Southeast Florida Coral Reef Fishery-Independent Baseline Assessment: 2012-2016 Summary Report

A. Kirk Kilfoyle, Brian K. Walker, Kurtis Gregg, Dana P. Fisco, and Richard E. Spieler

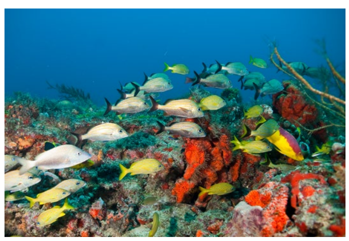
SEDAR82-RD56

Received: 9/21/2022

Southeast Florida Coral Reef Fishery-Independent Baseline Assessment: 2012-2016 Summary Report








National Oceanic and Atmospheric Administration
Coral Reef Conservation Program

Southeast Florida Coral Reef Fishery-Independent Baseline Assessment: 2012-2016 Summary Report

Prepared By: A. Kirk Kilfoyle[1], Brian K. Walker[1], Kurtis Gregg[2], Dana P. Fisco[1], and Richard E. Spieler[1]

[1]Nova Southeastern University Oceanographic Center, 8000 North Ocean Drive, Dania Beach, FL 33004

[2]ERT, Inc. *in support of* NOAA-Fisheries Service, Habitat Conservation Division, 400 N. Congress Avenue, Suite 110, West Palm Beach, FL 33401

January 4, 2018

Completed in Fulfillment of Contract
NA15OAR4320064

National Oceanic and Atmospheric Administration
Coral Reef Conservation Program
400 N. Congress Avenue, Suite 120
West Palm Beach, FL 33401

This report should be cited as follows:

Kilfoyle, A.K., Walker, B.K., Gregg, K., Fisco, D.P. and R.E. Spieler. 2017. Southeast Florida Coral Reef Fishery-Independent Baseline Assessment: 2012-2016 Summary Report. National Oceanic and Atmospheric Administration. 121 p.

This project received funding under multiple awards from NOAA Coral Reef Conservation Program and The Florida Department of Environmental Protection. The statements, findings, conclusions, and recommendations are those of the authors and do not necessarily reflect the views of NOAA or FDEP.

Photos: Kirk Kilfoyle; Illustrations: Diane Rome Peebles

Executive Summary

Reef fishes are an integral component of the southeast Florida coral reef ecosystem that provide as yet unmeasured ecologic and economic benefits to the region. Effective management of coral reef ecosystem components relies on datasets having sufficient spatial and temporal resolution to discern patterns for comparisons now and in the future. Until recently, there was no long-term data collection in place to assess the condition of reef fish resources of the northern Florida Reef Tract (FRT) (northern Miami-Dade, Broward, Palm Beach, and Martin counties). An assessment plan for the northern portion of the Florida reef tract was designed through a joint cooperative effort by scientists at the University of Miami, Rosenstiel School of Marine and Atmospheric Science (UM-RSMAS), National Oceanic and Atmospheric Administration - Southeast Fisheries Science Center (NOAA - SEFSC), Nova Southeastern University Oceanographic Center (NSUOC), in consultation with the Florida Fish and Wildlife Conservation Commission (FWC). This report is a synoptic overview of a five-year dataset that encompasses the collective sampling effort from all partner agencies, and includes survey results from 1,360 sites/Primary Sampling Units (PSUs) sampled during the 2012-2016 time period. The majority of the field work was accomplished through funding provided to NSUOC by the NOAA Coral Reef Conservation Program (CRCP) and the Florida Department of Environmental Protection – Coral reef Conservation Program (FDEP-CRCP). Significant amounts of data were also collected by multiple Southeast Florida Coral Reef Initiative (SEFCRI) partner agencies that were able to dedicate their time and resources to the project. Field sampling for each year began in May and ran through October.

During the five-year study period, >1.2 million individual fish representing 305 species and 70 families were recorded. Total mean density for all sites and strata combined for all five years was 176 fishes/SSU (±4.6 SEM) (Second-Stage Sample Unit = SSU or site, 177 m^2). Multivariate analyses showed patterns in the reef fish communities associated with benthic habitats. Water depth, reef type, bottom relief, and location were the primary determinants of reef fish distribution, with differences in assemblages between shallow (≤10 m) and deep (>10m) sites, high and low relief, and between multiple assemblage regions. In addition, the results indicate that regional populations of many commercially and recreationally important species are severely depleted, with large reproductively active adults being the most heavily exploited and in need of greater protection from fishing pressure.

The dataset provides opportunities for further mining to examine individual species and reef fish assemblage correlations with a host of abiotic and biotic variables. Thus, from both management and ecological-sciences perspectives, these data are a valuable resource. It is already clear there are significant differences in the geographic distribution of reef fishes at local and regional scales. There are interacting strata and latitudinal differences in total reef fish abundance, species distribution, sizes, and assemblage structure. The combination of data from all five years provides a complete regional fishery-independent baseline.

Acknowledgements

The success of this project can be attributed to the cooperative partnerships forged between multiple key agencies, universities, and individuals who have a vested interest in maintaining the health and sustainability of the coral reef ecosystems of southeast Florida. We thank James Bohnsack and Jeremiah Blondeau from the ***National Oceanic and Atmospheric Administration – National Marine Fisheries Service – Southeast Fisheries Science Center (NOAA-NMFS-SEFSC)*** *and Steven Smith, David Bryan, and Natalia Zurcher from the* ***University of Miami – Rosenstiel School of Marine and Atmospheric Science (UM-RSMAS)*** *for their essential guidance and analytical support throughout this process. By sharing expertise gained through development and refinement of the parent RVC program during its evolution in the Florida Keys and Dry Tortugas, they have strengthened the results of this baseline monitoring effort along the northern Florida Reef Tract.*

Many well-qualified scientific divers from multiple partner agencies (several of whom also had larger additional roles in this project) lent their time and resources to help make this project a safe, productive, and successful endeavor. Special recognition goes to the ***Florida Fish and Wildlife Conservation Commission (FWC) Tequesta Laboratory*** *and associates for their invaluable assistance with sampling the northern-most portion of the survey domain: Erick Ault, Jeff Beal, Anderson Berry, James Brodbeck, Grant Stoecklin, and Jim Whittington; and* **St. Lucie Inlet Preserve State Park:** *Ernest Cowen and Charles Jabaly. For assistance in sampling the southernmost portion of the survey domain:* ***Florida Department of Environmental Protection – Coral Reef Conservation Program (FDEP-CRCP):*** *Jennifer Jordan-Báez, Meghan Balling, Karen Bohnsack, Christopher Boykin, Gina Chiello, William Fisher, Melissa Gil, Julio Jimenez, Jamie Monty, Jenna Sansgaard, Melissa Sathe, Mollie Sinnott, Kristina Trotta, Katherine and Ori Tzadik, Joanna Walczak, Lauren Waters, Daron Willison, and Ana Zangroniz; and* ***Miami-Dade County – Department of Environmental Resources Management (DERM):*** *James Brown, Kevin Iglesias, Iman Olguin-Lira, Damaso Rosales, Rebecca Ross, Melissa Sathe, Jon Sidner, and Sara Thanner. Additional contributions were made by the* ***Broward County – Natural Resources Planning and Management Division (NRPMD):*** *Kenneth Banks, Courtney Kiel, Pat Quinn, and Angel Rovira;* ***FDEP-West Palm Environmental Resource Permit Program (ERP)****: Irene Arpayaglou; and* ***Coastal Eco-Group, Inc.****: Jenny Stein. Divers from* ***Nova Southeastern University Oceanographic Center (NSUOC)*** *include: Benjamin Barker, Cameron Baxley, James Brown, Brian Ettinger, Joshua Fredrick, Peter Grasso, Robert Jermain, Lystina Kabay, Morgan Knowles, Adam Nardelli, Keri O'Neil, Allison Patranella, and Shara Teter.*

Special thanks to our partners at ***NOAA-SEFSC*** *(Tom Adam, Jeremiah Blondeau (again), Joseph Contillo, Tara Dolan, James Europe, Jack Javech, Mike Judge, David McClellan, Lindsey Morrison, Benjamin Ruttenberg, and Brian Teare) and from* ***UM-RSMAS*** *(David Bryan, again) for additional training, logistical, and diving assistance. We thank Dana Wusinich-Mendez from* ***NOAA-CRCP*** *for her advice and assistance while navigating through the funding process, and Francisco Pagan from* ***FDEP-CRCP*** *for providing valuable feedback during preparation of this document.*

Table of Contents

List of Figures

List of Tables

List of Appendices

List of Acronyms

APT	All Purpose Tool
ANOSIM	Analysis of Similarity
ANOVA	Analysis of Variance
CREIOS	Coral Reef Ecosystem Integrated Observing System
CRCP	Coral Reef Conservation Program
DERM	Department of Environmental Resource Management
FDEP	Florida Department of Environmental Protection
FDOU	Fishing Diving and Other Uses
FKNMS	Florida Keys National Marine Sanctuary
FRRP	Florida Reef Resilience Program
FRT	Florida Coral Reef Tract
FWC	Florida Fish and Wildlife Conservation Commission
GIS	Geographic Information Systems
LAS	Local Action Strategy
LIDAR	Light Detection and Ranging
MDS	Multi-Dimensional Scaling
NFWF	National Fish and Wildlife Federation
NMFS	National Marine Fisheries Service
NRPMD	Natural Resources Planning and Management Division
NOAA	National Oceanic and Atmospheric Administration
NSUOC	Nova Southeastern University Oceanographic Center
PSU	Primary Sampling Unit
QA/QC	Quality assurance and quality control
RSMAS	University of Miami Rosenstiel School of Marine and Atmospheric Science
RVC	Reef Fish Visual Census
SEDAR	Southeast Data Assessment and Review
SEFSC	Southeast Fisheries Science Center
SEFCRI	Southeast Florida Coral Reef Initiative
SE FL	Southeast Florida
SIMPER	Similarity Percentages
SSU	Second-stage Sample unit
USCRTF	U.S. Coral Reef Task Force

1. INTRODUCTION

The ecosystem services of the Florida Reef Tract (FRT), including the diverse reef fish assemblage that it supports, have direct links to the health of both the state and local economies (Johns et al., 2001; Johns et al., 2004). Yet, it is widely believed and increasingly supported by multiple studies that abundance and average size of many commercially and recreationally important fishes have been reduced in the northern FRT (Ferro et al., 2005; Johnson et al., 2007; Ault and Franklin, 2011; Gregg, 2013a). Furthermore, a wide array of other acute and chronic anthropogenic impacts is applying increasing levels of additional stress to the entire reef system (e.g., coastal construction projects, sedimentation, ship groundings and anchor damage, water pollution, and other water quality issues) (Banks et al. 2008; Jordan et al., 2009; Behringer et al., 2011; Walker et al., 2012; Gregg, 2013b). These impacts are closely linked to the growing human population that resides in the highly developed and densely populated coastal region of southeast Florida. Because reef fishes are an important biologic, ecologic, and economic resource of the marine ecosystem, reef fish population trends and the associated driving forces need to be examined closely to understand and effectively manage these resources in a sustainable manner. In 1979, fishery-independent monitoring of reef fish populations began in the Florida Keys (the southern portion of the FRT from Dry Tortugas to Biscayne National Park). However, until recently there was no comparable fishery-independent data collection in place to assess the status of reef fish resources associated with the northern portion of the FRT (central and northern Miami-Dade, Broward, Palm Beach, and Martin counties).

Under the guidance of the U.S. Coral Reef Task Force (USCRTF), the Florida Department of Environmental Protection (FDEP) and the Florida Fish and Wildlife Conservation Commission (FWC) coordinated the formation of a team of marine resource professionals (local, state, regional, and federal), scientists, non-governmental organization representatives, and other coral reef stakeholders. This group, known as the Southeast Florida Coral Reef Initiative (SEFCRI) Team, gathers to develop local action strategies to address threats to the coral reef ecosystems in Miami-Dade, Broward, Palm Beach, and Martin Counties.

The SEFCRI Team identified the need for the development of a fishery-independent monitoring program for southeast Florida's coral reefs in 2004. This management need was again identified by stakeholders, managers, and scientists in 2008 during the Florida Reef Resilience Program (FRRP) Workshop, and most recently by managers and scientists at the National Oceanic and Atmospheric Administration (NOAA) Atlantic/Caribbean Coral Reef Ecosystem Integrated Observing System Workshop, and at Florida's Strategic Management Priorities Workshop. The need for fishery-independent information was confirmed in 2008 as contractors began gathering fishery-dependent and independent data for SEFCRI Local Action Strategy (LAS) Fishing, Diving, and Other Uses (FDOU) Project 18 & 20A: *Fisheries Resource Status and Management Alternatives for the Southeast Florida Region*. The contractors found several "snapshot" fishery-independent datasets in two of the four counties within the four-county SEFCRI region. With one exception (Ferro et al., 2005), these datasets mainly focused on artificial reef fish populations, and were only collected for one to two years. Preliminary results from Project 18 & 20A indicated that spatially and temporally explicit fishery-independent assessment on southeast Florida coral reefs was lacking and existing "snapshot" data could not be used to determine reef fish status and trends off of southeast Florida. Thus, the development of a fishery-independent assessment program for the region was recommended (Ault and Franklin, 2011). During the FWC review of the Ault and Franklin (2011) report, recommendations were provided to guide the development of the fishery independent sampling methodology and data analyses via email (L. Gregg, personal comment, March 21, 2017).

In 2011, Nova Southeastern University Oceanographic Center (NSUOC) received funding to develop a training program aimed at building the capacity to conduct a large-scale assessment of reef fish populations in southeast Florida. The assessment project was designed through a joint cooperative effort by scientists at the University of Miami Rosenstiel School of Marine and Atmospheric Science (UM-RSMAS) and NOAA-Southeast Fisheries Science Center (NOAA-SEFSC), in consultation with FWC, with the goal to effectively build on the success of the fishery-independent monitoring program implemented in the Florida Keys and apply it to the northern portion of the FRT. A robust statistical design and sampling plan for an initial region-wide survey was developed by UM-RSMAS, with archival data and additional assistance being provided by scientists at NSUOC and the FWC (FDEP-CRCP Project 3A). Data acquired in the assessment has enabled resource managers to examine the Florida Coral Reef Tract on a holistic scale and to more accurately assess the status of the reef fish resources, as well as contribute to system-wide multi-species stock assessments.

While the majority of the field work for this project was accomplished through funding granted to NSUOC, a significant portion of the data were collected by multiple partner agencies that were able to dedicate their time and resources to the project: NOAA-SEFSC, NOAA-Fisheries Southeast Region, Habitat Conservation Division (HCD), Florida Department of Environmental Protection-Coral Reef Conservation Program (FDEP-CRCP), Miami-Dade County-Department of Environmental Resources Management (DERM), Broward County-Natural Resources Planning and Management Division (NRPMD), and the FWC Tequesta laboratory. This report is a compilation of the five-year data collection from all partner agencies, and includes data from 232, 324, 308, 209, and 285 sites sampled in 2012, 2013, 2014, 2015, and 2016, respectively. The combination of data from all five years provides a complete regional baseline dataset from which the fishery-independent assessment is conducted.

2. PROJECT GOALS AND OBJECTIVES

The main goal of this project is implementation of a statistically robust, habitat-based, tiered fishery-independent sampling protocol designed to meet two main objectives: 1) to determine the current status of southeast Florida reef fish populations which will enable detection of changes in these populations in response to future management strategies, and 2) to provide a seamless integration with the existing Reef Fish Visual Census (RVC) program data which will allow for the entire FRT to be evaluated in a holistic manner. In addition, this project is intended to continue fostering beneficial partnerships among NSUOC, NOAA National Marine Fisheries Service (NMFS), NOAA-CRCP, FDEP-CRCP, FWC, and other Florida Keys and SEFCRI partner agencies and organizations. In addition to providing quantitative reef fish data to researchers and managers in southeast Florida, products of this work are already being used by the NOAA-CRCP. The National Coral Reef Monitoring Program (NCRMP) field sampling is conducted every other year (even years) in Florida. The SEFCRI RVC data were included in the NCRMP datasets in 2014 and 2016. The SEFCRI Fishery-Independent Baseline Assessment will provide NCRMP with an excellent foundation from which to base future reef fish assessments and any subsequent management actions.

Implementation of the SEFCRI Fishery-Independent Baseline Assessment included: project planning, sample allocation, diver training, coordination with southeast Florida partners, in-water field work/data collection, data entry, data quality assurance and quality control (QA/QC), data analysis, Geographic Information Systems (GIS) analyses, report writing, and determination of sites for each survey season.

3. METHODOLOGY

3.1. Study Area and Design

The study area included all previously mapped marine benthic hardbottom and coral reef habitats along the southeast Florida coastline shallower than 33 m from Government Cut in Miami-Dade County to the northern border of Martin County (Figure 1). The study area for the Florida Keys RVC survey spans south from Biscayne Bay National Park through the Florida Keys. The sampling design for the northern portion of the FRT was created with local stakeholder input in a separate FDEP-CRCP project by Ault et al. (2012). The plan adapted the stratified, random statistical sampling design developed and implemented for the Florida Keys reef fish monitoring plan (Smith et al., 2011).

The reef-scape was gridded into 100-m cells, referred to herein as primary sampling units (PSUs). Each PSU was divided into four 50x50 m grid cells to acquire second-stage randomized data collection locations with the PSU (Figure 2). A PSU is synonymous with a “site” throughout the remainder of this document. At each second-stage data collection point multiple data collections (fish counts) occurred. During the analysis, an arithmetic mean for adjacent counts from each buddy team was calculated to determine fish density per data collection area (177 m^2). This area is referred to herein as a second-stage unit (SSU).

Each PSU and SSU was characterized by three main strata types, which combined are termed herein as map strata: coral reef ecosystem biogeographic subregion, benthic habitat type, and topographic slope (Table 1). The coral reef ecosystem biogeographic subregions defined in Walker (2012) and Walker and Gilliam (2013) were used to divide the study area into ecologically relevant regions. Grid cells were characterized according to which region the majority of the unit resided. Benthic habitat maps from previous efforts were used to determine the majority habitat type in each PSU and SSU (Riegl et al., 2005; Walker et al., 2008; Walker, 2009; Walker, 2013). Benthic habitat maps contained more detail than was practical for the stratification, therefore *a priori* decisions were made to combine more specific habitats into broader strata (Table 2). Since topographic complexity also affects local fish distributions (Walker et al., 2009), topographic slope was included in the stratification as a surrogate for larger scale (10s of meters) topographic complexity. The slope was calculated in ArcGIS using high resolution LIDAR (Light Detection and Ranging) data. LIDAR data were analyzed for slope where all areas greater than 5° were considered "high slope". A single polygon layer of these areas was created and used to determine if the PSU and SSU majority were classified as high or low slope.

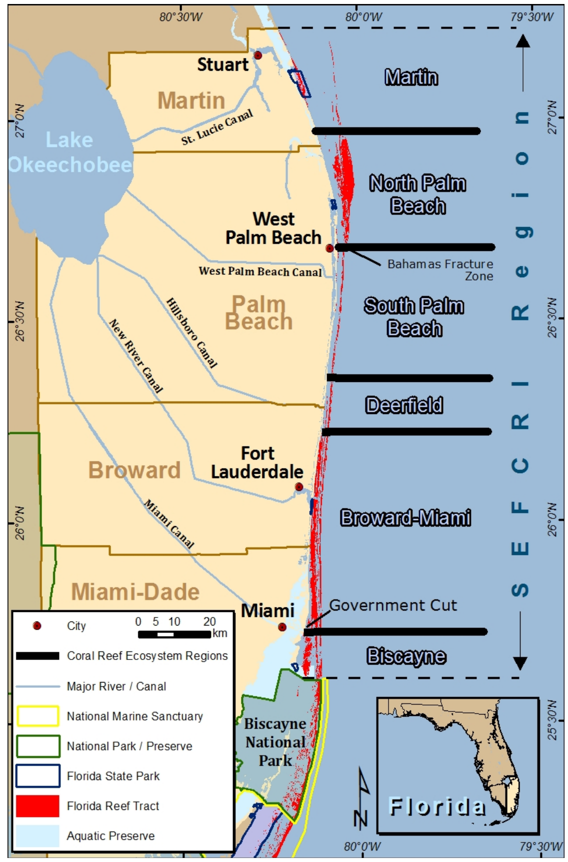


Figure 1. Study area included all hardbottom and reef habitats between the northern boundary of Martin County to Government Cut in Miami-Dade County.

The map strata were used to parse the region into finer categories to optimize survey locations for the eight targeted fishery species (see Table 4, page 11). A pure randomized design would take many more surveys to acquire the necessary data on the desired species, whereas a strategically targeted design is much more efficient (Smith et al., 2011). In the Florida Keys, this strategy has been used effectively to optimize data collection by capturing the variability of species by habitat strata and allocating more sample sites to those strata with higher variation. In the case of the northern portion of the FRT, initially there was not much regional information available about the fisheries species to inform the survey design, thus the proportion of benthic habitats were used (Ault et al., 2012). Subsequent years used previously collected data to aid in the site allocations. When including the biogeographic subregions, slope, and benthic habitat types, there were too many individual categories to be practical in the stratified random design and many were not thought to pertain to the targeted species (see section 3.4). For example, the subtle differences between Colonized Pavement-Shallow and Ridge-Shallow benthic communities and geomorphology were not thought to be major factors affecting species distribution. Therefore, certain benthic habitats were combined into what were intended to be more relevant strata, such as the nearshore habitats (NEAR). Combining the benthic habitats into habitat strata resulted in thirty-one map strata that were used in the sampling allocations (Table 1).

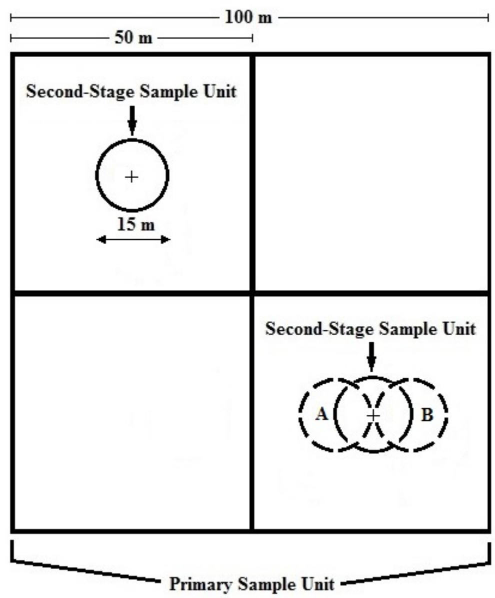


Figure 2. Illustration of Primary Sample Unit (PSU) and Second-Stage Sample Units (SSUs). Selection of 2 individual target SSUs was accomplished by a randomization of the 4 cells within the PSU. The dashed circles represent a buddy pair (A and B). [modified from Smith et al., 2011]

It was initially estimated that 360 PSUs could be visited each year with a combined effort from all partner agencies. Site allocations for each stratum were guided by the proportional distribution of strata in the sampling frame. Each stratum was given a minimum of 5 sites. Then the remaining sites were distributed proportionally by the strata area. Extremely large strata were limited to 50 sites. There were no other special strata that needed to be accommodated within the southeast Florida area survey frame, unlike the Florida Keys and Dry Tortugas annual surveys, which have been conducted largely within the boundaries of protected areas or special use zones. Once the total number of target sites for each stratum was determined, the corresponding number of PSUs was randomly chosen based on equal probability of selection from the survey frame using NOAA's sampling design tool for ArcGIS (http://coastalscience.noaa.gov/-projects/detail?key=185). Then, two of the four SSUs in each chosen PSU were randomly selected. The center location of the two chosen SSUs were the sample sites for that PSU. Then each point was evaluated in GIS and, if necessary, moved to the nearest target habitat within the SSU (where possible). In most cases, the points were not moved, but occasionally the sites targeting high relief needed to be adjusted to ensure that divers could find the appropriate habitat immediately instead of having to swim around looking for it. In cases where no suitable habitat was nearby, the point was discarded, and a suitable alternate was chosen. Appendix 1 displays the actual survey locations for all five years (2012-2016).

Table 1. *Map strata for the site randomization to optimize survey outcomes. The biogeographic subregions, habitat strata, and slope were used to define these areas. See Table 2 for habitat strata details.*

Subregion	**Habitat Strata**	**Slope**
Broward-Miami	INNR	High
Broward-Miami	INNR	Low
Broward-Miami	MIDR	High
Broward-Miami	MIDR	Low
Broward-Miami	NEAR	High
Broward-Miami	NEAR	Low
Broward-Miami	OFFR	High
Broward-Miami	OFFR	Low
Broward-Miami	PTDP	High
Broward-Miami	PTDP	Low
Broward-Miami	PTSH	N/D
Deerfield	MIDR	High
Deerfield	MIDR	Low
Deerfield	NEAR	Low
Deerfield	OFFR	High
Deerfield	OFFR	Low
Deerfield	PTDP	High
Deerfield	PTDP	Low
South Palm Beach	NEAR	Low
South Palm Beach	OFFR	High
South Palm Beach	OFFR	Low
South Palm Beach	PTDP	High
South Palm Beach	PTDP	Low
South Palm Beach	PTSH	N/D
North Palm Beach	DPRC	High
North Palm Beach	DPRC	Low
North Palm Beach	NEAR	Low
Martin	NEAR	High
Martin	NEAR	Low
Martin	RGDP	High
Martin	RGDP	Low

Table 2. Mapped benthic habitat classes and stratification habitat codes for this study, and major categories for the benthic habitat map in the southeast Florida region.

Map Habitat Class	Habitat Strata
Deep Ridge Complex	DPRC
Linear Reef-Inner	INNR
Linear Reef-Middle	MIDR
Linear Reef-Outer	OFFR
Ridge-Deep	OFFR (RGDP in Martin County only) *
Ridge-Shallow	NEAR
Other Delineations (Artificial, dredged inlets, sand borrow areas)	OTHR
Aggregated Patch Reef-Deep	PTDP
Aggregated Patch Reef-Shallow	PTSH
Patch Reef	PTSH <20m; PTDP >20m
Colonized Pavement-Deep	OFFR
Colonized Pavement-Shallow	NEAR
Unconsolidated Sediment	SAND
Scattered Coral/Rock in Sand	PTSH <20m; PTDP >20m
Seagrass	SGRS
Spur and Groove	OFFR
No Map Data	UNKW

*The Ridge-Deep was included in the OFFR strata for the southern portion of the reef tract. However, in Martin County, it was recognized as distinctly different and thus kept as a separate stratum.

Throughout the four-county region, a total of 360 primary and 216 alternate sites were selected in 2012. For 2013, a slightly different strategy was employed, using 360 primary/core, 105 secondary/tier 2, and 216 alternate sites selected. Core target sites were prioritized and completed before the tier 2 sites to ensure a minimum number of sites in each stratum were targeted in case all sites were not surveyed. Over the course of the 2013 field season almost every site on both the core and tier 2 lists were sampled. Due to the success of the 2013 sampling season, the secondary site strategy was abandoned in 2014, and 350 primary sites were selected. In 2015, sample size was reduced to 224 primary sites due to reduced funding. In 2016, additional funding was procured, and the number of primary sites was increased to 290.

3.2. Data Collection

Assessing population size and community level or species-specific trends of coral reef fishes is inherently difficult because of many factors. Reef fishes are speciose, exhibit various morphological and behavioral traits, have patchy distributions, many are highly mobile, some are crepuscular or nocturnal, and they occur in heterogeneous and diverse habitats. These factors can make it difficult to determine optimal or standardized survey methods, and as a result, many different visual survey methods have developed over time that are designed to provide researchers with the ability to assess fish populations at varying levels of precision. In recent years, much progress has been made regarding standardization of survey methodology among multiple academic, scientific and regulatory entities that routinely monitor and conduct research on the coral reefs found within the territorial waters of United States (Brandt et al., 2009). The most widely utilized method for assessing populations of coral reef fishes has become the stationary point-count (Bohnsack and Bannerot, 1986). This method was developed in the Florida Keys in 1979, and has been the utilized as the standard means of fishery-independent visual survey data collection for the Florida Keys RVC project since 1999 (Colvocoresses and Acosta, 2007). During a point-count, the survey diver establishes a location at the center of an imaginary cylinder 15 m in diameter (177 m^2) that encircles a column of water extending from the seabed to the sea surface. During a Reef Visual Census (RVC) point-count (RVC count and point-count are used synonymously throughout the remainder of this document), for the first five minutes only species names are recorded, with the exception of any highly migratory or target species (groupers, snappers, etc.), which are enumerated as soon as they are seen. It is the species encountered during the first five minutes that are most critical for establishing a representative "snapshot" of the area as it existed when the divers entered the water. For the second five minutes, the numbers and size ranges (mean, min, max) (fork length) of each species are filled in, with new species being added to the list as they are encountered. Additional members of species that were observed during the first five minutes that enter the survey area after their initial observation are not recorded a second time.

All visual assessment methods have biases (pros and cons) that are associated with the individual technique. Advantages of the RCV point-count method include: 1) data collection is non-destructive, 2) the ability to be easily replicated and randomized, 3) data are fishery-independent, 4) the ability to observe and characterize the community as a whole, 5) production of data that are amenable to rigorous statistical analysis, and 6) the ability to be quickly and economically employed. Some items that are considered as potential biases of commonly employed visual survey methods, including point-counts, are the tendency to over- or under-estimate numbers of fish, especially in terms of density and diversity of small, cryptic fishes, exceptionally abundant schooling fishes, and in highly complex habitats and when species richness is very high (Harvey et al., 2002; Harvey et al., 2003; Edgar et al., 2004; Colvocoresses and Acosta, 2007). In addition, it only assesses species that are readily visible during the day, and in the case of this project, only during the summer months (May-October). However, one of the goals of a well-designed fishery-independent monitoring program is to establish and maintain a consistent sampling method which will track and quantify relative changes in abundance/density/species richness/diversity over space and time. The RVC method meets the goals of generating useful data with moderate logistical requirements. Creating a completely accurate representation of a complex biological community is neither an essential goal for most management needs, nor a realistic goal due to the stochastic nature of community structure. The stratified sampling design implemented in this project is specifically designed to generate sample sizes adequate to allow for meaningful statistical comparisons within the observed range of abundance levels and within the boundaries of the survey domain.

Task methodology followed established methods from the FDEP-CRCP Project 3A report: Development of a Coral Reef Fishery-Independent Assessment Protocol for the Southeast Florida Region (Ault et al., 2012), and the RVC report: A Cooperative Multi-agency Reef Fish Monitoring Protocol for the Florida Keys Coral Reef Ecosystem (Brandt et al., 2009). In addition to assessing reef fish populations, the RVC protocol included a rapid characterization of multiple benthic habitat features with each point-count. Divers were equipped with a standardized 1-meter "All Purpose Tool" (APT) that was used to aid in size estimation of fishes and assessment of benthic habitat components. Benthic habitat features surveyed after each point-count included: substrate slope, max vertical hard and soft relief, surface relief coverage of hard and soft features, abiotic footprint, biotic cover by major organismal category, habitat type, underwater visibility and cylinder radius, water temperature, and current strength.

Abundance and distribution of reef fishes has been shown to fluctuate on a seasonal basis within the southeast Florida region, with greater abundances for many species being the norm during the summer months (Bohnsack et al., 1994; Sherman et al., 1999; Walker et al., 2002; Jordan et al., 2004). Therefore, data collection took place only within the months of May through October each year, with the exception of 2016 which pushed into early November due to frequent unfavorable marine conditions. The total number of sites and yearly percent contribution made by each agency (Table 3) does not account for the contribution that some divers made while conducting surveys from other partner agency vessels in order to increase sampling efficiency.

Table 3. Yearly sampling effort by total number of PSUs sampled and percentages of the total yearly effort for all partner agencies.

Agency	2012	2013	2014	2015	2016	Total
NSUOC	163 (70%)	193 (59%)	202 (66%)	130 (62%)	187 (65%)	875 (64.3%)
FWC Tequesta	7 (3%)	16 (5%)	50 (16%)	40 (19%)	38 (13%)	151 (11.1%)
FDEP-CRCP	16 (7%)	16 (5%)	23 (7%)	27 (13%)	39 (14%)	121 (8.9%)
NOAA-SEFSC	19 (8%)	87 (27%)	0 (0%)	0 (0%)	0 (0%)	106 (7.8%)
Miami-Dade Co.	15 (6%)	7 (2%)	24 (8%)	10 (5%)	12 (4%)	68 (5.0%)
Broward Co.	10 (4%)	6 (2%)	9 (3%)	2 (1%)	10 (4%)	37 (2.7%)
FDEP-West Palm	2 (1%)	0 (0%)	0 (0%)	0 (0%)	0 (0%)	2 (0.1%)
Totals	232	325	308	209	286	1360

3.3. Data Entry and Quality Control

Efforts to ensure maximum quality of the data were maintained throughout all levels of the data collection, entry, and verification process in order to create the most accurate database possible. This began with a review of the data sheet immediately following each dive, during which the diver consulted with their dive buddy and the other dive team (when applicable) about each entered variable to detect questionable or unreasonable entries, discrepancies, or missing data. Divers were encouraged to enter their data as soon as possible upon returning from the field, ideally the same or next day, but no longer than one week in order to give the diver the ability to best recall the specifics of each dive, detect any potential errors that were not caught on the boat, and prevent errors that would be caused by rushing to enter a large amount of data from an entire season at the last minute. Upon reaching the end of the sampling season, the lead data manager from each partner agency was responsible for generating proofing sheets which served as an aid to finding and correcting errors to the dataset during the quality assurance/quality control process. Once all errors were identified and corrected, the final version of the data (i.e., sample, species, and substrate files, boat log, diver log, and environmental data) for each agency was submitted to NSUOC for the final data merge and verification procedures. Once final data from each agency were compiled, the RVC Annual Master Spreadsheet file was created. This file consisted of merged (via Merge2.0.exe program) ASCII sample, substrate and species data outputs from the RVC data entry program, along with a combined version of the Boat/Field and Water Quality/Environmental logs, each of which became one of four individual worksheets within the completed RVC Annual Master Spreadsheet file. The next step involved performing an in-depth cross check of each of the four worksheets to locate any missing samples or incorrectly entered data, outliers, unlikely sizes and numbers of particular species, and any other dubious entries. Questionable elements discovered during this process were typically resolved by contacting the individual diver(s) who collected the data. A final rigorous verification procedure followed which scrutinized the habitat and substrate data, comparing the observed results to the GIS database.

3.4. Data Analysis

A descriptive ecological analysis that includes species inventory, density, and frequency of occurrence of all fish species observed was performed on the 2012-2016 dataset. This analysis followed established methods adopted from a previous RVC report (Brandt et al., 2009). Each of the aforementioned metrics was partitioned by individual strata (reef fish assemblage region, subregion, habitat type, and slope). Density was reported in terms of mean "SSU Density", which is the average of the data collections conducted in each secondary survey location (usually 2, rarely 1 or 3). This standardized each data collection to a single area of 177 m^2. For analyses presented in this report, species that were recorded past the 10 minute mark during a survey were filtered out unless they were highly unlikely to have entered the survey area during that time period (such as small and/or cryptic/sedentary species, etc.). In addition, an initial exploration into the trends of distribution and abundance throughout the greater Florida Reef Tract (combining data from the northern portion of the FRT with that from the Florida Keys and Dry Tortugas) was undertaken for a selection of economically important target species.

Of particular interest in the northern portion of the FRT, and one of the primary motivating factors for this project, is the population status of commercially and recreationally important reef fish species. Therefore, a subset of eight target species, based on their status as species of economic importance and their estimated level of exploitation in southeast Florida, were selected for an in-depth evaluation of average density and percent occurrence at different life-stages (pre-exploited and exploited) and average length of the exploited phase individuals (Lbar). The minimum legal size-limit or size at reproductive maturity (for unregulated species) was used as a measure for pre-exploited versus exploited and varied by species (Table 4) (FWC, 2017a). Fish with a fork length (FL) less than the specified length were considered as "pre-exploited" (not targeted in recreational or commercial fishing) and larger fish as "exploited". The species were: Red Grouper (*Epinephelus morio*), Mutton Snapper (*Lutjanus analis*), Gray Snapper (*Lutjanus griseus*), Yellowtail Snapper (*Ocyurus chrysurus*), White Grunt (*Haemulon plumierii*), Bluestriped Grunt (*Haemulon sciurus*), Hogfish (*Lachnolaimus maximus*), and Gray Triggerfish (*Balistes capriscus*). Although not widely recognized as a species of economic value, the growing presence of invasive Red Lionfish (*Pterois volitans/miles* complex) is of great concern as a potential threat to local reef fish resources, including juveniles of many commercially and recreationally important species. For these reasons, an additional analysis was performed that evaluated population levels of Red Lionfish in southeast Florida.

Table 4. List of commercially and recreationally important species and their minimum legal size of harvest (exploited lengths) in Atlantic waters from state and federal jurisdictions during the study period. Note: White and Bluestriped Grunts are unregulated species [FWC, 2017a].

Common Name	**Scientific Name**	**State**	**Federal**
Gray Triggerfish	*Balistes capriscus*	12" (30 cm) FL	14" (35 cm) FL
Red Grouper	*Epinephelus morio*	20" (50 cm) TL	
White Grunt	*Haemulon plumierii*	8" (20 cm) FL	
Bluestriped Grunt	*Haemulon sciurus*	8" (20 cm) FL	
Hogfish	*Lachnolaimus maximus*	12" (30 cm) FL	
Mutton Snapper	*Lutjanus analis*	16" (40 cm) TL	
Gray Snapper	*Lutjanus griseus*	10" (25 cm) TL	12" (30 cm) TL
Yellowtail Snapper	*Ocyurus chrysurus*	12" (30 cm) TL	

3.5 Reef Fish Assemblage Regions

Ecosystem based management calls for moving away from single species management to a more holistic management approach. Since southeast Florida reefs span northward along an ecological transition between the temperate north and subtropical south (Walker, 2012), a detailed study was conducted to investigate regional reef fish assemblage biogeography to elucidate the spatial relationships of reef assemblages throughout southeast Florida (Kilfoyle et al., 2014; Fisco, 2016). This study included the first three years of RVC data (2012-2014) from southeast Florida. Fisco (2016) statistically defined seven reef fish assemblage regions between St. Lucie Inlet and Government Cut based on the relative density of all species at each site, between all sites (Figure 3). Sites that were more similarly grouped together had consistent spatial locations throughout the survey domain. The main factors dividing the assemblages were depth, reef type, bottom relief, and location. Thus, for the final report herein, we provide the data analyses using the reef fish assemblage biogeographic regions as data analysis strata to further investigate regional differences and provide an ecosystem-based context to the study instead of solely focusing on individual species.

The data analysis strata (i.e., reef fish assemblage biogeographic regions) are as follows:

Deep Martin High (DMAH) – Sites deeper than 10 meters in the Martin Coral Reef Ecosystem Region with an average vertical relief greater than 0.3 meters.

Deep Martin Low (DMAL) – Sites deeper than 10 meters in the Martin Coral Reef Ecosystem Region with an average vertical relief less than 0.3 meters.

Deep North Palm Beach (DNPB) – Sites deeper than 10 meters in the North Palm Beach Coral Reef Ecosystem Region.

Deep South Palm Beach Miami (DSPM) – Sites deeper than 10 meters in the South Palm Beach, Deerfield, and Broward-Miami Coral Reef Ecosystem Regions.

Shallow Martin (SMAR) – Sites 10 meters or shallower in the Martin Coral Reef Ecosystem Region.

Shallow North Palm Beach Deerfield (SPBD) – Sites 10 meters or shallower in the North Palm Beach, South Palm Beach, and Deerfield Coral Reef Ecosystem Regions.

Shallow Broward-Miami (SBRM) – Sites 10 meters or shallower in the Martin Coral Reef Ecosystem Region.

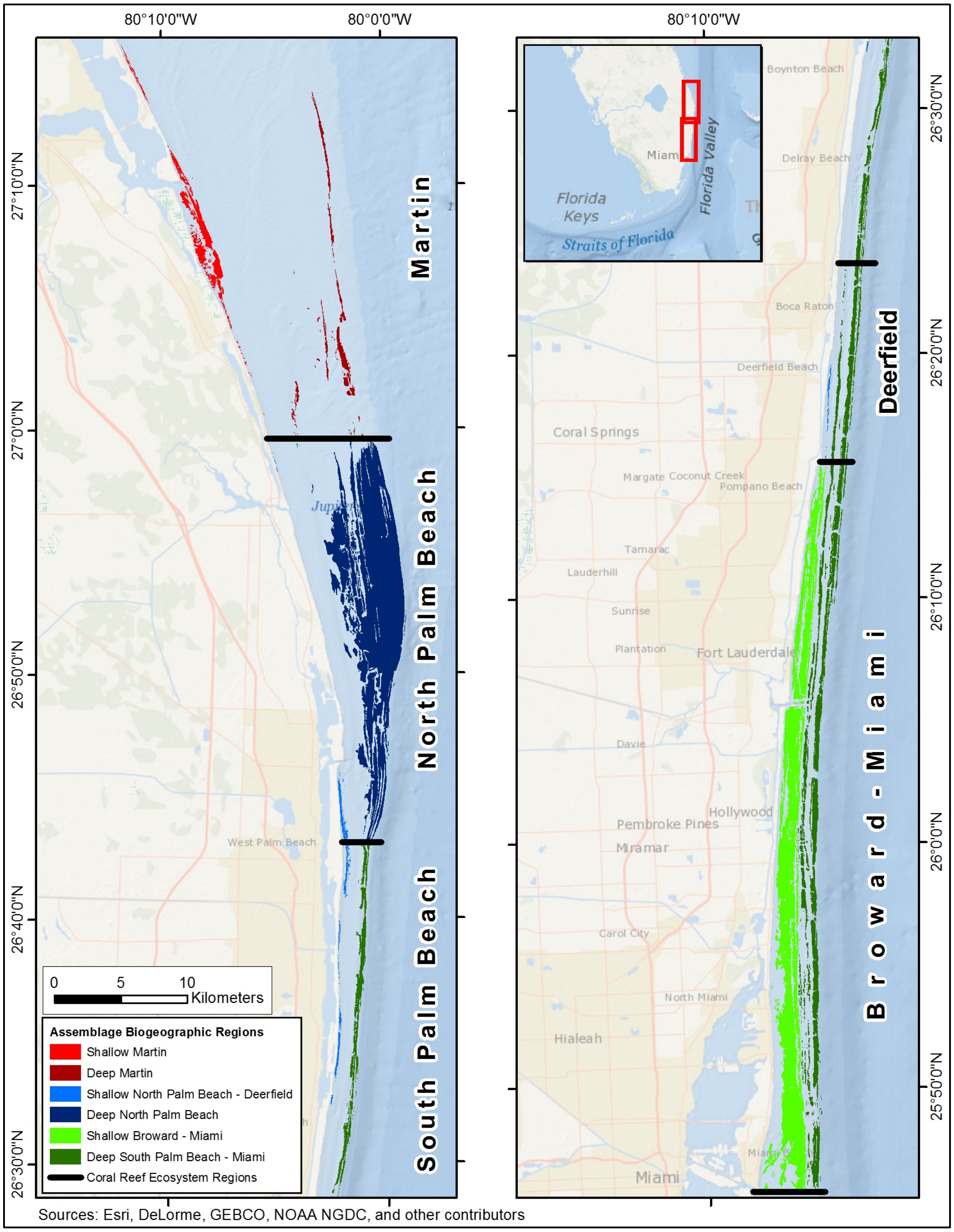


Figure 3. Map of the southeast Florida Reef Tract with Reef Fish Assemblage Biogeographic Regions indicated by habitat color. Shallow sites are lighter shades and Deep sites are darker. The Coral Reef Ecosystem Regions of Walker (2012) and Walker and Gilliam (2013) are labeled and divisions are indicated by dark bars.

4. RESULTS AND DISCUSSION

During the combined 2012-2016 sampling period, a total of 1,360 sites (PSUs) were surveyed by 65 divers from 11 agencies over the course of 5,290 dives. In 2012, 42 divers from 7 partner agencies conducted 881 dives, completing surveys at 232 sites. In 2013, 34 divers from 6 partner agencies conducted 1,226 dives, completing surveys at 325 sites. In 2014, 35 divers from 6 partner agencies conducted 1,213 dives, completing surveys at 308 sites. In 2015, 29 divers from 9 partner agencies conducted 830 dives, completing surveys at 209 sites. In 2016, 33 divers from 7 partner agencies conducted 1,140 dives, completing surveys at 286 sites.

4.1. Fish Assemblage

Over the course of the five-year study period, 1,238,951 fish representing 305 species from 70 families were counted (215 in 2012, 253 in 2013, 243 in 2014, 230 in 2015, and 234 in 2016). Out of those 305 species, 184 were recorded every year. Of the 121 species that were seen less frequently, 50 were small cryptic or nocturnal species, 10 were solitarily occurring elasmobranchs, 10 were large sportfishes, 7 were temperate-associated species, and many of the rest are considered as uncommonly or infrequently encountered. Comparatively, 214 species have been recorded from 13 years of annual monitoring (2001-2013) at repeated monitoring sites within Broward County (Gilliam et al., 2014) and a compiled total of 354 species (although not all reef associated) have been recorded in coastal marine habitats in Broward County from multiple projects over the course of the past 20+ years (Spieler et al., unpublished data). To further extend the comparison, there were 347 species recorded in RVC surveys in the Florida Keys and 370 species in the Dry Tortugas during the same 2012-2016 time-frame.

4.1.1. Fish Density and Species Richness

Total mean density for all years, species, sites, and strata combined was 176 ±4.6 SEM fishes/SSU. When mean SSU density between reef fish assemblage regions is compared (Figure 4), there was a general increase in density moving from north to south for the deep assemblage regions (DMAL, DNPB, DSPM), with the exception of the Deep Martin High (DMAH) assemblage region, which had the highest mean density of all the regions. This was partially attributable to the presence of high numbers of Mackerel and Round Scad (*Decapterus macarellus* and *D. punctatus*, respectively) that were recorded in 2013. There was little difference in mean density and rugosity between Shallow Martin (SMAR) and Shallow Broward-Miami (SBRM) (although there were considerable differences in assemblage structure; see section 4.1.2), but Shallow North Palm Beach-Deerfield (SPBD) was higher than both.

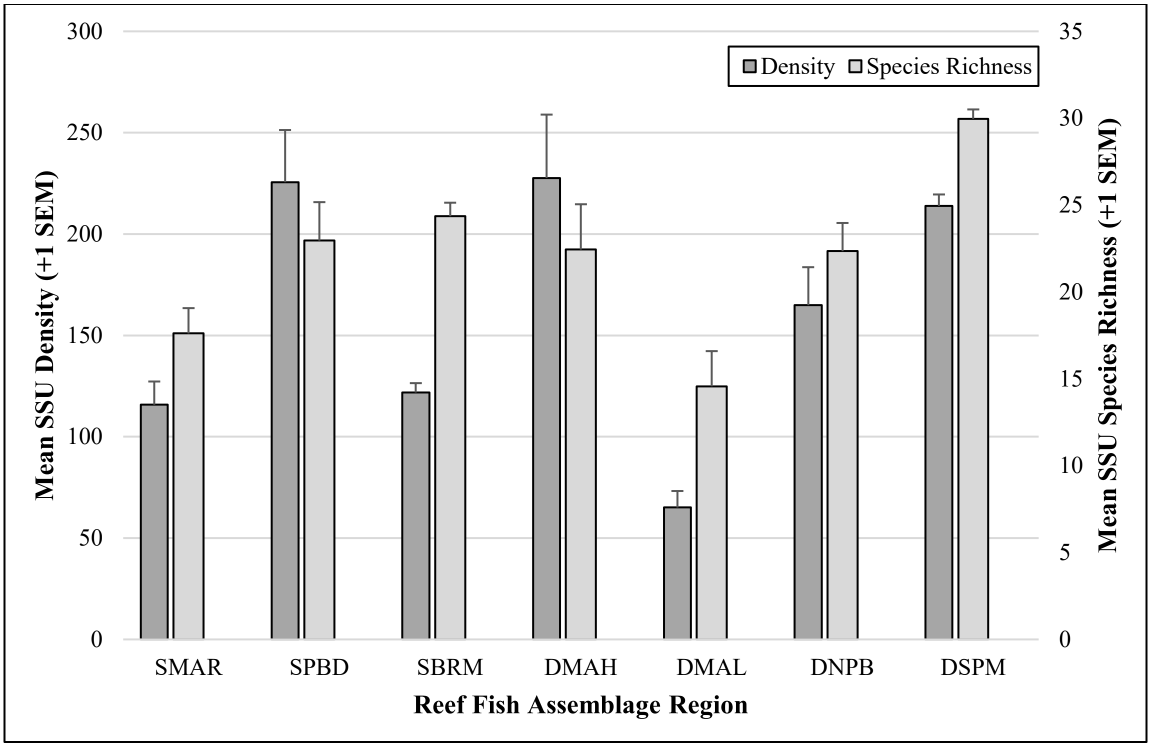


Figure 4. Mean SSU density and species richness by Reef Fish Assemblage Region: Shallow Martin (SMAR, N=100), Shallow Palm Beach-Deerfield (SPBD, N=137), Shallow Broward Miami (SBRM, N=761), Deep Martin High (DMAH, N=89), Deep Martin Low (DMAL, N=64), Deep North Palm Beach (DNPB, N=379), Deep South Palm Beach-Miami (DSPM, N=1,133).

Mean species richness for all years, sites, and strata combined was 24.8 ±0.6 species/SSU, and remained fairly similar among the five years of the study. For 2012 mean species richness was 25.6 ±0.5 species/SSU, in 2013 it was 23.8 ±0.4 species/SSU, in 2014 it was 26.3 ±0.4 species/SSU, in 2015 it was 25.7 ±0.5 species/SSU, and in 2016 it was 27.3 ±0.4 species/SSU. When species richness by reef fish assemblage region was compared (Figure 4), the deep regions closely resembled that of mean density with increasing species richness moving north to south, except for Deep High-relief in Martin County. Richness between the shallow regions was lower in Martin than the two regions further south which were similar to each other.

