Coral reef ecosystems of Reserva Natural de La Parguera (Puerto Rico): Spatial and temporal patterns in fish and benthic communities (2001-2007)

A cooperative investigation between NOAA and the University of Puerto Rico

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Coral reef ecosystems of Reserva natural de La Parguera (Puerto Rico): Spatial and temporal patterns in fish and benthic communities (2001-2007)

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About this Document

The report provides a spatial and temporal characterization of the fish and benthic communities of southwestern Puerto Rico, primarily within the La Parguera Natural Reserve. The reserve is a multi-use area that spans the continental shelf from the extensive mangrove forests fringing the shoreline to the complex shelfedge coral reefs that support a diverse and productive fish community. The coral reef ecosystem of La Parguera supports a locally important fishery, as well as recreational activities such as boating, snorkeling and diving. The data and synthesis in this report are intended to provide essential baseline biological information to support future management decision making. The project is a component of NOAA's Caribbean Coral Reef Ecosystem Monitoring (CREM) project of NOAA's Coral Reef Conservation Program (CRCP) and was conducted through an ongoing multi-agency collaboration between NOAA's Center for Coastal Monitoring and Assessment Biogeography Branch (CCMA-BB), the University of Puerto Rico and the Puerto Rico Government's Department of Natural and Environmental Resources (DNER).

This Technical Memorandum is part of a series of reports that focus on providing a quantitative spatial and temporal characterization of living marine resources and benthic communities associated with marine protected areas in the U.S. Caribbean. This project complements the National Coral Reef Ecosystem Monitoring Program's (NCREMP) Coral Reef Ecosystem Monitoring grants awarded to DNER by CRCP. This project was funded by NOAA's CRCP and National Centers for Coastal Ocean Science's CCMA.

Related projects include:

Caribbean Coral Reef Ecosystem Monitoring
http://ccmaserver.nos.noaa.gov/ecosystems/coralreef/reef_fish.html

Development of Reef Fish Monitoring Protocols to Support the National Park Service Inventory and Monitoring Program
http://ccmaserver.nos.noaa.gov/ecosystems/coralreef/fish_protocol.html

Coral bleaching and recovery observed at Buck Island, St. Croix, U.S. Virgin Islands, October and December, 2005
http://ccma.nos.noaa.gov/ecosystems/coralreef/coral_bleaching.html

National Coral Reef Ecosystem Montoring Program
http://ccma.nos.noaa.gov/ecosystems/coralreef/coral_grant.html

Benthic Habitat Mapping of Puerto Rico and the U.S. Virgin Islands
http://ccma.nos.noaa.gov/ecosystems/coralreef/usvi_pr_mapping.html

Seafloor Characterization of the U.S. Caribbean - *R/V* Nancy Foster Missions
http://ccma.nos.noaa.gov/products/biogeography/usvi_nps/overview.html

All photographs provided in this document were taken by NOAA/NOS/NCCOS/Center for Coastal Monitoring Assessment Biogeography Branch in Puerto Rico unless otherwise indicated.

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Executive Summary

Since 1999, NOAA's Center for Coastal Monitoring and Assessment, Biogeography Branch (CCMA-BB) has been working with federal and territorial partners to characterize monitor and assess the status of the marine environment in southwestern Puerto Rico. This effort is part of the broader NOAA Coral Reef Conservation Program's (CRCP) National Coral Reef Ecosystem Monitoring Program (NCREMP). With support from CRCP's NCREMP, CCMA conducts the "Caribbean Coral Reef Ecosystem Monitoring project" (CREM) with goals to: (1) spatially characterize and monitor the distribution, abundance and size of marine fauna associated with shallow water coral reef seascapes (mosaics of coral reefs, seagrasses, sand and mangroves); (2) relate this information to *in situ* fine-scale habitat data and the spatial distribution and diversity of habitat types using benthic habitat maps; (3) use this information to establish the knowledge base necessary for enacting management decisions in a spatial setting; (4) establish the efficacy of those management decisions; and (5) develop data collection and data management protocols. The monitoring effort of the La Parguera region in southwestern Puerto Rico was conducted through partnerships with the University of Puerto Rico (UPR) and the Puerto Rico Department of Natural and Environmental Resources (DNER). Project funding was primarily provided by NOAA CRCP and CCMA.

In recent decades, scientific and non-scientific observations have indicated that the structure and function of the coral reef ecosystem in the La Parguera region have been adversely impacted by a wide range of environmental stressors. The major stressors have included the mass *Diadema* die off in the early 1980s, a suite of hurricanes, overfishing, mass mortality of *Acropora* corals due to disease and several coral bleaching events, with the most severe mass bleaching episode in 2005. The area is also an important recreational resource supporting boating, snorkeling, diving and other water based activities. With so many potential threats to the marine ecosystem several activities are underway or have been implemented to manage the marine resources. These efforts have been supported by the CREM project by identifying marine fauna and their spatial distributions and temporal dynamics. This provides ecologically meaningful data to assess ecosystem condition, support decision making in spatial planning (including the evaluation of efficacy of current management strategies) and determine future information needs. The ultimate goal of the work is to better understand the coral reef ecosystems and to provide information toward protecting and enhancing coral reef ecosystems for the benefit of the system itself and to sustain the many goods and services that it offers society. This Technical Memorandum contains analysis of the first seven years of fish survey data (2001-2007) and associated characterization of the benthos. The primary objectives were to quantify changes in fish species and assemblage diversity, abundance, biomass and size structure and to provide spatially explicit information on the distribution of key species or groups of species and to compare community structure across the seascape including fringing mangroves, inner, middle, and outer reef areas, and open ocean shelf bank areas.

Methods:

For each biannual survey mission, sample sites were selected via a stratified random design using hard and soft bottom habitat types delineated in NOAA's benthic habitat map (Menza et al., 2006). Fish were surveyed during daylight hours along 25 m long by 4 m wide belt-transects for a fixed duration of 15 minutes. All species observed were identified to the lowest possible taxonomic level and their abundance was counted and grouped by size class. To quantify benthic habitat, five 1 m^2 quadrats were randomly placed along the transects and used to examine the relatively fine-scale biotic and abiotic components of the seascape (e.g., coral cover, macroalgal cover, etc.). A total of 1,167 fish surveys (572 from hardbottom, 437 from softbottom and 158 from mangrove habitats) were used in this analysis.

The raw data were summarized by habitat type to examine differences in richness, abundance and biomass in different habitat types and were mapped across the study region to examine across shelf patterns in community and species attributes. Changes occurring between years and by season over the duration of the study were summarized and tested to detect periods with significant change. The data and synthesis provides essential baseline biological data via: 1) a comprehensive spatial characterization of resources and habitat condition, and 2) an examination of increases and decreases in the abundance of fish species and components of the benthic habitat across the study area.

Major findings:

Diversity hotspots

- Highest coral cover was observed along the shelf edge southward of Terrumote I and around El Palo Reef. Cover of *Acropora palmata* (4.1-10.4%) was highest around El Palo Reef. Highest cover of *Montastraea annularis* complex (25.1-30.3%) was highest south-west of Turrumote I.
- Hot spots of coral species richness (10-14 species per 100 m^2) and diversity (H = 1.7-2.5) occurred offshore but were scattered throughout hard bottom habitat types.
- Highest rugosity (0.81-1) occurred between Margarita Reef and El Palo Reef and at Romero.
- Highest species richness of 41 fish per 100 m^2 area was recorded near the eastern shelf edge over colonized pavement with sand channels.
- Hotspots of high fish species richness, high fish biomass and high herbivore abundance and richness co-occurred along the shelf edge and around the complex of patch reefs between Margarita Reef and El Palo Reef.

Benthic habitat

- Hardbottom habitat types were dominated by algae, with an average of 5% live stony coral cover across the study area
- *Montastraea* and *Porites* spp. were the most common stony corals, with *M. annularis* complex comprising 35% of total coral cover, and *Porites astreoides,* the most frequently observed species occurring at 62% of hardbottom sites
- *Acropora* species were rarely observed, but *Acropora cervicornis* exhibited a wider spatial distribution than *A. palmata*
- Temporal analysis revealed that live coral cover varied significantly among some sampling years, but overall live coral cover decreased over the sampling period (2001-2007)
- Mangrove prop roots provided a substrate for colonization by a wide diversity of epiphytic organism

Fish

- Across the La Parguera study area a total of 210 fish species were identified to species level, with at least another 14 fish identified to genera
- Highest fish densities were associated with mangroves, with the assemblages composed mostly of juveniles

- Twenty-five of the 30 most abundant fish species in mangroves were also observed over coral reefs indicating a high level of multi-habitat use, but with fish body length markedly smaller in mangroves than on coral reefs. This can be indicative of size dependent ontogenetic habitat shifts, particularly for grunts (Haemulidae), snapper (Lutjanidae), parrotfish (Scaridae) and barracuda (Sphyraenidae)

- Based on body size, mangroves appeared to function as an intermediate habitat type for some grunts and snappers, with smallest fish associated with seagrasses, larger fish in mangroves and the largest mean length recorded for fish on coral reefs

- The small-bodied parrotfish species, striped parrotfish (*Scarus iseri*) and redband parrotfish (*Sparisoma aurofrenatum*) were the most commonly occurring fish across the seascapes at La Parguera

- Although rainbow parrotfish (*Scarus guacamaia*) is thought to have a high dependence on mangroves and coral reefs, only two individuals were observed between 2001 and 2007 in the La Parguera region confirming the rarity of the species in the U.S. Caribbean and its status as vulnerable according to the IUCN red list of threatened species

- Sightings of many large-bodied fish species targeted by the fishery have declined substantially over the past 25 years based on comparison between 1980-1981 and 2001-2007. These include rainbow parrotfish, midnight parrotfish (*Scarus coelestinus*), Nassau grouper (*Epinephelus striatus*), tiger grouper (*Mycteroperca tigris*), rock hind (*Epinephelus adscensionis*), red hind (*Epinephelus guttatus*), coney (*Cephalopholis fulva*), graysby (*Cephalopholis cruentata*), yellowtail snapper (*Ocyurus chrysurus*), lane snapper (*Lutjanus synagris*), mahogany snapper (*Lutjanus mahogoni*), dog snapper (*Lutjanus jocu*) and queen triggerfish (*Balistes vetula*)

- The largest snappers seen in the study area were markedly smaller than the maximum known size for the species, particularly schoolmaster (*Lutjanus apodus*), *O. chrysurus* and gray snapper (*Lutjanus griseus*)

- From a total of 1,167 surveys (572 from hardbottom) over seven years, no *M. tigris* or yellowfin (*Mycteroperca venenosa*) were observed and only two *E. striatus*; two black grouper (*Mycteroperca bonaci*); two *E. adscensionis* and 43 *E. guttatus* were observed

- Small-bodied groupers including *C. cruentata* (n=246) and coney *C. fulva* (n=81) were significantly more abundant than large-bodied groupers

- None of the grouper species observed had attained the maximum known size for their species, with maximum length for *Cephalopholis* species in the La Parguera region estimated at 30 cm fork length (FL) compared with a maximum known for *C. fulva* of 41 cm total length (TL) and 43 cm TL for *C. cruentata*. The largest *E. guttatus* was approximately 50% of the maximum recorded size

- Grunt (Haemulidae) abundance and biomass was highest for mangroves in close proximity to coral reefs and seagrass beds. All common species of grunt showed a strong across shelf size-dependent distribution, with the majority of juveniles in lagoonal nearshore areas and adults in deeper mid- and outer-shelf zones. Juveniles and adults did also co-occur at several sites across the shelf indicating some flexibility in the strategy for ontogenetic segregation

- Five sharks (two species) and three stingrays were the only sharks and rays observed within transects at La Parguera between 2001 and 2007

- Total fish biomass, herbivore biomass, grouper abundance, parrotfish and wrasse abundance were significantly higher in 2007 than in 2001

- Grouper biomass in 2007 was significantly lower than in 2001

- Only *C. fulva* and *S. aurofrenatum* increased in abundance consecutively for more than three years between 2001 and 2007

- Total snapper density and *L. apodus* density decreased from 2002 to 2005.

- The most striking inter-annual difference occurred between 2003 and 2004, whereby 65 metrics (approximately 80% of all metrics) decreased, with five decreasingly significantly; followed by 70% of metrics increasing the following year (2004-2005). A specific cause was not identified

- 65% of the 20 most abundant fish exhibited higher density in summer than in winter. Density of *Thalassoma bifasciatum* (bluehead wrasse) and *Chromis cyanea* (blue chromis) was more than 50% higher in summer. However, grunts, *Haemulon aurolineatum* (tomtate) and *Haemulon flavolineatum* (french grunt), and *S. iseri* were more abundant in winter

- Mean biomass for grouper, snapper, parrotfish and grunts was lower in the winter than summer

- Planktivores and piscivores were more abundant in summer than winter

Recommendations:

Management plans are urgently required for the La Parguera Natural Reserve to effectively balance human resource usage with conservation objectives. Survey data from this study indicates that the coral reef ecosystem within and around the La Parguera Natural Reserve is impacted by multiple stressors, with low live coral cover, high macroalgal abundance and a depleted population of large-bodied species resulting in shifts in species dominance.

The data collected through the NCREMP monitoring program and synthesized in this report together with related publications, CRES and UPR studies have made the La Parguera Natural Reserve one of the best studied regions in the Caribbean.

These scientific survey data and associated map products provide a template of environmental data to support a process of marine zoning that could include conservation areas and ‘no-take’ reserves for selected areas, while maintaining access to other areas for human activities. In this report, diversity hotspots and distributions of key species have been identified and mapped and the population status of harvested species has been quantified, providing the prerequisites for the development of successful spatial management strategies. The shelfedge environment and the complex coral reef ecosystems between Margarita Reef and El Palo Reef support highest fish species diversity and high abundance for many species and should receive special management attention. Historical comparison, however, revealed that many of the large-bodied fish species that are targeted by the fishery have markedly declined in abundance since 1980. The very low abundance of large-bodied groupers and parrotfish is likely to have a major impact to ecosystem functioning and should receive priority attention. The shift towards declining live coral cover and increasing algal abundance over much of the region is usually associated with a degrading coral reef ecosystem. Mangroves are an important habitat type in this region and function as part of an interconnected mosaic of habitats for many fish species. Efforts to protect and restore mangroves, particularly those that are in close proximity to seagrasses and coral reefs will be beneficial to diversity and productivity. Restoration targets can be set for rebuilding depleted populations of large-bodied fish and for creating suitable water quality conditions to help restore benthic habitat structure and coral reef architecture. The Natural Reserve is a unique coral reef ecosystem in Puerto Rico, due to its relatively sheltered position on a wide and shallow section of the insular shelf of southern Puerto Rico, but like many such systems it is vulnerable to over-use and influenced by a rapidly changing local, regional and global environment. Comprehensive and strategic management is now urgently needed to restore ecological integrity to the La Parguera Natural Reserve and to ensure the long-term sustainability of the coral reef ecosystem for current and future generations.

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Resumen

Desde el 1999 el programa de Bio-geografía del Centro para Monitoreo y Evaluación Costera (CCMA-BB por sus siglas en inglés) de la Administración Nacional de Océanos y Atmosfera (NOAA por sus siglas en inglés) ha colaborado con agencias federales y territoriales para caracterizar, monitorear y evaluar la condición del ambiente marino del suroeste de Puerto Rico. Este esfuerzo es parte del Programa de Conservación de Arrecifes de Coral (CRCP por sus siglas en inglés) bajo el Programa Nacional de Monitoreo de Ecosistemas de Arrecifes de Coral (NCREMP por sus siglas en inglés) de NOAA. Con el apoyo del NCREMP del CRCP el CCMA lleva a cabo el proyecto titulado “Monitoreo de Ecosistemas de Arrecifes de Coral del Caribe” (CREM por sus siglas en inglés) cuyas metas incluyen: (1) caracterizar espacialmente y monitorear la distribución, abundancia y tamaño de la fauna asociada con los paisajes de arrecifes de coral (mosaico compuesto de arrecifes de coral, hierbas marinas, arenales y manglares); (2) relacionar esta información con datos de hábitat a pequeña escala y la distribución y diversidad de tipos de hábitat utilizando mapas de hábitat bénticos; (3) utilizar esta información para establecer una base de conocimiento sobre la cual formular medidas de manejo con un marco espacial; (4) evaluar la efectividad de medidas de manejo; y (5) desarrollar protocolos par la colección y distribución de datos de monitoreo. Los esfuerzos de monitoreo de la región de La Parguera, en el suroeste de Puerto Rico se llevaron a cabo en colaboración con la Universidad de Puerto Rico (UPR) y el Departamento de Recursos Naturales y Ambientales (DRNA) de Puerto Rico. La subvención de este proyecto proviene principalmente de NOAA CRCP y CCMA.

En tiempos recientes, observaciones científicas y no-científicas han indicado que la estructura y funcionamiento del ecosistema de arrecifes de coral de la región de La Parguera se han afectado por una gama de impactos ambientales. Algunos de estos impactos incluyen la mortandad masiva del erizo negro (Diadema) a principios de los 1980’s, varios huracanes, la sobre-explotación pesquera, mortandades significativas del coral Acropora por enfermedades y eventos de blanqueamientos, el mas severo de estos en el 2005. El área es también un importante recurso recreativo donde se llevan a cabo actividades relacionadas con el mar incluyendo la navegación, careteo o ‘snorkeling’ y buceo entre otras. Ante la variedad de amenazas potenciales al ecosistema marino se han implementado actividades para manejar el recurso marino. Estos esfuerzos han recibido el apoyo del proyecto CREM al identificar la dinámica temporal y la distribución espacial de la fauna marina. Este estudio provee datos ecológicos que sirven para evaluar la condición del ecosistema, apoyan la toma de decisiones en planificación espacial (incluyendo la evaluación de la efectividad de estrategias de manejo) y permite identificar las necesidades de información futuras. La meta fundamental de este trabajo es lograr entender el ecosistema de arrecifes de coral y proveer información para proteger y mejorar el mismo para el beneficio del ecosistema al igual que mantener los servicios y bienes que este provee a la sociedad.

Este memorando técnico contiene un análisis de los primeros siete años (2001-2007) de censos de peces y caracterización béntica. El objetivo principal fue cuantificar cambios en la diversidad, abundancia, biomasa y estructura de tamaños de los peces y proveer información espacial explícita de la distribución de especies claves y grupos específicos. Esta información sirve a su vez para comparar la estructura de la comunidad a lo largo del paisaje marino compuesto de múltiples hábitats, incluyendo los manglares de franja, arrecifes en distintas posiciones de la plataforma y áreas de plataforma oceánica.

Métodos:

Para cada misión de muestreo semi-anual se seleccionaron sitios mediante un diseño aleatorio estratificado entre hábitats de fondos duros y blandos (arenosos) según el mapa de hábitat béntico de NOAA (Menza et al., 2006). Los censos de peces se llevaron a cabo durante el día en transectos de banda de 25 por 4 metros durante 15 minutos. Todas las especies observadas se identificaron al nivel taxonómico mas bajo posible y las abundancias se cuantificaron en clases de tamaños. Para estimar la composición del hábitat béntico, cinco cuadratas de un metro cuadrado se colocaron al azar en el transecto y se cuantificaron los componentes bióticos y abióticos (por ejemplo cobertura de coral, algas, etc.). Para este análisis se utilizaron los datos de 1,167 censos de peces, de los cuales 572 provienen de fondos duros y 595 de fondos blandos.

Los datos se resumieron por tipo de hábitat para examinar diferencias en la riqueza, abundancia y biomasa de peces en distintos hábitats bénticos y se mapearon a lo largo del área de estudio para examinar patrones en la distribución de los atributos de comunidades y especies. Se resumieron cambios entre años y por temporadas durante la duración del estudio para identificar mudanzas significativos entre periodos de muestreo. La síntesis de estos datos provee una base biológica mediante: 1) la caracterización espacial comprensiva de los recursos y la condición del hábitat y 2) los patrones de cambios en abundancias de peces y los componentes bénticos de hábitats en el área de estudio.

Resultados sobresalientes:

Centros de diversidad

- La cobertura de coral más alta se observo a lo largo del veril al sur de Cayo Turrumote I y en áreas circundantes al Cayo El Palo. La cobertura de *Acropora palmata* (4.1-10.4%) fue mayor en áreas de Cayo El Palo mientras la cobertura del complejo de *Montastraea annularis* (25.1-30.3%) fue mayor al suroeste de Cayo Turrumote I

- Puntos de alta riqueza de especies (10-14 especies por 100m^2) y diversidad (H = 1.7-2.5) de corales se observaron lejos de la costa en distintas áreas de hábitats de fondos duros

- La rugosidad mayor (0.81-1) se observó en un área entre el Cayo Margarita y Cayo El Palo al igual que en Cayo Romero

- Se detectaron áreas donde la riqueza de especies de peces fue más alta (41 especies por 100m^2) cerca del veril en hábitat de pavimento colonizado con canales de arena

- Puntos con alta riqueza de especies y alta biomasa de peces al igual que alta riqueza de especies y abundancia de peces herbívoros coincidieron a lo largo del veril y alrededor del grupo de arrecifes de parche localizados entre el Cayo Margarita y Cayo El Palo

Hábitat béntico

- Los hábitats duros tuvieron una cobertura béntica dominada por algas con un promedio de 5% de cobertura de coral vivo a lo largo del área de estudio

- Los hábitats duros tuvieron una cobertura béntica dominada por algas con un promedio de 5% de cobertura de coral vivo a lo largo del área de estudio

- Los corales más comunes pertenecen a los géneros *Montastraea* y *Porites*, donde *M. annularis* compuso el 35% de la cobertura total y *Porites astreoides* se observó mas frecuentemente en 62% de los sitios de muestreo de hábitats duros

- Ejemplares del genero Acropora fueron observados muy pocas veces y entre estos *Acropora cervicornis* tuvo una distribución espacial más amplia que *A. palmata*

- El análisis temporal revelo que la cobertura de coral vivo varió significativamente entre algunos años, pero en general declinó a lo largo del muestreo (2001-2007)

- Las raíces de mangles sumergidas proveen un sustrato béntico para la colonización de especies epífitas

Peces

- A lo largo del área de estudio en La Parguera se identificaron peces de 210 especies perteneciente a 14 géneros

- Las densidades más altas se encontraron en los manglares donde los juveniles dominaron el conjunto de peces

- Veinticinco de las 30 especies más abundantes en los manglares también se observaron en los arrecifes de coral lo cual indica el uso de múltiples tipos de hábitats por estas especies. Además las tallas de peces en los manglares fueron menores a los arrecifes. Esto indica que están ocurriendo cambios ontogenéticos (a lo largo de su ciclo de vida) en los tipos de hábitat para las familias roncos (Haemulidae), pargos (Lutjanidae), loros (Scaridae) y picúas (Sphyraenidae)

- Basado en las tallas de los peces, los manglares aparentan tener una función de hábitat intermedio para algunos roncos y pargos, donde las tallas menores están asociadas a las hierbas marinas, las tallas intermedias en los manglares y el tamaño promedio mayor se encuentra en los arrecifes de coral

- Las especies más comunes a lo largo de todo el paisaje marino de La Parguera fueron los loros pequeños *Scarus iseri* y *Sparisoma aurofrenatum*

- Aunque se ha demostrado una alta dependencia de loro guacamayo (*Scarus guacamaia*) hacia los manglares y los arrecifes de coral, solamente se observaron dos individuos en La Parguera entre el 2001 y 2007 lo cual revalida que la especie se considere escasa en el Caribe de Estados Unidos y vulnerable según la lista de especies amenazadas de la Unión Internacional para la Conservación de la Naturaleza (UICN)

- El avistamiento de varias especies de importancia comercial y de gran tamaño ha disminuido sustancialmente durante los pasados 25 años (basado en comparaciones entre 1980-81 y 2001-07) incluyendo a las siguientes especies: *Scarus guacamaia, Scarus coeruleus, Epinephelus striatus, Mycteroperca tigris, Epinephelus adscencionis, Epinephelus guttatus, Cephalopholis cruentata, Cephalopholis fulva, Ocyurus chrysurus, Lutjanus synagris, Lutjanus mahogoni, Lutjanus jocu* y *Balistes vetula*

- Los pargos más grandes observados en La Parguera fueron de tallas menores al tamaño máximo reportado para estas especies, particularmente para *Lutjanus apodus*, *O. chrysurus* y *Lutjanus griseus*

- De la totalidad de 1,167 censos (572 en fondos duros) a lo largo de siete años no se observó ningún ejemplar de Mycteroperca tigris ni *Mycteroperca venenosa* y solamente se observaron dos ejemplares de *E. striatus*, *Mycteroperca bonaci* y *E. adscensionis* mientras se cuantificaron únicamente 43 Epinephelus guttatus

- Los meros de menor tamaño, incluyendo *C. cruentata* (n=246) y *C. fulva* (n=81) fueron significativamente mas abundantes que los meros de gran tamaño

- Ninguna de las especies de meros observadas presentó el tamaño máximo reportado para su especie. Por ejemplo el tamaño máximo observado para especies de *Cephalopholis* en La Parguera se estimó en 30 cm (largo horquilla) aunque la talla máxima reportada para *C. fulva* es de 41 cm (largo total) y para *C. cruentata* es de 43 cm (largo total). La talla mayor observada en La Parguera para *E. guttatus* fue aproximadamente 50% del tamaño máximo reportado en la literatura científica

- La abundancia y biomasa de roncos (haemulidos) fue más alta en los manglares cercanos a arrecifes de coral y hierbas marinas. Todas las especies comunes de roncos demostraron un cambio en la distribución de tamaños que aumentó en sitios cercanos al veril, por ejemplo la mayoría de los juveniles se encontraron en lagunas cercanas a la costa y los adultos en áreas mas profundas de zonas intermedias y lejanas de la costa. Los juveniles y adultos coincidieron en varios lugares a lo largo de la plataforma lo cual indica alguna flexibilidad en su segregación ontogenética

- Solamente se observaron cinco tiburones (de dos especies) y tres rayas en La Parguera entre el 2001 y 2007

- La biomasa total y la biomasa de herbívoros, al igual que la abundancia de meros, loros y lábridos fueron significativamente mayores en 2007 que durante el 2001

- La biomasa de meros fue significativamente menor el 2007 que durante el 2001

- Se observó un aumento consecutivo de más de tres años (entre 2001 y 2007) en la abundancia de *C. fulva* y *S. aurofrenatum*

- La densidad total de pargos y la densidad de *L. apodus* declinó entre 2002 y 2005

- La diferencia entre años más sobresaliente ocurrió entre 2003 y 2004 donde declinaron 65 métricas (aproximadamente 80% de todas las métricas), de las cuales cinco declinaron significativamente; seguido por un aumento de 70% de las métricas en el año siguiente (2004-2005)

- Se observaron densidades más altas en verano que invierno para 65% de las 20 especies más abundantes. Por ejemplo la densidad de Thalassoma bifasciatum y *Chromis cyanea* fue 50% más alta durante el verano. Mientras tanto los roncos *Haemulon aurolineatum* y *Haemulon flavolineatum* al igual que el loro *S. iseri* tuvieron mayores abundancias en temporada de invierno.

- La biomasa promedio para meros, pargos, loros y roncos fue menor en temporada de invierno en comparación con verano

- Los planctívoros y picívoros fueron más abundantes durante temporada de verano en comparación con invierno

Recomendaciones:

Para alcanzar un balance efectivo entre los usos del recurso marino por los humanos y los objetivos de conservación se requieren urgentemente unos planes de manejo para la Reserva Natural de La Parguera. Los datos del presente estudio indican que el ecosistema de arrecifes de coral dentro y alrededor de la Reserva Natural de La Parguera esta afectada por múltiples amenazas causando una baja cobertura de coral, mayor abundancia de algas y escasez de peces de gran tamaño lo cual resulta en una mudanza de las especies dominantes.

Los datos colectados por el programa de monitoreo de N-CREMP y la síntesis de este informe al igual que otras publicaciones, e investigaciones de CRES y la UPR han establecido la Reserva Natural de La Parguera como una de las regiones más estudiadas del Caribe. Los resultados científicos al igual que los mapas proveen un borrador para comenzar un proceso de zonificación marina que puede incluir áreas de conservación, reservas marinas de no pesca, y garantizar el acceso a lugares para actividades recreativas. Los lugares de mayor diversidad y distribución de especies o grupos claves han sido identificados y cartografiados. Además se ha evaluado la condición de las poblaciones de especies de importancia comercial lo cual provee un pre-requisito para el desarrollo de estrategias de manejo y planificación espacial. El ambiente del veril y el ecosistema de arrecife de coral entre los cayos La Margarita y El Palo parecen tener mayor diversidad y abundancia para varias especies de peces lo cual merece atención especial por los manejadores. En una comparación histórica se destaca que la mayoría de los peces de importancia comercial de gran tamaño han declinado marcadamente en abundancia desde los años 80. Las abundancias bajas de los meros y loros de gran tamaño puede tener impactos a la función del ecosistema y deben recibir atención prioritaria.

El cambio en la composición béntica hacia mayor cobertura de algas en vez de corales en la mayoría de la región esta asociado a una degradación del ecosistema de arrecife de coral. Los manglares son un hábitat importante en esta región y tiene una función en el mosaico de hábitats que están entrelazados por el uso de varias especies. Los esfuerzos por proteger y restaurar los manglares, particularmente aquellos que se encuentran cercanos a las hierbas y los arrecifes serán de mayor beneficio para la diversidad y la productividad. Se deben establecer metas para recuperar las poblaciones de peces de gran tamaño y crear condiciones optimas de calidad de agua para ayudar a restaurar el hábitat béntico y la arquitectura del arrecife de coral. La Reserva Natural de La Parguera es un ecosistema de arrecifes único en Puerto Rico por su posición relativamente protegida en la plataforma insular de Puerto Rico, pero como muchos otros sistemas es vulnerable al mal uso y es influenciado por cambios ambientales regionales y globales. Actualmente se destaca la necesidad urgente de unas estrategias de manejo comprensivo para restaurar la integridad ecológica de la Reserva Natural de La Parguera y asegurar la sustentabilidad a largo plazo del ecosistema de arrecifes de coral para generaciones actuales y venideras.

Referencias

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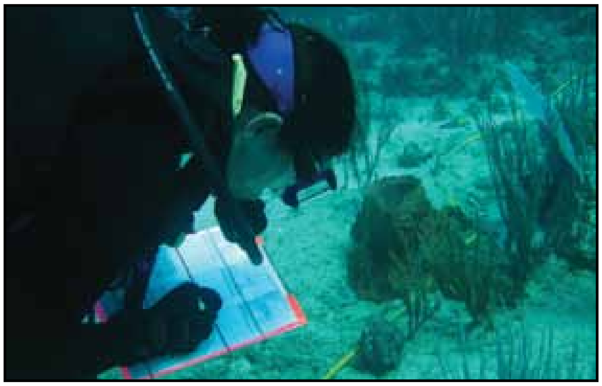
1. Introduction

La Parguera, Puerto Rico



Diver collecting benthic habitat composition data

Since 1999, NOAA's Center for Coastal Monitoring and Assessment Biogeography Branch (CCMA-BB) has been working with federal and territorial partners to characterize, monitor and assess the status of the marine environment in southwestern Puerto Rico. This effort is part of the broader NOAA Coral Reef Conservation Program's (CRCP) National Coral Reef Ecosystem Monitoring Program (NCREMP). With support from CRCP's NCREMP, CCMA conducts the "Caribbean Coral Reef Ecosystem Monitoring project" (CREM) with goals to: (1) spatially characterize and monitor the distribution, abundance, and size of marine fauna associated with shallow water coral reef seascapes (mosaics of coral reefs, seagrasses, sand and mangroves); (2) relate this information to *in situ* fine-scale habitat data and the spatial distribution and diversity of habitat types using benthic habitat maps; (3) use this information to establish the knowledge base necessary for enacting management decisions in a spatial setting; (4) establish the efficacy of those management decisions; and (5) develop data collection and data management protocols. Since 2002, CREM surveys have contributed to the Coral Reef Ecosystem Studies (CRES) program in the La Parguera region. CRES was a five-year research program funded through NOAA's Center for Sponsored Coastal Ocean Research and coordinated by the Department of Marine Sciences of University of Puerto Rico (UPR) to define and understand causes and effects of reef degradation, and provide managers with information and tools to aid in reversing the degradation of U.S. Caribbean coral reef ecosystems. The monitoring effort for the La Parguera region was conducted through partnerships with UPR and the Puerto Rico Department of Natural and Environmental Resources (DNER). The intention of this report is to provide a comprehensive spatial and temporal characterization to support management decision making and baseline data to facilitate the development of an effective adaptive management plan for the Reserva Natural La Parguera.

1.1. Introduction to the La Parguera study region

The coral reef ecosystem of the La Parguera region of southwestern Puerto Rico is a complex spatial mosaic of habitat types dominated by coral reefs, seagrasses, macroalgal beds, unconsolidated sediments and mangroves. The broad shelf and coastal embayment at La Parguera provides a sheltered shallow-water environment that has facilitated the development of a diverse and productive seascape, with important ecological, economic and cultural value. The mangrove forests of the La Parguera region are some of the most extensive in Puerto Rico and have been protected as insular forests since 1943 (Aguiler-Perera et al., 2006). In 1979, the La Parguera region (327 km^2) was designated as a Natural Reserve (NR), known as the Reserva Natural La Parguera, becoming the second marine protected area to be established (after Aguirre in 1918) in Puerto Rico. The La Parguera NR is managed by DNER Bureau of Coastal Reserves and Refuges (BCRR) as a multiple use zone and currently has no restrictions on fishing and no formal management plan. In the 1980s, La Parguera was proposed as a potential multi-use marine sanctuary under the National Marine Sanctuaries Act, but the proposal was abandoned due to insufficient local agreement (Valdés-Pizzini, 1990; Aguiler-Perera et al., 2006). In addition, a small "no-take" marine reserve (7.6 km^2) has been proposed for Turrumote, south of Corral (Figure 1.1).

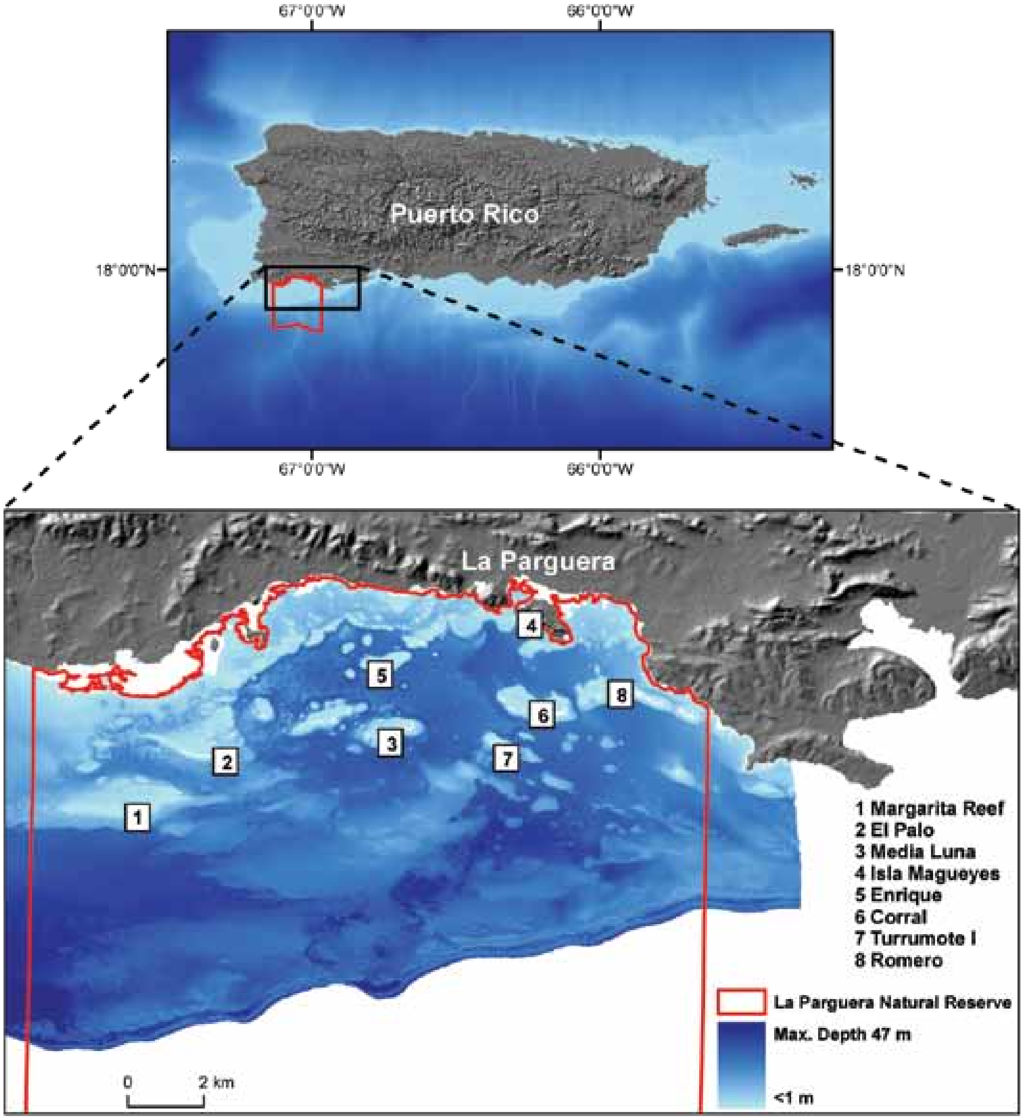


Figure 1.1. La Parguera study region in southwest Puerto Rico showing the boundary of the Natural Reserve and the bathymetry from airborne laser altimetry light and detection data (LiDAR) collected in 2006. Several prominent coral reefs, islands and mangrove cays are labeled.

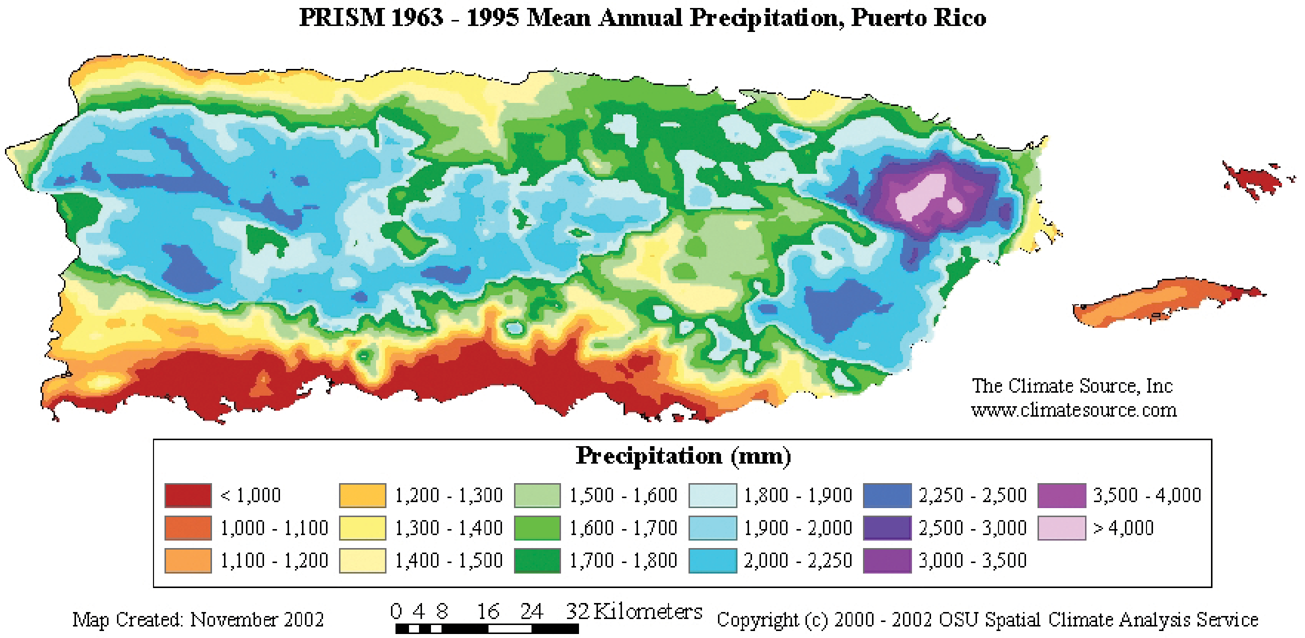


Figure 1.2. Modeled precipitation based on rainfall data from 108 National Weather Service stations. Lowest annual precipitation is recorded for the La Parguera region with 770 mm yr^{-1}. Source: http://www.climatesource.com/pr/fact_sheets/prppt_xl.jpg

The well-developed coral reef ecosystem at La Parguera exists in an area with relatively low human population size and coastal development, an absence of local rivers and the lowest rainfall in Puerto Rico (Figure 1.2).

Marine geological studies have revealed that the extensive coral reefs along the 8-10 km insular shelf developed over the past 10,000 years and were originally dominated by branching corals (*Acropora palmata*) in deeper shelf edge waters (Hubbard et al., 2008). Then in the Holocene period, reef development tracked sea level rise shoreward to create the distinct across shelf spatial arrangement of coral reef ecosystems observed today. This process led Hubbard et al. (2008) to classify coral reefs across the shelf at La Parguera into inner shelf reefs (some of which support mangrove stands), mid-shelf reefs and shelf edge reefs. The classification of inner, mid and outer shelf zones was adopted in this report to describe the spatial patterns in fish distribution across the shelf. Figure 1.3 shows examples of the complex structural features that exist across the shelf at La Parguera including collections of patch reefs, mangrove islands and shelfedge ridges and valleys.

Mangrove islands of La Parguera, PR

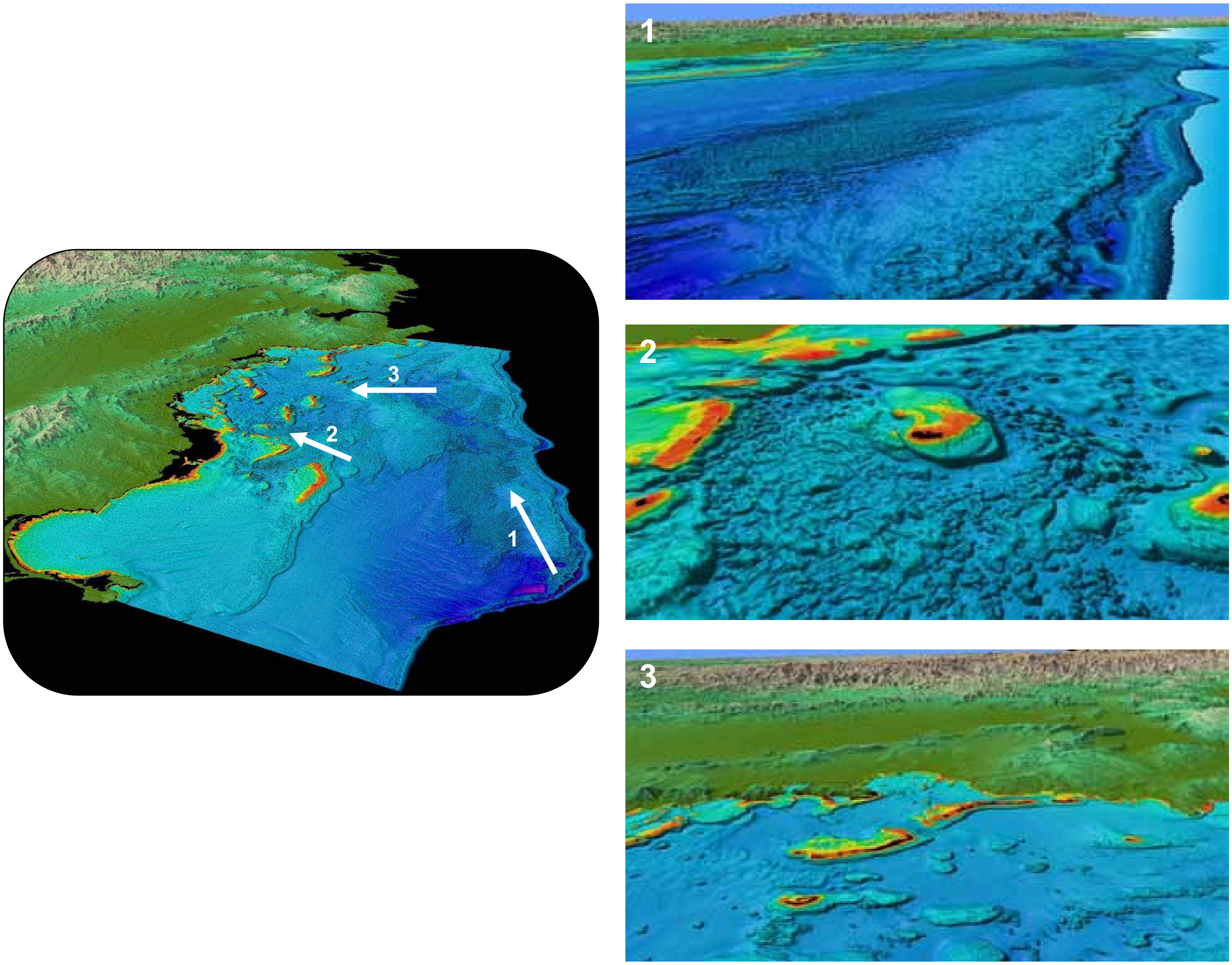


Figure 1.3. A 3-dimensional visualization of the LiDAR bathymetry for the La Parguera region showing (1) the ridges and valleys along the shelf edge; (2) the highly heterogeneous patch reefs that exist around El Palo reef, and (3) the distinctive mangrove cays that separate the lagoonal areas from the mid shelf zone.

Tides in the study area are diurnal and of low amplitude (<25 cm). The circulation of water is generally westward (Ojeda, 2002), with velocities of 10-15 cm/s (Hensley et al., 1994). Internal waves breaking on the shelf edge create nutrient pulses and periodically reduce water temperature. Offshore winds can cause an eastward subsurface current on the La Parguera shelf (Tyler and Sanderson, 1996).

1.2 Ecosystem change and the impact of multiple stressors

Until the past few decades, La Parguera coral reef ecosystems had been subjected to relatively few land-based stressors, but increasing deforestation of the coastal limestone hills, coastal resort development and housing have been linked to increased sedimentation and nutrient input to the system (Warne et al., 2005; Ryan et al., 2008). A time series of aerial photographs from 1936, 1963 and 2005 showing the La Parguera town and surrounding hills reveals an increase in urban development, roads and modification to the coastline, watershed and coastal vegetation (Figure 1.4).

Coastal development along mangrove lined coastline at La Parguera

Multiple interacting stressors including sedimentation, nutrient runoff, elevated sea water temperature and fishing are changing the structure and function of coral reef ecosystems of La Parguera (Garcia-Sais et al., 2005, 2008; Ballantine et al., 2008). Deterioration in water quality due largely to increased nutrients and turbidity as a result of land-based sources has been reported as a primary threat to nearshore coral reef ecosystems in South West Puerto Rico (Garcia-Sais et al., 2005, 2008). Bush (1991) estimated that 90 percent of the river sediment discharged at the coast is transported to the shelf edge and slope within a few months due to dominant across-shelf processes. Storm events can also redistribute sediment. Suspended sediment yields during the passage of Hurricane Georges (September 20-25, 1998) were highest in the watersheds on the south and southwest coast (Rio Portugues) yielding 7,800 tonnes/km^2 and Rio Jacquas yielding 3,700 tonnes/km^2 (Larsen and Webb, 2009). Sediment sampling (Figure 1.5) has demonstrated an overall doubling of terrestrial material in the marine sediments of backreef areas of La Parguera over the last century (Ryan et al., 2008). In southwest Puerto Rico, turbidity is a strong positive linear predictor of live coral cover along the shelf (Bejarano-Rodríguez, 2006) and has resulted in a compression of coral depth zonation, accompanied by changes in the relative abundance of coral species, which is directly related to individual species tolerance to sediment stress (Acevedo et al.,1989). Significant environmental changes have been observed, but relatively little is known about the interaction between ocean circulation, water clarity and the resilience of biotic communities across the shelf.

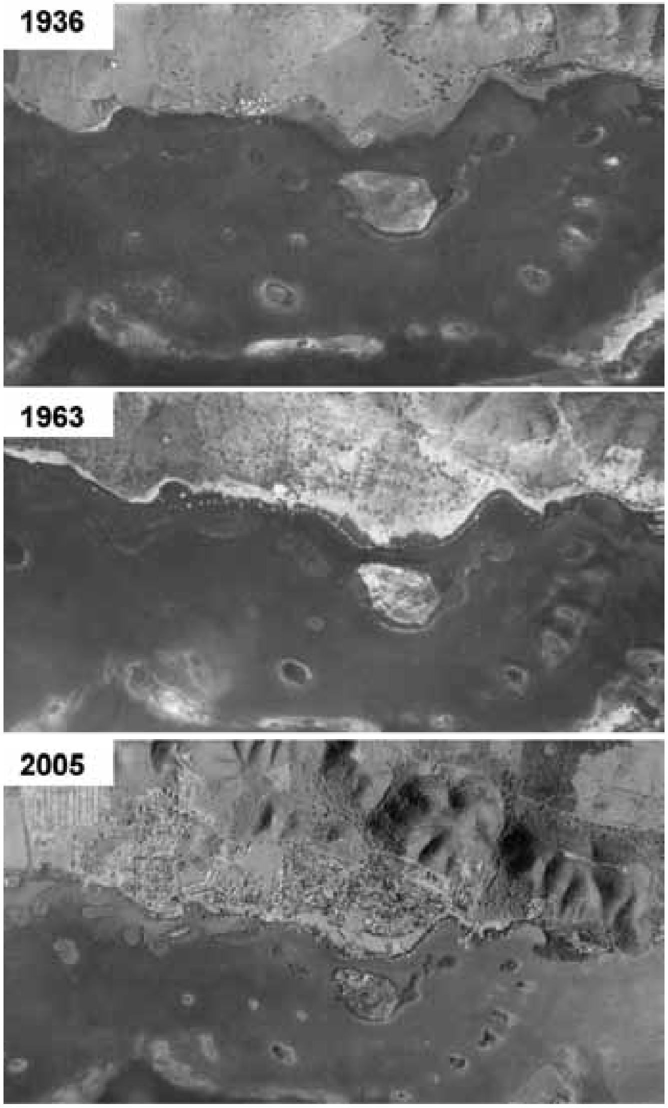


Figure 1.4. Aerial photography from 1936, 1963 and 2005 showing the growth of urban development in the La Parguera area and modifications to vegetation in the watersheds, coastline and islands. Source: http://www.log.furg.br/WEBens/morelock/clnpho.htm#par.