The top ten most densely recorded species averaged over all years and strata in order of decreasing SSU density ($\overline{D}$) were: Bicolor Damselfish (*Stegastes partitus*), Bluehead Wrasse (*Thalassoma bifasciatum*), Masked/Glass Goby (*Coryphopterus personatus/hyalinus*), Tomtate (*Haemulon aurolineatum*), unidentified/juvenile Grunts (*Haemulon* spp.), Creole Wrasse (*Clepticus parrae*), Yellowhead Wrasse (*Halichoeres garnoti*), Ocean Surgeonfish (*Acanthurus bahianus*), Redband Parrotfish (*Sparisoma aurofrenatum*), and Slippery Dick Wrasse (*Halichoeres bivitattus*). Unidentified herring (*Jenkinsia* sp.) were technically ranked 9th in the top 10 list, but they were confined to only nine recordings from the first three years, and four observations consisted of schools of 150-5,400 individuals. Since they are not strongly linked to reef environments and a few schools skewed their densities, this species was considered as an outlier for the purposes of this list/report.

The top ten most frequently observed species (e.g., frequency of occurrence ($\bar{P}$)) were as follows (in decreasing order): Sharpnose Pufferfish (*Canthigaster rostrata*), Bluehead Wrasse, Bicolor Damselfish, Ocean Surgeonfish, Doctorfish (*Acanthurus chirurgus*), Yellowhead Wrasse, Redband Parrotfish, Slippery Dick Wrasse, Reef Butterflyfish (*Chaetodon sedentarius*), and Blue Tang (*Acanthurus coeruleus*).

During the five-year study period, a total of 26 species were encountered in the southeast Florida region that had not been previously recorded in the Florida Keys or Dry Tortugas RVC surveys. These were added to the master species list used for the RVC data entry program. Following the 2012 surveys, seven species were added: Spotted Burrfish (*Chilomycterus reticulatus*), Atlantic Bumper (*Chloroscombrus chrysos*), Flying Gurnard (*Dactyloscopus volitans*), Sharptail Eel (*Myrichthys breviceps*), Goldspotted Eel (*Myrichthys ocellatus*), Atlantic Guitarfish (*Rhinobatos lentiginosus*), and Black Brotula (*Stygnobrotula latebricola*). Following the 2013 surveys, seven more species were added: Whitebone Porgy (*Calamus leucosteus*), Black Seabass (*Centropristis striata*), Mottled Mojarra (*Eucinostomus lefroyi*), Oyster Toadfish (*Opsanus tau*), Blackwing Searobin (*Prionotus rubio*), Banded Rudderfish (*Seriola zonata*), and Rough Scad (*Trachurus lathami*). Following the 2014 surveys, eight more species were added: Dwarf Goatfish (*Upeneus parvus*), Tiger shark (*Galeocerdo cuvier*), Chestnut Moray (*Enchelycore carychroa*), Red Snapper (*Lutjanus campechanus*), Palometa (*Trachinotus goodei*), Cownose Ray (*Rhinoptera bonasus*), Freckled Soapfish (*Rypticus bistrispinus*), and Bank Seabass (*Centropristis ocyurus*). Following the 2015 surveys, four more species were added: Orbicular Batfish (*Platax orbicularis*) (exotic), Southern Stargazer (*Uranoscopus y-graecum*), Bantam Bass (*Parasphyraenops incisus*), and Bank Butterflyfish (*Prognathodes aya*). No confirmed new species were added to the list following the 2016 surveys.

4.1.2. Fish Community Regional Habitat Associations

Multivariate analyses showed patterns in the reef fish assemblages associated with the reef habitat strata and region (Figure 5). The MDS shows the sites categorized by the strata where the shallow sites are circles and the deep sites are triangles or squares. The colors represent the different regions and match the spatial assemblage region illustrations in Figure 3. Red and garnet are Martin County region sites. Blue and navy are mostly Palm Beach and Deerfield region sites. Light and dark green are mainly Broward and Miami region sites. The MDS plot shows patterns where different colors and shapes populate different parts of the plot. For example, the Deep South Palm Beach to Miami (DSPM) sites (dark green triangles) are all clustered in the bottom center, whereas the Shallow Martin (SMAR) sites (red circles) are in the upper right quadrant. It is difficult to visualize with so many sites and so many categories in a two-dimensional plot. The separation is more obvious in three dimensions where the plot can be actively turned. Another obvious data pattern is the spread of sites. These results are similar to previously reported analyses on a large reef fish dataset for northern Broward County

(Walker et al., 2009). Walker et al. (2009) found that fish communities were more tightly clustered (similar) in the deeper communities and more variable in the shallow habitats. These new data analyses show that the pattern is not only depth driven, but varies between assemblage regions. The deep North Palm Beach and Martin sites (DNPB, DMAL, DMAH) and Shallow Martin, Shallow Palm Beach Deerfield, and Shallow Broward Miami sites (SMAR, SPBD, SBRM) all have considerable spread in the data that overlap with other assemblage regions. The Deep South Palm Miami (DSPM) sites are the most clustered in the MDS, which indicates this group has less variability.

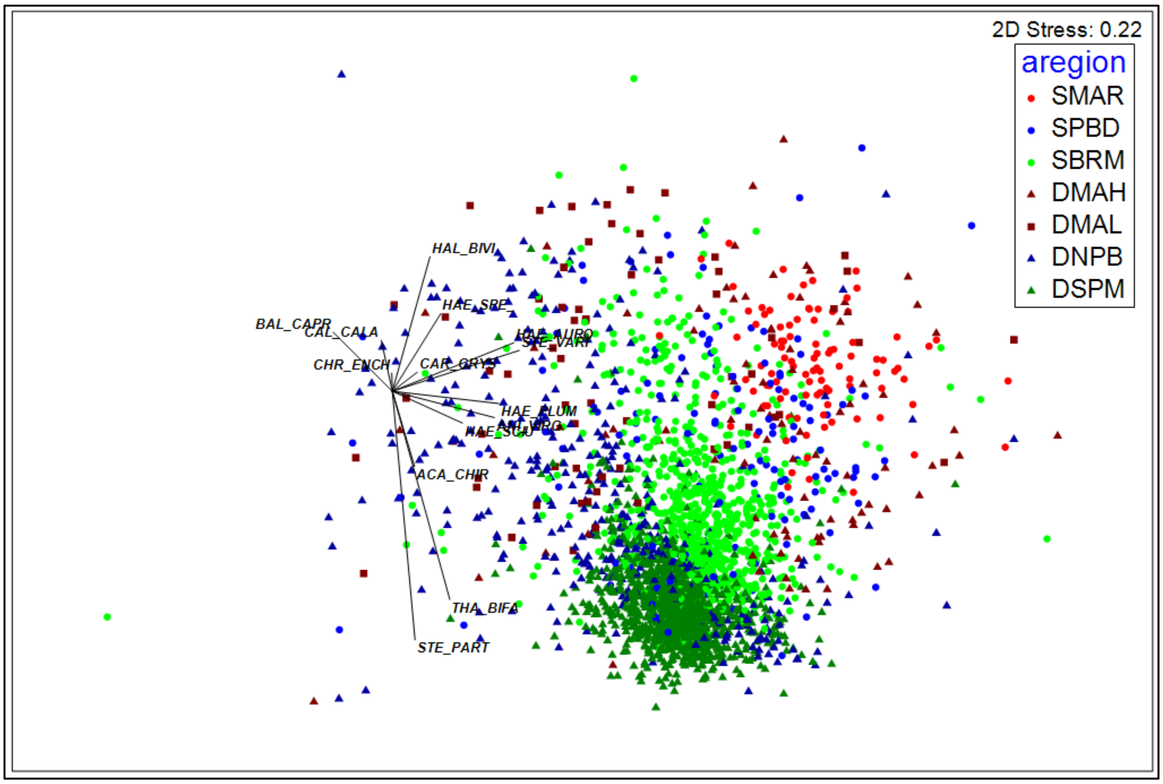


Figure 5. MDS plot of all sites (2012 – 2016) categorized by Reef Fish Assemblage Region. “S” indicates Shallow (≤10 m) and “D” indicates Deep (>10 m) assemblage regions. Shallow sites are denoted by solid circles and the Deep sites are solid triangles. Solid squares are Deep Low Relief. Lines indicate Pearson correlations of select species (chosen from SIMPER analyses) showing their influence on site similarities in the plot.

A better way to visualize the data is a through bootstrap means plot (Figure 6). This plot is analogous to a box plot in univariate analyses. In this case, a mean plot positon for a random subset of points is calculated for each category (assemblage region). This was repeated forty-three times to generate a mean and point cloud of means of each assemblage region indicating about 95% of the possible data means. Tightly clustered means indicate less variability and vice versa. Point clouds widely separated indicate more variability and overlapping point clouds indicate no difference between those groups. The bootstrap means plot of the reef fish assemblage regions shows distinct differences between the point cloud means of all categories. It also shows depth and regional spatial patterns. The shallow sites are all on the top of the plot and the deep sites on the bottom, indicating they are more similar to each other than to any of the deep sites. The plot also shows the northern regions on the left, the central regions in the middle, and the southern regions on the right. This pattern agrees with the regional directionality and transition from the temperate north to the subtropical south (Walker 2012; Walker and Gilliam, 2013; Fisco, 2016).

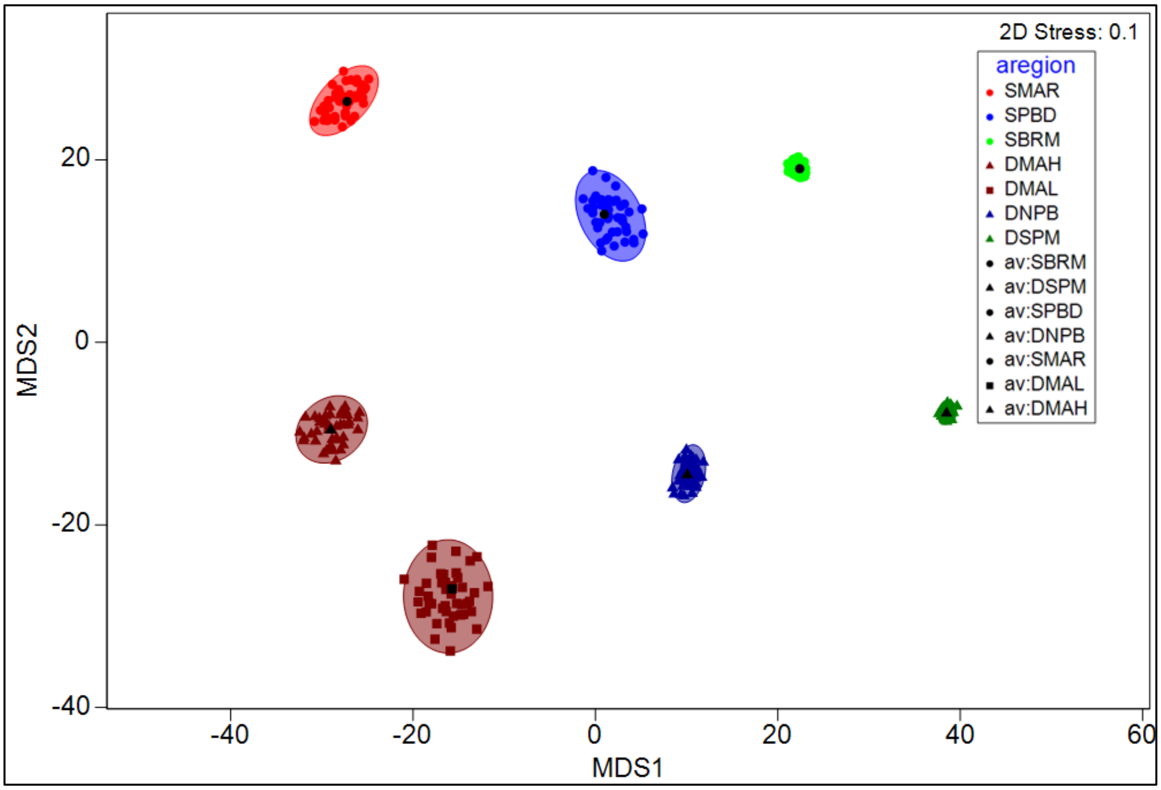


Figure 6. Plot of forty-three bootstrap means of the Reef Fish Assemblage Regions using all site data (2012 – 2016). "S" indicates Shallow (≤ 10 m) and "D" indicates Deep (> 10 m) Assemblage Regions. Shallow sites are denoted by solid circles and the Deep sites are solid triangles. Solid squares are Deep Low Relief. Point cloud indicates 95% about the mean. Point cloud separation indicates distinct categories.

All analysis of similarity (ANOSIM) pairwise comparisons between fish communities from reef fish assemblage regions were significant (Table 5). The R statistic, which indicates the strength of the difference (where 1 is the strongest and 0 is weakest), ranged from 0.974 between DSPM and SMAR to 0.176 between SPBD and DNPB. As expected, the strongest difference in the ANOSIM was between the Deep South Palm Beach Miami and the Shallow Martin reefs. Eight of the reef fish assemblage region comparisons exhibited strong differences (R stat > 0.5). Six comparisons exhibited moderate differences (R stat > 0.3), and two were weak (R stat < 0.3). The weakest comparisons were Shallow Palm Beach Deerfield with Deep North Palm Beach (R stat = 0.176) and Shallow Palm Beach Deerfield with Shallow Martin (R stat = 0.288), indicating that the assemblages in these regions were less distinct.

Similarity percentage analyses (SIMPER) identified the species that contributed most to the ANOSIM differences. These data are presented in Tables 6 and 7. The percentages in these tables are of transformed data (log(X+1)), therefore they are not the actual fish densities but rather the relative densities that show the differences in the analyses, which were also performed on transformed data. For this reason, these numbers should not be used when trying to compare actual densities. For example, the transformed density of *Halichoeres bivitattus* in Shallow Martin was 1.88 whereas the non-transformed SIMPER mean density was 10.15.

Table 5. Analysis of similarity comparisons of reef fish assemblages between Reef Fish Assemblage regions. The higher the R statistic, the stronger the difference.

Significant ANOSIM Pairwise Tests **Reef Fish Assemblage Regions**	**R Statistic**	**Significance Level %**
SBRM, DSPM	0.397	0.1
SBRM, SPBD	0.385	0.1
SBRM, DNPB	0.377	0.1
SBRM, SMAR	0.626	0.1
SBRM, DMAL	0.727	0.1
SBRM, DMAH	0.673	0.1
DSPM, SPBD	0.755	0.1
DSPM, DNPB	0.58	0.1
DSPM, SMAR	0.974	0.1
DSPM, DMAL	0.935	0.1
DSPM, DMAH	0.906	0.1
SPBD, DNPB	0.176	0.1
SPBD, SMAR	0.288	0.1
SPBD, DMAL	0.396	0.1
SPBD, DMAH	0.359	0.1
DNPB, SMAR	0.423	0.1

Table 6. Transformed similarity percentages between the Martin County Reef Fish Assemblage regions showing the species contributing the most to defining each region.

SMAR

Average similarity: 36.52

Species	Av. Trans Dens	Av. Sim	Sim/ SD	Cont%	Cum%
HAL_BIVI	1.88	5.70	1.33	15.6	15.60
ANI_VIRG	1.68	5.26	1.35	14.39	30.00
STE_VARI	1.40	4.11	1.19	11.26	41.26
HAE_AURO	1.85	3.45	0.67	9.45	50.71
ACA_CHIR	1.05	2.64	0.90	7.24	57.95
STE_LEUC	0.97	2.36	0.96	6.47	64.42
HAE_SPE_	1.43	1.94	0.47	5.31	69.73
HAE_PLUM	0.85	1.70	0.72	4.65	74.39

DMAL

Average similarity: 24.20

Species	Av. Trans Dens	Av. Sim	Sim/ SD	Cont%	Cum%
BAL_CAPR	1.28	4.57	1.06	18.89	18.89
HAL_BIVI	1.08	3.32	0.75	13.7	32.59
STE_PART	0.89	1.84	0.64	7.61	40.20
CAL_CALA	0.72	1.80	0.59	7.45	47.65
HAE_AURO	1.27	1.80	0.38	7.44	55.09
THA_BIFA	0.87	1.49	0.51	6.14	61.23
CHR_ENCH	0.76	1.43	0.41	5.9	67.13
ACA_CHIR	0.57	0.87	0.46	3.61	70.74

DMAH

Average similarity: 32.70

Species	Av. Trans Dens	Av. Sim	Sim/ SD	Cont%	Cum%
HAE_AURO	3.45	8.36	1.28	25.56	25.56
CHR_ENCH	1.32	2.14	0.55	6.54	32.10
BAL_CAPR	1.02	1.88	0.63	5.75	37.85
ACA_CHIR	0.97	1.83	0.93	5.59	43.44
STE_PART	1.29	1.77	0.75	5.42	48.86
ANI_VIRG	1.00	1.54	0.71	4.72	53.58
THA_BIFA	1.27	1.48	0.65	4.53	58.11
CAL_CALA	0.77	1.28	0.48	3.93	62.04
CAR_CRYS	0.95	1.12	0.64	3.42	65.46
STE_VARI	0.77	1.05	0.66	3.20	68.66
LUT_GRIS	0.87	1.01	0.58	3.08	71.74

Table 7. Transformed similarity percentages between the Deep and Shallow Reef Fish Assemblage regions south of Martin County showing the species contributing the most to defining each region.

SPBD

Average similarity: 26.72

Species	Av. Trans Dens	Av.Sim	Sim/SD	Cont%	Cum.%
HAL_BIVI	1.75	3.78	0.93	14.16	14.16
THA_BIFA	1.82	2.72	0.86	10.18	24.35
ACA_BAHI	1.39	2.67	1.06	10	34.34
HAE_SPE_	1.77	2.29	0.52	8.58	42.92
STE_PART	1.51	1.68	0.64	6.28	49.2
CAN_ROST	1.04	1.52	0.80	5.7	54.89
ABU_SAXA	1.2	1.37	0.51	5.14	60.03
ACA_CHIR	0.95	1.34	0.63	5.03	65.06
STE_VARI	0.73	0.98	0.63	3.66	68.72
ANI_VIRG	0.68	0.66	0.52	2.46	71.17

DNPB

Average similarity: 25.08

Species	Av. Trans Dens	Av.Sim	Sim/SD	Cont%	Cum.%
STE_PART	1.96	3.17	0.87	12.64	12.64
THA_BIFA	1.74	2.56	0.76	10.22	22.86
HAL_BIVI	0.94	1.93	0.55	7.69	30.54
BAL_CAPR	0.76	1.70	0.52	6.79	37.34
ACA_CHIR	0.93	1.63	0.70	6.51	43.84
SPA_ATOM	0.88	1.53	0.55	6.12	49.96
HAL_GARN	1.08	1.50	0.66	5.99	55.95
XYR_SPLE	0.66	1.44	0.41	5.74	61.69
CAN_ROST	0.71	1.40	0.76	5.58	67.27
ACA_BAHI	0.86	1.37	0.65	5.46	72.73

SBRM

Average similarity: 36.34

Species	Av. Trans Dens	Av.Sim	Sim/SD	Cont%	Cum.%
THA_BIFA	2.25	5.16	1.36	14.21	14.21
HAL_BIVI	1.48	3.78	1.17	10.39	24.6
ACA_BAHI	1.65	3.76	1.37	10.34	34.94
SPA_AURO	1.50	3.06	1.15	8.43	43.37
STE_PART	1.66	2.58	0.83	7.11	50.48
CAN_ROST	0.89	1.90	1.09	5.23	55.7
HAL_MACU	1.03	1.74	0.79	4.8	60.51
ACA_CHIR	1.01	1.72	0.71	4.72	65.23
HAE_PLUM	0.88	1.38	0.72	3.79	69.02
SCA_ISER	0.89	1.01	0.53	2.77	71.79

DSPM

Average similarity: 45.45

Species	Av. Trans Dens	Av.Sim	Sim/SD	Cont%	Cum.%
STE_PART	3.51	8.24	3.58	18.14	18.14
THA_BIFA	2.89	6.20	2.42	13.64	31.78
HAL_GARN	1.93	3.84	1.78	8.46	40.24
SPA_AURO	1.69	3.01	1.46	6.61	46.85
CAN_ROST	1.49	2.96	1.87	6.51	53.36
ACA_BAHI	1.58	2.72	1.20	5.98	59.34
CHA_SEDE	0.97	1.92	1.60	4.23	63.58
ACA_CHIR	1.25	1.90	0.90	4.18	67.76
COR_PERS	1.67	1.52	0.51	3.35	71.11

A fundamental principle in ecology assumes abiotic and biotic variables influence the distribution of all organisms, including marine fishes (Putman and Wratten, 1984; Recksiek et al., 2001). Abiotic variables such as temperature, salinity, depth, current, topographic complexity, and ecological processes including recruitment, competition, food availability, and predation all play roles in determining fish species distribution and abundance (Sale, 1991a and b). Where temperature and salinity are relatively consistent, other variables can structure associated fish assemblages, such as benthic habitat type and topographic complexity (Luckhurst and Luckhurst, 1978; Hixon and Beets, 1989; Ferrell and Bell, 1991; McCoy and Bell, 1991; McClanahan, 1994; Appeldoorn et al., 1997; Chabanet et al., 1997; Charton and Ruzafa, 1998; Friedlander and Parrish, 1998; Friedlander et al., 2003; Gratwicke and Speight, 2005 a and b). If a relationship exists between benthic habitats and reef fish assemblages, then the habitat map can be used as a surrogate map for the reef fish assmeblages. These analyses show that the southeast Florida reef fish assemblages are geographically distributed. These relationships are strongly related to depth, location along the coast, and to a lesser extent topographic complexity. The benthic habitat maps can be categorized to represent these strata; therefore, the benthic habitat maps can be used as a surrogate to map their distributions.

Depth is one of the most influential factors to the reef fish assemblage structure in this study which is typical in many local studies (Ferro et al., 2005; Walker et al., 2009; Gilliam et al., 2015). Correlations between fish assemblage attributes (i.e., density and/or species richness) and depth of habitat have also been documented on other coral reef habitats around the world (Luckhurst and Luckhurst, 1978; Gilmore et al., 1981; Friedlander and Parrish, 1998; Newman and Williams, 2001; Aguilar-Perera and Appeldoorn, 2008; Grober-Dunsmore et al., 2008). The southeast Florida benthic assemblages are also related to depth and correspond to the benthic habitat map classifications (Gilliam and Walker, 2011; B. Walker and Klug, 2014). Thus, the nearshore habitats, Ridge-Shallow, Colonized Pavement-Shallow, and Inner Reef were combined into one shallow strata to illustrate the reef fish assemblage based on its relationship to these benthic habitats.

Reef fish assemblage distribution also varied with geographic location. Along the southeast FRT, the amount and extent of distinct benthic habitats attenuates northward (Walker, 2012) and the benthic macroalgal (Lapointe, 2007) and coral (Moyer et al., 2003; Walker and Gilliam, 2013) assemblages vary with latitude. Many studies have related habitat structure to the structure of the reef fish assemblages (Friedlander and Parrish, 1998; Jones and Syms, 1998; Tuya et al., 2011). Along continental coasts, north-south faunal latitudinal boundaries fluctuate as warm-temperate and cold-temperate regions overlap in zones of transition. In these zones, species of different faunas comingle to various extents depending on yearly shifts in the oceanographic climate (Ebeling and Hixon, 1991). Variation in thermal regimes, either seasonal or with depth, may enhance local diversity in transitional zones between temperate and subtropical waters by promoting the co-occurrence of cool and warm water species (Stephens and Zerba, 1981). Southeast Florida is located at the convergence of the subtropical and temperate climate zones (Chen and Gerber, 1990; Lugo et al., 1999). Shifts in benthic assemblages are explained by differences in temperature regimes along the southeast Florida coast. Analyses of bottom temperatures along the reef tract show significant cold-water upwelling occurs more frequently and with higher intensity in the regions north of an area referred to as the Bahamas Fracture Zone (BFZ) (Gramer et al., 2017), a geological feature that coincides with the end of historical outer reef growth and where the Florida Current diverges from the coast (Klitgord et al., 1984). The division between DSPM and DNPB regions is situated along the BFZ, above which, the continental shelf widens and the Florida Current diverges from the coast. This divergence carries the warm tropical waters into the Gulf Stream and boundary eddies form causing the frequent episodes of cold water upwelling (Walker and Gilliam, 2013). This geologic feature affects the hydrography which in turn affects the benthic and fish assemblage distributions. Therefore, the habitat maps were split by these ecological zones and combined into strata relavant to reef fish assemblage biogeography.

Topography played a role in defining the assemblage regions, but to a lesser extent than depth and geographic location. Correlations of topographic relief have been documented for a range of coral reef fish metrics in many coral reef ecosystems, spanning multiple habitats and/or depths (Luckhurst and Luckhurst, 1978; Parrish et al., 1985; Friedlander and Parrish, 1998; Pittman et al., 2007; Pittman et al., 2009; Walker et al., 2009). Structural complexities in substrate can provide benefits to a variety of reef fishes. Live coral and other invertebrates living in the substrate can serve as a food source for some fish (Parrish et al., 1985; Friedlander and Parrish, 1998) while the structural complexity can serve as protection from physical or predatory stress (Hixon, 1991). The strongest difference in assemblages between low and high relief was between the Martin Low and Martin High assemblage regions where the Martin Low sites exhibited lower density and richness than the Martin High sites. In all other assemblage regions, relief did not play a significant role in differentiating the assemblages. This is expected, as there are many other stronger factors that influence total assemblage composition. This relationship could also change depending on the relative abundance of more rugosity-depedent fish. For example, snappers and groupers are typically in higher abundance in more topographically complex habitats (Sluka et al., 2001). More snappers and groupers on the reef might differentiate the assemblages into more topographic strata. The benthic habitat maps do not currently include a spatial representation of high and low topography in the region. It is possible to do with the available bathymetric datasets, however the relationship between reef fish assemblage and GIS-derived topography is quite low (Walker et al., 2009). The reef fish assemblages are likely responding to a finer scale topography that is not captured in the present bathymetry. The present bathmetry might still be a useful surrogate for illustrating the high relief areas, because they were used to identify strata in the survey design.

This study found a latitudinal decrease in the richness of the assemblages from a tropical assemblage in the south to a temperate assemblage in the north. This may be due to a decrease in the level of environmental variability through which reef species are able to survive and persist (Stephens and Zerba, 1981; Ebeling and Hixon, 1991). Tropical to temperate latitudinal differences in reef fish assemblages have been reported along the northern coast of Florida in the Indian river lagoon system (Gilmore et al., 1981). While their study includes inland habitats and assemblages north of the present study area, Gilmore et al (1981) noted that the warm-temperate Carolinian and the tropical Caribbean fish faunas overlap considerably in the east central Florida aquatic fish assemblages they studied. They proposed that the fishes of the Indian River Lagoon region in east central Florida originated in the Caribbean faunal province and apparently came into the region via the Florida Current while the warm-temperate Carolinian fishes distribution must be explained by adult migration with some aid from larval fishes transported via southbound counter-currents of the Florida Current and other inshore water mass movements (Gilmore et al., 1981). This study demonstrates that the transition between these two climate zones within the ichthyofaunal assemblages is present in habitats further south than the Gilmore et al. (1981) study covered. Typical coral reef fishes live among existing coral in relatively shallow tropical water where temperatures rarely drop below 20°C (Ebeling and Hixon, 1991). Over the course of a two year study, the temperature recorded in Martin County was below 20°C for about 2100 hours whereas the temperature recorded in the southern areas of the southeast FRT was below 20°C for approximately 300 hours (Walker and Gramer, *in prep*). This temperature regime difference is likely affecting the assemblage constituents. Examples of known geographic ranges for some of the species driving the differences in assemblages between the southernmost Shallow Broward-Miami and Deep South Palm Beach-Miami assemblage regions and the northernmost Shallow Martin, Deep Martin Low and High assemblage regions are displayed in Figure 7 (Kaschener et al., 2013). The species found in high densities at the northernmost sites (right) have ranges that extend much farther north, indicating they live in colder water temperatures throughout most of their range. The ranges of the species found in much higher densities farther south (left) diminish rapidly to the north indicating they are less tolerant of colder conditions (i.e., more tropical). For example, two of the species that have higher densities in the Shallow Martin, Deep Martin Low and High assemblage regions, Tomtate (*Haemulon aurolineatum*) and Spottail Seabream (*Diplodus holbrookii*), are found from 43° N to 33° S and 40° N to 20° N, respectively, whereas two of the species with higher densities in the South Palm Beach, and Broward-Miami assemblage regions, Bluehead Wrasse (*Thalassoma bifasciatum*) and Redband Parrotfish (*Sparisoma aurofrenatum*), are only found 33° N to 8° N and 32° N to 7° N, respectively (Robins and Ray, 1986). One species, Black Seabass (*Centropristis striata*), was observed 47 times in the Deep Martin Low and High assemblage regions combined and only three times in the North Palm Beach and South Palm Beach-Miami assemblage regions combined. The Black Seabass is described as a temperate species with a range from Maine to northeastern Florida that can reach extreme southern Florida during cold winters (Robins and Ray, 1986). Interestingly, none of the samples were conducted in winter, but cold-water upwelling is known to occur.

Finally, we compared the reef fish assemblage regions with the Coral Reef Ecoregions of Walker (2012) and Walker and Gilliam (2013) to understand the differences. One big distinction is that the reef fish assemblage regions differ depending on habitat depth. The Shallow Broward-Miami assemblage region (SBRM) aligns with the Broward-Miami Ecoregion and the Martin assemblage region (SMAR) aligns with the Martin Ecoregion; however, Shallow North Palm Beach Deerfield (SPBD) habitats span the North Palm Beach, South Palm Beach, and Deerfield Coral Reef Ecoregions. The Deep South Palm Beach – Miami assemblage region (DSPM) spans the Broward-Miami, Deerfield, and South Palm Beach Ecoregions, stopping at the BFZ. The Deep North Palm (DNPB) and Deep Martin (DMAH and DMAL) assemblage regions match the North Palm and Martin Ecoregions.

The Coral Reef Ecoregions were based on habitat type and morphologies and not by one particular group of species. It is not surprising that certain groups conform in some places and not others. Klug (2015) found that in the shallow habitats, coral communities and benthic cover supported the Biscayne and Broward-Miami separation. Walker and Gilliam (2013) found benthic community differences across the shelf and with habitats further south. A comprehensive regional benthic assessment has yet to be conducted on a scale that would facilitate a similar benthic analysis as those conducted herein. Such an analysis would elucidate how the communities and other major benthic functional groups (corals, algae, and gorgonians) are distributed in relation to the Coral Reef Ecoregions and Reef Fish Assemblage Regions defined herein.

These analyses enable an ecosystem management approach to the southeast Florida reef tract. Ecosystem management calls for moving away from single species management to a more holistic management approach. The separation of fishery-independent data by the relative density of all species in the assemblage focuses on the broader regional differences in fish assemblages instead of the distribution of individual species. It provides distribution maps of the different assemblages and defines those assemblages based on their statistical similarities. This facilitates dividing regional management actions based on the ecosystem (and not arbitrary geopolitical boundaries, region-wide, or by individual species distributions). Ecosystem management may also rely on ecosystem modeling to answer specific management questions (Grüss et al., 2016). These models rely heavily on various types of data for their formulation, calibration, validation, and use in scenario analyses (Grüss et al., 2017). The quality of their predictions is reliant on the quality of the available input data. Thus, improving the collection and compilation of distribution maps is critical to ensure that the predictions are sufficiently reliable to inform decisionmaking (Grüss et al., 2016; Tarnecki et al., 2016). If coupled with a similar regional benthic analysis on empirical data, these analyses would provide a much-needed understanding of the southeast Florida reef fauna biogeography for ecosystem mamangement and decisionmaking.

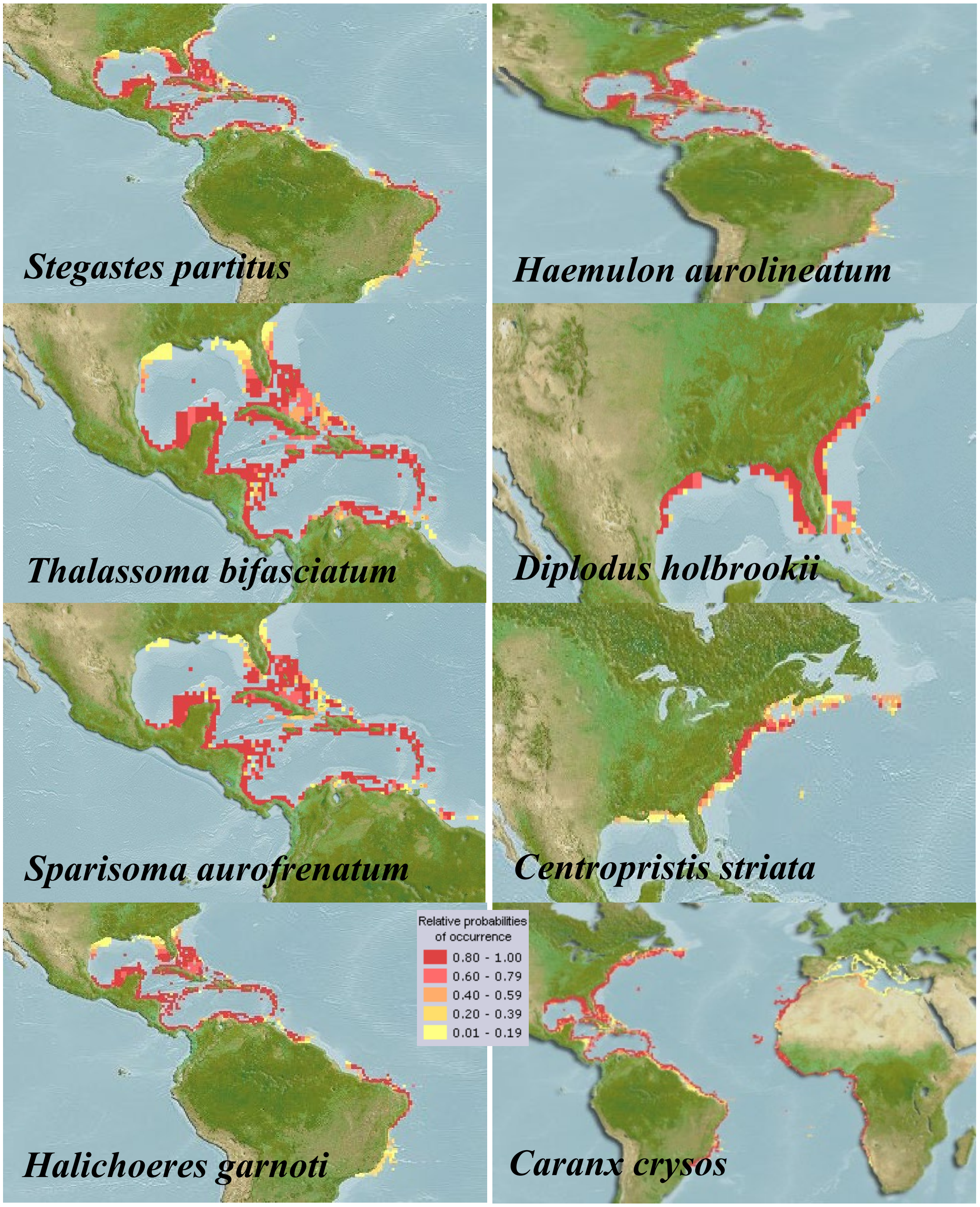


Figure 7. Examples of known ranges for some of the species driving the differences in assemblages between the southernmost Broward-Miami region and the northernmost Martin region (Kaschener et al., 2013).

4.1.3. Exploited Species

Most of the exploited species evaluated here indicated a cosmopolitan but unequal distribution across all strata and varying degrees of inter-annual variation (see Kilfoyle et al., 2015). Among the eight target species, with all years and strata combined, White Grunt and Gray Triggerfish exhibited the highest densities (Figure 8). Yellowtail Snapper, Bluestriped Grunt, and Gray Snapper were ranked in the middle, while Hogfish, Mutton Snapper, and Red Grouper exhibited the lowest densities. Density for many species was variable when broken down by reef fish assemblage region (Figure 9). When the data for each species were divided into separate pre-exploited and exploited phases, it was clear that for many of these species (Red Grouper, Mutton Snapper, Gray Snapper, Yellowtail Snapper, Hogfish, and Gray Triggerfish) that the pre-exploited phase was largely responsible for the majority proportion of each population within the southeast Florida region (see Figures 12, 19, 26, 33, 55, 62). This was further confirmed by partitioning of the data into discrete size classes (by 5 cm increments) and plotting the total number of observations from each (Figures 13, 20, 27, 34, 56, 63). In contrast to the aforementioned species, with White and Bluestriped Grunts it appears that both pre-exploited and exploited phase life-stages were responsible for driving the observed trends (Figures 40, 41, 47, and 48). It is noteworthy that many of the exploited species have a very low proportion of newly settled and early juvenile size ranges (0-5 cm). This indicates that either nursery areas were not sampled, the point-count methodology was not ideal for detecting newly settled or juvenile members of these and other species, recruitment was low, or some combination thereof.

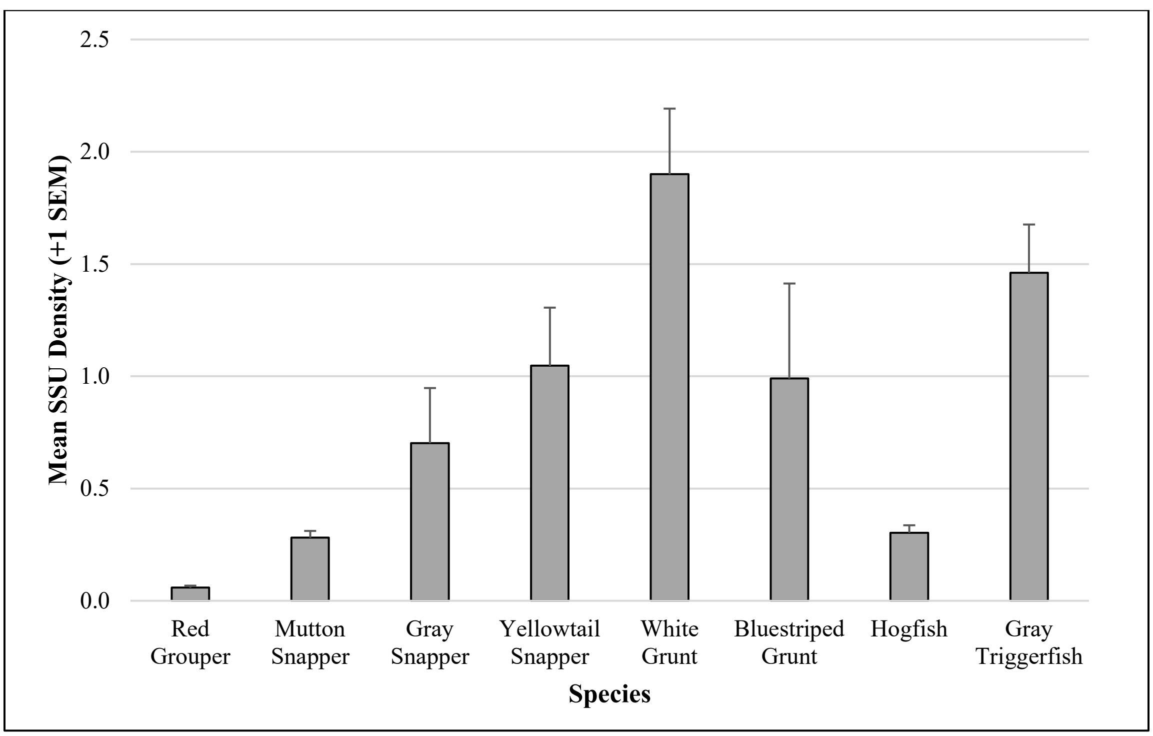


Figure 8. Mean SSU density for exploited species, with all years and strata combined.

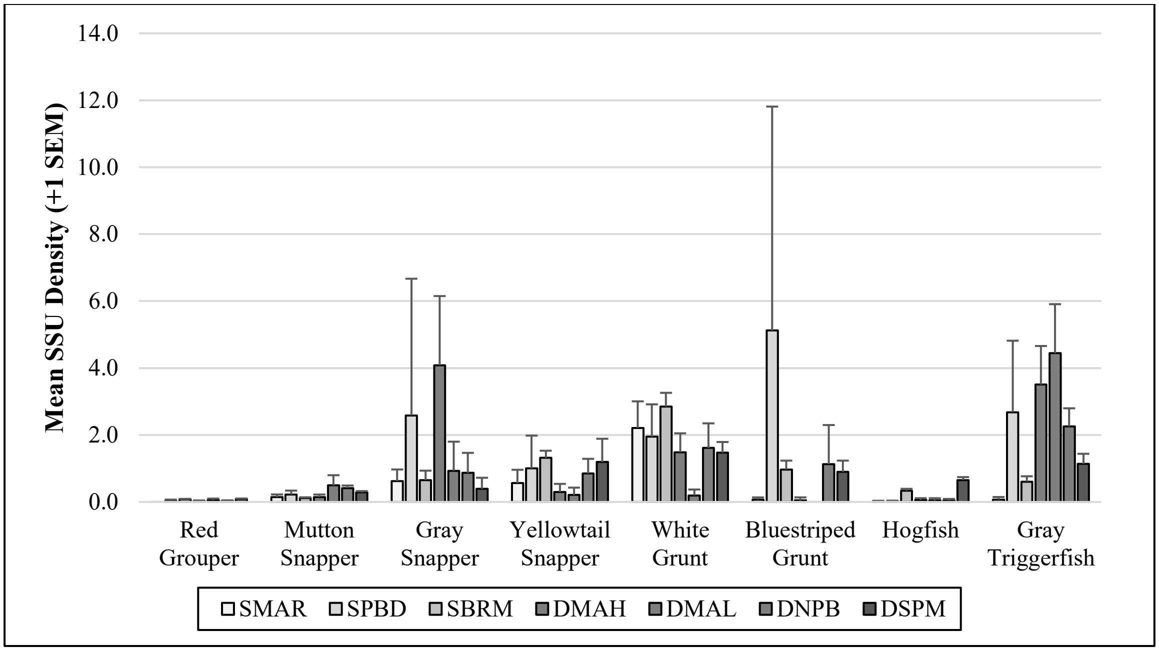


Figure 9. Mean SSU density for exploited species by Reef Fish Assemblage Region, with all years combined.

During the five-year survey period, the following species were encountered in low numbers (≤50 individuals total) (Table 8): Tarpon (*Megalops atlanticus*); Common Snook (*Centropomus undecimalis*); Cobia (*Rachycentron canadum*); Greater Amberjack (*Seriola dumerili*). There were also multiple species of groupers and snappers: groupers - Rock Hind (*Epinephelus adscensionis*), Red Hind (*E. guttatus*), Goliath (*E. itajara*), Black (*Mycteroperca bonaci*), Gag (*M. microlepis*), and Scamp (*M. phenax*); snappers - Blackfin (*Lutjanus buccanella*), Red (*L. campechanus*), Cubera (*L. cyanopterus*), and Dog (*L. jocu*). Several large schools (20-50 individuals) of >1 m TL Great Barracuda (*Sphyraena barracuda*) were observed during the last two years of the study period (although they were not recorded during surveys and are therefore absent in both the dataset and analyses) (K. Kilfoyle, personal observations).

Interestingly, none (zero) of the following species known to exist in southeast Florida were recorded: groupers - Speckled Hind (*E. drummondhayi*), Warsaw (*E. nigritus*), Snowy (*E. niveatus*), Nassau (*E. striatus*), Yellowmouth (*M. interstitialis*), Tiger (*M. tigris*), Yellowfin (*M. venenosa*), Yellowedge (*Hyporthodus flavolimbatus*), and Misty (*H. mystacinus*); snappers - Black (*Apsilus dentatus*), Queen (*Etelis oculatus*), Silk (*L. vivanus*), and Wenchman (*Pristipomoides macropthalmus*). Although many of these species are primarily associated with deeper habitats (>33 m) beyond the scope of this survey, several species have been documented from low-relief hardbottom habitats and vessel-reefs in Broward County in the 50-120 m depth range during remotely operated vehicle (ROV) surveys (Bryan et al., 2013).

Table 8. Species of economic importance: the total number of fish from all years combined (using mean SSU density totals), total number of legal/exploited phase individuals (Expl.), the percentage of legal/exploited phase individuals, average (%) Density ($\bar{D}$) (fish/SSU), average Percent Occurrence ($\bar{P}$) per SSU, the mean, minimum, and maximum observed total lengths (M [Min, Max]), and the minimum legal/exploited sizes. Species are listed in phylogenetic order and lengths are listed in centimeters unless otherwise noted.

Species	Total	Expl.	%	D	P	M (Min, Max)	Min. Expl. Size
Tarpon	8	n/a	n/a	0.002	0.003	134 (100, 200)	catch-and-release
Lionfish	404	n/a	n/a	0.1	0.13	21 (3, 43)	unregulated
Common Snook	31	30	96.8	0.01	0.005	75 (65, 88)	71.1 (28")
Black Seabass	492	4	0.8	0.08	0.02	19 (7, 41)	33.0 (13")
Coney	82	n/a	n/a	0.03	0.04	19 (6, 37)	unregulated
Graysby	662	n/a	n/a	0.2	0.24	17 (3, 45)	unregulated
Red Hind	20	n/a	n/a	0.009	0.01	21 (9, 36)	unregulated
Rock Hind	22	n/a	n/a	0.007	0.01	21 (7, 40)	unregulated
Goliath Grouper	38	n/a	n/a	0.01	0.009	167 (90, 250)	prohibited
Red Grouper	180	16	8.9	0.06	0.09	36 (4, 90)	50.8 (20")
Black Grouper	22	2	9.1	0.009	0.01	40 (7, 75)	61.0 (24")
Gag Grouper	15	2	13.3	0.005	0.008	37 (17, 90)	61.0 (24")
Scamp Grouper	35	2	5.7	0.008	0.01	28 (10, 55)	50.8 (20")
Cobia	3	2	66.7	0.0009	0.002	103 (80, 125)	83.8 (33")
Greater Amberjack	46	2	4.4	0.02	0.007	36 (13, 95)	71.1 (28")
Blackfin Snapper	7	0	0.0	0.002	0.002	10 (3, 19)	30.5 (12")
Cubera Snapper	3	2	80.0	0.0006	0.0009	41 (20, 50)	30.5 (12")
Dog Snapper	10	8	80.0	0.004	0.005	36 (25, 51)	30.5 (12")
Gray Snapper	1926	613	31.8	0.7	0.1	22 (2, 46)	25.4 (10")
Lane Snapper	2719	1312	48.3	0.8	0.08	16 (1, 38)	20.3 (8")
Mahogany Snapper	66	0	0.0	0.02	0.01	18 (5, 29)	30.5 (12")
Mutton Snapper	671	72	10.7	0.3	0.3	34 (3, 82)	45.7 (18")
Red Snapper	3	0	0.0	0.0001	0.00001	32 (18, 50)	50.8 (20")
Schoolmaster Snapper	166	31	18.7	0.07	0.02	23 (7, 39)	25.4 (10")
Vermillion Snapper	238	12	5.0	0.04	0.9	20 (3, 34)	30.5 (12")
Yellowtail Snapper	2693	232	8.6	1.1	0.3	18 (1, 45)	30.5 (12")
White Grunt	5057	1893	37.4	1.9	0.4	18 (2, 45)	20.3 (8")
Bluestriped Grunt	2869	1246	43.4	1.0	0.2	20 (2, 37)	20.3 (8")
Hogfish	1024	252	24.6	0.3	0.2	25 (3, 60)	30.5 (12")
Great Barracuda	81	n/a	n/a	0.03	0.02	113 (35, 200)	unregulated
Cero Mackerel	71	n/a	n/a	0.04	0.04	44 (25, 80)	unregulated
Gray Triggerfish	3569	266	7.4	1.5	0.4	21 (4, 46)	35.6 (14")

These figures and those that follow are meant to serve only as a general comparison of the fishes within the geographic and temporal survey domain of this study (i.e., 0-33 m depth, natural reef and hardbottom habitats only, diurnal, May through October). It is, however, important to note that this survey targets the most easily accessible and heavily utilized coastal underwater habitats that provide foundational support for economically important activities such as recreational fishing/spearfishing, scuba diving, and tourism. Many factors may contribute to observed differences between individual species densities, such as: trophic level, habitat preference, habitat availability, life history characteristics, growth rates, behavioral tendencies (solitary vs. schooling), biogeographical distributions, seasonality, and depth. For example, in comparing the observed density of Red Grouper to White Grunt, one must take into account that the grouper, as a slow-growing and generally solitary upper trophic level predator that can grow quite large, would be expected to exhibit a lower encounter rate relative to the smaller and lower trophic level schooling grunt, even in the absence of fishing pressure or habitat degradation.

The latitudinal distribution pattern of smaller populations of exploited and non-target species in the northern FRT is not clear (Appendix 2). In some cases, this distribution may indicate some substrate associated with the species is heterogeneously distributed, with more in the north than in the south, or vice versa. For example, Staghorn Coral (*Acropora cervicornis*) and mangroves, which are associated with Threespot Damselfish (*Stegastes planifrons*), Gray Snapper, and Great Barracuda (*Sphyraena barracuda*) abundances, respectively, are sparsely available or highly localized in the northern portion of the FRT (Nagelkerken et al., 2000; Precht et al., 2010). In other cases, the northern portion of the FRT may simply represent the northernmost or southernmost part of a species' range due to habitat preference or seasonality (e.g., Gag, Black, and Nassau Groupers, Black Seabass, Whitebone Porgy). However, some local populations appear egregiously low in southeast Florida in comparison to the southernmost tracts, and these species would likely benefit from immediate management attention.

4.1.4. Exploited Species: Red Grouper

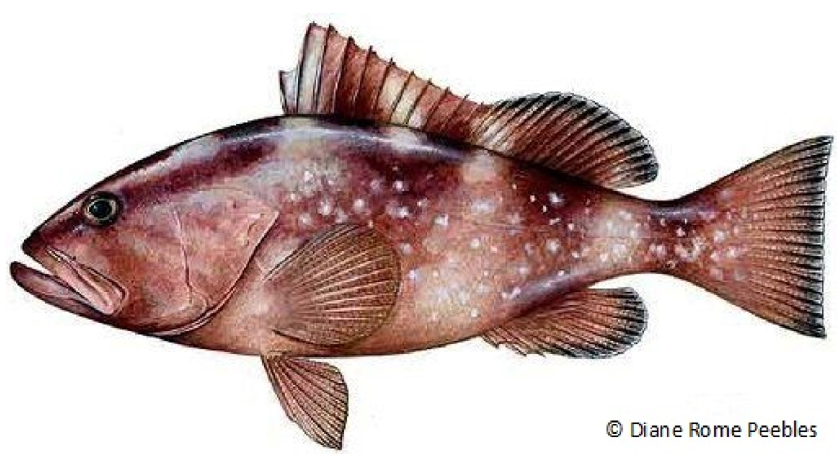


Figure 10. Red Grouper (*Epinephelus morio*).

Groupers (family Serranidae) are considered the most economically valuable finfish group in Florida (Jory and Iversen, 1989). The Red Grouper is one of the most commonly encountered large serranids on the coral reef and hardbottom habitats of southeast Florida. It is closely related to the Nassau Grouper (*Epinephelus striatus*) which is exceedingly rare or altogether absent in southeast Florida and was not seen during this study. The Red Grouper center of abundance has been reported as the West Florida Shelf and eastern Gulf of Mexico, as well as the Yucatan Peninsula, but it is considered as a primarily continental species, with a wider distribution (North Carolina to Brazil) than most other western central Atlantic groupers (Roe, 1976; Heemstra and Randall, 1993). Moe (1969) summarized the offshore movement of Red Grouper using evidence from mark and recapture studies. As juveniles and subadults they reside in shallow water habitats (3-18 m). An ontogenetic shift occurs around 40 to 45 cm SL and 4 to 6 years of age (which coincides with the onset of sexual maturity), and they move offshore to depths greater than 36 m (this may vary depending upon local habitat availability). Their maximum habitat depth is about 120 m, although there have been reports of Red Grouper occurring below 200 m depth. They typically exhibit strong site fidelity as juveniles and subadults, but as they mature they may migrate, with some reported as traveling as far as 28-72 km from their original tagging locations (Moe, 1969).

Large members of the grouper family share many life history traits that are widely believed to increase their vulnerability to exploitation, such as: carnivorous diet, slow growth, large size at reproductive maturity, long life-span (30+ years for Red Grouper), relative ease of capture, and ability to be harvested with a wide range of fishing gear (Manooch, 1987; Ralston, 1987). They also have a reputation for being inquisitive, often approaching scuba divers at close range and remaining nearby; a behavioral characteristic that makes them particularly susceptible to spearfishing. Most groupers, including the Red Grouper, are protogynous sequential hermaphrodites, beginning life as females and transitioning to males later on. However, there are many factors that may contribute to the timing for sex change within this family (density dependence, environmental fluctuations, etc.), and the driving mechanisms are still poorly understood. Stock assessments need to account for protogynous sex change, as this life history trait, when accompanied by size or age-selective harvest, applies a greater amount of pressure on males for protogynous species when compared to gonochoristic species (Bannerot, 1984; Bannerot et al., 1987; Alonzo et al., 2008). In this situation, it is entirely possible that the potential for population growth may become limited by the lower proportion of males (Coleman et al., 2011; SEDAR, 2017). Red Groupers are not reported to form large spawning aggregations (unlike Nassau, Goliath, Gag, and Black Grouper), but may form small polygynous spawning groups dispersed over wide areas (Zatacoff et al., 2004). Whether these occur in well-defined areas is still unknown, but if they do those areas would be prime candidates for some level of management for protection of spawning aggregations.

Red Grouper was the 77th most frequently observed and 119th most abundant species, with an average percent occurrence ($\bar{P}$) of 8.9% and average density ($\bar{D}$) of 0.05 fishes/SSU (Appendix 2). Comparatively, there were fewer Red Groupers in southeast Florida than the Florida Keys ($\bar{P}$=13.8, $\bar{D}$=0.11), with a large disparity compared to the

Dry Tortugas ($\bar{P}$=65.2, $\bar{D}$=0.60). The sample size was relatively small for this species in the southeast Florida region. Of the 5,290 surveys, only 398 Red Groupers were recorded. Red Grouper densities by reef fish assemblage region were low in both shallow and deep habitats (Figure 11). Density of exploitation phases by region revealed the majority of the population consisted of pre-exploited fish (<50 cm TL), with more of the exploited sizes in the deeper assemblage regions (Figure 12). Length frequency analysis showed that 64% of the population was between 30 and 44 cm TL (Figure 13), while the exploited phase (≥50 cm) comprised only 8.4% of the observed population. Abundance at length by reef fish assemblage region indicated a general increase in the pre-exploited phase towards the southern end of the sampling domain (Figure 14). The largest portion of the population was found in the Deep South Palm Beach Miami (DSPM) and Shallow Broward-Miami (SBRM) regions: 42% and 36%, respectively.

Reports from the eastern Gulf of Mexico estimated that length at 50% reproductive maturity (L_{50}) for this species was 40-50 cm, and length at 50% transition from female to male occurred at 80-90 cm around age 13 (Collins, 2002). A study in North and South Carolina reported L_{50} was 2.4 years and 48.7 cm TL, and age at 50% transition was 7.2 years at 69.0 cm TL (Burgos et al., 2007). Jory and Iverson (1989) reported that Red Groupers in south Florida usually begin changing from female to male between the ages of 5 and 10 at lengths ranging from 27.5-50.0 cm, and females mature at 4 to 6 years of age, reaching their greatest reproductive potential at 8 to 12 years.

The minimum legal size of harvest in the state of Florida is 20” TL (50.8 cm in both state and federal waters) (FWC, 2017a). In this study, the average recorded size for Red Grouper was 35.3 cm, and the average size of the exploited-phase (Lbar) was 57.8 cm. These results support previous findings which suggest that spawning stock has been declining since the mid-2000s in the southeastern US, and that the stock is experiencing overfishing and is currently overfished (SEDAR, 2017).

The lack of large individuals within the southeast Florida region survey domain (with a cutoff at 33 m depth) might be attributable to naturally occurring ontogenetic shifts that would place the larger individuals beyond the reach of the survey. However, comparison of the southeast Florida region length frequency and abundance at length against those from the Florida Keys and Dry Tortugas suggests otherwise (Figures 15 and 16). The presence of greater numbers of the exploited size classes in the Florida Keys and Dry Tortugas, where protection from harvest exists in certain areas, could imply that fishing pressure is a major contributor to the lower observed densities from the southeast Florida region where no protected areas have been established. However, there are also differences in nursery, juvenile, and adult habitat availability between the northern and southern portions of the FRT, which may be major contributing factors as well.

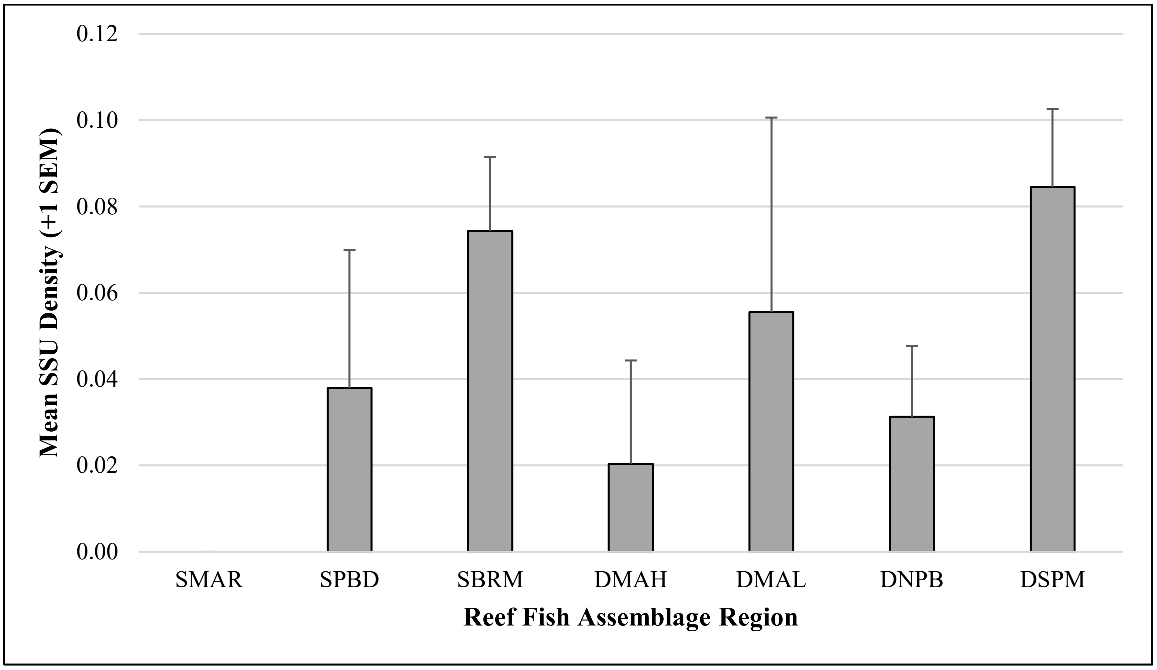


Figure 11. Red Grouper mean SSU density by Reef Fish Assemblage Region, with all years combined.

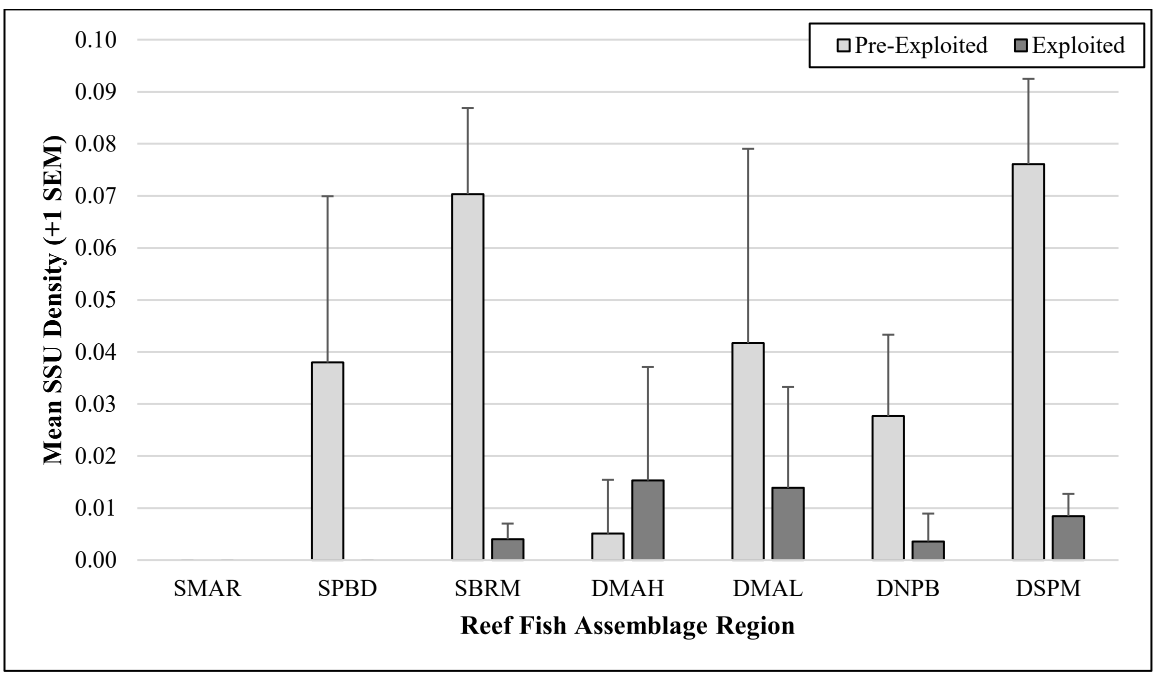


Figure 12. Mean SSU density of Red Grouper by Reef Fish Assemblage Region; pre-exploited and exploited life-stage comparison with all years combined.

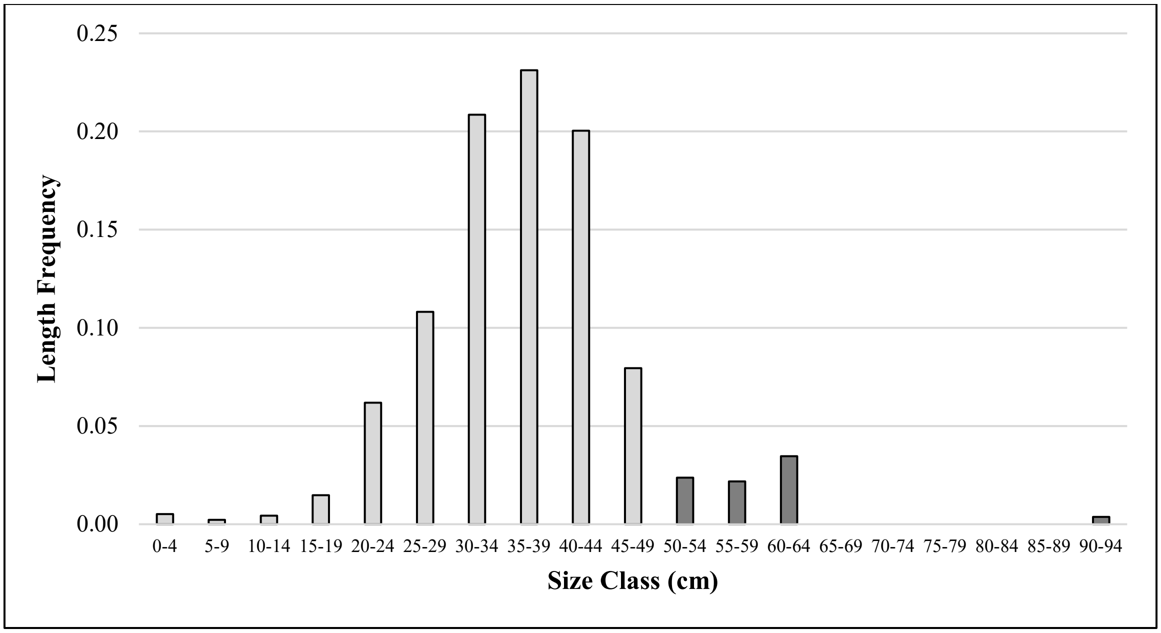


Figure 13. Domain-wide length frequency of Red Grouper by size class, with all years and strata combined. Darker gray indicates exploited size classes; minimum legal size of harvest for this species is 20” TL (50.8 cm in state and federal waters).

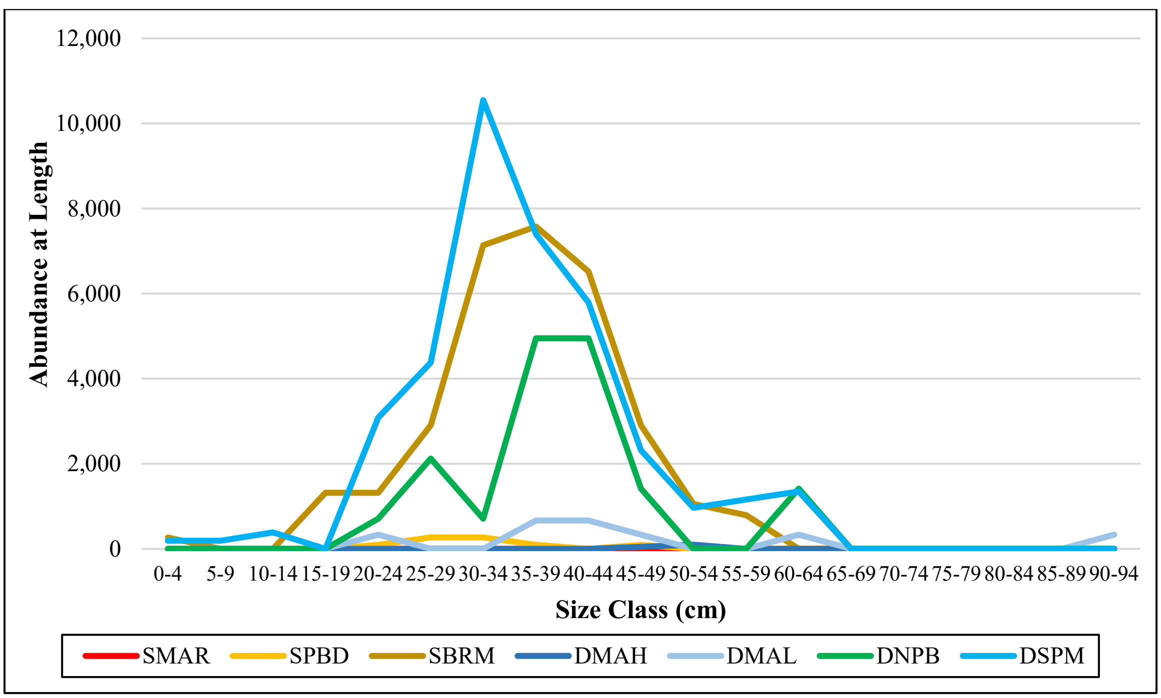


Figure 14. Length frequency comparison for Red Grouper by total estimated assemblage region-wide population size (abundance at length) for southeast Florida, with all years combined.

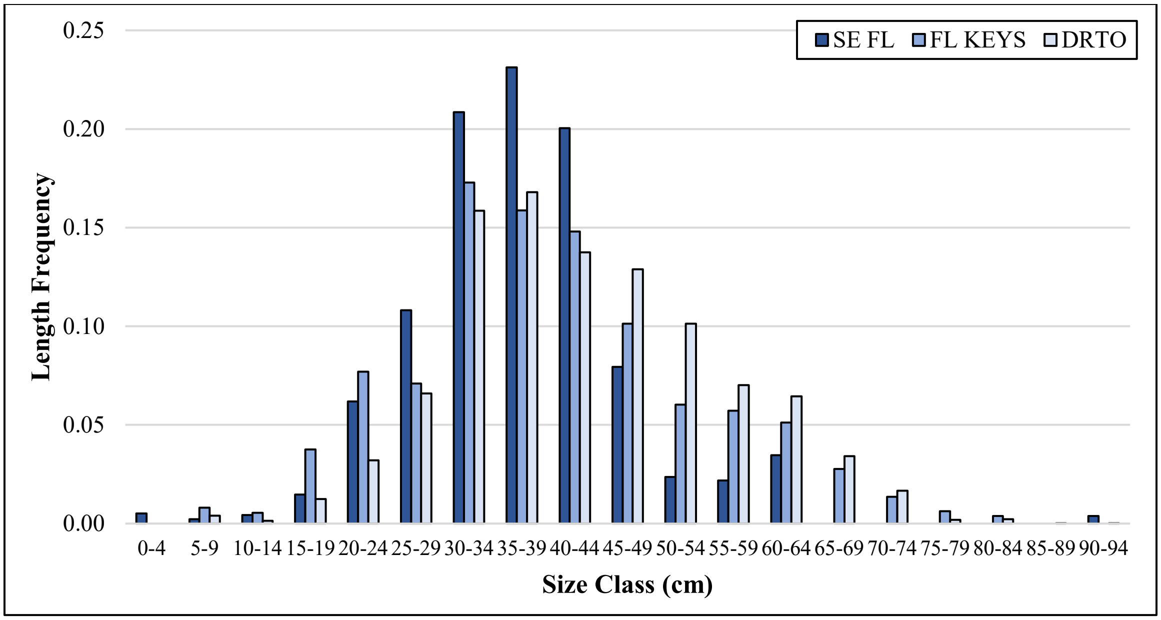


Figure 15. Length frequency comparison for Red Grouper, by proportion (all bars sum to 1 for each region regardless of sample size or number of fish counted), for the southeast Florida region, Florida Keys, and Dry Tortugas during the 2012-2016 time-period.

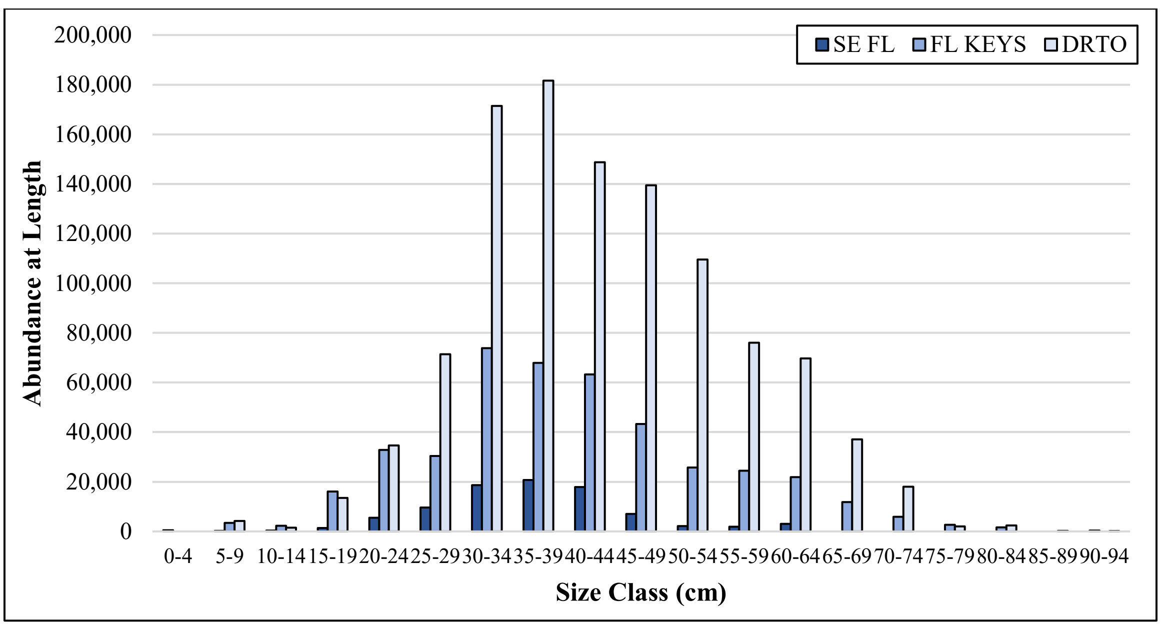


Figure 16. Length frequency comparison for Red Grouper, by total estimated domain-wide population size (abundance at length), for the southeast Florida region, the Florida Keys, and the Dry Tortugas for the 2012-2016 time period.

4.1.5. Exploited Species: Mutton Snapper

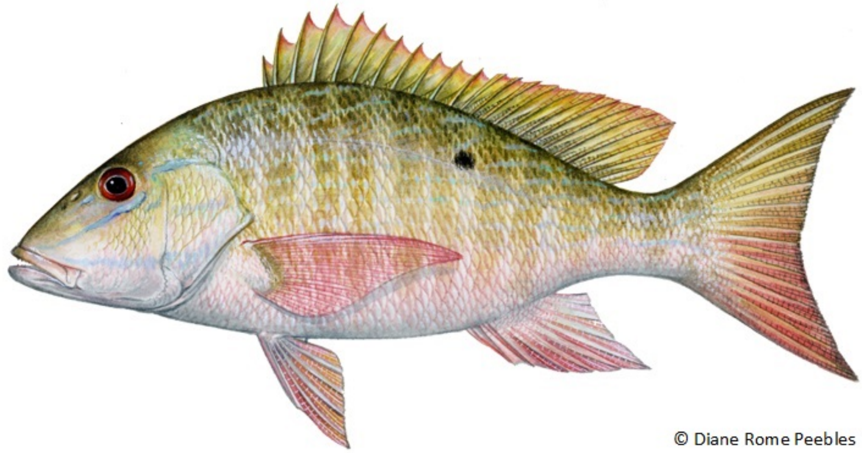


Figure 17. Mutton Snapper (*Lutjanus analis*).

One of the largest and most prized snapper species (Lutjanidae) along the eastern coast of Florida, Mutton Snapper have a geographic range that extends from Massachusetts to Brazil, but are most abundant in the Florida Keys, Bahamas, and the Caribbean (Vergara, 1977; Allen 1985; Cervignon, 1993; Burton, 2002). Most landings are reported from Cape Canaveral through the Florida Keys and Dry Tortugas, with landings north of this region being generally uncommon (Cuellar et al., 1996). Juveniles are known to utilize mangroves and seagrass beds as nursery habitats in bays and estuaries, although they are also found along with subadults on various sand and hardbottom habitats. Adults are usually found in more complex habitats offshore, such as coral reefs, ledges, and rocky outcroppings (Allen, 1985; Nagelkerken et al., 2000; Burton, 2002). Mutton Snappers have a reputation among the diving and spearfishing community as being wary and difficult to approach. Typically, they remain solitary until spawning season, when they form large spawning aggregations and display high site fidelity at specific locations (Domeier and Colin, 1997). While nocturnal foraging is the norm for many snapper species, Mutton Snapper are reported to feed continuously throughout the day (Watanabe et al., 2001).

Mutton Snapper was the 25th most frequently observed and 61st most abundant species, in this study with an average percent occurrence ($\bar{P}$) of 30.1% and average density ($\bar{D}$) of 0.28 fishes/SSU (Appendix 2). On average, southeast Florida had comparable numbers of Mutton Snapper as the Florida Keys ($\bar{P}$=24.7, $\bar{D}$=0.26) and Dry Tortugas ($\bar{P}$=39.3, $\bar{D}$=0.30). Comparison of Mutton Snapper densities by reef fish assemblage region revealed a high degree of consistency within the shallow and deep assemblage regions with no apparent increase in density moving either north or south through the survey domain (Figure 18). There was also greater mean density in the deeper regions than the shallow, with the exception of the Deep Martin High (DMAH), which was more similar to the shallow assemblage regions. Comparison of population densities by their exploitation phases revealed a similar pattern between assemblage regions, with comparable numbers in the shallow and deep regions, and more of both the pre-exploited and exploited phases in the deep regions (Figure 19). The proportion of the exploited phase was also larger in the deep regions than the shallow.

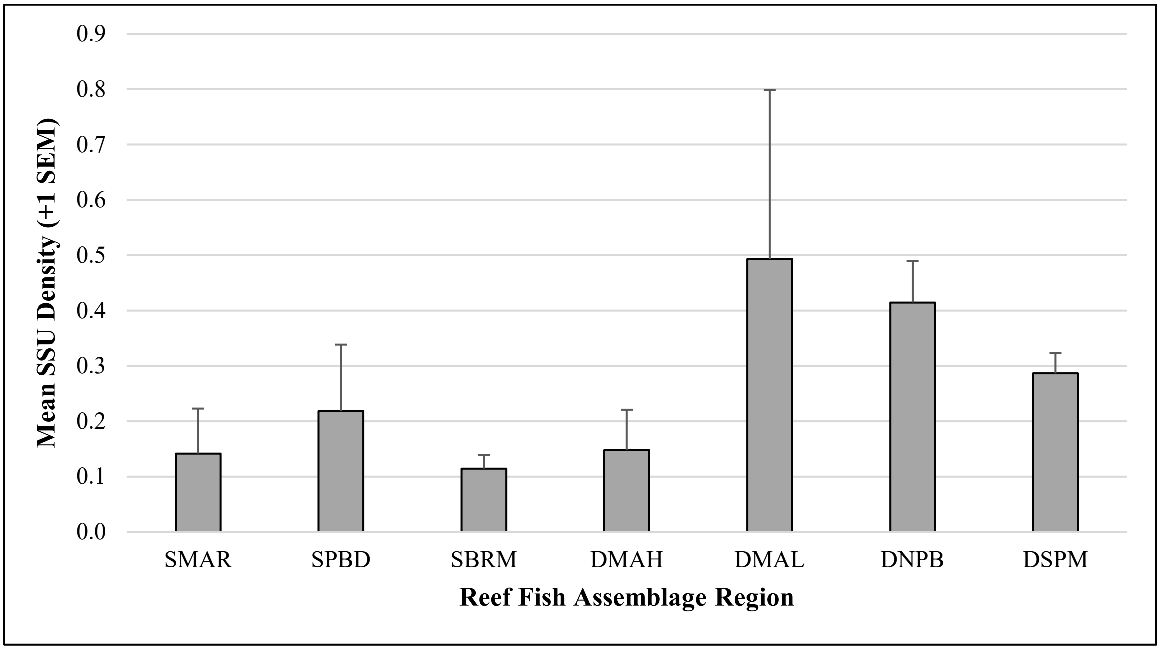


Figure 18. Mutton Snapper mean SSU density by Reef Fish Assemblage Region, with all years combined.

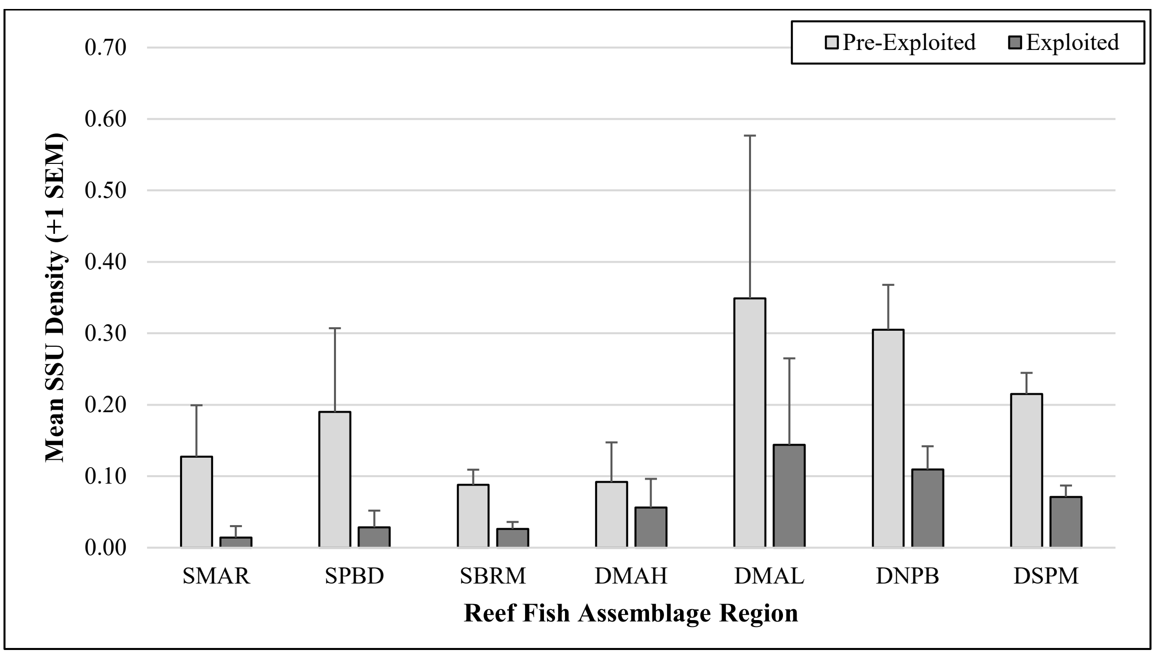


Figure 19. Mean SSU density of Mutton Snapper by Reef Fish Assemblage Region; pre-exploited and exploited life-stage comparison with all years combined.

Length frequency analyses for Mutton Snapper in southeast Florida again showed the majority of the population was classified as the pre-exploited phase, with the exploited phase contributing 24.9% of the total (Figure 20). Abundance at length by reef fish assemblage region indicated there are more Mutton Snapper in the Deep North Palm Beach (DNPB) assemblage region, followed by the Deep South Palm Beach Miami (DSPM) and the Shallow Broward Miami (SBRM) regions (Figure 21). Examination of length frequency curves along the entire FRT indicated similarity between the southeast Florida region and the Florida Keys, but with considerable differences to the Dry Tortugas (Figure 22). The latter was represented by a greater proportion of the larger size classes, but also, interestingly, fewer of the smaller size classes. In terms of overall population size, the Florida Keys was greater than both the southeast Florida region and Dry Tortugas, but both southeast Florida and the Florida Keys are skewed towards the pre-exploited phase (Figure 23). Perhaps not surprisingly, the largest recorded Mutton Snapper in the state of Florida was caught in the Dry Tortugas in 1998 (IGFA, 2017a).

Female Mutton Snapper are reportedly larger than the males throughout their range. The estimated size at 50% maturity (L_{50}) has been reported as 33.0 cm FL for males and 41.4 cm FL for females, while the size at full reproductive maturity has been reported as 38.0 to 43.0 cm FL for males and 45.0 to 47.0 cm FL for females (Figuerola-Fernández and Torres-Ruiz, 2001; Watanabe et al., 2005; Cummings, 2007). The minimum legal size of harvest for this species in the state of Florida during this study was 16" TL (40 cm in both state and federal waters) (FWC, 2017a). The average size for Mutton Snapper in this study was 34.0 cm, and the average size of the exploited-phase (Lbar) was 44.9 cm.

The Mutton Snapper fishery peaked in the late 1980's, and has been in a general decline ever since (Watanabe et al., 2005). The International Union for Conservation of Nature (IUCN) Red List has classified the Mutton Snapper as 'near threatened', with populations still on a decreasing trajectory (Lindeman et al., 2016). In 1992, a minimum size limit of 12" (30.5 cm) was established, which was changed to 16" (40.7 cm) in 1995. Burton (2002) examined catch curves and length frequency data before and after enactment of both regulations and found no differences in modal ages. This is a slow growing and long-lived species, with a maximum age approaching or exceeding 29 years (Burton, 2002; Watanabe et al., 2005). If sufficient numbers of Mutton Snapper are unable to survive to maturity, the risk of overexploitation becomes greater (Mason and Manooch, 1985). Decreasing landing trends, increased mortality, and regulations that apparently allow for continued harvest of spawning age adults are all indications that populations of this species are depleted. In January 2017 the FWC decided to take a more precautionary approach to management of Mutton Snapper and changed the minimum size of harvest from 16" FL (40.7 cm) to 18" (45.7 cm). These changes to the regulations were intended to provide more spawning opportunities for Mutton Snapper before they are harvested, and thereby increase overall population levels state-wide.

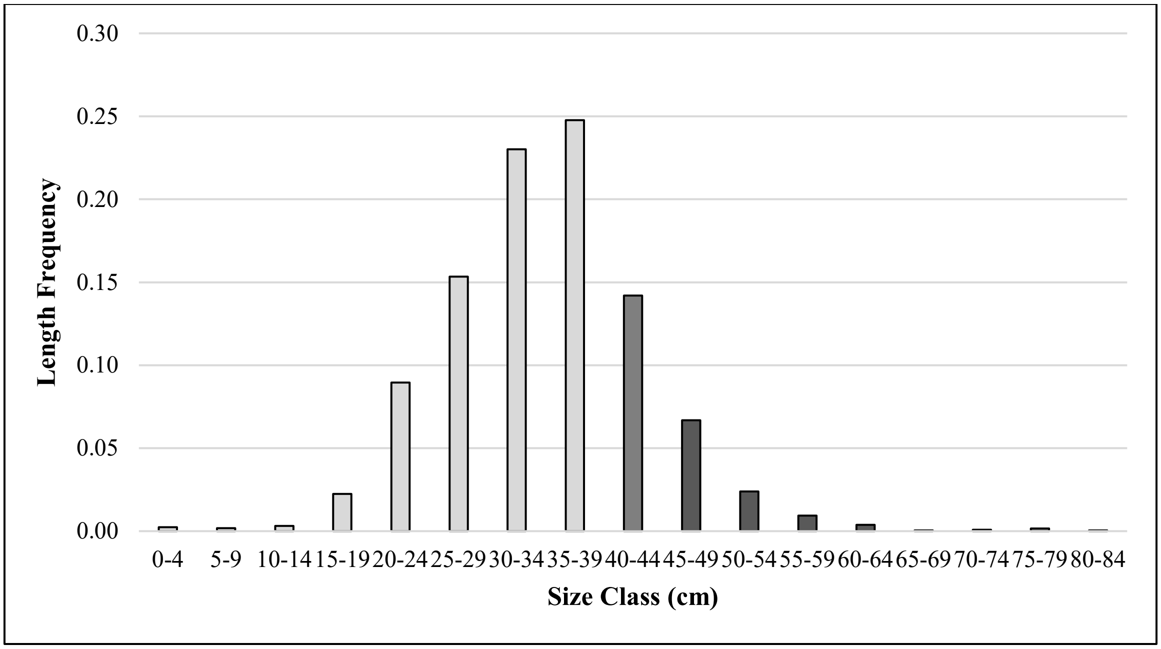


Figure 20. Domain-wide length frequency of Mutton Snapper by size class. Darker gray indicates exploited size classes; legal minimum size of harvest for this species was 16" TL (40.6 cm in state and federal waters) during the study period.

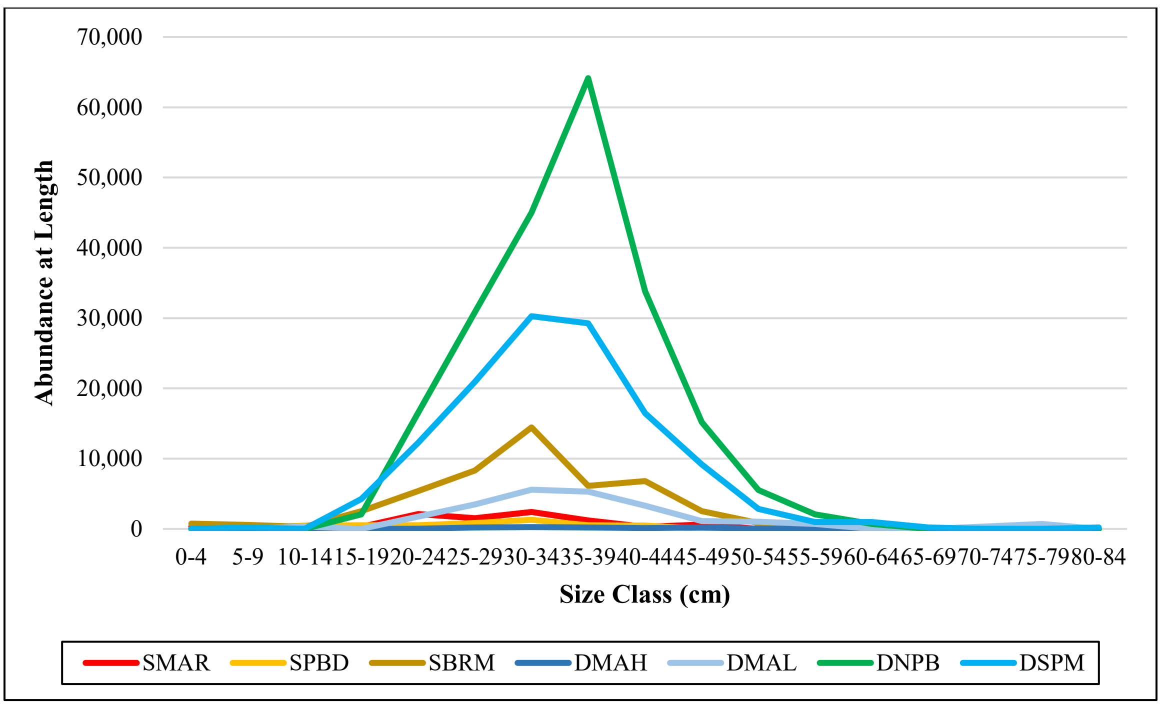


Figure 21. Length frequency comparison for Mutton Snapper by total estimated assemblage region-wide population size (abundance at length) for southeast Florida, with all years combined.

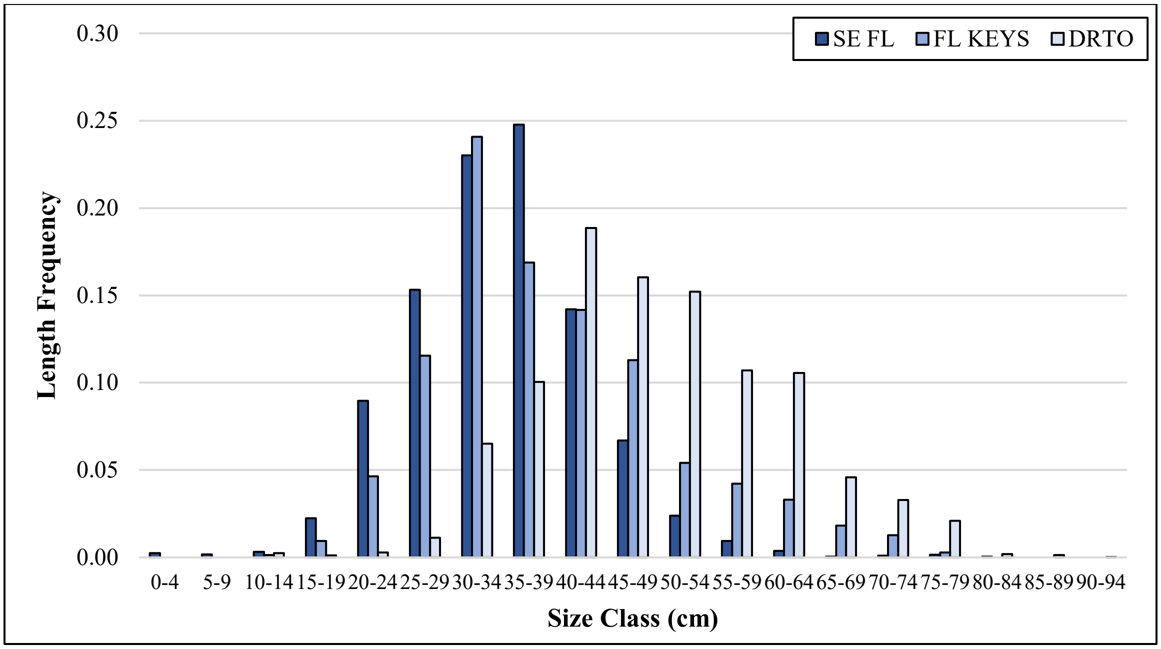


Figure 22. Length frequency comparison of Mutton Snapper, by proportion (all bars sum to 1 for each region regardless of sample size or number of fish counted), for the southeast Florida region, Florida Keys, and Dry Tortugas during the 2012-2016 time period.

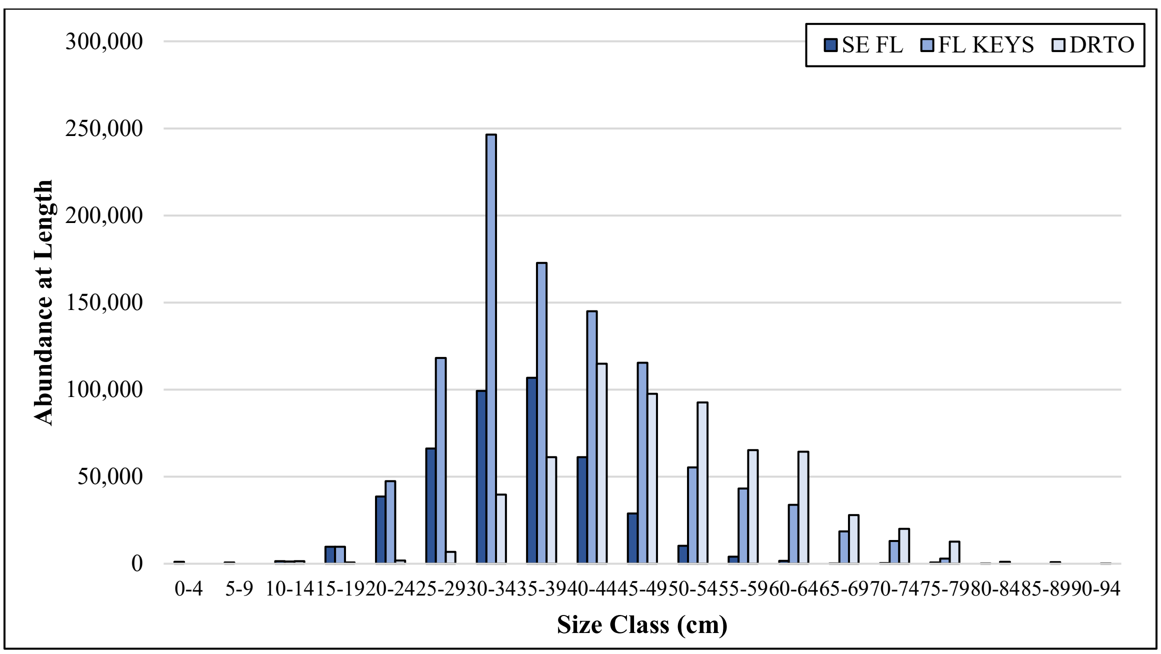


Figure 23. Length frequency comparison for Mutton Snapper, by total estimated domain-wide population size (abundance at length), for the southeast Florida region, the Florida Keys, and the Dry Tortugas for the 2012-2016 time period.

4.1.6. *Exploited Species: Gray Snapper*

Figure 24. Gray Snapper (*Lutjanus griseus*).

A moderately sized snapper relative to many other members of its genus, the Gray Snapper has a wide distribution in the western Atlantic from Florida to Brazil, including Bermuda, the Caribbean and the northern Gulf of Mexico (Robins et al., 1986; Burton, 2011). While juveniles have been reported as far north as Massachusetts and larvae have been found in the inlets of North Carolina, adults are rarely caught in North Carolina (Sumner et al., 1911; Hettler and Barker, 1993; Burton, 2001). It is believed that the larvae and juveniles found in the more northern locales are potentially Gulf Stream exports which would not survive the colder water temperatures in the winter (Burton, 2001). Gray Snapper exhibit diverse habitat usage during their life cycle. Spawning occurs offshore during the new moon phase of July and August (Domeier et al., 1998, Burton, 2001). Currents carry the eggs and larvae onshore where they mature into juveniles and smaller adults (Burton et al., 2001). At about three or four years of age, the small adults move offshore and begin to exhibit a distinct diel migration pattern (Rutherford et al., 1983; Luo et al., 2009). Nocturnally, Gray Snapper frequent shallow seagrass beds to feed while during the day they reside around coral reefs, shipwrecks, rocky outcroppings and ledges, and other artificial or natural live bottom areas that form irregular, complex habitats (Miller and Richards, 1979; Burton, 2001; Luo et al., 2009). Gray Snapper also exhibit seasonal migration patterns whereby during the spawning season mature adults move offshore to the spawning grounds. When they utilize habitats close to shore, they are subject to heavy recreational fishing pressure. When they move offshore, they are still heavily fished from head boats and by commercial fishers. Because of the diverse life history of the Gray Snapper, it is important for resource managers to implement management plans that include both inshore and offshore habitats.

Gray Snapper was the 61st most frequently observed and 40th most abundant species in this study, with an average percent occurrence ($\bar{P}$) of 9.3 and average density ($\bar{D}$) of 0.35 fishes/SSU (Appendix 2). On average, southeast Florida had a lower abundance of Gray Snapper than the Florida Keys ($\bar{P}$=23.9, $\bar{D}$=1.65) and Dry Tortugas ($\bar{P}$=19.0, $\bar{D}$=1.80). Mean Gray Snapper densities by reef fish assemblage region were consistent between the shallow and deep assemblage regions, with the exception of peaks in the Shallow Palm Beach Deerfield (SPBD) and Deep Martin High (DMAH) regions, and with no apparent increase in density moving either north or south through the survey domain (Figure 25). Comparison of the exploitation phases revealed a similar pattern between assemblage regions, with comparable numbers of both pre-exploited and exploited-phases within the shallow and deep regions (Figure 26). The SPBD and DMAH peaks (and associated high error bars) are the result of several large schools that were encountered within those assemblage regions. The length frequency curve for this species in the southeast Florida region indicated that the majority of the population was in the pre-exploited phase, with the exploited phase comprising 27.9% of the population (Figure 27). Length frequency curves for each reef fish assemblage region indicated the majority of the southeast Florida Gray Snapper population resided in the Deep North Palm Beach (DNPB) region, followed by the Deep South Palm Beach Miami (DSPM) and Shallow Broward Miami (SBRM) regions (Figure 28). Examination of length frequency curves along the entire FRT indicated similarity between all three regions, but with a greater proportion of the larger size classes present in the Florida Keys and Dry Tortugas (Figure 29). In terms of overall population size, the Florida Keys was greater than both the southeast Florida region and Dry Tortugas (Figure 30), but the Florida Keys seemed to harbor a greater contingent of subadults.

Domeier et al. (1996) reported the minimum size of reproductive maturity as 18.2 cm SL for males and 19.8 cm SL for females, with both sexes maturing at the same size; 100% maturity was attained by 24.0 cm SL. The minimum legal size of harvest for this species in the state of Florida is 10” TL (25.0 cm) (FWC, 2017a). Throughout Florida, most Gray Snapper attain this minimum legal size by three years of age (Burton, 2001). In this study, the average size for this species was 22.3 cm, and the average size of the exploited-phase (Lbar) was 29.2 cm.