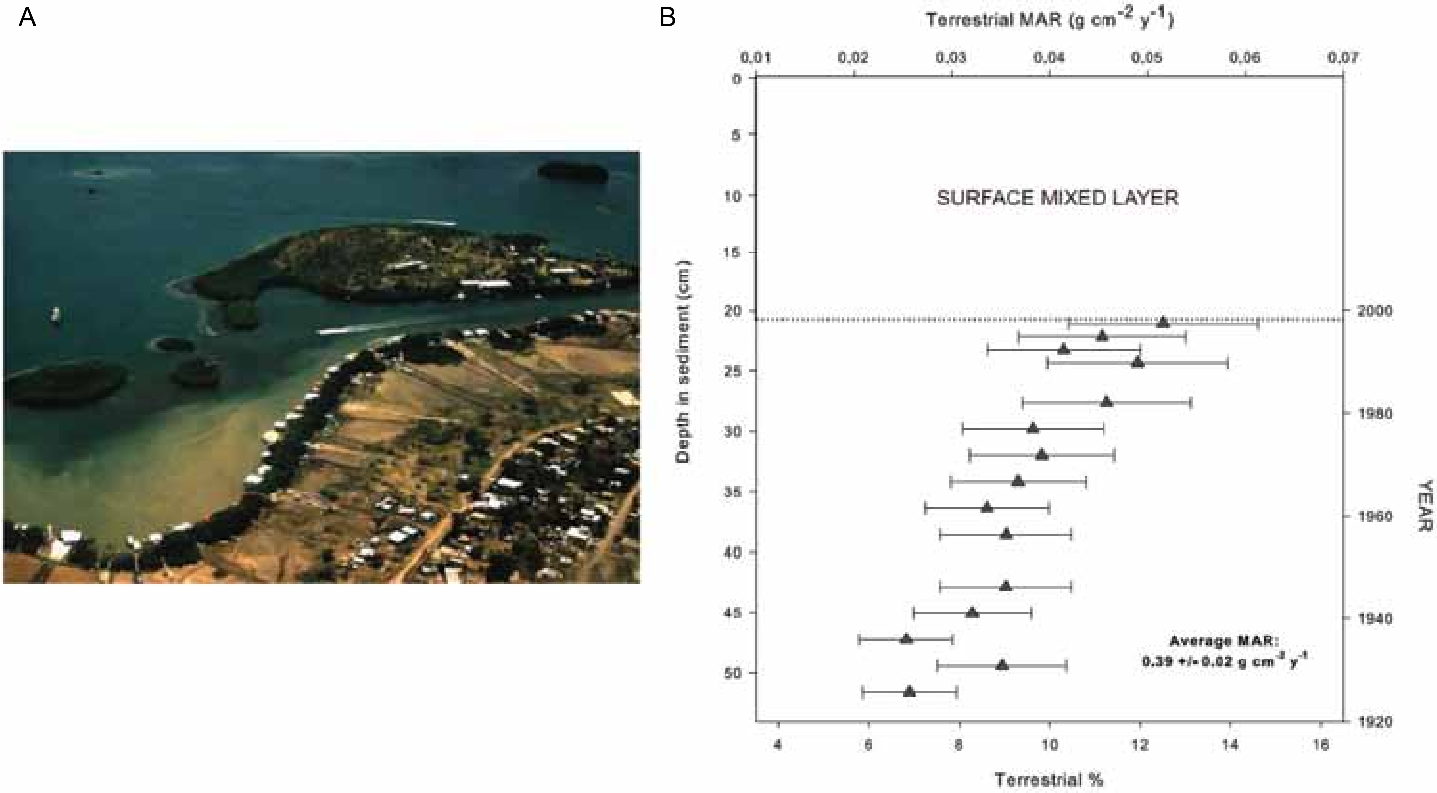
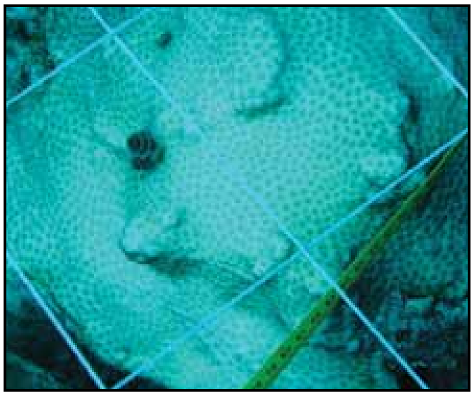


Figure 1.5. A) Aerial photograph showing sediment runoff in the 1980s along the developed shoreline at La Parguera. Source: http://www.log.furg.br/WEBens/morelock/clnpho.htm#par. B) The depth distribution of the terrestrial sediment percentage and terrestrial mass accumulation rate in a sediment core collected from a back-reef setting near Corral Reef, La Parguera. A gradual increase in both variables is observed with depth and time, with an overall doubling of terrestrial material over the last century. Source: Ryan et al., 2008.

In addition, disease and bleaching have led to substantial deterioration of coral reefs in the past two decades. White plague II and Caribbean yellow band diseases (YBD) are the most widespread and damaging diseases affecting scleractinian corals in Puerto Rico. White plague was first reported from La Parguera in 1995 (Bruckner and Bruckner, 1997) and since 1999 has been reported with increasing frequency from both shallow and deep coral reefs across the shelf at La Parguera. Disease, particularly white band, has caused a decline in *A. palmata* since it was first reported in the early 1980s (Davis et al., 1986). For more details on monitoring programs and results for the region see data in Garcia-Sais et al. (2008). Mass bleaching events have occurred periodically in the Caribbean, when anomalously high sea surface temperatures (SST) have persisted for extended periods of time, usually in late summer and fall. At least seven mass bleaching events have been reported in Puerto Rico and the U.S. Virgin Islands (USVI) since 1985, four of which occurred between 1998 and 2006 (Figure 1.6). Assessment of the 2005 bleaching event in southwest and west Puerto Rico revealed that 65% of 4,000 corals examined exhibited bleaching, with highest occurrence of bleaching found in La Parguera (42% fully bleached, 31% partially bleached; Garcia-Sais et al., 2008). In Puerto Rico, the combined effect of the 2005 bleaching event and subsequent disease, led to a failure of sexual reproduction in *Acropora* spp. and *Montastraea* spp. in 2006 (Ballantine et al., 2008).

Bleached *Siderastrea sidera* colony



Bleached *Montastraea* sp. colony

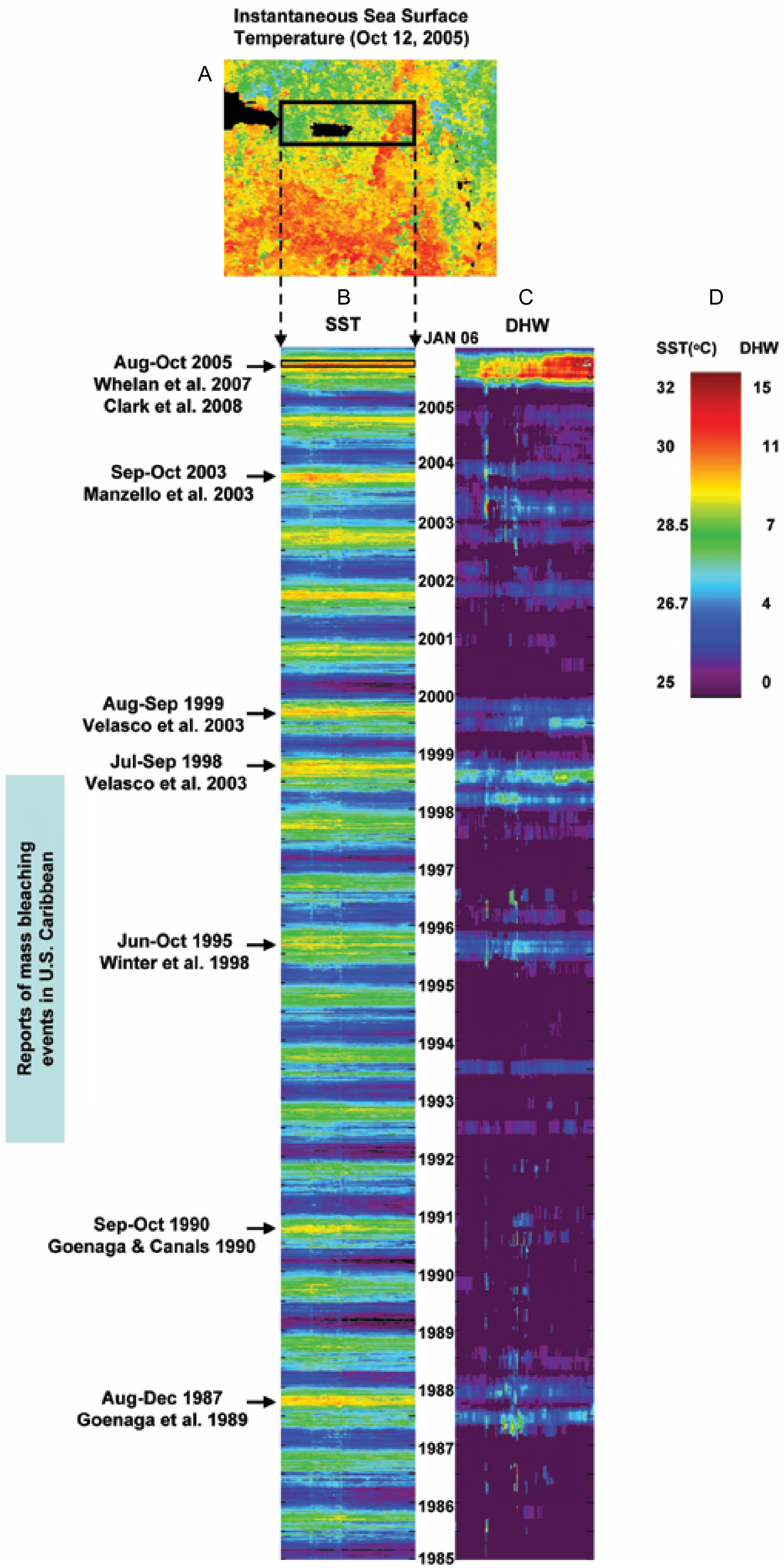


Figure 1.6. A) Shows instantaneous sea surface temperature (SST) during one of the days that coincided with the 2005 mass bleaching event recorded in the U.S. Caribbean (Clark et al., 2009). The SST data were recorded by the Advanced Very High Resolution Radiometer (AVHRR). B) A space-time chart showing the SST across the region between 1985 and 2006. Wider orange and red horizontal bands indicate greater persistence of high SST and alignment with bleaching records show that bleaching events typically occur where high water temperatures persist for weeks or months. C) A space-time chart showing degree heating weeks (DHW), a measure of cumulative thermal stress. For example, if the current temperature is above the maximum expected summertime temperature for a period of two weeks, the site would receive a rating of 2 DHWs. D) Legend for space-time charts. Source: SST data processed by Varis Ransi of NOAA CCMA Remote Sensing Group.

Reduction in the abundance of many of the key predators and herbivores on coral reef ecosystems through deterioration of habitat, disease and extraction by the fishery is likely to have a major influence on ecosystem integrity and ecological processes, but the overall ecological consequences for coral reef ecosystem structure and function in southwest Puerto Rico are not known. The integrity of coral reef ecosystems and the sustainability of commercially exploited fish species are key resource management concerns. Substantial reductions in commercial fish catches have been recorded for coral reef associated fishes over the past two decades (Matos-Caraballo, 2004; Garcia-Sais et al., 2005). Landings of demersal reef fishes peaked in 1979 at 2,400 metric tonnes (mt) then declined to 394 mt in 1988 (Appeldoorn and Myers, 1993). Spiny lobster (*Panulirus argus*) landings followed a similar trend (Bohnsack et al., 1991). In addition, the fishery has shown many of the other classic signs of overfishing which include shifts to catching smaller sized individuals and recruitment failures (Matos-Caraballo, 2004). This has occurred even though the number of active commercial fishers has decreased from 1,758 in 1996 to 1,163 in 2002. Surveys with fishers reveal that the insular shelf is the most heavily fished area, with coral reef associated fish being the primary targets. For instance, 85% of the fishers target resources on the continental shelf and 94% target “reef fishes”. Fishing pressure on the southern shelf increased from 70% in 1996 to 83% in 2002, but the fishing on the shelf edge decreased in the same period. A decrease in net use and traps has occurred and an increase in “hook-and-line” between 1996 and 2002 and most fishers now use a wide range of gears to diversify their catch. In 2002, the La Parguera region had the highest number of fishers on the south coast with a total of 26 full time and 37 part time utilizing 47 fishing vessels (Matos-Caraballo, 2004). Seventy-four percent of the fishers from the La Parguera region believe that the fishery has worsened over time with the perceived causes being overfishing (31%), pollution (13%) and habitat destruction (11%; Figure 1.7). The shallow water reef fish fishery in Puerto Rico primarily targets grunts (Haemulidae), groupers (large-bodied Serranidae), goatfish (Mullidae), parrotfish (Scaridae) and snappers (Lutjanidae; Caribbean Fishery Management Council [CFMC], 1985; see Appendix A). Triggerfish, squirrelfish, hogfish, porgies and trunkfish combined represent approximately 15% of the total catch (Appendix A).

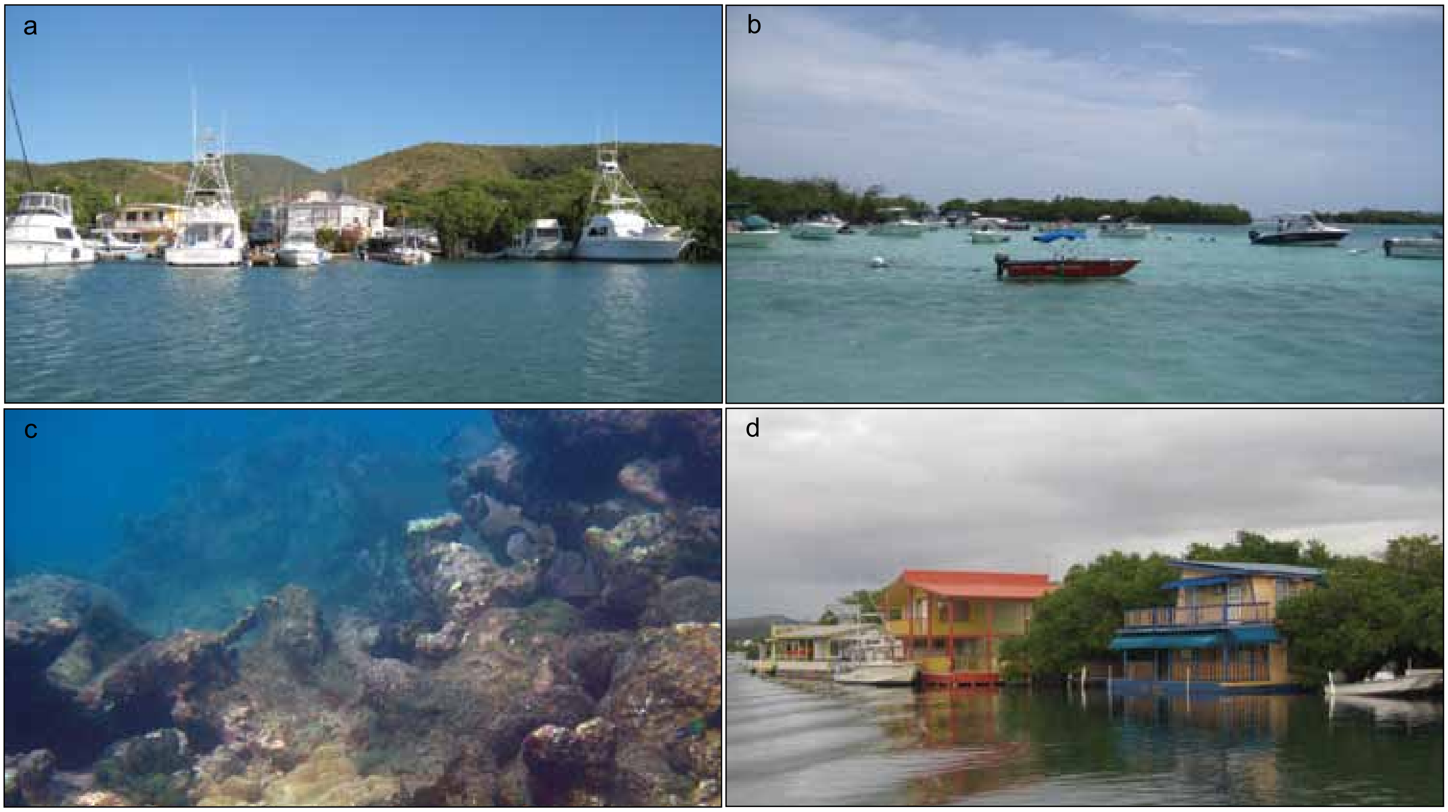


Figure 1.7. Examples of perceived causes of fishing decline according to the majority of fishers from the La Parguera region. a) fishing pressure; b) noise/recreation; c) habitat degregation, and d) habitat destruction.

A recent numerical evaluation of the sustainability of a wide range of important fishery species in Puerto Rico indicated that the majority of species are fished at unsustainable levels (Ault et al., 2008). Of the 25 fish species assessed, 16 were below the spawning potential ratio of 30%, implying that the stocks are no longer sustainable at current exploitation levels. Therefore, the Ault et al. (2008) study using length-based assessment corroborates local fishers knowledge. At time of writing, only three species (Nassau grouper, *Epinephelus striatus*, goliath grouper, *Epinephelus itajara*, and queen conch, *Strombus gigas*) were designated as overfished by NOAA's National Marine Fisheries Service (NMFS), but under the Sustainable Fisheries Act, National Standard 1 Amendments, the yellowtail snapper (*Ocyurus chrysurus*) and several large-bodied grouper including misty (*Epinephelus mystacinus*), red (*Epinephelus morio*), yellowedge (*Epinephelus flavolimbatus*), tiger (*Mycteroperca tigris*) and yellowfin (*Mycteroperca venenosa*) groupers should also be designated as overfished, while several species of parrotfish, triggerfishes and boxfishes would be considered "at risk" (Appeldoorn, 2008).

1.3. Benthic habitat mapping in southwest Puerto Rico

CCMA BB initiated benthic mapping activities in 1998 and developed a benthic habitat map with a 1 acre (4,000 m^2) minimum mapping unit (MMU) that used a hierarchical classification scheme of 21 distinct benthic habitat types within eight geomorphological zones (Figure 1.8). Habitat classes and boundaries were delineated based on visual interpretation of aerial photography and extensive ground-truthing (Kendall et al., 2001). The classification scheme included ZONE (geomorphological zone: lagoon, bank, etc.); HABITAT (broad substratum type: uncolonized sediments, submerged vegetation, colonized hardbottom, etc.); TYPE (mangrove, seagrass, linear reef, etc.); MODIFIERS (percentage cover of aquatic vegetation: patchy/continuous or <30% seagrass, etc.); and DESCRIPTORS (a combined description of HABITAT, TYPE and MODIFIER). The zone/habitat approach to classifying benthic structure was developed in consultation with CFMC, Dr. Kenyon Lindeman and CCMA BB to support multiple needs in resource management and ecosystem-based science. A relatively large "UNKNOWN" area remained unclassified due to poor

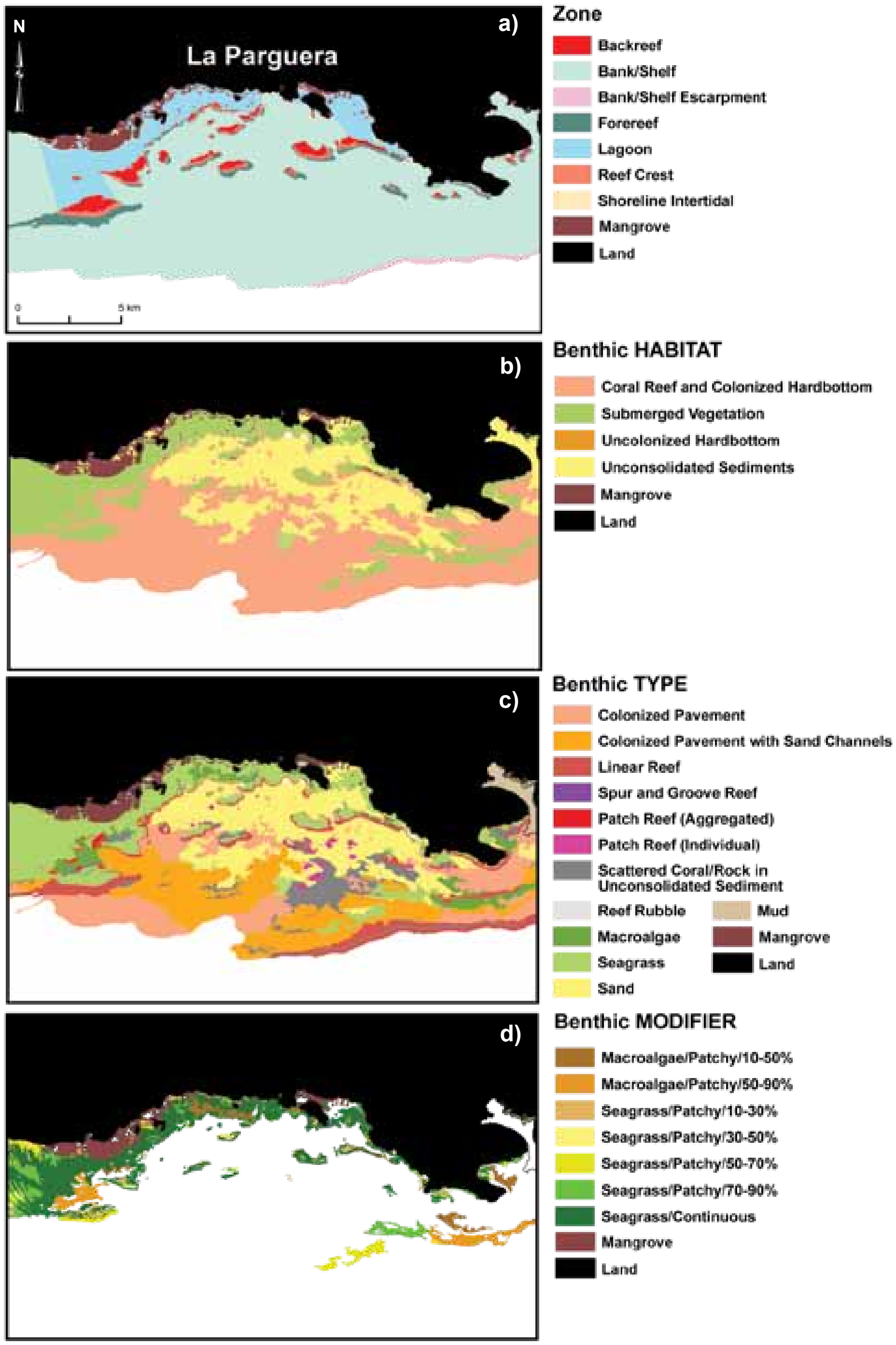


Figure 1.8. NOAA's benthic habitat map showing four levels of the hierarchical classification scheme: 1) ZONE; 2) HABITAT; 3) TYPE and 4) MODIFIER. Source: Kendall et al., 2001.

light penetration in the water column. In 2005, spectral analysis of Landsat Thematic Mapper data (CCMA-BB) and locally available side-scan sonar data (Triana, 2002) was used to classify the deeper water “unknown” area to provide a complete coverage for the La Parguera study site. The benthic habitat map includes coral reef ecosystems to a depth of approximately 35 m and therefore does not provide benthic information for the large proportion of the La Parguera NR that exists in deeper (>100 to >1,500 m) shelf edge waters extending to the 9 mile NR boundary.

In 2006, hydrographic Light Detection and Ranging (LiDAR) data were collected to map bathymetry in southwestern Puerto Rico using the ADS Mk II Airborne System (Figure 1.9). The 900 Hertz (1,064 nm) A Nd: Yag laser acquired 4x4 m spot spacing and 200% seabed coverage. In total, 265 square nautical miles of LiDAR were collected between -20 m (topographic) up to 50 m (depth). This data provided a bathymetric surface and reflectivity surface. More details on the methods can be found via online metadata files (http://ccma.nos.noaa.gov/products/biogeography/lidar_pr/welcome.html).

To highlight the changes in surface complexity across the region we measured surface rugosity as the ratio between the horizontal surface and the actual convoluted 3-dimensional surface bathymetry. Areas with high change in surface complexity occured at reef and shelf edges and where clusters of patch reefs existed and spur and groove formations. Relatively high topographic complexity (orange/red in Figure 1.9) was found along the shelf edge, the fringing coral reefs of the backreef zone (innershelf reefs) and linear midshelf reefs, the highly heterogeneous patch reefs at El Palo and north of Margarita Reef (Figure 1.9) and the large expanse of colonized pavement with sand channels extending east-west parallel to the shelf edge. This study and earlier analyses have revealed that measures of surface complexity are useful and cost-effective predictors of the distribution of fish species, fish diversity and biomass and coral diversity and abundance (Pittman et al., 2009).

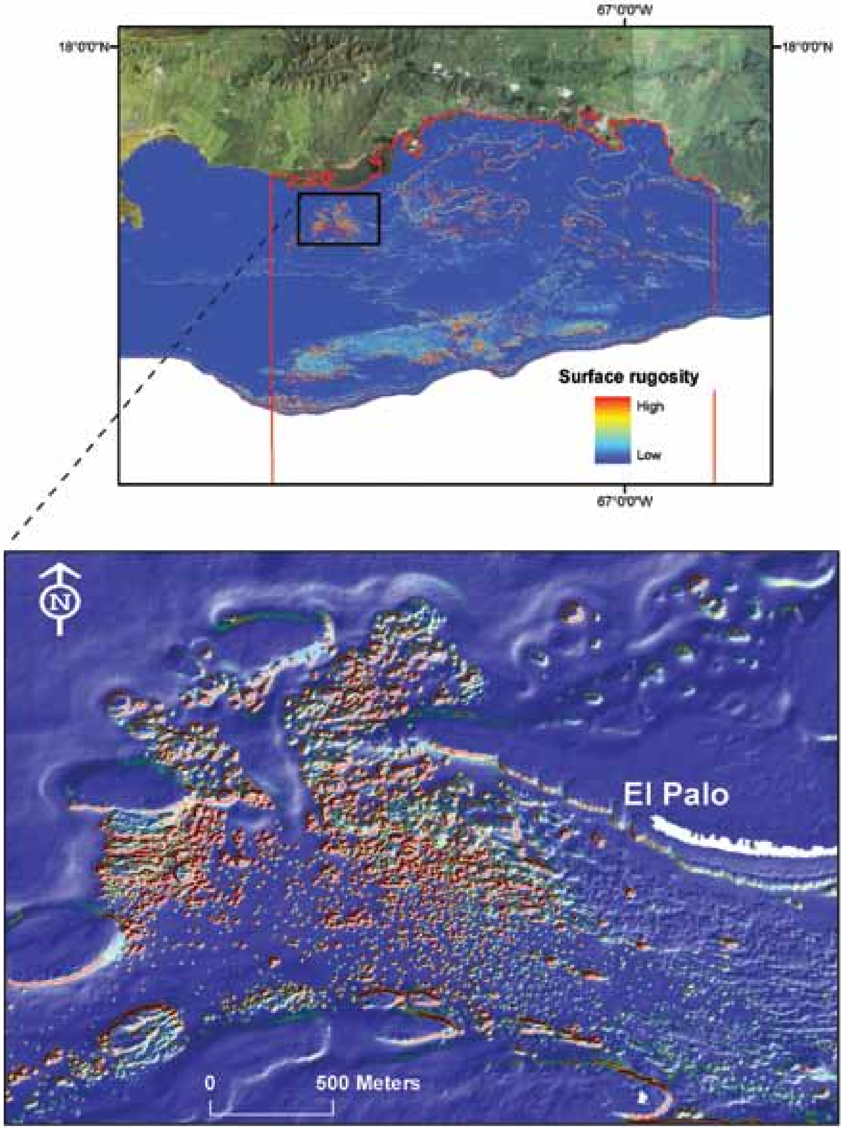


Figure 1.9. Surface rugosity calculated from the LiDAR bathymetry to show the pattern of topographic variability across the study region. A distinctive area with high rugosity created through the clustering of linear and patch reefs exists between El Palo and Margarita Reefs. The La Parguera Natural Reserve encompasses almost all of the shallow water high complexity coral reef in the region.

1.4. References

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Chapter 2. Benthic Composition

2.1. Introduction

Coral reef ecosystems in Puerto Rico comprise a system of connected mangrove forests, seagrass beds, unconsolidated sediments and coral reefs and hardbottom substrates. These habitats provide shelter and sustenance to fishes and invertebrates that form the basis of important fisheries in the region. A major goal of CCMA's Caribbean Coral Reef Ecosystem Monitoring (CREM) project is to characterize benthic composition and correlate such information to spatial and temporal patterns in the distribution of fish and invertebrate populations. Benthic characterizations provide the basis for identifying species-habitat relationships, increasing understanding of spatial patterns in the distributions of habitats, and illustrating important and crucial linkages for the successful management of coral reef fisheries and other important resources. This chapter provides baseline estimates of benthic substrate composition of hardbottom, seagrass, macroalgae, mangrove and unconsolidated habitats in La Parguera, Puerto Rico as defined by Kendall et al. (2001). More specifically, data are presented to characterize the types, distributions and percent cover of benthic flora and fauna within mapped substrates.

Figure 2.1. NOAA diver recording benthic habitat composition within the randomly placed 1 m^2 quadrat along the belt transect.

2.2. Methods

2.2.1. Survey Data

Underwater visual surveys were conducted for benthic composition data collection within the La Parguera study area in southwest Puerto Rico. At each site, benthic data were collected from five 1 m^2 quadrats randomly placed along a 25 m belt transect used for fish census (Figure 2.1). For detailed methods, see Appendix D.

Table 2.1 provides a list of measured benthic composition variables and types of measurements collected. In addition to abiotic and biotic benthic cover, data were collected on queen conch (*Eustrombus gigas*), long-spined sea urchin (*Diadema antillarum*), Caribbean spiny lobster (*Panulirus argus*) and marine debris. Collection of abundance and distribution data on these selected macro invertebrates began in 2004, with their abundance being recorded only if individual(s) were observed within the 4 x 25 m transect (Appendix B). Marine debris data collection began in January 2007 (Figure 2.2). Between 2001 and 2004, benthic habitats were categorized according to the habitat types defined by (Kendall et al., 2001; Figure 2.3) for sampling allocation

Figure 2.2. Marine debris.

Table 2.1. Abiotic and biological variables measured to characterize benthic assemblages along fish transects in the southwest Puerto Rico study region.

	Measurements		
Benthic Biota	**Cover (%)**	**Height (cm)**	**Abund. (#)**
Abiotic			
Hardbottom	X	X	
Sand	X		
Rubble	X		
Fine sediment	X		
Rugosity			
Water depth			
Biotic			
Corals (by species)	X		
Macroalgae	X	X	
Seagrass (by species)	X	X	
Gorgonians			
Sea rods, whips and plumes	X	X	X
Sea fans	X	X	X
Encrusting form	X		
Sponges			
Barrel, tubes, rope, vase morph.	X	X	X
Encrusting morphology	X		
Other benthic macrofauna			
Anemonies and hydroids	X		X
Tunicates and zoanthids	X		
Macro-invertebrates			
Queen conch (by sexual maturity)			X
Spiny lobster			X
Long-spined urchin			X
Mangroves			
Prop roots			X
Prop roots colonized by algae			X
Prop roots colonized by sponges			X
Prop roots colonized by other			X

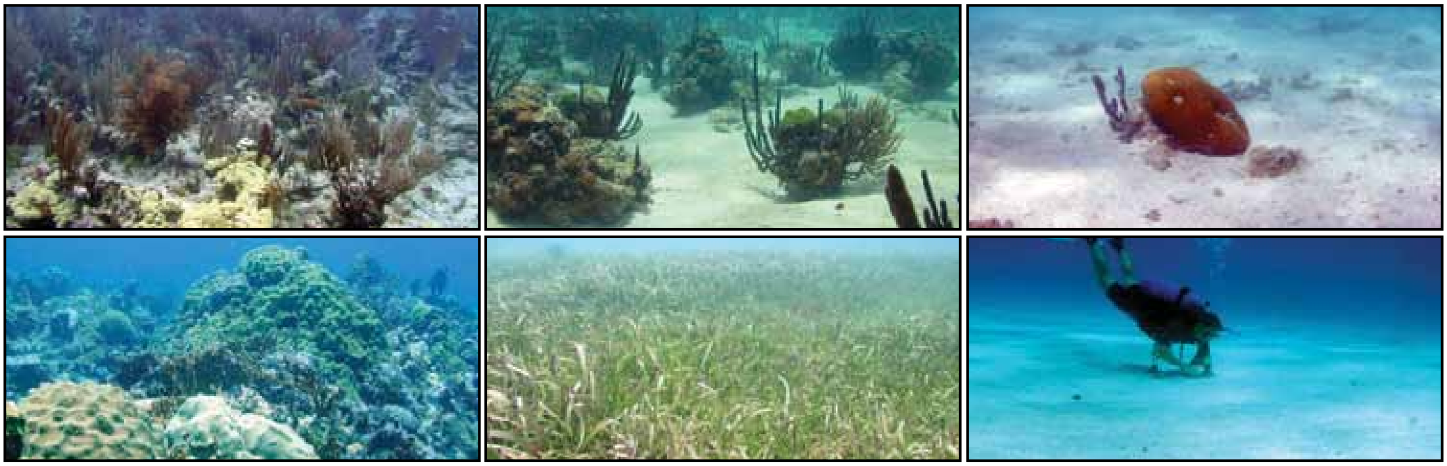


Figure 2.3. A selection of habitat types designated in the hierarchical classification scheme of NOAA's benthic habitat map (Kendall et al., 2001) for the U.S. Caribbean (clockwise from left to right): colonized pavement, patch reef, scattered coral and/or rock, linear reef, seagrass and sand.

and design. Between 2004 and 2007 however, habitats were re-categorized into three habitat types (hard, soft and mangrove) for sampling and data collection.

2.2.2. Analytical Methods

In situ data on the cover of benthic biota were summarized from 1,167 surveys (approximately 5,835 quadrats) during 2001 to 2007 (Tables 2.2 and 2.3). Surveyed sites were located within four broad thematic habitat types in the study area: colonized coral reef and hardbottom areas (hereafter hardbottom habitats; n=572), seagrass and algal communities (hereafter submerged aquatic vegetation or SAV; n=272), unconsolidated sediments (sand and mud habitats; n=91) and mangroves (n=158). An additional 74 sites that were sampled were mapped as unknown habitat (Table 2.2). These sites had no biotic cover and were excluded from subsequent analyses. Note that during any single mission, the number of surveys conducted within hardbottom types varied and was relatively low for the least abundant habitat types (colonized bedrock and reef rubble) but high for more abundant habitats such as colonized

Table 2.2. Number of hardbottom benthic habitat sites surveyed in the southwest Puerto Rico study area by mission and mapped habitat type from 2001-2007. Mapped habitat categories are from Kendall et al. (2001).

Sample period	Colonized Pavement	Col. pav. w/ sand channels	Linear reef	Patch reef	Scattered coral/rock in sand	Bedrock/ reef rubble	Sand/ mud	Seagrass/ macroalgae	Unknown*	Mangrove	Total
2001 Winter	1	2	2	3	1		1	11	1	7	29
2001 Spring	2	3	2	2	1		8	13		7	38
2001 Fall	2	4	1		3		3	16		6	35
2002 Winter		4	5		2	1	5	28	3	9	57
2002 Summer	3	3	7	3	3		4	12	1	9	45
2003 Winter	5	7	6	6	5		8	22	1	15	75
2003 Spring	4	13	9	5	11		12	22	1	13	90
2003 Fall	5	13	6	7	12		8	26		13	90
2004 Spring	11	14	4	7	6		9	26		13	90
2004 Summer	10	16	4	5	10		11	20		13	89
2005 Winter	7	18	7	4	7	1	14	19		13	90
2005 Summer	10	23	12	2	6		2	10	15		80
2006 Winter	12	23	8		8		1	11	16	10	89
2006 Summer	14	22	8	2	6	1	2	14	11	10	90
2007 Winter	14	22	9	2	12			10	11	10	90
2007 Summer	10	25	10	5	1		3	12	14	10	90
Total	**110**	**212**	**100**	**53**	**94**	**3**	**91**	**272**	**74**	**158**	**1,167**

**Unknown refers to benthic substrates that were unmappable by Kendall et al. (2001) because of turbid conditions.*

pavement and patch reefs (Table 2.3). Hence observed differences in benthic composition or the lack thereof among hardbottom habitat types should be interpreted with caution, given that mean estimates of metrics from least abundant habitats were more variable compared with estimates from more abundant habitats.

Table 2.3. The number of benthic habitat sites surveyed by mapped habitat type for the southwestern Puerto Rico study region from 2001-2007. Mapped habitat categories are from Kendall et al. (2001).

Mapped habitat types	Mapped Area (km^2)	% Mapped Area	# sites surveyed	Area surveyed (km^2)	% area surveyed
Colonized Bedrock	212.76	0.2%	2	0.2	0.09
Colonized Pavement	18,505.23	16.5%	110	11	0.06
Col. Pavement w/ Sand Channels	32,058.04	28.6%	212	21.2	0.07
Linear Reef	11,306.15	10.1%	100	10	0.09
Patch Reefs (Aggregate & Individual)	2,738.33	2.4%	53	5.3	0.19
Reef Rubble	270.98	0.2%	1	0.1	0.04
Scattered Coral/Rock in sand	10,345.36	9.2%	94	9.4	0.09
Unconsolidated sediments	6,620.71	5.9%	91	9.1	0.14
Macroalgae	5,160.26	4.6%	40	4	0.08
Seagrass	24,692.53	22.1%	234	23.4	0.09
Total	**111,910.3624**	**100%**	**937**	**93.7**	**0.08**

Many benthic variables were measured during the surveys, but data analyses for this report focused primarily on describing broad-scale spatial patterns and temporal trends in the area abundance (percent cover) of the sessile biotic components as described in Table 2.1. Specifically, data were analyzed to examine the following: 1) benthic habitat composition of broad thematic habitat types and more resolved hardbottom habitat types; 2) broad-scale seascape patterns in cover of live coral, macro algae and seagrasses; and 3) temporal trends in live scleractinian (hard) coral and algal cover.

2.2.2.1. Characterizing spatial distributions of benthic biotic components among habitats

Estimates of percent cover (mean ± standard error [SE]) of selected benthic biota were calculated for each site. Sites were used as independent sample units and were considered replicates within survey missions and habitat types. Multiple quadrat measurements (percent cover) for biota within each transect were averaged using the equation:

$$\Sigma(Q_i - n) / n$$

where Q_i = quadrat i, and n is the total number of quadrats. Average site values were then used to calculate means and SE of measured variables per 100 m^2 for each habitat type. Standard errors of means represent variability among sites rather than variability among quadrats within a site. Differences in the cover of benthic biota among habitat types were determined by using a series of One-Way non-parametric ANOVA (Wilcoxon) tests to identify significant differences among habitat types (Zar, 1999). When significant differences were found, non-parametric multiple pair-wise comparisons were used to determine the pairs of habitat types that were significantly different (Zar, 1999).

2.2.2.2. Characterizing seascape spatial patterns

Site values of benthic community metrics averaged from quadrat data were interpolated and mapped using inverse distance weighting (IDW) to guide interpretation of spatial patterns within the broader seascape of the study region. IDW is a method of interpolation that estimates values by averaging the values of sample data points in a predetermined neighborhood. In this case, the interpolation used a minimum of 10 neighbors. Only hardbottom sites were used to map typical hardbottom features such as coral, gorgonian, and sponge percent cover. Similarly, only softbottom sites were used to map seagrass percent cover. Some caution should be taken with resulting mapped patterns especially in areas of low point abundance and at the edges of the study area.

2.2.2.3. Characterizing temporal trends in live coral and algal cover

Characterization of temporal trends in benthic composition metrics were based on data collected from 2001 to 2007. Survey missions were categorized as occurring during winter or spring (December through June), summer (August), or fall (September through October). Average site values were then used to calculate means and SE of measured variables per 100 m^2 for each survey mission. Standard errors of means represent variability among sites rather than variability among quadrats within a site. Temporal trends in mean estimates of live coral and algal cover were determined by using two statistical approaches. First, a series of One-Way non-parametric ANOVA (Wilcoxon) tests were done to determine if significant differences occurred among the sampling periods, and non-parametric multiple pair-wise comparisons were used to identify pairs of sampling periods that were significantly different. Second, overall temporal trends in mean estimates of live coral and algal cover were determined by using the nonparametric Jonckheere Test (JT) for Ordered Alternatives to examine whether or not significant change occurred in percent coral cover and algae between 2001 and 2007. The JT statistical procedure assumed that there were no differences in coral and algae cover among sampling periods and tested against a postulated sequential increase or decrease in those benthic metrics across sampling periods. The JT test also assumed that estimates of means were derived from random independent samples and that estimates of variance were homogeneous across sampling periods and the frequency distribution of data was similar among periods. To calculate the test statistic, the k(k-1)/2 Mann-Whitney U counts were derived with the following equation (Wolfe and Hollander, 1999):

$$U\mu v = \sum_{i=1}^{n\mu} \sum_{j=1}^{nv} \phi(Xiu, Xjv), 1 \le \mu < v \le k$$

Where Ø$_{(a,b)=1}$ if a < b, or 0 if otherwise. The Jonckheere test Statistic (J) was then calculated as the sum of these U counts and was compared against a significance threshold ($J_{\alpha = 0.05}$) that was dependent on the number of sampling periods and number of sites surveyed within each sampling period (Wolfe and Hollander 1999). If $J_{metric} > J_{\alpha} = {}_{0.05}$, we concluded that estimates of mean cover were not equal across sampling periods, and that there was an overall sequential increase or decrease in mean estimates during the study period. Additionally, residual plots of coral and algae percent cover were examined to determine if variance estimates were homogenous and comparable over time.

Montastraea annularis complex and gorgonians

2.3. Results

2.3.1. Benthic habitat composition

Estimates of percent cover (mean ± SE) of selected benthic organisms are reported for four habitat types observed in the study area: hardbottom habitats, SAV communities, sand and mud habitats, and mangroves. Data are presented at these broader thematic habitat resolutions because the number of surveyed sites varied among the more resolved habitat types and was extremely low for rare habitats such as bedrock and reef rubble (Table 2.3). Comparisons presented here are intended to characterize broad differences in benthic composition among the four habitat types.

2.3.1.1. Characterization of colonized hardbottom habitat types

Pavement habitat was the most spatially extensive habitat type (45.1% of the study area) and was therefore the most intensively surveyed, followed by seagrass (22.1%) and linear reefs (10.1%; Table 2.2). The remainder of the area surveyed comprised of scattered coral and rock in sand, unconsolidated sediments, macro algae, patch reefs, reef rubble and bedrock (Table 2.2); Hardbottom habitats combined formed a larger proportion (67.4%) of the study region than did soft bottom areas.

Overall, mean benthic cover on hardbottom habitats was 60.5 ± 1.3%. Generally, hardbottom habitat types were dominated by algae (27.0 ± 1.0% "turf-like" algae [turf algae], 14.0 ± 0.7% macroalgae, and 1.3 ± 0.3% crustose coralline algae [CCA]; Figure 2.4). The next most abundant benthic group was the gorgonians (soft corals), which had an average percent cover of 7.4 ± 0.4% (Figure 2.4). Mean live scleractinian coral cover averaged 5.3 ± 0.3% across the study area. Cyanobacteria and filamentous algae were grouped as a single component and had a mean cover of 1.4 ± 0.2%. Other benthic organisms observed on hardbottom habitats included sponges (2.3 ± 0.1%), seagrasses (1.7 ± 0.3%) and hydroids such as fire coral (0.3 ± 0.1%; Figure 2.4).

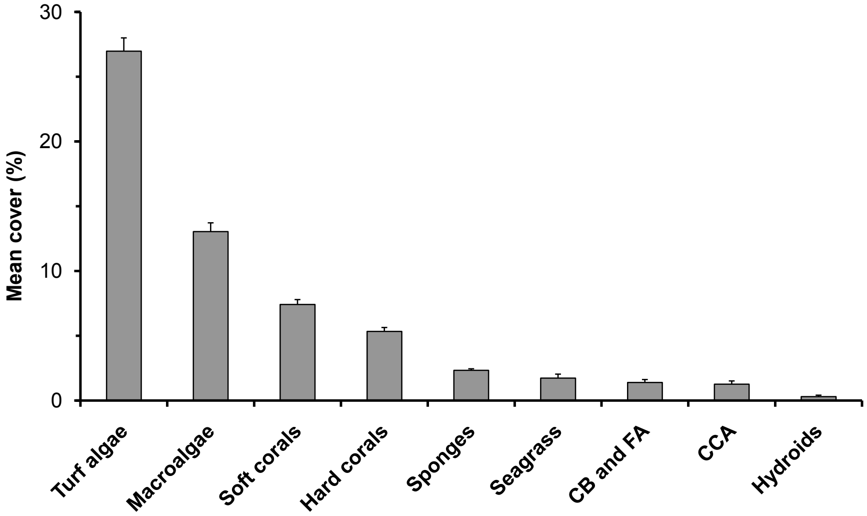


Figure 2.4. Mean (± SE) percent cover for key benthic components on hardbottom sites (n=572) in the southwest Puerto Rico study region from 2001-2007. CCA=crustose coralline algae; CB and FA=cyanobacteria and filamentous algae).

Benthic composition was variable among the hardbottom habitat types surveyed (Figure 2.5). The mean percent cover of live scleractinian coral was highest on linear reef (6.8 ± 0.7%, n=100), followed by colonized pavement (6.1 ± 0.4%), patch reef (4.8 ± 0.7%), but was lowest on reef rubble (1.2 ± 0%, n=1; Figure 2.5). Fire corals were the most commonly recorded hydroid species on pavement, linear reef, and patch reef habitat types, with highest percent cover occurring

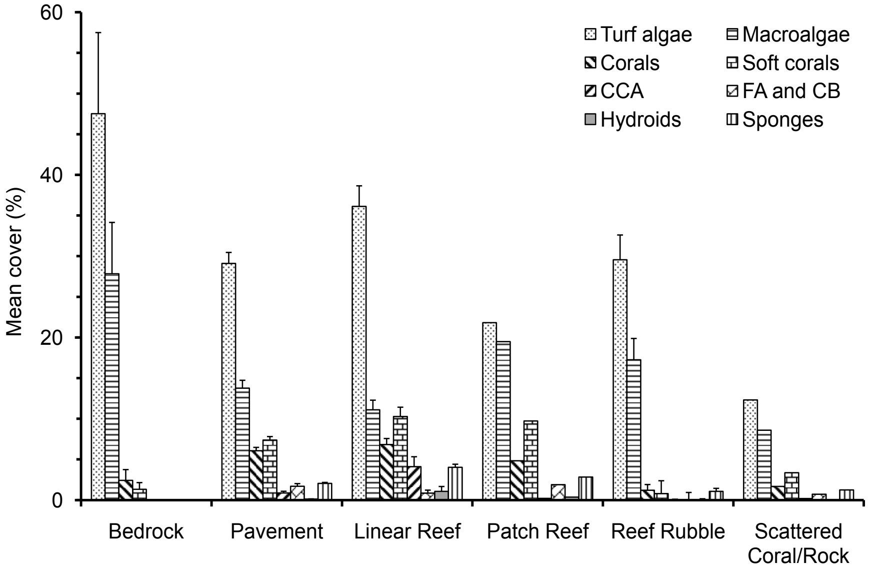
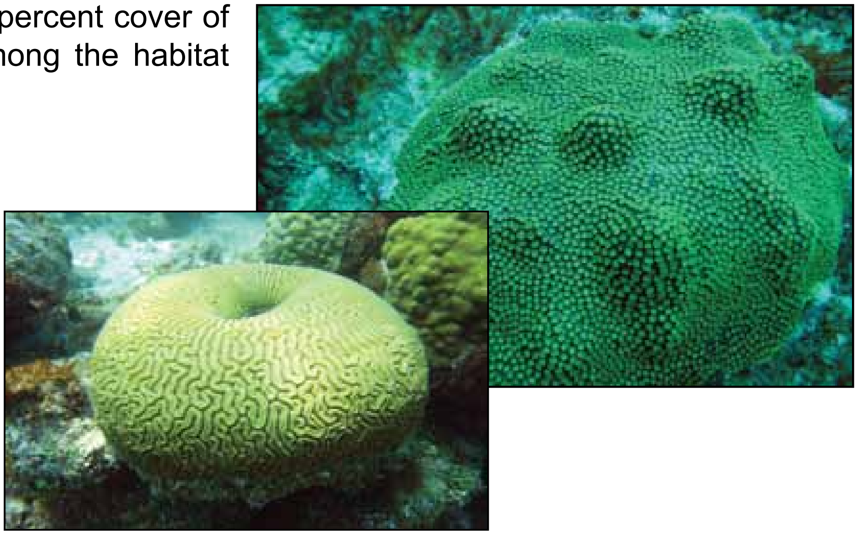


Figure 2.5. Vertical bars represent mean (± SE) percent cover for Key components of benthic community on hardbottom habitat types (n=572) in the southwest Puerto Rico study region from 2001-2007. Ovals represent global mean (± SE) cover of key components from all hardbottom habitats combined.

on linear reef habitats (1.1 ± 0.6%). The percent cover of sponges and fire corals were similar among the habitat types surveyed (Figure 2.5).


Diploria labyrintheformes (left) and *Montastraea annularis* complex (right)

Live scleractinian coral cover included at least 24 coral genera, but only nine with a mean cover greater than 0.01% (Figure 2.6). The two coral genera with the highest mean cover were *Montastraea* spp. (2.3 ± 1.9%) and *Porites* spp. (0.8 ± 0.1%). Three genera (*Siderastrea* spp., *Diploria* spp. and *Agaricia* spp.) had the next highest mean cover (0.4 ± 0.0%) on hardbottom throughout the study region. *Acropora* corals had a mean cover of 0.2 ± 0.1%, (Figures 2.6 and 2.7). Boulder brain coral (*Colphophyllia natans*) was not as commonly observed and had a mean cover of 0.2 ± 0.1%). Other genera observed on hardbottom habitats were rare and included *Cladocora, Dendrogyra, Dichocoenia, Eusmilia, Favia, Helioceris, Isophyllastrea, Isophyllia, Madracis, Manicina, Mycetophyllia, Oculina, Mussa, Scolymia, Solenastrea* and *Stephanocoenia*. These corals had a combined mean cover of 0.6 ± 0.1% (Figure 2.6).

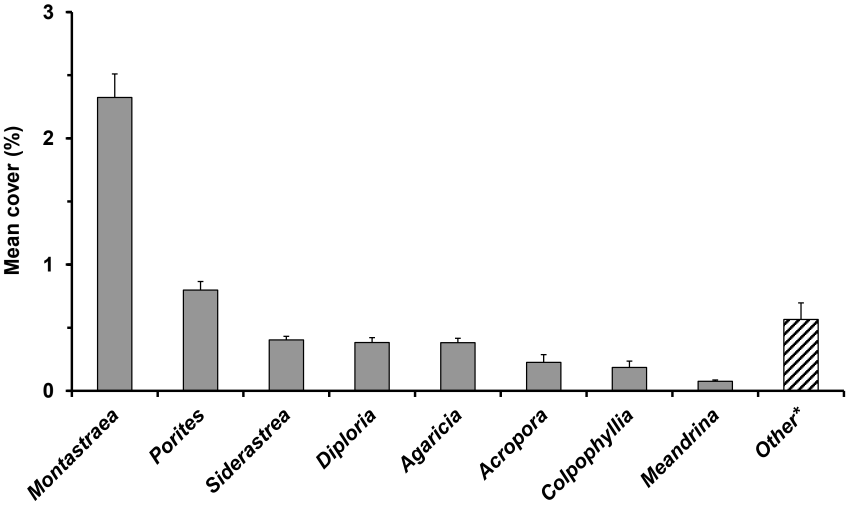


Figure 2.6. Aerial mean (± SE) percent cover of coral genera found across hardbottom sites in the southwest Puerto Rico study region from 2001-2007. Asterisks (*) indicates the other genus: Claocora, Dendrogyra, Dichocoenia, Eusmilia, Favia, Heloceris, Isophyllastrea, Isophyllia, Madracis, Manicina, Mycetophyllia, Oculina, Musa, Scolymia, Solenastrea *and* Stephanocoenia.