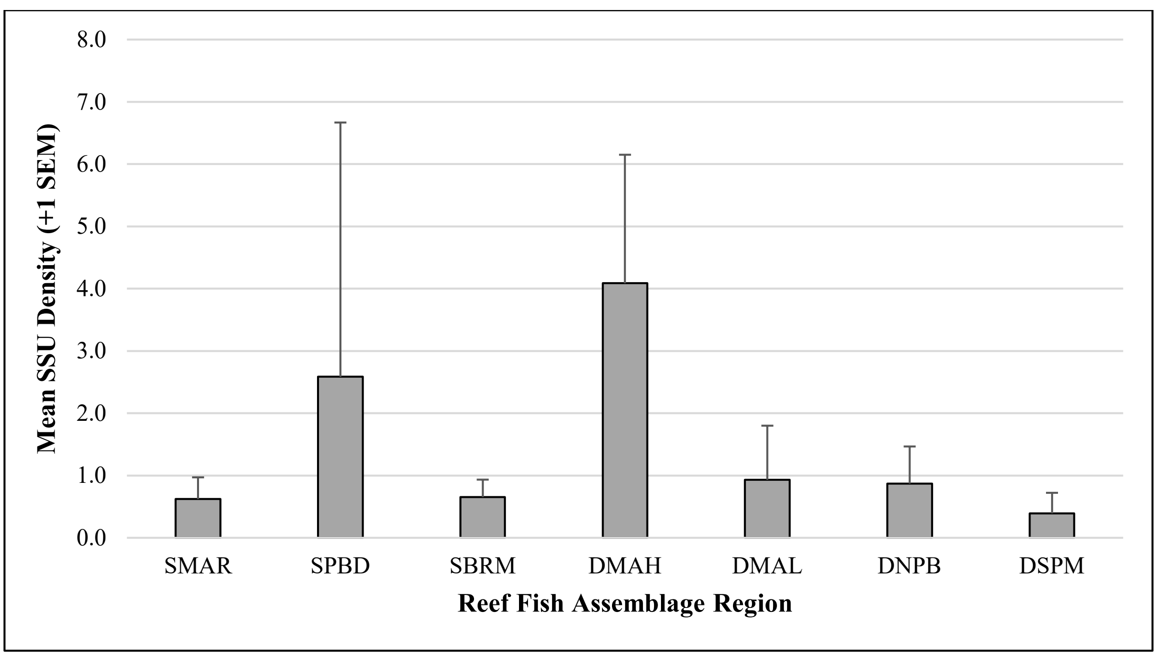


Figure 25. Gray Snapper mean SSU density by Reef Fish Assemblage Region, with all years combined.

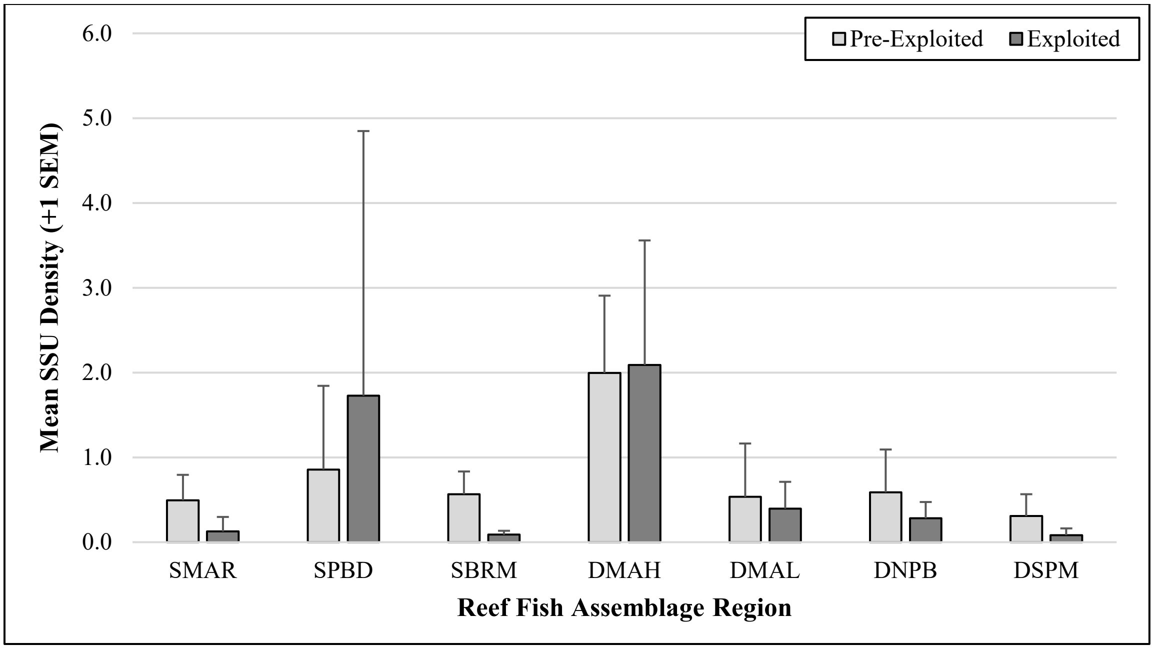


Figure 26. Mean SSU density of Gray Snapper by Reef Fish Assemblage Region; pre-exploited and exploited life-stage comparison with all years combined.

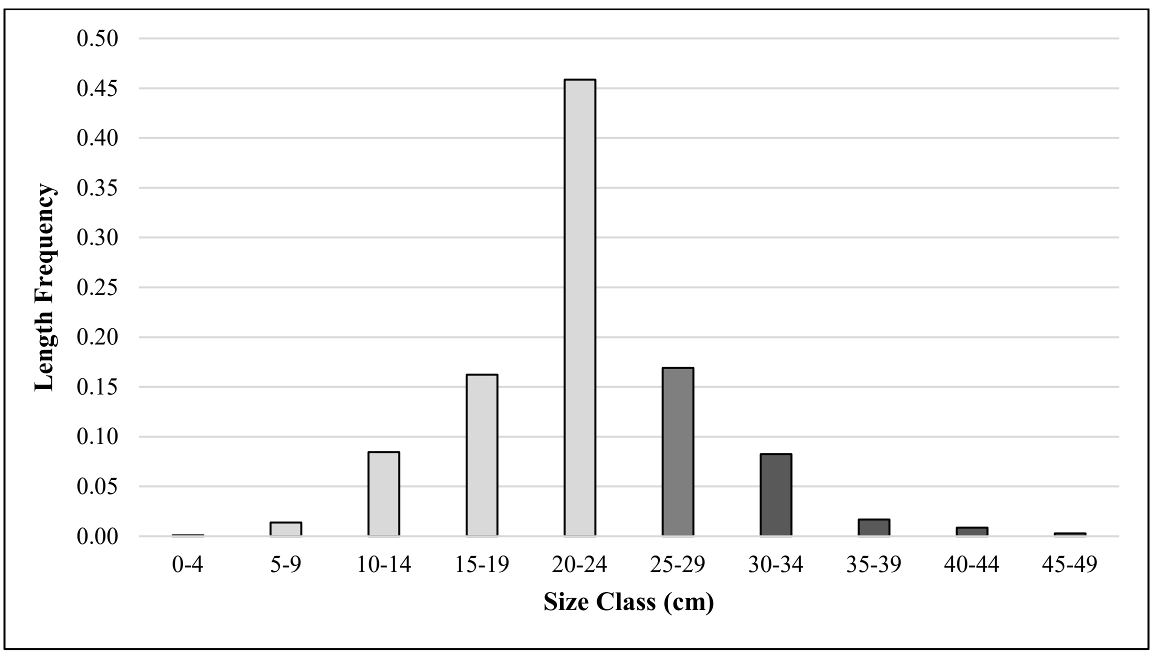


Figure 27. Length frequency of Gray Snapper by size class. Darker gray indicates exploited size classes; minimum legal size of harvest for this species is 10-12" TL (25.4 cm in state waters; 30.5 cm in federal waters).

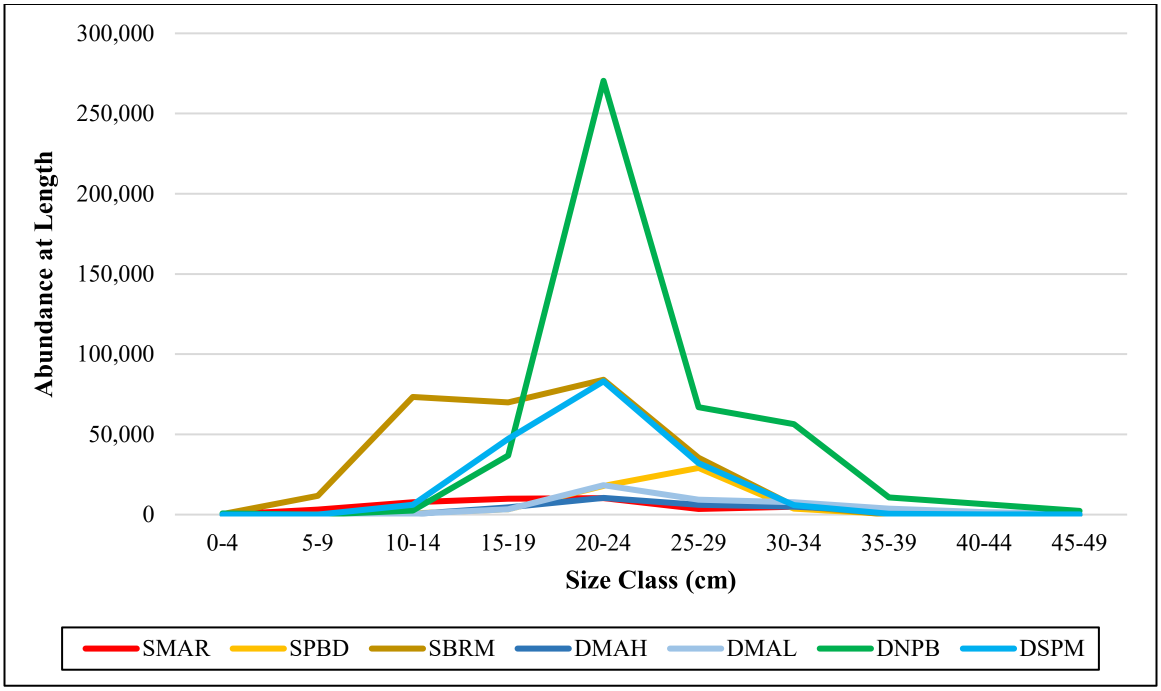


Figure 28. Length frequency comparison for Gray Snapper by total estimated assemblage region-wide population size (abundance at length) for southeast Florida, with all years combined.

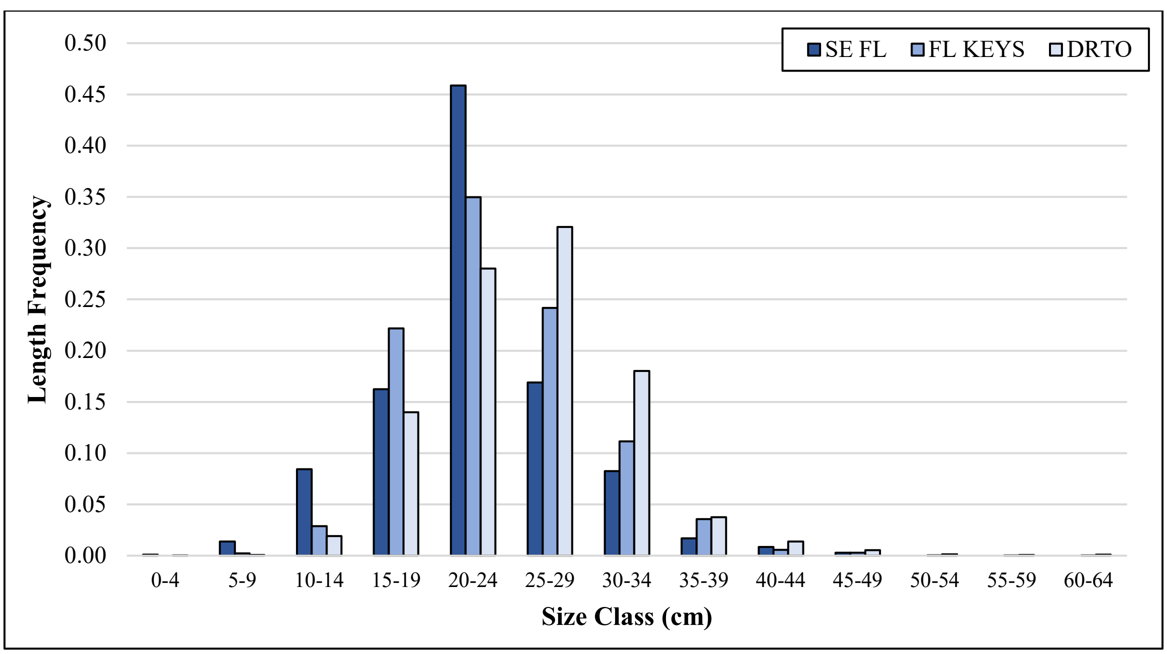


Figure 29. Length frequency comparison Gray Snapper, by proportion (all bars sum to 1 for each region regardless of sample size or number of fish counted), for the southeast Florida region, Florida Keys, and Dry Tortugas during the 2012-2016 time period.

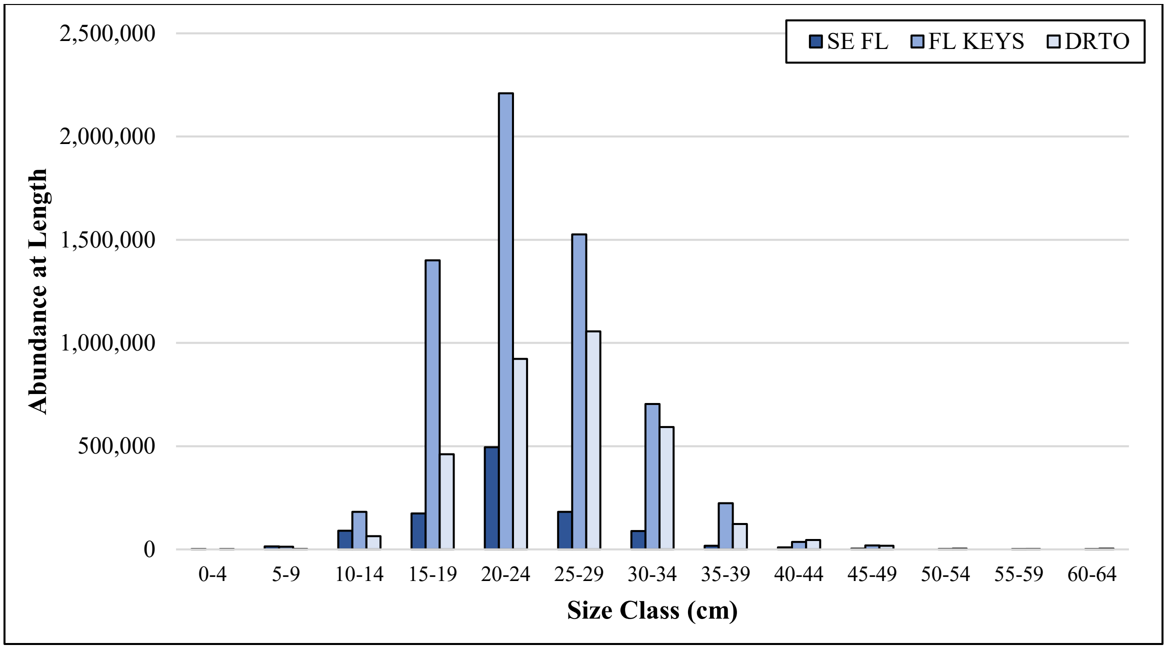


Figure 30. Length frequency comparison for Gray Snapper, by total estimated domain-wide population size (abundance at length), for the southeast Florida region, the Florida Keys, and the Dry Tortugas for the 2012-2016 time period.

4.1.7. Exploited Species: Yellowtail Snapper

Figure 31. Yellowtail Snapper (*Ocyurus chrysurus*).

A relatively small planktophagous species, compared to many other snappers, Yellowtail Snapper tend to feed on more pelagic organisms than the previously discussed Mutton or Gray Snappers (Bortone and Williams, 1986; Domeier and Clarke, 1992). Yellowtail Snapper are found at depths of 10-70 m from Massachusetts to Brazil but are most abundant in the coral reef habitats of south Florida, the Bahamas, and the Caribbean (Thompson and Munro, 1983; Manooch and Drennon, 1987; O'Hop et al., 2012). While Yellowtail Snapper populations in south Florida appear to be propagated by local production rather than upcurrent transport, some low-level of recruitment may be occurring from the western Caribbean (Yucatan peninsula and the Campeche banks) to the Florida Keys through the Loop Current (O'Hop et al., 2012).

Yellowtail Snapper was the 27th most frequently observed and 30th most abundant species, in this study with an average percent occurrence ($\bar{P}$) of 27.9% and average density ($\bar{D}$) of 1.04 fishes/SSU (Appendix 2). On average, southeast Florida had a lower abundance of Yellowtail Snapper than the Florida Keys ($\bar{P}$=64.6, $\bar{D}$=4.87) and Dry Tortugas ($\bar{P}$=80.5, $\bar{D}$=8.80). Mean Yellowtail Snapper densities by reef fish assemblage region were consistent across the shallow and deep assemblage regions, with an apparent increase in density moving north to south through the survey domain (Figure 32). Comparison of the exploitation phases revealed that the pre-exploited phase is responsible for the majority of the surveyed population, with more juveniles and subadults in the shallower assemblage regions and an increasing number moving north to south (Figure 33). The greatest densities of the exploited phase were seen in the Shallow Palm Beach Deerfield (SPBD) assemblage region, although, as with the Gray Snapper, many of the larger peaks were the result of multiple large schools. The length frequency curve for this species in the southeast Florida region indicates that the majority of the population was in the pre-exploited phase (Figure 34). Estimated abundance at length by reef fish assemblage region indicated the majority of the larger Yellowtail Snapper population resided in the Deep North Palm Beach (DNPB) assemblage region, with a population of physically smaller but equally abundant snappers in the Deep South Palm Beach Miami (DSPM) and Shallow Broward Miami (SBRM) regions (Figure 35).

Examination of length frequency curves along the entire FRT indicated a high degree of similarity among regions, but with more juveniles in the Florida Keys (Figure 36). In terms of overall population size, the Florida Keys and Dry Tortugas were both considerably larger than the southeast Florida region (Figure 37); however, all three regions were heavily skewed towards the pre-exploited phase. Nursery habitat provided by mangroves has been shown to strongly influence community structure of fishes on neighboring coral reefs (Mumby et al., 2004). The greater availability of mangrove habitat in the Florida Keys and Dry Tortugas compared to the southeast Florida region may in part be responsible for the smaller population noted in the latter. Spawning can occur in most months of the year (in the Florida Keys ripe fish have been observed year-round) but is most likely to peak between April and August (Collins and Finucane, 1989; Figuerola et al., 1998; McClellan and Cummings, 1998). Settlement of Yellowtail Snapper into seagrass habitats occurs when the larvae reach a standard length of approximately 2.0 cm (Bortone and Williams, 1986; Bartels and Ferguson, 2006). Studies show that juveniles smaller than 15.0 cm FL are found primarily in seagrasses, then as they grow larger, an ontongenetic shift occurs and they move to shallow coral reef areas (Nagelkerken et al., 2000).

Yellowtail Snapper have been heavily fished for many years in south Florida, yet Garcia et al. (2003) found that neither the growth pattern nor age structure of the population appears to have changed in the last 20 years. However, Acosta and Beaver (1998) noted that the overall mean size of Yellowtail Snapper in the Florida Keys decreased from 43.0 cm TL in 1985 to 36.0 cm TL in 1996 and larger Yellowtail Snapper were demonstrably less common in recent landings than they were in the late 1970s (Garcia et al., 2003). Head boat fisheries reportedly harvest Yellowtail Snapper mainly of ages 2 to 6 years

(99% of catch) which range in size from 23.5 to 44.6 cm FL. Yellowtail Snapper larger than 45.0 cm and older than 8 years old are rarely found in head boat catches. Commercial fisheries report harvesting all ages, although the majority range from 1 to 7 years old and from 22.0 to 50.0 cm FL. The minimum legal size of harvest for this species in the state of Florida is 12" TL (30.0 cm) in both state and federal waters (FWC, 2017a). Most Yellowtail Snappers attain sexual maturity at a size range of around 18.0 to 35.0 cm FL and at about 3 to 5 years of age (Bortone and Williams, 1986). Size at reproductive maturity in Florida has been reported as 23.0 cm TL and about 1.7 years of age. Elsewhere in the Caribbean, Figuerola et al. (1998) reported the length at 50% maturity (L_{50}) of 22.4 cm FL from Puerto Rico, and Trejo-Martinez et al. (2011) estimated an L_{50} of 21.3 cm FL from the Campeche Banks in Mexico. In this study, the average size of Yellowtail Snapper was 19.5 cm, the average size of the exploited-phase (Lbar) was 33.2 cm and, in total, 9.2% of the population qualified as being in the exploited-phase (≥30 cm).

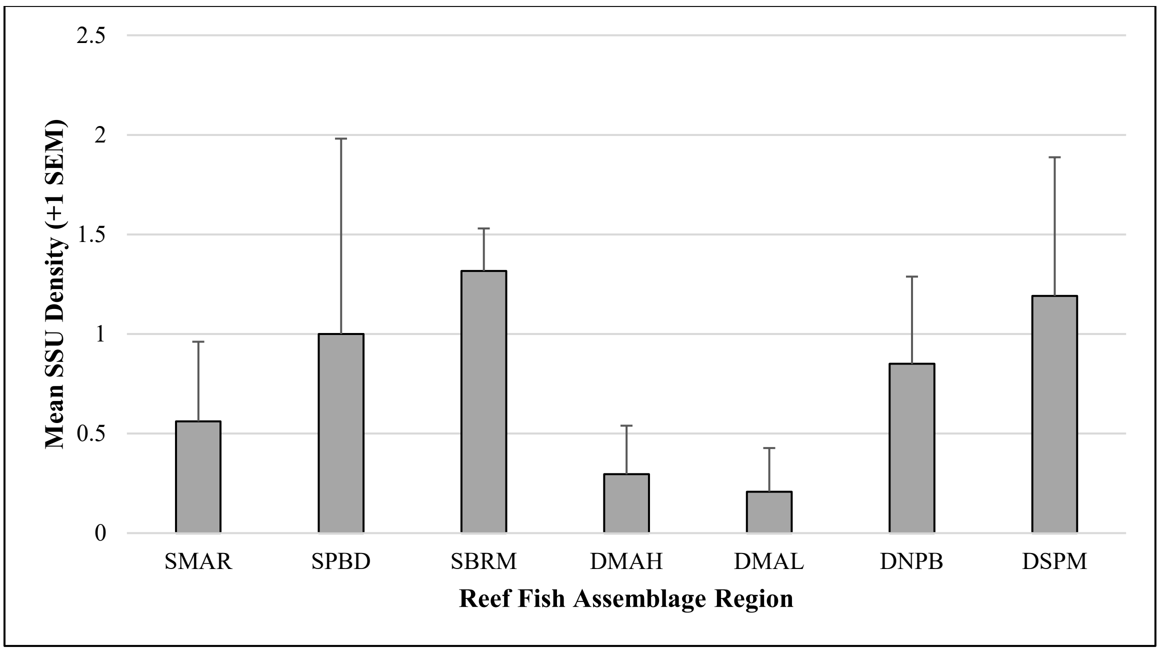


Figure 32. Yellowtail Snapper mean SSU density by Reef Fish Assemblage Region, with all years combined.

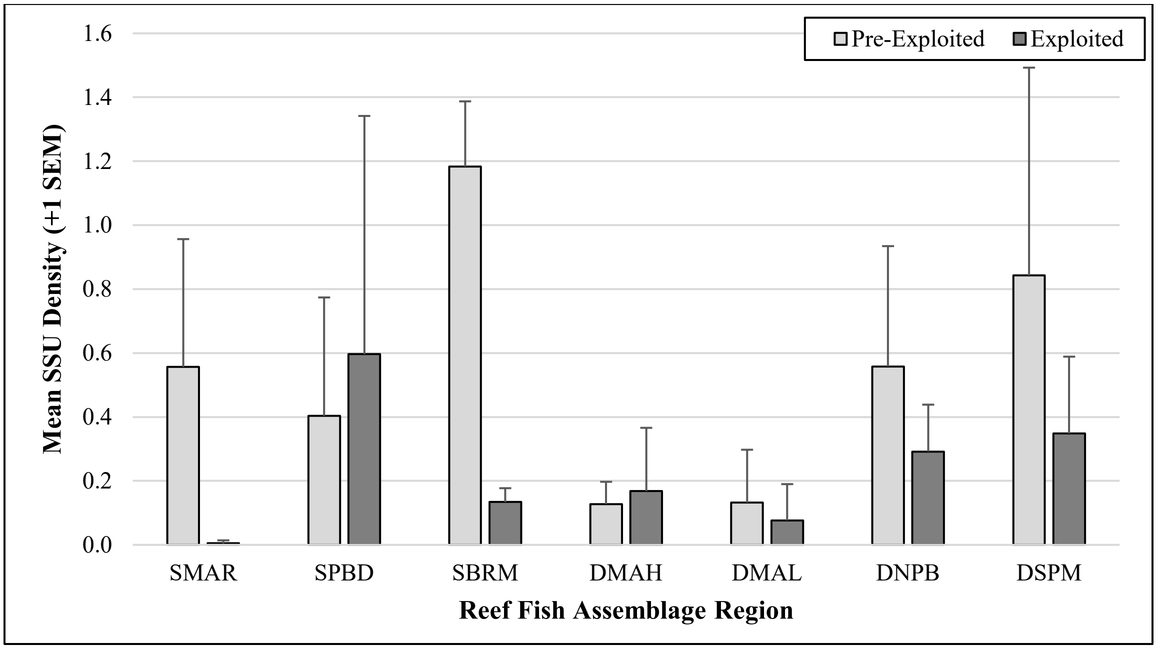


Figure 33. Mean SSU density of Yellowtail Snapper by Reef Fish Assemblage Region; pre-exploited and exploited life-stage comparison with all years combined.

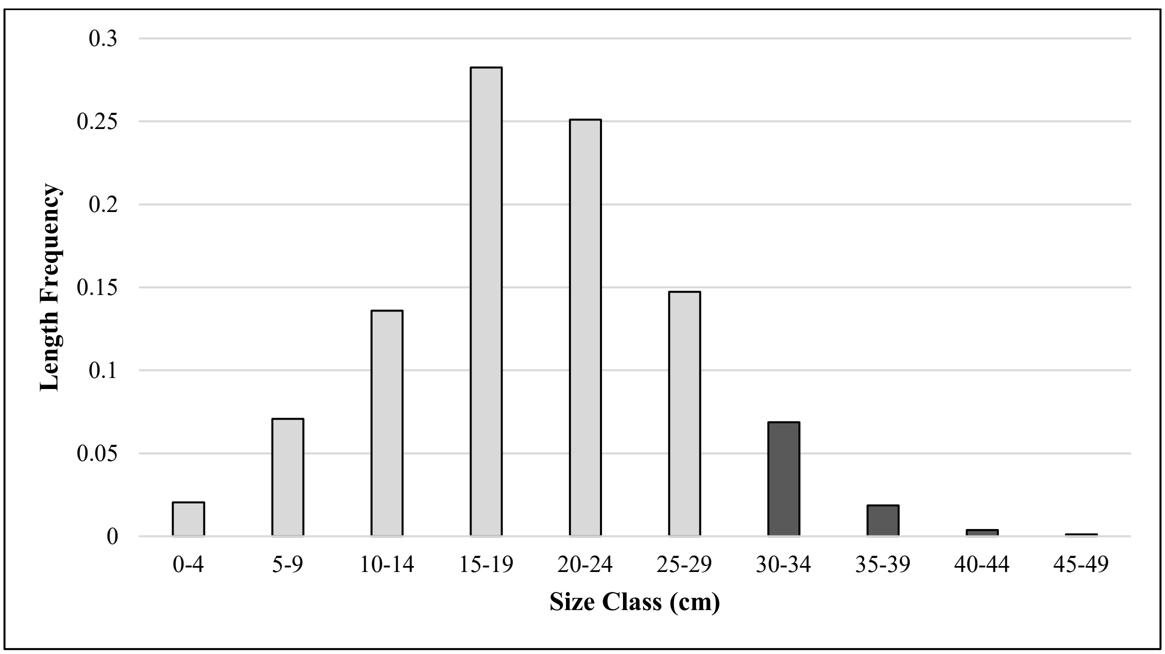


Figure 34. Length frequency of Yellowtail Snapper by size class, with all years and strata combined. Darker gray indicates exploited size classes; legal minimum size of harvest for this species is 12" TL (30.5 cm in state and federal waters).

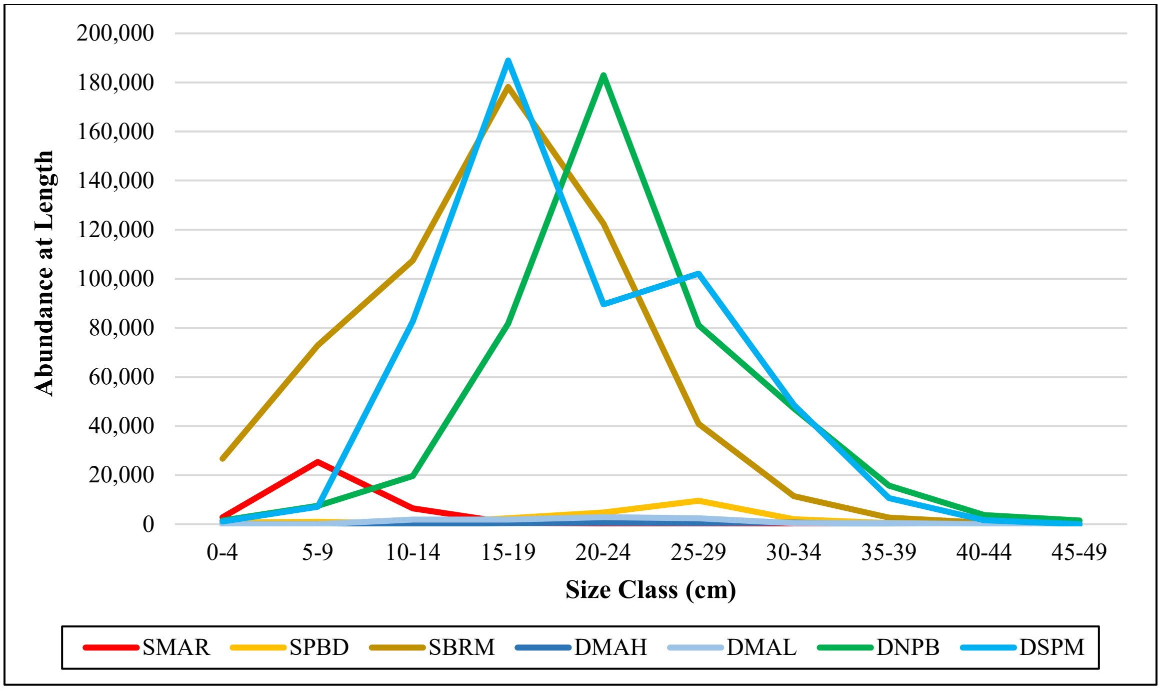


Figure 35. Length frequency comparison for Yellowtail Snapper by total estimated assemblage region-wide population size (abundance at length) for southeast Florida, with all years combined.

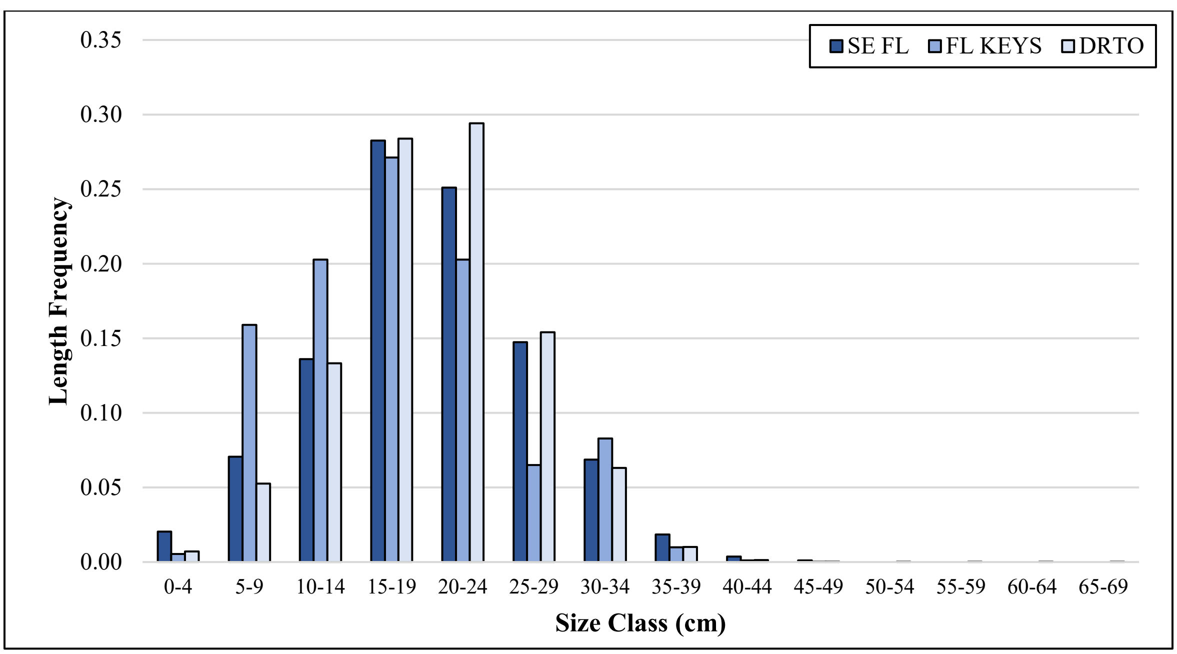


Figure 36. Length frequency comparison Yellowtail Snapper by proportion (all bars sum to 1 for each region regardless of sample size or number of fish counted), for the southeast Florida region, Florida Keys, and Dry Tortugas during the 2012-2016 time period.

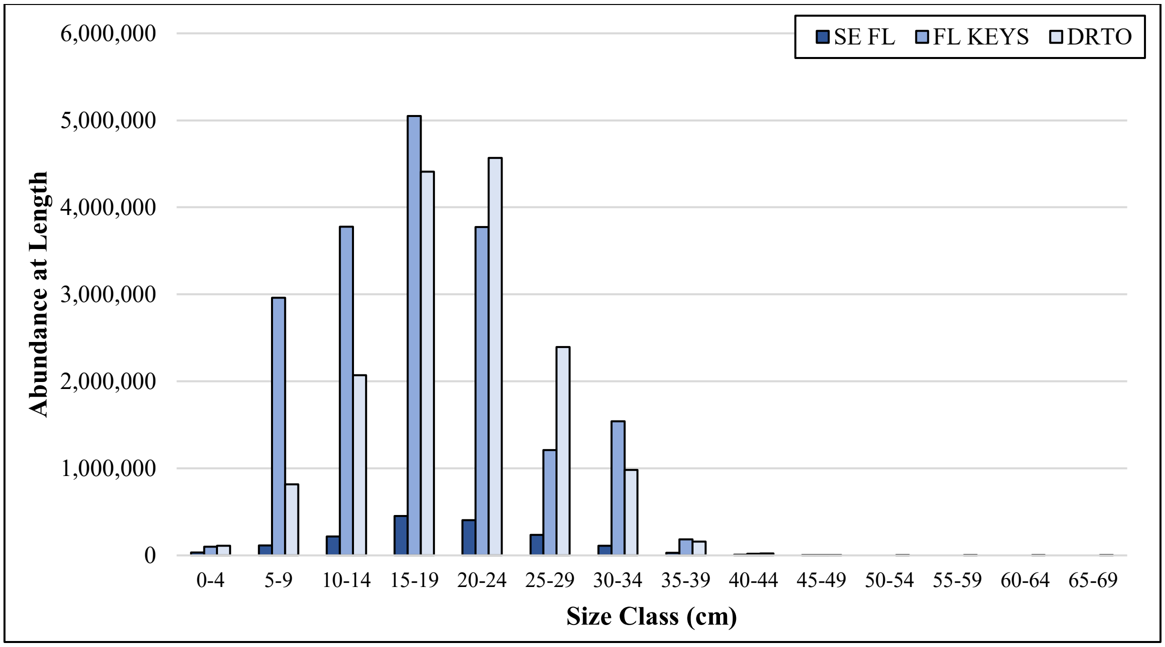


Figure 37. Length frequency comparison for Yellowtail Snapper, by total estimated domain-wide population size (abundance at length), for the southeast Florida region, the Florida Keys, and the Dry Tortugas for the 2012-2016 time period.

4.1.8. Exploited Species: White Grunt

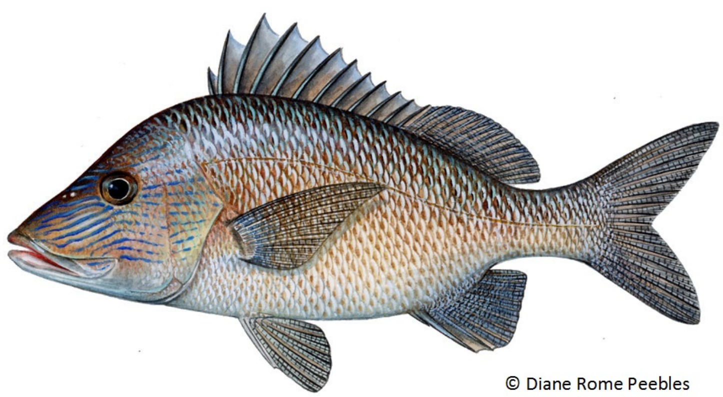


Figure 38. White Grunt (*Haemulon plumierii*).

Grunts (Haemuldiae) are some of the most commonly encountered fishes on the coral reefs of southeast Florida. Although there is no minimum legal size of harvest or bag limit in the state of Florida for White Grunt [*this species is classified as 'unregulated' in both state and federal waters* (FWC, 2017a)], the National Marine Fisheries Service considers this species as important to both commercial and recreational fisheries (NMFS, 1998). White Grunt occurs from Virginia (Chesapeake Bay) to Brazil (Smith, 1997), and along with the closely related Tomtate (*Haemulon aurolineatum*) occupy the northernmost range of all grunts in the Western Atlantic (Gaut and Munro, 1983). While considered abundant off of North Carolina and South Carolina, the White Grunt occurs infrequently off of Georgia and northeastern Florida, becoming more abundant off of Palm Beach County and through the Florida Keys (Potts and Manooch, 2001). The White Grunt has a maximum reported length of 53 cm in Georgia (IGFA, 2017b) to 58.9 cm in the Carolinas (Potts and Manooch, 2001), and may live as long as 27 years (Murie and Parkyn, 2005). Murphy et al. (1999) states that >95% of the total annual harvest of this species in the southeastern United States is landed in Florida, and about 85-90% of those landings are from the west coast of Florida (including Monroe County and the Florida Keys). Comparisons of White Grunt from North and South Carolina to southeast Florida indicate that the Florida grunts only grow to 2/3 the size of the Carolina grunts, and that growth rates in Florida may also be slower (Potts and Manooch, 2001). There are considerable differences in availability of suitable hardbottom habitat and proximity to the coast between the Carolinas and Florida, with more shallow water habitats and easier access to the resource by anglers in Florida (Darcy, 1983).

White Grunt size at reproductive maturity in the southeast United States has been reported as 16.9 to 24.1 cm TL (L_{50} = 16.7 cm TL) for females, and 17.3 to 27.7 cm TL (L_{50} = 18.6 cm TL) for males (Padgett, 1997). However, a report by de Silva and Murphy (1999) pointed out information on the reproductive life history of this species from southeast Florida is lacking and suggested that more studies are required. Other age and growth studies of White Grunt have made assumptions on life history characteristics based on data collected elsewhere (Murphy et al., 1999; Potts, 2000; Potts and Manooch, 2001) which can lead to assessment inaccuracies. A genetics study utilizing mitochondrial DNA variation in the White Grunt has revealed three distinctive lineages: 1) a northern type that ranges from the Carolinas through the Florida Keys and Gulf of Mexico, 2) a southern type that is found in the Florida Keys, Mexico, and Puerto Rico, and 3) a third type found exclusively in Trinidad (Chapman et al., 1999). Interestingly, the study noted that ~95% of the grunts that were examined from the Florida Keys belonged to the second group; the mixing of two genetic forms in south Florida has implications on how the stock should best be defined and managed.

White Grunt was the 14th most frequently observed and 19th most abundant species in this study, with an average percent occurrence ($\bar{P}$) of 43.8% and average density ($\bar{D}$) of 1.89 fishes/SSU (Appendix 2). On average, southeast Florida had fewer White Grunts than the Florida Keys ($\bar{P}$=75.3, $\bar{D}$=10.02) and Dry Tortugas ($\bar{P}$=90.9, $\bar{D}$=6.70). White Grunt densities by reef fish assemblage region were consistent across strata, with the exception of Deep Martin Low (DMAL) (Figure 39). For the purposes of this report, the minimum size of the exploited phase for this species is considered as 8” FL (20 cm). The average size for this species in the southeast Florida region was 18.0 cm, and the average size of the exploited-phase (Lbar) was 24.1 cm. In total, 43.5% of the White Grunt population qualified as exploited-phase (≥20 cm). The exploitation phase comparisons showed higher densities of pre-exploited fish in the shallower assemblage regions and the density of the exploited phase fish increased incrementally in more southern regions (Figure 40). A full 35% of the population were classified as the exploited phase, although the largest size classes were nearly absent (Figure 41). Estimated abundance by length, by reef fish assemblage region, showed a greater portion of the southeast Florida region White Grunts resided in the Shallow Broward-Miami (SBRM) region, followed by the Deep North

Palm Beach (DNPB) and Deep South Palm Beach-Miami (DSPM) regions (Figure 42). Population length frequency estimates were similar along the entire FRT (Figure 43). However, estimated population size in the Florida Keys surpassed both southeast Florida and the Dry Tortugas (Figure 44). The nursery habitat provided by mangroves has been shown to strongly influence community structure of fishes on neighboring coral reefs (Mumby et al., 2004). A recent study in the Florida Keys on mangrove-associated reef fish and habitat mapping data show the populations on the reef are influenced by the amount of mangrove habitat within a 25-km radius (Shideler et al., 2017). Thus, the greater availability of mangrove habitat (which serves as critical nursery habitat for many species, including several grunt species) in the Florida Keys compared to the southeast Florida region may explain, in part, the difference in the population sizes (Shideler et al., 2017).

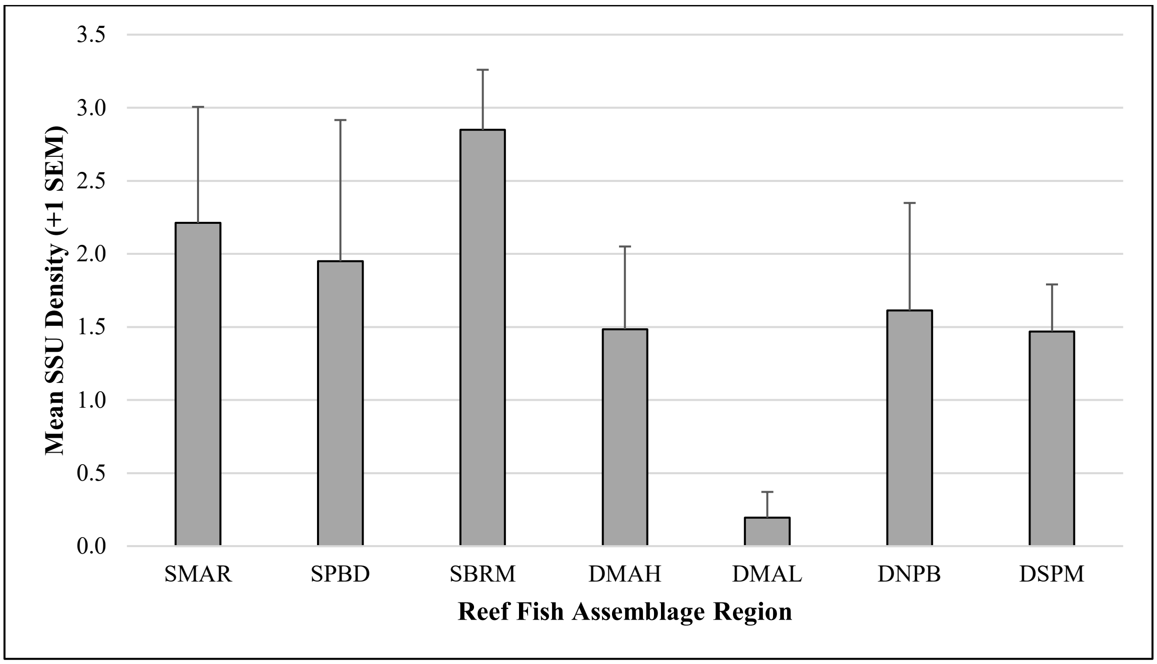


Figure 39. White Grunt mean SSU density by Reef Fish Assemblage Region, with all years combined.

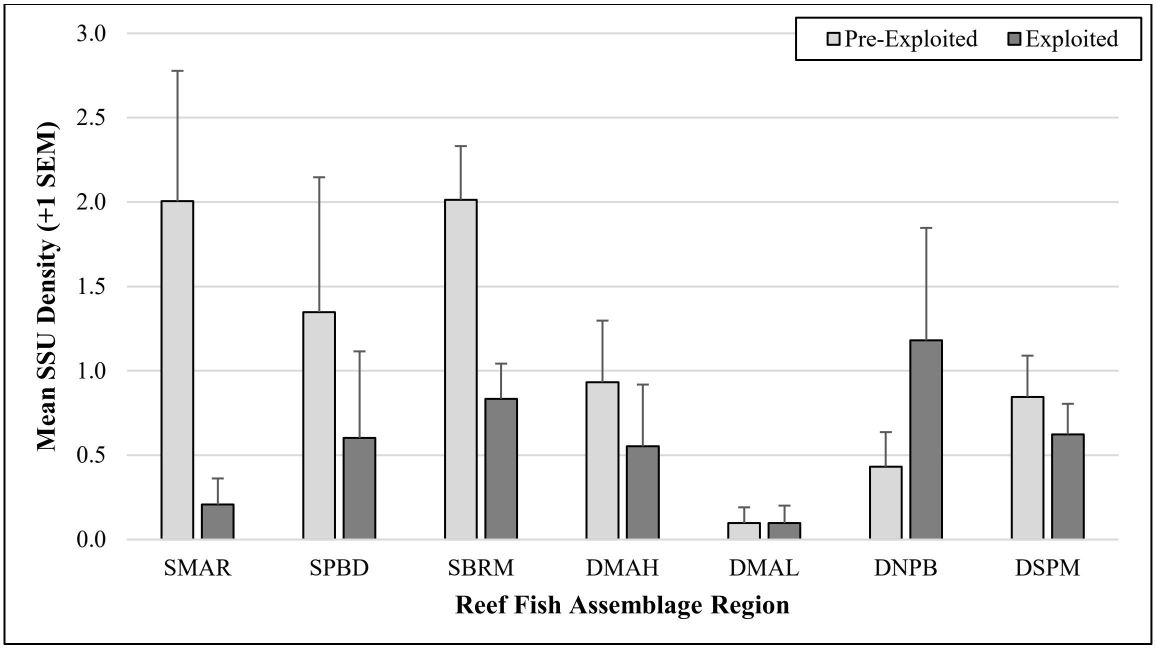


Figure 40. Mean SSU density of White Grunt by Reef Fish Assemblage Region; pre-exploited and exploited life-stage comparison with all years combined.

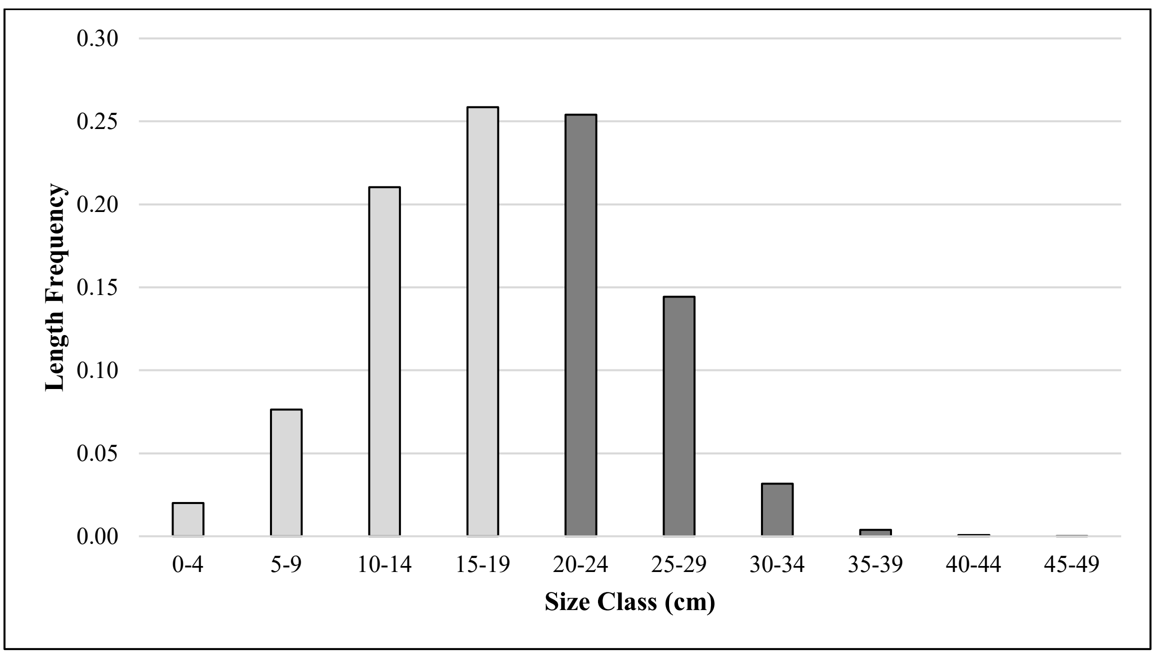


Figure 41. Domain-wide length frequency of White Grunt by size class, with all years and strata combined. Darker gray indicates exploited size classes; estimated minimum size of the exploited phase for this species is 8" TL (20.3 cm, unregulated in state and federal waters).

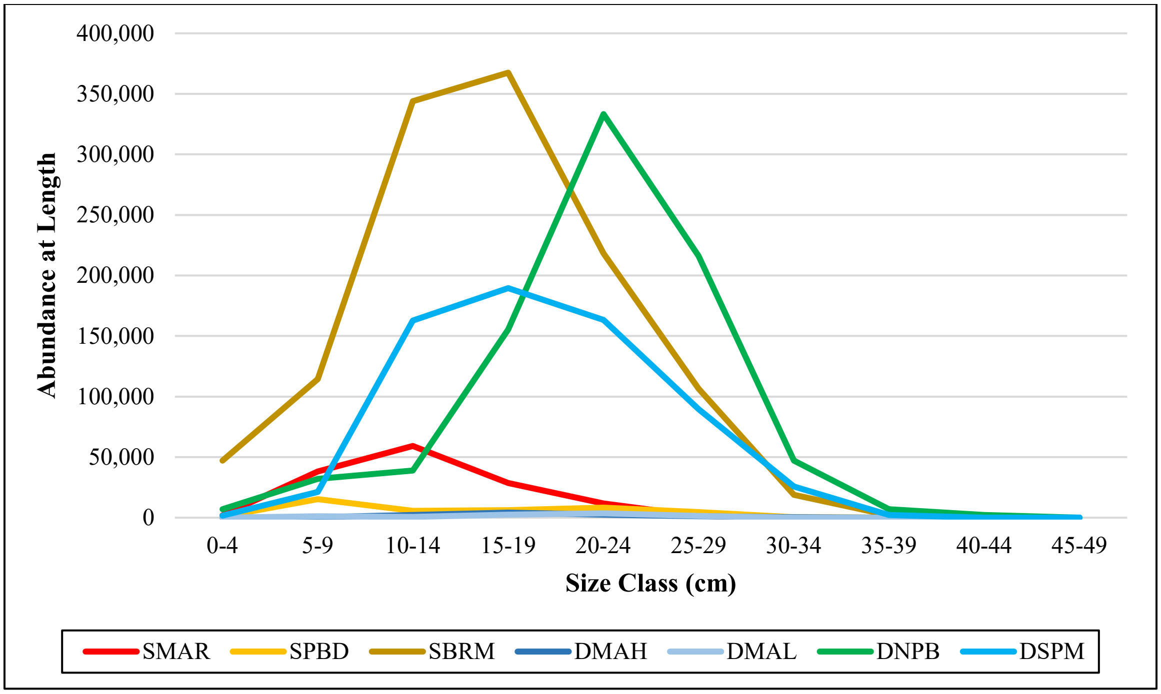


Figure 42. Length frequency comparison for White Grunt by total estimated assemblage region-wide population size (abundance at length) for southeast Florida, with all years combined.

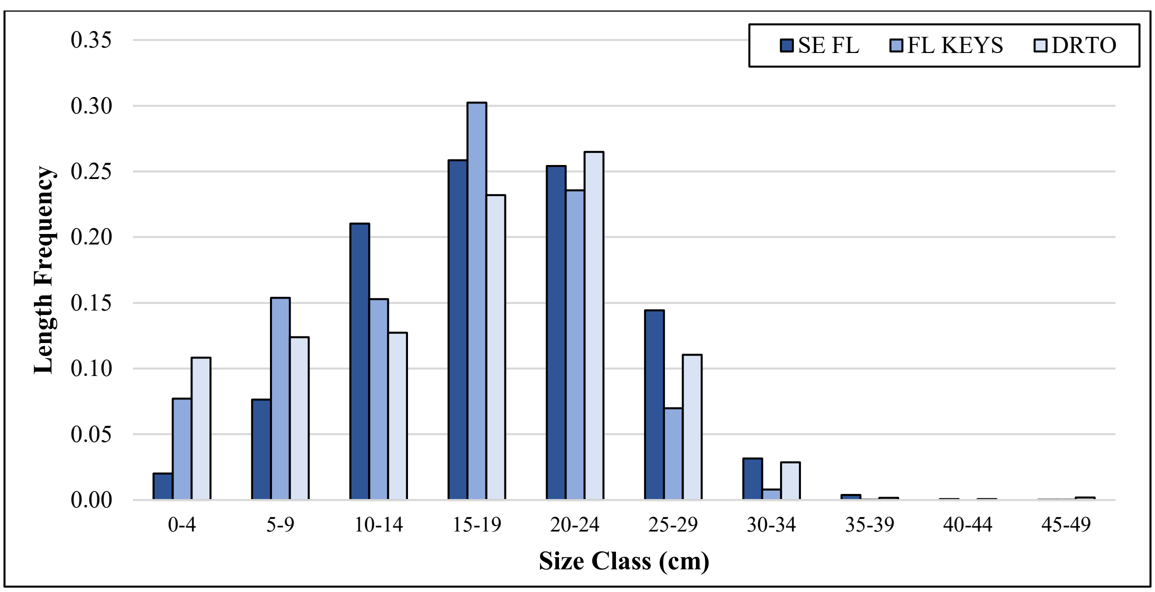


Figure 43. Length frequency comparison for White Grunt, by proportion (all bars sum to 1 for each region regardless of sample size or number of fish counted), for the southeast Florida region, Florida Keys, and Dry Tortugas during the 2012-2016 time period.

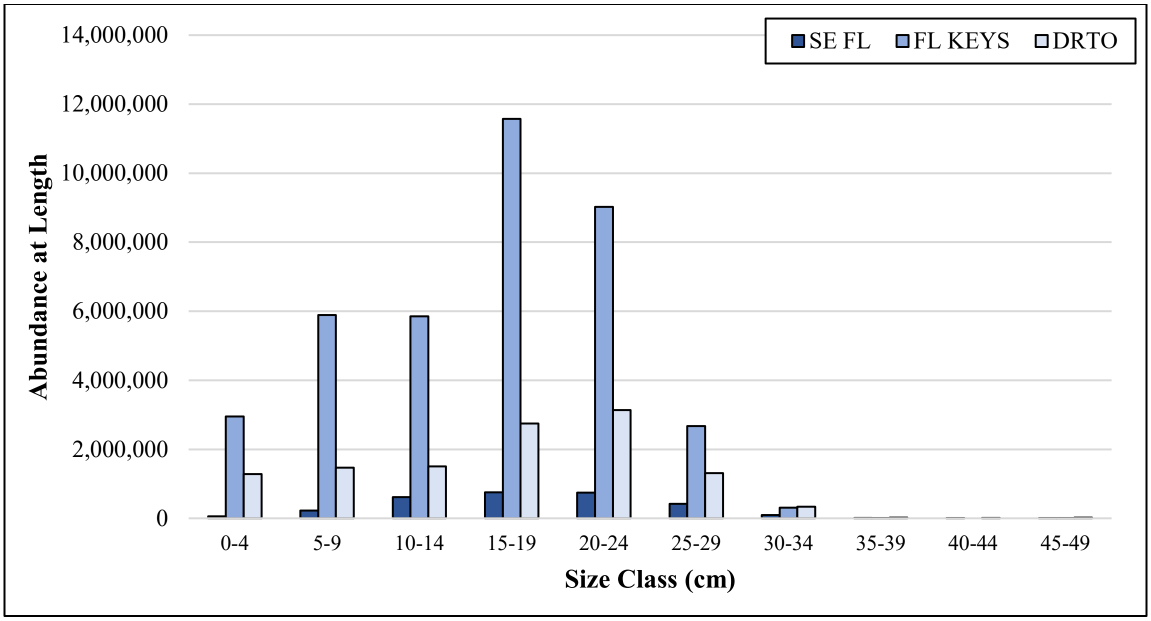


Figure 44. Length frequency comparison for White Grunt, by total estimated domain-wide population size (abundance at length), for the southeast Florida region, the Florida Keys, and the Dry Tortugas for the 2012-2016 time period.

4.1.9. Exploited Species: Bluestriped Grunt

Figure 45. Bluestriped Grunt (*Haemulon sciurus*).

Compared to the other species discussed in this report, not as much information is available in the literature for the Bluestriped Grunt. There is no minimum legal size of harvest in the state of Florida for this species; it is classified as 'unregulated' in both state and federal waters (FWC, 2017a). Bluestriped Grunts were not as commonly encountered as White Grunts, ranking as the 47th most frequently observed and 31st most abundant species in this study, with an average percent occurrence ($\bar{P}$) of 18.0% and average density ($\bar{D}$) of 0.98 fishes/SSU (Appendix 2). There were fewer Bluestriped Grunts in southeast Florida than the Florida Keys ($\bar{P}$=32.8, $\bar{D}$=2.69) but more than the Dry Tortugas ($\bar{P}$=9.6, $\bar{D}$=0.30). Mean Bluestriped Grunt densities were consistent between reef fish assemblage regions, with the exception of Shallow Palm Beach Deerfield (SPBD) where it was higher primarily due to the pre-exploited phase (Figures 46 and 47). Size at reproductive maturity (L_m) has been reported as 18.5 to 20.0 cm from southwest Cuba and Jamaica, respectively (Billings and Munro, 1974; García-Cagide et al., 1994), although reproductive life history information from south Florida has yet to be determined for this species.

For the purposes of this report, the approximate size of the exploited phase is considered as 8" FL (20 cm). The average size for this species in the southeast Florida region was 18.9 cm, and the average size of the exploited-phase (Lbar) was 24.0 cm. The length frequency analysis indicated more than half (57.8%) of the population was classified as the exploited phase (≥20 cm) (Figure 48). Abundance at length estimates, by reef fish assemblage region, showed that the largest portion of the population were in the Deep North Palm Beach (DNPB), Deep South Palm Beach-Miami (DSPM), and Shallow Broward-Miami (SBRM) regions (Figure 49). Bluestriped Grunts were relatively sparse in the far northern assemblage regions (Martin County). Similar to the White Grunt, the length frequency data along the entire FRT indicated that population levels were similar throughout (Figure 50). In addition, again similar to the White Grunt, the Florida Keys appeared to support a larger population of Bluestriped Grunts than southeast Florida or the Dry Tortugas (Figure 51). The Bluestriped Grunt may be even more reliant upon the availability of mangroves for nursery habitat than the White Grunt (Nagelkerken et al., 2001; Morinière et al., 2002; Morinière et al., 2004; Faunce and Serafy, 2007), so a similar pattern of greater abundance in the Florida Keys is not surprising. Faunce and Serafy (2007) noted that settlement and grow-out for Bluestriped Grunts in the Florida Keys occurs in seagrass beds for the first 8-10 months of their life cycle, followed by expansion into mangrove habitats at 10-12 cm. Interestingly, we found that the proportion of the smallest size classes (0-4 and 5-9 cm) were higher in southeast Florida compared to the Florida Keys and Dry Tortugas (Figure 51). This may be indicative of differential resource utilization in southeast Florida due to comparatively lower availability of seagrasses and mangrove habitats.

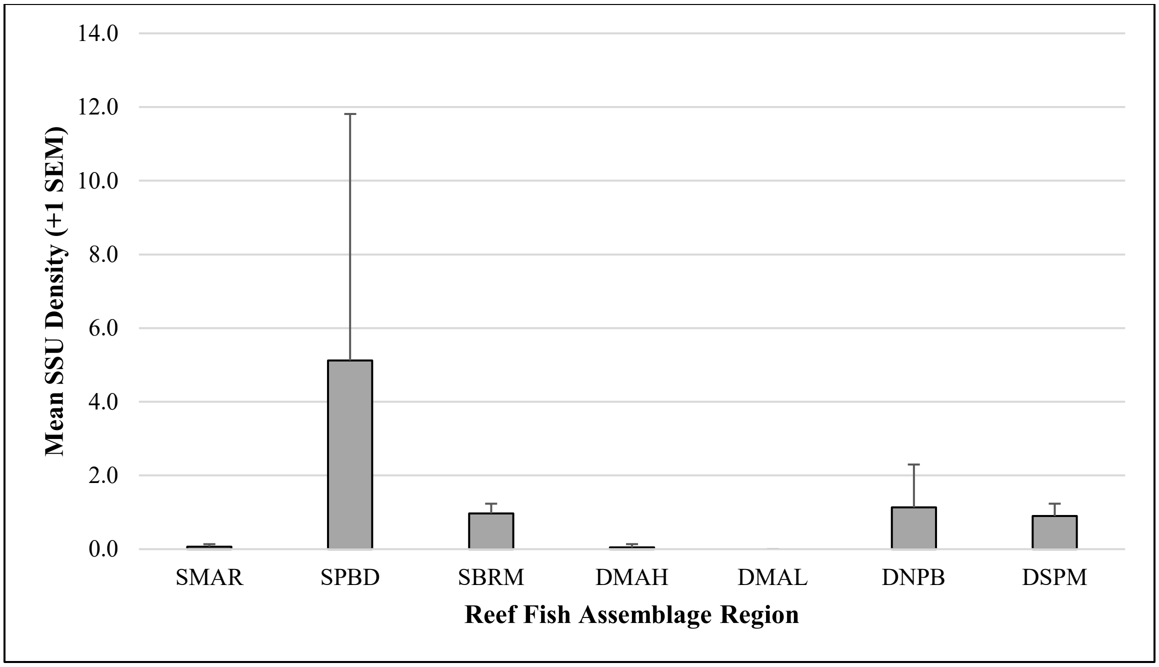


Figure 46. Bluestriped Grunt mean SSU density by Reef Fish Assemblage Region, with all years combined.

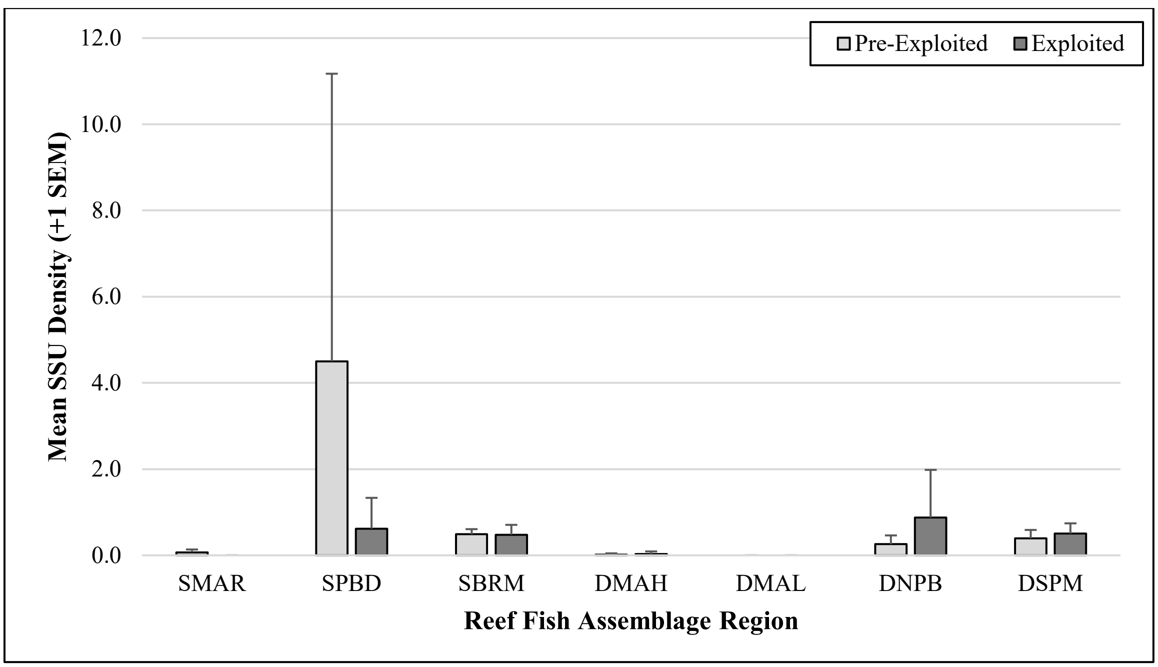


Figure 47. Mean SSU density of Bluestriped Grunt by Reef Fish Assemblage Region; pre-exploited and exploited life-stage comparison with all years combined.

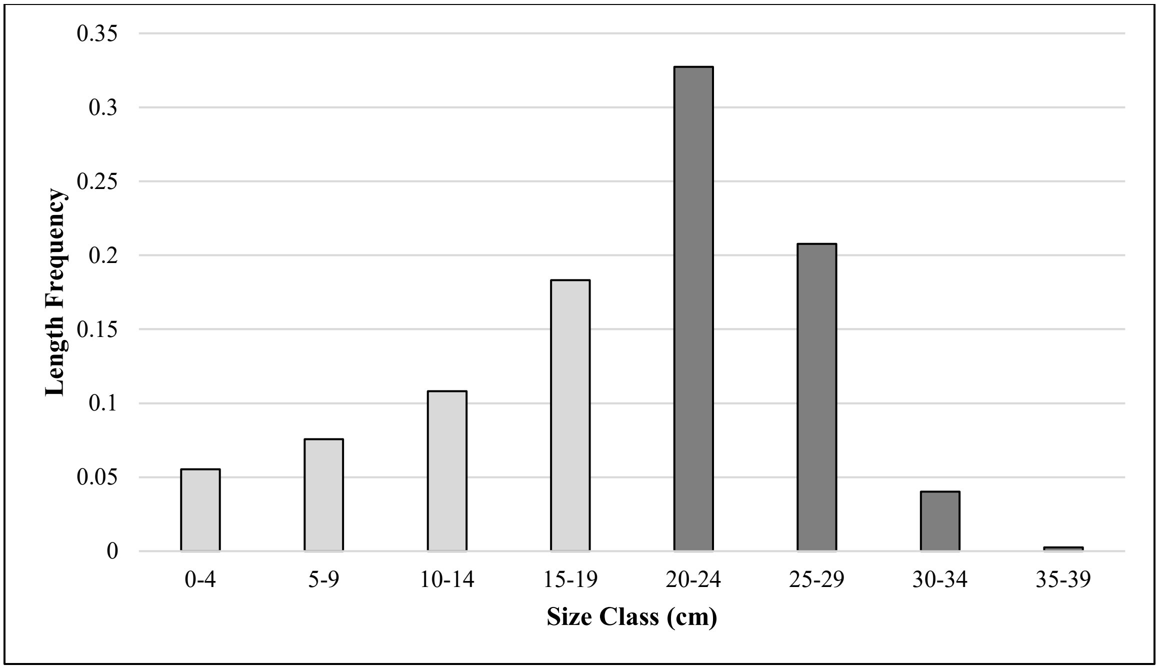


Figure 48. Domain-wide length frequency of Bluestriped Grunt by size class, with all years and strata combined. Darker gray indicates exploited size classes; estimated minimum size of the exploited phase for this species is 8” TL (20.3 cm, unregulated in state and federal waters).

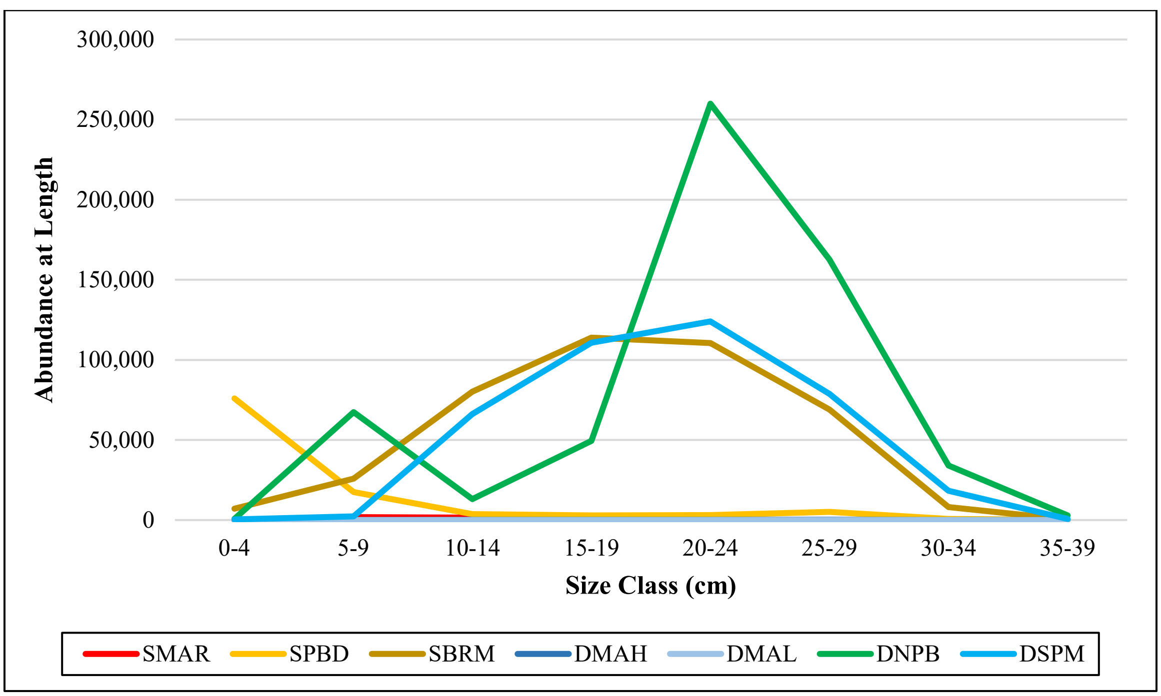


Figure 49. Length frequency comparison for Bluestriped Grunt by total estimated assemblage region-wide population size (abundance at length) for southeast Florida, with all years combined.

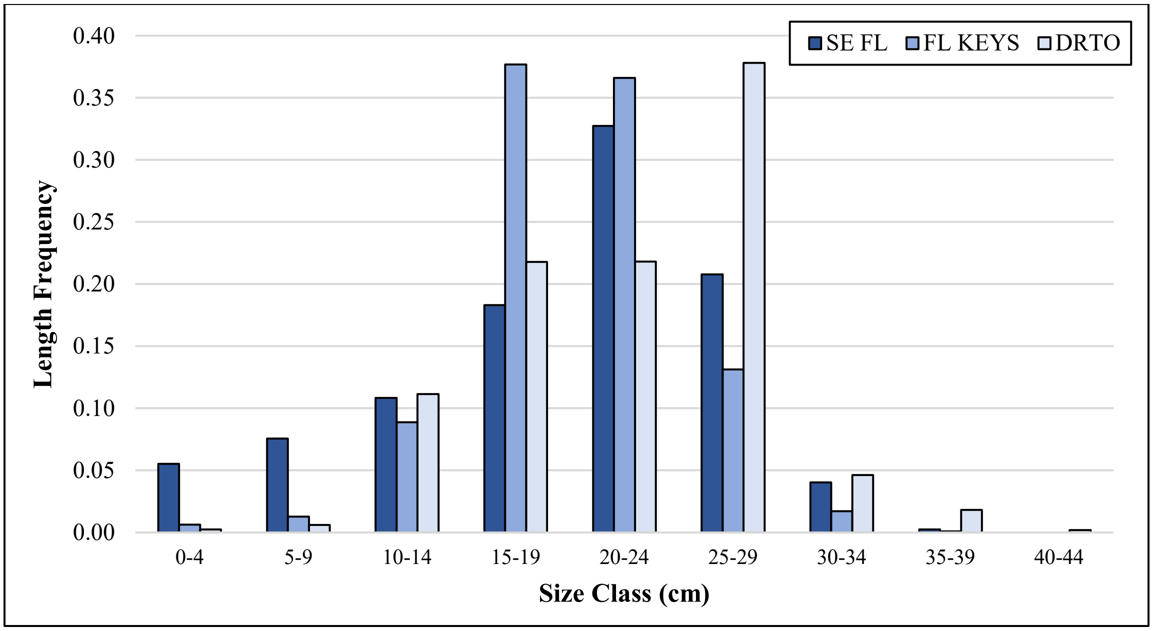


Figure 50. Length frequency comparison for Bluestriped Grunt, by proportion (all bars sum to 1 for each region regardless of sample size or number of fish counted), for the southeast Florida region, Florida Keys, and Dry Tortugas during the 2012-2016 time period.

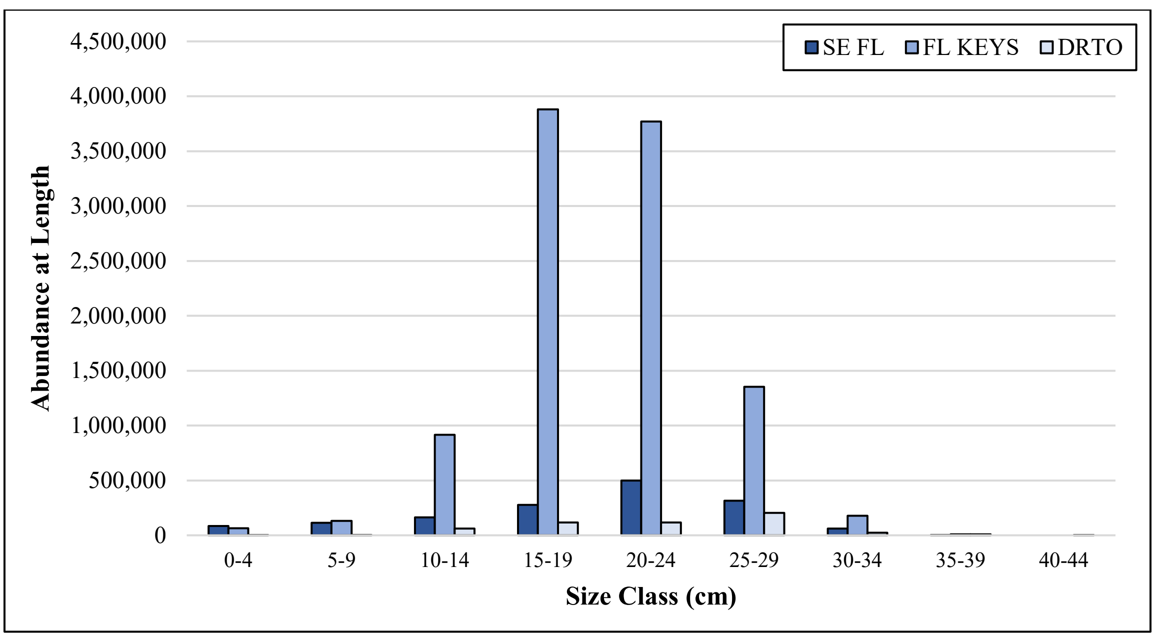


Figure 51. Length frequency comparison for Bluestriped Grunt, by total estimated domain-wide population size (abundance at length), for the southeast Florida region, the Florida Keys, and the Dry Tortugas for the 2012-2016 time period.

4.1.10. Exploited Species: Hogfish

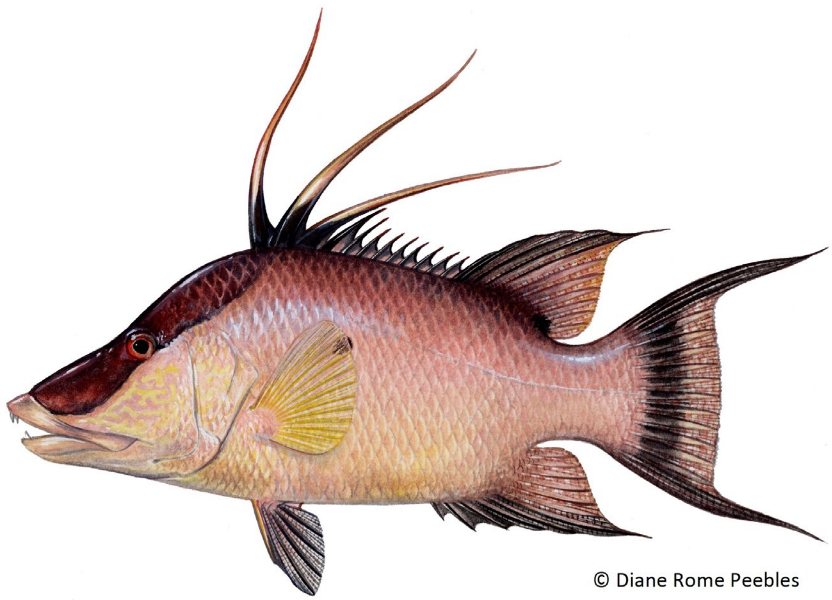


Figure 52. Hogfish (*Lachnolaimus maximus*).

Hogfish are the largest and most commercially valuable member of the wrasse family (Labridae) in the state of Florida (McBride et al., 2001). Hogfish are relatively easy to approach underwater, making them a prime target for spearfishing. While this species occurs from Nova Scotia to Brazil (Froese and Pauly, 2017), about 70% of the reported US commercial Hogfish landings during 2000-2004 were from Florida; fishery-dependent data for this species is unavailable because divers are responsible for the majority of the harvest (Kingsley, 2004; McBride et al., 2008). Hogfish were unregulated until 1994, at which point the legal minimum size of harvest was set at 12" FL (30.5 cm) in both state and federal waters (FWC, 2017c). In November 2016, FWC approved new regulations for Hogfish in state waters, including lowering the daily bag limit from five to one, and increasing the Atlantic recreational and commercial minimum size limit to 16" FL (40.6 cm) (FWC, 2017c). These changes to the regulations went into effect in August 2017 and were intended to provide more spawning opportunities for Hogfish before they are harvested, thereby increasing overall population levels state-wide.

There were some concerns about the previous (12") size limit due to the unique reproductive life history of this species. Hogfish are monandric protogynous hermaphrodites, meaning they begin life as females and then change to males after first passing through a functional female phase after reaching maturity (McBride et al., 2001; McBride and Johnson, 2007). One of the most famous and prominent physical characteristics of Hogfish is the pronounced change in snout morphology associated with transformation into the male terminal phase; a process that has been shown to take 1-2 years to complete (Figure 53) (McBride et al., 2001; McBride et al., 2007). This species is also haremic, with one male typically spawning with multiple females within its territory (groups of 10 to 15 have been reported) (Colin, 1982; McBride and Johnson, 2007). The harvest of a terminal phase male can potentially reduce the spawning activity and reproductive output of the harem it was associated with on a temporary basis. Sex change for this species is a linear but complex process, linked to both social structure and spawning seasonality, and taking several months to complete. The minimum size at which transformation from female to male occurs is 30 cm, but all females will eventually change to males if they survive long enough (may live >20 years) (McBride and Johnson, 2007). The size of 50% maturity (L_{50}) for females in south Florida is 16.9 cm and 42.5 cm for males (McBride et al., 2008). In the Florida Keys, more Hogfish reproductive behavior has been observed at unfished sites compared to fished sites (Munoz et al., 2010). This may suggest a disruption in social structure at the fished sites and emphasizes the potential ability for marine protected areas to maintain the reproductive output of this and other commercially and recreationally important species that have complex social and mating systems and high site fidelity.

Figure 53. Hogfish stages of maturity and sexual dimorphism.

Hogfish was the 35th most frequently observed and 56th most abundant species in this study, with an average percent occurrence ($\bar{P}$) of 23.5% and average density ($\bar{D}$) of 0.30 fishes/SSU (Appendix 2). Although Hogfish has been reported to reach sizes exceeding 90 cm FL (McBride and Richardson, 2007), the average size recorded in this study was 24.6 cm. On average, southeast Florida had fewer Hogfish than the Florida Keys ($\bar{P}$=70.3, $\bar{D}$=1.69) and Dry Tortugas ($\bar{P}$=45.7, $\bar{D}$=0.50). Mean Hogfish densities by reef fish assemblage region were consistent across the northern assemblage regions and increased moving south into the Shallow Palm Beach-Miami (SBRM) and Deep South Palm Beach Miami (DSPM) regions (Figure 54).

The average size of the exploited-phase (Lbar) was 34.6 cm. In total, 25.1% of the Hogfish population qualified as exploited-phase (≥30 cm), and only 8.8% of those were larger than the L_{50} reported for males. Comparisons of the exploitation phases revealed a similar pattern between assemblage regions, with comparable numbers of each in the northern portion, and the greatest portion of both pre-exploited and exploited phase individuals in the SBRM and DSPM assemblage regions (Figure 55). The majority of the population was in the pre-exploited phase, but the exploited phase contained over 25% of the population (Figure 56). The estimated abundance at by length by assemblage region showed that the Deep South Palm Beach Miami (DSPM) region had the greatest portion of the population, followed by the Shallow Broward Miami (SBRM) region (Figure 57). The length frequency data were similar between southeast Florida and the Florida Keys. The Dry Tortugas Hogfish population size structure was noticeably different, containing a much higher proportion of the larger size classes, but, interestingly, fewer of the smaller size classes (Figure 58). The estimated population density in the Florida Keys, however, was much greater than both southeast Florida and the Dry Tortugas (Figure 59).

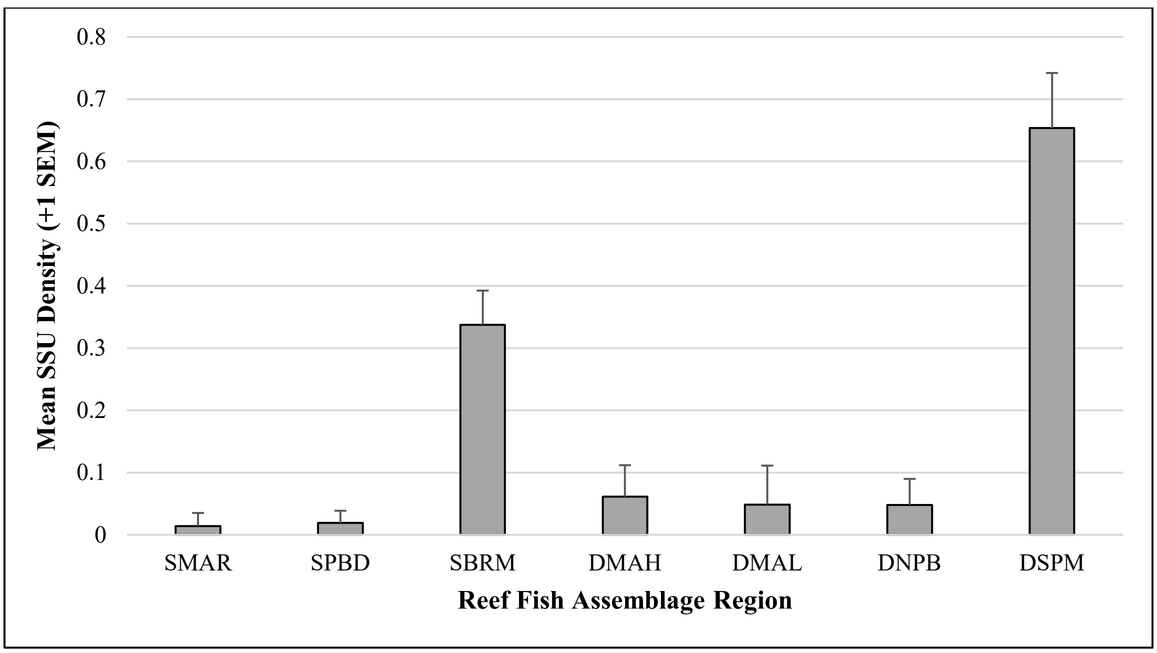


Figure 54. Hogfish mean SSU density by Reef Fish Assemblage Region, with all years combined.

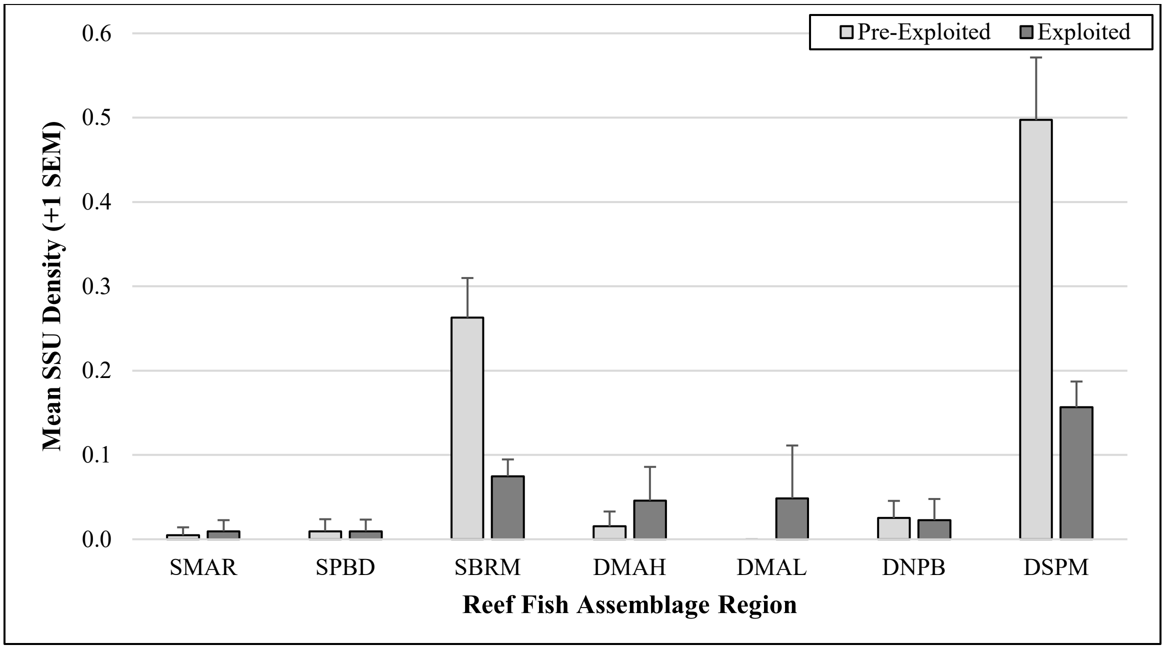


Figure 55. Mean SSU density of Hogfish by Reef Fish Assemblage Region; pre-exploited and exploited life-stage comparison with all years combined.

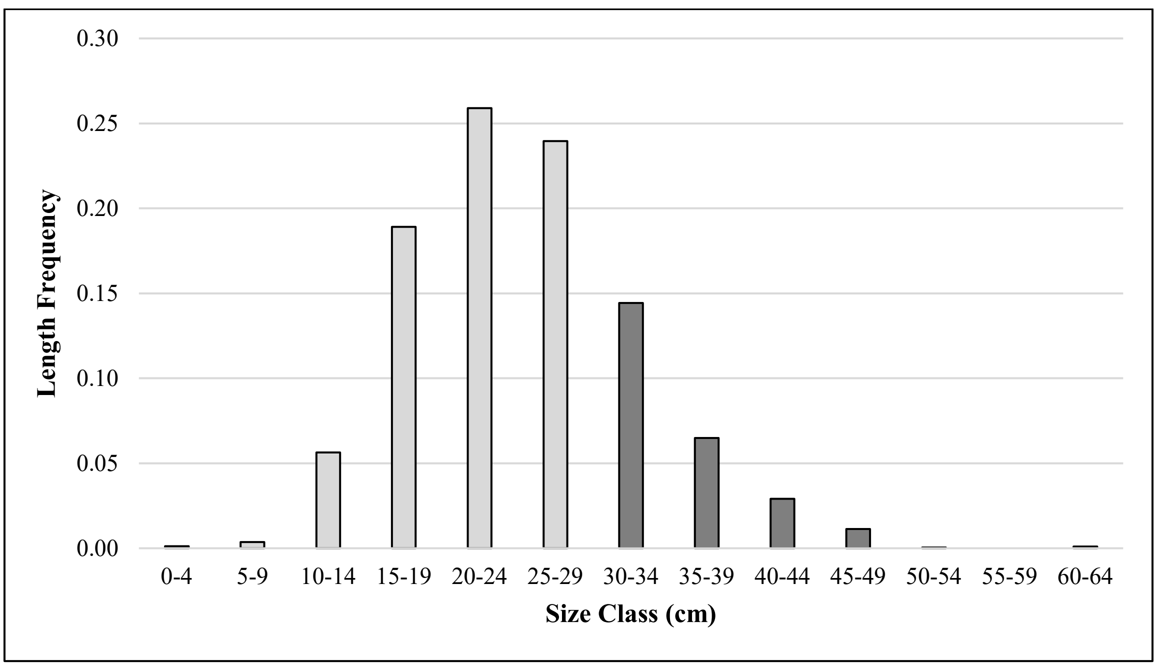


Figure 56. Domain-wide length frequency of Hogfish by size class. Darker gray indicates exploited size classes; minimum legal size of harvest for this species is 12” TL (30.5 cm in state and federal waters).

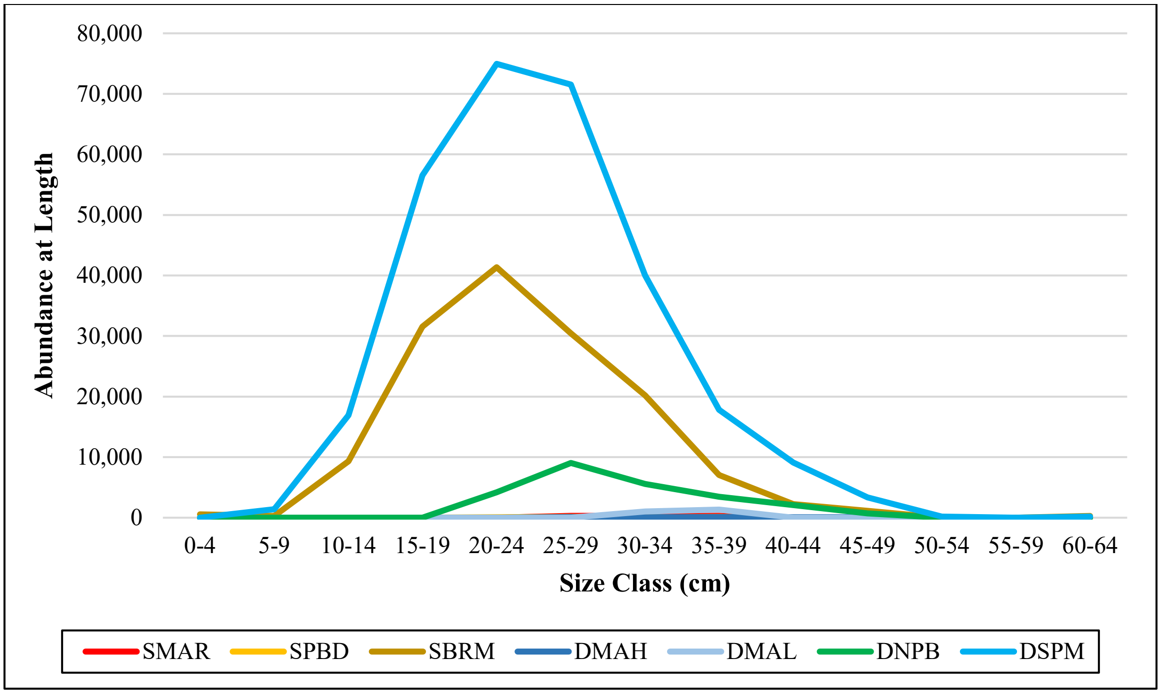


Figure 57. Length frequency comparison for Hogfish by total estimated assemblage region-wide population size (abundance at length) for southeast Florida, with all years combined.

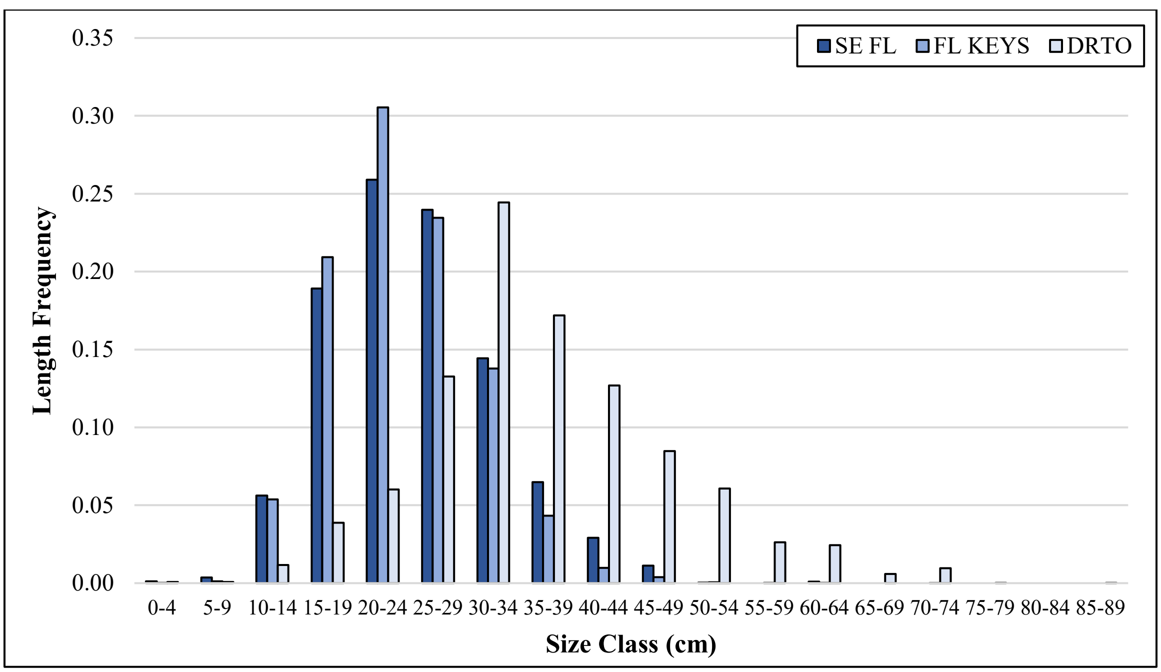


Figure 58. Length frequency comparison for Hogfish, by proportion (all bars sum to 1 for each region regardless of sample size or number of fish counted), for the southeast Florida region, Florida Keys, and Dry Tortugas during the 2012-2016 time period.

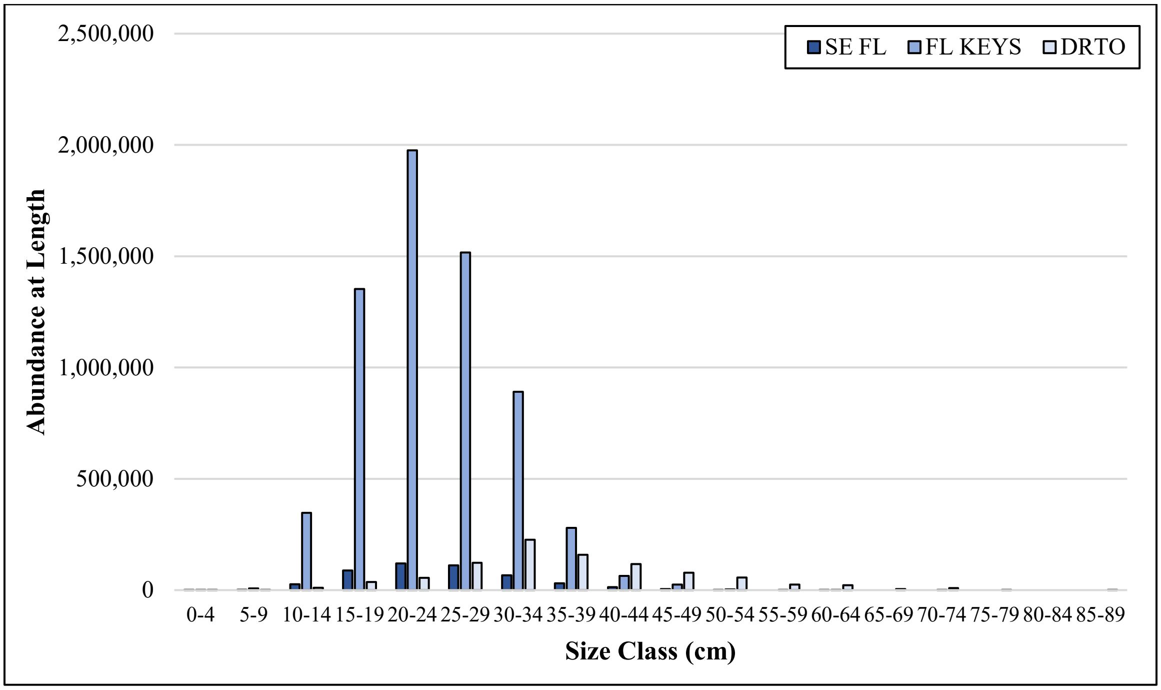


Figure 59. Length frequency comparison for Hogfish, by total estimated domain-wide population size (abundance at length), for the southeast Florida region, the Florida Keys, and the Dry Tortugas for the 2012-2016 time period.

4.1.11. Exploited Species: Gray Triggerfish

Figure 60. Gray Triggerfish (*Balistes capriscus*).

As snapper and grouper fisheries have become depleted and more tightly regulated, formerly untargeted species such as the Gray Triggerfish (Balistidae) have become increasingly valuable and more frequently harvested (Kelly, 2014). Their curiosity, aggression, and dentition are legendary among both the fishing and diving community.

Gray Triggerfish generally settle out of the pelagic environment onto benthic habitats during their first year once they reach ~100 cm FL (Burton et el., 2015). Post-settlement and early-stage juveniles are frequently associated with floating rafts of *Sargassum* sp. seaweed in the pelagic environment, which serves as nursery habitat for many other species as well (Settle, 1993; Wells and Rooker, 2004). When winds and currents drive large quantities of *Sargassum* onshore in southeast Florida, the local reefs (especially the nearshore habitats) may experience large pulses in recruitment of these species, including Gray Triggerfish. There were a few occasions in this study where large aggregations of small individuals (<15 cm) were recorded in shallow-water nearshore habitats of <5m depth, indicating these habitats may serve as essential habitat for younger size classes (note the peak for the SPBD assemblage region in Figures 9 and 10). Burton et al. (2014) reported that Gray Triggerfish grow moderately fast, attaining an average size of ~35.0 cm by age three. In another fishery-independent study in the southeastern U.S., the L_{50} for males was reported at 18.4 cm, with all males >27.0-28.0 cm being mature. For females, L_{50} was 17.7 cm, with all females >25.1-26.0 cm being mature (Kelly-Stormer et al., 2017). The minimum legal size of harvest in the state of Florida is 12" FL (30.5 cm) and 14" (35.4 cm) in federal waters (FWC, 2017a).