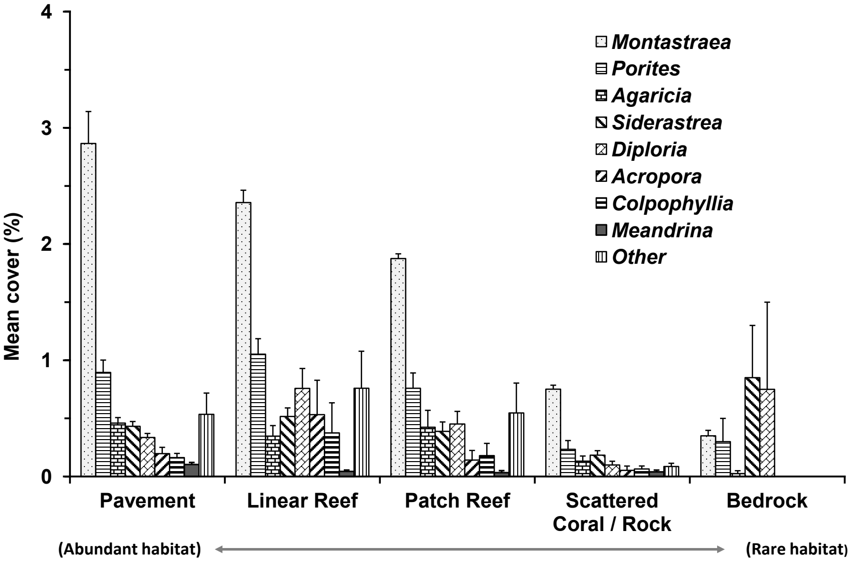


Figure 2.7. Mean (± SE) percent cover of coral genera by hardbottom habitat type in the southwest Puerto Rico study region from 2001-2007.

The distribution of coral genera among hardbottom habitats reflected the overall spatial patterns of coral distribution observed on all hardbottom habitats combined. *Montastraea* and *Porites* species were the most ubiquitous corals, with the highest mean coral cover in the most abundant hardbottom habitats: pavement, linear reef and patch reefs habitats (Figure 2.6). Additionally, *Montastraea* and *Porites* were the genera with the most coral cover for all hardbottom habitats except for colonized bedrock, where only two sites were surveyed. Acroporid coral cover generally was low, but was highest on linear reef habitat (Figure 2.7). Other coral genera such as *Diploria*, *Siderastrea* and *Colpophyllia* appeared to have similarly low cover in both abundant and rare hardbottom habitats (Figure 2.7). Interestingly, several genera were not observed on colonized bedrock habitats, but this pattern may have reflected the low number of colonized bedrock sites surveyed (n=2).

2.3.1.2. Characterization of submerged aquatic vegetation (SAV) habitat types

The benthic organisms observed in SAV habitats had total cover of 48.4 ± 2%. Seagrasses had the highest mean cover (28.1 ± 1.8%) in habitats classified as SAV followed by macroalgae (12.4 ± 0.9%; Figure 2.8). Four seagrass species were observed, of which *Thalassia testudinum* (turtle grass) had the most cover (21.2 ± 1.7%; Figure 2.9). Some turf algae (4.2 ± 0.8% cover) was observed in SAV habitat, most likely colonizing fragments of hard substrates that are commonly observed in seagrass beds. Cyanobacteria and unidentified filamentous algae (2.3 ± 0.6%) were also observed colonizing seagrasses, macroalgae, and patches of hardbottom substrates encountered in SAV habitats. Other organisms found inhabiting SAV habitats included sponges, gorgonians, corals, CCA algae and hydroids such as fire corals. These organisms were rare with mean estimates of cover less than 0.4 ± 0.1%.

2.3.1.3. Characterization of unconsolidated sediment types

Overall, the total benthic cover on unconsolidated sediments was low (18.5 ± 2.6%). Most of the cover observed on this habitat type was turf algae (6 ± 1.6%), followed by seagrass (5.7 ± 1.4%), and macroalgae (4.6 ± 0.7%; Figure 2.10). Hard and soft corals, sponges, crustose coralline algae, cyanobacteria and filamentous algae were also observed, but their mean cover was less than 0.7 ± 0.3%. These organisms were often encountered on small patches of hard substrate that often occurred within unconsolidated sediment habitats. Unlike seagrass habitats in which *T. testudinum* dominated, *Syringodium filiforme*[A]

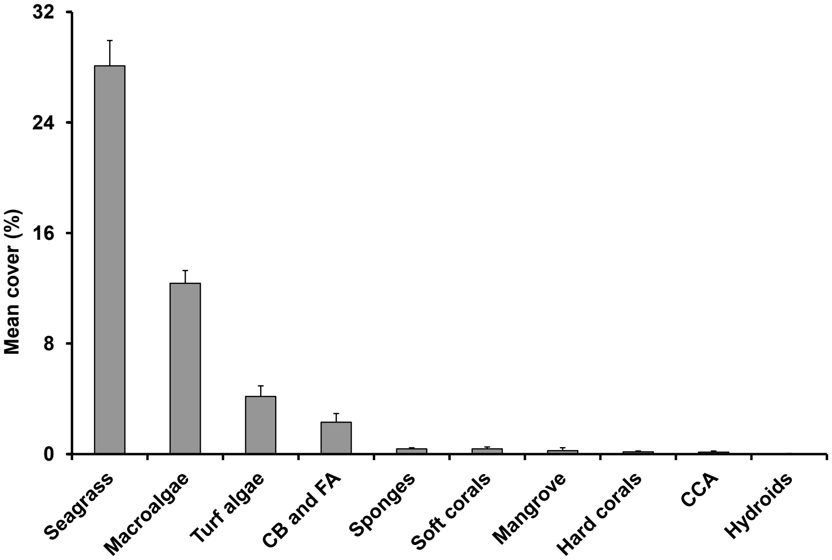


Figure 2.8. Mean (± SE) percent cover of key benthic components on submerged aquatic vegetation (SAV) sites (n=272) in the study region from 2001-2007. CCA=crustose coralline algae; CB and FA=cyanobacteria and filamentous algae.

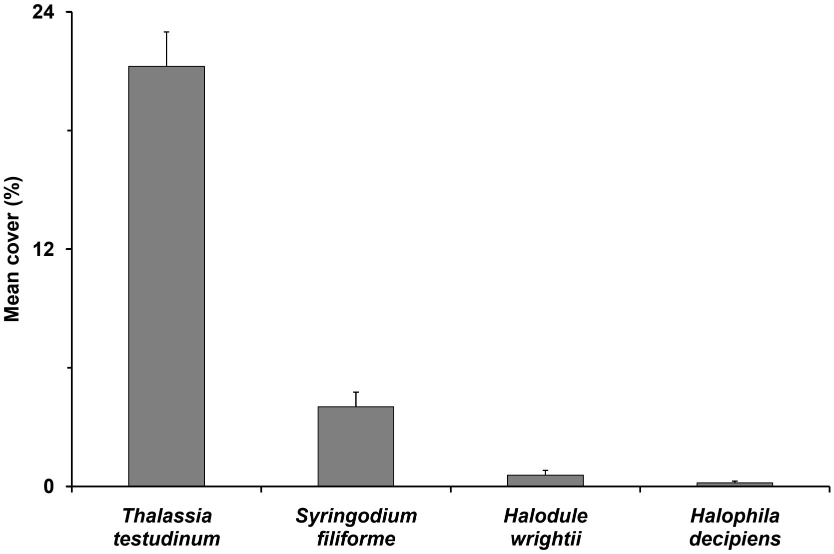


Figure 2.9. Mean (± SE) percent cover of seagrass species observed on SAV sites (n=272) in the southwest Puerto Rico study region from 2001-2007.[A]

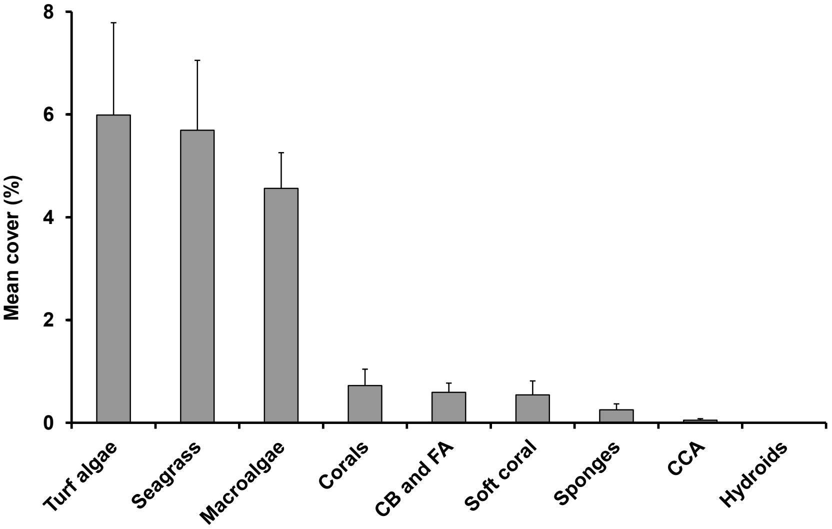


Figure 2.10. Mean (± SE) percent cover of key benthic components on unconsolidated sediment sites (n=91) in the study region from 2001-2007. CCA= crustose coralline algae; CB and FA=cyanobacteria and filamentous algae.

[A] The taxonomic name for *Halodule wrightii* has been recently changed to *Halodule beaudettei* and *Syringodium filiforme* has recently changed to *Syringodium isoetifolium*.

(manatee grass) was the most dominant of the seagrass species in unconsolidated sediments habitats, with a mean cover of 3.1 ± 1.0% (Figure 2.11).

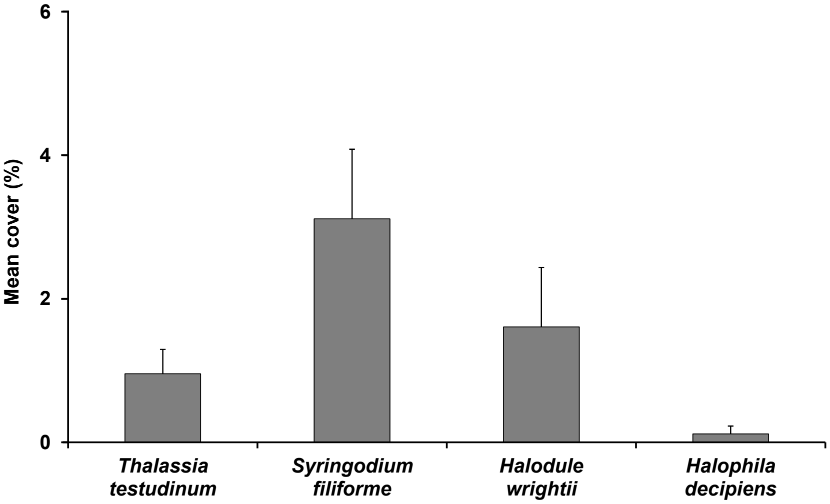


Figure 2.11. Mean (± SE) percent cover of seagrass species observed on unconsolidated sediment sites (n=91) in the southwest Puerto Rico study region from 2001-2007.

2.3.1.4. Characterization of benthic substrates in mangrove habitats

Benthic substrates within mangrove habitats were colonized by benthic organisms and had an overall mean benthic cover of 54.9 ± 2.7%. At many sites, mangrove prop roots provided a major benthic substrate for colonization by epiphytic organisms. Macroalgae had the highest benthic cover (14.5 ± 1.5%) and was the most dominant sub-aquatic benthic organism identified on mangrove prop roots and abiotic benthic substrates (Figure 2.12). One commonly encountered feature of benthic substrates in mangrove habitats was a thick layer of detritus, which had a mean cover of 7.8 ± 1.2% (Figure 2.12). Other organisms found colonizing mangrove habitats included sponges, hard and soft corals, CCA and hydroids such as fire corals, turf algae, seagrasses, cyanobacteria and filamentous algae.

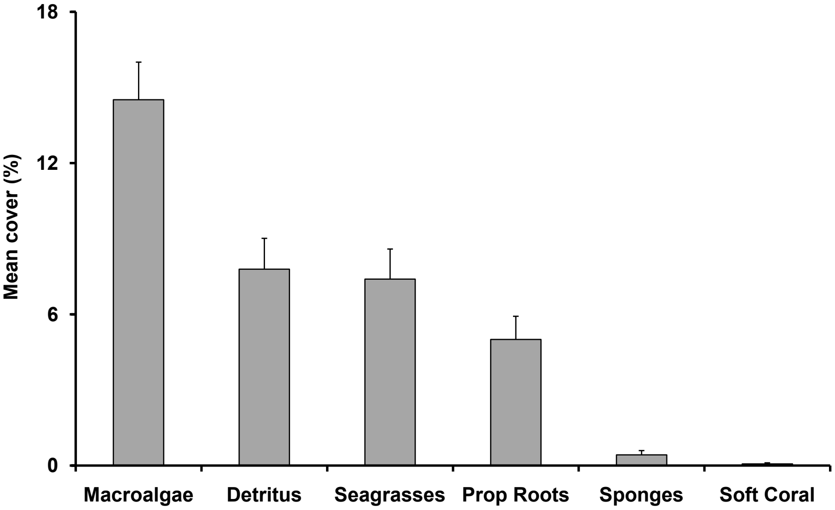


Figure 2.12. Mean (± SE) percent cover of key benthic components on benthic substrates at mangrove sites (n=158) in the southwest Puerto Rico study region from 2001-2007.

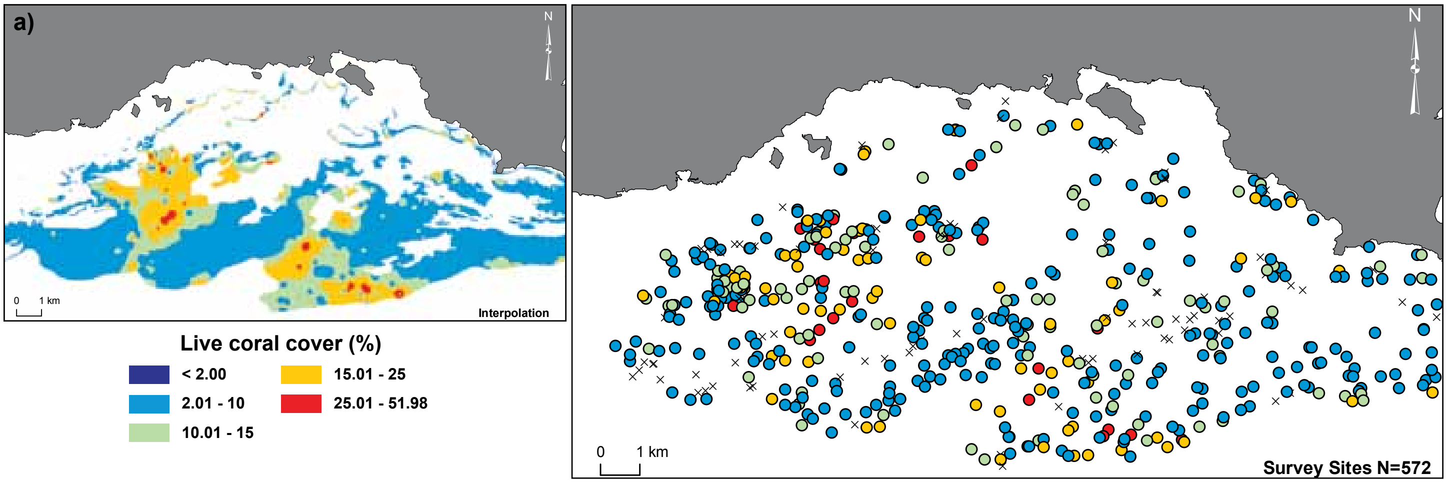
2.3.2. Spatial distribution patterns in benthic cover

The following sections describe broad spatial patterns in metrics for live corals, algae, gorgonians, sponges and seagrasses that were derived from interpolations of percent cover data. Maps of interpolated distributions are useful in that they help elucidate broad-scale patterns (e.g., the degree of patchiness and location of hotspots) in the seascape that are not discernible from point-data. Interpolations for corals and rugosity were confined to hardbottom areas leaving softbottom areas as a white space in the interpolated maps. Several distinct spatial patterns were observed and are described below.

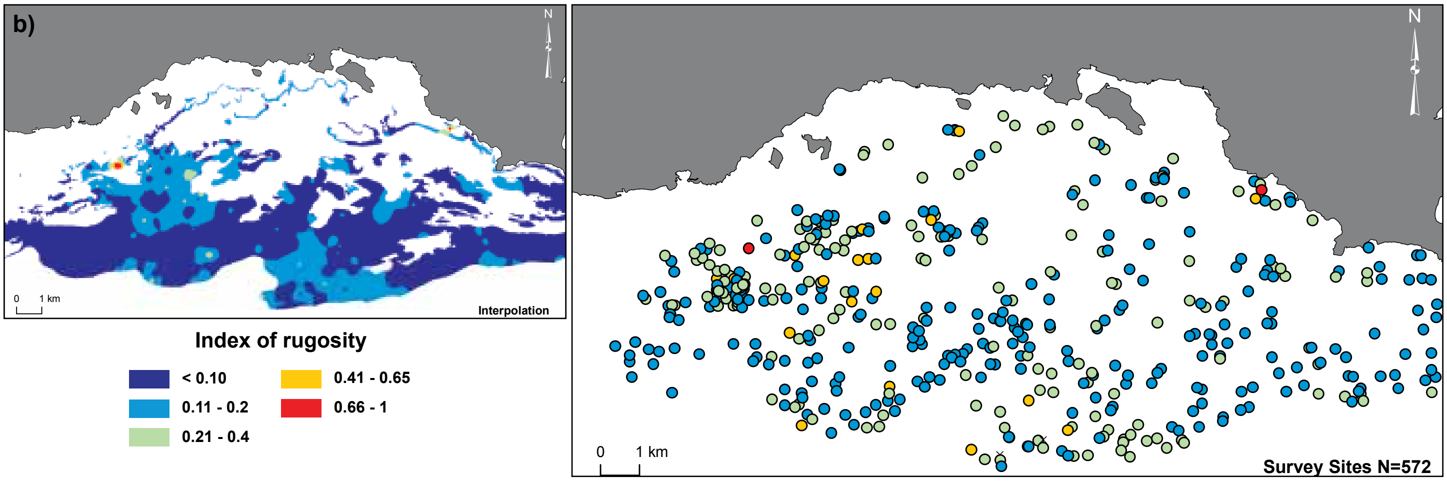
2.3.2.1. Spatial patterns in live coral cover


Diploria strigosa

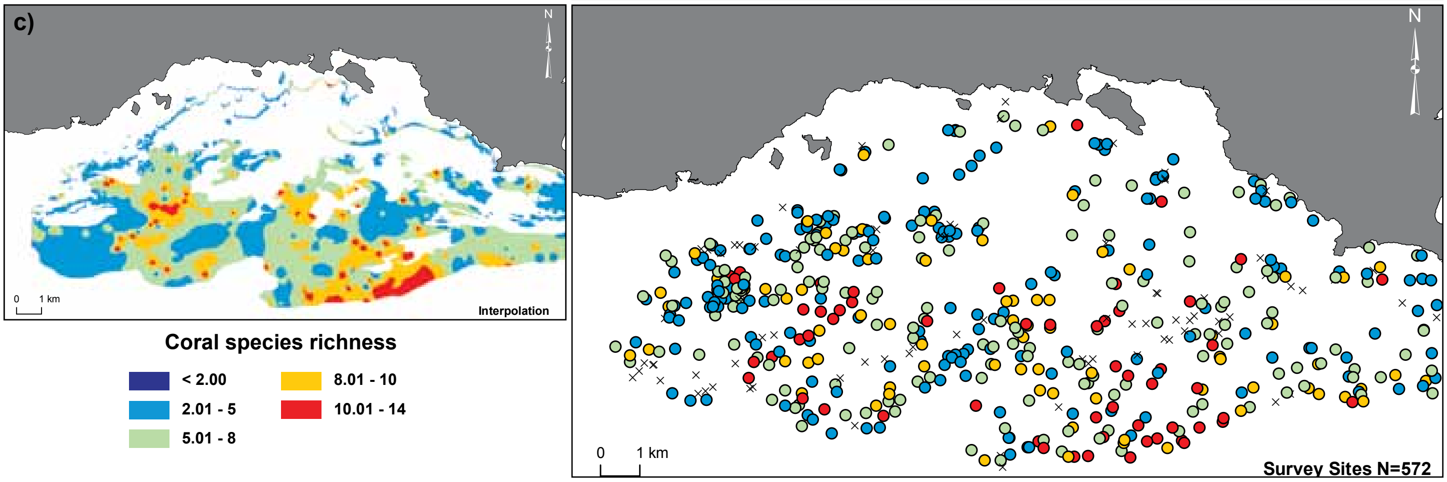
Interpolated live coral data reveal patchy areas of higher (10-15%) live coral cover along the nearshore fringing reefs in the center of the study area (Figure 2.13a). More robust areas of high coral cover were located in the western portion of the study area with cover ranging from 10-37%. Two smaller patches of high live coral cover were evident on the bank shelf with cover ranging from 10-52%. These areas were associated with increased benthic rugosity (Figure 2.13b) and also featured higher coral species richness



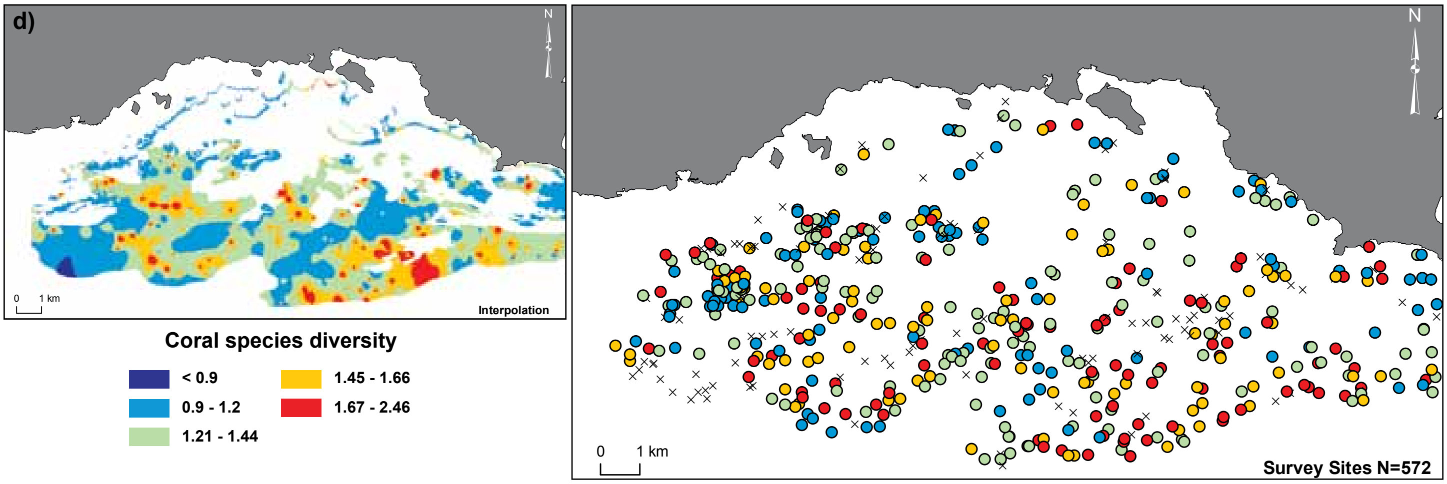
Live coral cover (%)



Index of rugosity



Coral species richness



Coral species diversity

Figure 2.13. Maps of the interpolated (left map) and spatial (right map) distributions for: (a) live percent coral cover, (b) benthic rugosity, (c) coral species richness and (d) coral species diversity.

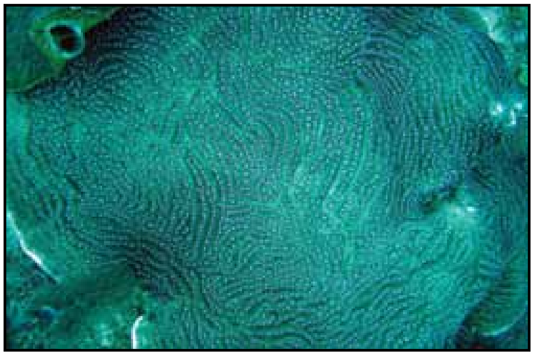
Table 2.4. Summary statistics for hard coral species found in surveys in the southwest Puerto Rico study area from 2001-2007.

Species	**Measurements**	
	Frequency occurance	**% cover** Mean ± **SE**
Acropora cervicornis	0.11	0.18 (0.06)
Acropora palmata	0.01	0.04 (0.02)
Agaricia agaricites	0.13	0.06 (0.01)
Agaricia fragilis	0.00	0.00 (0.00)
Agaricia lamarcki	0.01	0.01 (0.00)
Agaricia spp.	0.38	0.31 (0.03)
Cladocora arbuscula	0.00	0.00 (0.00)
Colpophyllia natans	0.16	0.18 (0.05)
Dendrogyra cylindricus	0.02	0.02 (0.01)
Dichocoenia stokesii	0.17	0.03 (0.00)
Diploria clivosa	0.15	0.12 (0.02)
Diploria labyrnithiformis	0.12	0.06 (0.01)
Diploria spp.	0.03	0.03 (0.01)
Diploria strigosa	0.28	0.18 (0.03)
Eusmilia fastigiata	0.02	0.00 (0.00)
Favia fragum	0.05	0.01 (0.00)
Helioceris cucullata	0.02	0.03 (0.01)
Isophyllastrea rigida	0.00	0.00 (0.00)
Isophyllia sinuosus	0.01	0.00 (0.00)
Madracis mirabilis	0.01	0.00 (0.00)
Madracis spp.	0.04	0.01 (0.00)
Manicina areolata	0.04	0.01 (0.00)
Meandrina meandrites	0.22	0.07 (0.01)
Millepora alcicornis	0.24	0.07 (0.01)
Millepora aliciae	0.00	0.00 (0.00)
Millepora complanata	0.04	0.01 (0.00)
Millepora decactis	0.03	0.01 (0.00)
Millepora spp.	0.20	0.22 (0.11)
Montastraea annularis complex	0.49	1.89 (0.17)
Montastraea cavernosa	0.47	0.43 (0.04)
Mussa angulosa	0.01	0.00 (0.00)
Mycetophyllia daanana	0.00	0.00 (0.00)
Mycetophyllia ferox	0.03	0.01 (0.00)
Mycetophyllia lamarckiana	0.01	0.00 (0.00)
Mycetophyllia reesi	0.00	0.00 (0.00)
Mycetophyllia spp.	0.03	0.01 (0.00)
Oculina diffusa	0.01	0.01 (0.01)
Porites asteroides	0.62	0.56 (0.04)
Porites branneri	0.01	0.01 (0.01)
Porites colonensis	0.01	0.01 (0.00)
Porites porites	0.32	0.20 (0.04)
Porites spp.	0.01	0.03 (0.03)
Scleractinia	0.05	0.36 (0.12)
Scolymia cubensis	0.00	0.00 (0.00)
Scolymia spp.	0.04	0.01 (0.00)
Siderastrea radians	0.31	0.16 (0.02)
Siderastrea siderea	0.42	0.21 (0.02)
Siderastrea spp.	0.05	0.03 (0.01)
Solenastrae spp.	0.02	0.00 (0.00)
Stephanocoenia intercepta	0.16	0.03 (0.01)

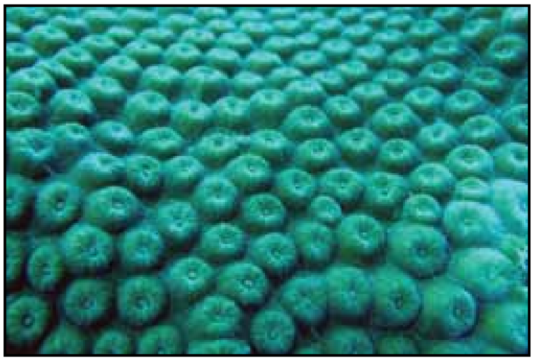
(Figure 2.13c). These areas of higher benthic rugosity and live coral cover contained six coral species on average with maximum richness (10-14 species) typically observed on the bank shelf. Coral species diversity (Figure 2.13d) appeared greatest on the bank shelf with values ranging from 1.4-2.5 per 100 m^2. In general, diversity declined with decreasing depth (shallower) and rugosity.

The broad study area exhibited overall low live coral cover (5%) on hardbottom with scattered patches of high cover and species richness. Live coral cover was dominated by five species (*Montastraea annularis* complex, *Porites astreoides*, *Montastraea cavernosa*, *Agaricia* spp. and *Siderastrea siderea*) that accounted for over 60% of the total coral cover in the study area (Table 2.4). These species also varied in percent cover and frequency of occurrence among hardbottom habitat types (Table 2.5).

Porites astreoides



Agaricia lamarckii



Montastrea cavernosa

Table 2.5. Summary statistics for coral species found in hardbottom sites by habitat type in the southwest Puerto Rico study area from 2001-2007.

	Col. Bedrock (n=2)		Col. Pavement (n=322)		Linear Reef (n=100)		Patch Reef (n=53)		Reef Rubble (n=1)		Scat. Coral/Rock in Sand (n=94)	
Species	**Freq. occur**	**% cover Mean ± SE**	**Freq. occur**	**% cover Mean ± SE**	**Freq. occur**	**% cover Mean ± SE**	**Freq. occur**	**% cover Mean ± SE**	**Freq. occur**	**% cover Mean ± SE**	**Freq. occur**	**% cover Mean ± SE**
A. cervicornis			0.14	0.16 (0.04)	0.09	0.42 (0.29)	0.08	0.14 (0.08)			0.03	0.05 (0.04)
A. palmata			0.01	0.04 (0.03)	0.04	0.11 (0.07)						
A. agaricites	0.50	0.03 (0.03)	0.16	0.07 (0.02)	0.12	0.05 (0.02)	0.15	0.06 (0.03)			0.02	0.03 (0.02)
A. fragilis			0.01	<0.01 (<0.01)								
A. lamarcki			0.02	0.02 (0.01)	0.01	<0.01 (<0.01)						
Agaricia spp.			0.44	0.37 (0.05)	0.41	0.30 (0.09)	0.32	0.37 (0.15)	1.00	0.24	0.17	0.11 (0.04)
C. arbuscula					0.01	<0.01 (<0.01)						
C. natans			0.17	0.16 (0.04)	0.17	0.38 (0.26)	0.15	0.18 (0.11)			0.09	0.06 (0.03)
D. cylindricus			0.02	0.03 (0.02)	0.01	0.01 (0.01)					0.01	0.01 (0.01)
D. stokesii	0.50	0.05 (0.05)	0.21	0.04 (0.01)	0.17	0.04 (0.01)	0.09	0.01 (<0.01)			0.07	0.01 (<0.01)
D. clivosa	0.50	0.75 (0.75)	0.16	0.10 (0.02)	0.18	0.22 (0.07)	0.21	0.19 (0.07)			0.03	0.01 (0.01)
D. labyrinthiformis			0.13	0.07 (0.02)	0.17	0.06 (0.02)	0.13	0.09 (0.04)			0.03	0.02 (0.01)
Diploria spp.			0.02	0.03 (0.01)	0.05	0.04 (0.02)					0.02	<0.01 (<0.01)
D. strigosa			0.31	0.14 (0.02)	0.39	0.43 (0.15)	0.23	0.17 (0.07)	1.00	0.30	0.09	0.06 (0.03)
E. fastigiata			0.02	<0.01 (<0.01)	0.02	<0.01 (<0.01)					0.01	0.00 (<0.01)
F. fragum			0.03	0.01 (<0.01)	0.08	0.02 (0.01)	0.15	0.04 (0.02)				
H. cucullata			0.03	0.05 (0.02)							0.02	0.03 (0.02)
I. rigida			<0.01	<0.01 (<0.01)	0.01	0.01 (0.01)						
I. sinuosus			0.01	<0.01 (<0.01)	0.02	<0.01 (<0.01)						
M. decactis			0.03	0.01 (<0.01)	0.04	<0.01 (<0.01)					0.01	<0.01 (<0.01)
M. mirabilis			0.01	<0.01 (<0.01)	0.01	<0.01 (<0.01)	0.02	0.01 (0.01)				
Madracis spp.			0.05	0.01 (<0.01)	0.05	0.01 (<0.01)					0.03	<0.01 (<0.01)
M. areolata			0.04	0.01 (<0.01)	0.06	0.01 (0.01)	0.02	<0.01 (<0.01)			0.02	0.01 (<0.01)
M. meandrites			0.29	0.10 (0.02)	0.18	0.04 (0.01)	0.09	0.03 (0.02)			0.13	0.04 (0.02)
M. alcicornis			0.26	0.05 (0.01)	0.30	0.12 (0.03)	0.30	0.21 (0.12)			0.11	0.02 (0.01)
M. complanata			0.03	0.01 (<0.01)	0.10	0.03 (0.01)	0.04	0.00 (<0.01)			0.02	<0.01 (<0.01)
Millepora spp.			0.19	0.07 (0.02)	0.35	0.92 (0.60)	0.21	0.16 (0.11)			0.09	0.02 (0.01)
M. annularis cmplx	0.50	0.35 (0.35)	0.57	2.40 (0.26)	0.49	1.74 (0.38)	0.53	1.54 (0.38)	1.00	0.20	0.20	0.56 (0.26)
M. cavernosa			0.56	0.47 (0.05)	0.45	0.62 (0.15)	0.43	0.33 (0.10)			0.20	0.19 (0.06)
M. angulosa			0.01	<0.01 (<0.01)	0.01	<0.01 (<0.01)						
M. aliciae			<0.01	<0.01 (<0.01)								
M. danaana			<0.01	<0.01 (<0.01)								
M. ferox			0.04	0.02 (0.01)	0.02	0.01 (<0.01)	0.02	0.03 (0.03)				
M. lamarckiana			0.01	<0.01 (<0.01)							0.01	0.01 (0.01)
M. reesi			0.01	<0.01 (<0.01)								
Mycetophyllia spp.			0.04	0.01 (<0.01)	0.02	0.01 (0.01)	0.04	0.01 (0.00)				
O. diffusa			0.01	0.01 (0.01)							0.01	0.01 (0.01)
P. asteroides	1.00	0.10 (<0.01)	0.67	0.62 (0.05)	0.77	0.75 (0.08)	0.66	0.56 (0.11)	1.00	0.10	0.24	0.15 (0.05)
P. branneri			0.01	<0.01 (<0.01)	0.02	0.04 (0.04)					0.01	<0.01 (<0.01)
P. colonensis			<0.01	0.01 (0.01)	0.02	0.01 (0.01)					0.01	0.01 (0.01)
P. porites	0.50	0.20 (0.20)	0.34	0.21 (0.07)	0.34	0.25 (0.08)	0.42	0.20 (0.05)	1.00	0.14	0.14	0.07 (0.03)
Porites spp.			0.01	0.05 (0.05)	0.02	0.01 (<0.01)						
Scleractinia			0.04	0.36 (0.18)	0.11	0.68 (0.32)	0.09	0.44 (0.26)			0.02	0.02 (0.02)
S. cubensis							0.02	0.01 (0.01)				
Scolymia spp.			0.05	0.01 (<0.01)	0.02	0.00 (<0.01)	0.04	<0.01 (<0.01)			0.04	0.01 (<0.01)
S. radians	0.50	0.65 (0.65)	0.34	0.17 (0.03)	0.40	0.26 (0.06)	0.26	0.14 (0.05)	1.00	0.22	0.15	0.07 (0.03)
S. siderea			0.50	0.25 (0.03)	0.46	0.23 (0.04)	0.38	0.23 (0.07)			0.18	0.07 (0.02)
Siderastrea spp.	0.50	0.20 (0.20)	0.05	0.02 (0.01)	0.05	0.03 (0.01)	0.06	0.02 (0.01)			0.05	0.05 (0.03)
Solenastraea spp.			0.02	<0.01 (<0.01)	0.01	0.00 (<0.01)	0.02	0.01 (0.01)			0.02	<0.01 (<0.01)
S. intercepta	0.50	0.10 (0.10)	0.18	0.04 (0.01)	0.20	0.04 (0.01)	0.09	0.02 (0.02)			0.10	0.02 (0.01)

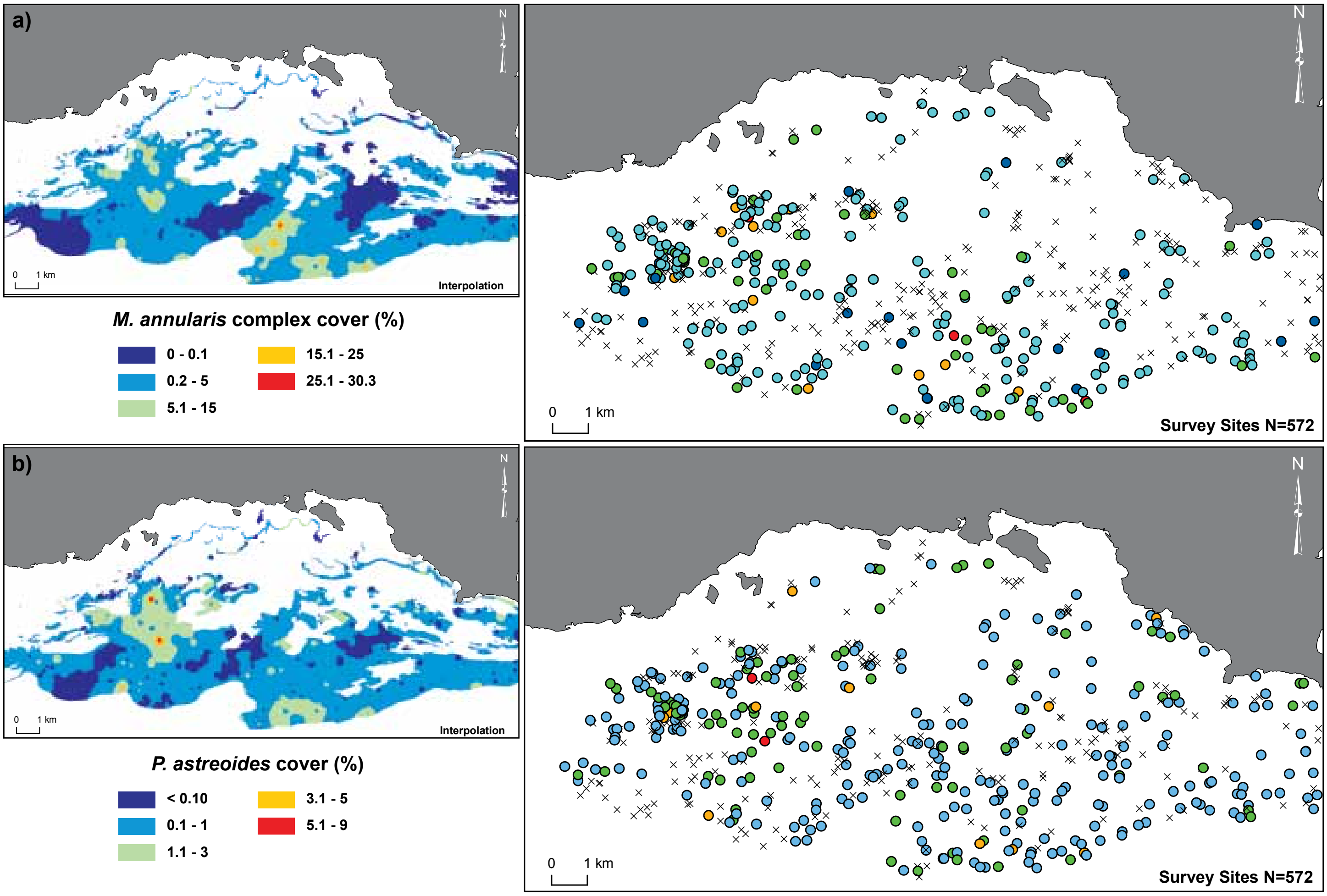


Figure 2.14. Maps of the interpolated (left map) and spatial (right map) distributions for live percent cover for coral species: (a) Montastraea annularis *complex and (b)* Porites astreoides.

M. annularis complex was the most dominant coral taxa (35% of total cover; 45% frequency of occurrence) with areas of highest cover (5-25%) located in the central and western portions of the study area (Figure 2.14a). Highest coral cover (>25%) occurred on the bank shelf in relatively deeper waters (>15 m). *P. astreoides* was the most frequently observed coral taxa (62%) with extensive presence within the study area but limited in terms of cover (Figure 2.14b). Overall, cover was low (0.1-1%) with localized patches of high cover (5-9%) associated with increased rugosity on the bank shelf. *M. cavernosa* (Figure 2.15a) was observed in nearly 50% of surveys (Table 2.4) and exhibited low cover throughout the study area.

Isolated patches of high cover (4-12%) were observed along the nearshore fringing reefs and moderate cover (1-4%) was patchily distributed throughout the study area. Colonies of *Agaricia* spp. were common (38%), but generally low in cover throughout the study area (Figure 2.15b). Highest cover (3-8%) was generally observed at moderate depths (6-18 m) and high rugosity. *S. siderea* was commonly observed (42%) but cover was low throughout the study area (Figure 2.15c). Small isolated patches of cover >1% were widely dispersed but not spatially associated with depth or rugosity.

Acropora cervicornis

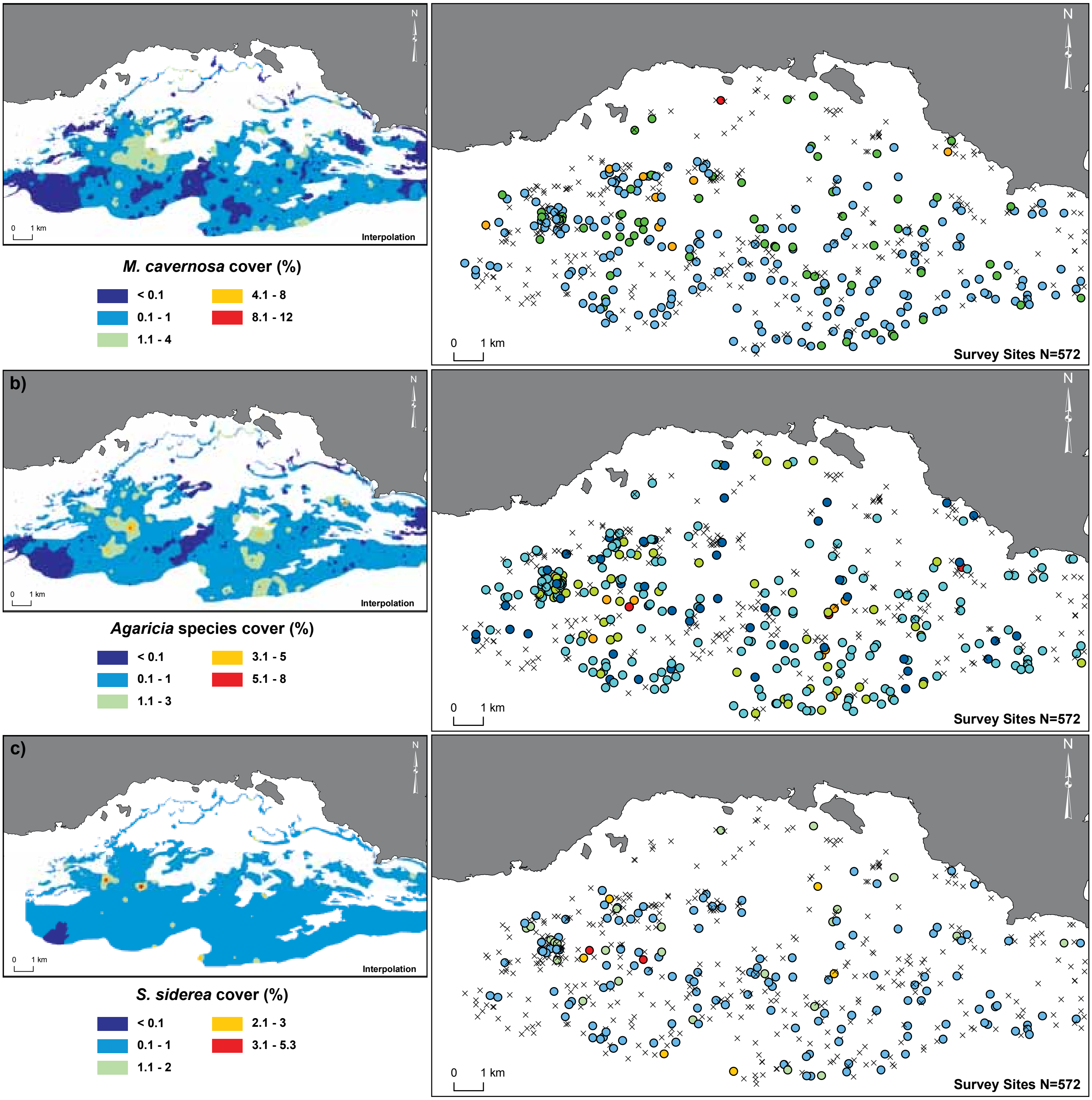


Figure 2.15. Maps of the interpolated (left map) and spatial (right map) distributions for live percent cover for coral species: (a) Montastraea cavernosa*, (b)* Agaricia *species and (c)* Siderastrea siderea*.*

Threatened *Acropora* species were rare or uncommon (*Acropora cervicornis* 11%, *Acropora palmata* 1%) with patchy distribution throughout the study area (Figure 2.16). Colonies were generally observed at depths <10 m and both exhibited highest cover in relatively close proximity in the western portion of the study area. *A. cervicornis* (Figure 2.16a) was more widely distributed than *Acropora palmata*, (Figure 2.16b) but both were low overall. It should be noted, our sampling approach was not optimized for detecting *Acropora* species.

Acropora palmata

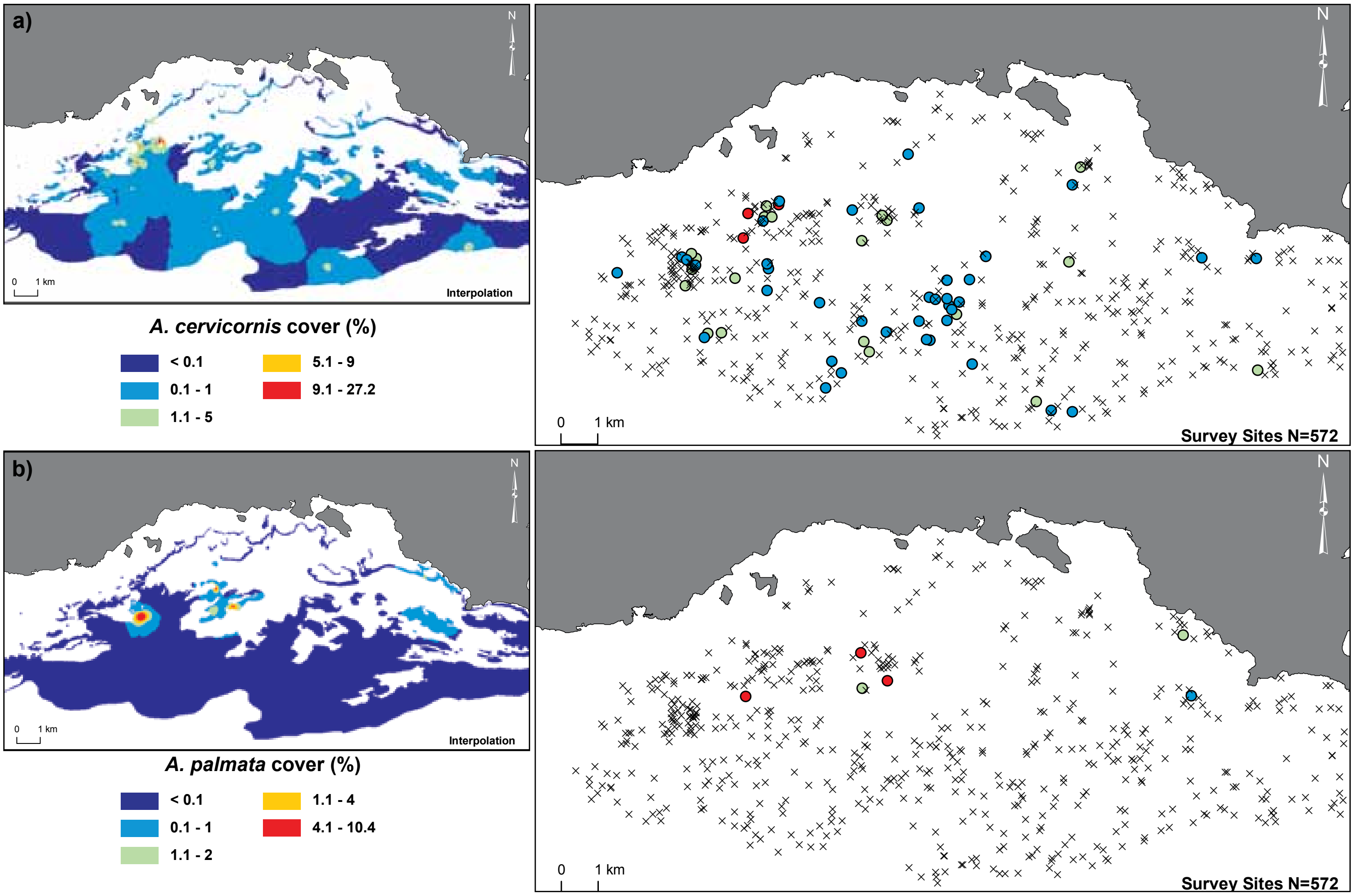


Figure 2.16. Maps of the interpolated (left map) and spatial (right map) distributions for live percent cover for coral species: (a) Acropora cervicornis *and (c)* Acropora palmata.

2.3.2.2. Spatial patterns in macroalgae cover

Macroalgae (Figure 2.17a) cover was similar on both hard and soft bottom habitats (mean=12-14%) with considerable variability in the shallow waters close to the mangrove shoreline and the edge of the bank shelf. Macroalgal cover was highest (60-90%) along the western edge of the study area exhibited by broad shallow (<4 m) sand flats. Significant macroalgal cover was also observed in and adjacent to the mangroves that line the shore and some offshore islands. Turf algae were a significant component to the hardbottom benthic community. Mean cover (30%) was nearly twice that of macroalgal cover and combined accounted for 40-45% of the hardbottom cover. Turf algae (Figure 2.17b) were widely distributed across the hardbottom seascape and higher cover was spatially associated with high rugosity. Soft corals (gorgonians and sea fans) were common hardbottom features (Figure 2.17c), in particular on patch reefs and colonized pavement. Cover ranged from 0-58% and, in general, declined from the shore to the edge of the bank shelf. High soft coral cover was associated with high rugosity. Sponge cover ranged from 0-30% and was variable on all hardbottom types (Figure 2.17d). The general pattern indicated higher cover at greater depth and higher rugosity. Sponge cover appeared to be greater in the central and eastern portions of the study area.

Macroalgae

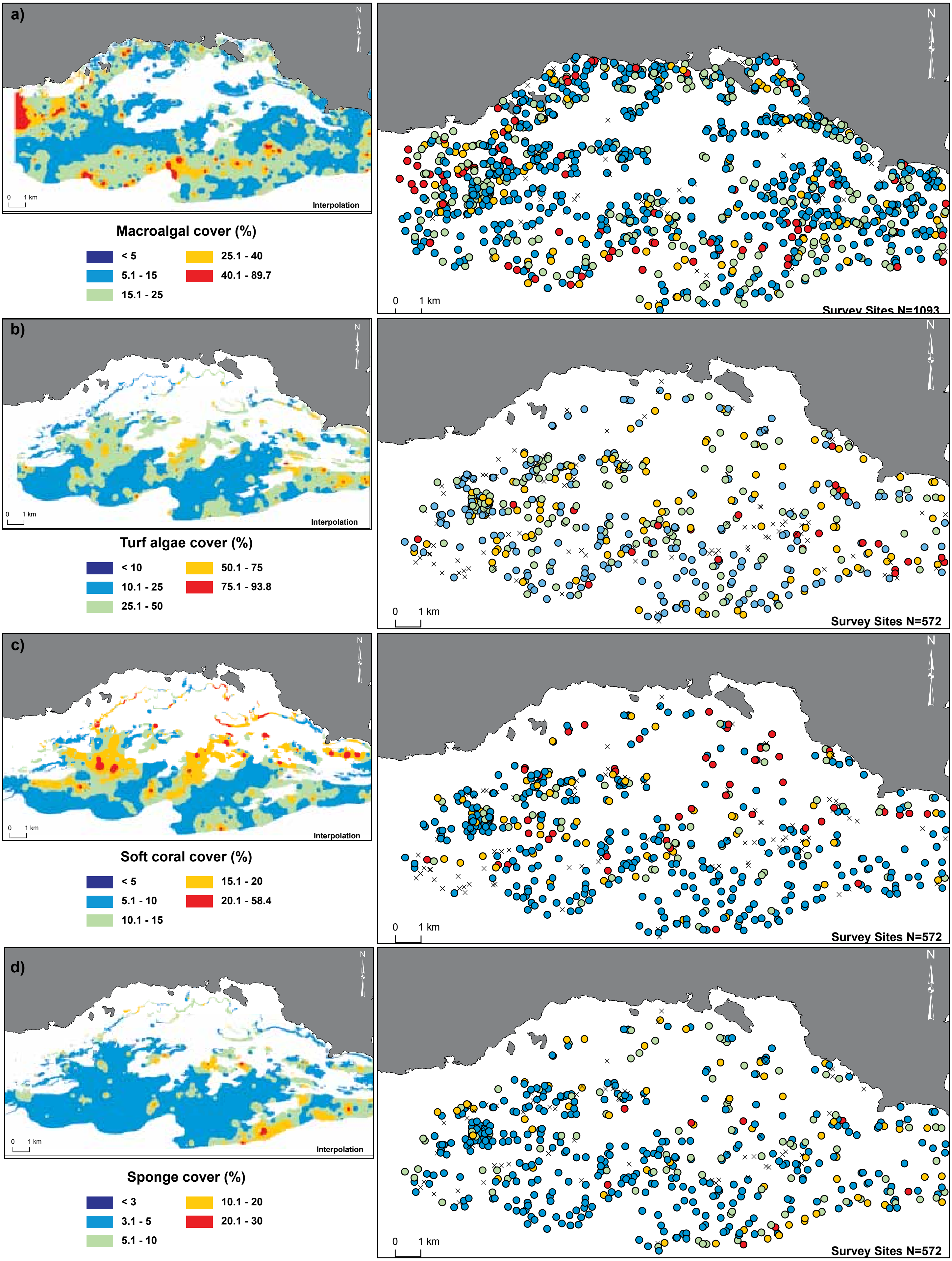


Figure 2.17. Maps of the interpolated (left map) and spatial (right map) distributions for: (a) macroalgae, (b) turf algae, (c) soft coral cover and (d) sponges.

2.3.2.3. Spatial patterns in seagrass cover

Thalassia testudinum

Seagrasses were common on the softbottom habitats in the study area (Figure 2.18a). Seagrass cover was most extensive close to shore and declined toward the shelf break. Large continuous areas of seagrass are present nearshore, adjacent to mangroves and inside the fringing reefs. In this zone, seagrass was predominantly *T. testudinum* that was abundant to depths of approximately 16 m (Figure 2.18b). *T. testudinum* cover ranged from 0-96% and occurred primarily in the central and western portion of the study area. *S. filiforme* (Figure 2.18c), was uncommon (<10% cover) at depths <10 m, but surpassed *Thalassia* with increasing depth where cover ranged between 30-85%.

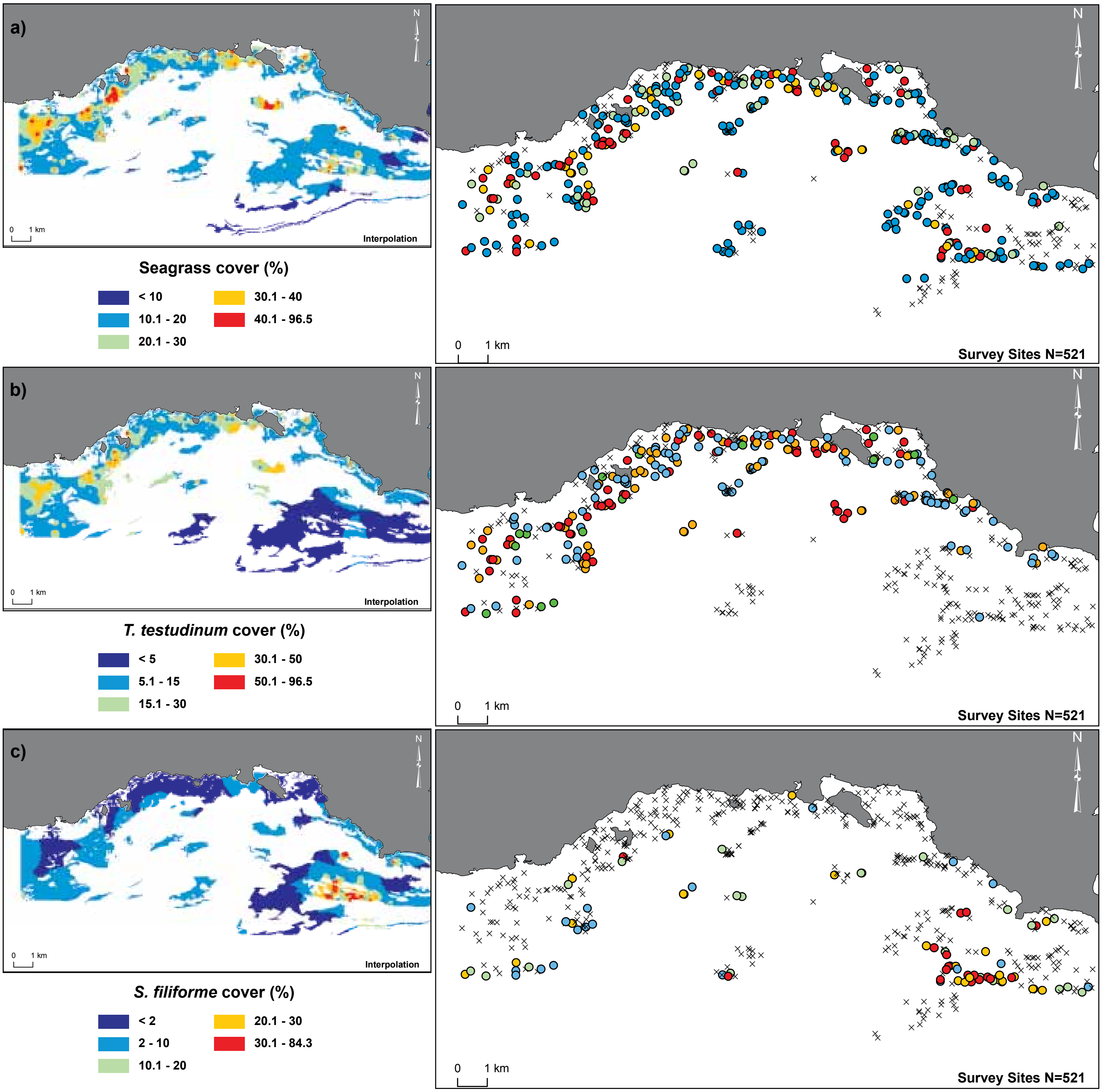


Figure 2.18. Maps of the interpolated (left map) and spatial (right map) distributions for: (a) seagrass, (b) Thalassia testudinum *and (c)* Syringodium filiforme.

2.3.3. Temporal patterns in benthic composition (2001-2007)

Temporal trends in the percent cover of live coral and algae were based on multiple pair-wise comparisons among sampling periods. A common approach to determining temporal trends in coral reef communities is the use of permanent sites and stations that are revisited periodically to examine successive changes in selected metrics. The data used in this study instead were collected using a stratified sampling design to randomly select sites that were never revisited. Thus, observed temporal trends reflect changes in average conditions within a habitat type between sampling periods rather than changes at specific sites. An underlying premise is that randomly selected sites are representative of the habitat strata from which they were selected.

2.3.3.1. Temporal patterns in live coral cover

Examination of differences in mean live coral cover among sampling periods revealed interesting patterns, which indicated that there has been a general decline in overall coral cover (Figures 2.19). Except for February 2002, mean estimates among sites during earlier missions (January 2001 through June 2006) were greater than those of later missions (December 2002 through August 2007). The higher variance in coral cover observed during earlier missions may have resulted from smaller sample size (n<19) during earlier missions compared with later missions (n>29). However, examination of residual plots of percent coral cover ordered from Winter 2001 to Summer 2007 indicates that the frequency distribution and variance estimates of coral cover were similar across sampling periods and that the statistical populations can be compared across years. Further statistical analysis revealed that there was a consistent overall decline in live coral cover from Winter 2001 through Summer 2007, and the decline was significant for the five most abundant genera: *Montastraea*, *Porites*, *Agaricia*, and *Siderastrea* (Figure 2.19, Table 2.6).

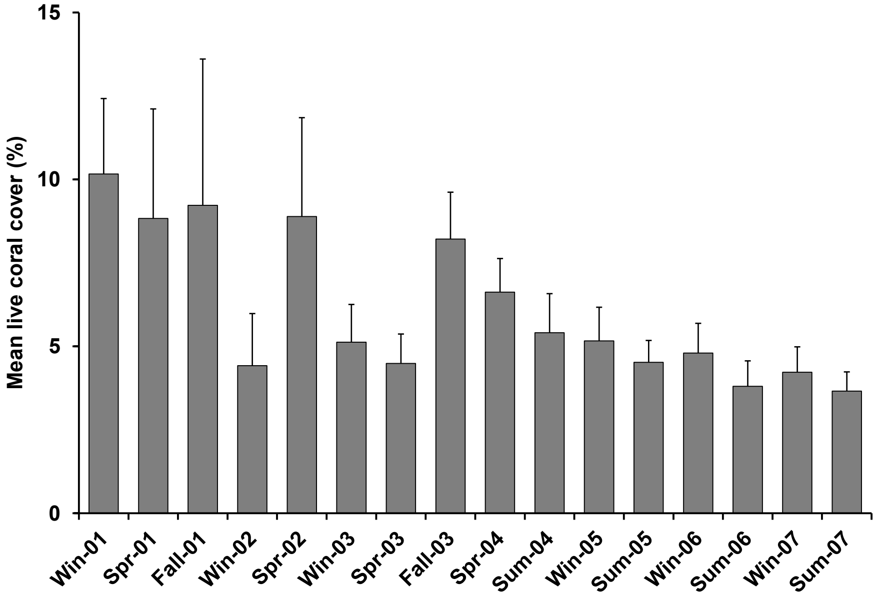


Figure 2.19.Seasonal and inter-annual patterns in mean (± SE) percent live coral cover of on hardbottom habitats over a seven year period in the southwest Puerto Rico study region.

Table 2.6. Results of non-parametric ordered comparisons (JT test) to determine trends in percent live cover of the five most abundant coral genera in SW Puerto Rico from Winter 2001 through Summer 2007. N = 572.

Hard genus	J	Z	P	N
All hard corals	113,946.50	16.9	< 0.0001	572
Montastraea	95,339.90	9.0	< 0.0001	572
Diploria	85,418.50	4.8	< 0.0001	572
Porites	90,439.00	6.7	< 0.0001	572
Siderastrea	94,496.50	8.6	< 0.0001	572
Agaricia	81,357.50	2.8	0.0029	572

When coral cover data were analyzed separately for different habitat types, statistically significant differences among years were detected. Live coral cover varied significantly among sampling periods within two hardbottom habitat types: colonized pavement and colonized pavement with sand channels habitats (Table 2.7). Non-parametric multiple comparisons of mean live coral cover among sampling periods within colonized pavement

Table 2.7. Results of non-parametric ANOVA (Wilcoxon test) to determine significant differences in percent cover of live coral among sampling periods within each in hardbottom habitat in the southwest Puerto Rico study region.

Hardbottom habitat types	N	X^2	DF	P
Colonized Pavement	110	29.7	14	0.008*
Col. Pavement with Sand Channels	212	33.3	15	0.004*
Linear Reef	100	17.0	15	0.317
Patch Reef (Aggregated)	27	11.6	9	0.237
Patch Reef (Individual)	26	8.1	10	0.621
Scattered Coral/Rock in Sand	94	12.3	15	0.654

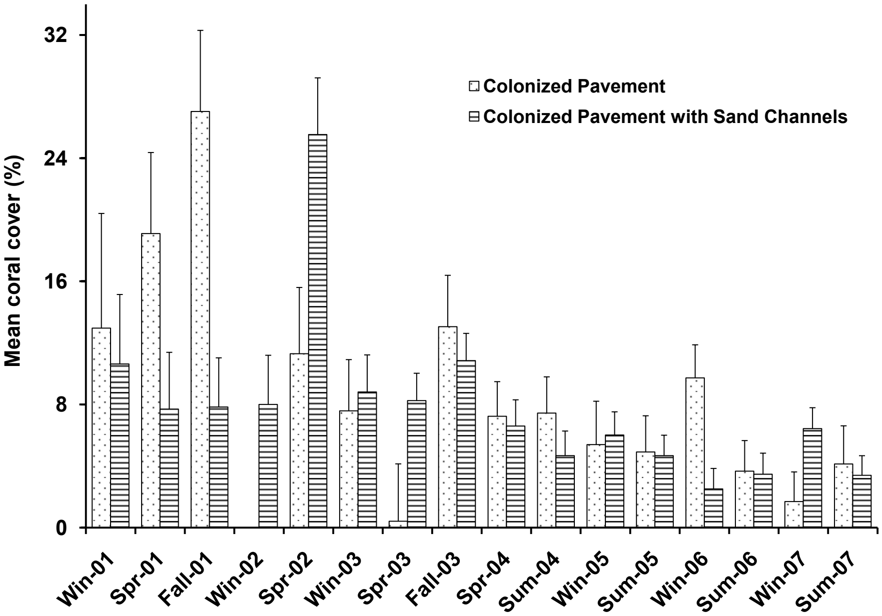


Figure 2.20. Seasonal and inter-annual patterns in mean (± SE) percent live coral cover on two hardbottom habitat types over a seven year period in the southwest Puerto Rico study region. Means with similarly shaded arrows were not significantly different from each other (p<0.05).

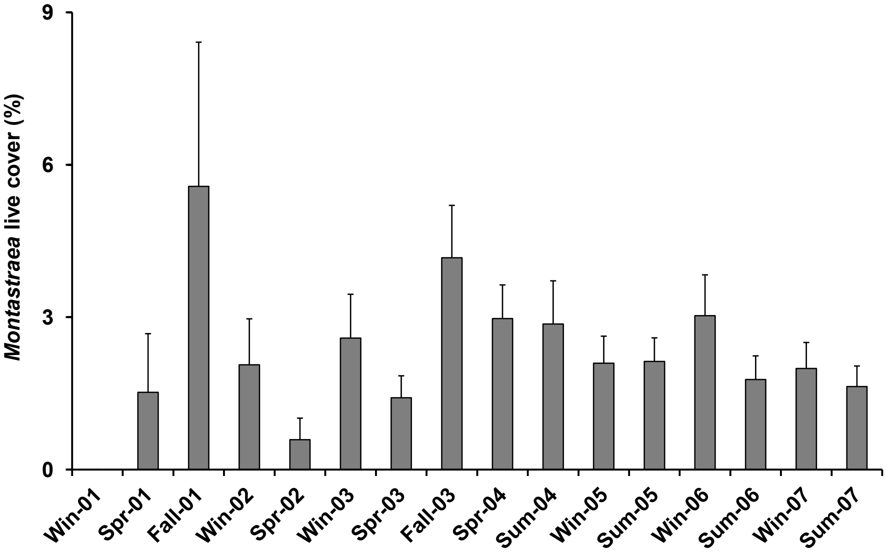


Figure 2.21. Seasonal and inter-annual patterns in mean (± SE) percent live cover of Montastraea *spp. on hardbottom habitats over a seven year period in the southwest Puerto Rico study region. Means with similarly shaded arrows were not significantly different from each other (p<0.05).*

Table 2.8. Results of non-parametric ordered comparisons (JT test) to determine trends in percent live cover of coral within habitat types in southwest Puerto Rico from Winter 2001 through summer 2007.

Habitat type	**J**	**Z**	**P**	**N**
Colonized pavement	4,321.50	8.2	< 0.0001	110
Colonized pavement with Sand channels	18,429.50	15.9	< 0.0001	212
Linear reef	4,340.50	12.2	< 0.0001	100
Patch reef (aggregated)	292.50	5.6	< 0.0001	27
Patch reef (individual)	279.50	5.9	< 0.0001	26
Scattered Coral / rock in unconsolidated sediment	2,461.50	3.2	0.0	94

with sand channels revealed that mean live coral cover was significantly higher during September 2003 compared with January 2006 and August 2007 (p<0.05, Figure 2.20). Non-parametric multiple pair-wise comparisons of mean live coral in colonized pavement habitats did not reveal which sampling periods were statistically different, although a non-parametric One-Way ANOVA indicated statistically significant variation among sampling periods (Table 2.7; Figure 2.20). In terms of overall trends, there was a decreasing trend in hard coral cover from Winter 2001 through Summer 2007 in five habitats (colonized pavement, colonized pavement with Sand channels, Linear reef, Patch reef, and scattered coral/rock in unconsolidated sediment (Table 2.8).

Non-parametric Wilcoxon tests revealed that mean live cover of *M. annularis* complex also varied significantly among sampling periods (x^2=39.0, p=0.0006). The temporal trend in mean estimates suggested that live *Montastraea* cover may have decreased through time (Figure 2.21). Non-parametric multiple pair-wise comparisons revealed that observed live cover of *M. annularis* complex was significantly higher during later sampling periods (September 2003 through August 2007) compared with June 2002 (p<0.05; Figure 2.21). However, there was an overall significant declining trend in cover of *M. annularis* complex during the study period, and the decline was strongest from winter 2003 through summer 2007 (J=95,339, P<0.0001, N=572).

Further analysis revealed disparate results for significant differences among sampling periods in live coral cover of the five most common coral genera on hardbottom habitat types surveyed in La Parguera (Table 2.9).

All five genera (*Montastraea*, *Porites*, *Siderastrea*, *Diploria* and *Agaricia*) showed significant differences among sampling periods within at least one hardbottom habitat (p<0.05; Table 2.10). Significant differences in mean live cover of these coral genera among sampling periods were detected for pavement and linear reef habitats, which were the more common habitats in La Parguera, and also were the ones most intensely surveyed (Table 2.10).

Table 2.9. Results of non-parametric ANOVA (Wilcoxon test) to determine significant differences in percent live cover of coral genera among sampling periods within each in hardbottom habitat in the southwest Puerto Rico study region.

Hard bottom Habitat	*Montastraea*	*Porites*	*Siderastrea*	*Diploria*	*Agaricia*
Colonized Pavement	0.009	0.064	0.193	0.001	0.070
Col. Pav. w/ Sand Channels	0.098	0.003	0.027	0.837	0.001
Linear Reef	0.174	0.051	0.037	0.110	0.009
Patch Reef (Aggregated)	0.409	0.400	0.141	0.511	0.555
Patch Reef (Individual)	0.271	0.249	0.133	0.571	0.162
Scattered Coral/Rock in Sand	0.747	0.972	0.470	0.231	0.849

Table 2.10. A-E. Results of non-parametric ordered comparisons (JT test) to determine trends in percent live cover of coral genera within habitat types in southwest Puerto Rico from Winter 2001 - Summer 2007.

A) ***Montastraea***

Habitat type	J	Z	P	N
Colonized pavement	3,573.50	4.4	< 0.0001	110
Colonized pavement with Sand channels	14,141.50	7.6	< 0.0001	212
Linear reef	3,426.50	6.9	< 0.0001	100
Patch reef (aggregated)	256.00	4.1	< 0.0001	27
Patch reef (individual)	244.00	4.4	< 0.0001	26
Scattered Coral / rock in unconsolidated sediment	2,089.00	0.6	0.271	94

B) ***Diploria***

Habitat type	J	Z	P	N
Colonized pavement	2,421.50	-2.0	0.022	110
Colonized pavement with Sand channels	12,650.00	4.8	< 0.0001	212
Linear reef	3,130.50	5.1	< 0.0001	100
Patch reef (aggregated)	198.00	2.0	< 0.0001	27
Patch reef (individual)	223.50	3.5	0.0002	26
Scattered Coral / rock in unconsolidated sediment	2,046.00	0.3	0.367	94

C) ***Porites***

Habitat type	J	Z	P	N
Colonized pavement	3,034.00	1.5	0.0636	110
Colonized pavement with Sand channels	13,330.00	6.0	< 0.0001	212
Linear reef	3,525.00	7.3	< 0.0001	100
Patch reef (aggregated)	251.00	3.9	< 0.0001	27
Patch reef (individual)	243.50	4.4	< 0.0001	26
Scattered Coral / rock in unconsolidated sediment	2,147.00	1.1	0.1368	94

D) ***Siderastrea***

Habitat type	J	Z	P	N
Colonized pavement	3,313.50	3.0	0.0012	110
Colonized pavement with Sand channels	14,924.00	9.1	< 0.0001	212
Linear reef	3,381.00	6.5	< 0.0001	100
Patch reef (aggregated)	199.50	1.8	0.0398	27
Patch reef (individual)	221.50	3.4	0.0003	26
Scattered Coral / rock in unconsolidated sediment	2,101.00	0.7	0.2447	94

E) ***Agaricia***

Habitat type	J	Z	P	N
Colonized pavement	3,123.00	2.0	0.0202	110
Colonized pavement with Sand channels	12,094.50	3.6	0.0001	212
Linear reef	2,786.50	3.0	0.0012	100
Patch reef (aggregated)	187.00	1.3	0.105	27
Patch reef (individual)	196.50	2.4	0.0079	26
Scattered Coral / rock in unconsolidated sediment	2,026.50	0.1	0.4524	94

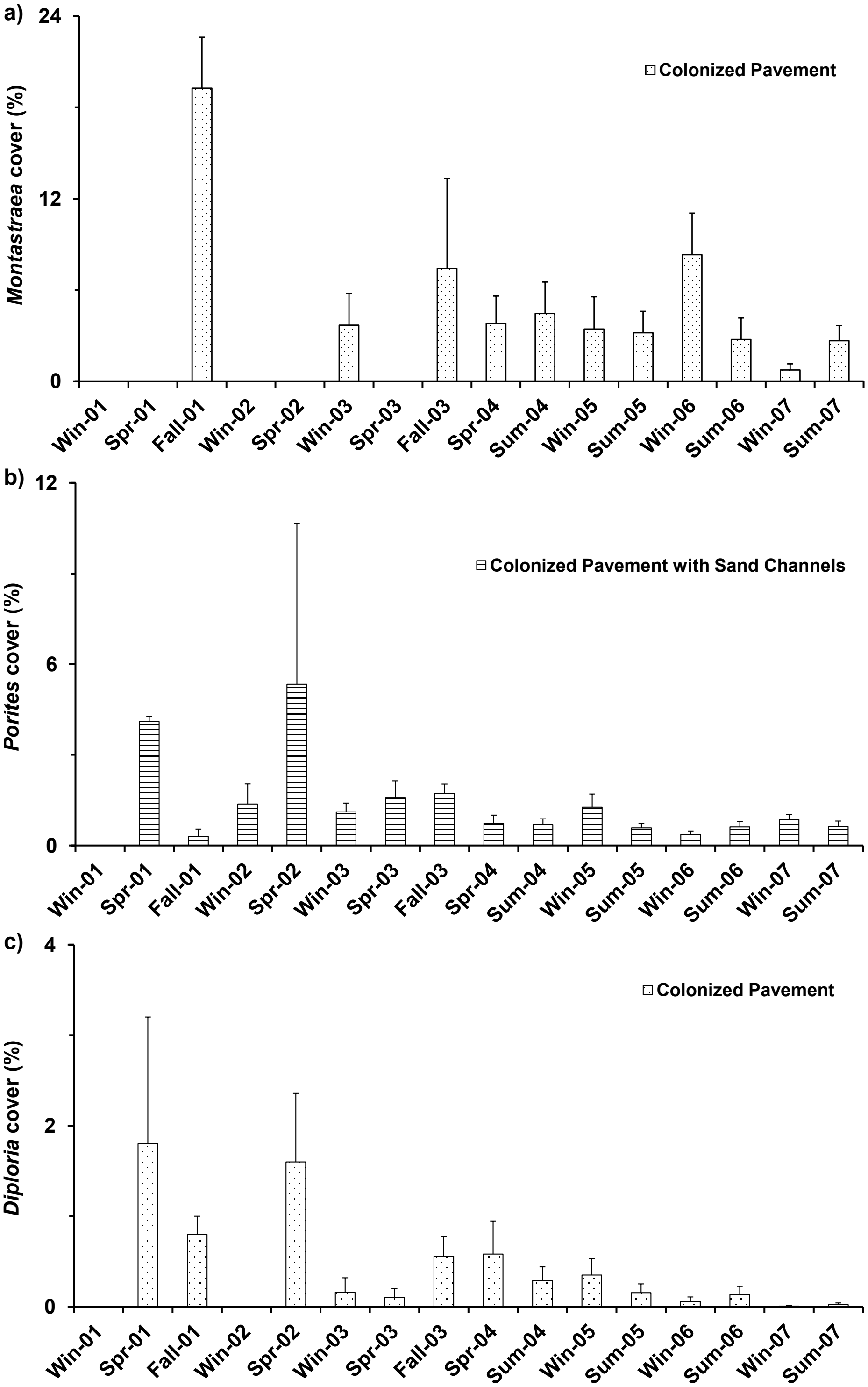


Figure 2.22. Seasonal and inter-annual patterns in mean (± SE) percent live cover of three of the five most abundant coral genera: a) Montastraea *spp., b)* Porites *spp. and c)* Diploria *spp., on hardbottom habitat types over a seven year period. Arrows indicate non-parametric pair-wise comparisons where significant differences among sampling periods were found. Means with similarly shaded arrows were not significantly different from each other (p< 0.05).*

Non-parametric multiple pair-wise comparisons between sampling periods revealed significant differences in mean live cover for the most abundant coral genera within a few hardbottom habitat types. Temporal variability in mean live cover of *Montastraea* spp. was significant for colonized pavement (p=0.009), and appeared to be lower during later sampling periods compared with earlier ones. Mean live cover of *Montastraea* spp. was 19.3 ± 2.4%, in October 2001 (n=2) but over time gradually decreased to 2.7 ± 1.0 (n=15) by August 2007 (Figure 2.22a). On colonized pavement with sand channel hardbottom habitats, mean live cover of *Porites* spp. was 4.1 ± 0.2% in May 2001 (n=3) but gradually decreased significantly to 0.4 ± 0.1% (n=23) by January 2006 (Figure 2.22b). Likewise, mean live cover of *Diploria* spp. in colonized pavement habitat was 1.6 ± 0.8% (n=3) in June 2002 but was significantly lower (p<0.05) in January 2006 (0.1 ± 0.1%, n=12). During January 2007 and August 2007, live cover of *Diploria* spp. was essentially 0% (Figure 2.22c).

On both linear reef and colonized pavement with sand channel hardbottom habitats, mean live cover of *Siderastrea* spp. varied significantly among sampling periods, although it appeared not to increase or decrease over time. Yet multiple comparison tests detected significant pair-wise differences among sampling periods only for linear reef habitats, where mean live cover of *Siderastrea* spp. was higher in August 2006 and August 2007 compared with June 2002 (p<0.05; Figures 2.23a). Similarly, mean live cover of *Agaricia* spp. on linear reef habitats was essentially 0% in June 2006 (n=7) but was significantly higher (0.7 ± 0.3%, n=9) by May 2003 (Figure 2.23b). Significant pair-wise differences in mean live cover of *Agaricia* spp. on colonized pavement with sand channel habitats were not detected among sampling period (Figure 2.23b). Interestingly, significant declining trends in percent cover were observed for all five genera (*Montastraea*, *Porites*, *Siderastrea*, *Diploria* and *Agaricia*) in most habitats (Table 2.10 A-E).

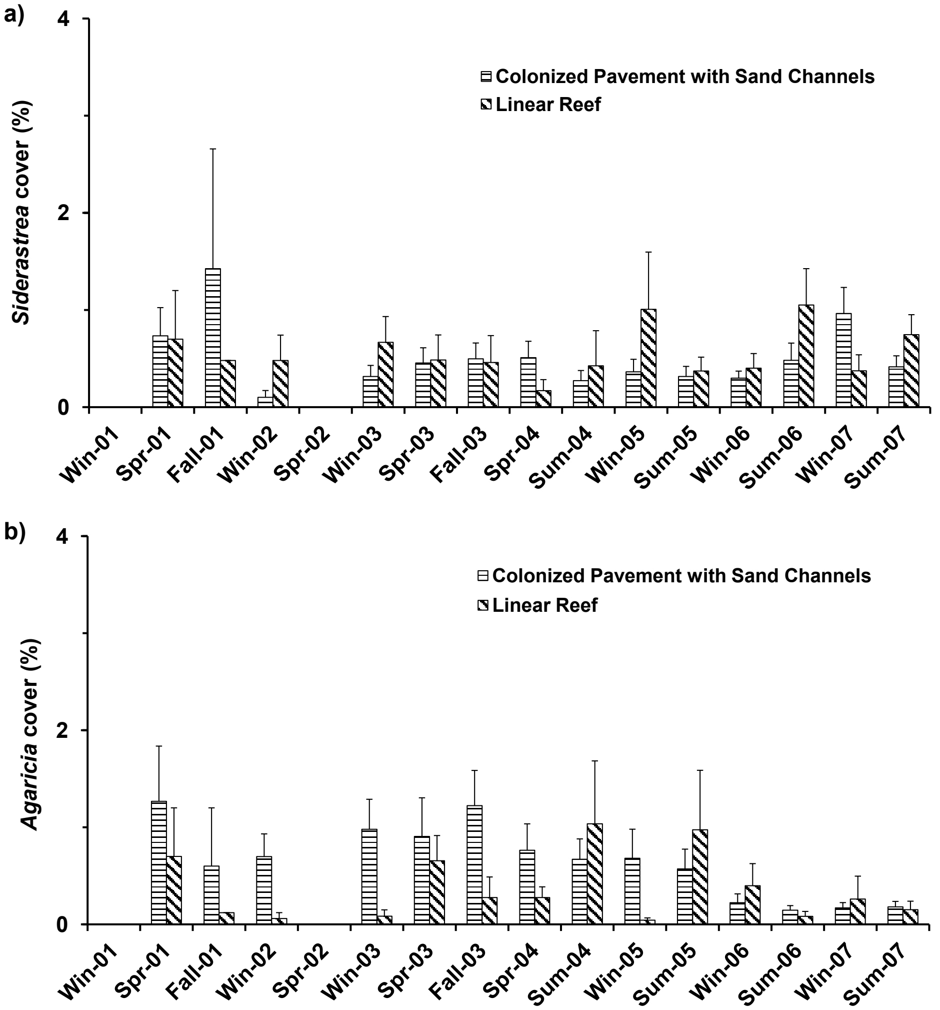


Figure 2.23. Seasonal and inter-annual patterns in mean (± SE) percent live cover of two of the five most abundant coral genera: a) Siderastrea *spp. and b)* Agaricia *spp., on hardbottom habitat types over a seven year period. Arrows indicate non-parametric pair-wise comparisons where significant differences among sampling periods were found. Means with similarly shaded arrows were not significantly different from each other (p< 0.05).*

2.3.3.2. Temporal patterns in algal cover

Mean algal cover varied significantly among sampling periods (Table 2.11). Non-parametric ordered comparisons indicated that there was a significant declining trend in cover of various types of algae (Table 2.12); whereas, non-parametric multiple comparison tests revealed several pair-wise differences in mean algal cover among years (Table 2.13a-d). Mean cover of macroalgae followed a sinusoidal pattern with highs of 31.9 ± 7.9% in January 2001, 17.4 ± 2.1% in March 2004, and 21.7 ± 2.8% in August 2008 (Figure 2.24a). Mean cover of turf algae exhibited a similar pattern, with higher peaks in mean cover occurring during February and December 2002 and lower peaks in cover during January and August 2005 (Figures 2.24b). Temporal pattern in mean cover of cyanobacteria/filamentous algae and CCA suggest that episodic blooms alternated with periods of low cover of those algal types between January 2001 and August 2007 (Figures 2.24c and 2.24d).

Table 2.11. Results of non-parametric ordered comparisons (JT test) to determine trends in percent cover of algal types from Winter 2001 through summer 2007 in southwest Puerto Rico study region. N = 422.

Algal Group	J	Z	P	N
All algae	78,954.5	26.41	< 0.0001	422
Turf algae	66,370.5	17.70	< 0.0001	422
Crustose coralline algae	47,279.0	5.06	< 0.0001	422
Filamentous algae	52,848.0	8.76	< 0.0001	422
Macro algae	67,059.0	18.16	< 0.0001	422

Table 2.12. Results of non-parametric ANOVA (Wilcoxon test) to determine significant differences in percent live cover of algal types among sampling periods within each in hardbottom habitat in the southwest Puerto Rico study region.

Algae types	X^2	DF	P
All algal types	56.8	15	0.000001
Macroalgae	60.8	15	0.000000
Turf algae	87.5	15	0.000000
Cyanobacteria / Filamentous algae	132.7	15	0.000000
Crustose coralline algae	90.6	15	0.000000

Table 2.13. Multiple non-parametric pair-wise comparisons of mean percent algae cover for: a) macroalgae, b) turf algae, c) cyanobacteria/ filamentous algae and d) crustose coralline algae among sampling periods (2001-2007) in the southwest Puerto Rico study region. Green boxes with dots indicate significant pair-wise differences (p<0.05).

Win-01	Spr-01	Fall-01	Win-02	Spr-02	Win-03	Spr-03	Fall-03	Spr-04	Sum-04	Win-05	Sum-05	Win-06	Sum-06	Win-07	Sum-07
Win-01															
	Spr-01														
		Fall-01													
			Win-02												
				Spr-02											●
					Win-03			●	●						●
						Spr-03									
							Fall-03								
								Spr-04				●			
									Sum-04						
										Win-05					
											Sum-05				●
												Win-06			●
													Sum-06		●
														Win-07	●
															Sum-07

a) Macroalgae

Win-01	Spr-01	Fall-01	Win-02	Spr-02	Win-03	Spr-03	Fall-03	Spr-04	Sum-04	Win-05	Sum-05	Win-06	Sum-06	Win-07	Sum-07
Win-01						●	●				●	●			
	Spr-01														●
		Fall-01													●
			Win-02						●						●
				Spr-02											●
					Win-03				●			●	●	●	●
						Spr-03									●
							Fall-03								●
								Spr-04	●						●
									Sum-04						
										Win-05					
											Sum-05				
												Win-06			●
													Sum-06		●
														Win-07	●
															Sum-07

b) Turf algae

Win-01	Spr-01	Fall-01	Win-02	Spr-02	Win-03	Spr-03	Fall-03	Spr-04	Sum-04	Win-05	Sum-05	Win-06	Sum-06	Win-07	Sum-07
Win-01						●	●								●
	Spr-01						●								●
		Fall-01		●		●	●								●
			Win-02				●								●
				Spr-02				●				●			●
					Win-03	●	●								●
						Spr-03		●	●			●			
							Fall-03	●	●			●			
								Spr-04							●
									Sum-04						●
										Win-05					●
											Sum-05				●
												Win-06			●
													Sum-06		●
														Win-07	●
															Sum-07

c) Cyanobacteria / filamentous algae

Win-01	Spr-01	Fall-01	Win-02	Spr-02	Win-03	Spr-03	Fall-03	Spr-04	Sum-04	Win-05	Sum-05	Win-06	Sum-06	Win-07	Sum-07
Win-01															
	Spr-01									●				●	
		Fall-01													
			Win-02			●				●	●			●	
				Spr-02						●				●	
					Win-03										
						Spr-03		●	●				●		
							Fall-03							●	
								Spr-04		●	●			●	
									Sum-04	●	●			●	
										Win-05					
											Sum-05		●	●	
												Win-06		●	
													Sum-06		
														Win-07	●
															Sum-07

d) Crustose coralline algae

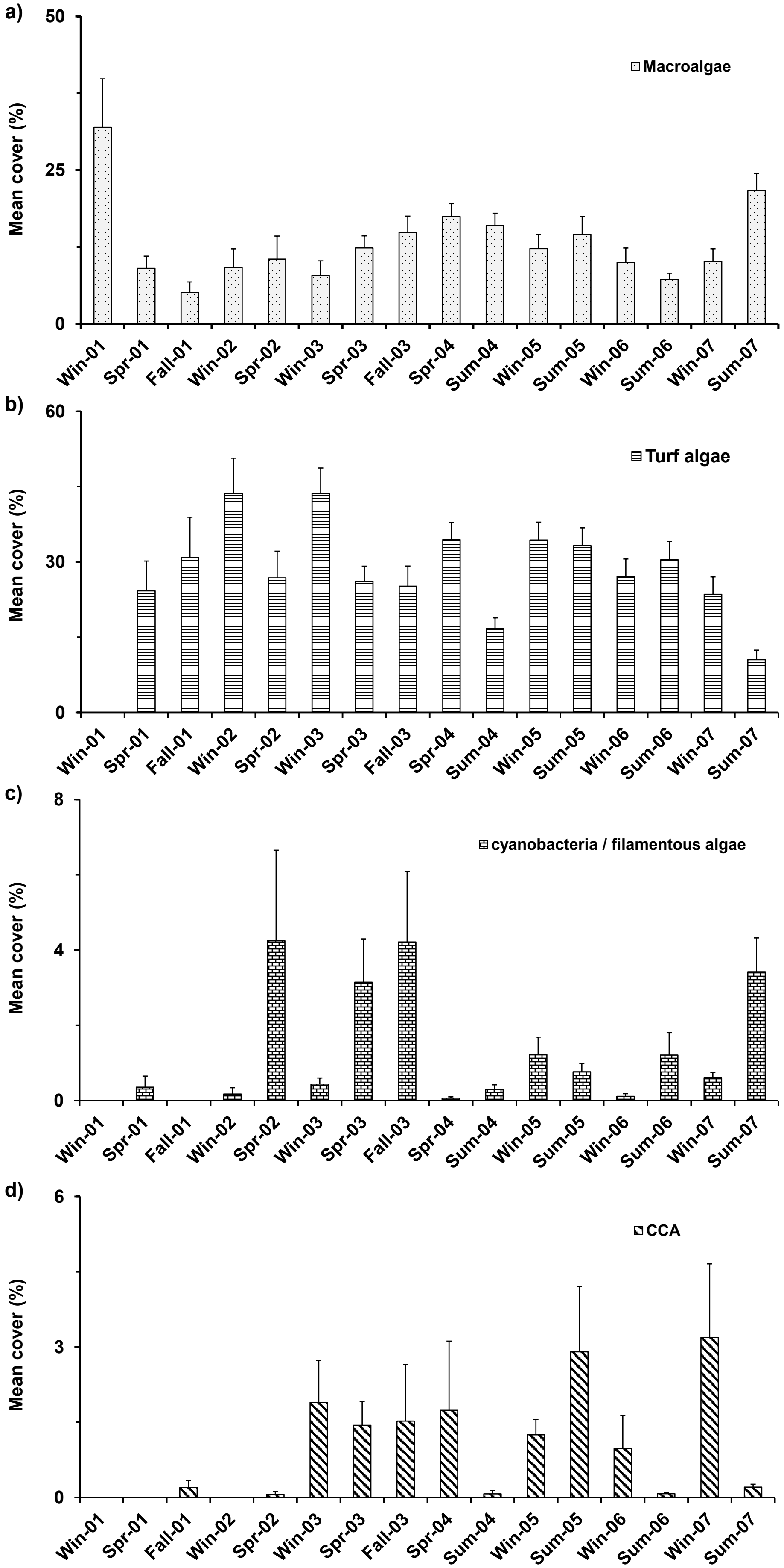


Figure 2.24. Seasonal and inter-annual patterns in mean (± SE) percent cover of algal categories: a) macroalgae, b) turf algae, c) cyanobacteria/filamentous algae and d) crustose coralline algae on hardbottom habitat types over a seven year period. Results of non-parametric comparisons of means among sampling periods are shown in Tables 2.13a-d.

2.3.4. Abundance and distribution of macroinvertebrates

*2.3.4.1. Queen conch (*Eustrombus gigas*)*

Queen conch (Eustrombus gigas)

A total of 45 individual queen conch (*Eustrombus gigas*; 34 immature and 11 mature) were recorded from 35 of 618 survey sites in the La Parguera study area since conch data collection began in August 2004 (Table 2.14). Seventy six percent of all conch observed were immature, and these occurred at 26 of the 35 sites (Table 2.14). Mature conch were recorded at 10 of the 35 sites, comprising approximately 24% of all conch recorded (Table 2.14). Immature and mature conch were found across the shelf, with no clear inshore-offshore pattern of spatial or habitat segregation, although immature and mature conch were present together at only one site (Figure 2.25). In contrast to other regions of the U.S. Caribbean, extensive areas of seagrass did not appear to support markedly higher occurrence of conch than other habitat types.

Table 2.14. Number of total surveys, surveys where conch were observed and individuals counted during 2004-2007. Conch totals per year represent two missions per year, January and August, except in 2004: conch data collection began in August 2004.

Year	Habitat type	Number of sites where conch were observed	Number of immature conch	Number of mature conch	Total number of conch	Total number of sites surveyed
2004	Hard	3	2	1	3	**89**
	Soft	5	8	0	8	
	Mangrove	1	1	0	1	
2005	Hard	1	1	0	1	**170**
	Soft	2	4	0	4	
	Mangrove	1	1	0	1	
2006	Hard	4	4	1	5	**179**
	Soft	11	10	5	15	
	Mangrove	0	0	0	0	
2007	Hard	4	2	2	4	**180**
	Soft	3	1	2	3	
	Mangrove	0	0	0	0	

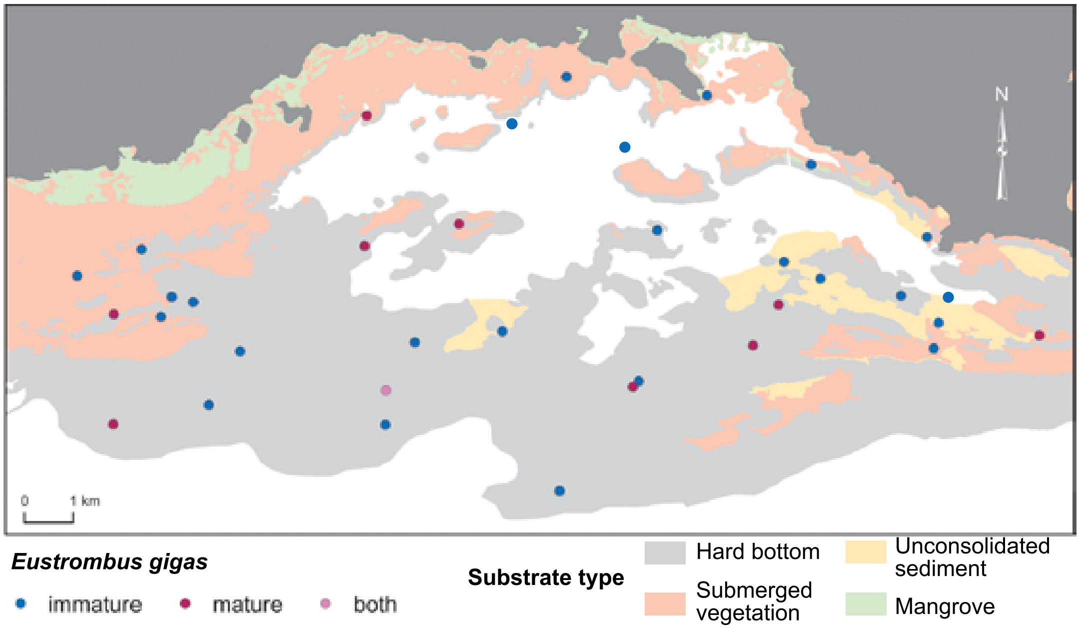


*Figure 2.25. Distribution map of queen conch (*Eustrombus gigas*) in the southwest Puerto Rico study area.*

E. gigas was most abundant in 2006 (n=20) and least abundant in 2005 when no adults were recorded (n=6; Table 2.14). In almost all years abundance was higher in softbottom habitat than hardbottom and lowest in mangroves. Highest density of mature conch was 2 individuals per 100 m^2 of softbottom (Figure 2.26b). The greatest number (n=3) of conch observed at any one site occurred twice: at an unconsolidated sediment site during the January 2005 field mission and again at an unconsolidated sediment site during the August 2006 field mission (Figure 2.26a). Only immature *E. gigas* were recorded during mangrove surveys in 2004 and 2005 (Figure 2.27); no mature *E. gigas* were recorded in mangroves. Immature *E. gigas* were also more likely to be encountered over hardbottom than were mature conch (Figure 2.27a).

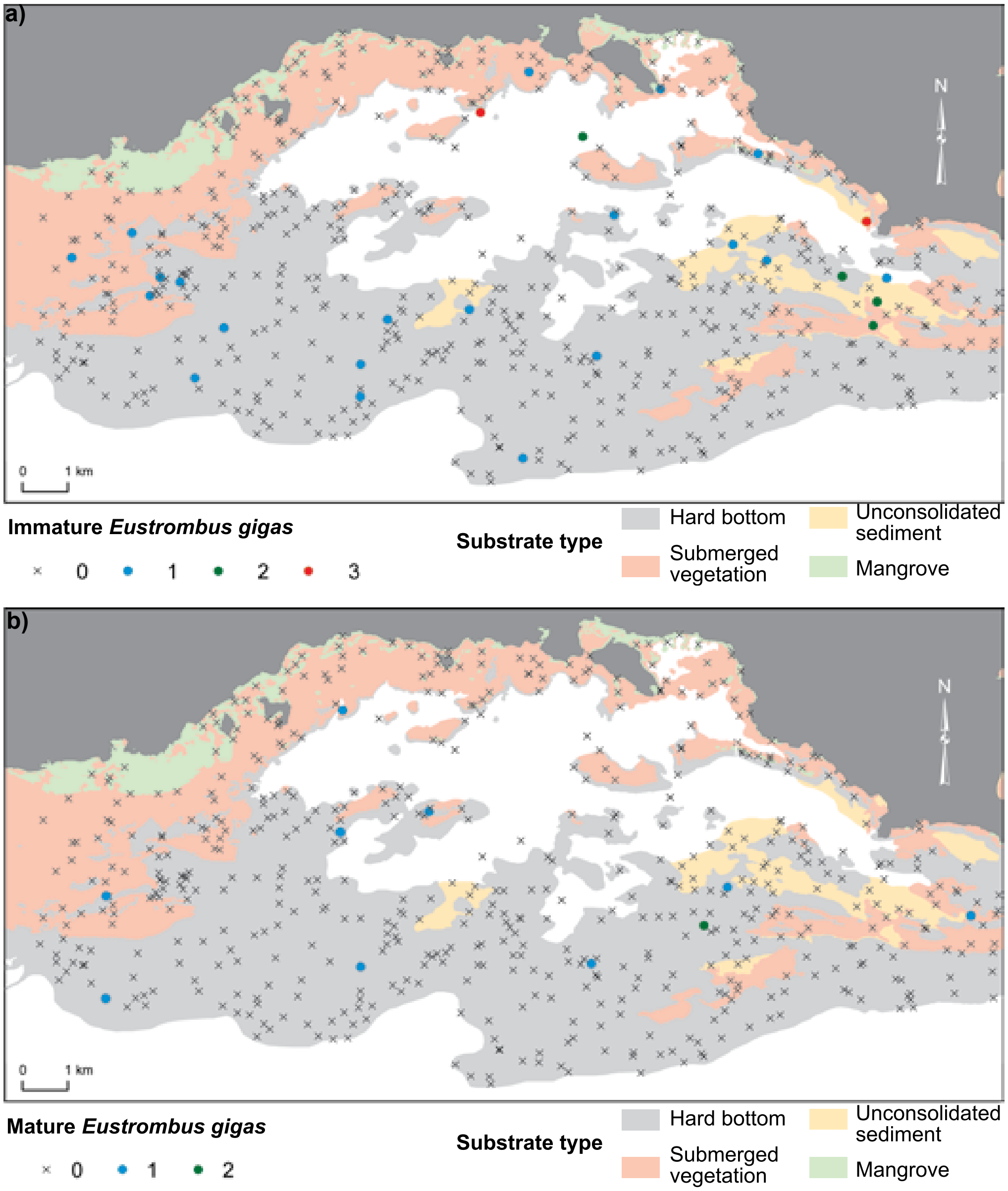


*Figure 2.26. Distribution maps of a) immature and b) mature queen conch (*Eustrombus gigas*) in the southwest Puerto Rico study area.*

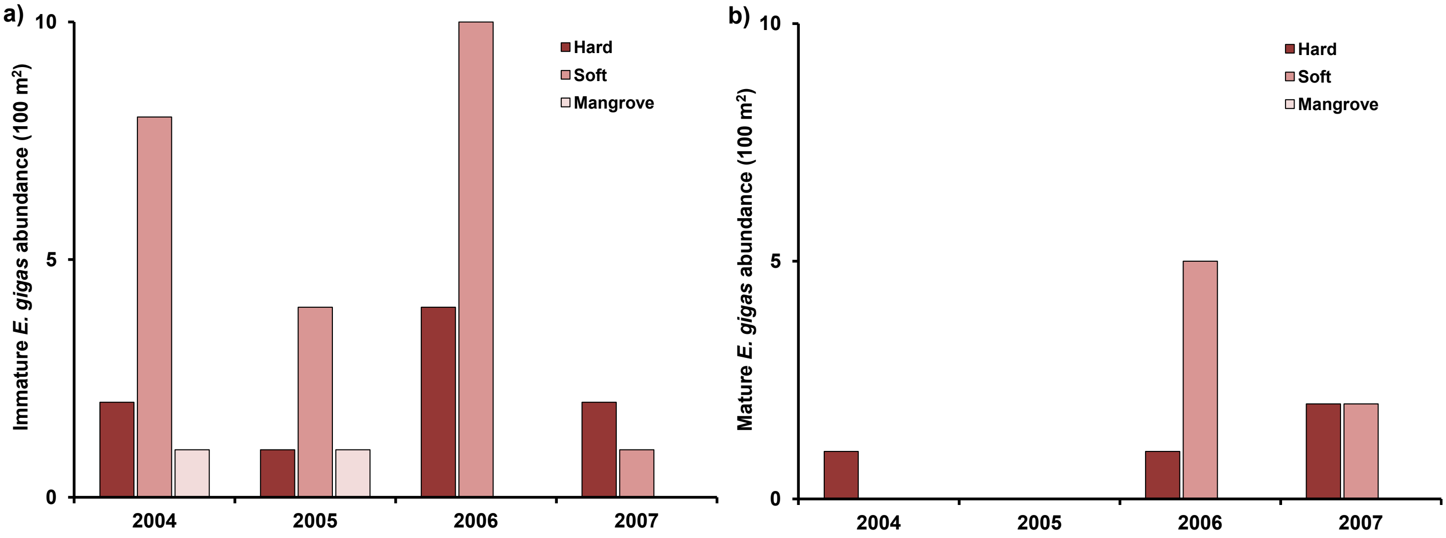


*Figure 2.27. Abundance of a) immature and b) mature queen conch (*Eustrombus gigas*) in the southwest Puerto Rico study area.*

*2.3.4.2. Long-spined sea urchin (*Diadema antillarum*)*

Long-spined sea urchins (*Diadema antillarum*) have been included in habitat surveys since 2006 and have been observed primarily at hardbottom sites (Table 2.15). Across the study area, *D. antillarum* were observed at relatively few sites (n=12) out of 314 surveys, but were clustered in large numbers. A total of 192 individuals from 12 surveys sites were recorded between 2006 and 2007 (Table 2.15). The highest abundance at a single site was 75 individuals observed along a linear reef site. The one softbottom site surveyed with *D. antillarum* present was a seagrass habitat surveyed in August 2006, with 30 individuals recorded, also the second highest density for *D. antillarum* at any one site (Table 2.15).