Gray Triggerfish spawning in the southeastern U.S. occurs from April through September, with peak spawning in June and July (Kelly-Stormer et al., 2017). This species is a haremic and territorial nesting species, with one male reproducing with multiple females (Simmons and Szedlmayer, 2012). The nest is protected by the male for 24-48 hours until the larvae enter the pelagic stage (MacKichan and Szedlmayer, 2007). During this study, there were multiple accounts of aggregations (>20 individuals) of Gray Triggerfish larger than 25 cm, however no reproductive behavior or nesting areas were seen. To date there are no reports of this species spawning in southeast Florida.

Gray Triggerfish was the 18th most frequently encountered and 24th most abundant species in this study, with a mean percent occurrence ($\bar{P}$) of 40.9% and mean density ($\bar{D}$) of 1.46 fishes/SSU (Appendix 2). While this species' geographic range extends from Nova Scotia to Argentina (Robins and Ray, 1986), the percent occurrence in the Florida Keys and Dry Tortugas was below 10% [Florida Keys ($\bar{P}$=8.2, $\bar{D}$=0.13); Dry Tortugas ($\bar{P}$=0.5, $\bar{D}$=0.00)]. Comparison of Gray Triggerfish densities by reef fish assemblage region indicated that the majority of the population resided in deeper habitats, with a general increase in density moving north (Figure 61). Comparison of population densities by their exploitation phases revealed much the same pattern for both pre-exploited and exploited phases (Figure 62). Length frequency analysis showed that the majority of the population was in the pre-exploited phase, with the exploited phase comprising 24% of the population (Figure 63).

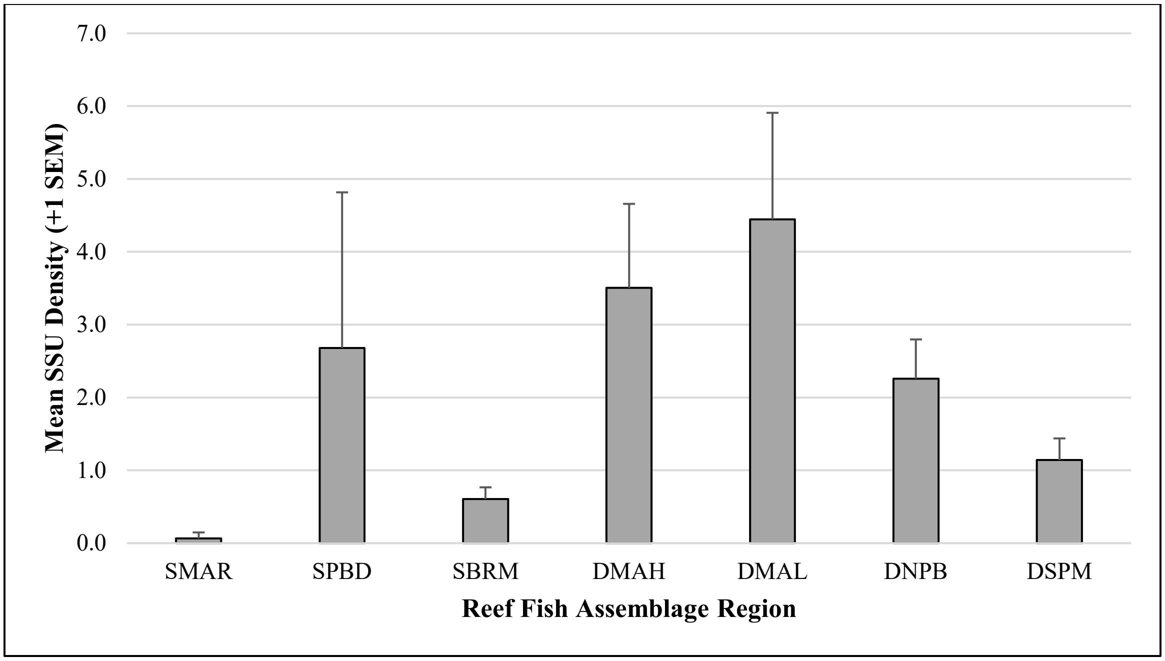


Figure 61. Gray Triggerfish mean SSU density by Reef Fish Assemblage Region, with all years and life-stages combined.

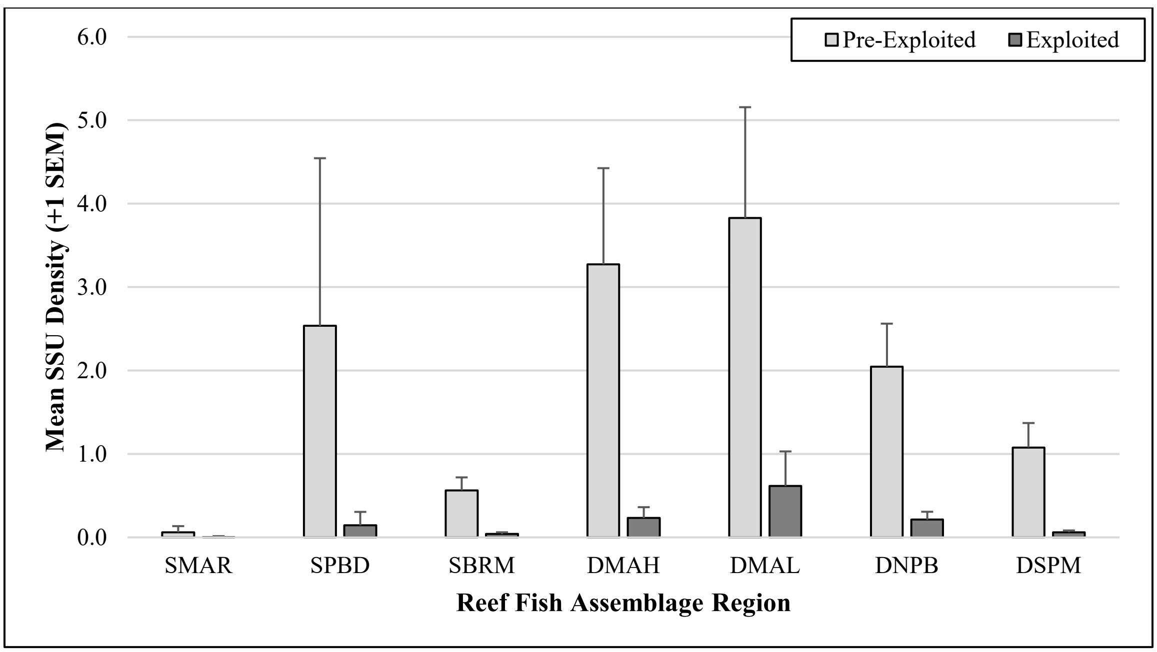


Figure 62. Mean SSU density of Gray Triggerfish by Reef Fish Assemblage Region; pre-exploited and exploited life-stage comparison with all years combined.

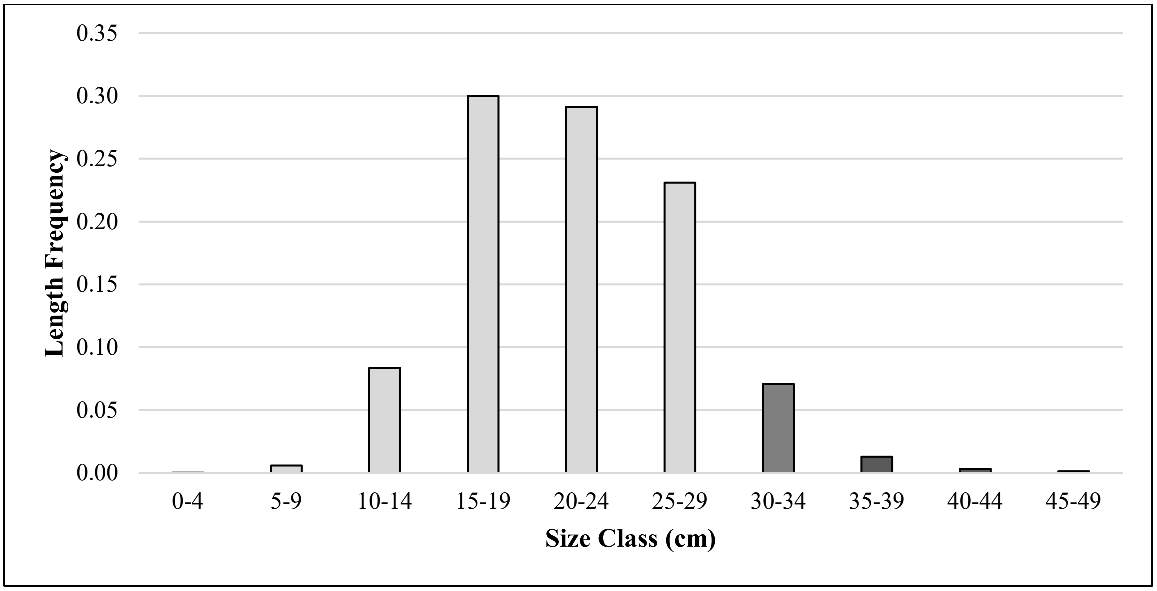


Figure 63. Domain-wide length frequency of Gray Triggerfish by size class, with all years and strata combined. Darker gray indicates exploited size classes; minimum legal size of harvest for this species is 12-14” TL (30.5 cm in state waters; 35.6 cm in federal waters).

The estimated abundance at length curves by reef fish assemblage region indicated that the greatest portion of the Gray Triggerfish population in the southeast Florida region was in the Deep North Palm Beach (DNPB) region (Figure 64). Their estimated abundance declined in other regions both north and south of DNPB. This may be due to their relatively high abundance on the Deep Ridge Complex habitat and its broad expanse. Interestingly, a comparison of the estimated abundance at length curves for this species from southeast Florida, the Florida Keys, and Dry Tortugas indicated a greater abundance of the larger size classes (>20 cm) in the Dry Tortugas (Figure 65), although the total estimated population size was miniscule compared to the Florida Keys and southeast Florida (Figure 66). Curiously, both southeast Florida and Florida Keys estimated length at abundance analyses displayed similar declines in larger size classes (>30 cm), with only a small fraction of the overall population represented by the exploited phase: 8.8% and 6.6%, respectively.

The average recorded size for this species in the southeast Florida region was 21.6 cm, and the average size of the exploited-phase (Lbar) was 32.5 cm. In total, 8.8% of the Gray Triggerfish population qualified as exploited-phase (≥30 cm). The largest individual recorded was 46 cm FL from the RGDP habitat in the Deep Martin Low (DMAL) assemblage region at a depth of 24 m. This compares well with previously reported maximum total lengths of 60 cm and 44 cm (Harmelin-Vivien and Quero, 1990; Figueiredo et al., 2002).

In another study, a greater proportion of the larger size classes consisted of males (Kelly-Stormer et al., 2017). If the decline of exploited individuals in the Florida Keys and southeast Florida is from fishing, this activity may be targeting a disproportionate number of males, which in turn may have implications on the spawning success of the females in certain areas. For long-term sustainability, it is important to know if the sex ratio in the exploited phase is being affected by fishing and how that is impacting the population's reproductive behaviors and success, as well as its genetic diversity. Other studies have recommended increased fishery-independent sampling off southern Florida to improve understanding of timing and spawning locations for this and other commercially important reef fishes (Farmer et al., 2017).

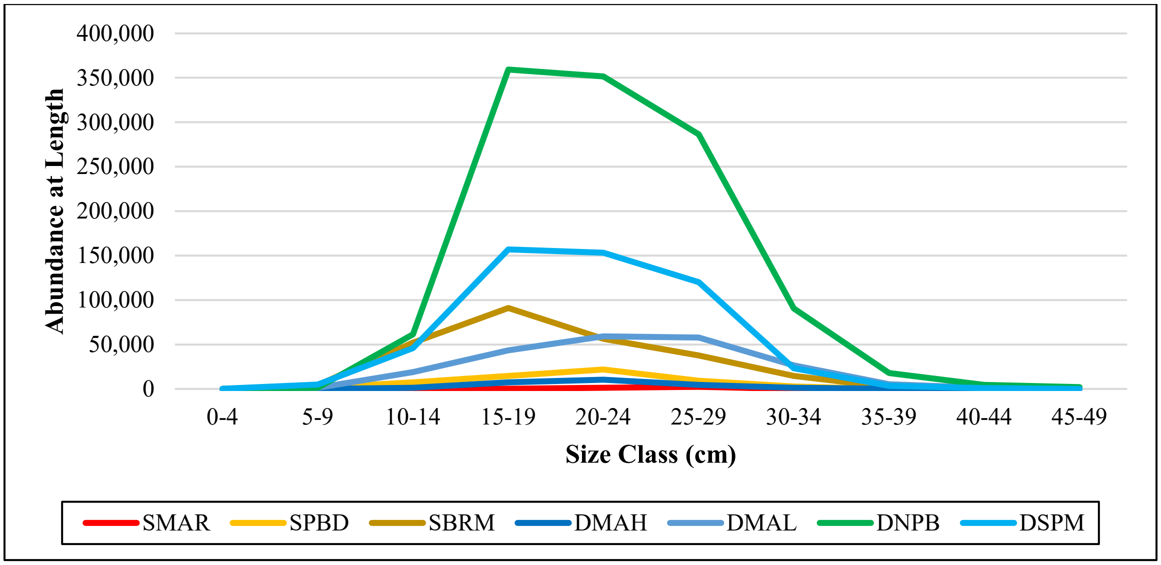


Figure 64. Length frequency comparison for Gray Triggerfish by total estimated assemblage region-wide population size (abundance at length) for southeast Florida, with all years combined.

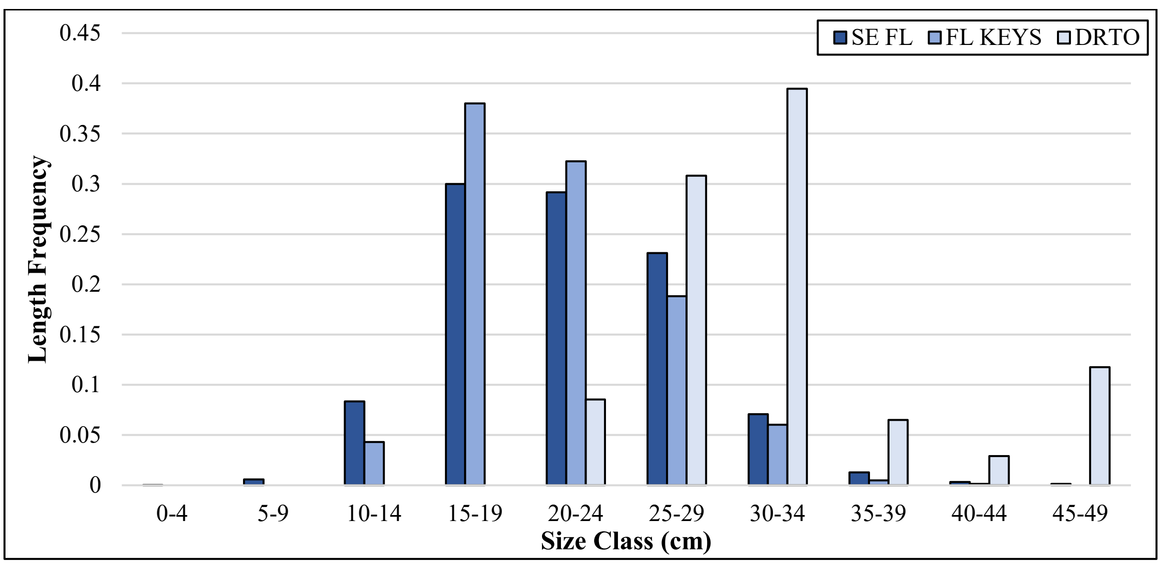


Figure 65. Length frequency comparison for Gray Triggerfish, by proportion (all bars sum to 1 for each region regardless of sample size or number of fish counted), for the southeast Florida region, Florida Keys, and Dry Tortugas during the 2012-2016 time-period.

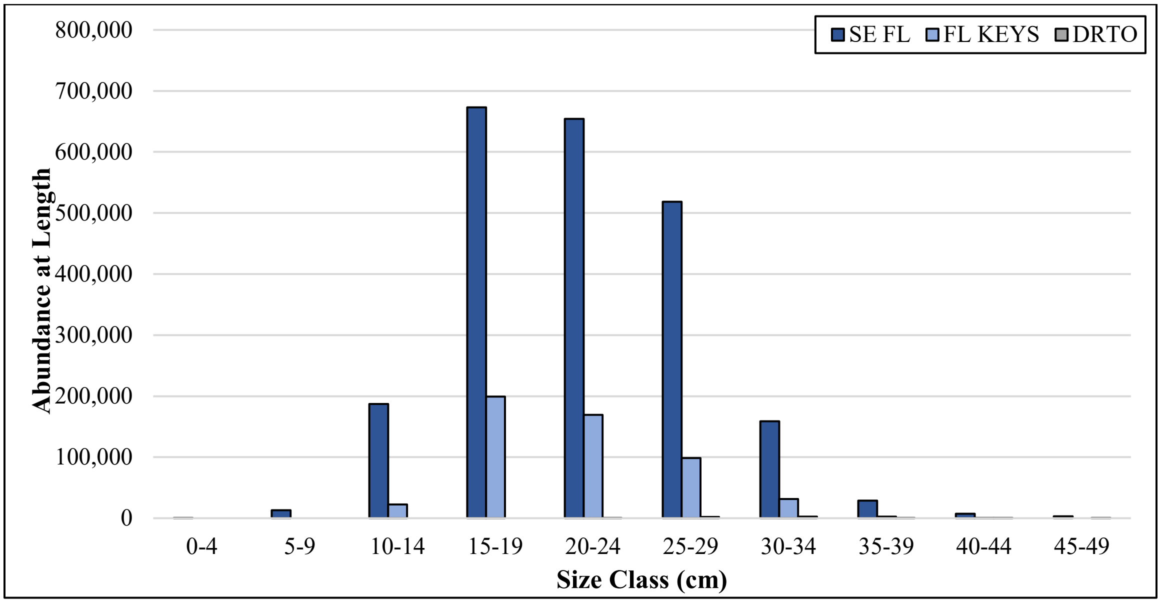


Figure 66. Length frequency comparison for Gray Triggerfish, by total estimated domain-wide population size (abundance at length), for the southeast Florida region, the Florida Keys, and the Dry Tortugas for the 2012-2016 time period.

4.1.12. Invasive Species: Lionfish

Figure 67. Red Lionfish (*Pterois volitans*).

Due to the rapid spread of the invasive Red Lionfish in the Western Atlantic and the increasing level of concern that stakeholders in southeast Florida have regarding impacts on native species, a brief discussion of the data collected for this species during this study is included here. While the Red Lionfish is now a well-established invasive species throughout Florida and the greater Caribbean, there were several other exotic species encountered during this study, including the Orbicular Batfish (*Platax orbicularis*), Sailfin Tang (*Zebrasoma velifer*), and Yellowtail Tang (*Z. xanthurum*), but they were mostly single occurrences. A great amount of research has been devoted to understanding the spread and resultant impact of lionfishes on populations of native species (both as potential prey items and competitors for resources) (Albins and Hixon, 2008; Morris and Akins, 2009; Green et al., 2012). Removal efforts aimed at either reducing local populations or eradicating them completely have been strongly encouraged, but there is still much debate about the effectiveness of these efforts and what other management strategies should be implemented to reduce their impact.

Red Lionfish (technically *Pterois* spp. = *Pterois volitans/miles* complex; both are present in Florida and *P. volitans* is more common, but visual differentiation is challenging) were the 58th most frequently observed species in this study, with mean percent occurrence ($\bar{P}$) of 12.5% and mean density ($\bar{D}$) of 0.12 fish/SSU. There were relatively few Red Lionfish encountered in shallow water, with the vast majority being recorded from the deeper assemblage regions (Figure 68). The length frequency curve for this species in the southeast Florida region indicated that this species is a textbook case of a healthy population, with a very symmetrical curve and larger size classes being well represented (Figure 69). When length frequency curves were compared for each reef fish assemblage region, there were more Red Lionfish in the Deep South Palm Beach Miami (DSPM) assemblage region, followed by the Deep North Palm Beach (DNPB) and Deep Martin Low (DMAL) regions (Figure 70).

Examination of length frequency curves along the entire FRT indicated a high degree of similarity between all regions, but with a greater percentage of juveniles showing up in southeast Florida region and Florida Keys surveys (Figure 71). In terms of overall population size, the Florida Keys appears to have the largest, followed by the southeast Florida and the Dry Tortugas (Figure 72). Interestingly, the two largest recorded lionfish captured in the Caribbean have been reported from Islamorada, Florida; one measuring 47.4 cm (18.6”) was landed in 2013 (Aguilar-Perera et al., 2013) and another at 47.7 cm (18.8”) in 2015 (FWC, 2017b).

Currently, there is no minimum legal size of harvest for this species in the state of Florida (FWC, 2017a). However, L_{50} has been reported as 19.0 cm TL for females (which is approximately 40% of their maximum size) and 10.0 cm TL for males (Morris, 2009; Gardener et al., 2015). In this study, the average size for Red Lionfish was 21.3 cm, and the average size of the exploited-phase (Lbar) was 26.6 cm. In total, 62% of the Red Lionfish population in southeast Florida may be classified as reproductively mature (≥19 cm). This clearly has implications for the long-term fate of this species in the region. If the length frequency curves for the regional population of Red Lionfish closely resembles the presumed standard for how things should look for a healthy population, then this has serious implications for ongoing eradication and management efforts. In addition, this serves as a sobering comparison against the population structures for several of the previously discussed exploited species, such as: Red Grouper, Mutton Snapper, Yellowtail Snapper, Hogfish, and Gray Triggerfish, which may justifiably be considered as depleted within the survey domain.

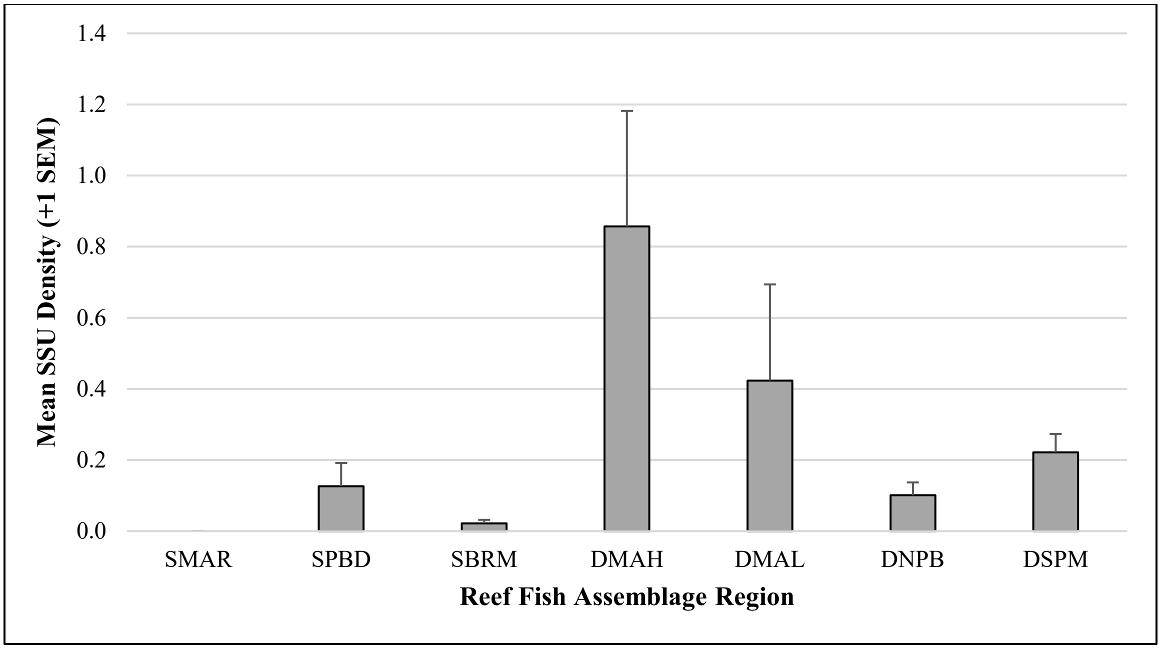


Figure 68. Mean SSU density of Red Lionfish by Reef Fish Assemblage Region, with all years combined.

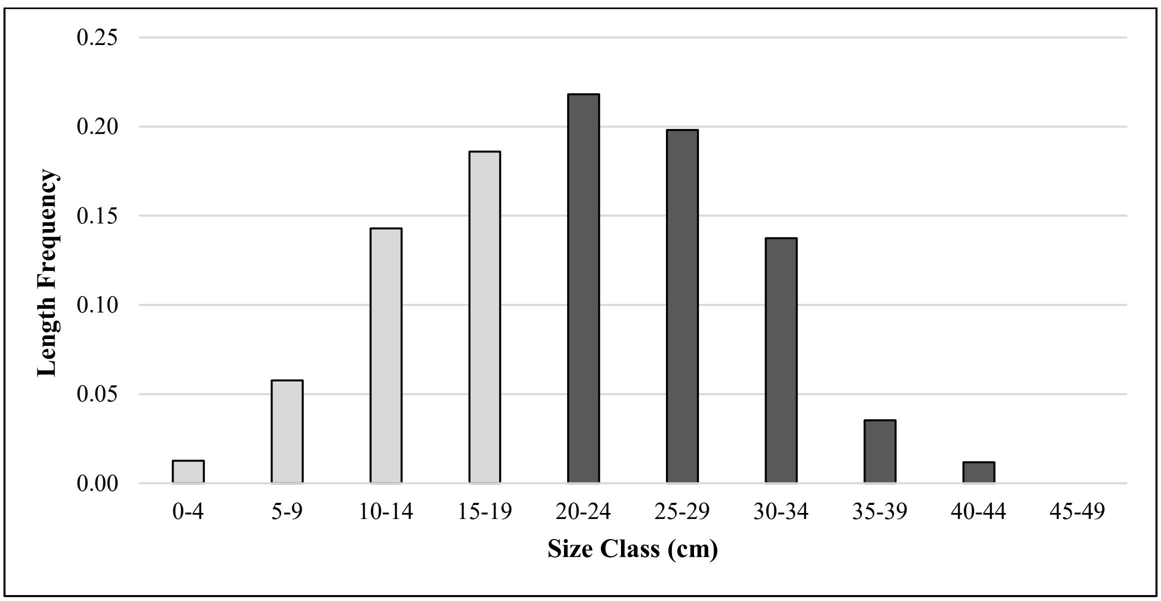


Figure 69. Domain-wide length frequency of Red Lionfish by size class, with all years and strata combined. Darker gray bars indicate size of reproductive maturity (L_m), which has been reported as 7.5” TL (19 cm) for females.

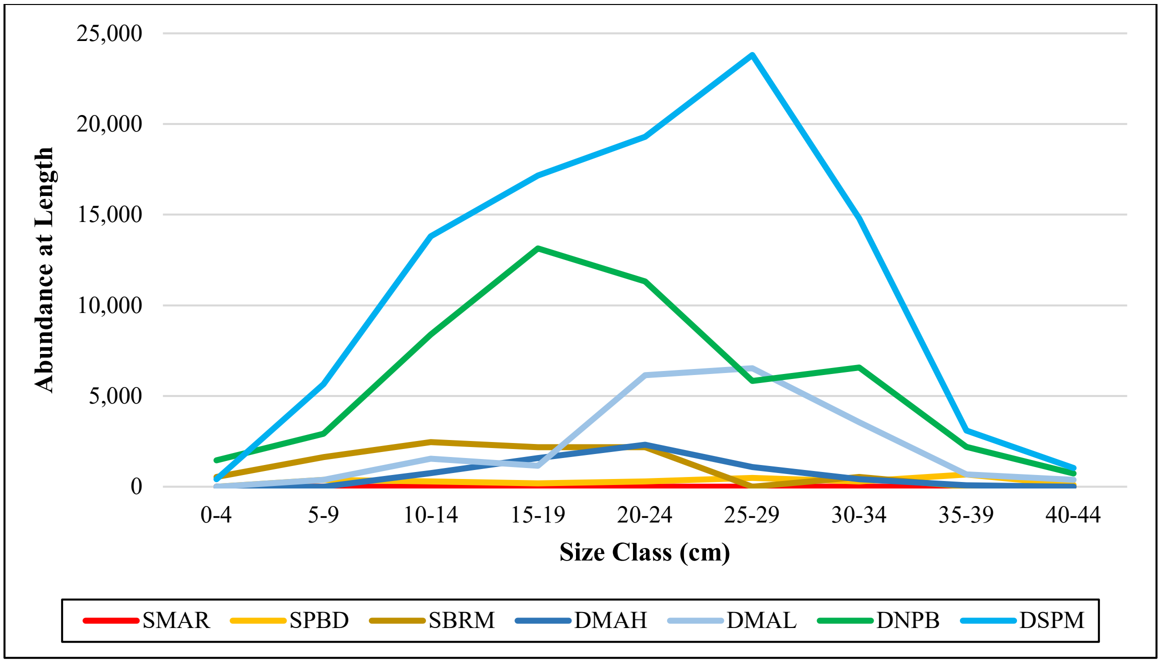


Figure 70. Length frequency comparison for Red Lionfish by total estimated assemblage region-wide population size (abundance at length) for southeast Florida, with all years combined.

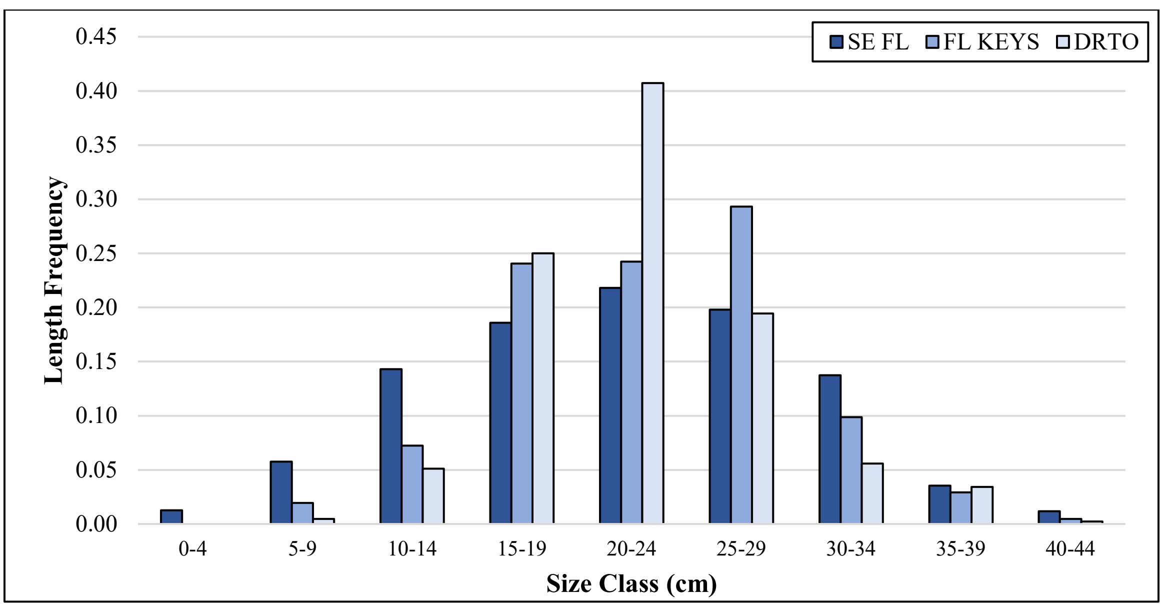


Figure 71. Length frequency comparison of Red Lionfish, by proportion (all bars sum to 1 for each region regardless of sample size or number of fish counted), for the southeast Florida region, Florida Keys, and Dry Tortugas during the 2012-2016 time period.

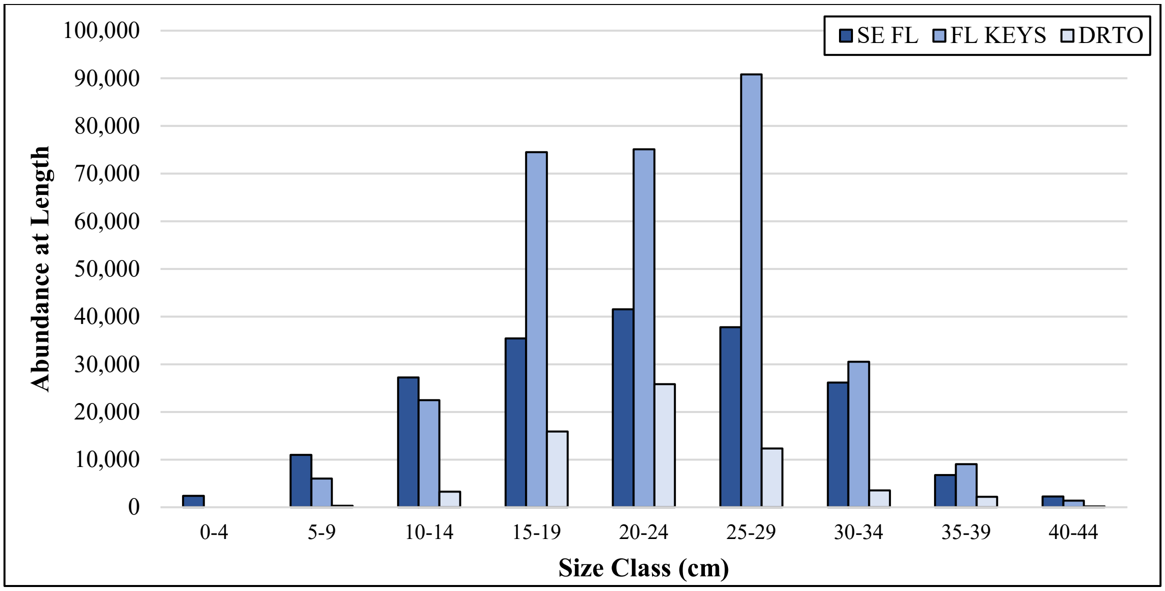


Figure 72. Length frequency comparison for Red Lionfish by total estimated domain-wide population size (abundance at length), for the southeast Florida region, the Florida Keys, and the Dry Tortugas for the 2012-2016 time period.

4.1.13. Other Species of Management Interest: Fishes ≥100 cm TL

Despite many years of exploitation and size-selective extraction, there are still some large commercially and recreationally important fishes residing in southeast Florida. Several large groupers, sharks, and other fishes were encountered during survey dives over the five-year study period. However, sometimes they were not present in the actual survey area or during the survey time period, but rather they were observed while transitioning between stations or while looking around after the surveys were completed (Figure 73).

The largest fishes recorded in the dataset for all years and strata combined, in descending order of greatest observed individual length for all fishes ≥100 cm TL (and the total number ≥100 cm seen) are listed as follows: Nurse Shark (*Ginglymostoma cirratum,* 29), Giant Manta Ray (*Manta birostris,* 1), Bull Shark (*Carcharias leucas,* 4), Tiger Shark (*Galeocerdo cuvier,* 1), Caribbean Reef Shark (*Carcharias perezei,* 13), Goliath Grouper (*Epinephelus itajara,* 37), Lemon Shark (*Negaprion brevirostris,* 2), Scalloped Hammerhead Shark (*Sphyrna lewini,* 2), Spotted Eagle Ray (*Aetobatus narinari,* 6), Sandbar Shark (*Carcharias plumbeus,* 2), Green Moray Eel (*Gymnothorax funebris,* 16), Tarpon (*Megalops atlanticus,* 8), Great Barracuda (*Sphyraena barracuda,* 60), Atlantic Sailfish (*Istiophorus platypterus,* 1), Southern Stingray (*Dasyatis americanus,* 4), Bluespotted Coronetfish (*Fistularia tabacaria,* 11), African Pompano (*Alectis ciliaris,* 2), Cobia (*Rachycentron canadum,* 2), King Mackerel (*Scomberomorus cavalla,* 1), Spotted Moray Eel (*Gymnothorax moringa,* 7), Purplemouth Moray Eel (*Gymnothorax vicinus,* 1), Atlantic Guitarfish (*Rhinopterus lentiginosus,* 1), and Yellow Jack (*Carangoides bartholomaei*, 2). Recordings of fishes >100 cm in length were well distributed throughout the entire survey domain, although the largest were from the North Palm

Beach and Martin subregions, mainly due to more frequent encounters with sharks and Goliath Groupers.

Figure 73. A large (estimated ~100 cm TL) Gag Grouper (*Mycteroperca microlepis*) photographed on an RVC dive while transitioning between survey locations in South Palm Beach in late October 2016.

4.1.14. Temporal Considerations

This baseline report thoroughly explores the relationships between coral reef fish density and assemblage structure throughout the SEFCRI region over the course of five years. Although inter-annual variation and temporal fluctuations were not the focus of this report, they were nevertheless present and should not be overlooked. Significant changes in the direction and magnitude of population change and overall resource status may take longer than five years to reveal themselves. For example, 38% of the Queen Triggerfish (*Balistes vetula*) encountered in this study were recorded in 2016 alone. Many factors can contribute to temporal differences in community structure and abundance of reef fishes. Reef fish assemblages are influenced by a combination of abiotic and biotic variables, such as: reef morphology, water chemistry, temperature, depth, current regimes, terrestrial influences (i.e. runoff, sedimentation, nutrient levels), extreme weather events (hurricanes, cold snaps), large scale climate changes, benthic community composition, stochastic settlement and recruitment dynamics (i.e. larval supply, predation, competition, etc.), and changes in biogeographic distribution of species. In addition, anthropogenic impacts (water quality/pollution, coastal construction and development, habitat loss) and associated management practices (beach nourishment, fishing regulations) are an influential presence in the coastal marine environment. Many reef fish populations fluctuate on seasonal or multi-year scales in response to a combination of the aforementioned variables. Because population levels can fluctuate greatly from year to year, understanding of how biotic and abiotic variables interact with one another and change in response to management practices will be improved by the evaluation of long-term datasets, such as those being generated by continuing monitoring efforts efforts along the FRT through the National Coral Reef Monitoring Program (NCRMP). Furthermore, because effective management of fish resources requires effective monitoring of populations of early life-stages and their habitats (i.e. nearshore hardbottom and ledges), we recommend this be considered in future surveys. In addition, better fishery-independent data from hardbottom habitats below 33 m and artificial reefs are needed to better understand local populations and trends.

5. FINDINGS AND CONCLUSIONS

Comparison of data from all subregions of the FRT showed a pattern of increasing percent occurrence and density, and similar Lbar, for most but not all target species from southeast Florida through the Florida Keys and into the Dry Tortugas. Likewise, this comparison leads to the finding that many species of fisheries interest are depleted in the northern portion of the FRT as they are in the Florida Keys. These results validate assertions that many target species are being locally depleted, with large reproductively mature adults being the prime targets. Some species are affected more critically than others due to the combination of slow growth, complex life histories, and behavioral tendencies that have resulted in severely reduced abundance and truncated size class distributions. Some species' densities may be more dependent upon regional differences in availability of essential nursery and juvenile habitat, while status of others may be driven more by geographical differences in distribution. Protection from exploitation is also playing a part, as demonstrated by data from the Dry Tortugas. The southeast Florida fishery-independent dataset does not depict a pristine condition that provides a target for preservation. Rather, the dataset provides a picture of fish assemblages that have already experienced substantial anthropogenic impacts; it provides a critical baseline for management strategies aimed at improvement of valuable and heavily utilized natural resources.

Healthy reefs support not only more robust populations of economically important species, but also an entire recreational diving industry that thrives on the diversity and abundance of reef fish assembalges associated with the high-relief ledges and hardbottom habitats common in southeast Florida. On a recreational dive boat, it is not uncommon to find spearfishers alongside photographers, scientists, wreck divers, and new student divers. Some areas are heavily utilized as a multi-use resource. For example, consider the relatively narrow strip of hardbottom/coral reef habitat off of southern Palm Beach County. Head (fishing) boats and dive boats typically come out at least twice a day every day (weather permitting) and target the same reef lines and bottom features, thereby repeatedly targeting the same reef fish resources over and over again. It is no coincidence that some of the most valued target species accessible to the majority of divers and fishers are species that exhibit the most severaly truncated length frequency curves (Red Grouper, Mutton Snapper, and Hogfish). It is also not coincidental that some of these species also share unique life history traits (i.e., slow growth, large size at reproductive maturity, long life span, sex change) that exacerbate the issue. The combination of biology and high demand can largely explain why single-species fisheries management technqiues on the stock-scale alone have proven inadequate in ameliorating the effects of fishing pressure and local depletions of highly valued reef fish. Most stakeholders agree high fishing pressure on spawning aggregations is not conducive to maintaining reproduction necessary for sustaining reef fish populations. With these factors in mind, along with the limited number of effective management options currently available to work with, it seems that considering establishment of marine protected areas or other place-based management approaches to protect spawning aggregations and meet management goals in southeast Florida is perhaps an eventuality that should be more thoroughly discussed sooner rather than later.

6. REFERENCES

Acosta, A. and R. Beaver. 1998. Fisheries, Growth, and Mortality of Yellowtail Snapper, *Ocyurus chrysurus*, in the Florida Keys, Florida, U.S.A. Proceedings of the Gulf and Caribbean Fisheries Institute 50, 851-870.

Aguilar-Perera, A. and R.S. Appeldoorn. 2008. Spatial distribution of marine fishes along a cross-shelf gradient containing a continuum of mangrove-seagrass-coral reefs off southwestern Puerto Rico. Estuarine, Coastal and Shelf Science 76(2), 378-394.

Aguilar-Perera, A., Perera-Chan, L. and L. Quijano-Puerto. 2013. Record Body Size for the Red Lionfish, Pterois volitans (Scorpaeniformes), in the Southern Gulf of Mexico. Gulf and Caribbean Research 25, 121-123.

Albins, M.A. and M.A. Hixon. 2008. Invasive Indo-Pacific lionfish *Pterois volitans* reduce recruitment of Atlantic coral-reef fishes. Marine Ecology Progress Series 367, 233-238.

Allen, G.R. 1985. FAO Species Catalog Vol. 6. Snappers of the World. An Annotated and Illustrated Catalogue of Lutjanid species known to Date. FAO Fisheries Synopsis 6(125), 208 p.

Alonzo, S.H., Ish, T., Key, M., MacCall, A.D. and M. Mangel. 2008. The importance of incorporating protogynous sex change into stock assessments. Bulletin of Marine Science 83, 163-179.

Appeldoorn, R.S., Recksiek, C.W., Hill, R.L., Pagan, F.E. and G.D. Dennis. 1997. Marine protected areas and reef fish movements: the role of habitat in controlling ontogenetic migration. Proceedings of the 8th International Coral Reef Symposium, Vol. 2. 1917-1922.

Ault, J.S. and E.C. Franklin. 2011. Fisheries resource status and management alternatives for the southeast Florida region. Report to FDEP. Miami Beach, Florida. 105 p.

Ault, J.S., Smith, S.G., Zurcher, N., Bryan, D.R. and J. Blondeau. 2012. CRCP Project 3A: Development of a coral reef fishery-independent assessment protocol for the southeast Florida region. Florida Department of Environmental Protection. Miami, Florida. 67 p. Retrieved from: www.dep.state.fl.us/coastal/programs/coral/reports/-DEP_CRCP_3a_Report.pdf (June 6, 2014).

Banks, K.E., Riegl, B.M., Richards, V.P., Walker, B.K., Helmle, K.P., Jordan, L.K.B., Phipps, J., Shivji, M., Spieler, R.E. and R.E. Dodge. 2008. The reef tract of continental Southeast Florida (Miami-Dade, Broward, and Palm Beach Counties, USA). In: Riegl B. and R.E. Dodge (eds.) Coral Reefs of the USA. Springer-Verlag, Dordrecht, 125-172.

Bannerot, S.P. 1984. The dynamics of exploited groupers (Serranidae): An investigation of the protogynous hermaphroditic reproductive strategy. Ph.D. Dissertation, University of Miami, Coral Gables, Florida. 393 p.

Bannerot, S.P., Fox, W.W. Jr. and J.E. Powers. 1987. Reproductive strategies and the management of snappers and groupers in the Gulf of Mexico and Caribbean. In: J.J. Polovina and S. Ralston (eds.) Tropical snappers and groupers: biology and fisheries management. Westview Press, Boulder, Colorado, 561-603.

Bartels, C.T. and K.L. Ferguson. 2006. Preliminary Observations of Abundance and Distribution of Settlement-Stage Snappers in Shallow, Nearshore Seagrass Beds in the Middle Florida Keys. Proceedings of the Gulf and Caribbean Fisheries Institute 57, 235-247.

Behringer, D.C., Swett, R.A. and T.K. Frazer. 2011. Determining coral reef impacts associated with boat anchoring and user activity in southeast Florida. Florida Department of Environmental Protection – Coral Reef Conservation Program, Miami Beach, Florida. 66 p.

Billings, V.C. and J.L. Munro. 1974. The biology, ecology, exploitation and management of Caribbean reef fishes: Pomadasydae (grunts). Part 5. Research Report of the Zoological Department of the University of the West Indies 3, 1-128.

Bohnsack, J.A. and S.P. Bannerot. 1986. A stationary visual census technique for quantitatively assessing community structure of coral reef fishes. NOAA Technical Report NMFS 41.

Bohnsack, J.A., Harper, D.E., McClellan, D.B. and M. Hulsbeck. 1994. Effects of reef size on colonization and assemblage structure of fishes at artificial reefs off southeastern Florida, USA. Bulletin of Marine Science 55(2-3), 796-823.

Bortone, S.A. and J.L. Williams. 1986. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (South Florida): gray, lane, mutton and yellowtail Snappers. U.S. Fish and Wildlife Service Biological Report 82(11.52). U.S. Army Corps of Engineers, TR EL-82-4, 18 p.

Brandt, M.E., Zurcher, N., Acosta, A., Ault, J.S., Bohnsack, J.A., Feeley, M.W., Harper, D.E., Hunt, J., Kellison, T., McClellan, D.B., Patterson, M.E. and S.G. Smith. 2009. A cooperative multi-agency reef fish monitoring protocol for the Florida Keys coral reef ecosystem. Natural Resource Report NPS/SFCN/NRR – 2009/150. National Park Service, Fort Collins, Colorado.

Bryan, D.R., Kilfoyle, K., Gilmore, R.G. and R.E. Spieler. 2013. Characterization of the mesophotic reef fish community in south Florida, USA. Journal of Applied Ichthyology 29, 108-117.

Burgos, J.M., Sedberry, G.R., Wynaski, D.M. and P.J. Harris. Life history of red grouper (*Epinephelus morio*) off the coasts of North Carolina and South Carolina. Bulletin of Marine Science 80(1), 45-65.

Burton, M.L. 2001. Age, growth, and mortality of gray snapper, *Lutjanus griseus*, from the east coast of Florida. Fishery Bulletin 99(2), 254 p.

Burton, M.L. 2002. Age, growth and mortality of mutton snapper, Lutjanus analis, from the east coast of Florida, with a brief discussion of management implicatins. Fisheries Research 59, 31-41.

Burton, M.L., Potts, J.C., Carr, D.R., Cooper, M. and J. Lewis. 2014. Age, growth, and mortality of gray triggerfish (*Balistes capriscus*) from the southeastern United States. Fishery Bulletin 113(1), 27-39.

Chabanet, P., Ralambondrainy, H., Amanieu, M., Faure, G. and R. Galzin. 1997. Relationships between coral reef substrata and fish. Coral Reefs 16(2), 93-102.

Charton, J.A. and A.P. Ruzafa. 1998. Correlation between habitat structure and a rocky reef fish assemblage in the southwest Mediterranean. Marine Ecology 19(2), 111-128.

Chen, E. and J.F. Gerber. 1990. Climate. pp. 11-34 *in* Myers, R.L. and J.J. Ewel (eds.) Ecosystems of Florida. University of Central Florida Press, Orlando.

Coleman, F.C., Koenig, C.C., Huntsman, G.R., Musick, J.A., Eklund, A.M., McGovern, J.C., Chapman, R.W., Sedberry, G.R. and C.B. Grimes. 2011. Long-lived Reef Fishes: The Grouper-Snapper Complex. Fisheries 25(3), 14-21.

Colin, P.L. 1982. Spawning and larval development of the hogfish, *Lachnolaimus maximus* (Pisces: Labridae). Fishery Bulletin-United States, National Marine Fisheries Service (USA) 80, 853-862.

Collins, L.A. and J.H. Finucane. 1989. Reproductive biology of yellowtail snapper, *Ocyurus chrysurus*, from the Florida Keys. US Department of Commerce, NOAA, NMFS, SEFSC, Panama City Laboratory, PCL Contribution 89-11, 23 p.

Colvocoresses, J.and A. Acosta. 2007. A large-scale field comparison of strip transect and stationary point count methods for conducting length-based underwater visual surveys of reef fish populations. Fisheries Research 85(1-2), 130-141.

Cuellar, N., Sedberry, G.R., Machowski, D.J., and M.R. Collins. 1996. Species composition, distribution and trends in abundance of snappers of the southeastern USA, based on fishery-independent sampling. In: Arreguin-Sánchez, F., Munro, J.L., Balgos, M.C. and D. Pauly (eds.) Biology, Fisheries and Culture of Tropical Snappers and Groupers, ICLARM Conference Proceedings 48, 59–73.

Cummings, N. 2007. Important aspects of the life history of mutton snapper, *Lutjanus analis*, with emphasis on populations in the Caribbean. SEDAR-14 Assessment Workshop Report No. 2.