*Table 2.15. Abundance of long-spined sea urchins (*Diadema antillarum*) observed within transects by year in the southwest Puerto Rico study area.*

Year	Habitat type	Sites surveyed	Sites observed	Total individuals
2006	Hard	91	7	64
2006	Soft	41	1	30
2006	Mangrove	20	0	0
2007	Hard	100	4	98
2007	Soft	42	0	0
2007	Mangrove	20	0	0
Total		**314**	**12**	**192**

*2.3.4.3. Caribbean spiny lobster (*Panulirus argus*)*

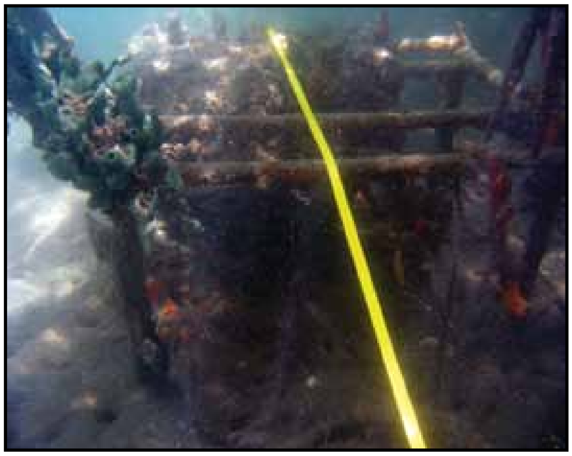
A total of 27 Caribbean spiny lobster (*Panulirus argus*) were observed at 14 of 469 (approximately 3.0%) sites from 2005-2007. Lobster were observed on all three substrate types in 2005, on hard and mangrove sites in 2006, and only on hardbottom habitats in 2007 (Table 2.16). The highest abundance at one survey was seven individuals recorded at a mangrove survey site in August 2006 (Table 2.16).

*Table 2.16. Abundance of Caribbean spiny lobster (*Panulirus argus*) observed within transects by year in the southwest Puerto Rico study area.*

Year	Habitat type	Sites surveyed	Sites observed	Total individuals
2005	Hard	90	2	2
2005	Soft	52	1	2
2005	Mangrove	13	2	6
2006	Hard	91	4	5
2006	Soft	41	0	0
2006	Mangrove	20	1	7
2007	Hard	100	4	5
2007	Soft	42	0	0
2007	Mangrove	20	0	0
Total		**469**	**14**	**27**

2.3.5. Marine debris

Collection of marine debris data began in January 2007. One piece of cloth was recorded in January 2007, and four pieces of debris were seen in August 2007. In addition to the cloth, debris found included a champagne bottle, a plastic cylinder, a PVC rectangular frame and a glass bottle. The area of all debris equaled 1,014 cm^2 with a total affected area of 1,054 cm^2. Debris were colonized by a combination of cyanobacteria, turf algae, sponges, crabs and macroalgae. The sites where debris was recorded ranged in average depth from 1 to 12 meters. Habitat types included hardbottom, scattered coral and rock in sand, and mangrove. The largest amount of debris was seen at the mangrove site on the north side of Magueyes, where a large PVC rectangular frame and a plastic cylinder were found. A champagne bottle was seen on the north side of Media Luna in the sand of a hardbottom site; a glass bottle was north of Corral on a hardbottom site with some sand, and the cloth was found toward the shelf edge on hardbottom.



Marine debris in the mangroves of La Parguera.

2.3.6. Summary of results

- Colonized pavement habitat was the most common habitat type occurring in La Parguera.
- Hardbottom habitat types were dominated by algae, followed by gorgonians and stony corals.
- The cover of live stony coral averaged about 5% on hardbottom throughout the study area.
- Linear reefs represented 10% of the study area, but had the highest percent cover of live coral (approximately 7%).
- Live stony coral cover comprised of at least 24 genera, but only nine had a mean cover greater than 0.01.
- *Montastraea* and *Porites* spp. were the most ubiquitous coral genera among habitat types and had the highest mean cover among observed genera. *M. annularis* complex had the most coverage (35% of total coral cover) whereas *P. asteroides* was the most frequently observed species (62% occurrence).
- On average, *T. testudinum* was the most abundant seagrass species in SAV habitats whereas *S. filiforme* had the most coverage in unconsolidated sediments such as sand or mud.
- Mangrove prop roots provide a major benthic substrate for colonization by epiphytic organism, with macroalgae having the highest coverage of benthic organisms observed thereon.
- Live coral cover was patchy in spatial distribution, and interpolations revealed that live coral cover and taxa richness correlated positively with substrate rugosity.
- *Acropora* species were rarely observed, most likely due to the sampling approach used in this study; *A. cervicornis* had a wider spatial distribution compared with *A. palmata*.
- Highest coral cover was observed along the shelf edge southward of Terrumote I and around El Palo. Cover of *A. palmata* (4.1-10.4 %) was highest around El Palo, whereas highest cover of *M. annularis* complex (25.1-30.3%) wast south-west of Turrumote I.
- Hot spots of coral species richness (10-14 species per 100 m^2) and diversity (H = 1.7-2.5) occurred offshore, but were scattered throughout hard bottom habitat types.
- Highest rugosity (0.8-1) occurred between Margarita Reef and El Palo and at Romero.
- Temporal analysis revealed that live coral cover varied significantly among sampling years. Overall, there was a trend of decreasing coral cover over time.
- Mean algal cover varied significantly over time, with different seasonal patterns being observed for various algal types.
- Densities of *E. gigas* were relatively low, with 76% of observed conch being immature and below the legal size class for the fishery.
- *D. antillarum* were only observed at a few sites, but occurred in large numbers when they were encountered.
- *P. argus* were infrequently observed; with the highest number of individuals were observed in mangrove during 2006.
- The largest piece of debris was seen on the northern side of Magueyes, where a large PVC rectangular frame and a plastic cylinder were found. Debris were colonized by a combination of algae, cyanobacteria, sponges, crabs and shrimp.


Shrimp

2.4. Discussion

2.4.1. Colonized hardbottom habitats: benthic characterization and spatial patterns

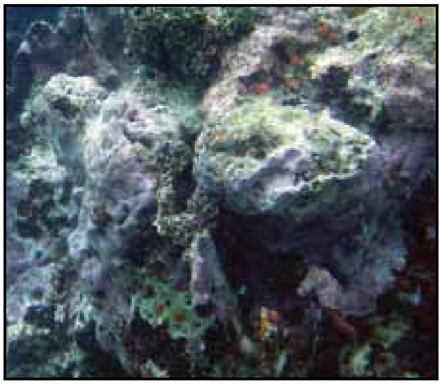
Marine benthic maps of nearshore environments have become an important tool for conservation and management of biological resources. Due to cost restraints, such maps often are produced at low resolutions (i.e., large scales) that typically do not capture the full spectrum of spatial variation in the distribution and composition of benthic resources. A good example of this is the minimum mapping unit (MMU) of 0.4 ha (1 acre or approximately 4,047 m^2) used to develop benthic maps of La Parguera, Puerto Rico and which resulted in the identification of 26 distinct benthic habitat types (Kendall et al., 2001). However, some softbottom polygons (i.e., habitat types) included unidentified benthic features such as sand halos and patch reefs that were smaller than the MMU. These unidentified features are known to influence the spatial distribution and occurrence of marine fauna at multiple scales (Parrish, 1989; Kendall et al., 2003; Chittaro, 2004). Likewise, spatial patterns in benthic composition can be influenced by marine fauna such as fishes and invertebrates at spatial scales more resolved than a MMU of 0.4 ha (Helfman, 1978, 1982; Meyer et al., 1983; Pittman et al., 2004; Burkepile and Hay, 2008). By quantitatively characterizing temporal and sub-meter spatial variation in benthic composition and physical attributes of mapped polygons, this study provided additional information for use in elucidating species-habitat relationships, understanding spatial patterns in the distribution of marine fauna, and identifying faunal effects on benthic composition.

Although the composition of benthic substrates varied spatially within and among habitat types, some general spatial patterns in occurrence and cover of benthic organisms were observed. For example, turf algae – defined as a multispecific assemblage of small filamentous algae – was the most extensively occurring benthic organism group within all hardbottom habitat types, followed by macroalgae and a low occurrence of CCA. A widely accepted hypothesis is that the abundance of algae on coral reefs typically is controlled by herbivory (Steneck and Dethier, 1994). Under high rates of herbivory, coral reefs and hardbottom substrates generally are characterized by low-biomass algal assemblages dominated by turf, with low cover of CCA, and macroalgae (Steneck and Dethier, 1994). In contrast, under low rates of herbivory, macroalgae forms the dominant algal cover type, followed by turf algae, and few crustose coralline algae (Steneck and Dethier, 1994).


Dictyota sp. surrounding *Dichocoenia stokesi*

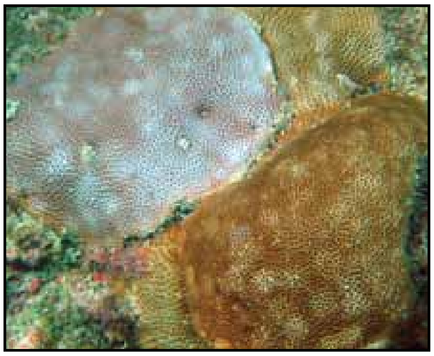
The extensive coverage of turf algae observed during this study suggests that rates of herbivory on reefs and hardbottom areas in la Parguera may be high. This seems counter-intuitive, given the widely accepted belief that a Caribbean-wide decline in the abundance of *D. antillarum* along with serial over-fishing have reduced herbivory and have resulted in a phase-shift from coral-dominated to macroalgal-dominated reefs in the Caribbean (Lessios et al., 1984; Gardner et al., 2003; Hughes, 1994). During this study, the abundance of long-spined sea urchins was relatively low, but small-bodied herbivorous fishes were numerically abundant and dominated fish assemblages across the shelf (Fish chapter, this report). Idjadi et al. (2006) implicated structural refugia from long-lived colonies of *M. annularis* complex in a phase-shift reversal from macroalgae-dominated back to coral-dominated reefs in Jamaica. Bechtel et al. (2006) also implicated a resurgence of *D. antillarum* and an abundance of other echinoids in an observed decline in macroalgal cover and a corresponding increase in scleractinian coral cover on a reef in Jamaica. Based on our data from La Parguera, it is possible that rates of herbivory from relatively abundant fishes and other observed echinoids may have provided enough herbivory to keep the cover of macroalgae relatively low while turf algae flourished.


M. annularis complex


Crustose coralline algae

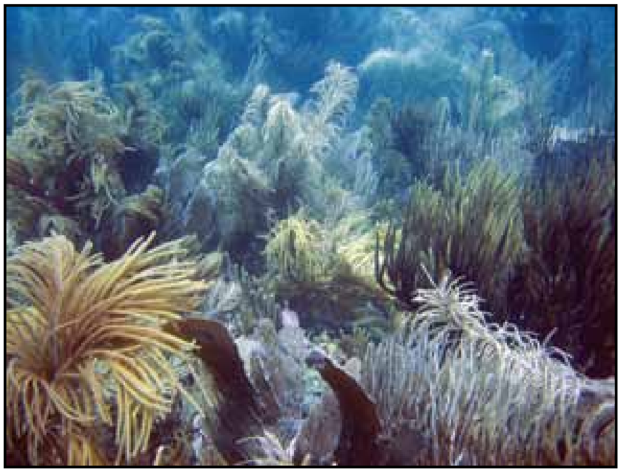
The high cover of turf algae on reef and hardbottom areas in La Parguera also may be due to relatively quick growth rates that provide a competitive advantage to this algal group over slower-growing coral and crustose coralline algae. CCA play a crucial role in coral reef ecology by contributing calcium carbonate to reef structure and by facilitating the settlement and colonization of scleractinian corals (Steneck and Dethier, 1994). Turf algae trap and stabilize unconsolidated sediments, but also rapidly overgrow and kill underlying coralline algae and coral colonies through encroachment (Steneck and Dethier, 1994). In La Parguera, the coverage of CCA was very low, indicating that overgrowth of CCA and corals may have occurred, and that very little suitable substrates exist for newly settling coral colonies.

Another general pattern in benthic composition observed during this study was the low average cover of live scleractinian coral (approximately 5-7%) on reef and hardbottom areas and the virtual absence of *Acropora* spp. thickets on forereefs (0.5-6 m depth) in La Parguera. Such low coral live cover is now typical of most reefs in the Caribbean and has resulted from the synergistic effect of natural and anthropogenic factors operating over the past three decades (Gardener et al., 2003). Acroporid populations were formally very abundant on reefs near La Parguera (Weil et al., 2002). Since the late 1970s however, successive disease outbreaks, periodic hurricanes, bleaching events, predators, and increased anthropogenic activities have contributed to the demise of acroporid and other coral populations in La Parguera (Weil et al., 2002). Following the major outbreak of white band disease, several tropical cyclones decimated shallow-water acroporid populations in La Parguera between 1979 and 1998 and thereby removed a major source of structural complexity from those coral reefs (reviewed in Garcia-Sais et al., 2005, 2008; Weil et al., 2002). No major hurricane has hit Puerto Rico since 1998, but per this study, reefs at depths less than 18 m in La Parguera remain depauperate of acroporid corals and hence also their important function of providing high structural complexity for fish and other organisms. Most recently, the major Caribbean-wide bleaching event in 2005 resulted in additional coral mortality, such that coral cover declined by 40-60% (Garcia-Sais et al., 2008).


Diseased *S. sidera*

Interpolations of this study's synoptic estimates of live coral cover summed across species revealed a few areas of relatively high cover that could be considered hot spots of live coral (see Figure 2.13a, pg. 21). These hotspots may be refuge areas where demographic processes have resulted in coral populations that are resilient to multiple synergistic stressors. If so, corals at these locations are more likely to persist longer in the future than corals at other locations. Additionally, the locations of such hotspots corresponded with areas of relatively high rugosity, coral species richness, and diversity (Figures 2.13b, c, pg. 21). Protection of such hotspots may benefit ecosystem conservation. Interestingly, the five most dominant species in terms of coral cover in La Parguera were two frame building species (*M. cavernosa* and *M. annularis* complex) and three more weedy species (*P. astreoides*, *Agaricia* spp. and *S. siderea*). Frame building corals are important in that they provide structural complexity and are also major contributors to reef growth and persistence, whereas weedy species provide very little complexity and contribute relatively little to reef growth (Hoegh-Guldberg et al., 2007). If these hotspots are to be selected for increased management and protection from anthropogenic stressors, further work is needed to understand the physical and oceanographic properties that correlate with their enhanced ecological features.


S. radians



Dense gorgonian habitat

An interesting feature of reef and hardbottom areas in La Parguera, is the preponderance of gorgonians (sea rods, sea whips sea plumes, and sea fans) at sites throughout the shelf. Also known as soft corals, gorgonians were commonly observed intermingled with other sessile benthic organisms, and often dominated colonized pavement and patch reef habitats in terms of cover. At some colonized pavement sites for example, the canopy created by the high density and cover of gorgonians virtually obscured the sea floor such that other sessile benthic organisms were not visible. Percent cover of soft corals also correlated positively with rugosity, probably because through niche partitioning, more rugose areas provide a wider variety of benthic substrate for settlement and colonization by soft corals than less rugose areas (MacArthur and Levins, 1964). Although they contribute minimally to calcium carbonate accretion and reef growth, soft corals represent important ecosystem components on the La Parguera shelf in that they provide important habitat for fishes and feeding sea turtles (Gratwicke et al., 2006; Blumenthal et al., 2009). Additionally, soft corals can affect the spatial distributions of other sessile benthic organisms because they are superior competitors for space, and they are able to secrete allelo-chemicals that deter growth of other benthic organisms (Fenical, 1987; Harvell and Fenical, 1989).

2.4.2. Softbottom habitats: benthic characterization and spatial patterns

Analysis of the benthic maps used in this study showed that softbottom habitats comprised approximately 33% of the study area (22.1% seagrass, 4.6% macroalgae and 5.9% unconsolidated sediments (Table 2.3, pg. 15). As shown by the spatial interpolations of synoptic estimates from this study, softbottom areas in La Parguera exhibited a zonation pattern typical of Caribbean shallow-water ecosystems; seagrass percent cover was highest near the shore but decreased toward deeper offshore areas. Similarly, the spatial distributions of the two most commonly occurring seagrass species also were zoned. *T. testudinum* dominated shallow water near shore areas down to a depth of 16 m, whereas *S. filiforme* dominated deeper areas offshore. Such zonation patterns result generally from decreasing nearshore-to-offshore gradients of nutrients and light penetration. Sponges and native coral species (e.g., *Dichocoenia stokesii*) also were observed frequently in seagrass and macro algae habitats. Calcareous macroalgae (e.g., *Halimeda* spp., *Udotea* spp., and *Penicillus* spp.) were commonly encountered on soft bottom habitats, but their percent cover was low relative to those of seagrass and more foliose algae such as *Lobophora*, *Dictyota*, and *Padina* spp. Nevertheless calcareous algae are ecologically important to coral reef communities because their skeletal remains (e.g., *Halimeda* spp.) are a major component of carbonate sediments occurring within coral reef ecosystems (Hubbard et al., 1981; Drew, 1983).

T. testudinum

S. filiforme

Interestingly, benthic organisms typical of coral reefs and hardbottom substrates (e.g., turf algae, CCA, scleractinian corals and gorgonians) occasionally were encountered within areas mapped as softbottom. The atypical occurrence of these reef-associated organisms within softbottom polygons was likely an artifact of differences between the scale at which map polygons were delineated and the scale at which benthic data were collected. An MMU of 0.4 ha did not allow delineation of reef and hardbottom patches less than 0.4 ha that were encountered within areas mapped as softbottom. Thus, our fine-scale 1 m^2 quadrat benthic surveys on these hardbottom patches that occurred within softbottom areas provide additional data that may be crucial for understanding observed relationships between faunal species and their mapped habitats.

Several studies have shown that softbottom habitats are ecologically important components of coral reef ecosystems. For example, reef fishes are known to migrate from reef and hard bottom areas, forage on adjacent non coral reef habitats (sand, seagrasses, algal plains), and they represent a trophic pathway of energy transfer among habitats (McFarland et al., 1979; Meyer et al., 1983). Furthermore, several landscape analyses have correlated various seagrass metrics with increased probability of juvenile grunt occurrence on reef and hardbottom areas in St. Croix (Kendall et al., 2003), higher sighting frequencies of groupers on hardbottom habitats in the Florida Keys (Jeffrey, 2004), and increased fish abundance and species richness in mangrove communities in Puerto Rico (Pittman et al., 2007a). Several other studies have demonstrated that both vegetated and non-vegetated softbottom areas are known to provide habitat and food for several coral reef fishery species, endangered and threatened species, and many other marine organisms (Parrish, 1989; Nagelkerken et al., 2000; Dahlgren and Marr, 2004; Adams et al., 2006). Fine-scale benthic characterizations, such as those conducted during this study, should provide additional information to further explain these faunal species-habitat relationships.

2.4.3. Benthic characterization of mangrove habitats

In La Parguera, mangroves are very distinctive intertidal and near-shore habitat features which are dominated by red mangrove (*Rhizopora mangle*). Extensive mangrove stands occur along the shoreline as well as form islands in the back reef lagoon areas of the La Parguera coral reef ecosystem, where they support abundant fish populations (Christensen et al., 2003; Pittman et al., 2007b). The tidal range in La Parguera is less than 0.5 m, thus most prop roots seaward of the mangrove community are continuously submerged. Our benthic characterizations of mangrove habitats found extensive cover of benthic organisms, particularly algae, sponges and hydroids, on both the seafloor and as epiphytes on submerged prop roots. The high mean cover (>50%) of vegetation observed on benthic substrates indicates that the La Parguera mangroves are highly productive systems which may be providing enough food and nutrients to support resident fish and invertebrate populations. Additionally, many sites had thick deposits of detritus, suggesting that mangroves were performing their function of entrapping sediments.


Sponges on mangrove prop root


Tunicates on mangrove prop root


Mussels on mangrove prop root

The benthic characterizations of mangrove habitats reported in this study have provided information that could be used as covariates to explain spatial variance in fish populations among mangroves and other benthic habitats. Further research is necessary however to identify additional characteristics of mangrove communities that may vary spatially and may be influencing benthic composition and faunal distributions. For example, mangroves are major producers of detritus that may be contributing to offshore productivity (Odum and Heald, 1972, 1975). However, rates of productivity and generation of detritus in mangroves will vary based on location, oceanographic properties, composition of submerged vegetation, and frequency of tidal flushing.

2.4.4. Temporal trends in benthic composition on hardbottom habitats

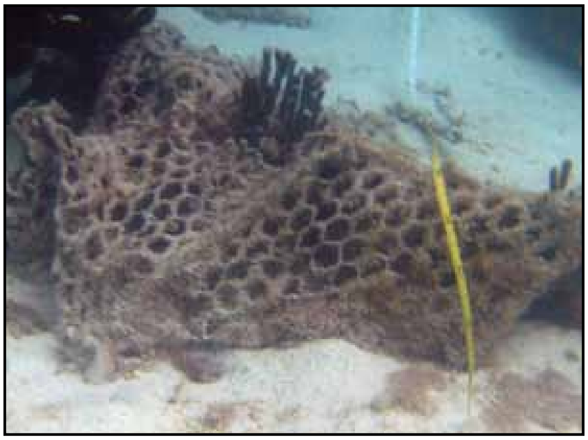
Temporal analysis of data on percent cover revealed a general decrease in live coral in La Parguera, particularly on pavement habitats between fall of 2003 and summer of 2007. Our observations of a temporal decline in live coral cover at La Parguera are consistent with those of other recent studies from Puerto Rico and the USVI. Temporal declines ranging from 40-50% in live coral were reported at several sites in Puerto Rico including reefs off Isla Desecheo, Mayguez, Guanica and Ponce, with most of the loss occurring after the 2005 bleaching event (Garcia-Sais et al., 2008). At Buck Island, St. Croix, mean estimates of live coral cover on reefs were lowest in 2006 after four years of observations (Pittman et al., 2008; Clark et al., 2009). Similarly, in St. John, average live coral cover declined from 21.4% in 2005 to 8% by October 2007 (Miller at al., 2009). Much of the reported loss in live coral occurred in a few species, namely *M. annularis* complex, *C. natans* and *Agaricia agaricites* (St. John), *M. annularis* complex (Puerto Rico), and *M. annularis* complex and *Agaricia* spp. (Buck Island, St. Croix). After the drastic decline in acroporid corals, *Montastraea* remained one of the most abundant coral species in La Parguera (per this study) and in other areas of the U.S. Caribbean (Garcia-Sais et al., 2008; Rothenberger et al. 2008; Miller et al., 2009). Given their dominance and key ecological roles as reef-building species, the recent declines in the cover of *Montastraea* species represent a severe degradation to already fragile reef ecosystems.


Colpophyllia natans

Such coral loss and reef degradation have ultimate ecological and economic consequences. However, there is still a lack of understanding about the ecosystem properties that confer resilience and sustainable ecological function to coral reefs (Done, 1992). Consequently, further research is needed to identify areas within near-shore ecosystems with physical and ecological properties that correlate well with enhanced ecosystem resilience to multiple stressors. Identification of such areas can help managers design and manage protected areas to promote ecosystem conservation. Our characterizations of benthic composition and descriptions of spatial patterns at La Parguera provide a foundation for identifying locations with enhanced ecological properties that may be resistant and resilient to manageable anthropogenic stressors such as over-fishing, land-based sources of pollution, and habitat destruction.

2.4.5. Marine debris

Marine debris, as defined by NOAA's Marine Debris program, is "any persistent solid material that is manufactured or processed and directly or indirectly, intentionally or unintentionally, disposed of or abandoned into the marine environment or the Great Lakes" (http://marinedebris.noaa.gov/info/welcome.html, accessed 26 Jan 2010). There are several types of marine debris from a variety of sources that have multiple impacts on the marine environment. Marine debris is composed mainly of plastics, glass, metal, rubber, derelict fishing gear and derelict vessels. The types of marine debris commonly seen in the waters around La Parguera include plastics, glass, rubber and derelict fishing gear.


Derelict trap

Marine debris originates from ocean-based and land-based sources including cargo ships, fishing and other vessels, littering, dumping, poor waste management, storm water discharge and extreme natural events such as hurricanes and floods. The effects of marine debris are difficult to measure but affect the aesthetics of shorelines and coastal environments world-wide. Debris can also cause damage by scouring, breaking or smothering important and fragile marine habitats and entangling wildlife such as sea turtles, whales and fish. Ingestion of marine debris can prove fatal to animals by leading to loss of nutrition, internal injury or blockage and starvation. Marine debris can transport organisms from native habitats to non-native ones, serving as the vector for potentially catastrophic alien species introductions. Marine debris can be dangerous to safe navigation of vessels and has in the past caused vessel damage and loss of navigation abilities of large NOAA research vessels.

The debris data collected in La Parguera from 2006-2007 revealed only four locations of debris. It is important to recall that only debris on the 25 x 4 m transect was recorded, explaining the low number of sites with debris. Several (unquantified) pieces of debris were seen outside transects.

The marine debris around La Parguera may have a detrimental economic impact due to degradation of the marine habitat that tourism and human health depend upon. Boat damage, water pollution from storm runoff, and health issues are all potential effects. Further studies into the amount, distribution and affects of marine debris around the study area will provide useful information for future action and protection of this marine environment.

2.4.6. Macroinvertebrates

*2.4.6.1. Queen conch (*E. gigas*)*

Queen conch are ecologically important components of faunal assemblages that occur in Caribbean coastal ecosystems. They are important herbivores that feed on micro- and macro-algae (Stoner ,1997), become prey for several marine organisms such as reptiles, fishes, and crustaceans (Randall, 1964), and their populations support commercial fisheries that were valued as much as U.S. $30 million in 1992 (Appeldoorn and Rodriguez, 1994). However, Caribbean-wide declines in annual queen conch landings – most likely from overfishing and habitat degradation – resulted in the species being listed as commercially threatened and being protected under Appendix II of CITES in 1992 (Wells et al., 1985; Appeldoorn, 1994). Protection under Appendix II means that management of queen conch stocks and monitoring of exports are necessary to prevent extinction of the species.

Reported commercial landings of queen conch from Puerto Rico decreased from 402,510 pounds in 1983 to 90,947 pounds in 1992 (CFMC, 1996). Although reported conch landings subsequently increased in 1993 to 164,612 pounds and to as much as 281,378 pounds by 2002, the Puerto Rican queen conch fishery has not rebounded to 1983 levels (SEDAR, 2007). It is likely that annual landings of queen conch from La Parguera mirrored trends in the reported conch landings for Puerto Rico. Queen conch is a major fishery in La Parguera. Conch meat (locally known as "carucho") is an easily accessible source of protein for locals because extensive shallow seagrass and algal beds typically inhabited by conch are in close proximity to coastal villages and towns. Additionally, conch meat is a staple on the menu of many local restaurants (pers. obs). Conch assemblages in La Parguera are fished both by local artisanal (recreational) fishers for home consumption (pers. obs) and by commercial fishers from Puerto Rico’s west coast (Appeldoorn, 1992). Landings from west coast fishers accounted for a substantial proportion of conch landings reported for Puerto Rico (Appeldoorn, 1992; SEDAR, 2007). It is unclear what proportion of conch landings from La Parguera are taken by artisanal fishers because most of their catch is unreported, but artisanal catch was estimated to be 35% of commercial landings in 1986 (CFMC, 2005; Matos-Caraballo, 2004; Diaz, 2007). If this estimate is realistic, then it is reasonable assume that both artisanal and commercial fishing over the past few decades has had negative demographic consequences on conch assemblages in La Parguera.

Concern that historical over-fishing may have been decimating queen conch populations resulted in the implementation of federal and territorial regulations during the past 16 years to reduce the harvest of conch from U.S. Caribbean territories (CFMC, 1996). Implemented regulations applicable to Puerto

Rico include: (1) the requirement of a territorial fishing license and permit since 1994 to catch conch; (2) prohibiting the use of hookahs as fishing gear within Puerto Rico's Exclusive Economic Zone (EEZ) since 1997 and within territorial waters since 2004; (3) a legal size limit of 23 cm (9 inches) or greater in total shell length and lip thickness greater than 3 mm (1/8th of an inch) the EEZ since 1997 and for territorial waters since 2004; (4) maximum daily quotas of 150 queen conch per commercial fisher or 450 per commercial fishing vessel and three queen conch per recreational fisher or 12 per recreational vessel from the EEZ since 1996 and from territorial waters since 2004; (5) complete closure of the queen conch fishery within Puerto Rico's EEZ since 1997; and (6) an annual seasonal closure from July 1st through September 30th within the EEZ since 1996 and within Puerto Rico's territorial waters since 1997 (CFMC, 1996; SEDAR, 2007). Interestingly, there was a seven-year delay in the implementation of daily quotas and gear restrictions in territorial waters compared with federal (EEZ) waters.

The federal and territorial regulations for queen conch were implemented to reduce harvest, with the expectation that if fishing mortality on conch assemblages was reduced, then *in situ* abundance and occurrence of conch would increase over time. The data collected by this study suggest that regulations have not been effective in protecting queen conch assemblages at La Parguera. The densities and occurrence of queen conch observed in La Parguera region were relatively low. This study surveyed 6.18 ha and observed queen conch at only 6% of the 618 surveys region between August 2004 and August 2007. Average queen conch density was 0.073 individuals per 100 m^2, with highest density being two adult mature individuals per 100 m^2. Additionally, the maximum number of conch observed during a sampling mission was 12 individuals in August 2006. Not only was observed conch abundance very low in La Parguera, 76% of conch encountered were immature (i.e., their shells had not yet developed a lip) and were below the legal size class for the fishery.

Data from a previous study that surveyed 81 randomly selected sites in La Parguera observed similarly low densities of queen conch between May 1985 and April 1986. Torres Rosado (1987) visually surveyed 40.81 ha of bottom substrate and observed 331 queen conch, with a density of 0.08 conch per 100 m^2. Of the 331 queen conch observed, 227 (68%) were immature and had shells without lips. The similarity in queen conch densities observed during 1985-1986 and those observed during this study period (2001-2007), suggests that queen conch stocks were already depressed in 1985, and have not rebounded since then. Stock assessment studies indicate that queen conch requires minimum population densities of 56 individuals per ha for successful reproduction and recruitment (Stoner and Ray-Culp, 2000), although observations in the Florida Keys have also revealed that successful mating occurred when densities were 200 individuals per ha or higher (Bob Glazer, pers. comm). Density of queen conch was only eight conch per ha during 1985-86 (Torres Rosado, 1987) and only seven conch per ha during 2001-2007 (this study). It is very likely that densities of queen conch in La Parguera now are too low for successful reproduction and recruitment, which may be preventing queen conch abundance and occurrence from increasing.

E. gigas

It is possible too that the territorial and federal fishery regulations have reduced fishing mortality on queen conch, but that the demographic effects of this reduction are yet to be seen. The sites surveyed during this study were all within territorial waters, where daily quotas and gear restrictions have been in place for only six years; this may not have been sufficient time for queen conch abundance and occurrence to increase. Also, we have no comparable fishery-independent survey data on the abundance and occurrence of queen conch from the EEZ, where fishery regulations have been in effect for a longer time. According to local fishers, queen conch populations have rebounded within the EEZ. At recent stock assessment workshops (http://www.sefsc.noaa.gov/sedar/), west coast Puerto Rican fishers insist that at La Parguera, most conch occur off-shore toward the shelf edge in deeper waters at 30 m rather than at shallower depths where visual surveys for conch were conducted (Diaz, 2007). Benthic substrates that occur at depths greater than 30 m however, were beyond the sampling domain of this study.

*2.4.6.2. Long-spined sea urchin (*D. antillarum*)*

The long-spined sea urchin is another important component of faunal assemblages found in coral reef ecosystem. It is considered a major herbivore that controls macroalgae abundance on Caribbean coral reefs (Lessios et al., 1984; Lessios, 1998). Prior to the massive Caribbean-wide die-off of *Diadema* in the late 1980s, urchins were present throughout the wider Caribbean in most habitats including coral reefs, seagrass beds, rocky shores, softbottom and mangrove; they were abundant in shallow areas down to 15 m, with some found as deep as 40 m (Randall et al., 1964; Sammarco, 1972; Weil et al., 1984). The Caribbean-wide mass mortality of *D. antillarum* was estimated to be greater than 93% (Lessios et al., 1984; Lessios, 1988), and this had catastrophic effects on reef health (Knowlton, 2001). With the disappearance of these keystone herbivores from the reefs, many Caribbean reefs became dominated by macroalgae, and the recruitment of scleractinian corals was inhibited (Edmunds and Carpenter, 2001). Diseased urchins were first reported in Puerto Rico in January 1984 near Laurel reef in La Parguera (Vicente and Goenaga, 1984).

D. antillarum

A total of 192 long spined sea urchins (*D. antillarum*) were recorded on only 12 of 314 (3.8%) sites surveyed, and had an overall density 0.53 per 100 m^2. These observations suggest that the abundance of long-spined sea urchins on reefs at La Parguera is low, but the estimates being reported here may not precisely reflect *in situ* abundance. Sea-urchins can be very patchy in their distributions, and it may be that survey effort (i.e., the number of sites surveyed) was inadequate to characterize population abundance at La Parguera. For example, one of the twelve sites surveyed contained 75 of the 192 individuals recorded. In addition, long-spined sea urchins were enumerated as part of generalized surveys that were intended to characterize benthic composition but were not optimized to search and identify urchins only. For example, in extremely rugose and complex reef habitats, long-spined urchins could be located in deep crevices, holes or behind overhangs, where they would be undetected unless exhaustive searches were made to locate them.

Nevertheless, the occurrence of *D. antillarum* on La Parguera reefs is an indication of potential population recovery, given the recent mass mortality of the species. La Parguera has higher densities of urchins than other localities in the Caribbean, with large numbers of reproductive adults, abundant juveniles and no signs of disease or unhealthy individuals (Weil et al., 2005). Weil et al. (2005) also reported that the distribution of urchins across the La Parguera region is patchy, but suggested that recovery of local populations may be occurring. The sampling design used in this study and the data generated would be useful in detecting long-term increases in the sea urchin density, if such changes were to occur. More targeted surveys that monitor urchin population recovery may be required.

*2.4.6.3. Caribbean spiny lobster (*P. argus*)*

The stratified random sampling design utilized by this study also provided limited opportunities to conduct spiny lobster surveys and determine their broad-scale distribution in the La Parguera region. Lobsters were rarely encountered during surveys and were observed only at approximatrely 3.0% of 3 sites visited between 2005 and 2006. However, sightings of spiny lobsters during surveys were opportunistic rather than deliberate, and exhaustive searches of overhangs and crevices were not conducted to determine lobster presence because of time dive time limitation. Thus, it is quite likely that the sightings reported from this study underestimated the frequency of lobster encounters in La Parguera.

Like queen conch however, spiny lobster is a treasured local delicacy in La Parguera, and local populations are targeted by both commercial and artisanal fishers. Stock assessments conducted by over the past two decades have indicated that the spiny lobster fishery in Puerto Rico has shown signs of overfishing; and landings, catch rates, and relative abundance have declined significantly since the fishery began in 1969 (Morris et al., 2004). Analysis of data on spiny lobster landings from Puerto Rico indicates that the fishery is concentrated around the southwestern shelf, which includes the La Parguera region (Valle-Esquivel, 2005).

Fishing pressure on spiny lobster is controlled under federal regulations implemented by the CFMC since 1985 and by territorial regulations implemented by DNER since 1936 (CFMC, 1985). Regulations include prohibiting the harvest of females with eggs and individuals measuring less than 9 cm (3.5 inches) in carapace length; barring the use of chemicals, explosives, poisons, drugs, spears, and hooks or similar devices to harvest lobsters; requiring the use of traps with self-destruct panels; and limiting entry into the fishery to only fishers with a permit (CFMC, 1985). It is doubtful whether these regulations have been effective however, given that between 1985 and 1989, undersized lobsters accounted for 40% of the total lobster catch from Puerto Rico (Bohnsack et al., 1991; Morris et al., 2004). In summary, better enforcement of existing regulations may be needed to improve lobster abundance over time, and a dedicated lobster-monitoring program will be needed to document long-term changes in lobster populations in La Parguera.

P. argus

2.5. References

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Chapter 3. Fish Communities, Groups and Species

3.1. Introduction


Balloonfish (*Diodon holocanthus*) in turtle grass.

Previous studies of fish distributions and diversity across the shelf at La Parguera have either been limited to selected coral reef locations or specific habitat types and have been brief snapshots of communities from discrete periods within a single year or two. Such data do not provide sufficient spatial information to develop comprehensive management plans across the complex multiple-habitat seascapes that characterize the La Parguera Natural Reserve. In addition, to determine changes over time, sampling must be conducted on a regular basis across the region. In response, the current Coral Reef Ecosystem Monitoring (CREM) project was designed to provide information on fish community structure and change across all components of the seascape and to examine inter-annual and seasonal changes and multi-year trends for fish species and assemblage biomass, abundance and diversity.

The CREM data were first utilized in 2003 to examine fish communities and their distribution across the shelf at La Parguera (Christensen et al., 2003). The study examined data on fish species distributions and size classes from multiple habitat types across multiple zones (lagoonal, etc.) and found that the type of habitat (i.e., mangrove, seagrass, hardbottom) was a better predictor of the spatial distribution of fish, particularly snappers (Lutjanidae) and grunts (Haemulidae), than the location across the shelf. For several fish species, distinct body size dependent distribution patterns were considered to be indicative of ontogenetic shifts in habitat use, with smallest individuals more abundant in seagrasses and mangroves and larger adults more abundant at coral reef sites. In similar studies, Aguilar-Perera and Appeldoorn (2007, 2008) also highlighted the importance of connectivity between mangroves, seagrasses and coral reefs, particularly for the many species of fish that undertake multiple habitat shifts in their life-cycle. In addition, CREM data were utilized to examine the ecological relevance of a mosaic of habitat types to fish (Pittman et al., 2007a) demonstrating that fish diversity and abundance in mangroves is influenced by the marine habitat types adjacent to mangroves. For instance, mangroves and seagrasses that coexisted in close proximity (≤100 m) supported higher fish species richness and abundance than mangroves with adjacent unvegetated sediments. For some species, such as yellowtail snapper (*Ocyurus chrysurus)*, the close proximity of both seagrasses and coral reefs explained the observed distribution patterns.

Other studies have found that the topographic complexity of the surrounding coral reef ecosystem at La Parguera is also important in explaining patterns of fish species diversity and abundance across the shelf. Work by Kimmel (1985) conducted in 1980 and 1981 provides a rare example of a spatially comprehensive fish-habitat study of fish communities from 21 discrete biotopes across the insular shelf at La Parguera. These data provide an important historical reference point with which to compare changes in fish species occurrence. Kimmel's work also highlighted the importance of integrating topographic complexity into the definition of habitat types and the potential for predicting fish assemblages from mapped habitat types. Recent studies in Puerto Rico using NOAA CCMA Biogeography Branch (CCMA-BB) fish survey data demonstrated the importance of topographic complexity for predicting fish species richness and other key metrics (Pittman et al., 2007b; Pittman et al., 2009) in the La Parguera region and neighboring USVI. Spatial predictive modeling using regression trees and geographic information system (GIS) was used to predict and map fish species richness across the entire seascape at La Parguera, with greater than 70% map accuracy (Appendix E).

While spatial patterns in species distributions are becoming better known, comparatively little is known about the temporal change that may be occurring in both the size and abundance of fish and macroinvertebrates. This report provides both a spatial and temporal characterization of fish community metrics, fish species (with emphasis on fishery species), macroinvertebrates and benthic habitat composition using underwater survey data collected during a seven year period (2001-2007). This section of the report focuses on the spatial distribution of species and community metrics (i.e., composition, species richness, biomass, abundance of assemblages) and the temporal patterns (2001-2007) across mosaics of habitat types in the study area at La Parguera, southwest Puerto Rico. The intention is to provide a spatial and temporal characterization for the area and does not therefore establish relationships between environmental structure and fish distributions, which will be the focus of subsequent publications. Fish communities are highly heterogeneous in time and space and can also function as indicators of ecosystem integrity and health. Examination of fish community composition and fish species distributions provides important baseline information for ecological studies, as well as, critical information to support resource management decision making with regard to understanding essential fish habitat, identifying where species of concern are located, identifying diversity and productivity hotspots, prioritizing activities in marine protection, mapping environmental sensitivity, designing restoration strategies and monitoring programs.

3.2. Methods

3.2.1. Survey Data

Fish surveys were conducted along a 25 m long by 4 m wide belt transect (100 m^2) using a fixed survey duration of 15 minutes (Menza et al., 2006; Figure 3.1). The number of individuals per species is recorded in 5 cm size class increments up to 35 cm using the visual estimation of fork length. Individuals greater than 35 cm are recorded as an estimate of the actual fork length to the nearest centimeter. A benthic habitat map was used to develop and implement a stratified-random sampling design based on two strata: hard and softbottom habitat types to minimize variance in population estimates and maximize the power to detect changes (Figure 3.2). A total of 1,167 fish surveys (572 from hardbottom and 595 from softbottom habitats, including mangroves) were used in this analysis. For a detailed description of CCMA-BB's fish census survey methods see Appendix D.

Figure 3.1. NOAA trained observer recording fish census data along the belt transect.

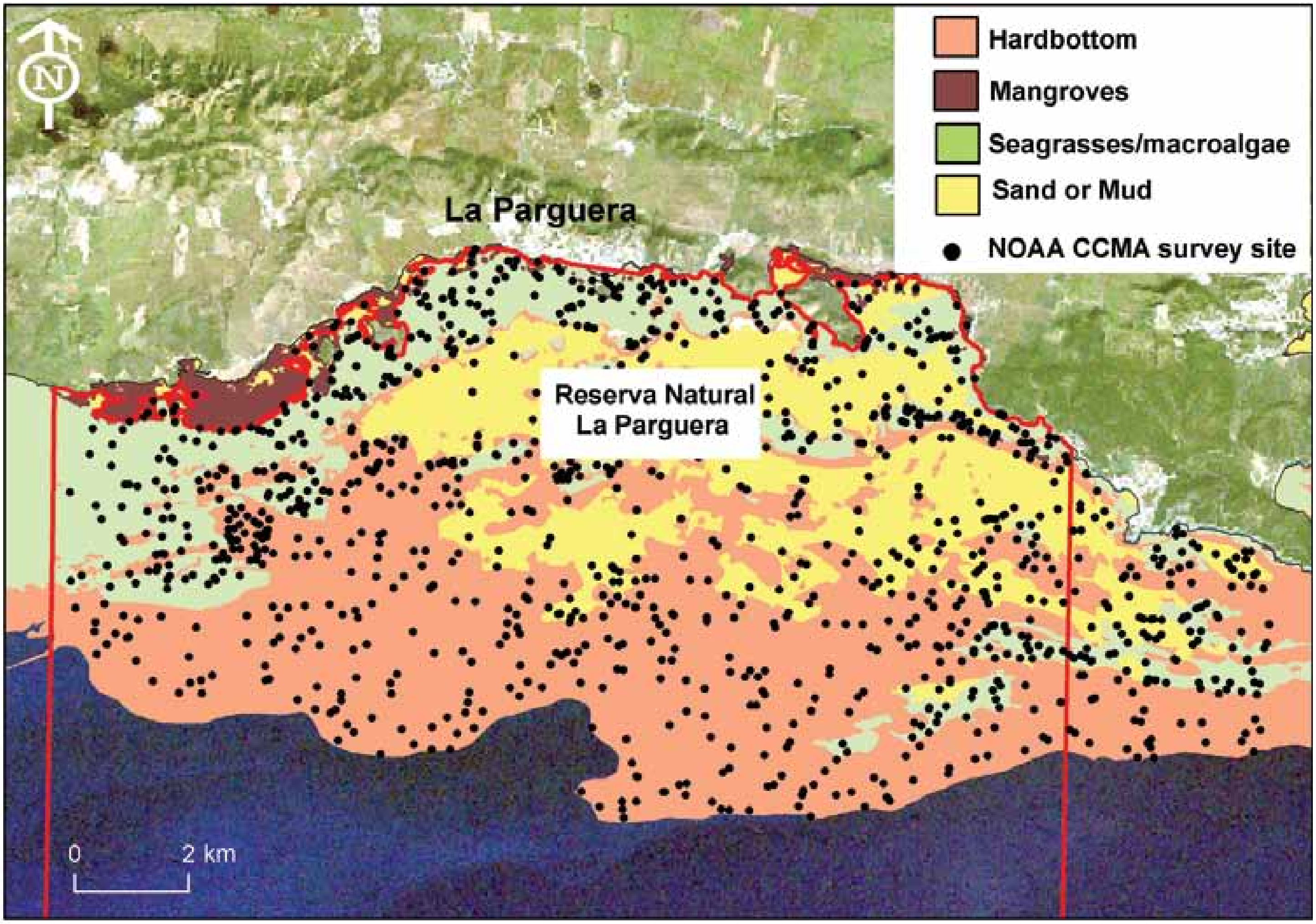


Figure 3.2. NOAA's benthic habitat map showing the major classes of habitat type across the study region of Reserva Natural La Parguera. Survey sites were allocated at spatially random locations distributed within the strata of hardbottom and softbottom benthic classes between 2001 and 2007.

3.2.2. Data analysis

3.2.2.1. Benthic composition and fish community composition

Differences and similarities in the species composition of communities between samples (often referred to as assemblage or community structure) were examined using a species-abundance by site data matrix. Infrequently observed fish that were not identified to species level were removed. The matrix was fourth-root transformed to ensure that rare and intermediate abundance species, in addition to the highly abundant species, played a significant role in determining patterns in community composition. The data was then used to construct a matrix of the percentage similarity in community composition between all pairs of sites using the Bray-Curtis Coefficient,

$$S'_{jk} = \left[1 - \frac{\sum_{i=1}^{n} X_{ij} - X_{ik}}{\sum_{i=1}^{n} X_{ij} + X_{ik}} \right]$$

where x_{ij} is the abundance of the *i*th species in the *j*th sample and where there are *n* species overall.

This algorithm is considered a robust estimator of ecological distance and has had widespread usage in ecology particularly for comparison of biological data on community structure (Faith et al., 1987). Its robustness is in part due to its exclusion of double zeros, that is, if two samples are missing the same species, they will not be regarded as similar based on the same absentees (Legendre and Legendre, 1998). This similarity coefficient reduces the comparison between all pairs of samples to single numerical values that are arranged in a secondary matrix from which pattern is examined.

An ANOSIM test, a multivariate version of Analysis of Variance (ANOVA; Primer v6; Clarke and Warwick, 1994), with 999 permutations was used to test for significant differences in fish assemblage composition between mapped classes at multiple thematic resolutions including: 1) HABITAT Structure - soft, hard and mangrove; 2) all habitat TYPES (see Table 3.1), and 3) modifiers for seagrass (e.g., classes of percentage cover). The R value is a better relative indicator of the amount of dissimilarity between groups than the significance test and is thus given greater emphasis here. The R is usually interpreted as the pairs of fish assemblage composition being: R<0.25 = barely separable; R>0.5 = overlapping, but clearly different and R>0.75 = well separated.

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

3.2.2.2. Species habitat association

Species-habitat associations were determined by overlaying fish survey points (start of the 25 m transect) on the NOAA benthic habitat map and linking to the class of habitat type at the point location.

3.2.2.3. Inter-annual Patterns

Inter-annual patterns were examined by comparing means using ANOVA and SigmaPlot© (SAS Institute, 2006) in a wide range of community metrics and individual species data amongst years. Data were tabulated and where means decreased significantly from one year to the next then a red arrow was assigned and if increased significantly then a green arrow was assigned. Consecutive years of significant decline or increase where denoted with double arrows.

3.2.2.4. Seasonal Patterns

Seasonal patterns across years (2004-2007) were examined by grouping fish data into winter (December-March) and summer field seasons (August) and then plotting means (± SE) using bar charts.

3.3. Results

3.3.1. Spatial distribution patterns and species-habitat associations

3.3.1.1. Fish assemblages

Overall a total of 210 species were identified to the species level, with at least another 14 fishes identified to genera. For the complete list of species size details and summary information refer to Appendix B. At the coarsest thematic resolution within the hierarchical classification scheme, using mapped classes of hardbottom, softbottom and mangroves, fish assemblages were significantly different (ANOSIM R=0.54, p<0.01). Greatest difference in fish assemblage composition was found for hardbottom versus mangroves (ANOSIM R=0.83, p<0.01), which were very distinct from one another. In contrast, softbottom and hardbottom fish assemblages exhibited overlap (ANOSIM R=0.32, p<0.01; Figure 3.3).

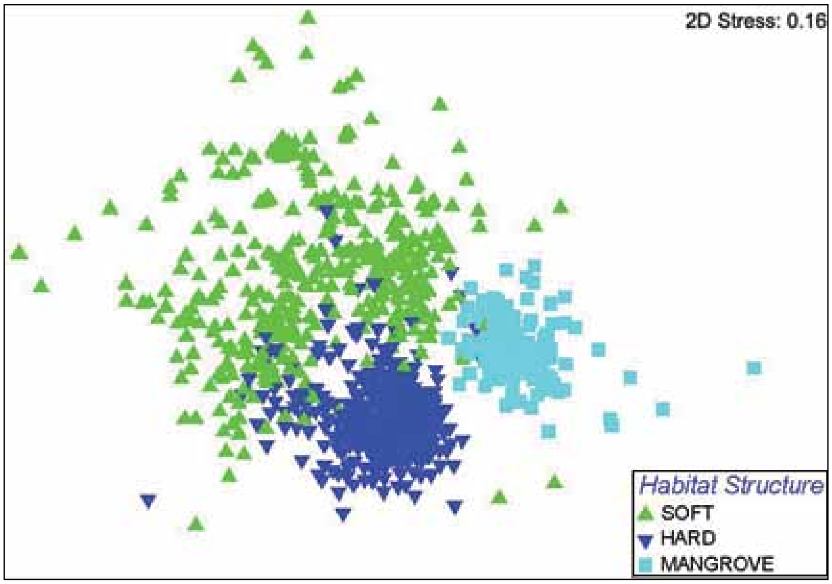


Figure 3.3. Multi-dimensional scaling plot (nMDS) showing between site similarity in fish assemblages composition. Points are individual survey sites that have been assigned a benthic habitat class based on geographical location within a mapped class of the NOAA benthic habitat map. Stress <0.2 indicates an adequate two-dimensional representation of the data.

At the more detailed thematic level of habitat TYPE (all hard and soft types included), fish assemblages were less distinct from one another (ANOSIM R=0.39, p<0.01).

Jenkinsia sp. and schoolmaster (*Lutjanus apodus*)

At the level of habitat TYPE, fish assemblages within mangroves were significantly well separated from all hardbottom and softbottom habitat types, with most overlap existing between fish assemblages of seagrasses and mangroves (Table 3.1). Fish assemblages associated with patch reefs were least distinct and showed high overlaps with all soft- and hardbottom habitat classes, but were more similar to other colonized hardbottom habitat types than to macroalgae and sand. Individual patch reefs and aggregated patch reefs were barely separable although a statistically significant difference was calculated. Fish assemblage composition of macroalgae and sand and macroalgae and seagrass habitat types were not significantly different and exhibited very high overlap. Fish assemblages associated with linear reefs, colonized pavement with sand channels, colonized pavement and the two classes of patch

Table 3.1. ANOSIM R values measuring fish assemblage similarities between samples grouped by benthic habitat TYPE. Uncommon habitat types were not shown. R values in bold are statistically significant (p=<0.01). Dark green= well separated, medium green= some overlap, light green= barely separable.