Darcy, G.H. 1983. Synopsis of biological data on the grunts *Haemulon aurolineatum* and *H. plumieri* (Pisces: Haemulidae). NOAA Tech. Report NMFS Circular 448. 37 p.

de Silva, J.A. and M.D. Murphy. 2001. A Summary of the Status of White Grunt *Haemulon plumieri* from the East Coast of Florida. Report to the Florida Fish and Wildlife Conservation Commission, Florida Marine Research Institute.

Domeier, M.L. and M.E. Clarke. 1992. A laboratory produced hybrid between *Lutjanus synagris* and *Ocyurus chrysurus* and a probable hybrid between *L. griseus* and *0. chrysurus* (Perciformes: Lutjanidae). Bulletin of Marine Science 50(3), 501-507.

Domeier, M.L., Koenig, C. and F. Coleman. 1996. Reproductive biology of gray snapper (*Lutjanus griseus*) with notes on spawning for other Western Atlantic snappers (Lutjanidae). In: Arreguín-Sánchez, F., Munro, J.L., Balagos, M.C. and D. Pauly (eds.) Biology and culture of tropical groupers and snappers. ICLARM Conference Proceedings 48, 449 p.

Domeier, M.L. and P.L. Colin. 1997. Tropical reef fish spawning aggregations: defined and reviewed. Bulletin of Marine Science 60, 698-726.

Domeier, M.L., Koenig, C. and F. Coleman. 1998. Reproductive Biology of the Gray Snapper (*Lutjanus griseus*), with Notes on Spawning for other Western Atlantic Snappers (Lutjanidae). pp. 189-201. In: F. Arreguin-Sanchez, J.L. Munro, M.C. Balgos and D. Pauly (eds.) Biology and culture of tropical groupers and snappers. ICLARM Conference Proceedings 48, 449 p.

Ebeling, A.W. and M.A. Hixon. 1991. Tropical and temperate reef fishes: Comparison of community structures. pp. 509-563 *in* Sale, P.F. (ed.) The ecology of fish on coral reefs. Academic Press, San Diego, California, USA.

Edgar, G.J., Barrett, N.S., and A.J. Morton. 2004. Biases associated with the use of underwater visual census techniques to quantify the density and size-structure of fish populations. Journal of Expeirmental Marine Biology and Ecology 308(2), 269-290.

Farmer, N., Heyman, W.D., Karnauskas, M., Kobara, S., Smart, T.I., Ballenger, J.C., Reichert, M.J.M., Wyanski, D.M., Tishler, M.S., Lindeman, K.C., Lowerre-Barbieri, S.K., Switzer, T.S., Solomon, J.J., McCain, K., Marhefka, M. and G.R. Sedberry. 2017. Timing and locations of reef fish spawning off the southeastern United States. PloS ONE 12(3): e0172968. Available at: https://doi.org/10.1371/-journal.pone.0172968.

Faunce, C.H. and J.E. Serafy. 2007. Nearshore hábitat use by gray snapper (*Lutjanus griseus*) and bluestriped grunt (*Haemulon sciurus*): environmental gradients and ontogenetic shifts. Bulletin of Marine Science 80(3), 473-495.

Ferrell, D.J. and J.D. Bell. 1991. Differences among assemblages of fish associated with Zostera capricorni and bare sand over a large spatial scale. Marine Ecology Progress Series 72(1/2), 15-24.

Ferro, F., Jordan, L.K.B. and R.E. Spieler. 2005. Spatial variability of the coral reef fish assemblages offshore Broward County, Florida. NOAA Technical Memorandum N MFS-SEFSC-532. 73 p.

Figueiredo, J.L. de, A.P. dos Santos, N. Yamaguti, R.A. Bernardes and C.L. Del Bianco Rossi-Wongtschowski, 2002. Peixes da zona econômica exclusiva da Região Sudeste-Sul do Brasil: Levantamento com Rede de Meia-Água. São-Paulo: Editora da Universidade de São Paulo; Imprensa Oficial do Estado, 242 p.

Figuerola, M., Matos-Caraballo, D. and W. Torres. 1998. Maturation and reproductive seasonality of four reef fish species in Puerto Rico. Proceedings of the Gulf and Caribbean Fisheries Institute 50, 938-968.

Figuerola-Fernández, M. and W. Torres-Ruiz. 2001. Aspectos de la biología reproductive de la sama (*Lutjanus analis*) en Puerto Rico y recomendaciones para su manejo. Informe Final Laboratorio de Investigaciones Pesqueras, Puerto Rico Departamento de Recursos Naturales y Ambientales.

Fisco, D. 2016. Reef Fish Spatial Distribution and Benthic Habitat Associations on the Southeast Florida Reef Tract. Master's thesis. Nova Southeastern University. Retrieved from: http://nsuworks.nova.edu/occ_stuetd/408 (July 12, 2017).

Friedlander, A.M. and J.D. Parrish. 1998. Habitat characteristics affecting fish assemblages on a Hawaiian coral reef. Journal of Expeirmental Marine Biology and Ecology 224(1), 1-30.

Friedlander, A.M., Brown, E.K., Jokiel, P.L., Smith, W.R. and K.S. Rodgers. 2003. Effects of habitat, wave exposure, and marine protected area status on coral reef fish assemblages in the Hawaiian archipelago. Coral reefs 22(3), 291-305.

FWC. 2017a. Florida Recreational Sportfishing Regulations. Retrieved from: http://www.eregulations.com/florida/fishing/saltwater/florida-saltwater-fishing-species-and-limits (June 25, 2017).

FWC. 2017b. Florida Lionfish State Records. (Retrieved from: http://myfwc.com/-fishing/saltwater/recreational/lionfish/state-records/ (June 30, 2017).

FWC. 2017c. Hogfish conservation measures and boundaries set. Retrieved from: http://-myfwc.com/news/news-releases/2016/november/17/hogfish/ (August 31, 2017).

Garcia, E.R., Potts, J.C., Rulifson, R.A., and C.S. Manooch. 2003. Age and growth of yellowtail snapper, *Ocyurus chrysurus*, from the southeastern United States. Bulletin of Marine Science 72(3), 909-921.

García-Cagide, A., Claro, R. and B.V. Koshelev. 1994. Reproducción. pp. 187-262. In: R. Claro (ed.) Ecología de los peces marinos de Cuba. Academia de Ciencias de Cuba, Instituto de Oceanografia y Centro de Investigationes de Quintana Roo, Quintana Roo, México.

Gardener, P.G., Frazer, T.K., Jacoby, C.A., and R.P.E. Yanong. 2015. Reproductive biology of invasive lionfish (*Pterois* spp.). Frontiers in Marine Science 2, 7 p.

Gaut, V.C. and J.L. Munro. 1983. The biology, ecology, and bionomics of the grunts, Pomadasyidae. pp. 110–141 In: J.L. Munro (ed.) Caribbean coral reef fishery resources. ICLARM, Manilla, Philippines. 276 p.

Gilliam, D.S. and B.K. Walker. 2011. Benthic Habitat Characterization for the South Florida Ocean Measurement Facility (SFOMF) Protected Stony Coral Species Assessment (pp. 54). Fort Lauderdale, FL: Prepared for Seaward Services, Inc.

Gilliam, D.S., Dodge, R.E., Spieler, R.E., Halperin, A.A., C. Walton and K. Kilfoyle. 2015. Marine Biological Monitoring in Broward County, Florida: Year 14 (2013) Annual Report. 121 p.

Gilmore, R.G., Donohoe, C.J., Cooke, D.W. and D.J. Herrema. 1981. Fishes of the Indian River Lagoon and adjacent waters, Florida. Fort Pierce, FL: Harbor Branch Foundation, Inc.

GMFMC. 2008. Reef fish amendment 30A: greater amberjack-revised rebuilding plan, accountability measures and gray triggerfish-establish rebuilding plan, end overfishing, accountability measures, regional management, management thresholds and benchmarks. Gulf of Mexico Fishery Management Council, Tampa, Florida.

Gramer, L.J., Hendee, J.C., Stamates, S.J., Walker, B.K. and R.K. Walter. 2017. Upwelling on the Southeast Florida Shelf: Summer sea temperature extremes and associated nutrient fluxes over a 17 year period. NOAA Technical Memorandum, Miami, FL, National Oceanic and Atmospheric Administration.

Gratwicke, B. and M.R. Speight. 2005. The relationship between fish species richness, abundance and habitat complexity in a range of shallow tropical marine habitats. Journal of Fish Biology 66(3), 650-667.

Gratwicke, B. and M.R. Speight. 2005. Effects of habitat complexity on Caribbean marine fish assemblages. Marine Ecology Progress Series 292, 301-310.

Green, S.J., Akins, J.L., Maljković, A. and I.M. Côté. 2012. Invasive Lionfish Drive Atlantic Coral Reef Fish Declines. PLoS ONE 7(3): e32596. Available at: https://doi.org/10.137/journal.pone.0032596.

Gregg, K. 2013a. Management considerations for the southeast Florida coral reef ecosystem. Report Prepared for: NOAA Fisheries Southeast Region – Habitat Conservation Division, West Palm Beach, Florida. 42 p.

Gregg, K. 2013b. Literature review and synthesis of land-based sources of pollution affecting essential fish habitats in southeast Florida. Report Prepared for: NOAA Fisheries Southeast Region – Habitat Conservation Division, West Palm Beach, Florida. 55 p.

Grober-Dunsmore, R., Frazer, T. K., Beets, J.P., Lindberg, W.J., Zwick, P. and N.A. Funicelli. 2008. Influence of landscapr structure on reef fish assemblages. Landscape Ecology 23(1), 37-53.

Grüss, A., Harford, W.J., Schirripa, M.J., Velez, L., Sagarese, S.R., Shin, Y. and P. Verley. 2016. Management strategy evaluation using the individual-based, multispecies modeling approach OSMOSE. Ecological Modelling 340, 86-105.

Grüss, A., Thorson, J.T., Babcock, E.A. and J.H. Tarnecki. 2017. Producing distribution maps for informing ecosystem-based fisheries management using a comprehensive survey database and spatio-temporal models. ICES Journal of Marine Science, fsx120, Retrieved from: https://doi.org/10.1093/icesjms/fsx120 (September 15, 2017).

Harmelin-Vivien, M.L. and J.C. Quéro. 1990. Balistidae. pp. 1055-1060. In: Quéro, J.C., Hureau, J.C., Karrer, C., Post, A. and L. Saldanha (eds.) Check-list of the fishes of the eastern tropical Atlantic (CLOFETA). JNICT, Lisbon; SEI, Paris, and UNESCO, Paris. Vol. 2.

Harvey, E., Fletcher, D. and M. Shortis. 2002. Estimation of reef fish length by divers and by stereo-video: A first comparison of the accuracy and precision in the field on living fish under operational conditions. Fisheries Research 57(3), 255-265.

Harvey, E., Fletcher, D., Shortis, M.R. and G.A. Kendrick. 2003. A comparison of underwater visual distance estimates made by scuba divers and a stereo-video system: implications for underwater visual census of reef fish abundance. Marine and Freshwater Research 55(6), 573-580.

Heemstra, P.C. and J.E. Randall. 1993. Groupers of the World. FAO Fisheries Synopsis, no.125, vol. 9. Food and Agriculture Organization, Rome.

Hettler, W.F., and D.L. Barker. 1993. Distribution and abundance of larval fishes at two North Carolina inlets. Estuarine, Coastal and Shelf Science 37(2), 161-179.

Hixon, M.A. and J.P. Beets. 1989. Shelter characteristics and Caribbean fish assemblages: experiments with artificial reefs. Bulletin of Marine Science 44(2), 666-680.

IGFA. 2017a. IGFA World Records Database. International Gamefish Association, Fort Lauderdale, USA. Retrieved from: http://wrec.igfa.org/WRecordsList.aspx?lc=AllTackle&cn=Snapper,%20mutton (May 15, 2017).

IGFA. 2017b. IGFA World Records Database. International Gamefish Association, Fort Lauderdale, USA. Retrieved from: http://wrec.igfa.org/WRecordsList.aspx?lc=AllTackle&cn=Grunt,%20white (April 04, 2017).

Johns, G.M., Leeworthy, V.R., Bell, F.W. and M.A. Bonn. 2001. Socioeconomic study of reefs in southeast Florida – Final Report. Hazen and Sawyer Environmental Engineers and Scientists.

Johns, G.M., Milon, J.W. and D. Sayers. 2004. Socioeconomic study of reefs in Martin County, Florida. Final Report. Hazen and Sawyer Environmental Engineers and Scientists.

Johnson, A. G. (1983). Age and growth of yellowtail snapper from South Florida. Transactions of the American Fisheries Society, 112(2A), 173-177.

Johnson, D.R., Harper, D.E., Kellison, G.T. and J.A. Bohnsack. 2007. Description and discussion of southeast Florida fishery landings, 1990-2000. NOAA Technical Memorandum NMFS-SEFSC-550. 64 p.

Jones, G.P. and C. Syms. 1998. Disturbance, habitat structure and the ecology of fishes on coral reefs. Austral Ecology 23(3), 287-297.

Jordan, L. K., Gilliam, D.S., Sherman, R.L., Arena, P.T., Harttung, F.M., Baron, R.M. and R.E. Spieler. 2004. Spatial and Temporal Recruitment Patterns of Juvenile Grunts (*Haemulon* spp.) in South Florida. Oceanography Faculty Proceedings, Presentations, Speeches, Lectures. Paper 65

Jordan, L.K.B., Banks, K.W., Fisher, L.E., Walker, B.K. and D.S. Gilliam. 2009. Elevated sedimentation on coral reefs adjacent to a beach nourishment project. Marine Pollution Bulletin 60, 261-271.

Jory, D.F. and E.S. Iversen. 1989. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates (South Florida) – Black, Red, and

Nassau Groupers. U.S. Fish and Wildlife Service Biological Report 82(11.110). U.S. Army Corps of Engineers, TR EL-82-4. 21 p.

Kaschner, K., Rius-Barile, J., Kesner-Reyes, K., Garilao, C., Kullander, S.O., Rees, T. and R. Froese. 2013. AquaMaps: Predicted range maps for aquatic species. Retrieved from: www.aquamaps.org (August 12, 2017).

Kelly, A.M. 2014. Age, Growth, and Reproduction of Gray Triggerfish *Balistes capriscus* Off the Southeastern U.S. Atlantic Coast. SEDAR41-RD36. Master's Thesis, College of Charleston, Charleston, NC. 54 p.

Kelly-Stormer, A., Shervette, V., Kolmos, K., Wyanski, D., Smart, T., McDonough, C. and M.J.M. Reichert. 2017. Gray Triggerfish Reproductive Biology, Age, and Growth off the Atlantic Coast of the Southeastern USA. Transactions of the American Fisheries Society 146(3), 523-538.

Kilfoyle, A.K., Freeman, J., Jordan, L.K.B., T.P. Quinn and R.E. Spieler. 2013. Fish assemblages on a mitigation boulder reef and neighboring hardbottom. Ocean and Coastal Management 75: 53-62.

Kilfoyle, K., Walker, B.K., Fisco, D.P., Smith, S.G., and R.E. Spieler. 2015. Southeast Florida Coral Reef Fishery-Independent Baseline Assessment – 2012-2014 Summary Report. Florida Department of Environmental Protection. 129 pp.

Kingsley, M.C.S. MS 2004. The hogfish in Florida: Assessment Review and Advisory Report. Southeast Data and Assessment Review. Prepared for the South Atlantic Fishery Management Council, the Gulf of Mexico Fishery Management Council, and the National Marine Fisheries Service., 15 p. Retrieved from: http://www.sefsc.noaa.gov/sedar/download/SEDAR6_AR2_Hogfish.pdf?id=DOCUMENT (July 01, 2017).

Klitgord, K.D., Popenoe, P. and H. Schouten. 1984. Florida: A Jurassic transform plate boundary. Journal of Geophysical Research 89(B9), 7753-7772.

Klug, K. 2015. Cross-Shelf and Latitudinal Benthic Community Investigation in the Nearshore Habitats of the Northern Florida Reef Tract. Master's thesis. Nova Southeastern University. 116 p. Retrieved from: http://nsuworks.nova.edu/-occ_stuetd/386 (March 03, 2017).

Lapointe, B. 2007. Comparative Ecology of Harmful Macroalgal Blooms in South Florida's Coastal Waters (CEHAB). Ft. Pierce FL: Harbor Branch Oceanographic Institution. 64 p.

Lindeman, K., Anderson, W., Carpenter, K.E., Claro, R., Cowan, J., Padovani-Ferreira, B., Rocha, L.A., Sedberry, G. and M. Zapp-Sluis. 2016. Lutjanus analis. The IUCN Red List of Threatened Species 2016: e.T12416A506350.

Luckhurst, B.E. and K. Luckhurst. 1978. Analysis of the influence of substrate variables on coral reef fish communities. Marine Biology 49(4), 317-323.

Lugo, A.E., Brown, S.L., Dodson, R., Smith, T.S. and H.H. Shugart. 1999. The Holidridge life zones of the conterminous United States in relation to ecosystem mapping. Journal of Biogeography 26(5), 1025-1038.

Luo, J., Serafy, J.E., Sponaugle, S., Teare, P.B. and D. Kieckbusch. 2009. Movement of gray snapper Lutjanus griseus among subtropical seagrass, mangrove, and coral reef habitats. Marine Ecology Progress Series 380, 255-269.

MacKichan, C.A. and S.T. Szedlmayer. 2007. Reproductive Behavior of the Gray Triggerfish, *Balistes capriscus*, in the Northeastern Gulf of Mexico. Proceedings of the Gulf and Caribbean Fisheries Institute 59, 231-236.

Manooch, C.S. and C.L. Drennon. 1987. Age and growth of yellowtail snapper and queen triggerfish collected from the US Virgin Islands and Puerto Rico. Fisheries Research 6(1), 53-68.

Manooch, C.S., III and D.L. Mason. 1987. Age and growth of the warsaw grouper and black grouper from the southeast region of the United States. Northeast Gulf Science 9(2), 65-75.

Mason, D. L. and C.S. Manooch, III. 1985. Age and growth of mutton snapper along the east coast of Florida. Fisheries Research 3, 93-104.

McBride, R.S., Johnson, M., Bullock, L., and F. Stengard. 2001. Preliminary observations on the sexual development of hogfish, *Lachnolaimus maximus* (Pisces: Labridae). Proceedings of the Gulf and Caribbean Fisheries Institute 52, 98-102.

McBride, R.S. and M.R. Johnson. 2007. Sexual development and reproductive seasonality of hogfish (Labridae: *Lachnolaimus maximus*), an hermaphroditic reef fish. Journal of Fish Biology 71(5), 1270-1292.

McBride, R.S., Thurman, P.E. and L.H. Bullock. 2008. Regional variations of hogfish (*Lachnolaimus maximus*) life history: consequences for spawning biomass and egg production models. Journal of Northwest Atlantic Fisheries Science 41, 1-12.

McClanahan, T.R. 1994. Kenyan coral reef lagoon fish: effects of fishing, substrate complexity, and sea urchins. Coral Reefs 13(4), 231-241.

McClellan, D.B. and N.J. Cummings. Fishery and biology of the yellowtail snapper, *Ocyurus chrysurus*, from the southeastern United States, 1962 through 1996. Proceedings of the Gulf and Caribbean Fisheries Institute 50, 827-850.

McCoy, E.D. and S.S. Bell. 1991. Habitat structure: the evolution and diversification of a complex topic. In: Bell, S.S., McCoy, E.D. and H.R. Mushinsky (eds.) Habitat Structure. Population and Community Biology Series, Vol. 8. Springer, Dordrecht. 3-27.

Miller, G.C. and W.J. Richards. 1979. Reef fish habitat, faunal assemblages, and factors determining distributions in the South Atlantic Bight. Proceedings of the Gulf and Caribbean Fisheries Institute 32, 124-130.

Moe, M.A., Jr. 1969. Biology of the red grouper Epinephelus morio (Valenciennes) from the eastern Gulf of Mexico. Florida Department of Natural Resources Marine Research Laboratory Professional Paper 10, 1-95.

Morinière, E.C., Pollux, B.J.A., Nagelkerken, I. and G. van der Velde. 2002. Post-settlement Life Cycle Migration Patterns and Habitat Preference of Coral Reef Fish that use Seagrass and Mangrove Habitats as Nurseries. Estuarine, Coastal and Shelf Science 55(2), 309-321.

Morinière, E.C., Nagelkerken, I., van der Meij, H. and G. van der Velde. 2004. What attracts juvenile coral reef fish to mangroves: habitat complexity or shade? Marine Biology 144(1), 139-145.

Morris, J.A. and J.L. Akins. 2009. Feeding ecology of invasive lionfish (*Pterois volitans*) in the Bahamian archipelago. Environmental Biology of Fishes 86(3), 389-398.

Moyer, R.P., Riegl, B., Banks, K. and R.E. Dodge. 2003. Spatial patterns and ecology of benthic communities on a high-latitude South Florida (Broward County, USA) reef system. Coral Reefs 22(4), 447-464.

Mumby, P.J., Riegl, B., Banks, K. and R.E. Dodge. 2003. Spatial patterns and ecology of benthic communities on a high-latitude South Florida (Broward County, USA) reef system. Coral Reefs 22(4), 447-464.

Mumby, P.J., Edwards, A.J., Arias-González, J.E., Lindeman, K.C., Blackwell, P.G., Gall, A., Gorczynska, M.I., Harborne, A.R., Pescod, C.L., Renken, H. and C.C. Wabnitz. 2004. Mangroves enhance the biomass of coral reef fish communities in the Caribbean. Nature 427(6974), 533-536.

Munoz, R.C., Burton, M.L., Brennan, K.J. and R.O. Parker. 2010. Reproduction, habitat utilization, and movements of hogfish (*Lachnolaimus maximus*) in the Florida Keys, U.S.A.: comparisons from fished versus unfished habitats. Bulletin of Marine Science 86(1), 93-116.

Murie, D.J. and D.C. Parkyn. 2005. Age and Growth of White Grunt (*Haemulon plumieri*): A Comparison of Two Populations along the West Coast of Florida. Bulletin of Marine Science 76(1), 73-93.

Murphy, M.D., Murie, D.J. and R.G. Muller. 1999. Stock assessment of white grunt from the west coast of Florida. Report to the Florida Fish and Wildlife Conservation Commission. 56 p.

Nagelkerken, I., van der Velde, G., Gorissen, M.W., Meijer, G.J., van‘t Hof, T. and C den Hartog. 2000. Importance of mangroves, seagrass beds and the shallow coral reef as a nursery for important coral reef fishes, using a visual census technique. Estuarine Coastal Shelf Science 51, 31–44.

Nagelkerken, I., Kleijnen, S., Klop, T., van den Brand, R.A.C.J., Cochret de la Moriniere, E. and G. van der Velde. 2001. Dependence of Caribbean reef fishes on mangroves and seagrass beds as nursery habitats: a comparison of fish faunas between bays with and without mangroves/seagrass beds. Marine Ecology Progress Series 214, 225-235.

Newman, S.J. and D.M. Williams. 2001. Spatial and temporal variation in assemblages of Lutjanidae, Lethrinidae and associated fish species among mid-continental shelf reefs in the central Great Barrier Reef. Marine and Freshwater Research 52(6), 843-851.

NMFS (National Marine Fisheries Service). 1998. NMFS Fisheries Statistics and Economics Division. Commercial and recreational fisheries statistics surveys via the internet.

O'Hop, J., Murphy, M. and D. Chagaris. 2012. The 2012 Stock Assessment Report for Yellowtail Snapper in the South Atlantic and Gulf of Mexico. Fish and Wildlife Conservation Commission, St. Petersburg, Florida. 341 p.

Padgett, S.M., Harris, P.J. and D.M. Wyanski. 1997. Analytical report on the age, growth, and reproductive biology of the white grunt, *Haemulon plumieri*, along the southeast Atlantic coast of the United States. MARMAP Program Contract No. 52WCNF6006013PW, South Carolina Department of Natural Resources, Charleston, South Carolina. 61 p.

Parrish, J.D., Callahan, M.W. and J.E. Norris. 1985. Fish trophic relationships that structure reef communities. Proceedings of the 5th International Coral Reef Symposium, vol. 4, 73-78.

Pittman, S.J., Christensen, J.D., Caldow, C., Menza, C. and M.E. Monaco. 2007. Predictive mapping of fish species richness across shallow-water seascapes in the Caribbean. Ecological Modelling 204(1-2), 9-21.

Pittman, S.J., Costa, B.M. and T.A. Battista. 2009. Using Lidar Bathymetry and Boosted Regression Trees to Predict the Diversity and Abundance of Fish and Corals. Journal of Coastal Research 53, 27-38.

Potts, J.C. 2000. Population assessment of two stocks of white grunt, *Haemulon plumieri*, from the Southeastern coast of the United States. NOAA report prepared for the South Atlantic Fishery Management Council. 66 p.

Potts, J.C. and C.S. Manooch, III. 2001. Differences in the age and growth of White Grunt (*Haemulon plumierii*) from North Carolina and South Carolina compared with Southeast Florida. Bulletin of Marine Science 68(1), 1-12.

Precht, W.F., Aronson, R.B., Moody, R.M. and L. Kaufman. 2010. Changing Patterns of Microhabitat Utilization by the Threespot Damselfish, *Stegastes planifrons*, on Caribbean Reefs. PLoS ONE 5(5): e10835. Available at: https://doi.org/-10.1371/journal.pone.0010835.

Putman, R.J. and S.D. Wratten. 1984. Principles of ecology. Croom Helm, London. 388 p.

Ralston, S. 1987. Mortality rates of snappers and groupers. In: J.J. Polovina and S. Ralston (eds.) Tropical snappers and groupers: biology and fisheries management. Westview Press, Boulder, Colorado, 375-404.

Recksiek, C.W., Murphy, B.R., Appledoorn, R.S. and K.C. Lindeman. 2001. Integrating Fish Fuana and Habitat Assessments: A Fundamental Step in Developing Fishery Reserve Design Criteria. Proceedings of the 52nd Gulf and Caribbean Fisheries Institute, 654-666.

Riegl, B., Walker, B., G. Foster and K. Foster, 2005. Development of GIS maps for southeast Florida coral reefs. Florida Department of Environmental Protection, Miami Beach, Florida, p. 69.

Robins, C.R. and G.C. Ray. 1986. A field guide to Atlantic coast fishes of North America. Houghton Mifflin Company, Boston, U.S.A., 354 p.

Roe, R.B. 1976. Distribution of snappers and groupers in the Gulf of Mexico and Caribbean Sea as determined from exploratory fishing data. Florida Sea Grant Report 17, 129-164.

Rutherford, E.S., Thue, E.B. and D.G. Buker. 1983. Population structure, food habits, and spawning activity of gray snapper, *Lutjanus griseus*, in Everglades National Park. South Florida Research Center Report SFRC-83/02. Homestead, Florida, 40 p.

Sale, P.F. 1991. The Ecology of Fishes on Coral Reefs. Academic Press, San Diego, CA. 754 pp.

Sale, P.F. 1991. Habitat structure and recruitment in coral reef fishes. In: Bell, S.S., McCoy, E.D., and H.R. Mushinsky (eds.) Habitat structure: The physical arrangement of objects in space. Chapman and Hall. 197-210.

SEDAR. 2006. SEDAR 9 - Stock assessment report 1. Gulf of Mexico Gray Triggerfish. Charleston, SC. 35 pp. Retrieved from: http://sefsc.noaa.gov/sedar (March 25, 2017).

SEDAR. 2017. SEDAR 53 – South Atlantic Red Grouper Assessment Report. SEDAR, North Charleston SC. 159 pp. Retrieved from: http://sedarweb.org.sedar-53 (August 24, 2017).

Settle, L.R. 1993. Spatial and temporal variability in the distribution and abundance of larval and juvenile fishes associated with pelagic Sargassum. M.S. thesis, University of North Carolina Wilmington, Wilmington, NC. 64 p.

Sherman, B.R., Gilliam, D.S. and R.E. Spieler. 1999. A preliminary examination of depth associated spatial variation in fish assemblages on small artificial reefs. Journal of Applied Ichthyology 15(3), 116-121.

Shideler, G.S., Araújo, R.J., Walker, B.K., Blondeau, J. and J.E. Serafy. 2017. Non-linear thresholds characterize the relationship between reef fishes and mangrove habitat. Ecosphere 8(9).

Sluka, R.D., Chiappone, M. and K.M.S. Sealey. 2001. Influence of habitat on grouper abundance in the Florida Keys. U.S.A. Journal of Fish Biology 58, 682-700.

Smith, C.L. 1997. National Audubon Society field guide to tropical marine fishes of the Caribbean, the Gulf of Mexico, Florida, the Bahamas, and Bermuda. Alfred A. Knopf, Inc., New York. 720 p.

Smith, S.G., Ault, J.S., Bohnsack, J.A., Harper, D.E., Luo, J. and D.B. McClellan. 2011. Multispecies survey design for assessing reef-fish stocks, spatially-explicit management performance, and ecosystem condition. Fisheries Research 109, 25-41.

Stephens, J.S. and K.E. Zerba. 1981. Factors affecting fish diversity on a temperate reef. Pp. 111-121 *in* Noakes, D.L.G. and J.A. Ward (eds.) Ecology and ethology of fishes. Developments in environmental biology of fishes, vol. 1. Springer, Dordrecht.

Sumner, F.B., Osburn, R.C., and L.J. Cole. 1911. A biological survey of the waters of Woods Hole. Bulletin of the United States Bureau of Fisheries 31, 549-794.

Tarnecki, J.H., Wallace, A.A., Simons, J.D. and C.H. Ainsworth. 2016. Progression of a Gulf of Mexico food web supporting Atlantic ecosystem model development. Fisheries Research 179, 237-250.

Thompson, R. and J.L. Munro. 1983. The biology, ecology and bionomics of the snapper, Lutjanidae. pp. 94-109. In: Munro, J.L. (ed.) Caribbean coral reef fishery resources. ICLARM Student Review 7, 276 p.

Trejo-Martínez, J., Brulé, T., Mena-Loría, A., Colás-Marrufo, T. and M. Sánchez-Crespo. 2011. Reproductive aspects of the yellowtail snapper *Ocyurus chrysurus* from the southern Gulf of Mexico. Journal of Fish Biology 79(4), 915-936.

Tuya, F., Wernberg, T. and M.S. Thomsen. 2011. The relative influence of local to regional drivers of variation in reef fishes. Journal of Fish Biology 79(1), 217-234.

Vergara, R.R. 1977. *Lutjanus analis* (Cuvier, 1828). In: Fischer, W. (Ed.), FAO Species Identification Sheets for Fishery Purposes. Western Central Atlantic (Fishing Area 31), Vol. 3. Food and Agriculture Organization of the United Nations, Rome, p. LUT Lut 13.

Walker, B.K., Henderson, B. and R.E. Spieler. 2002. Fish assemblages associated with artificial reefs of concrete aggregates or quarry stone offshore Miami Beach, Florida, USA. Aquatic Living Resources 15(2), 95-105.

Walker, B.K., Riegl, B. and R.E. Dodge. 2008. Mapping coral reef habitats in southeast Florida using a combined technique approach. Journal of Coastal Research 24(5): 1138-1150.

Walker, B.K. 2009. Benthic habitat mapping of Miami-Dade County: Visual interpretation of LADS bathymetry and aerial photography. Florida DEP report # RM069, Miami Beach, Florida, p. 31.

Walker, B.K., Jordan, L.K.B., Spieler, R.E., 2009. Relationship of reef fish assemblages and topographic complexity on southeastern Florida coral reef habitats. Journal of Coastal Research 53, 39-48.

Walker, B.K. 2012. Spatial analyses of benthic habitats to define coral reef ecosystem regions and potential biogeographic boundaries along a latitudinal gradient. PLoS ONE 7: e30466. Available at: https://doi.org/10.1371/journal.pone.0030466.

Walker, B.K., Gilliam, D.S., Dodge, R.E. and J. Walczak. 2012. Dredging and shipping impacts on southeast Florida coral reefs. Proceedings of the 12th International Coral Reef Symposium, 19A Human Impacts on Coral Reefs: General Session, Cairns, Australia, 9-13 July 2012.

Walker, B.K. and D.S. Gilliam. 2013. Determining the extent and characterizing coral reef habitats of the northern latitudes of the Florida reef tract (Martin County). PLoS ONE 8(11): e80439. Available at: https://doi.org/10.1371/journal.-pone.0030466.

Walker, B.K. and K. Klug. 2014. Southeast Florida Shallow-Water Habitat Mapping and Coral Reef Community Characterization. Florida DEP Coral Reef Conservation Program report, 71 p.

Walker, B.K. and L. Gramer. Latitudinal controls of coral reef community distribution on the Florida Reef Tract. *In prep*.

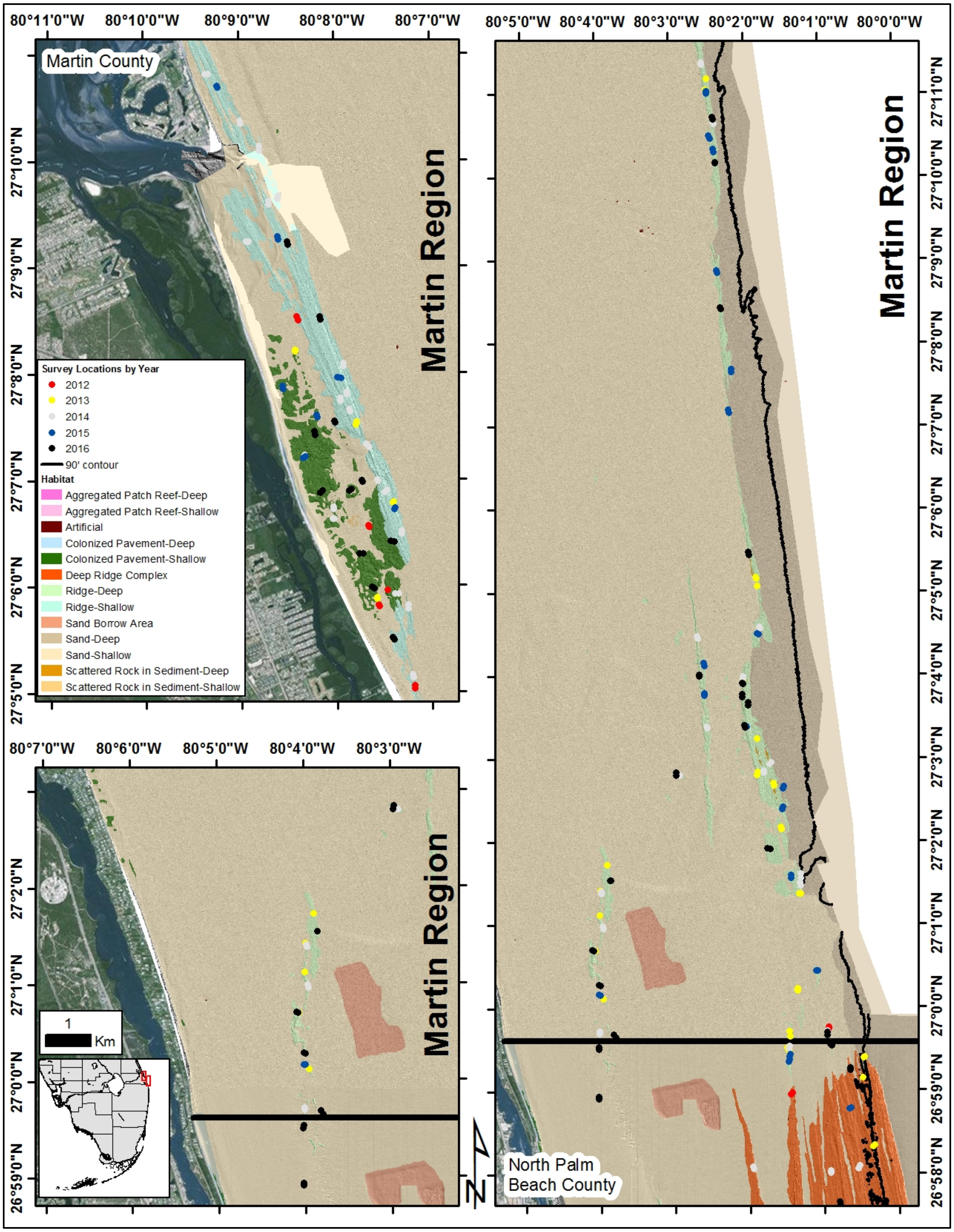
Watanabe, W.O., Ellis, S.C. and J. Chaves. 2001. Effects of dietary lipid and energy to protein ratio on growth and feed utilization of juvenile mutton snapper *Lutjanus analis* fed isonitrogenous diets at two temperatures. Journal of the World Aquaculture Society 32(1), 30-40.

Watanabe, W.O., Benetti, D.D., Feeley, M.W., Davis, D.A. and R.P. Phelps. 2005. Status of Artificial Propagation of Mutton, Yellowtail, and Red Snapper (family Lutjanidae) in the Southeastern United States. American Fisheries Society Symposium 46, 516-540.

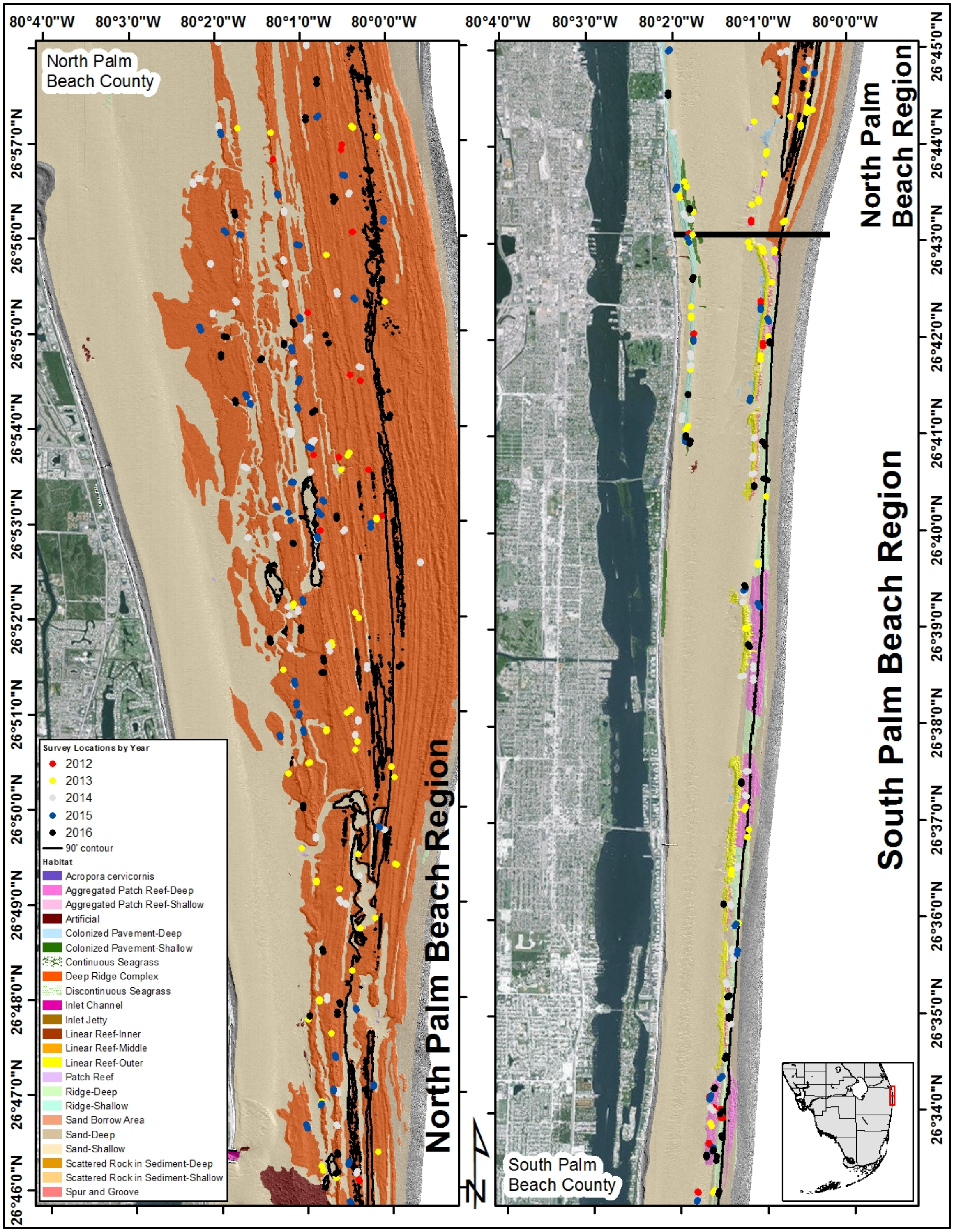
Wells, R.J. and J.R. Rooker. 2004. Spatial and temporal patterns of habitat use by fishes associated with Sargassum mats in the northwestern Gulf of Mexico. Bulletin of Marine Science 74(1), 81-99.

7. APPENDICES

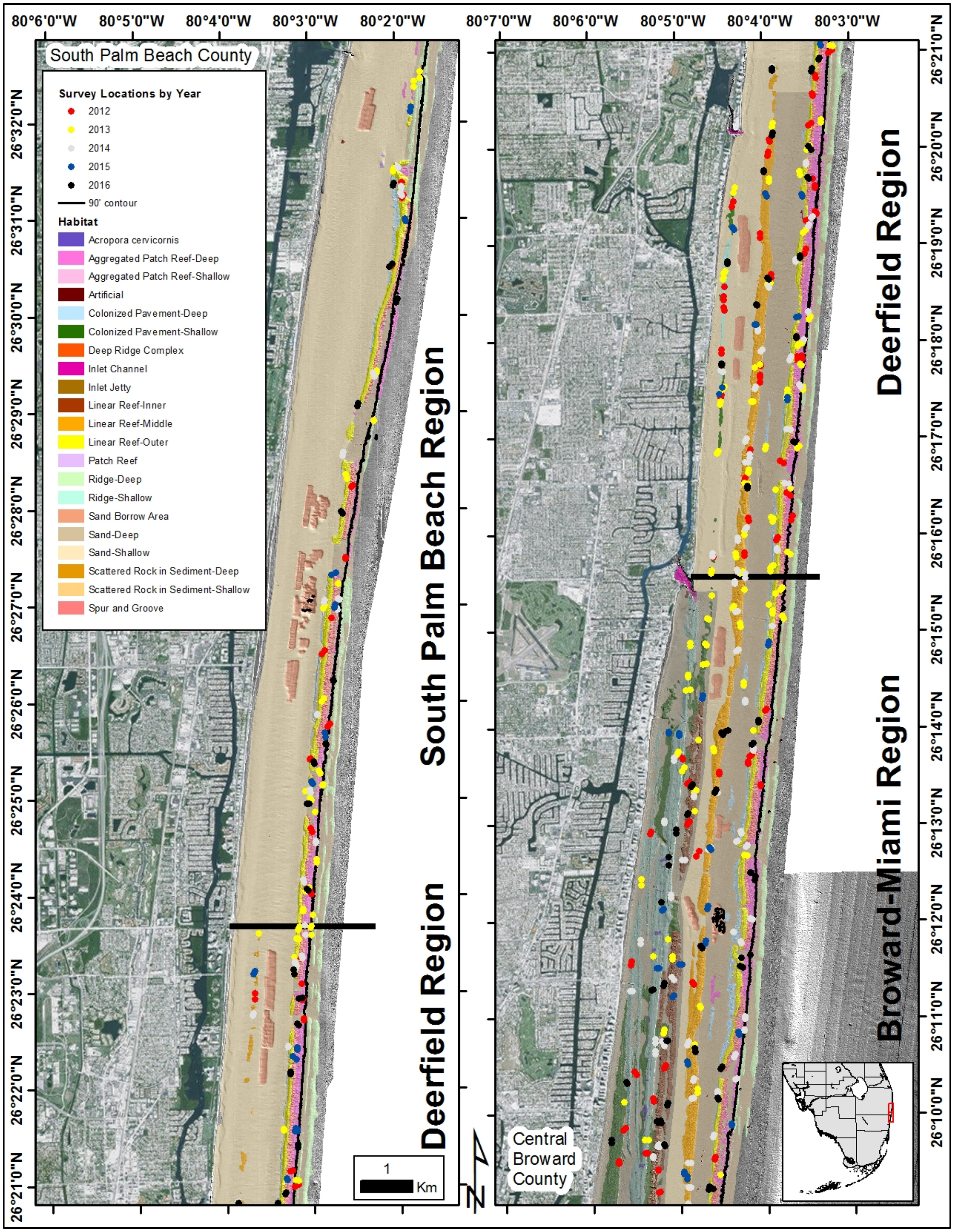
Appendix 1. Site Maps of actual survey locations from the combined 2012-2016 period.



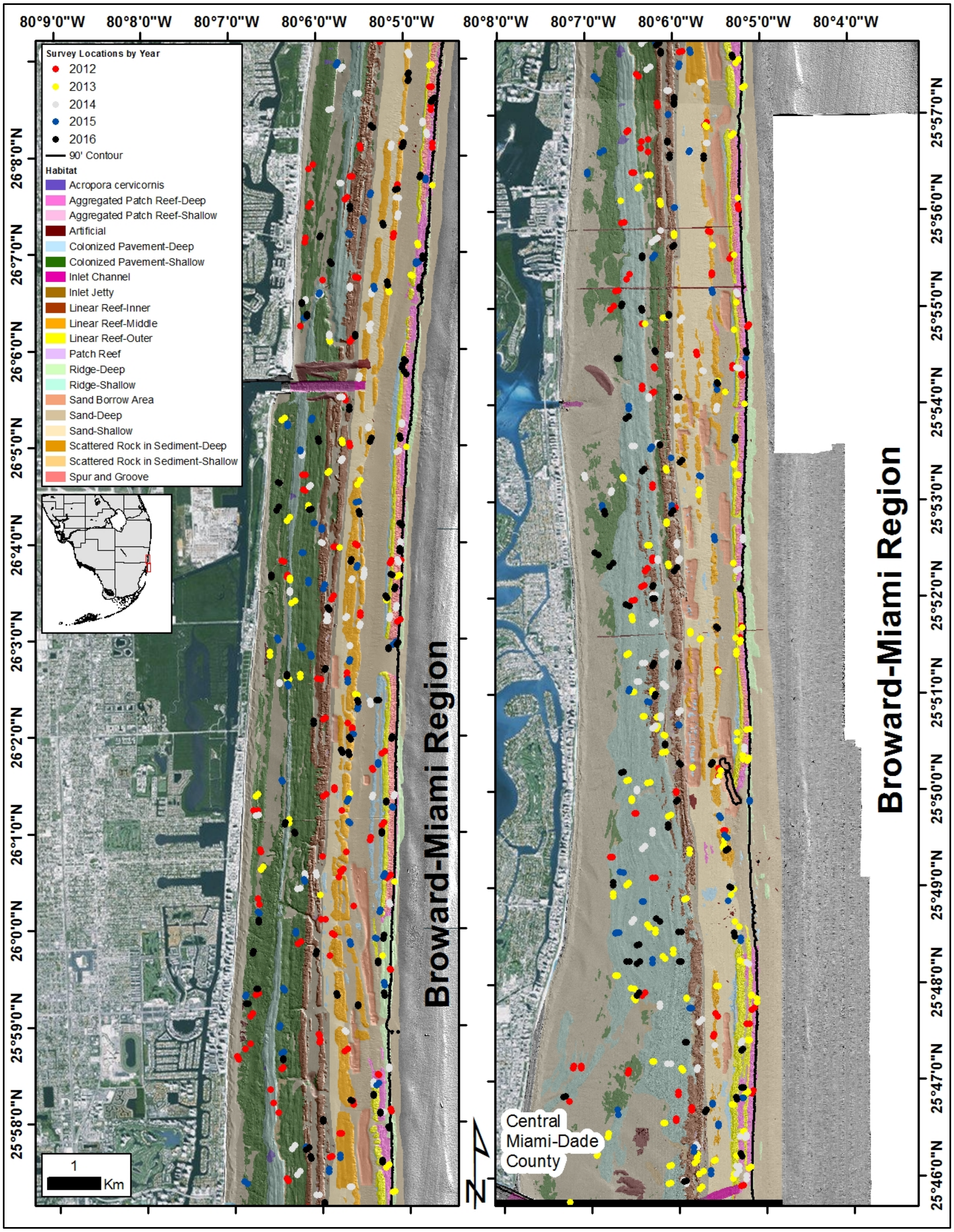
Appendix 1. (continued)



Appendix 1. (continued)



Appendix 1. (continued)


Survey Locations by Year
2012
2013
2014
2015
2016
90' Contour
Habitat
Acropora cervicornis
Aggregated Patch Reef-Deep
Aggregated Patch Reef-Shallow
Artificial
Colonized Pavement-Deep
Colonized Pavement-Shallow
Inlet Channel
Inlet Jetty
Linear Reef-Inner
Linear Reef-Middle
Linear Reef-Outer
Patch Reef
Ridge-Deep
Ridge-Shallow
Sand Borrow Area
Sand-Deep
Sand-Shallow
Scattered Rock in Sediment-Deep
Scattered Rock in Sediment-Shallow
Spur and Groove
Broward-Miami Region
Broward-Miami Region
Central Miami-Dade County
1
Km
N
80°9'0"W
80°8'0"W
80°7'0"W
80°6'0"W
80°5'0"W
80°8'0"W
80°7'0"W
80°6'0"W
80°5'0"W
80°4'0"W
26°8'0"N
26°7'0"N
26°6'0"N
26°5'0"N
26°4'0"N
26°3'0"N
26°2'0"N
26°1'0"N
26°0'0"N
25°59'0"N
25°58'0"N
25°57'0"N
25°56'0"N
25°55'0"N
25°54'0"N
25°53'0"N
25°52'0"N
25°51'0"N
25°50'0"N
25°49'0"N
25°48'0"N
25°47'0"N
25°46'0"N

Appendix 2. Average percent occurrence ($\bar{P}$) per SSU, average density ($\bar{D}$) per SSU, and survey precision (CV of $\bar{D}$) by species for the five-year period (2012-2016) for the SEFCRI region (5 annual surveys) as compared to data from 2012, 2014, and 2016 from the Florida Keys (3 annual surveys) and the Dry Tortugas (3 annual surveys). Species presented here had a mean percent occurrence ≥10% in one or more regions (94 species total). Species with values highlighted in pink were not observed with greater than 10% occurrence in the SEFCRI region. Species with values highlighted in gray were not observed with greater than 10% occurrence in the Florida Keys and Dry Tortugas.