Mapped habitat type	Patch Reef (Ind)	Patch Reef (Agg)	Linear Reef	Colonized Pavement	Col. Pav. w/ Sand Chan.	Seagrass	Macroalgae	Unconsol. Sediment
Patch Reef (Individual)								
Patch Reef (Aggregate)	**0.18**							
Linear Reef	0.05	-0.06						
Colonized Pavement	-0.03	-0.08	**0.07**					
Col. Pavement w/ Sand Channels	0.13	0.12	**0.12**	**0.11**				
Seagrass	0.16	0.10	**0.29**	**0.25**	**0.43**			
Macroalgae	**0.26**	**0.34**	**0.74**	**0.49**	**0.78**	0.09		
Unconsolidated Sediments	**0.24**	**0.21**	**0.56**	**0.40**	**0.71**	**0.28**	0.02	
Mangrove	**0.89**	**0.86**	**0.89**	**0.85**	**0.90**	**0.47**	**0.94**	**0.82**

reef were barely separable (Table 3.1). This result was confirmed when only hardbottom types were analyzed for pairwise differences (R=0.16), with the fish assemblages associated with the biologically and topographically complex habitat types showing no significant pairwise differences. Significant differences in fish assemblages only occurred between the most complex and the least complex habitat types (i.e. linear reef versus scattered coral/rock with sand). Highest dissimilarity in fish assemblages of hardbottom types occurred between individual patch reefs and scattered coral/rock in sand (80.1% mean dissimilarity). The higher abundance (>50% higher) of redband parrotfish (*Sparisoma aurofrenatum*) and striped parrotfish (*Scarus iseri*), and foureye butterflyfish (*Chaetodon capistratus*) on patch reefs versus scattered coral/rock explained the highest proportion of the dissimilarity. Most species were in higher abundance on patch reefs, except for very small-bodied benthic fish such as orangespotted goby (*Nes longus*), goldspot goby (*Gnatholepis thompsoni*) and bridled goby (*Coryphopterus glaucofraenum*), which were more abundant on scattered coral/rock and sandy substrata.

When pairwise differences were examined at the MODIFIER level for classes of seagrass cover and sand statistically significant difference was detected (Global R=0.54, p<0.01). Fish assemblages associated with continuous seagrass cover (>90%) were most different than other levels of cover including the sand habitat type (Table 3.2). Fish assemblages of continuous seagrass, however, did show some overlap with fish assemblages associated with sparser seagrass cover, but was most dissimilar to patchy seagrass (50-70%) and sand.

Table 3.2. ANOSIM R values measuring fish assemblage similarities between samples grouped by benthic habitat MODIFIER (i.e., % seagrass cover). Sand was included to examine differences in fish assemblages between classes of patchy seagrass and sand. R values in bold are statistically significant (p<0.01). Medium green=some overlap, light green=barely separable.

Mapped habitat type modifier	Continuous >90%	Patchy 70-90%	Patchy 50-70%	Patchy 30-50%	Patchy 10-30%
Continuous >90%					
Patchy 70-90%	**0.28**				
Patchy 50-70%	**0.52**	0.17			
Patchy 30-50%	0.18	0.27	**0.58**		
Patchy 10-30%	**0.36**	**0.34**	**0.47**	0.01	
Sand (Uncon. sed.)	**0.46**	0.05	0.04	0.11	**0.13**

Fish assemblages of the sparser 30-50% seagrass cover were not significantly different than assemblages associated with 10-30% cover and sand habitat types (Table 3.2). High variability in the response was evident, with assemblages of 50-70% seagrass being significantly different to 30-50% seagrass and 10-30 % seagrass, although some overlap in assemblage composition was evident. These data indicate that thresholds in seagrass cover may occur at the >90% level of cover, since fish assemblage composition was most different between this level of cover and almost all other classes exhibiting lower seagrass cover. Therefore, the presence of seagrasses with >90% cover makes a significant difference to the fish community highlighting the fact that not all seagrass beds provide equal function and spatial heterogeneity in benthic structure is very important for many fish.


Baitfish in mangroves

3.3.1.2. Fish community metrics

Highest fish abundance was observed in two zones: (i) the mangroves and some coral reefs in the nearshore lagoonal zone, as well as offshore mangrove islands within 3 km of the coastline; and (ii) colonized hardbottom in deeper waters (>10 m) on the shelf approximately 7-10 km from the coastline (Figure 3.4a). Lowest fish abundance was associated with unvegetated sediments (i.e., sand and muddy sand) across the study area (Figure 3.5a). The majority of fish in mangroves were juveniles of multi-habitat species including fish species that also utilize coral reefs (Table 3.3). Large schools (>1,000) of small-bodied planktivorous fish (Atherinidae, Clupeidae, Engraulidae) were observed at many (n=9) mangrove sites. Greatest concentrations of high fish biomass were found over colonized hardbottom at the most offshore portion of the study area and also at several nearshore sites including colonized hardbottom immediately east of Margarita Reef. Other high biomass sites were near the interface between colonized hardbottom and softbottom habitat types (Figure 3.4b).

Overall, mangroves supported intermediate levels of fish biomass and unvegetated sediments supported lowest mean fish biomass (Figure 3.4b). Similarly, highest fish species density (number of species

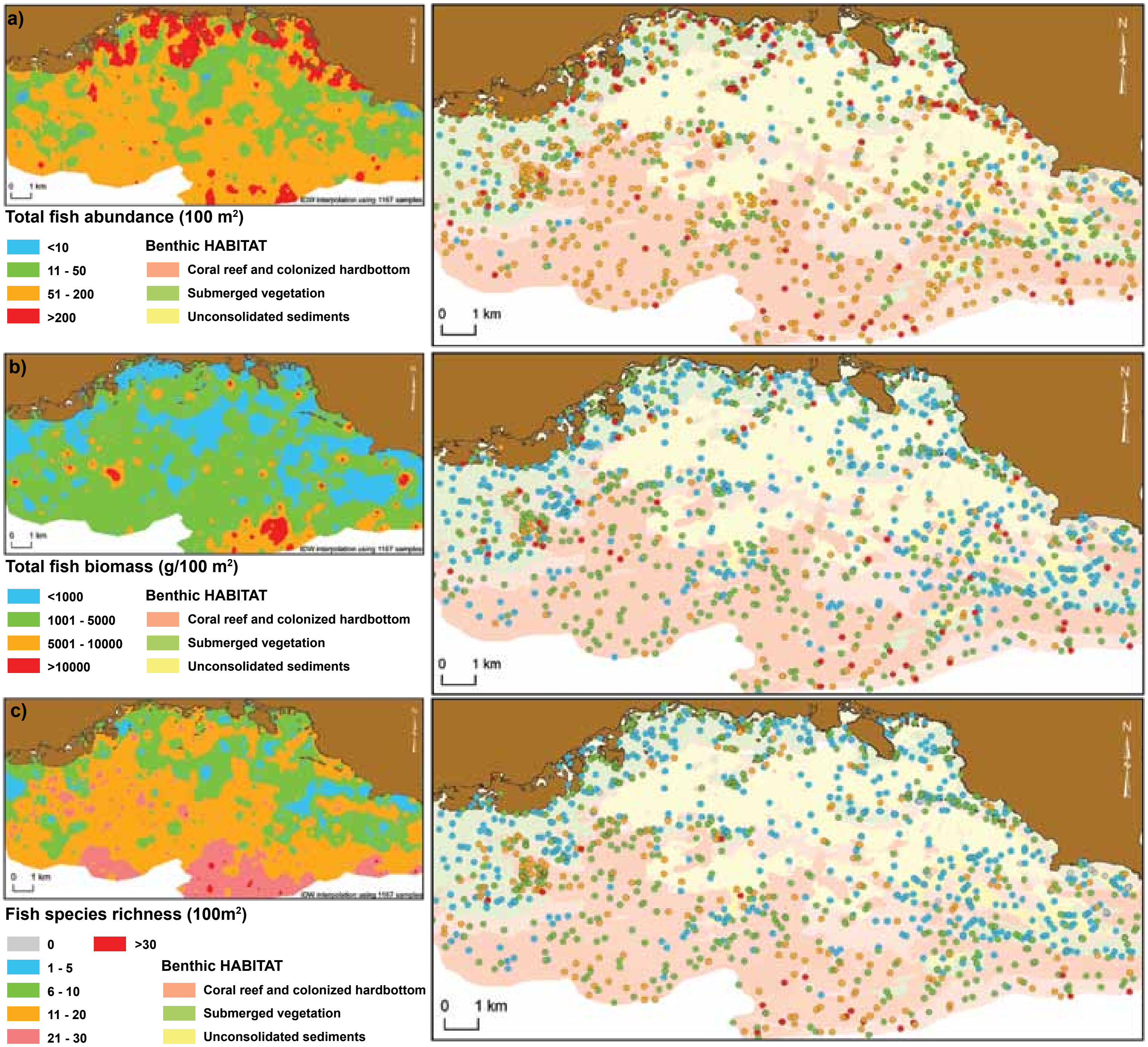


Figure 3.4. Maps of the interpolated (left map) and spatial (right map) distributions for: (a) total fish abundance, (b) total fish biomass and (c) total species richness.

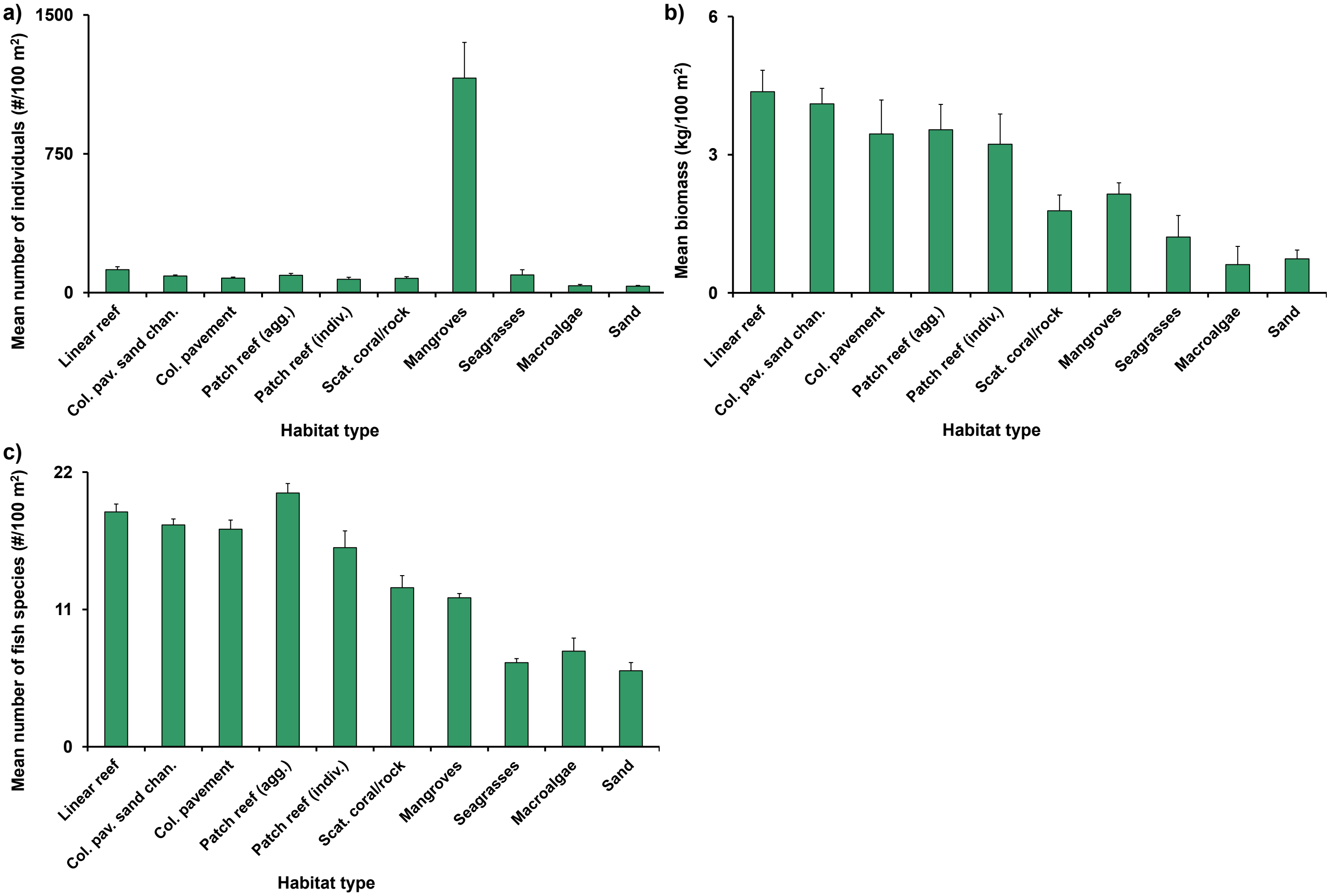


Figure 3.5. Comparison of mean (± SE) values by habitat type in the southwest Puerto Rico study area for: (a) total fish density, (b) total fish biomass and (c) total species richness.

per 100 m²) was observed over colonized hardbottom along the shelf edge and along the mangrove fringe closest to colonized hardbottom areas (Figure 3.4c). The majority of hardbottom sites supported fish species density of 11 or more fish per 100 m², with several sites in deeper water areas along the shelf edge and around El Palo and Margarita Reef supporting more than 30 fish species per 100 m². Lowest mean fish density was recorded over unvegetated sediments across the study area (Figure 3.5a).

Table 3.3. Percentage occurrence, range of major habitat types used and mean (± Standard Deviation) of fork length by habitat type for the 20 most abundant fish species/groups observed using mangroves in southwest Puerto Rico from 2001-2006. Habitat types used are indicated by M-mangroves, S-seagrasses, U-unvegetated sediments and C-coral reefs.

Species/Family	% occur	Habitats used	Mean (±SD) Fork Length, cm			
			Mangrove	Seagrass	Unvegetated	Coral reef
Jenkinsia spp.	51.9	MSUC	3.1 (0.6)	3.0 (0.0)	3.0 (0.0)	3.0 (0.0)
Atherinomorus spp.	14.2	M	3.0 (0.0)	- -	- -	- -
Clupeidae	14.2	MS	3.2 (1.0)	3.0 (0.0)	- -	- -
Haemulon flavolineatum	76.4	MSC	6.4 (3.3)	4.5 (2.6)	- -	13.4 (3.9)
Haemulon sciurus	77.2	MSUC	9.9 (5.0)	4.1 (2.7)	12.5 (0.0)	16.5 (7.5)
Engraulidae	2.4	M	3.0 (0.0)	- -	- -	- -
Lutjanus apodus	100	MSC	12.3 (6.4)	10.0 (3.5)	- -	17.4 (5.8)
Stegastes leucostictus	85.8	MSUC	5.9 (2.4)	4.5 (2.1)	4.8 (2.3)	5.8 (2.8)
Scarus iseri	35.4	MSUC	5.6 (3.2)	4.4 (2.1)	7.4 (6.0)	10.9 (6.0)
Abudefduf saxatilis	59.1	MC	5.8 (2.8)	- -	- -	8.8 (4.3)
Eucinostomus melanopterus	52.8	MUC	5.3 (2.8)	- -	9.2 (2.9)	7.8 (6.7)
Lutjanus griseus	45.7	MSC	16.7 (8.8)	12.5 (7.1)	- -	21.9 (8.7)
Gerres cinereus	52.7	MSUC	9.7 (5.2)	10.0 (3.5)	17.5 (0.0)	19.0 (5.2)
Sparisoma radians	35.4	MSUC	6.4 (3.9)	5.1 (2.8)	4.6 (2.8)	4.1 (2.1)
Haemulon spp.	20.5	MSUC	4.4 (4.1)	3.4 (1.2)	3.0 (0.0)	4.9 (4.7)
Chaetodon capistratus	58.3	MSUC	5.0 (2.4)	3.7 (1.6)	5.7 (2.5)	8.0 (2.9)
Coryphopterus glaucofraenum	14.7	MSUC	3.4 (1.3)	3.2 (0.9)	3.3 (1.1)	3.3 (1.8)
Sphyraena barracuda	65.4	MUC	18.6 (13.8)	- -	46.5 (61.5)	75.9 (40.9)
Acanthurus chirurgus	27.6	MSUC	7.3 (3.6)	6.4 (5.0)	11.1 (4.0)	14.9 (5.7)
Haemulon aurolineatum	5.5	MSUC	3.0 (0.0)	3.0 (0.0)	6.8 (5.4)	13.3 (6.3)

Herbivorous fish species richness was highly heterogeneous across hardbottom habitat types, with highest mean species richness along the shelf edge and around the El Palo and Margarita Reef region (Figure 3.6c). Although mean herbivore richness was relatively high in mangroves too (Figure 3.6c), this was elevated at offshore mangroves rather than the inshore fringing mangroves. Mean herbivorous fish species richness was more than 50% lower at seagrass/macroalgal sites than colonized hardbottom sites and lowest at unvegetated sediment sites (Figure 3.7c).

Herbivorous surgeonfish (*Acanthurus* species).

Mean herbivore fish biomass was markedly higher over colonized hardbottom than any other major habitat type (Figure 3.7b) and highest abundance was observed along the outer shelf sites and around the topographically complex collection of patch reefs at El Palo (Figure 3.6b). High herbivore fish

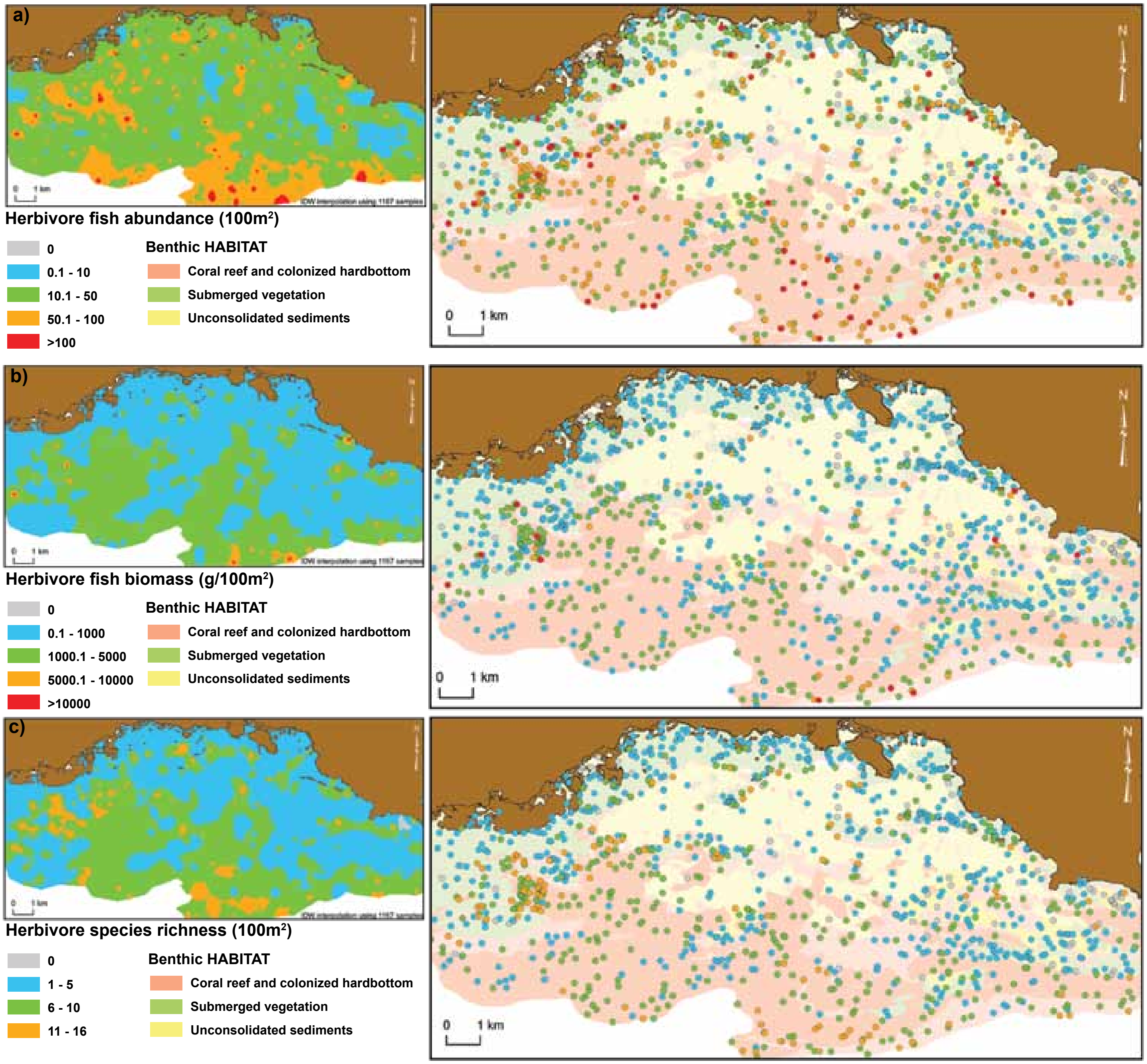


Figure 3.6. Maps of the interpolated (left map) and spatial (right map) distributions for herbivore: (a) abundance, (b) biomass and (c) species richness.

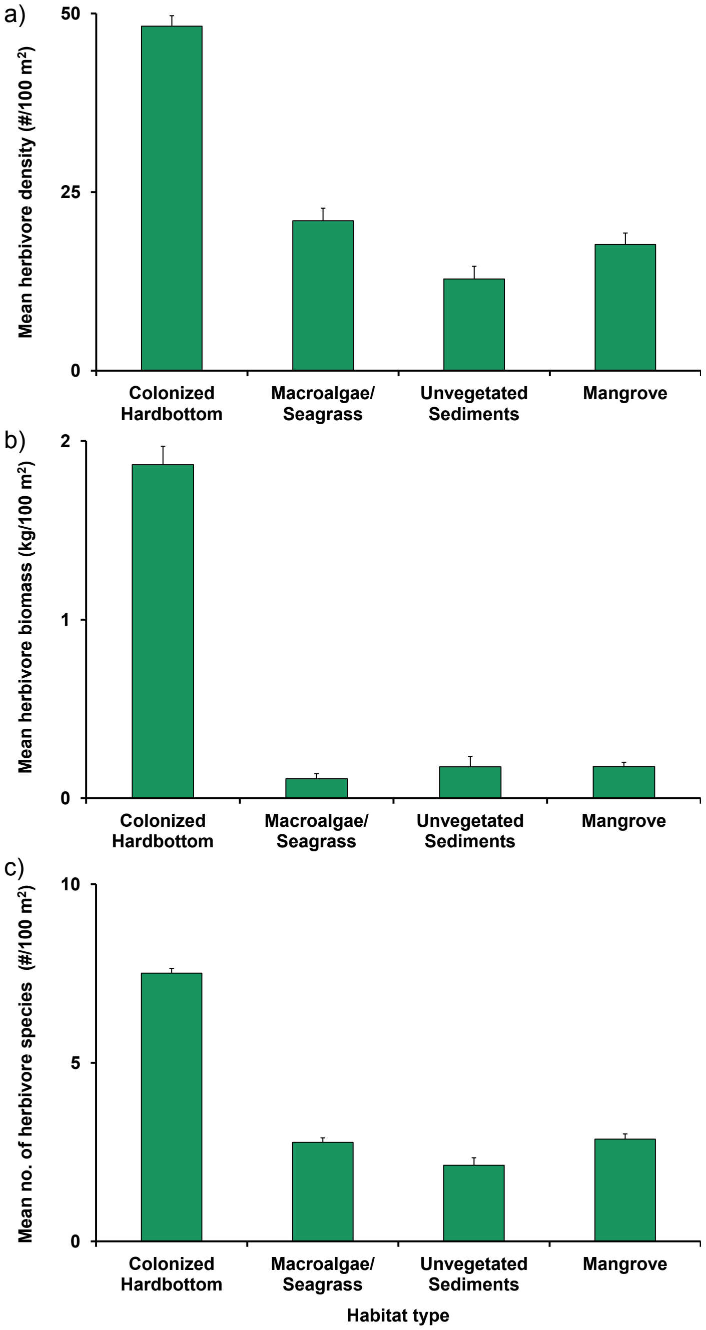


Figure 3.7. Comparison of mean (± SE) values by habitat type in the southwest Puerto Rico study area for herbivore: (a) density, (b) biomass and (c) species richness.

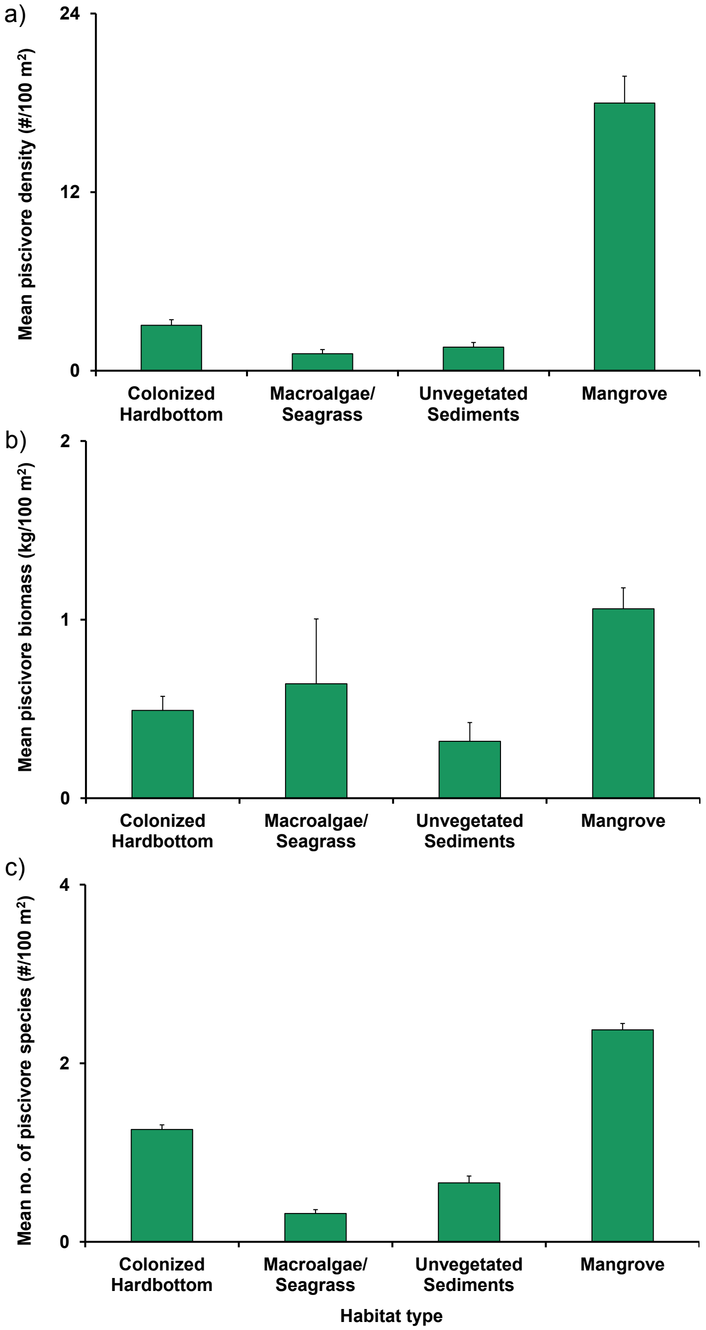


Figure 3.8. Comparison of mean (± SE) values by habitat type in the southwest Puerto Rico study area for piscivore: (a) density, (b) biomass and (c) species richness.

abundance was also observed across colonized hardbottom and seagrass habitat types in inshore lagoonal environments (Figure 3.6a and Figure 3.7a).

Mean herbivorous fish species richness (Figures 3.7c) was considerably higher than piscivorous fish species richness and no distinct areas of high piscivorous fish diversity were observed in the La Parguera study area. Mean piscivorous fish species richness was highest in mangroves and lowest in macroalgae/seagrass habitat (Figure 3.8c). Only 0.01% of all samples had more than five piscivorous fish species. Planktivorous fish species richness (Figure 3.10c) was also considerable lower than herbivorous fish richness, although areas of high planktivorous fish richness were more spatially distinct, with between six and 10 species observed in fish assemblages over colonized hardbottom sites along the shelf edge (Figure 3.11c). Overall, however, a higher mean diversity of planktivorous fish was found utilizing the mangrove edge (Figure 3.11c).

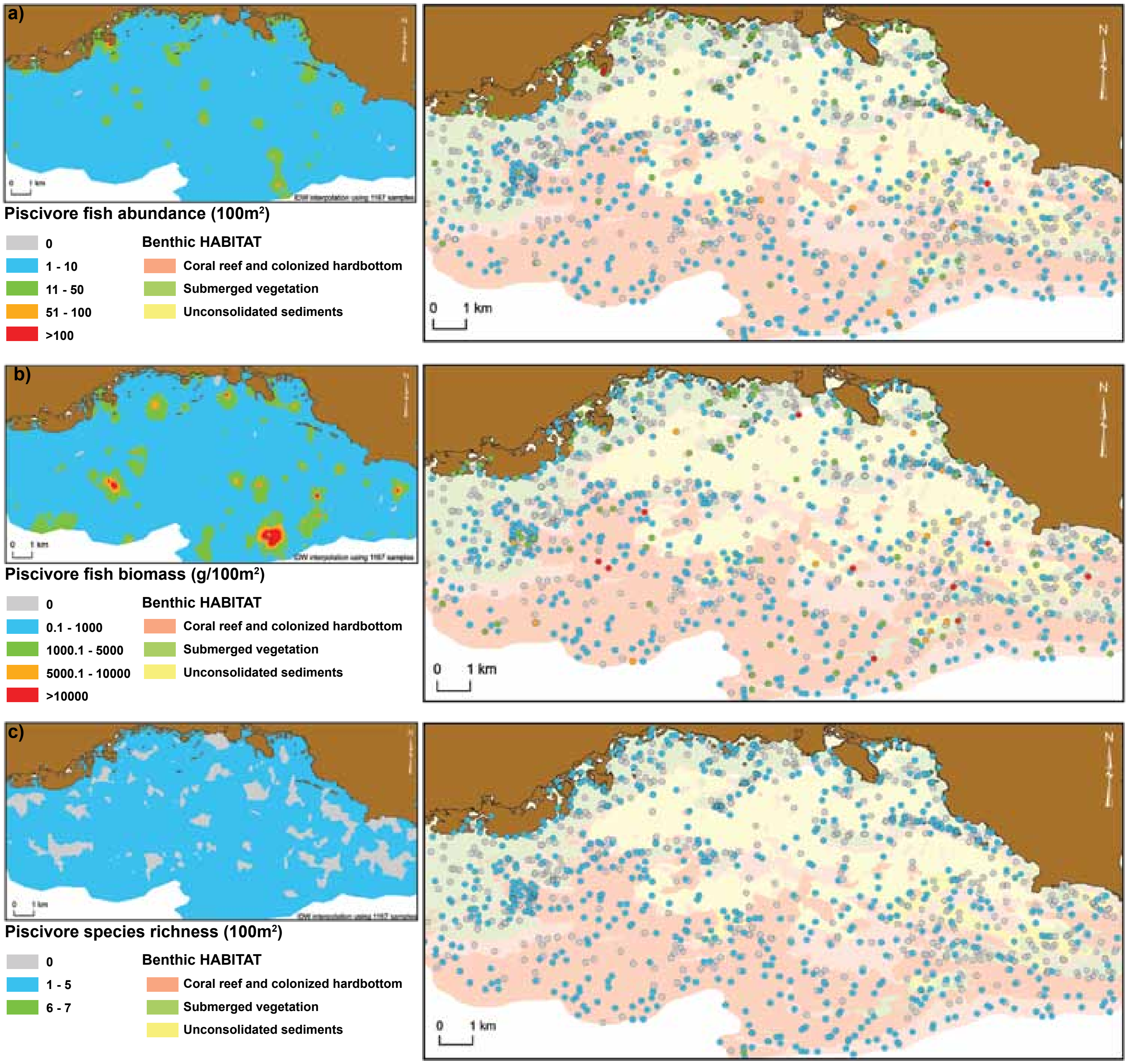


Figure 3.9. Maps of the interpolated (left map) and spatial (right map) distributions for piscivore: (a) abundance, (b) biomass and (c) species richness.

In contrast, piscivorous fish were most abundant in the mangroves (Figure 3.9a), but with some isolated sites in both midshelf and outershelf zones exhibiting high abundance (Figure 3.8a). The piscivorous fish observed within mangroves were primarily juveniles of fish that are piscivorous as subadults and adults. Several sites over colonized hardbottom that exhibited high biomass were sightings of cartilaginous fish such as sharks and rays (Figure 3.8b and Figure 3.9b). Highest mean biomass of piscivorous fish was recorded for mangroves (Figure 3.9b).

Piscivorous fish, cero mackrel (*Scomberomorus regalis*).

In the mangroves, planktivorous fish abundance was dominated by frequently observed schools of small-bodied silvery fish from families including Clupeidae, Atherinidae and Engraulidae (Figure 3.10a). These fish often swim in mixed species schools and could not be identified to species level using visual surveys. Mean biomass, however, was highest for colonized hardbottom habitat types particularly near the outershelf zone, where planktivorous fish included large numbers of creole wrasse (*Clepticus parrae*; Figure 3.11b).

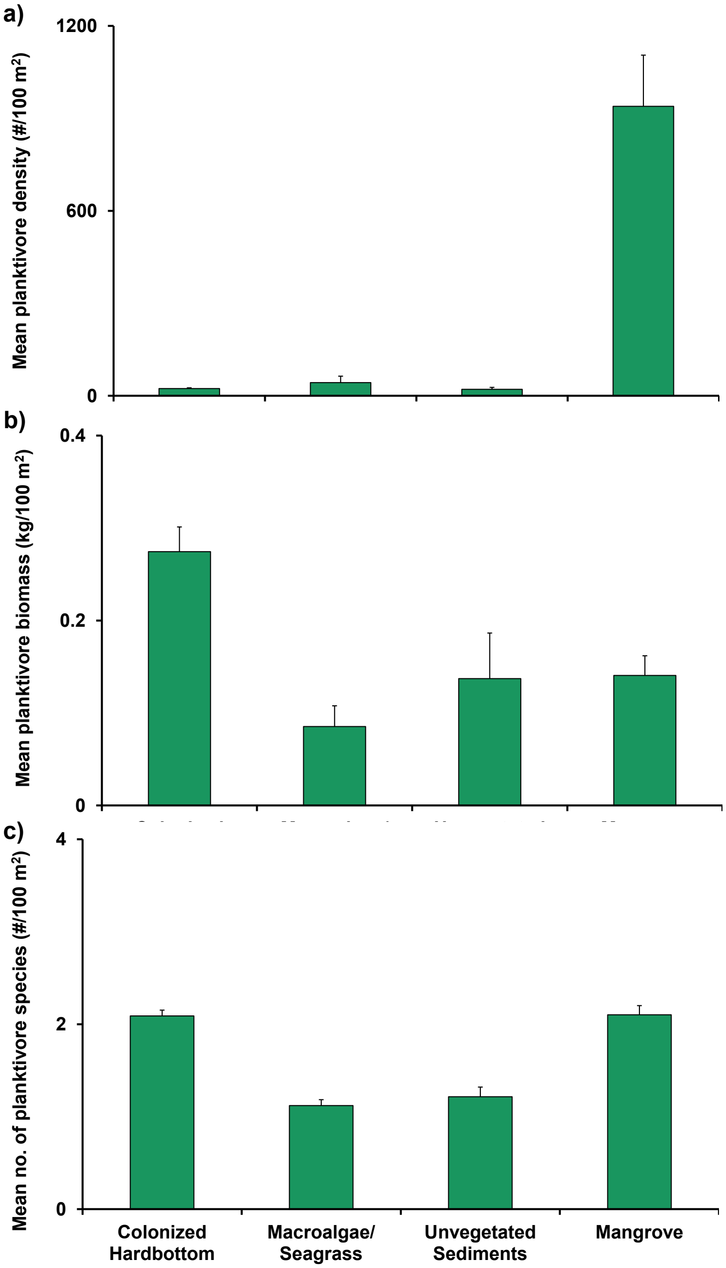


Figure 3.10. Comparison of mean (± SE) values by habitat type in the southwest Puerto Rico study area for planktivore: (a) density, (b) biomass and (c) species richness.

Planktivorous species, creole wrasse (*Clepticus parrae*).

Jenkinsia species in mangroves.

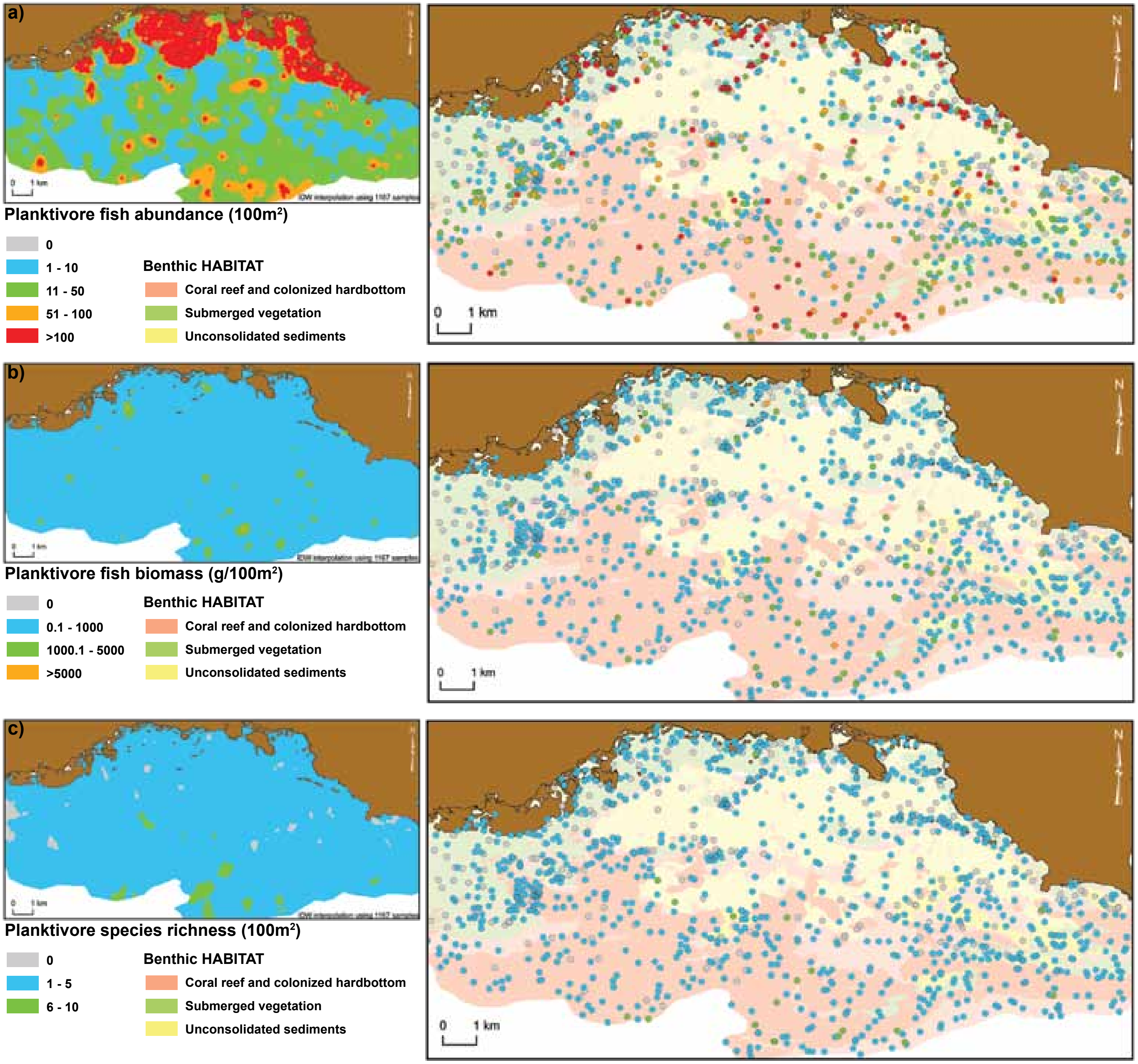


Figure 3.11. Maps of the interpolated (left map) and spatial (right map) distributions for planktivore: (a) abundance, (b) biomass and (c) species richness.

3.3.2. Taxonomic groups

Select Large-body Groupers (Serranidae)

Abundance and biomass of the large-bodied serranids or grouper (species from the Subfamily Epinephelinae; *Epinephelus* spp., *Cephalopholis* spp. and *Mycteroperca* spp.) was highest (7 individuals/100 m^2 and 1.4 kg/100 m^2, respectively) along the deeper water (>10 m) topographically complex shelf edge habitat types (linear reef and colonized pavement with sand channels; Figures 3.12a-b and 3.13a-b). Grouper were absent from mangroves and only found in very low abundance in macroalgal and seagrasses of nearshore and lagoonal zones (Figures 3.12a and 3.13a). For much of the colonized hardbottom across the study area, groupers were either absent or at very low densities (≤2/100 m^2). Biomass was highest over colonized pavement on the mid and outershelf (Figure 3.11b).

The smallest juveniles of the three most common grouper species were absent for coney (*Cephalopholis fulva*) and red hind (*Epinephelus guttatus*) and were very rarely seen for graysby (*Cephalopholis cruentata*; Figure 3.14a). *C. fulva* and *E. guttatus* were not observed until the 10-15 cm size class (subadults). In contrast, a larger proportion of the total sightings of *C. cruentata* were large juveniles and subadults (Figure 3.12a). The most frequently seen size classes of *C. fulva* were the 15-25 cm (measured in fork length [FL]) and 25-30 for *E. guttatus* (Figures 3.13 and 3.14b). No *Cephalopholis* species longer than 30 cm FL were recorded in the study area, although the maximum known size for *C. fulva* is 41 cm TL and for *C. cruentata* is 42.6 cm TL (Table 3.4). The maximum size for *E. guttatus* in this study was 40 cm FL compared with a maximum for the species of 76 cm TL (Table 3.4). Species of the genus *Mycteroperca* (*M. tigris; M. bonaci, M. venenosa, M. interstitialis*) were entirely absent from the surveyed sites between 2001-2007, even though tiger grouper (*Mycteroperca tigris*) and yellowfin grouper (*Mycteroperca venenosa*) may once have spawned along the shelf edge (Ojeda-Serrano, 2007).

*Table 3.4. Summary data on selected species from key fish families showing maximum size observed in the study region and maximum known size for the species found from 1,167 samples from 2001-2007. * Values from FishBase (http://www.fishbase.org). TL=total length; FL= fork length. Refer to Appendix C for detailed information.*

Species	Approx. size class at first maturity*	Max. known size*, TL	Max. size observed in PR, FL
Balistes vetula	20-25	60	40
Cephalopholis cruentata	15-20	42.6	30
Cephalopholis fulva	15-20	41	30
Epinephelus guttatus	20-25	76	40
Haemulon aurolineatum	15-20	25	25
Haemulon flavolineatum	15-20	30	30
Haemulon plumierii	15-20	53	30
Haemulon sciurus	15-20	46	35
Lutjanus apodus	20-25	67.2	45
Lutjanus griseus	25-30	89	65
Lutjanus mahogoni	15-20	48	30
Lutjanus synagris	20-25	60	65
Ocyurus chrysurus	20-25	86.3	40
Mulloidichthys martinicus	15-20	39.4	35
Pseudupeneus maculatus	15-20	30	30
Scarus iseri	10-15	35	35
Scarus taeniopterus	10-15	35	35
Sparisoma aurofrenatum	10-15	28	35
Sparisoma rubripinne	--	47.8	40
Sparisoma viride	15-20	64	50
Sphyraena barracuda	--	200	150

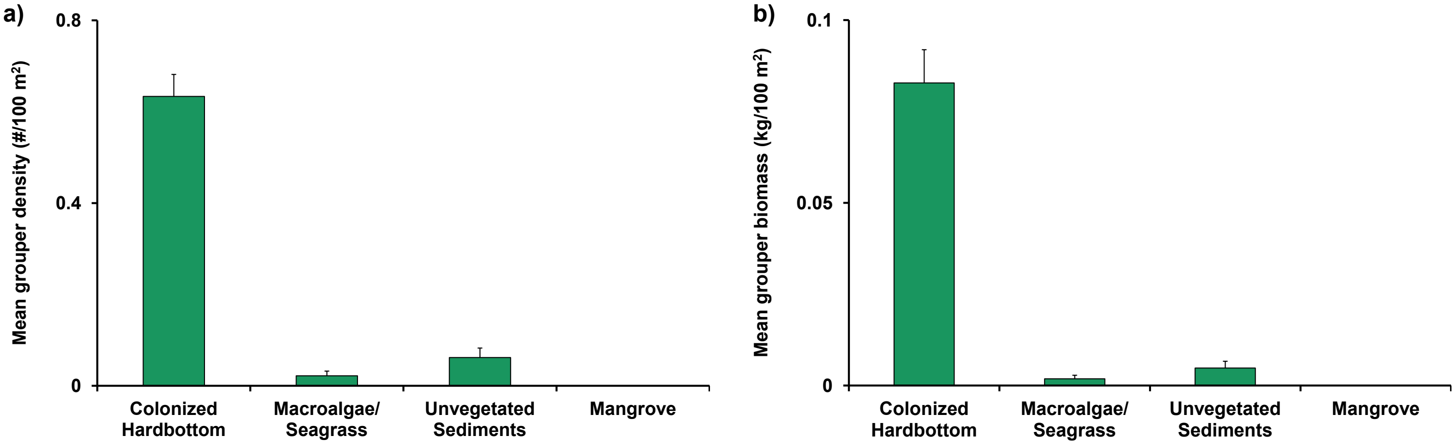


Figure 3.12. Comparison of mean (± SE) density and biomass by habitat type in the southwest Puerto Rico study area for select grouper (Serranidae).

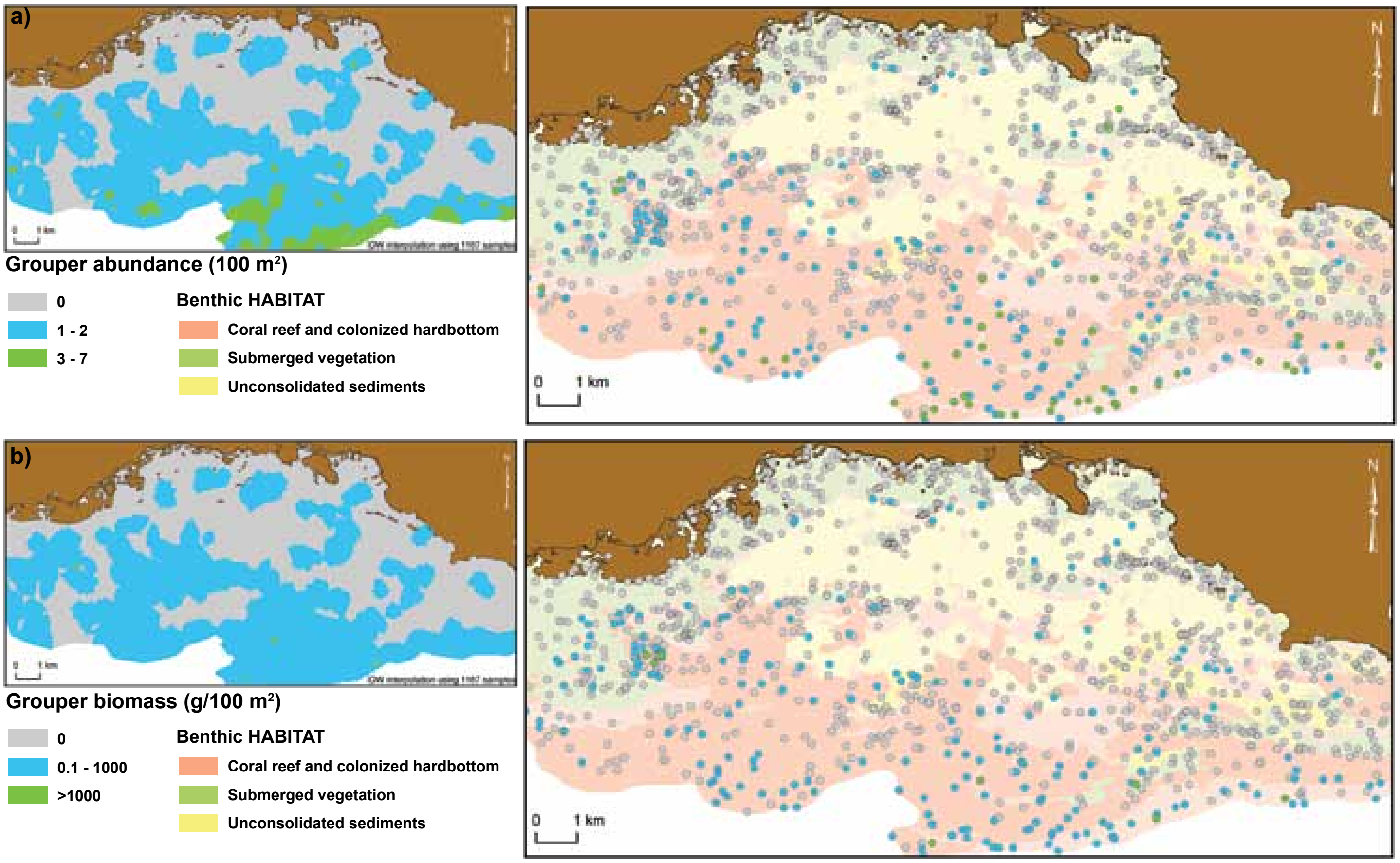


Figure 3.13. Maps of the interpolated (left map) and spatial (right map) distributions for select large-bodied grouper (Serranidae): (a) abundance and (b) biomass.

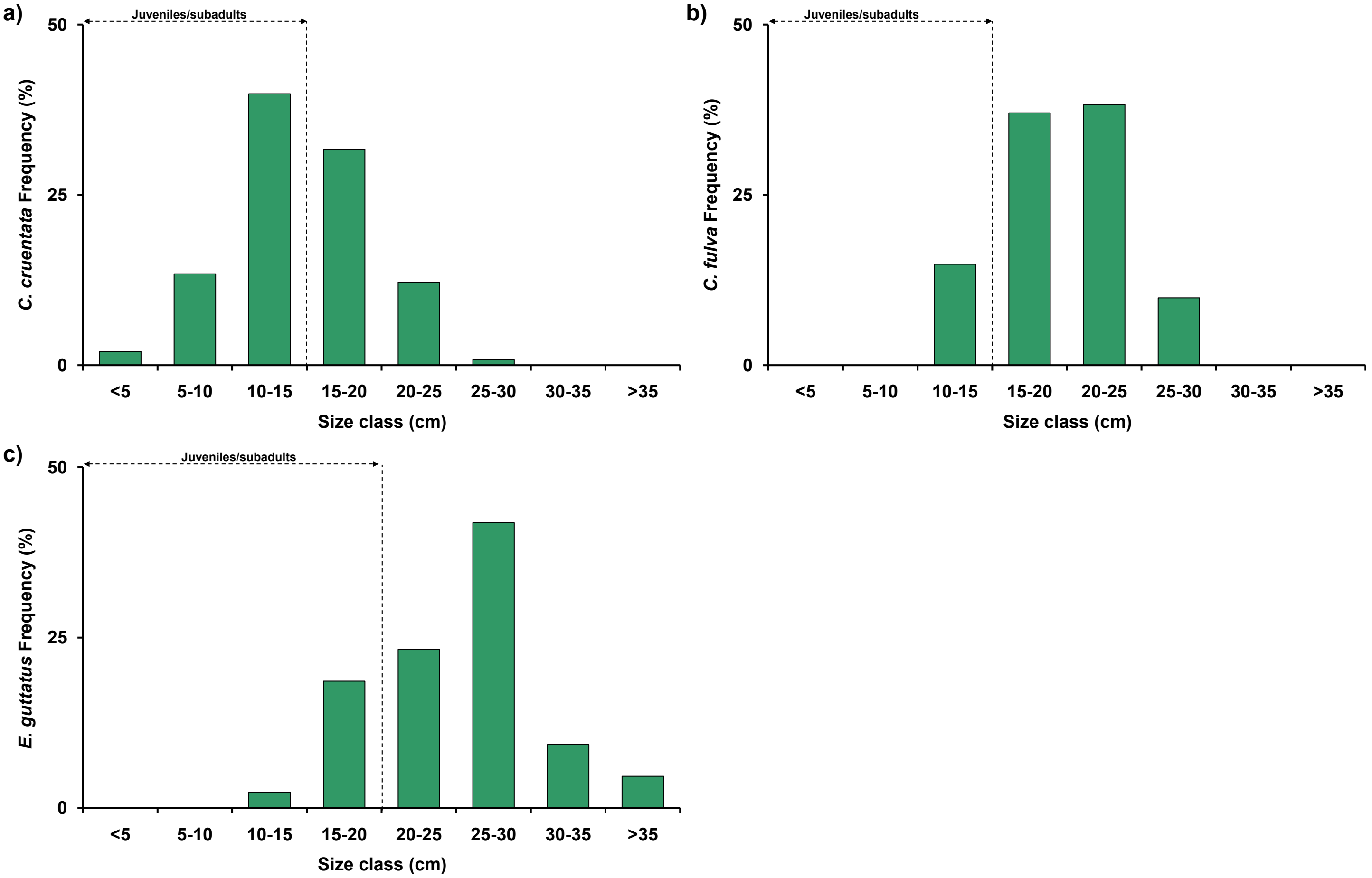


*Figure 3.14. Size frequency histogram for select grouper (Serranidae) in the southwest Puerto Rico study area. (a) graysby (*C. cruentata*), (b) coney (*C. fulva*) and (c) red hind (*E. guttatus*).*

Graysby (*Cephalopholis cruentata*)

C. cruentata abundance and biomass were highest over colonized hardbottom with higher occurrence on mid and outer shelf zones than the nearshore (Figures 3.15 and 3.16). Densities were relatively low across the study area, but with several high rugosity areas of contiguous coral reefs on the outer shelf supporting densities of >2 *C. cruentata* per 100 m^2 (Figure 3.15a). Abundance and biomass were comparatively low in seagrasses and unvegetated sandy areas (Figure 3.16).

Graysby (*Cephalopholis cruentata*) in the Flower Garden Banks.

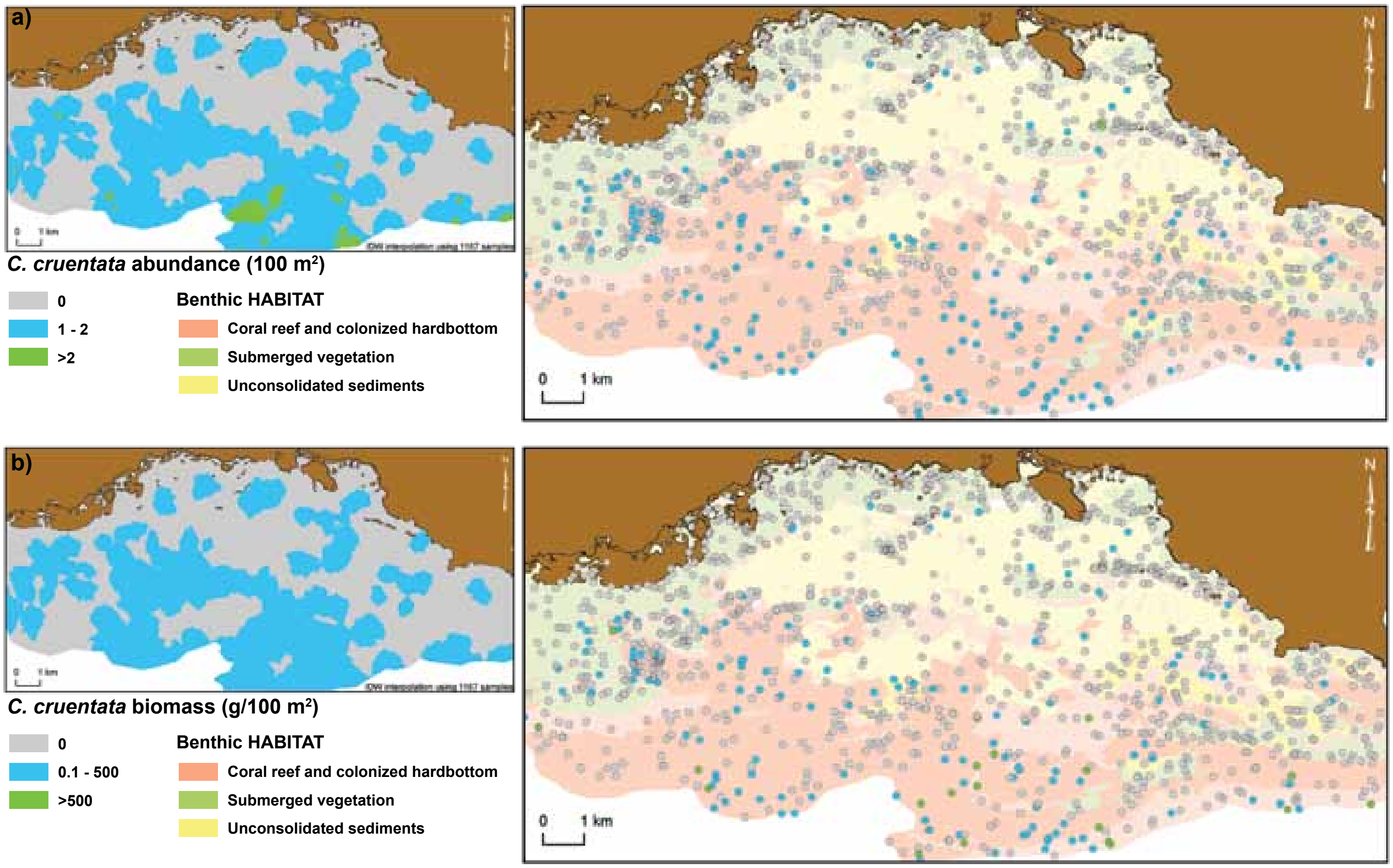


*Figure 3.15. Maps of the interpolated (left map) and spatial (right map) distributions for graysby (*C. cruentata*): (a) abundance and (b) biomass.*

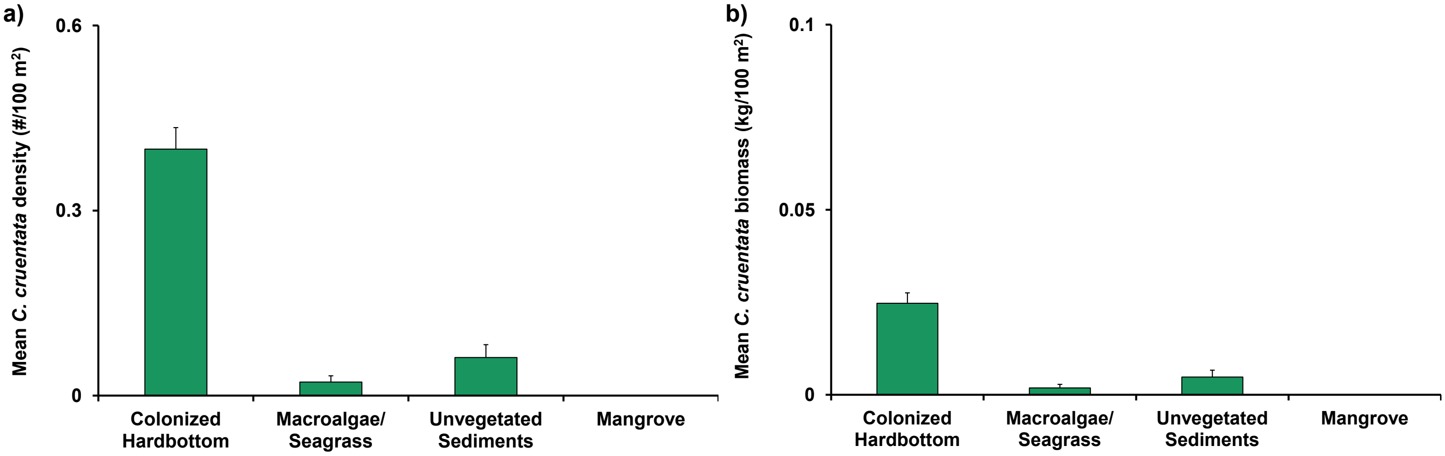


*Figure 3.16. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for graysby (*C. cruentata*).*

The majority of *C. cruentata* individuals observed were adults and subadults, with highest densities occurring in the topographically complex colonized pavement with sand channels, linear coral reefs and patch reefs habitat types (Figure 3.17). All life stages co-occurred in the same zones and habitat types across the shelf. No distinct inshore-offshore ontogenetic patterns were evident and juveniles exhibited no distinct preference for shallow inshore areas (Figure 3.17). *C. cruentata* were absent from mangroves and reef rubble and rarely seen over unvegetated sediments, macroalgae and seagrasses (Figure 3.17).

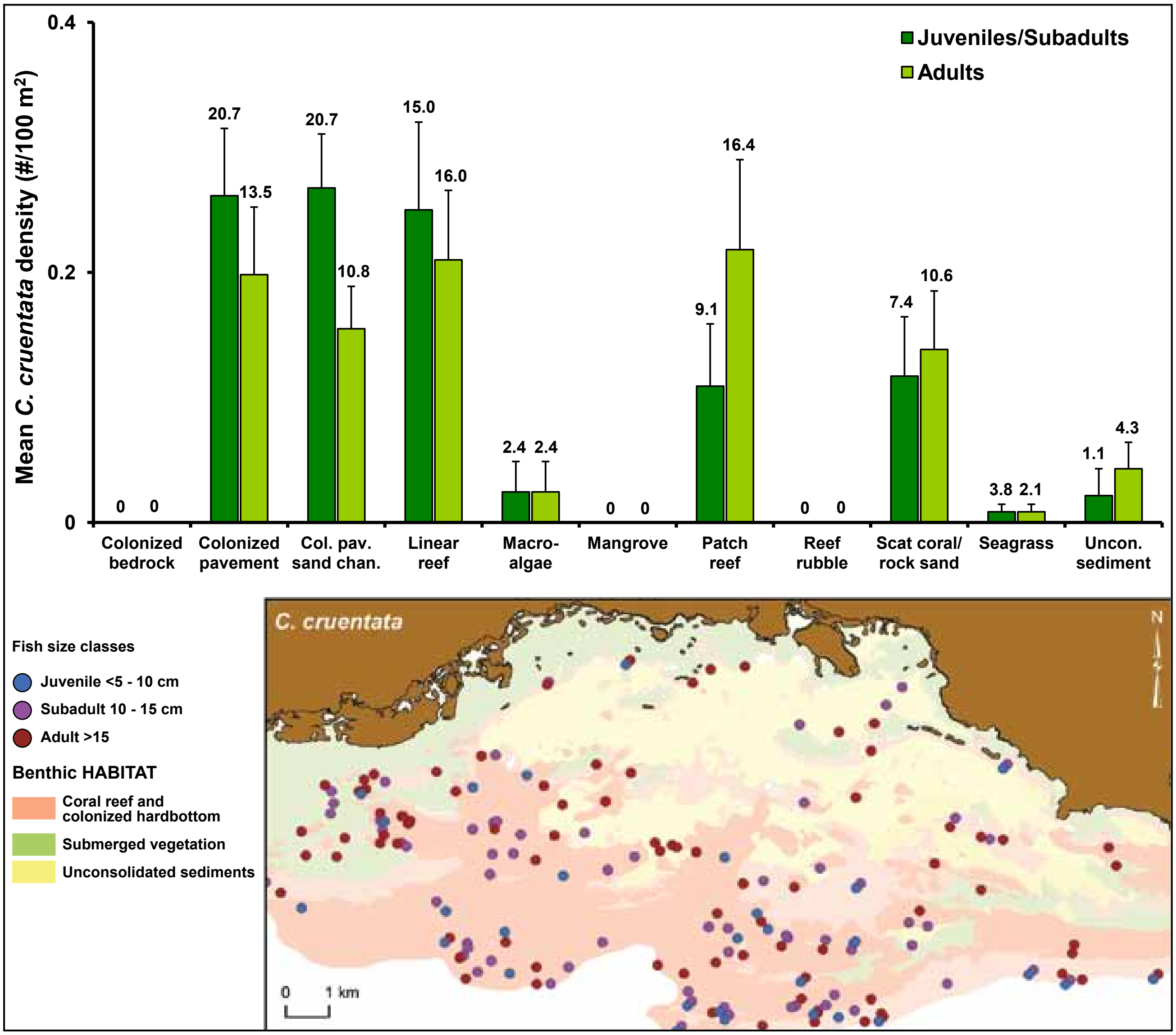


*Figure 3.17. Top: Mean density (±SE) for juvenile/subadult and adult by mapped habitat type for graysby (*C. cruentata*). Number above error bar indicates percent occurrence. Bottom: Spatial distribution of juvenile, subadult and adult for graysby (*C. cruentata*) in the southwest Puerto Rico study area.*

Coney (*Cephalopholis fulva*)

C. fulva exhibited a very restricted distribution within the study area with abundance and biomass highest on colonized hardbottom along the shelf edge (Figures 3.18 and 2.18). In contrast, in nearshore and mid-shelf zones only one *C. fulva* individual was observed (Figure 3.18). Seagrasses, mangroves and unvegetated soft sediments were not utilized during daylight hours (Figure 3.19).

Coney (*Cephalopholis fulva*)

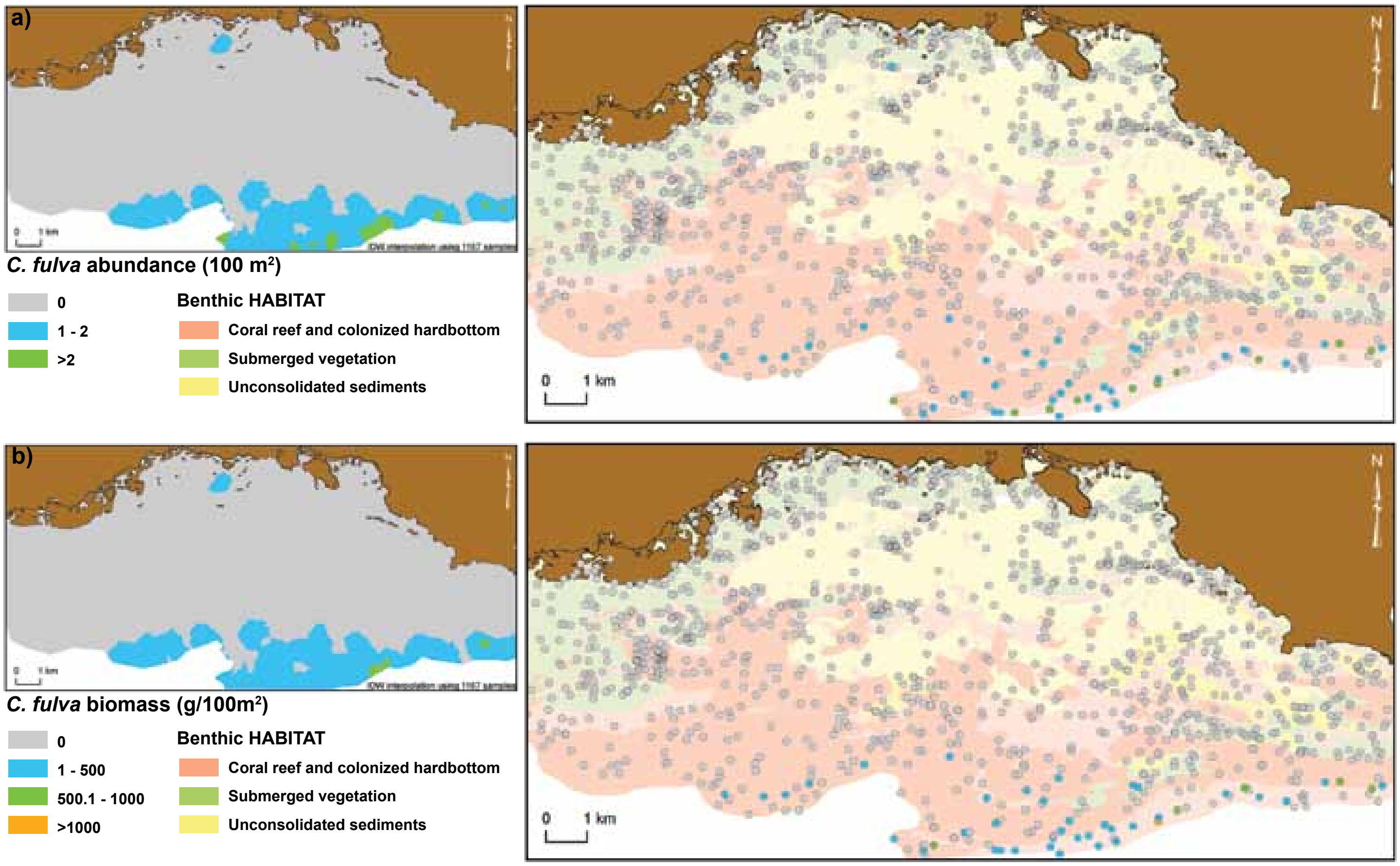


*Figure 3.18. Maps of the interpolated (left map) and spatial (right map) distributions for coney (*C. fulva*): (a) abundance and (b) biomass.*

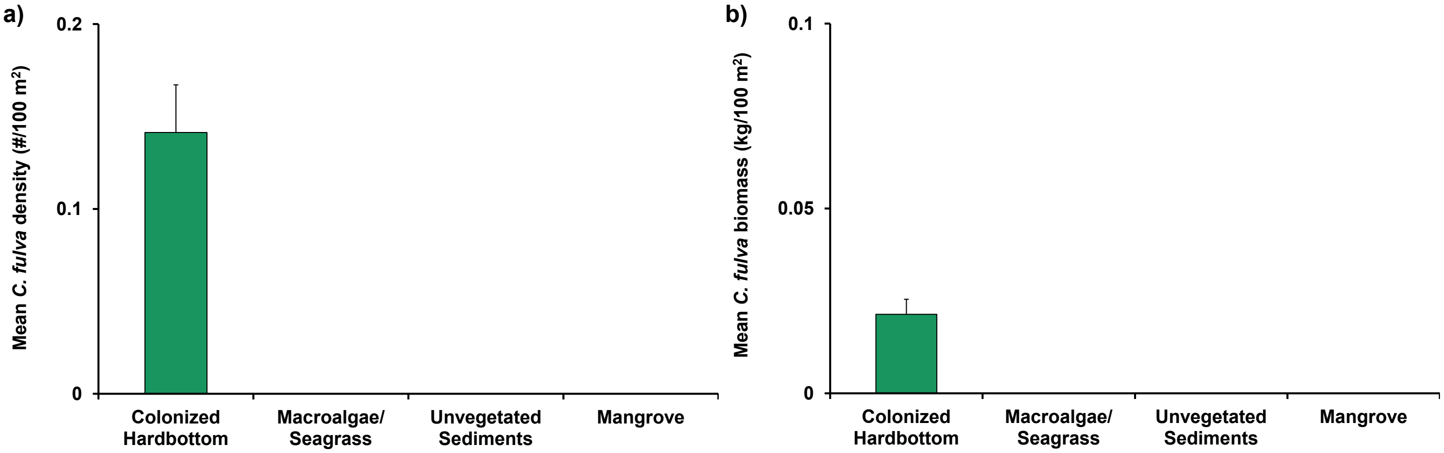


*Figure 3.19. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for coney (*C. fulva*).*

Juvenile *C. fulva* have not been observed in this monitoring program and very few subadults were sighted (only subadults are shown in the juvenile/subadult size class; Figures 3.14b and 3.20). Subadults and adults almost exclusively occupy outer shelf linear reef, colonized pavements with sand channels, colonized pavement and scattered corals that exist at the southern outer edge of the study area (Figure 3.20).

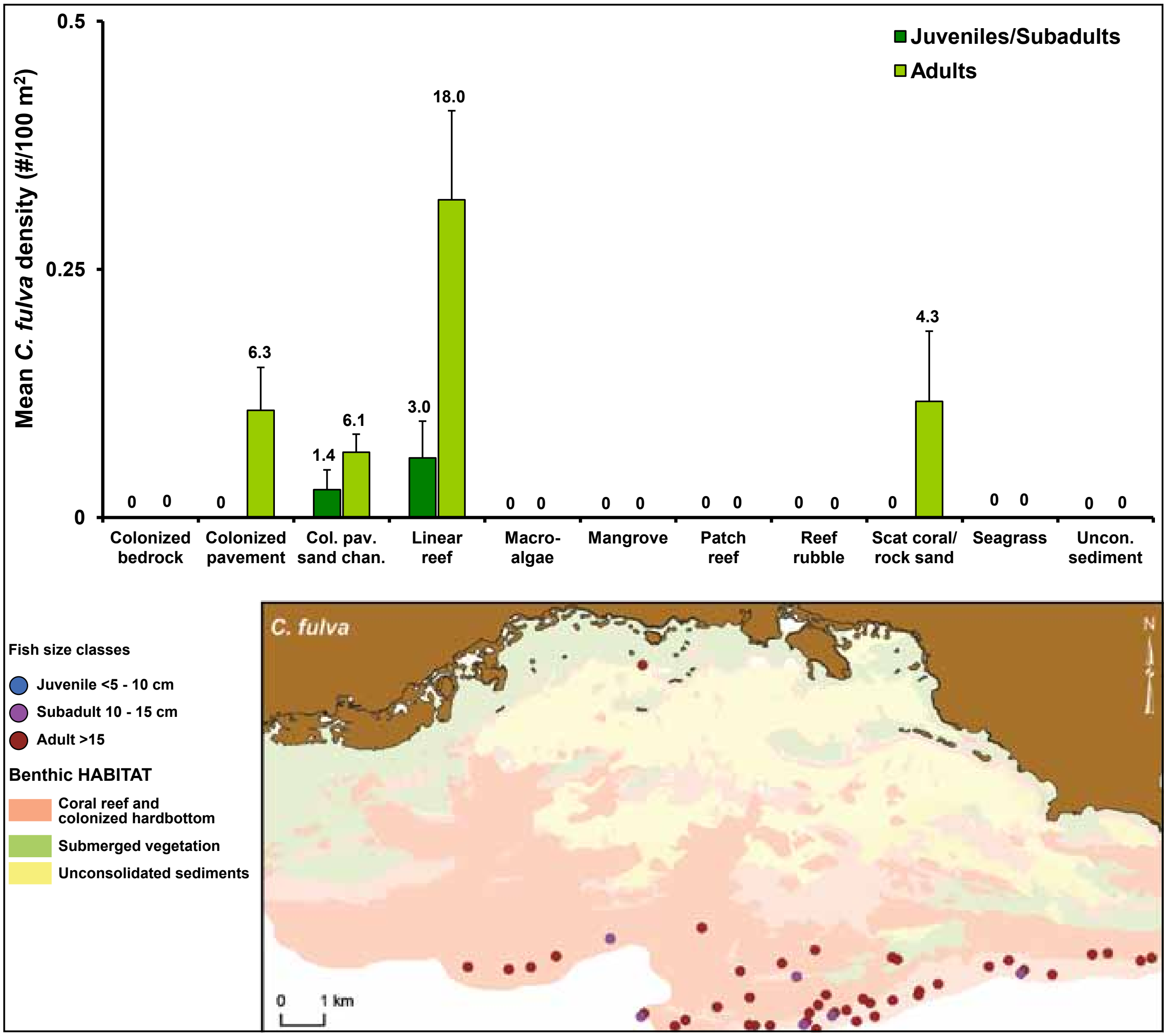


*Figure 3.20. Mean density (+ SE) for juvenile/subadult and adult by mapped habitat type for coney (*C. fulva*). Number above error bar indicates percent occurrence. Bottom: Spatial distribution of juvenile, subadult and adult for coney (*C. fulva*) in the southwest Puerto Rico study area.*

Red hind (*Epinephelus guttatus*)

E. guttatus exhibited a very patchy spatial distribution, with sightings confined to colonized hardbottom habitat types predominantly on the mid and outer shelf zones (Figure 3.21a; 3.22). Several individuals were also observed on coral reefs fronting the lagoonal zone. Highest biomass was estimated at sites along the topographically complex linear reefs and colonized pavement with sand channels habitat types at the shelf edge (Figure 3.21b).

Red hind (*Epinephelus guttatus*)

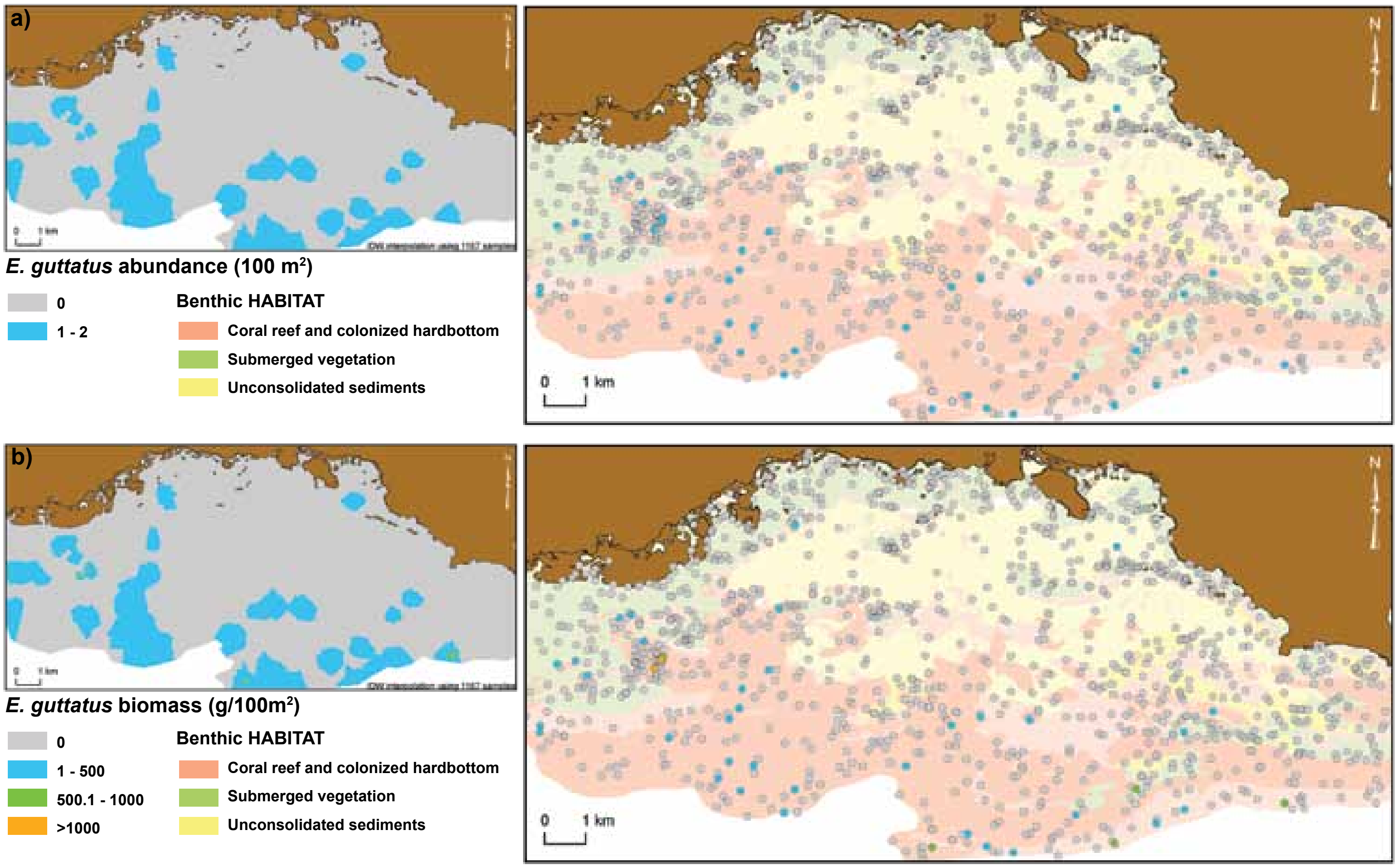


*Figure 3.21. Maps of the interpolated (left map) and spatial (right map) distributions for red hind (*E. guttatus*): (a) abundance and (b) biomass.*

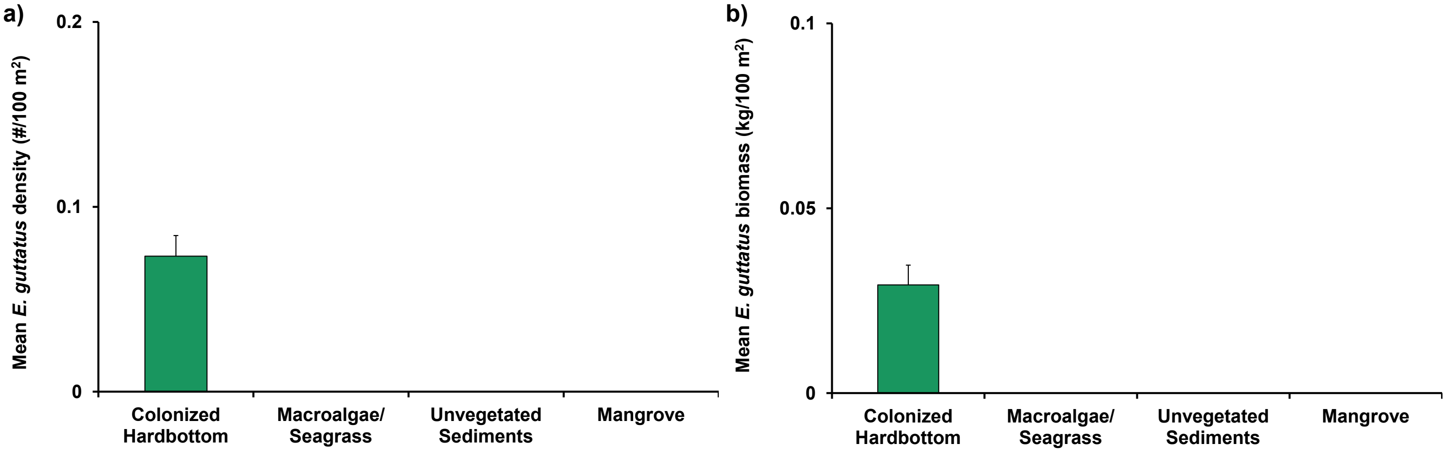


*Figure 3.22. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for red hind (*E. guttatus*).*

Juvenile *E. guttatus* have not been observed in this monitoring program (Figure 3.14c). Subadults and adults occurred predominantly in the mid and outer shelf zones, with highest densities of adults in the topographically complex patch reefs, colonized pavement with sand channels and linear reef habitat types (Figure 3.23). Subadults and adults showed no obvious geographical segregation. Five out of 11 habitat types were utilized and *E. guttatus* was absent from all softbottom habitat types, reef rubble, colonized bedrock and mangroves (Figure 3.23).

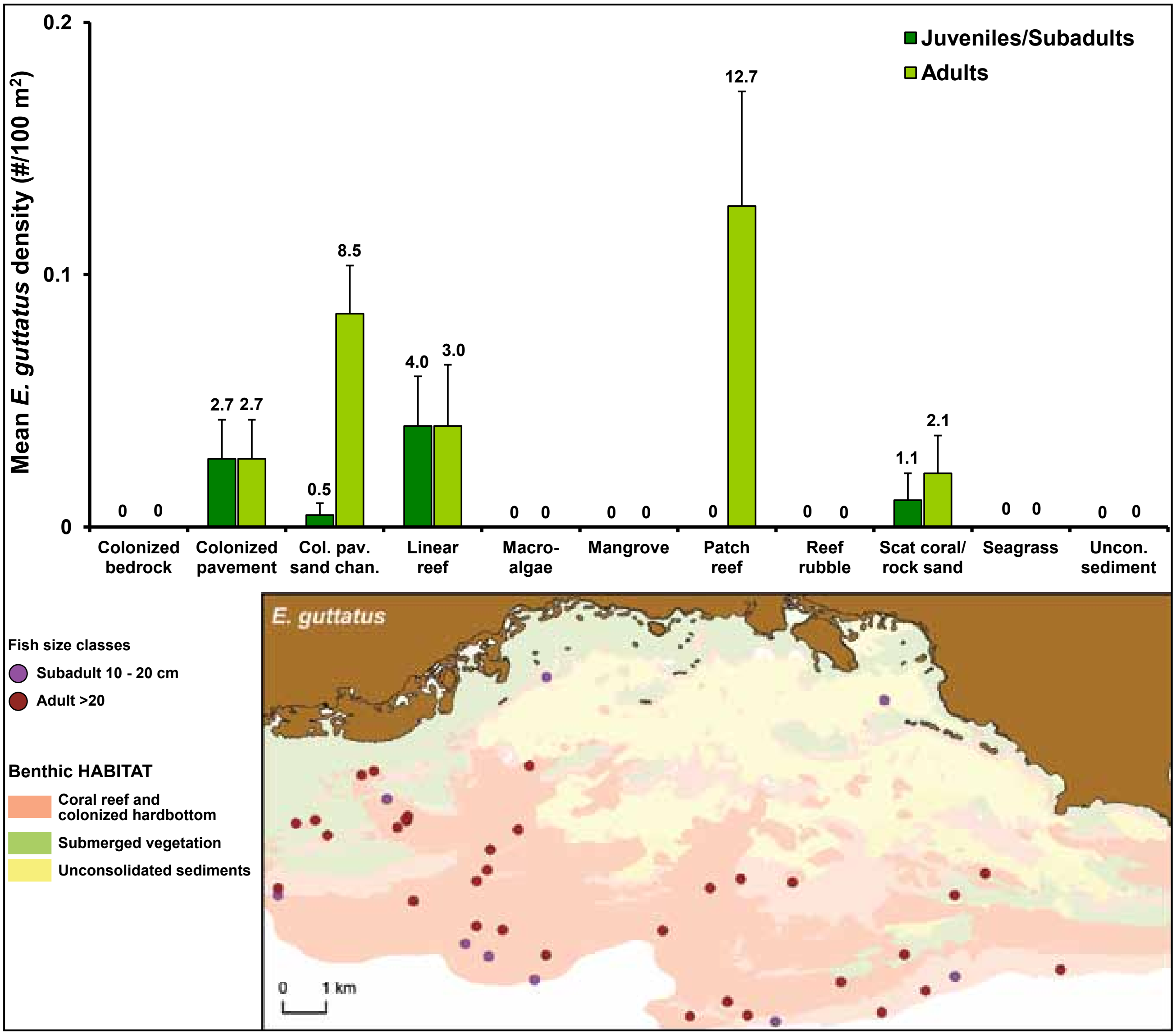


*Figure 3.23. Mean density (± SE) for juvenile/subadult and adult by mapped habitat type for red hind (*E. guttatus*). Number above error bar indicates percent occurrence. Bottom: Spatial distribution of subadult and adult for red hind (*E. guttatus*) in the southwest Puerto Rico study area.*

Snappers (Lutjanidae)

Lutjanids were widely distributed over the study region, but with abundance highest in nearshore areas, particularly fringing mangroves and mangrove cays (Figure 3.24a). Biomass was also highest in nearshore mangroves, although several hardbottom sites on the mid and outer shelf also exhibited high biomass represented by a few large-bodied individual snapper (Figure 3.24b). Lowest abundance and biomass were observed over soft bottomed habitat types (Figure 3.25).

L. apodus

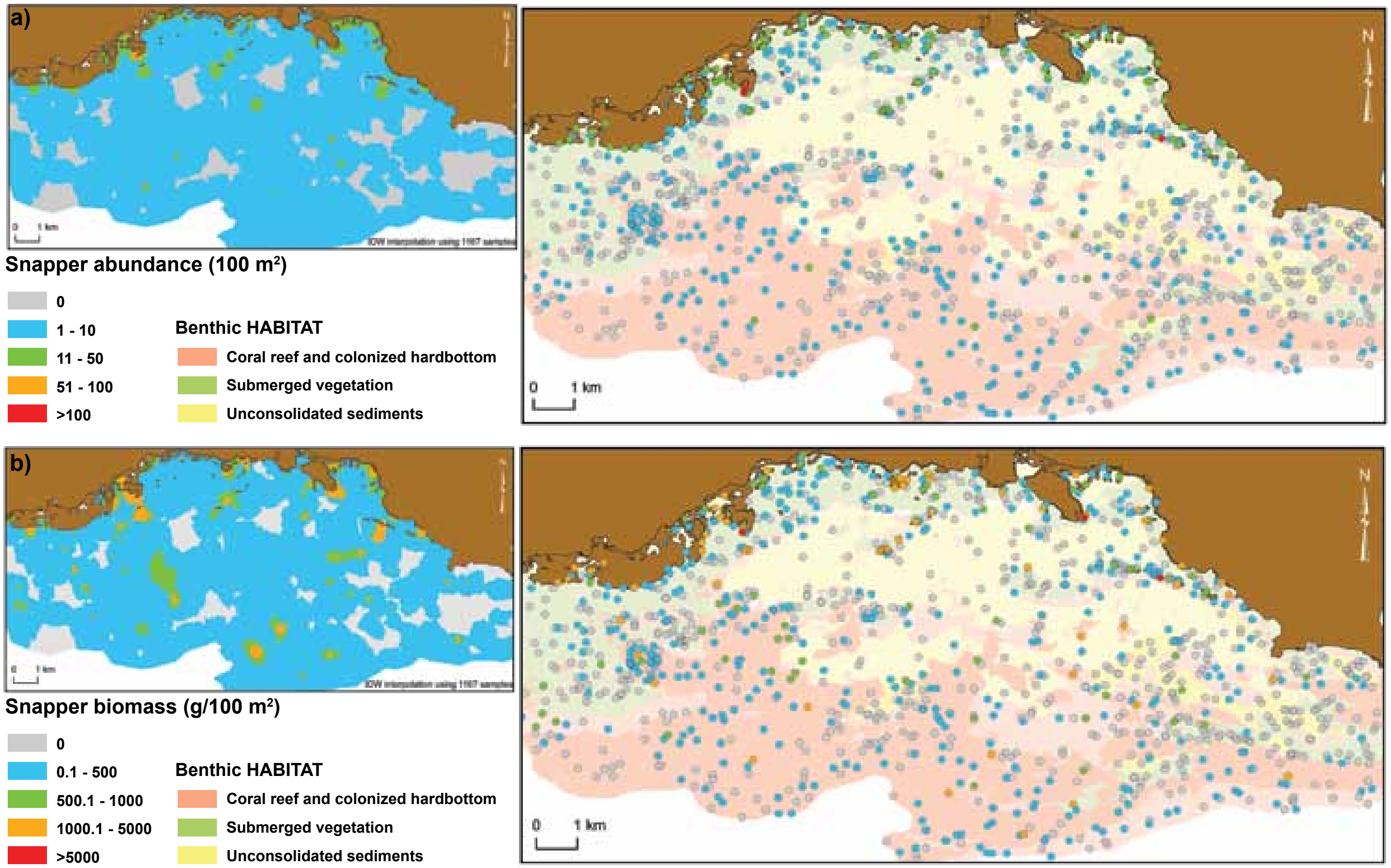


Figure 3.24. Maps of the interpolated (left map) and spatial (right map) distributions for snapper (Lutjanidae): (a) abundance and (b) biomass.

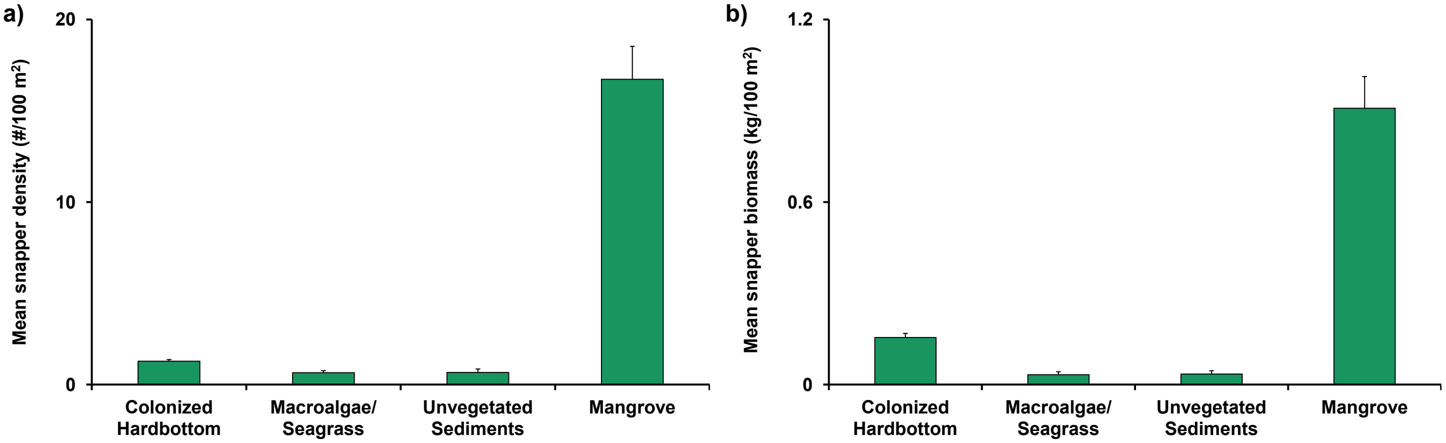


Figure 3.25. Comparison of mean (± SE) density and biomass by habitat type in the southwest Puerto Rico study area for snapper species (Lutjanidae).

The smallest size classes of lutjanids were well represented, particularly for lane snapper (*Lutjanus synagris*) with approximately 35% of all individuals <5 cm (Figure 3.26). Relatively few of the smallest juveniles were observed for mahogany snapper (*Lutjanus mahogoni*), but this species exhibited the largest proportion of adult individuals compared with other lutjanids (Figure 3.26c). The largest adult schoolmaster (*Lutjanus apodus*) and *L. synagris* were no larger than 30 cm FL (Figures 3.26a and 3.26d); the largest gray snapper (*Lutjanus griseus*) was 50 cm FL and the largest yellowtail snapper (*Ocyurus chrysurus*) was 40 cm FL (Figures 3.26b and 3.26e). The largest individuals observed in the study area were markedly smaller than the maximum known size for the species (Table 3.4). The largest *L. apodus* and *L. synagris* was less than 50% of the maximum size for the species. *L. mahogoni* was 62% of the maximum size for the species (Table 3.4). Some snapper species were very rarely seen. Only two blackfin snapper (*Lutjanus buccanella*), one cubera snapper (*Lutjanus cyanopterus*) and 13 dog snapper (*Lutjanus jocu*) were observed between 2001 and 2007.

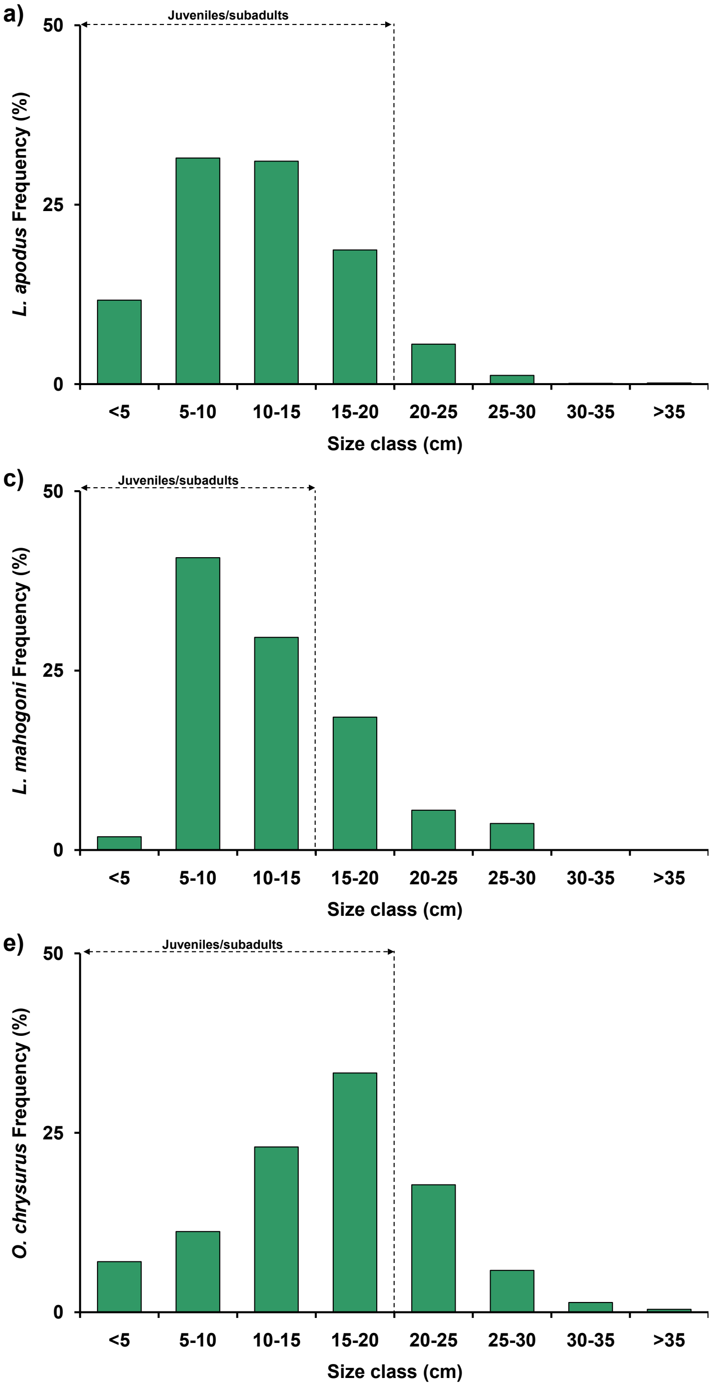


*Figure 3.26. Size frequency histogram for select snapper (Lutjanidae) in the southwest Puerto Rico study area. (a) schoolmaster (*L. apodus*), (b) gray snapper (*L. griseus*), (c) mahogany snapper (*L. mahogoni*), (d) lane snapper (*L. synagris*) and (e) yellowtail snapper (*O. chrysurus*).*

Schoolmaster (*Lutjanus apodus*)

L. apodus were most frequently observed relatively close to shore in the lagoonal zone, with mean abundance and biomass markedly higher in mangroves than any other habitat type (Figures 3.27 and 3.28). High biomass was found at several sites on mangrove cays and at one site over colonized hardbottom in the outer shelf zone (Figure 3.27).

L. apodus

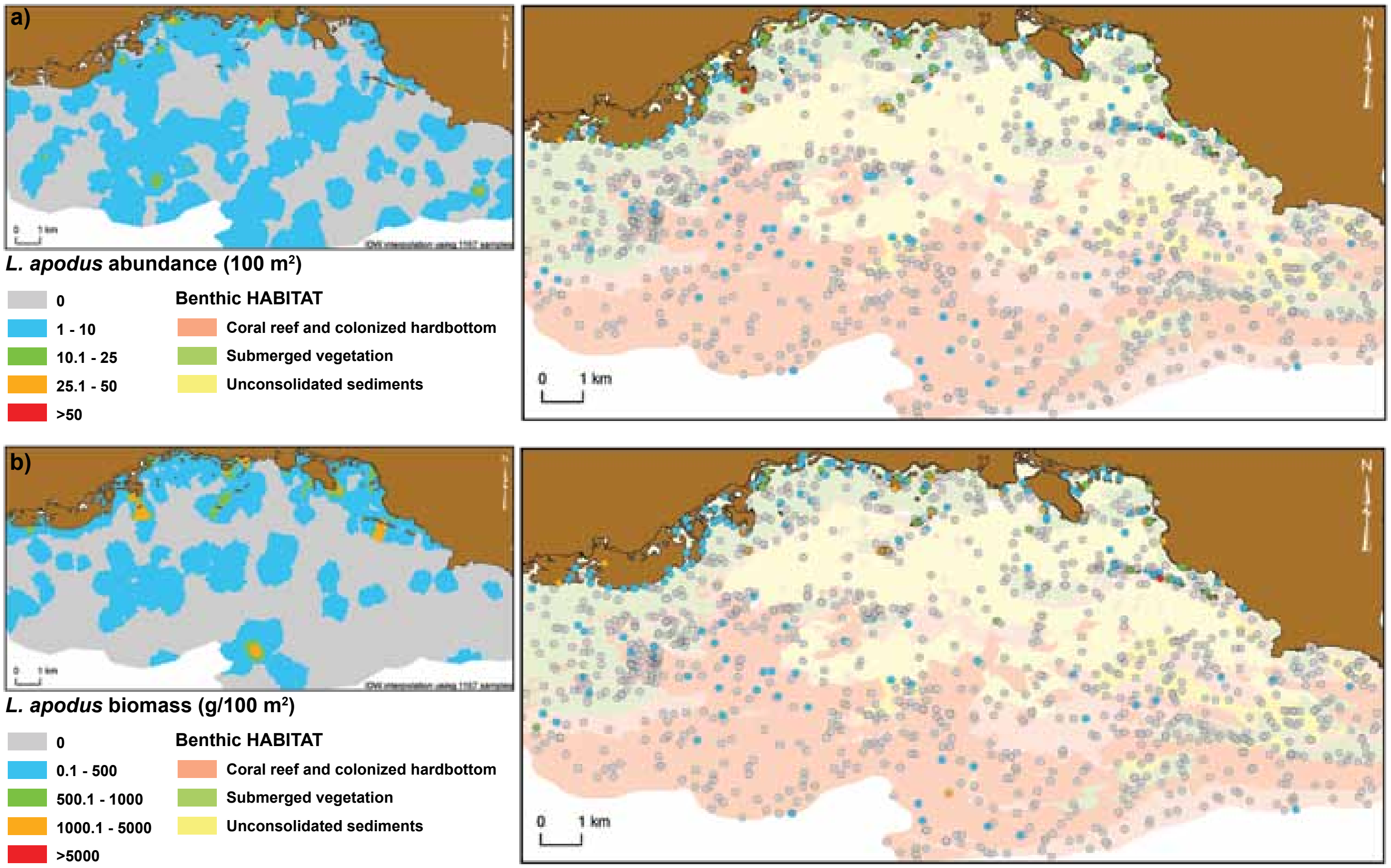


*Figure 3.27. Maps of the interpolated (left map) and spatial (right map) distributions for schoolmaster (*L. apodus*): (a) abundance and (b) biomass.*

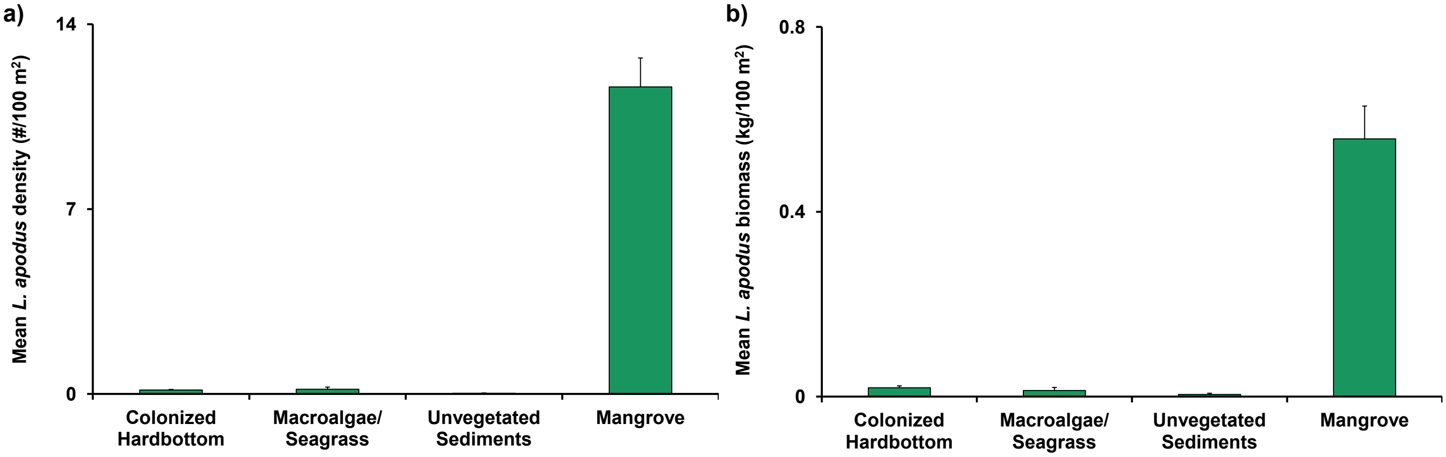


*Figure 3.28. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for schoolmaster (*L. apodus*).*

L. apodus juveniles and adults were observed in eight of 11 habitat types, with highest densities for all life stages recorded for mangroves (Figure 3.29). The majority of juvenile *L. apodus* were sighted within mangroves, but with some juveniles also seen at sites from the mid and outer shelf zones. Although *L. apodus* was observed across all shelf zones, occurrence was lowest in the outer shelf zone (Figure 3.29).