Species	**Family**	**SEFCRI**			**FLORIDA KEYS**			**DRY TORTUGAS**		
EXPLOITED		***P***	***D***	**CV(*D*)**	***P***	***D***	**CV(*D*)**	***P***	***D***	**CV(*D*)**
***Gray Triggerfish** (*Balistes capriscus*)	Balistidae	40.9	1.46	14.6	8.2	0.13	25.9	0.5	0.00	83.9
Bar Jack (*Caranx ruber*)	Carangidae	30.1	1.21	27.6	39.1	3.97	18.4	20.6	3.30	36.7
Porkfish (*Anisotremus virginicus*)	Haemulidae	48.8	2.15	22.3	37.7	0.79	14.9	17.5	0.30	22.6
Tomtate (*Haemulon aurolineatum*)	Haemulidae	15.4	9.33	29.9	16.3	13.85	29.3	16.2	14.20	24.6
French Grunt (*Haemulon flavolineatum*)	Haemulidae	18.4	3.00	24.1	39.2	3.50	14.7	16.5	1.40	32.8
***White Grunt** (*Haemulon plumieri*)	Haemulidae	43.8	1.89	15.3	75.3	10.02	10.8	90.9	6.70	14.9
***Bluestriped Grunt** (*Haemulon sciurus*)	Haemulidae	18.0	0.98	42.7	32.8	2.69	16.2	9.6	0.30	38.6
***Hogfish** (*Lachnolaimus maximus*)	Labridae	23.4	0.30	10.9	70.3	1.69	7.3	45.7	0.50	12.8
***Mutton Snapper** (*Lutjanus analis*)	Lutjanidae	30.1	0.28	10.8	24.7	0.26	13.5	39.3	0.30	12.4
***Gray Snapper** (*Lutjanus griseus*)	Lutjanidae	12.1	0.70	34.9	23.9	1.65	18.5	19.0	1.80	33.6
***Yellowtail Snapper** (*Ocyurus chrysurus*)	Lutjanidae	27.9	1.04	24.7	64.6	4.87	14.3	80.5	8.80	9.3
Graysby (*Cephalopholis cruentata*)	Serranidae	24.2	0.21	11.9	42.1	0.59	8.5	33.7	0.30	8.8
***Red Grouper** (*Epinephelus morio*)	Serranidae	8.9	0.05	15.6	13.8	0.11	18.5	65.2	0.60	8.5
Black Grouper (*Mycteroperca bonaci*)	Serranidae	1.3	0.00	59.0	12.7	0.09	15.7	17.6	0.10	13.9
Great Barracuda (*Sphyraena barracuda*)	Sphyraenidae	2.2	0.02	50.5	6.8	0.06	21.4	10.1	0.00	18.0
NON-TARGET & AQUARIUM										
Ocean Surgeon (*Acanthurus bahianus*)	Acanthuridae	74.7	4.86	5.9	81.7	3.84	5.1	62.7	1.30	10.0
Doctorfish (*Acanthurus chirurgus*)	Acanthuridae	69.3	3.46	7.0	53.3	2.15	10.6	32.4	0.50	18.2
Blue Tang (*Acanthurus coeruleus*)	Acanthuridae	51.3	1.56	10.2	81.2	3.69	10.0	77.9	2.30	8.0

Appendix 2. (continued)

Species	Family	SEFCRI			FLORIDA KEYS			DRY TORTUGAS		
NON-TARGET & AQUARIUM		***P***	***D***	**CV(*D*)**	***P***	***D***	**CV(*D*)**	***P***	***D***	**CV(*D*)**
Atlantic Trumpetfish (*Aulostomus maculatus*)	Aulostomidae	8.0	0.00	68.5	15.0	0.00	93.2	3.8	0.00	45.1
Seaweed Blenny (*Parablennius marmoreus*)	Blenniidae	9.7	0.12	22.1	4.5	0.04	28.6	18.1	0.10	24.5
Blue Runner (*Caranx crysos*)	Carangidae	12.7	1.13	38.9	5.4	0.78	60.0	1.5	0.10	78.0
Foureye Butterflyfish (*Chaetodon capistratus*)	Chaetodontidae	12.9	0.16	13.1	45.1	1.00	7.8	31.3	0.40	9.5
Spotfin Butterflyfish (*Chaetodon ocellatus*)	Chaetodontidae	26.3	0.32	8.7	46.9	0.74	6.8	52.3	0.60	7.2
Reef Butterflyfish (*Chaetodon sedentarius*)	Chaetodontidae	52.1	0.98	6.3	32.5	0.56	9.0	18.8	0.10	17.9
Banded Butterflyfish (*Chaetodon striatus*)	Chaetodontidae	4.7	0.04	25.0	12.8	0.12	14.5	6.5	0.00	28.9
Bridled Goby (*Coryphopterus glaucofraenum*)	Gobiidae	26.1	0.51	14.1	57.2	1.96	8.7	77.8	4.50	11.8
Masked Goby (*Coryphopterus personatus*)	Gobiidae	26.1	15.43	17.8	40.4	31.43	15.3	56.4	70.20	14.9
Neon Goby (*Elacatinus oceanops*)	Gobiidae	11.4	0.16	19.6	20.2	0.39	14.6	40.2	0.80	11.7
Goldspotted Goby (*Gnatholepis thompsoni*)	Gobiidae	13.9	0.15	16.5	18.1	0.26	17.2	13.2	0.10	26.2
Black Margate (*Anisotremus surinamensis*)	Haemulidae	11.1	0.16	30.6	3.3	0.02	29.3	0.7	0.00	76.7
Squirrelfish (*Holocentrus adscensionis*)	Holocentridae	19.6	0.19	15.2	10.3	0.11	19.6	14.7	0.10	30.8
Longspine Squirrelfish (*Holocentrus rufus*)	Holocentridae	4.4	0.03	28.0	4.7	0.07	32.9	16.8	0.20	22.0
Spanish Hogfish (*Bodianus rufus*)	Labridae	32.6	0.37	12.1	28.2	0.29	10.0	18.7	0.10	11.4
Creole Wrasse (*Clepticus parrae*)	Labridae	14.7	5.06	30.3	15.2	3.45	20.4	9.0	2.20	23.3
Slippery Dick (*Halichoeres bivittatus*)	Labridae	61.4	3.69	7.7	72.7	7.65	6.2	83.3	14.00	10.4
Yellowcheek Wrasse (*Halichoeres cyanocephalus*)	Labridae	11.0	0.09	21.5	1.0	0.00	44.0	7.0	0.00	36.2
Yellowhead Wrasse (*Halichoeres garnoti*)	Labridae	65.3	5.01	7.6	75.0	4.77	5.2	86.4	5.30	6.8
Clown Wrasse (*Halichoeres maculipinna*)	Labridae	42.6	1.65	9.8	60.0	3.06	6.6	59.8	2.10	11.5
Blackear Wrasse (*Halichoeres poeyi*)	Labridae	12.4	0.13	20.7	15.5	0.24	15.7	10.5	0.10	37.3
Puddingwife (*Halichoeres radiatus*)	Labridae	8.3	0.06	26.3	28.2	0.33	11.6	17.0	0.10	18.8
Bluehead (*Thalasoma bifasciatum*)	Labridae	79.5	19.34	9.1	93.1	19.89	4.4	92.5	14.20	6.8
Green Razorfish (*Xyrichtys splendens*)	Labridae	22.1	0.87	22.0	11.1	0.46	24.8	11.2	0.20	50.2
Rosy Blenny (*Malacoctenus macropus*)	Labrisomidae	4.2	0.03	34.0	3.0	0.02	27.5	10.3	0.10	23.5

Appendix 2. (continued)

Species	Family	SEFCRI			FLORIDA KEYS			DRY TORTUGAS		
NON-TARGET & AQUARIUM		***P***	***D***	**CV(*D*)**	***P***	***D***	**CV(*D*)**	***P***	***D***	**CV(*D*)**
Saddled Blenny (*Malacoctenus triangulatus*)	Labrisomidae	9.4	0.06	15.8	6.1	0.05	21.3	45.8	0.60	11.9
Schoolmaster Snapper (*Lutjanus apodus*)	Lutjanidae	1.5	0.07	99.2	11.9	0.48	24.9	4.0	0.00	25.1
Blue Goby (*Ptereleotris calliura*)	Microdesmidae	8.1	0.15	27.1	10.4	0.26	22.0	26.9	0.90	25.5
Scrawled Filefish (*Aluterus scriptus*)	Monacanthidae	16.5	0.17	15.5	9.2	0.11	28.0	2.7	0.00	78.4
Orangespotted Filefish (*Cantherhines pullus*)	Monacanthidae	12.0	0.07	12.0	1.5	0.00	35.9	0.7	0.00	61.6
Spotted Goatfish (*Pseudupeneus maculatus*)	Mullidae	42.1	0.83	12.2	35.6	0.61	11.5	61.8	1.00	11.0
Yellowhead Jawfish (*Opistognathus aurifrons*)	Opistognathidae	12.7	0.21	19.1	11.7	0.25	24.0	57.4	3.70	14.2
Honeycomb Cowfish (*Acanthostracion polygonia*)	Ostraciidae	8.7	0.05	18.3	0.8	0.00	47.5	0.2	0.00	57.6
Scrawled Cowfish (*Acanthostracion quadricornius*)	Ostraciidae	10.4	0.07	18.1	4.0	0.02	22.6	2.2	0.00	42.5
Smooth Trunkfish (*Rhinesomus triqueter*)	Ostraciidae	10.5	0.07	15.3	10.6	0.06	15.1	1.6	0.00	39.7
Blue Angelfish (*Holacanthus bermudensis*)	Pomacanthidae	23.1	0.22	12.8	24.2	0.23	12.4	51.4	0.50	9.3
Queen Angelfish (*Holacanthus ciliaris*)	Pomacanthidae	22.5	0.19	11.7	38.2	0.40	9.0	28.5	0.20	12.9
Townsend Angelfish (*Holacanthus townsendi*)	Pomacanthidae	2.3	0.01	35.8	5.4	0.03	20.6	13.2	0.00	19.4
Rock Beauty (*Holacanthus tricolor*)	Pomacanthidae	35.7	0.51	7.5	35.7	0.61	8.6	8.1	0.00	18.2
Gray Angelfish (*Pomacanthus arcuatus*)	Pomacanthidae	44.4	0.51	7.3	61.3	1.03	7.7	39.3	0.40	9.8
French Angelfish (*Pomacanthus paru*)	Pomacanthidae	29.0	0.28	12.2	26.5	0.24	10.4	11.8	0.10	21.8
Sergent Major (*Abudefduf saxatilis*)	Pomacentridae	13.5	1.10	21.8	18.7	1.45	15.4	10.2	0.30	21.9
Blue Chromis (*Chromis cyanea*)	Pomacentridae	23.1	2.38	20.0	32.4	2.59	11.8	22.6	1.30	29.7
Yellowtail Reeffish (*Chromis enchrysura*)	Pomacentridae	18.1	0.60	20.7	1.1	0.06	75.6	17.8	0.30	25.7
Sunshinefish (*Chromis insolata*)	Pomacentridae	22.0	2.08	24.2	10.2	0.83	43.2	13.0	0.40	33.1
Brown Chromis (*Chromis multilineata*)	Pomacentridae	10.3	1.17	26.0	14.4	1.82	26.6	10.3	1.20	28.7
Purple Reeffish (*Chromis scotti*)	Pomacentridae	13.3	1.47	33.5	8.6	0.69	32.4	53.1	13.60	14.8
Dusky Damselfish (*Stegastes adustus*)	Pomacentridae	7.4	0.14	32.6	12.3	0.18	16.1	8.6	0.10	26.0
Beaugregory (*Stegastes leucostictus*)	Pomacentridae	24.2	0.44	14.4	26.4	0.29	11.1	29.4	0.50	12.7
Bicolor Damselfish (*Stegastes partitus*)	Pomacentridae	78.7	25.65	8.9	85.8	21.42	5.3	75.5	11.30	9.5

Appendix 2. (continued)

Species	Family	SEFCRI			FLORIDA KEYS			DRY TORTUGAS		
EXPLOITED		***P***	***D***	**CV(*D*)**	***P***	***D***	**CV(*D*)**	***P***	***D***	**CV(*D*)**
Threespot Damselfish (*Stegastes planifrons*)	Pomacentridae	4.2	0.05	34.5	21.6	0.41	14.1	29.0	0.80	12.0
Cocoa Damselfish (*Stegastes variabilis*)	Pomacentridae	41.0	0.87	11.6	59.2	1.39	7.2	96.4	7.40	6.1
Bluelip Parrotfish (*Cryptotomus roseus*)	Scaridae	24.6	0.82	13.7	10.3	0.16	19.5	19.5	0.30	24.3
Midnight Parrotfish (*Scarus coeruleus*)	Scaridae	2.1	0.01	37.2	20.5	0.36	16.6	6.4	0.00	34.5
Rainbow Parrotfish (*Scarus guacamaia*)	Scaridae	3.6	0.06	81.5	15.5	0.22	23.2	2.4	0.00	37.2
Striped Parrotfish (*Scarus iseri*)	Scaridae	35.5	2.02	10.6	83.0	9.90	6.7	91.0	12.50	5.4
Princess Parrotfish (*Scarus taeniopterus*)	Scaridae	26.9	0.82	11.4	26.4	0.77	15.0	14.7	0.40	21.9
Greenblotch Parrotfish (*Sparisoma atomarium*)	Scaridae	44.8	1.69	12.4	42.7	1.17	9.6	59.4	2.10	12.9
Redband Parrotfish (*Sparisoma aurofrenatum*)	Scaridae	64.4	4.42	7.2	93.1	6.49	4.3	87.3	3.30	6.3
Redtail Parrotfish (*Sparisoma chrysopterum*)	Scaridae	10.9	0.13	25.5	28.7	0.52	31.4	25.0	0.30	23.4
Bucktooth Parrotfish (*Sparisoma radians*)	Scaridae	10.9	0.16	19.6	11.8	0.19	17.9	10.9	0.10	26.0
Yellowtail Parrotfish (*Sparisoma rubripinne*)	Scaridae	11.0	0.13	20.8	24.9	0.44	15.3	11.8	0.10	28.5
Stoplight Parrotfish (*Sparisoma viride*)	Scaridae	32.9	0.57	12.3	63.4	1.95	6.9	61.1	1.30	9.2
High-hat (*Pareques acuminatus*)	Sciaenidae	9.8	0.15	27.9	12.3	0.14	19.1	2.5	0.00	30.9
Red Lionfish (*Pterois volitans*)	Scorpaenidae	12.5	0.12	17.2	7.9	0.08	20.6	4.5	0.00	27.7
Barred Hamlet (*Hypoplectrus puella*)	Serranidae	2.7	0.01	29.3	5.7	0.03	24.4	30.5	0.20	11.7
Butter Hamlet (*Hypoplectrus unicolor*)	Serranidae	19.7	0.20	16.5	37.2	0.44	8.6	53.4	0.70	6.5
Lantern Bass (*Serranus baldwini*)	Serranidae	16.5	0.13	19.3	10.0	0.07	18.5	14.2	0.10	25.5
Tobaccofish (*Serranus tabacarius*)	Serranidae	12.4	0.12	14.3	6.4	0.05	25.1	19.5	0.20	24.2
Harlequin Bass (*Serranus tigrinus*)	Serranidae	35.7	0.40	8.5	38.9	0.44	8.5	35.9	0.30	12.9
Chalk Bass (*Serranus tortugarum*)	Serranidae	7.0	0.36	62.3	2.2	0.04	81.3	5.7	0.10	57.6
Saucereye Porgy (*Calamus calamus*)	Sparidae	20.0	0.28	17.5	48.0	0.77	9.7	83.3	1.80	6.9
Sharpnose Puffer (*Canthigaster rostrata*)	Tetraodontidae	79.7	2.60	5.9	64.9	1.27	7.1	41.7	0.30	10.1
Bandtail Puffer (*Sphoeroides spengleri*)	Tetraodontidae	11.1	0.08	14.2	1.6	0.01	35.2	0.5	0.00	91.9

Appendix 3. Percent Occurrence ($\bar{P}$), Mean Density ($\bar{D}$), and Coefficient of Variation (CV) for all species recorded in the SEFCRI region with all five years combined, in alphabetical order by family.

Common Name	Scientific Name	Family	P	D	CV(D)
Doctorfish	*Acanthurus chirurgus*	Acanthuridae	0.69	3.47	7.0
Blue tang	*Acanthurus coeruleus*	Acanthuridae	0.51	1.56	10.2
Unidentified Surgeonfish species	*Acanthurus* spp.	Acanthuridae	0.03	0.10	50.0
Ocean surgeon	*Acanthurus bahianus*	Acanthuridae	0.75	4.87	6.0
Cardinalfish species	*Astrapogon* spp.	Apogonidae	0.006	0.006	102.2
Barred cardinalfish	*Apogon binotatus*	Apogonidae	0.008	0.005	49.0
Flamefish	*Apogon maculatus*	Apogonidae	0.01	0.01	68.6
Twospot cardinalfish	*Apogon pseudomaculatus*	Apogonidae	0.02	0.03	50.2
Sawcheek cardinalfish	*Apogon quadrisquamatus*	Apogonidae	0.0002	0.0001	201.6
Belted cardinalfish	*Apogon townsendi*	Apogonidae	0.004	0.002	62.7
Reef silverside	*Hypoatherina harringtonensis*	Atherinidae	0.0002	0.05	201.6
Silverside species	*Menidia* sp.	Atherinopsidae	0.0006	0.25	142.0
Trumpetfish	*Aulostomus maculatus*	Aulostomidae	0.08	0.08	23.6
Queen triggerfish	*Balistes vetula*	Balistidae	0.02	0.02	45.5
Ocean triggerfish	*Canthidermis sufflamen*	Balistidae	0.03	0.02	38.2
Black durgon	*Melichthys niger*	Balistidae	0.0004	0.0002	142.5
Gray triggerfish	*Balistes capriscus*	Balistidae	0.41	1.46	14.7
Oyster toadfish	*Opsanus tau*	Batrachoididae	0.001	0.0009	148.1
Barred blenny	*Hypleurochilus bermudensis*	Blenniidae	0.002	0.0008	115.9
Redlip blenny	*Ophioblennius macclurei*	Blenniidae	0.001	0.001	151.3
Seaweed blenny	*Parablennius marmoreus*	Blenniidae	0.10	0.12	22.1
Molly miller	*Scartella cristata*	Blenniidae	0.003	0.004	89.5
Blenny species	Blenny spp.	Blenniidae	0.01	0.01	70.6
Eyed flounder	*Bothus ocellatus*	Bothidae	0.0004	0.001	197.8
Peacock flounder	*Bothus lunatus*	Bothidae	0.002	0.0008	141.2
Black brotula	*Stygnobrotula latebricola*	Bythitidae	0.0004	0.0004	197.8
Lancer dragonet	*Callionymus bairdi*	Callionymidae	0.0003	0.0002	143.6
African pompano	*Alectis ciliaris*	Carangidae	0.003	0.02	133.1
Bar jack	*Caranx ruber*	Carangidae	0.30	1.22	27.7
Jack species	*Caranx* spp.	Carangidae	0.007	0.05	94.6
Blue runner	*Caranx crysos*	Carangidae	0.13	1.13	38.9
Crevalle jack	*Caranx hippos*	Carangidae	0.003	0.06	135.5
Horse-eye jack	*Caranx latus*	Carangidae	0.0003	0.002	180.4
Black jack	*Caranx lugubris*	Carangidae	0.0003	0.0002	200.7
Scad species	*Decapterus* spp.	Carangidae	0.004	0.05	92.6
Mackerel scad	*Decapterus macarellus*	Carangidae	0.008	0.31	70.4
Round scad	*Decapterus punctatus*	Carangidae	0.02	0.89	47.4

Appendix 3. (continued)

Common Name	**Scientific Name**	**Family**	***P***	***D***	***CV(D)***
Rainbow runner	*Elagatis bipinnulata*	Carangidae	0.02	0.24	66.5
Leatherjack	*Oligoplites saurus*	Carangidae	0.0005	0.0007	143.8
Lookdown	*Selene vomer*	Carangidae	0.0004	0.0002	200.2
Greater amberjack	*Seriola dumerili*	Carangidae	0.007	0.02	112.0
Almaco jack	*Seriola rivoliana*	Carangidae	0.04	0.06	42.6
Jack species	*Seriola* spp.	Carangidae	0.0003	0.0007	187.9
Banded rudderfish	*Seriola zonata*	Carangidae	0.005	0.006	80.9
Permit	*Trachinotus falcatus*	Carangidae	0.001	0.001	161.3
Palometa	*Trachinotus goodei*	Carangidae	0.0004	0.0002	200.2
Rough scad	*Trachurus lathami*	Carangidae	0.0004	0.004	134.2
Yellow jack	*Carangoides bartholomaei*	Carangidae	0.07	0.22	67.0
Atlantic bumper	*Chloroscombrus chrysurus*	Carangidae	0.006	0.39	95.3
Blacktip shark	*Carcharhinus limbatus*	Carcharhinidae	0.0003	0.0003	200.7
Caribbean reef shark	*Carcharhinus perezei*	Carcharhinidae	0.01	0.01	72.3
Sandbar shark	*Carcharhinus plumbeus*	Carcharhinidae	0.0004	0.0004	142.5
Tiger Shark	*Galeocerdo cuvier*	Carcharhinidae	0.0008	0.0004	200.2
Lemon shark	*Negaprion brevirostris*	Carcharhinidae	0.001	0.0005	156.9
Bull shark	*Carcharhinus leucas*	Carcharhinidae	0.003	0.001	81.7
Common snook	*Centropomus undecimalis*	Centropomidae	0.0005	0.01	194.5
Secretary blenny	*Acanthemblemaria maria*	Chaenopsidae	0.0005	0.0003	144.9
Glass blenny	*Emblemaria diaphana*	Chaenopsidae	0.0004	0.0008	167.8
Sailfin blenny	*Emblemaria pandionis*	Chaenopsidae	0.01	0.008	56.8
Wrasse blenny	*Hemiemblemaria simulus*	Chaenopsidae	0.0002	0.0002	201.6
Roughhead blenny	*Acanthemblemaria aspera*	Chaenopsidae	0.006	0.003	56.7
Spotfin butterflyfish	*Chaetodon ocellatus*	Chaetodontidae	0.26	0.33	8.8
Reef butterflyfish	*Chaetodon sedentarius*	Chaetodontidae	0.52	0.98	6.4
Banded butterflyfish	*Chaetodon striatus*	Chaetodontidae	0.05	0.05	25.0
Longsnout butterflyfish	*Prognathodes aculeatus*	Chaetodontidae	0.0004	0.0002	142.5
Foureye butterflyfish	*Chaetodon capistratus*	Chaetodontidae	0.13	0.17	13.2
Redspotted hawkfish	*Amblycirrhitus pinos*	Cirrhitidae	0.003	0.002	73.5
Spanish sardine	*Sardinella aurita*	Clupeidae	0.002	0.18	118.7
Herring species	*Jenkinsia* spp.	Clupeidae	0.003	4.58	192.6
Brown garden eel	*Heteroconger longissimus*	Congridae	0.004	0.07	102.0
Flying gurnard	*Dactylopterus volitans*	Dactylopteridae	0.001	0.0005	119.7
Southern stingray	*Dasyatis americana*	Dasyatidae	0.006	0.003	66.6
Spotted burrfish	*Chilomycterus atinga*	Diodontidae	0.001	0.0006	91.6
Spotfin burrfish	*Chilomycterus reticulatus*	Diodontidae	0.0008	0.0004	118.4
Striped burrfish	*Chilomycterus schoepfii*	Diodontidae	0.002	0.002	102.8
Puffer species	*Diodon* spp.	Diodontidae	0.001	0.001	160.1

Appendix 3. (continued)

Common Name	Scientific Name	Family	*P*	*D*	*CV(D)*
Balloonfish	*Diodon holocanthus*	Diodontidae	0.07	0.04	14.7
Bridled burrfish	*Chilomycterus antennatus*	Diodontidae	0.0004	0.0002	197.8
Sharksucker	*Echeneis naucrates*	Echeneidae	0.02	0.02	42.1
Whitefin sharksucker	*Echeneis neucratoides*	Echeneidae	0.004	0.003	80.8
Remora	*Remora remora*	Echeneidae	0.006	0.004	70.9
Shark species	*Elasmobranch* spp.	Elasmobranchiomorphi	0.001	0.0007	142.0
Anchovy species	*Anchoa* spp.	Engraulidae	0.0003	0.0002	200.7
Atlantic spadefish	*Chaetodipterus faber*	Ephippidae	0.05	0.26	37.7
Cornetfish	*Fistularia tabacaria*	Fistulariidae	0.02	0.01	40.2
Silver mojarra	*Eucinostomus argenteus*	Gerreidae	0.0003	0.0005	200.7
Slender mojarra	*Eucinostomus jonesi*	Gerreidae	0.0005	0.002	142.6
Unidentified mojarra	*Gerreidae spp.*	Gerreidae	0.0003	0.0004	175.5
Mottled mojarra	*Ulaema lefroyi*	Gerreidae	0.0001	0.0003	202.7
Yellow fin mojarra	*Gerres cinereus*	Gerreidae	0.01	0.11	178.6
Nurse shark	*Ginglymostoma cirratum*	Ginglymostomatidae	0.02	0.01	32.4
White-eye goby	*Bollmania boqueronensis*	Gobiidae	0.0005	0.0003	144.9
Bridled goby	*Coryphopterus glaucofraenum*	Gobiidae	0.26	0.52	14.1
Masked goby	*Coryphopterus personatus*	Gobiidae	0.26	15.43	17.9
Goby species	*Coryphopterus* spp.	Gobiidae	0.02	0.02	51.9
Pallid goby	*Coryphopterus eidolon*	Gobiidae	0.005	0.003	73.6
Peppermint goby	*Coryphopterus lipernes*	Gobiidae	0.001	0.0007	142.1
Dash goby	*Ctenogobius saepepallens*	Gobiidae	0.004	0.003	72.3
Neon goby	*Elacatinus oceanops*	Gobiidae	0.11	0.16	19.6
Yellownose goby	*Elacatinus randalli*	Gobiidae	0.0008	0.0004	200.2
Yellowline goby	*Elacatinus horsti*	Gobiidae	0.002	0.001	90.5
Yellowprow goby	*Elacatinus xanthiprora*	Gobiidae	0.002	0.002	80.6
Sharknose goby	*Elecatinus evelynae*	Gobiidae	0.003	0.004	76.6
Goldspot goby	*Gnatholepis thompsoni*	Gobiidae	0.14	0.16	16.5
Goby species	*Gobiidae* spp.	Gobiidae	0.005	0.005	67.2
Seminole goby	*Microgobius carri*	Gobiidae	0.002	0.001	108.2
Banner goby	*Microgobius microlepis*	Gobiidae	0.0003	0.0006	200.7
Orangespotted goby	*Nes longus*	Gobiidae	0.0003	0.0002	200.7
Rusty goby	*Priolepis hipoliti*	Gobiidae	0.0008	0.0004	200.2
Colon goby	*Coryphopterus dicrus*	Gobiidae	0.05	0.04	35.5
Porkfish	*Anisotremus virginicus*	Haemulidae	0.49	2.16	22.4
Black margate	*Anisotremus surinamensis*	Haemulidae	0.11	0.17	30.6
Tomtate	*Haemulon aurolineatum*	Haemulidae	0.15	9.34	30.0
French grunt	*Haemulon flavolineatum*	Haemulidae	0.18	3.00	24.1
White grunt	*Haemulon plumieri*	Haemulidae	0.44	1.90	15.4

Appendix 3. (continued)

Common Name	Scientific Name	Family	*P*	*D*	*CV(D)*
Bluestriped grunt	*Haemulon sciurus*	Haemulidae	0.18	0.99	42.8
Grunt species	*Haemulon* spp.	Haemulidae	0.23	5.89	24.1
White margate	*Haemulon album*	Haemulidae	0.02	0.02	50.3
Caesar grunt	*Haemulon carbonarium*	Haemulidae	0.06	0.29	43.6
Smallmouth grunt	*Haemulon chrysargyreum*	Haemulidae	0.02	0.31	63.7
Spanish grunt	*Haemulon macrostomum*	Haemulidae	0.03	0.03	45.9
Cottonwick	*Haemulon melanurum*	Haemulidae	0.06	1.16	64.1
Sailor's choice	*Haemulon parra*	Haemulidae	0.07	0.20	48.7
Striped grunt	*Haemulon striatum*	Haemulidae	0.02	0.52	63.4
Boga	*Haemulon vittatum*	Haemulidae	0.002	0.12	170.6
Pigfish	*Orthopristis chrysoptera*	Haemulidae	0.002	0.002	104.1
Ballyhoo	*Hemiramphus brasiliensis*	Hemiramphidae	0.003	0.17	117.4
Squirrelfish species	*Holocentrus* spp.	Holocentridae	0.005	0.004	101.3
Longspine squirrelfish	*Holocentrus rufus*	Holocentridae	0.04	0.04	28.1
Blackbar soldierfish	*Myripristis jacobus*	Holocentridae	0.04	0.09	45.1
Reef squirrelfish	*Sargocentron coruscum*	Holocentridae	0.005	0.003	104.4
Dusky squirrelfish	*Sargocentron vexillarium*	Holocentridae	0.0005	0.0003	152.1
Squirrelfish	*Holocentrus adscensionis*	Holocentridae	0.20	0.20	15.3
Sailfish	*Istiophorus platypterus*	Istiophoridae	0.0008	0.0004	200.2
Bermuda Chub	*Kyphosus sectatrix*	Kyphosidae	0.10	0.49	45.7
Spanish hogfish	*Bodianus rufus*	Labridae	0.33	0.38	12.1
Creole wrasse	*Clepticus parrae*	Labridae	0.15	5.06	30.4
Slippery dick	*Halichoeres bivittatus*	Labridae	0.61	3.70	7.8
Yellowcheek wrasse	*Halichoeres cyanocephalus*	Labridae	0.11	0.10	21.5
Yellowhead wrasse	*Halichoeres garnoti*	Labridae	0.65	5.01	7.6
Clown wrasse	*Halichoeres maculipinna*	Labridae	0.43	1.65	9.8
Unidentified wrasse	*Halichoeres spp.*	Labridae	0.007	0.03	87.8
Painted wrasse	*Halichoeres caudalis*	Labridae	0.001	0.001	112.8
Rainbow wrasse	*Halichoeres pictus*	Labridae	0.009	0.009	95.3
Blackear wrasse	*Halichoeres poeyi*	Labridae	0.12	0.14	20.8
Puddingwife	*Halichoeres radiatus*	Labridae	0.08	0.06	26.3
Wrasse species	*Labridae* spp.	Labridae	0.005	0.01	128.6
Hogfish	*Lachnolaimus maximus*	Labridae	0.23	0.30	10.9
Bluehead wrasse	*Thalassoma bifasciatum*	Labridae	0.80	19.3	9.2
Rosy razorfish	*Xyrichtys martinicensis*	Labridae	0.03	0.13	60.3
Pearly razorfish	*Xyrichtys novacula*	Labridae	0.01	0.01	55.8
Green razorfish	*Xyrichtys splendens*	Labridae	0.22	0.88	22.0
Razorfish species	*Xyrichtys* spp.	Labridae	0.006	0.008	81.7
Spotfin hogfish	*Bodianus pulchellus*	Labridae	0.05	0.04	27.0

Appendix 3. (continued)

Common Name	Scientific Name	Family	*P*	*D*	*CV(D)*
Hairy blenny	*Labrisomus nuchipinnis*	Labrisomidae	0.02	0.02	39.0
Goldline blenny	*Malacoctenus aurolineatus*	Labrisomidae	0.0004	0.002	197.8
Dusky blenny	*Malacoctenus gilli*	Labrisomidae	0.001	0.004	177.5
Rosy blenny	*Malacoctenus macropus*	Labrisomidae	0.04	0.04	34.1
Saddled blenny	*Malacoctenus triangulatus*	Labrisomidae	0.09	0.07	15.8
Marbled blenny	*Paraclinus marmoratus*	Labrisomidae	0.001	0.002	184.7
Downy blenny	*Labrisomus kalisherae*	Labrisomidae	0.0001	0.00004	202.7
Schoolmaster snapper	*Lutjanus apodus*	Lutjanidae	0.02	0.07	99.3
Blackfin snapper	*Lutjanus buccanella*	Lutjanidae	0.002	0.002	81.6
Red snapper	*Lutjanus campechanus*	Lutjanidae	0.00001	0.0001	166.6
Cubera snapper	*Lutjanus cyanopterus*	Lutjanidae	0.0009	0.0006	184.2
Gray snapper	*Lutjanus griseus*	Lutjanidae	0.12	0.70	35.0
Dog snapper	*Lutjanus jocu*	Lutjanidae	0.005	0.004	87.8
Mahogany snapper	*Lutjanus mahogoni*	Lutjanidae	0.01	0.02	101.5
Snapper species	*Lutjanus* spp.	Lutjanidae	0.003	0.002	64.5
Lane snapper	*Lutjanus synagris*	Lutjanidae	0.08	0.75	61.0
Yellowtail snapper	*Ocyurus chrysurus*	Lutjanidae	0.28	1.05	24.8
Vermilion snapper	*Rhomboplites aurorubens*	Lutjanidae	0.009	0.04	83.0
Mutton snapper	*Lutjanus analis*	Lutjanidae	0.30	0.28	10.8
Sand tilefish	*Malacanthus plumieri*	Malacanthidae	0.07	0.05	19.9
Tarpon	*Megalops atlanticus*	Megalopidae	0.003	0.002	81.6
Giant manta	*Manta birostris*	Mobulidae	0.0003	0.0003	200.7
Scrawled filefish	*Aluterus scriptus*	Monacanthidae	0.17	0.17	15.5
Filefish species	*Aluterus* spp.	Monacanthidae	0.003	0.004	86.6
Orangespotted filefish	*Cantherhines pullus*	Monacanthidae	0.12	0.08	12.1
Whitespotted filefish	*Cantherhines macrocerus*	Monacanthidae	0.03	0.02	30.2
Fringed filefish	*Monacanthus ciliatus*	Monacanthidae	0.006	0.003	73.3
Slender filefish	*Monacanthus tuckeri*	Monacanthidae	0.04	0.03	25.7
Planehead filefish	*Stephanolepis hispidus*	Monacanthidae	0.04	0.03	21.3
Unicorn filefish	*Aluterus monoceros*	Monacanthidae	0.02	0.05	50.6
Orange filefish	*Aluterus schoepfii*	Monacanthidae	0.02	0.02	57.3
Spotted goatfish	*Pseudupeneus maculatus*	Mullidae	0.42	0.84	12.2
Dwarf goatfish	*Upeneus parvus*	Mullidae	0.0008	0.002	200.2
Yellow goatfish	*Mulloidichthys martinicus*	Mullidae	0.007	0.02	90.2
Chestnut moray	*Enchelycore carychroa*	Muraenidae	0.0008	0.0004	200.2
Green moray	*Gymnothorax funebris*	Muraenidae	0.01	0.008	39.4
Goldentail moray	*Gymnothorax miliaris*	Muraenidae	0.01	0.008	36.4
Spotted moray	*Gymnothorax moringa*	Muraenidae	0.05	0.02	21.8
Purplemouth moray	*Gymnothorax vicinus*	Muraenidae	0.009	0.004	48.7

Appendix 3. (continued)

Common Name	Scientific Name	Family	*P*	*D*	*CV(D)*
Viper moray	*Enchelycore nigricans*	Muraenidae	0.0006	0.0003	147.9
Spotted eagle ray	*Aetobatus narinari*	Myliobatidae	0.004	0.003	65.5
Lesser electric ray	*Narcine bancroftii*	Narcinidae	0.001	0.0008	120.6
Shortnose batfish	*Ogcocephalus nasutus*	Ogcocephalidae	0.001	0.01	190.7
Batfish species	*Ogcocephalus* spp.	Ogcocephalidae	0.001	0.0006	152.0
Sharptail eel	*Myrichthys breviceps*	Ophichthidae	0.01	0.006	55.4
Goldspotted eel	*Myrichthys ocellatus*	Ophichthidae	0.001	0.0006	105.1
Yellowhead jawfish	*Opistognathus aurifrons*	Opistognathidae	0.13	0.22	19.2
Banded jawfish	*Opistognathus macrognathus*	Opistognathidae	0.001	0.0007	110.6
Jawfish species	*Opistognathus* spp.	Opistognathidae	0.003	0.002	96.0
Dusky jawfish	*Opistognathus whitehursti*	Opistognathidae	0.005	0.003	55.6
Scrawled cowfish	*Acanthostracion quadricornis*	Ostraciidae	0.10	0.07	18.2
Honeycomb cowfish	*Acanthostracion polygonius*	Ostraciidae	0.09	0.05	18.4
Spotted trunkfish	*Lactophrys bicaudalis*	Ostraciidae	0.01	0.008	41.3
Trunkfish	*Lactophrys trigonus*	Ostraciidae	0.009	0.005	47.6
Smooth trunkfish	*Rhinesomus triqueter*	Ostraciidae	0.11	0.07	15.3
Gulf flounder	*Paralichthys albigutta*	Paralichthyidae	0.0003	0.0002	200.7
Glassy sweeper	*Pempheris schomburgkii*	Pempheridae	0.005	0.20	164.8
Blue angelfish	*Holacanthus bermudensis*	Pomacanthidae	0.23	0.23	12.8
Queen angelfish	*Holacanthus ciliaris*	Pomacanthidae	0.23	0.20	11.8
Townsend angelfish	*Holacanthus townsendi*	Pomacanthidae	0.02	0.01	35.8
Rock beauty	*Holacanthus tricolor*	Pomacanthidae	0.36	0.51	7.5
Gray angelfish	*Pomacanthus arcuatus*	Pomacanthidae	0.44	0.51	7.4
French angelfish	*Pomacanthus paru*	Pomacanthidae	0.29	0.29	12.3
Cherubfish	*Centropyge argi*	Pomacanthidae	0.08	0.15	24.7
Sergeant major	*Abudefduf saxatilis*	Pomacentridae	0.14	1.10	21.9
Blue chromis	*Chromis cyanea*	Pomacentridae	0.23	2.39	20.0
Yellowtail reeffish	*Chromis enchrysura*	Pomacentridae	0.18	0.60	20.8
Sunshinefish	*Chromis insolata*	Pomacentridae	0.22	2.08	24.3
Brown chromis	*Chromis multilineata*	Pomacentridae	0.10	1.18	26.1
Purple reeffish	*Chromis scotti*	Pomacentridae	0.13	1.47	33.5
Yellowtail damselfish	*Microspathodon chrysurus*	Pomacentridae	0.02	0.04	79.8
Dusky damselfish	*Stegastes adustus*	Pomacentridae	0.07	0.15	32.7
Longfin damselfish	*Stegastes diencaeus*	Pomacentridae	0.01	0.02	50.4
Beaugregory	*Stegastes leucostictus*	Pomacentridae	0.24	0.44	14.4
Bicolor damselfish	*Stegastes partitus*	Pomacentridae	0.79	25.66	9.0
Threespot damselfish	*Stegastes planifrons*	Pomacentridae	0.04	0.05	34.6
Damselfish species	*Stegastes* spp.	Pomacentridae	0.009	0.05	87.1
Cocoa damselfish	*Stegastes variabilis*	Pomacentridae	0.41	0.87	11.6

Appendix 3. (continued)

Common Name	Scientific Name	Family	*P*	*D*	*CV(D)*
Glasseye snapper	*Heteropriacanthus cruentatus*	Priacanthidae	0.02	0.04	98.6
Bigeye	*Priacanthus arenatus*	Priacanthidae	0.02	0.03	63.4
Blue dartfish	*Ptereleotris calliura*	Ptereleotridae	0.08	0.16	27.1
Hovering dartfish	*Ptereleotris helenae*	Ptereleotridae	0.04	0.08	31.5
Cobia	*Rachycentron canadum*	Rachycentridae	0.002	0.0009	112.6
Atlantic guitarfish	*Rhinobatos lentiginosus*	Rhinobatidae	0.002	0.001	112.2
Emerald parrotfish	*Nicholsina usta*	Scaridae	0.005	0.01	100.4
Midnight parrotfish	*Scarus coelestinus*	Scaridae	0.006	0.009	107.4
Blue parrotfish	*Scarus coeruleus*	Scaridae	0.02	0.02	37.3
Rainbow parrotfish	*Scarus guacamaia*	Scaridae	0.04	0.06	81.6
Striped parrotfish	*Scarus iseri*	Scaridae	0.36	2.03	10.6
Parrotfish species	*Scarus* spp.	Scaridae	0.03	0.05	42.1
Princess parrotfish	*Scarus taeniopterus*	Scaridae	0.27	0.82	11.5
Queen parrotfish	*Scarus vetula*	Scaridae	0.03	0.02	30.5
Greenblotch parrotfish	*Sparisoma atomarium*	Scaridae	0.45	1.69	12.5
Redband parrotfish	*Sparisoma aurofrenatum*	Scaridae	0.64	4.43	7.3
Redtail parrotfish	*Sparisoma chrysopterum*	Scaridae	0.11	0.13	25.5
Bucktooth parrotfish	*Sparisoma radians*	Scaridae	0.11	0.17	19.6
Yellowtail parrotfish	*Sparisoma rubripinne*	Scaridae	0.11	0.14	20.8
Parrotfish species	*Sparisoma* spp.	Scaridae	0.004	0.008	98.7
Stoplight Parrotfish	*Sparisoma viride*	Scaridae	0.33	0.58	12.3
Bluelip parrotfish	*Cryptotomus roseus*	Scaridae	0.25	0.83	13.7
Spotted drum	*Equetus punctatus*	Sciaenidae	0.02	0.01	39.4
Reef croaker	*Odontoscion dentex*	Sciaenidae	0.01	0.06	74.1
High-hat	*Pareques acuminatus*	Sciaenidae	0.10	0.16	27.9
Cubbyu	*Pareques umbrosus*	Sciaenidae	0.01	0.05	56.2
Drum species	*Sciaenidae* spp.	Sciaenidae	0.002	0.002	103.8
Jackknife fish	*Equetus lanceolatus*	Sciaenidae	0.007	0.007	71.8
Little tunny	*Euthynnus alletteratus*	Scombridae	0.01	0.03	83.7
King mackerel	*Scomberomorus cavalla*	Scombridae	0.002	0.001	83.2
Spanish mackerel	*Scomberomorus maculatus*	Scombridae	0.01	0.03	139.6
Cero	*Scomberomorus regalis*	Scombridae	0.04	0.04	27.6
Lionfish	*Pterois* spp.	Scorpaenidae	0.13	0.12	17.2
Spotted scorpionfish	*Scorpaena plumieri*	Scorpaenidae	0.06	0.04	19.1
Black seabass	*Centropristis striata*	Serranidae	0.02	0.08	74.5
Graysby	*Cephalopholis cruentata*	Serranidae	0.24	0.22	11.9
Coney	*Cephalopholis fulva*	Serranidae	0.04	0.03	29.8
Sand perch	*Diplectrum formosum*	Serranidae	0.04	0.06	43.1
Red grouper	*Epinephelus morio*	Serranidae	0.09	0.06	15.7

Appendix 3. (continued)

Common Name	Scientific Name	Family	*P*	*D*	*CV(D)*
Rock hind	*Epinephelus adscensionis*	Serranidae	0.01	0.007	37.3
Red hind	*Epinephelus guttatus*	Serranidae	0.01	0.009	76.8
Goliath grouper	*Epinephelus itajara*	Serranidae	0.009	0.01	72.3
Black hamlet	*Hypoplectrus nigricans*	Serranidae	0.001	0.0005	102.4
Tan hamlet	*Hypoplectrus randallorum*	Serranidae	0.002	0.001	89.8
Hamlet species	*Hypoplectrus* spp.	Serranidae	0.006	0.003	69.4
Butter hamlet	*Hypoplectrus unicolor*	Serranidae	0.20	0.21	16.6
Blue hamlet	*Hypoplectrus gemma*	Serranidae	0.02	0.01	38.2
Shy hamlet	*Hypoplectrus guttavarius*	Serranidae	0.0007	0.0004	118.4
Indigo hamlet	*Hypoplectrus indigo*	Serranidae	0.001	0.0005	102.4
Barred hamlet	*Hypoplectrus puella*	Serranidae	0.03	0.02	29.3
Wrasse bass	*Liopropoma eukrines*	Serranidae	0.00005	0.00002	202.9
Peppermint basslet	*Liopropoma rubre*	Serranidae	0.0004	0.0006	197.8
Black grouper	*Mycteroperca bonaci*	Serranidae	0.01	0.009	59.1
Gag	*Mycteroperca microlepis*	Serranidae	0.008	0.005	72.8
Scamp	*Mycteroperca phenax*	Serranidae	0.01	0.008	52.5
Atlantic creolefish	*Paranthias furcifer*	Serranidae	0.001	0.006	168.6
Freckled soapfish	*Rypticus bistrispinus*	Serranidae	0.03	0.0006	140.2
Whitespotted soapfish	*Rypticus maculatus*	Serranidae	0.02	0.02	38.1
Greater soapfish	*Rypticus saponaceus*	Serranidae	0.03	0.02	26.8
School bass	*Schultzea beta*	Serranidae	0.01	0.24	114.5
Grouper-sea bass species	*Serranidae spp.*	Serranidae	0.0009	0.0005	143.8
Orangeback bass	*Serranus annularis*	Serranidae	0.02	0.01	54.0
Lantern bass	*Serranus baldwini*	Serranidae	0.17	0.13	19.3
Tattler	*Serranus phoebe*	Serranidae	0.004	0.002	97.1
Belted sandfish	*Serranus subligarius*	Serranidae	0.02	0.02	55.9
Tobaccofish	*Serranus tabacarius*	Serranidae	0.12	0.12	14.3
Harlequin bass	*Serranus tigrinus*	Serranidae	0.36	0.41	8.5
Chalk bass	*Serranus tortugarum*	Serranidae	0.07	0.36	62.4
Mutton hamlet	*Alphestes afer*	Serranidae	0.002	0.002	135.4
Western Atlantic seabream	*Archosargus rhomboidalis*	Sparidae	0.0002	0.0004	201.6
Saucereye porgy	*Calamus calamus*	Sparidae	0.20	0.29	17.5
Sheepshead porgy	*Calamus penna*	Sparidae	0.07	0.09	34.4
Pluma porgy	*Calamus pennatula*	Sparidae	0.002	0.05	98.0
Littlehead porgy	*Calamus proridens*	Sparidae	0.09	0.09	27.4
Porgy species	*Calamus* spp.	Sparidae	0.08	0.10	34.0
Jolthead porgy	*Calamus bajonado*	Sparidae	0.02	0.02	38.3
Whitebone porgy	*Calamus leucosteus*	Sparidae	0.04	0.06	38.9
Knobbed porgy	*Calamus nodosus*	Sparidae	0.03	0.05	46.9

Appendix 3. (continued)

Common Name	Scientific Name	Family	*P*	*D*	*CV(D)*
Silver porgy	*Diplodus argenteus*	Sparidae	0.01	0.03	62.2
Spottail seabream	*Diplodus holbrookii*	Sparidae	0.04	0.18	47.0
Pinfish	*Lagodon rhomboides*	Sparidae	0.0007	0.003	137.6
Sheepshead	*Archosargus probatocephalus*	Sparidae	0.02	0.04	51.8
Great barracuda	*Sphyraena barracuda*	Sphyraenidae	0.02	0.03	50.5
Southern sennet	*Sphyraena picudilla*	Sphyraenidae	0.001	0.29	158.2
Scalloped hammerhead	*Sphyrna lewini*	Sphyrnidae	0.002	0.002	182.5
Bonnethead	*Sphyrna tiburo*	Sphyrnidae	0.0003	0.0002	200.7
Pipefish species	*Syngnathus* spp.	Syngnathidae	0.002	0.0009	188.5
Inshore lizardfish	*Synodus foetens*	Synodontidae	0.01	0.005	39.8
Sand diver	*Synodus intermedius*	Synodontidae	0.02	0.01	79.6
Diamond lizardfish	*Synodus synodus*	Synodontidae	0.0006	0.0006	198.6
Southern puffer	*Sphoeroides nephelus*	Tetraodontidae	0.0001	0.0002	202.9
Bandtail puffer	*Sphoeroides spengleri*	Tetraodontidae	0.11	0.08	14.3
Checkered puffer	*Sphoeroides testudineus*	Tetraodontidae	0.006	0.004	67.7
Sharpnose puffer	*Canthigaster rostrata*	Tetraodontidae	0.80	2.61	6.0
Blackwing searobin	*Prionotus rubio*	Triglidae	0.001	0.0006	123.8
Bandtail searobin	*Prionotus ophryas*	Triglidae	0.0004	0.0002	142.5
Unknown species	*Unknown* spp.	unknown	0.001	0.05	170.6
Yellow stingray	*Urobatis jamaicensis*	Urotrygonidae	0.05	0.03	19.4