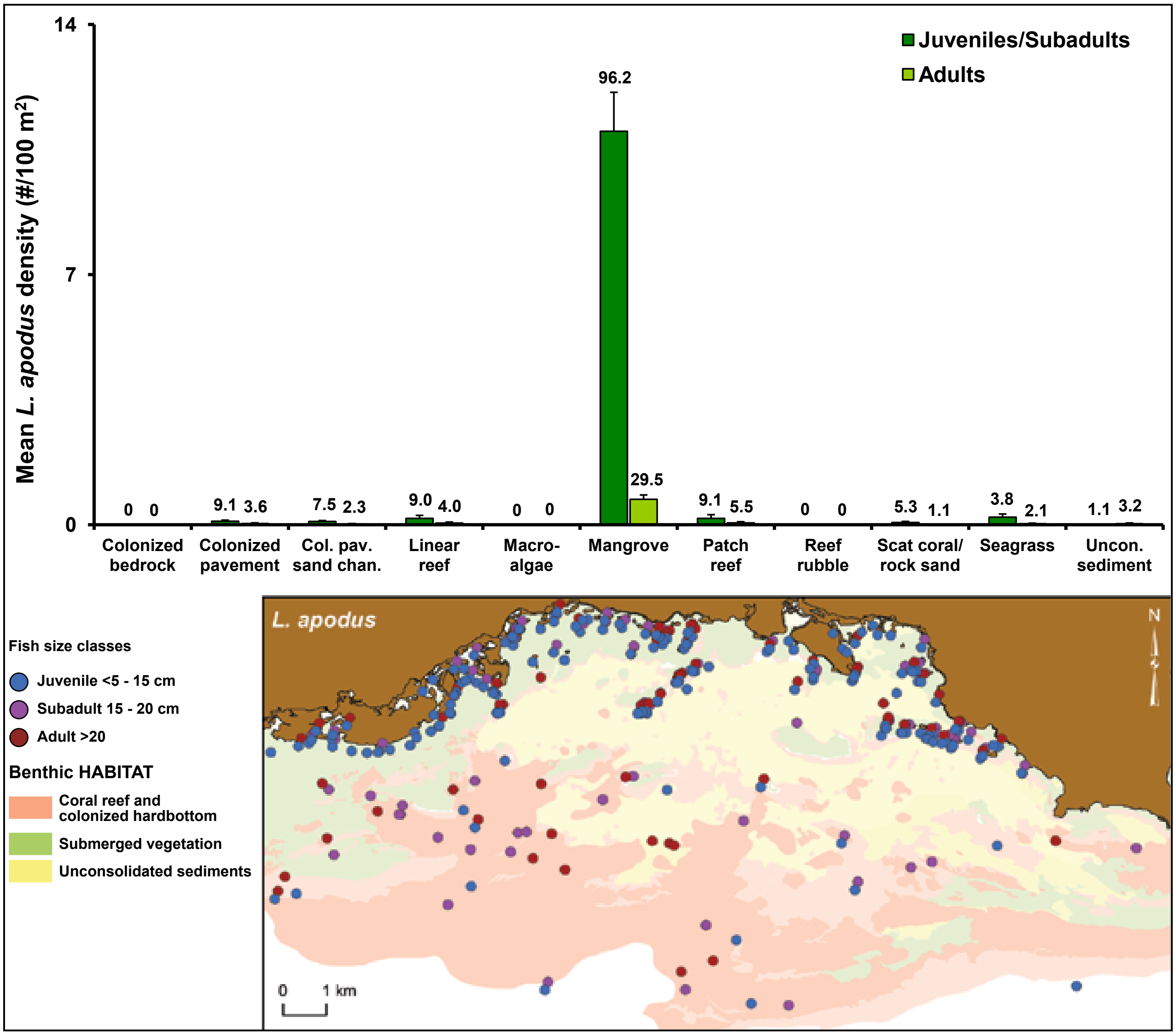


*Figure 3.29. Mean density (±SE) for juvenile/subadult and adult by mapped habitat type for schoolmaster (*L. apodus*). Number above error bar indicates percent occurrence. Bottom: Spatial distribution of juvenile, subadult and adult for schoolmaster (*L. apodus*) in the southwest Puerto Rico study area.*

Gray snapper (*Lutjanus griseus*)

L. griseus exhibited a relatively restricted distribution, with more than 95% of sightings occurring within mangroves, particularly along the fringe of the largest area of mangrove forest in the western portion of the study area (Figure 3.30). Highest biomass was also associated with mangroves (Figures 3.30b and 3.31b). *L. griseus* were very rarely observed on the mid and outer shelf. No individuals were observed over unvegetated sediments (Figure 3.31).

Gray snapper (*Lutjanus griseus*)

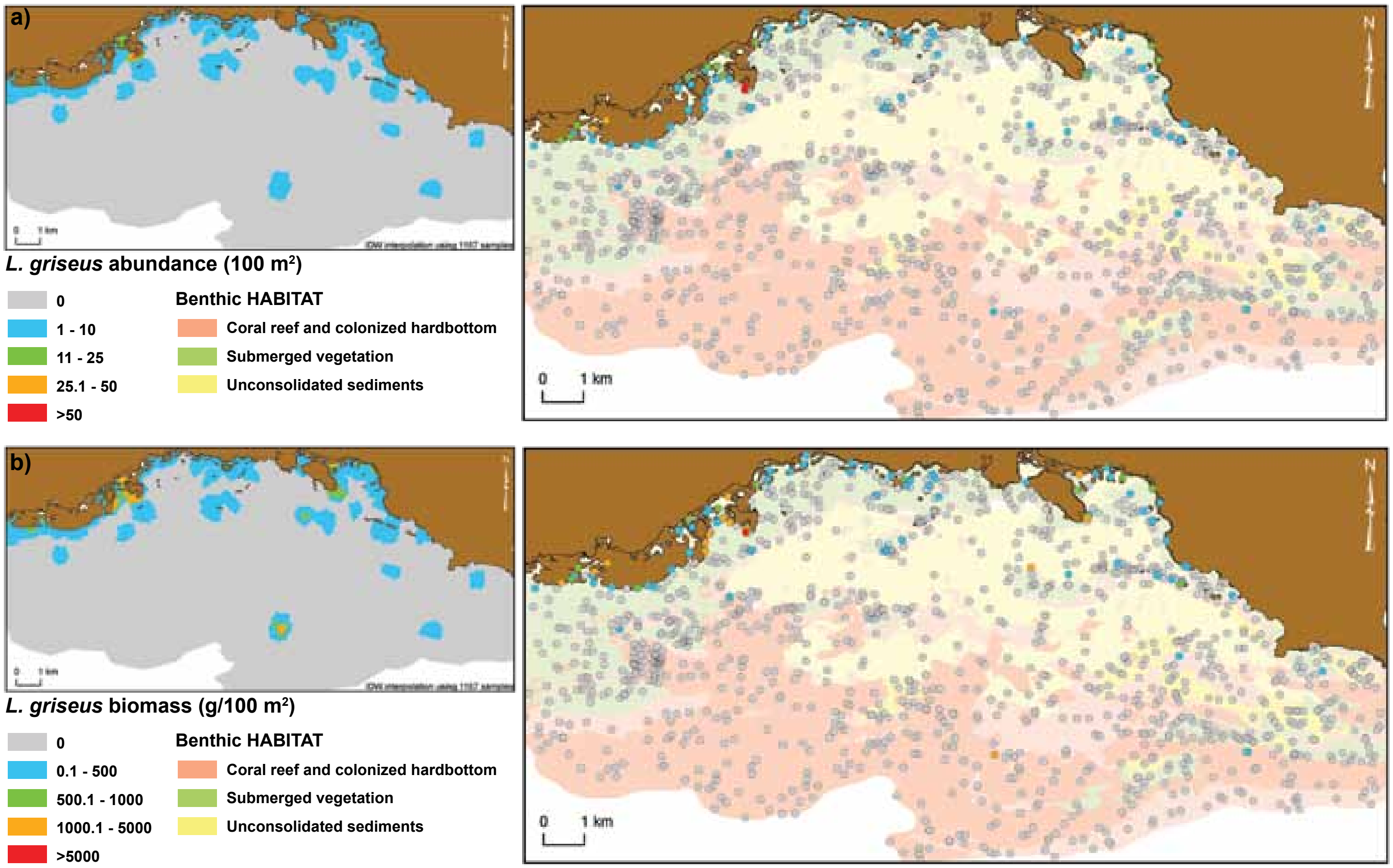


*Figure 3.30. Maps of the interpolated (left map) and spatial (right map) distributions for gray snapper (*L. griseus*): (a) abundance and (b) biomass.*

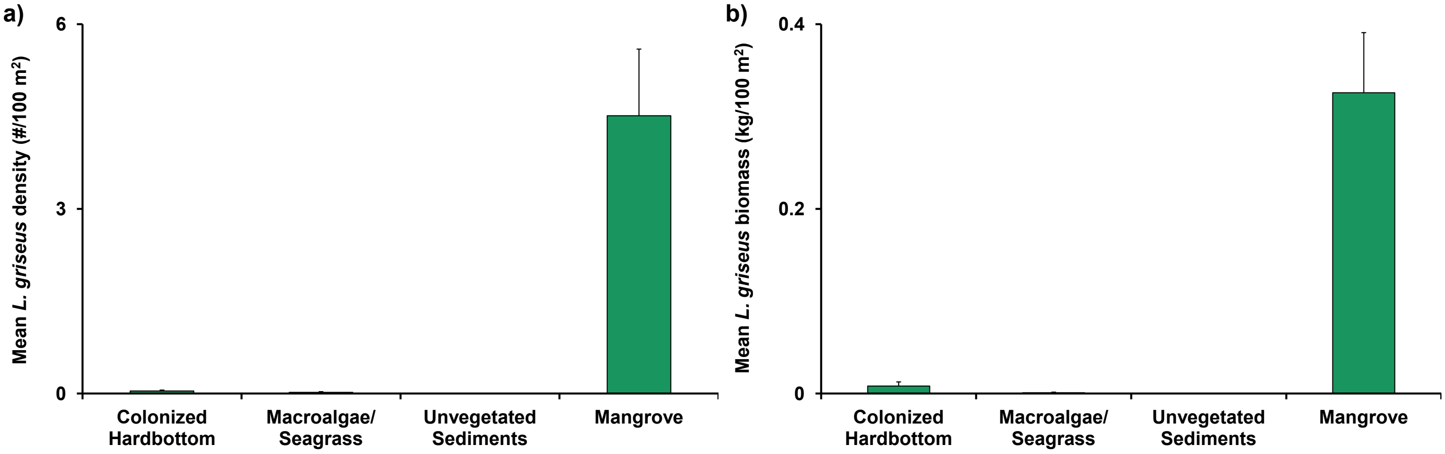


*Figure 3.31. Comparison of mean (+ SE) density and biomass by substrate type in the southwest Puerto Rico study area for gray snapper (*L. griseus*).*

L. griseus juveniles, subadults and adults coexisted both geographically and in the same habitat types across the study area (Figure 3.32). Most of the juveniles, subadults and adults were sighted in mangroves, but subadults and adults were also very occasionally seen over colonized hardbottom of the mid and outer shelf zones (Figure 3.32). *L. griseus* exhibited a more inshore distribution than did *L. apodus* (Figures 3.29 and 3.32). While juveniles/subadults were sighted in eight of 11 habitat types, adults exhibited a less general distribution being only observed in four of 11 habitat types (Figure 3.32).

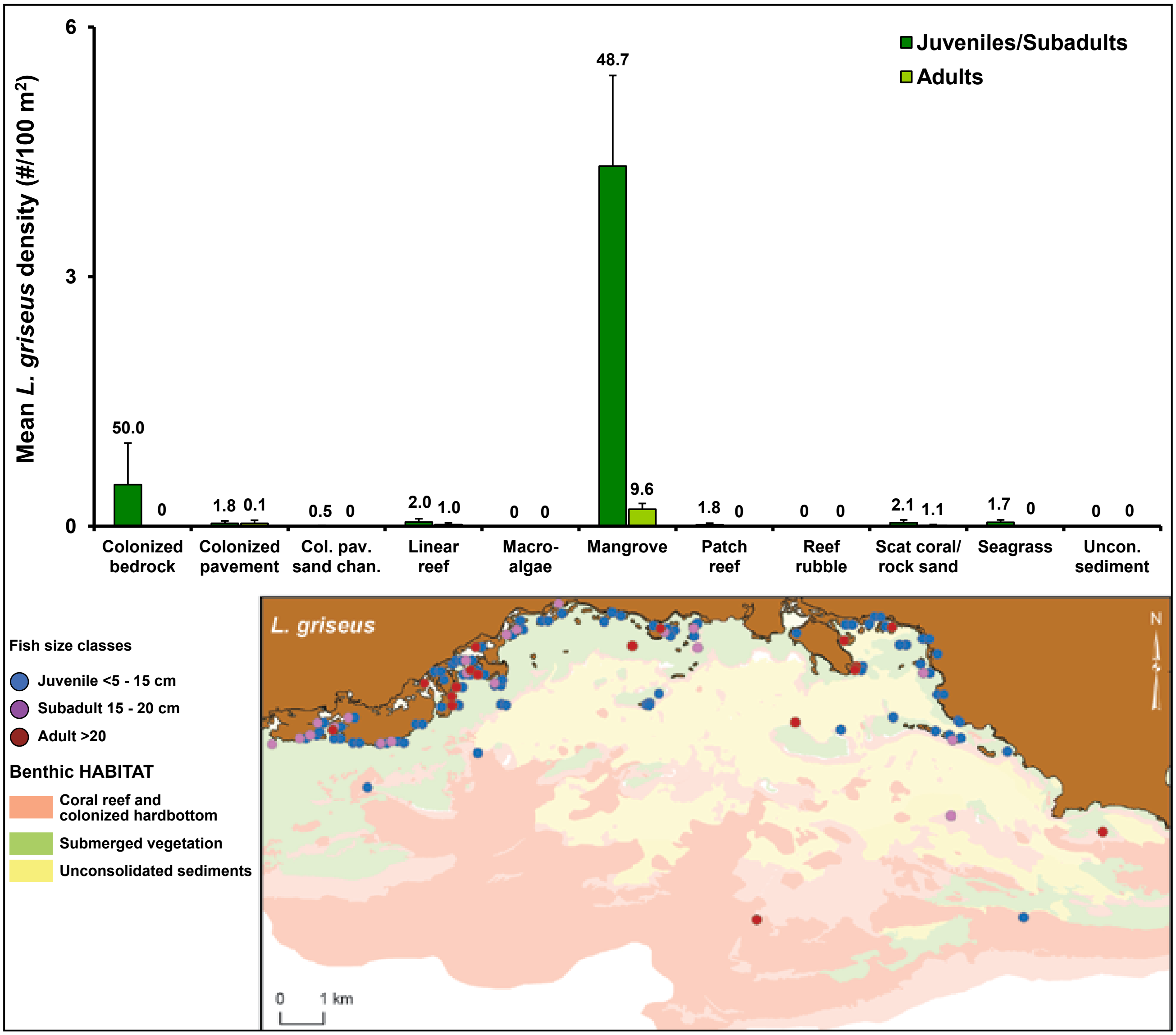


*Figure 3.32. Mean density (±SE) for juvenile/subadult and adult by mapped habitat type for gray snapper (*L. griseus*). Number above error bar indicates percent occurrence. Bottom: Spatial distribution of juvenile, subadult and adult for gray snapper (*L. griseus*) in the southwest Puerto Rico study area.*

Mahogany snapper (*Lutjanus mahogoni*)

L. mahogoni was infrequently sighted across the study area (Figure 3.33a), with highest mean abundance recorded for mangroves and lowest for seagrasses (Figure 3.34a). Highest biomass was recorded for unvegetated sediments (Figure 3.34b) all of which were in close proximity to colonized hardbottom areas (Figure 3.33b).

Mahogany snapper (*Lutjanus mahogoni*) in the Bahamas

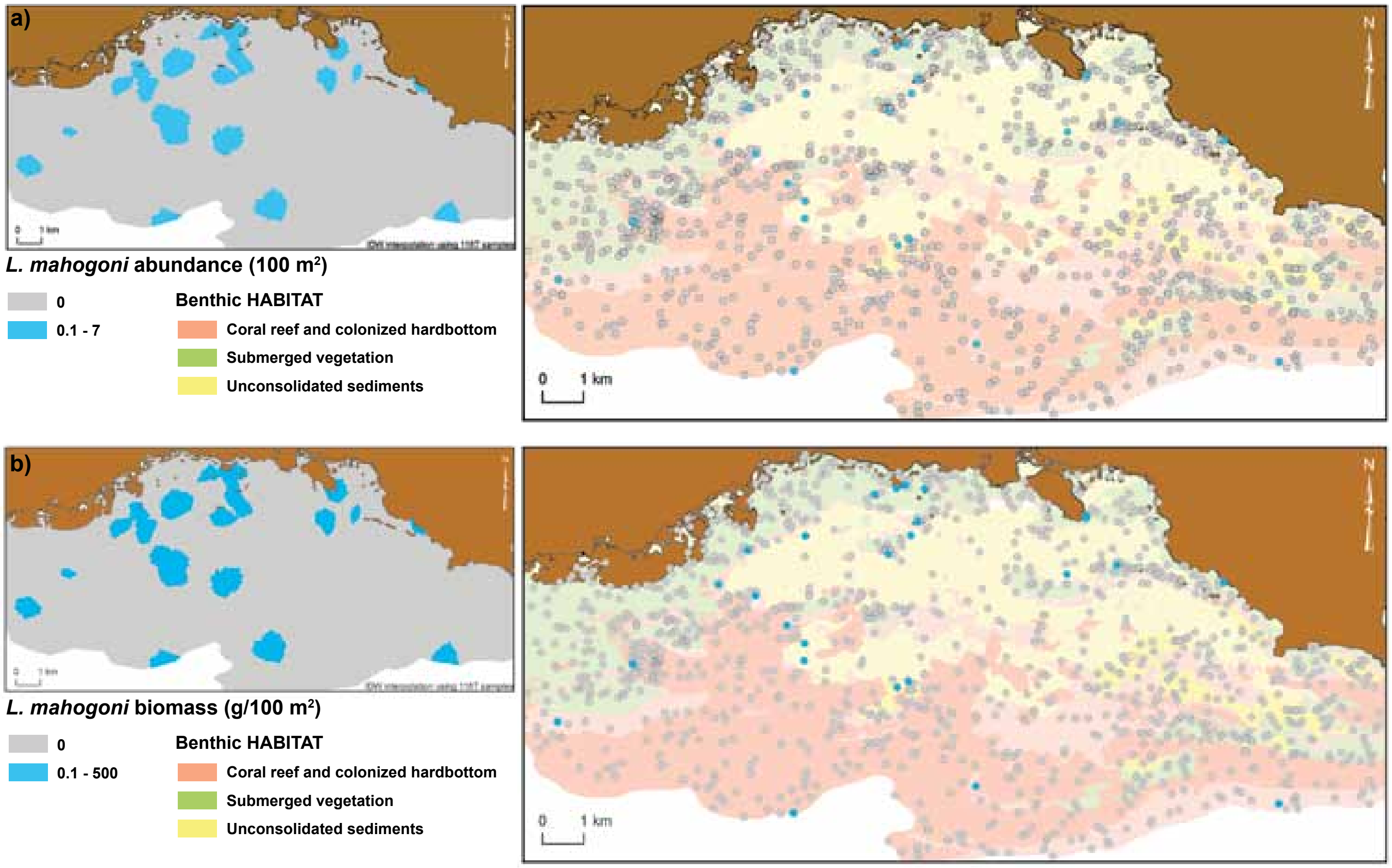


*Figure 3.33. Maps of the interpolated (left map) and spatial (right map) distributions for mahogany snapper (*L. mahogoni*): (a) abundance and (b) biomass.*

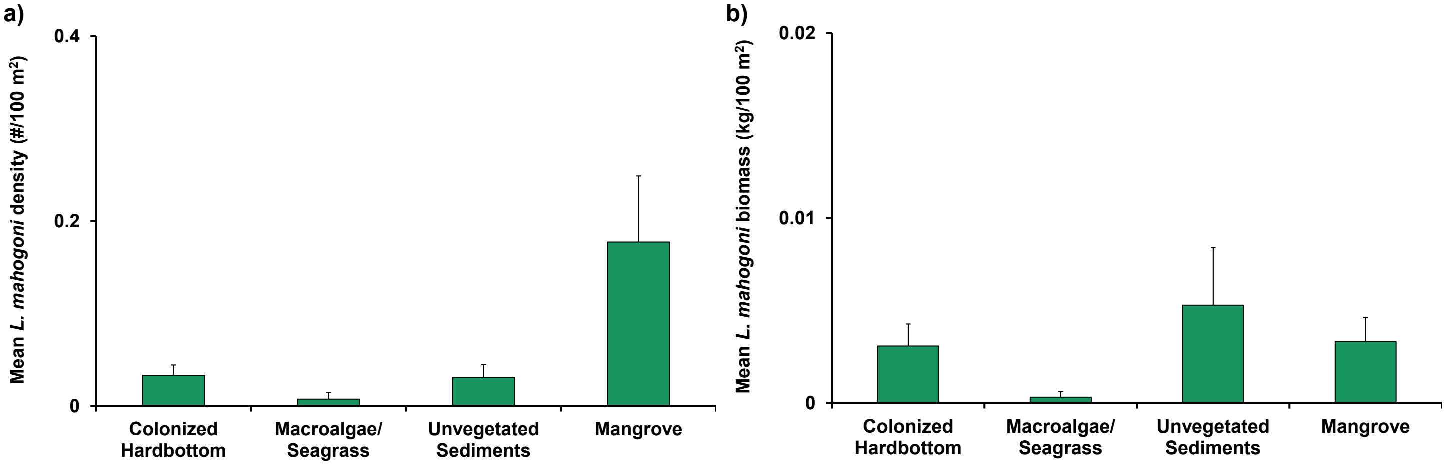


*Figure 3.34. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for mahogany snapper (*L. mahogoni*).*

L. mahogoni were relatively uncommon in the study area. Juveniles/subadults were associated with five of 11 habitat types and adults with six of 11 habitat types (Figure 3.35). Juveniles of *L. mahogoni* were found only in the nearshore and lagoonal zone, with highest mean densities in mangroves and linear reefs (Figure 3.35). In contrast, subadults and adults were sighted most frequently over patch reefs, linear reefs and unconsolidated sediments on the mid and outer shelf zones (Figure 3.35).

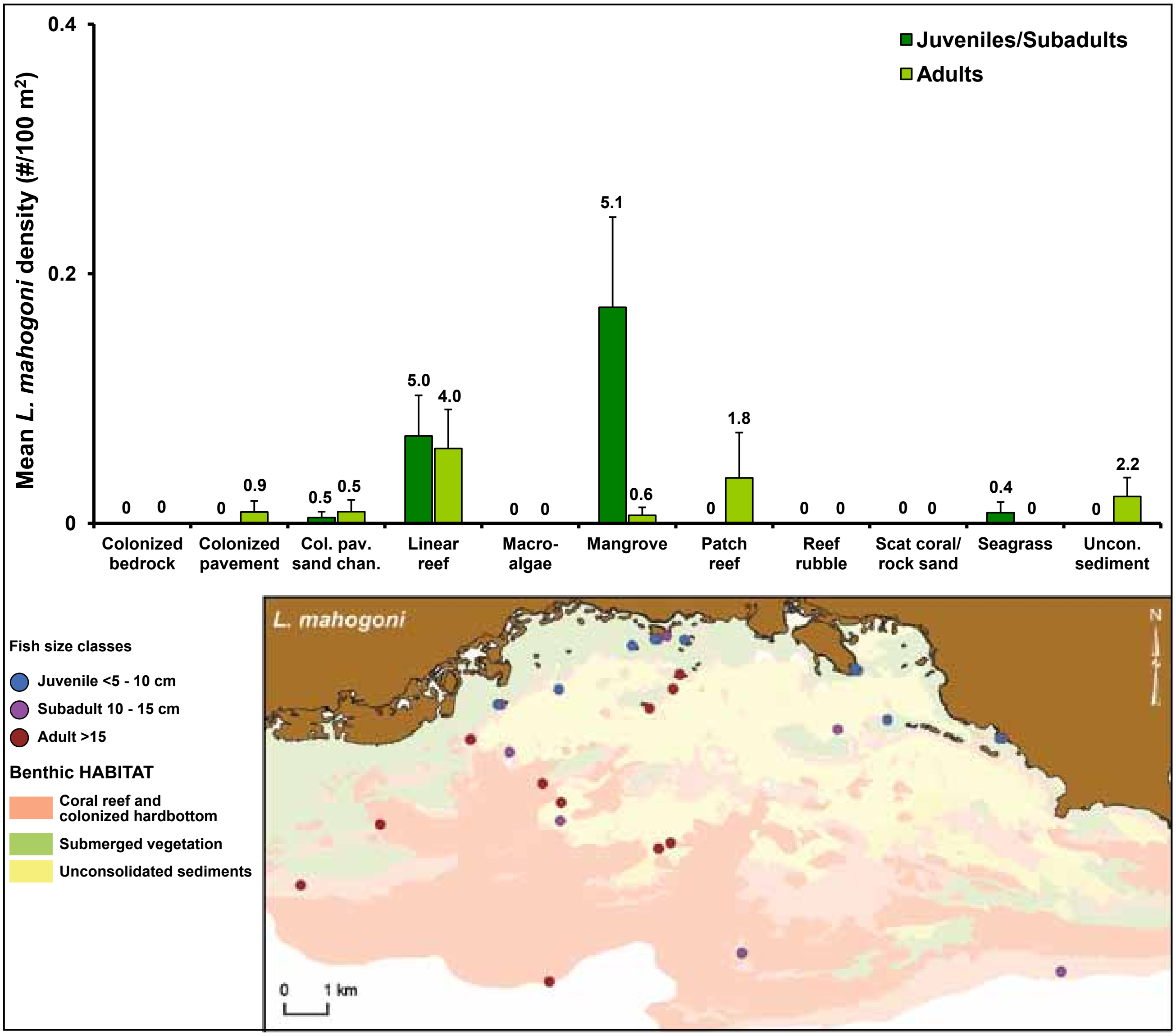


*Figure 3.35. Mean density (± SE) for juvenile/subadult and adult by mapped habitat type for mahogany snapper (*L. mahogoni*). Number above error bar indicates percent occurrence. Bottom: Spatial distribution of juvenile, subadult and adult for mahogany snapper (*L. mahogoni*) in the southwest Puerto Rico study area.*

Lane snapper (*Lutjanus synagris*)

L. synagris was observed across all major habitat types and occurred in lagoonal, mid shelf and outer shelf zones (Figure 3.36a). Mean abundance and biomass were highest over unvegetated sediments in close proximity to coral reefs (Figure 3.36b). Mean abundance in mangroves and seagrass beds was higher than for colonized hardbottom, but the reverse was true for mean biomass (Figure 3.37).

Lane snapper (*Lutjanus synagris*) in Honduras. Photo: Les Wilk

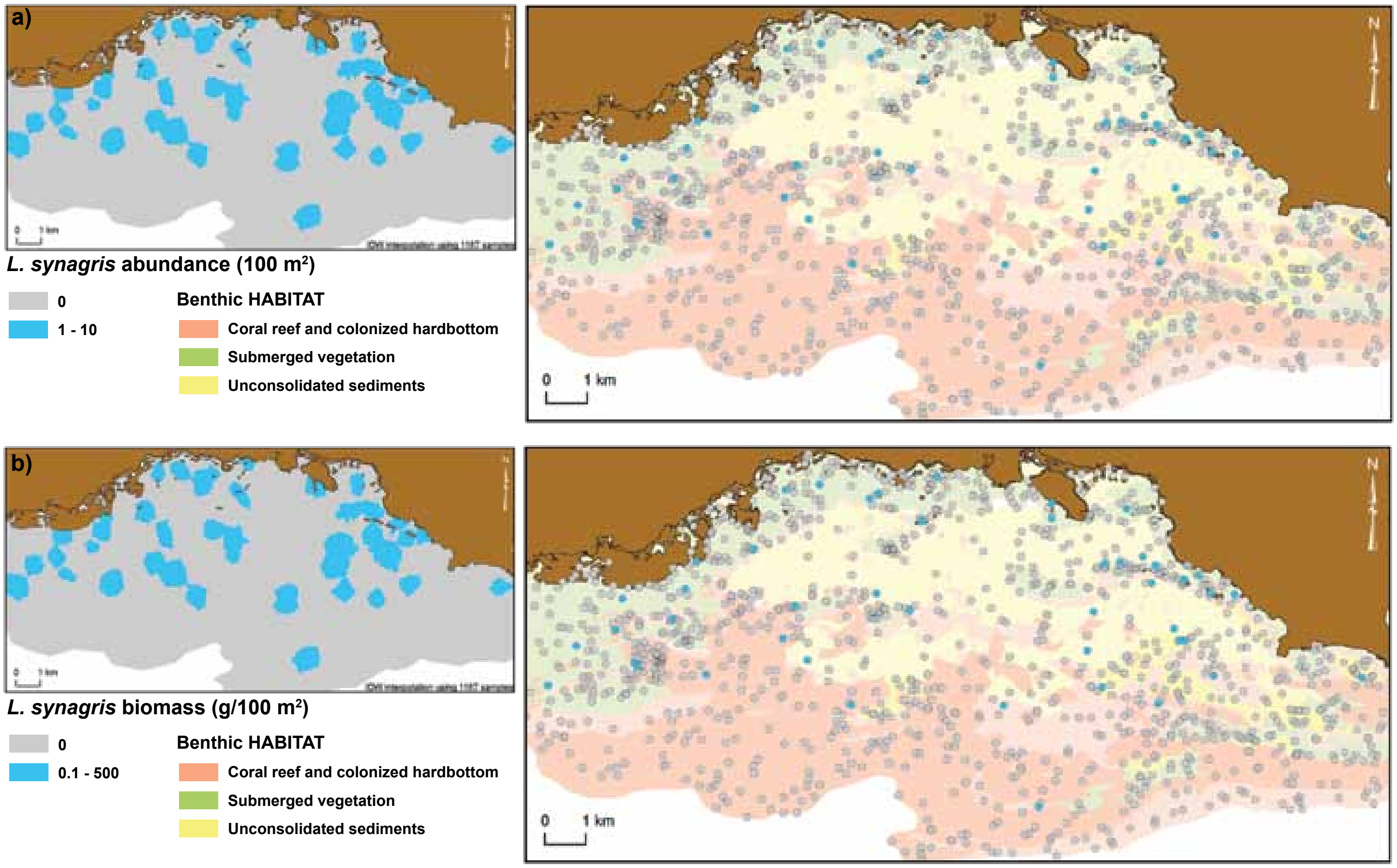


Figure 3.36. Maps of the interpolated (left map) and spatial (right map) distributions for lane snapper (L. synagris): (a) abundance and (b) biomass.

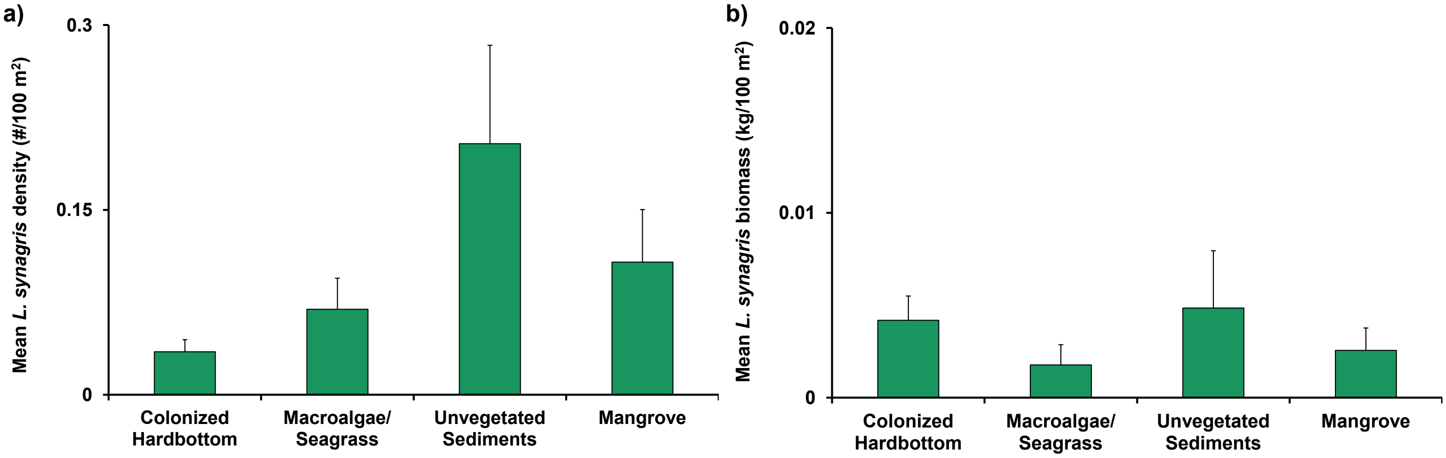


Figure 3.37. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for lane snapper (L. synagris).

Juvenile *L. synagris* were sighted in nearshore and lagoonal environments, as well as the mid-shelf zone, but were not observed on the outer shelf (Figure 3.38). Juveniles/subadults utilized a wide range of hard and softbottom habitat types, with highest densities in colonized bedrock, patch reefs and mangroves. Although juveniles were most frequently seen over colonized hardbottom, the sightings were always in close proximity to seagrasses. Adults were only sighted over colonized pavement with sand channels, colonized pavement and unconsolidated sediments mostly located farther offshore, near the boundary between extensive contiguous hardbottom and adjacent softbottom habitat types (Figure 3.38).

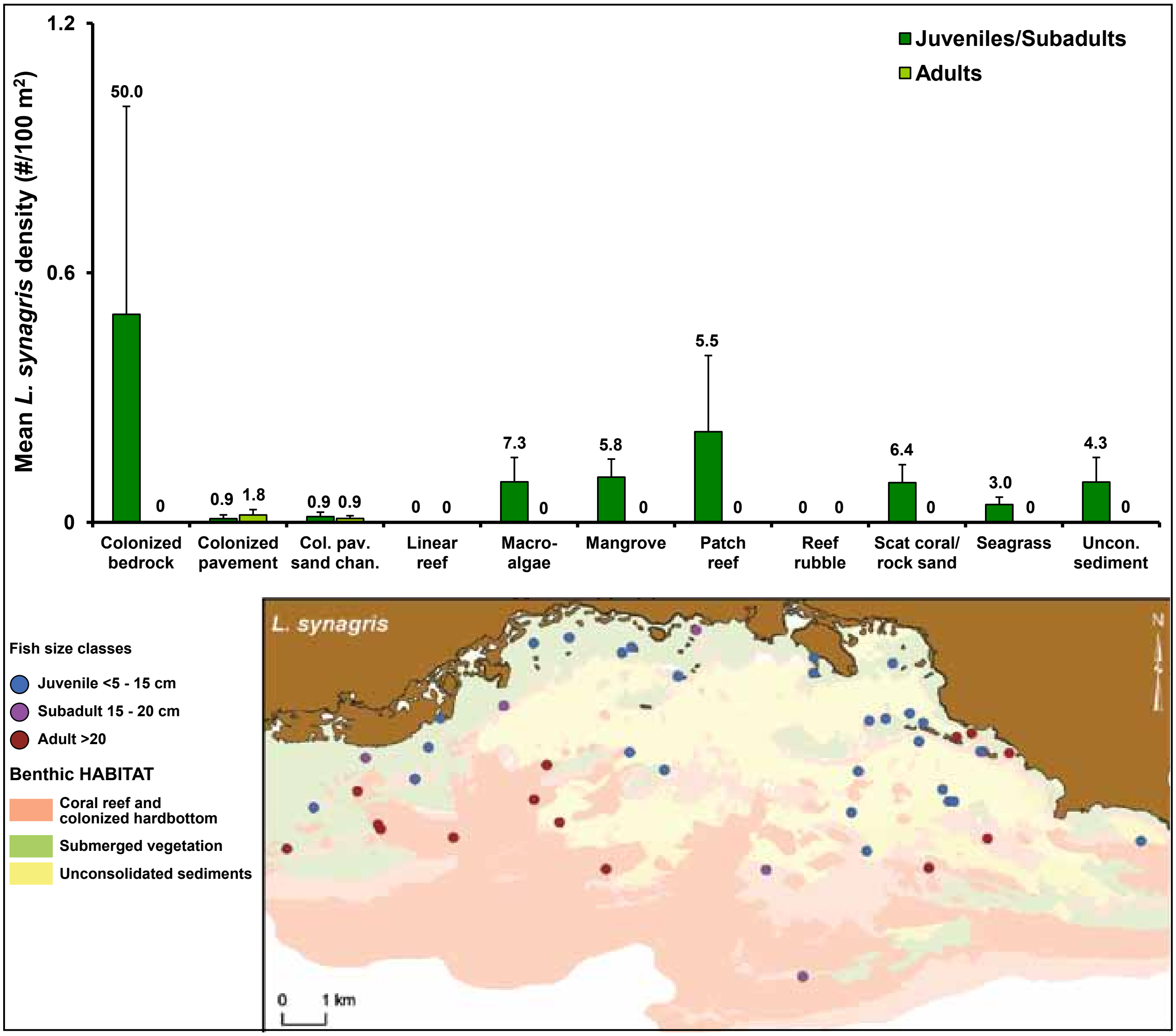


*Figure 3.38. Mean density (±SE) for juvenile/subadult and adult by mapped habitat type for lane snapper (*L. synagris*). Number above error bar indicates percent occurrence. Bottom: Spatial distribution of juvenile, subadult and adult for lane snapper (*L. synagris*) in the southwest Puerto Rico study area.*

Yellowtail snapper (*Ocyurus chrysurus*)

O. chrysurus is the most widely distributed and most abundant of the Lutjanidae in the La Parguera region, with highest mean abundance over colonized hardbottom then seagrasses and mangroves (Figures 3.39a and 3.40a). Highest mean biomass was also calculated for colonized hardbottom, with several high biomass sites occurring on the mid and outer shelf zones (Figure 3.39b). Lowest mean biomass was calculated for mangroves and seagrasses, while lowest mean abundance was for sites classified as unvegetated sediment (Figure 3.40b).

Yellowtail snapper (*Ocyurus chrysurus*)

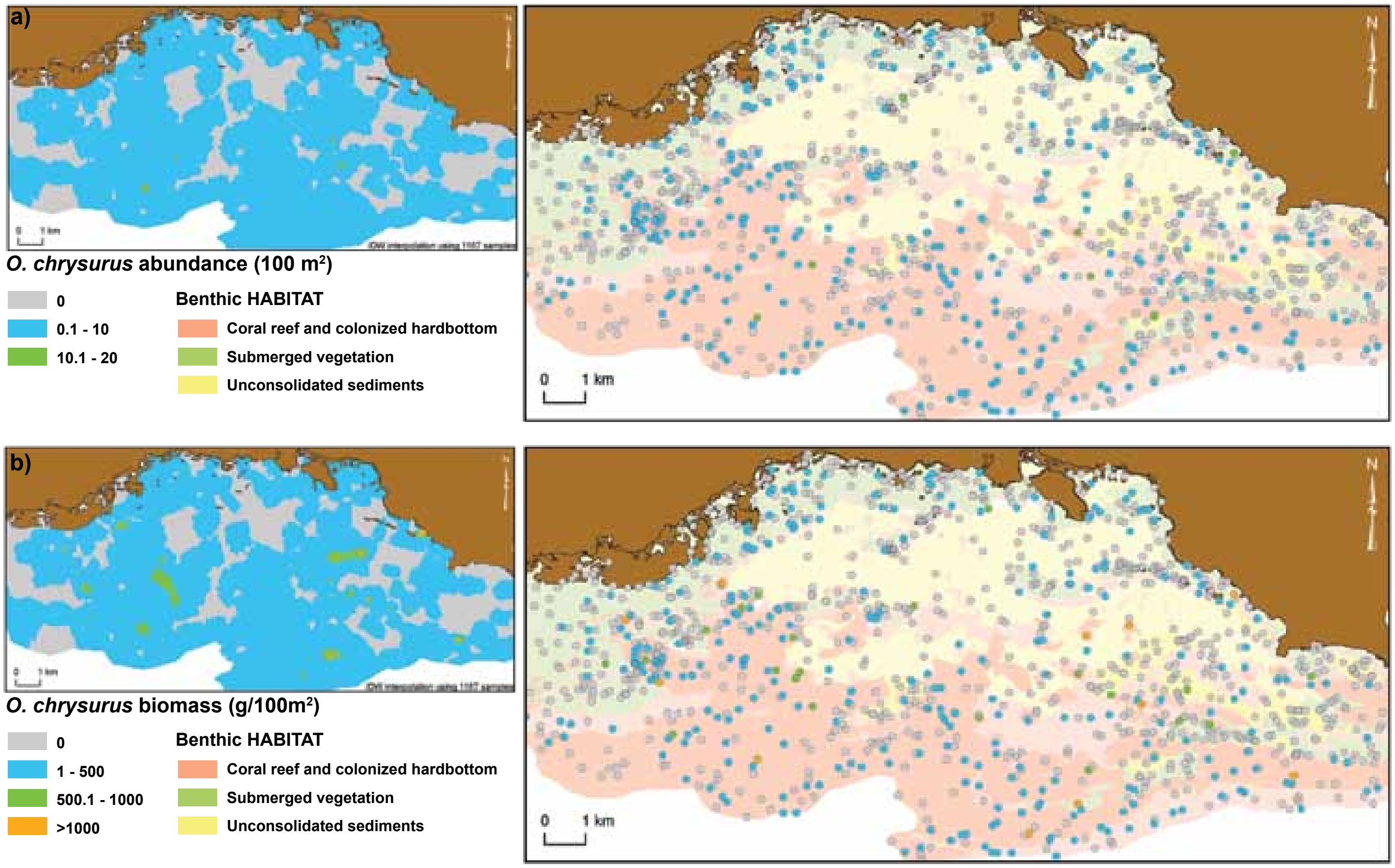


*Figure 3.39. Maps of the interpolated (left map) and spatial (right map) distributions for yellowtail snapper (*O. chysurus*): (a) abundance and (b) biomass.*

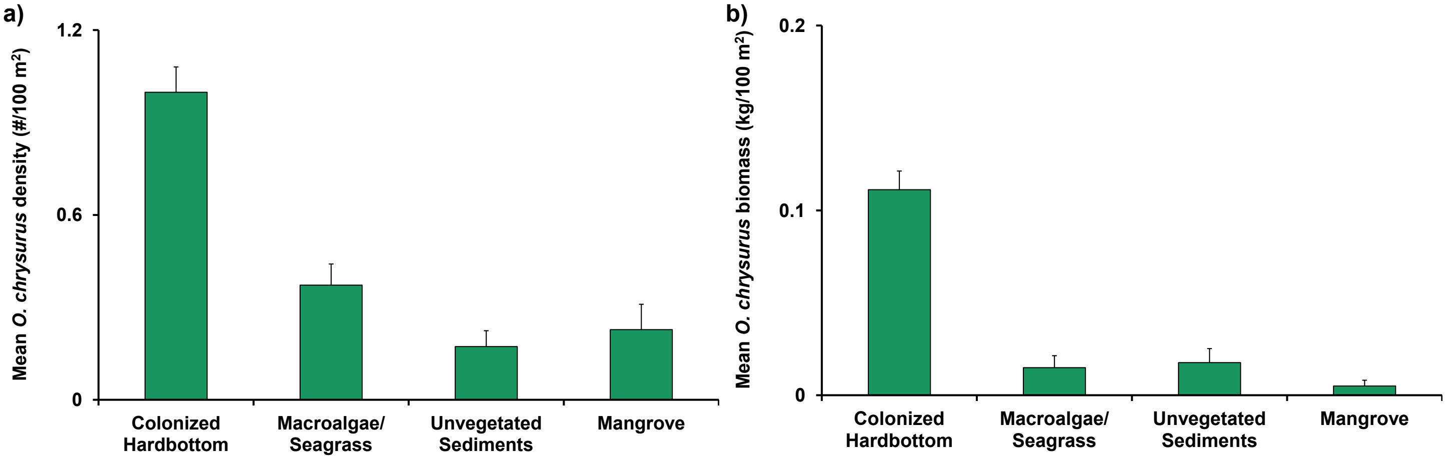


*Figure 3.40. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for yellowtail snapper (*O. chysurus*).*

O. chrysurus was the most ubiquitous of all Lutjanidae species, with juveniles sighted in 10 of the 11 habitat types and adults in eight of 11 habitat types (Figure 3.41). All life history stages coexisted across hardbottom habitat types (except reef rubble) of the study area. No geographical segregation was evident as juveniles were frequently sighted on the mid and outer shelf zones, however, juveniles utilized nearshore and lagoonal environments more frequently than did subadults and adults (Figure 3.41). In contrast to other common Lutjanidae species, adult *O. chrysurus* were not observed in mangroves (Figure 3.41). Juveniles and adults were observed at high occurrence over colonized pavement with sand channels, linear reefs and colonized pavement and juveniles were markedly more common over seagrasses than adults. High densities at colonized bedrock sites were due to relatively high abundance at only two sites (Figure 3.41).

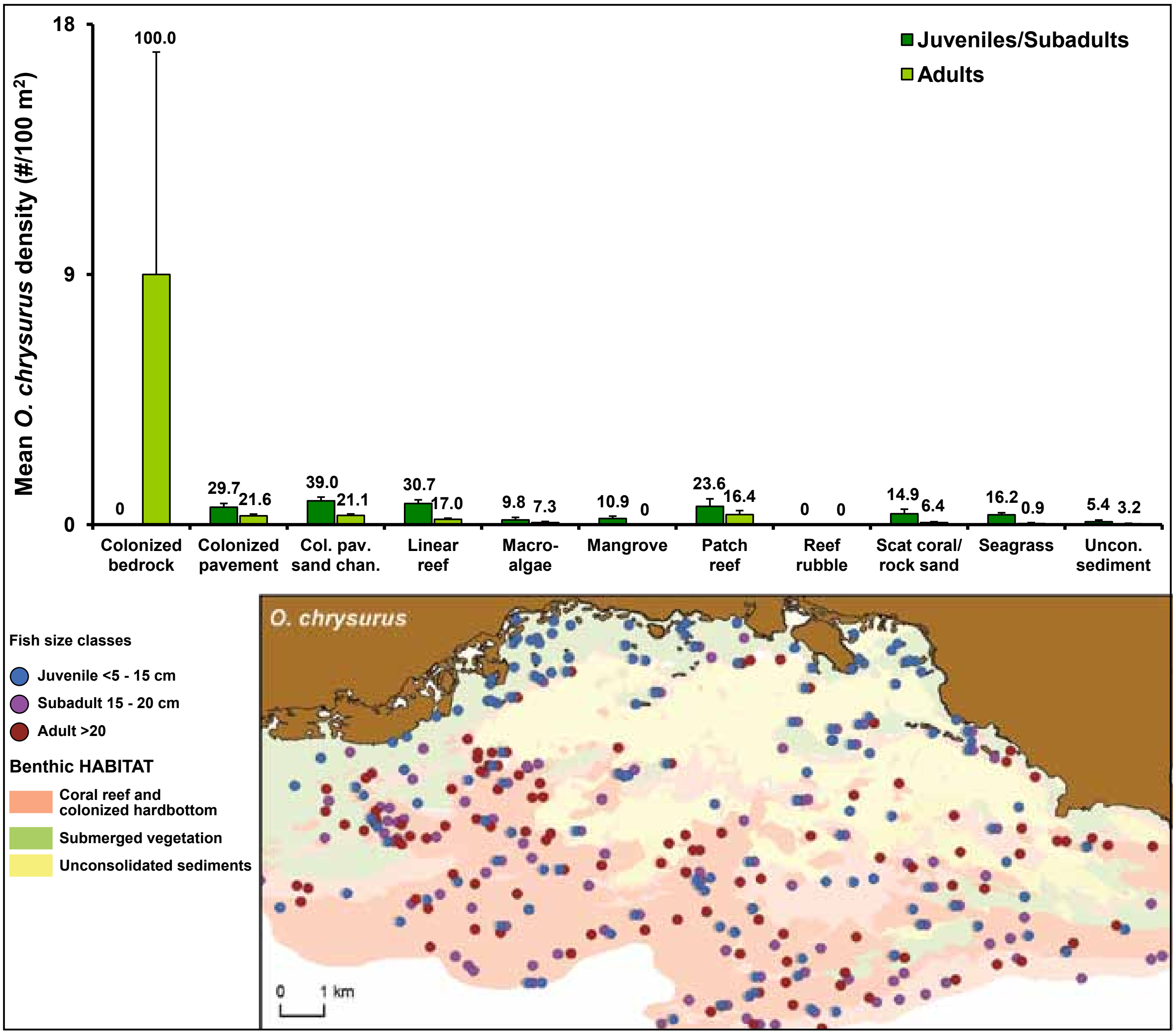


*Figure 3.41. Mean density (+ SE) for juvenile/subadult and adult by mapped habitat type for yellowtail snapper (*O. chysurus*). Number above error bar indicates percent occurrence. Bottom: Spatial distribution of juvenile, subadult and adult for yellowtail snapper (*O. chysurus*) in the southwest Puerto Rico study area.*

Parrotfish (Scaridae)

Scaridae abundance follows closely the distribution of colonized hardbottom, although the abundance is highly heterogeneous, with highest abundance across the large expanse of the most topographically complex colonized hardbottom habitat in the mid and outer shelf zones including El Palo and along some of the inner shelf linear reefs (Figures 3.42a and 3.43a). Very similar patterns were documented for Scaridae biomass, but with very low biomass for nearshore lagoonal habitat types such as fringing mangroves, sand and macroalgae and seagrasses (Figures 3.42b and 3.43b). This suggests that many of the Scaridae found in high abundance in lagoonal areas were small bodied individuals and juveniles. The largest Caribbean parrotfish were rare or absent from the study including rainbow

Redband (*Sparisoma aurofrenatum*), striped (*Scarus iseri*), and stoplight (*Scarus viride*) parrotfish

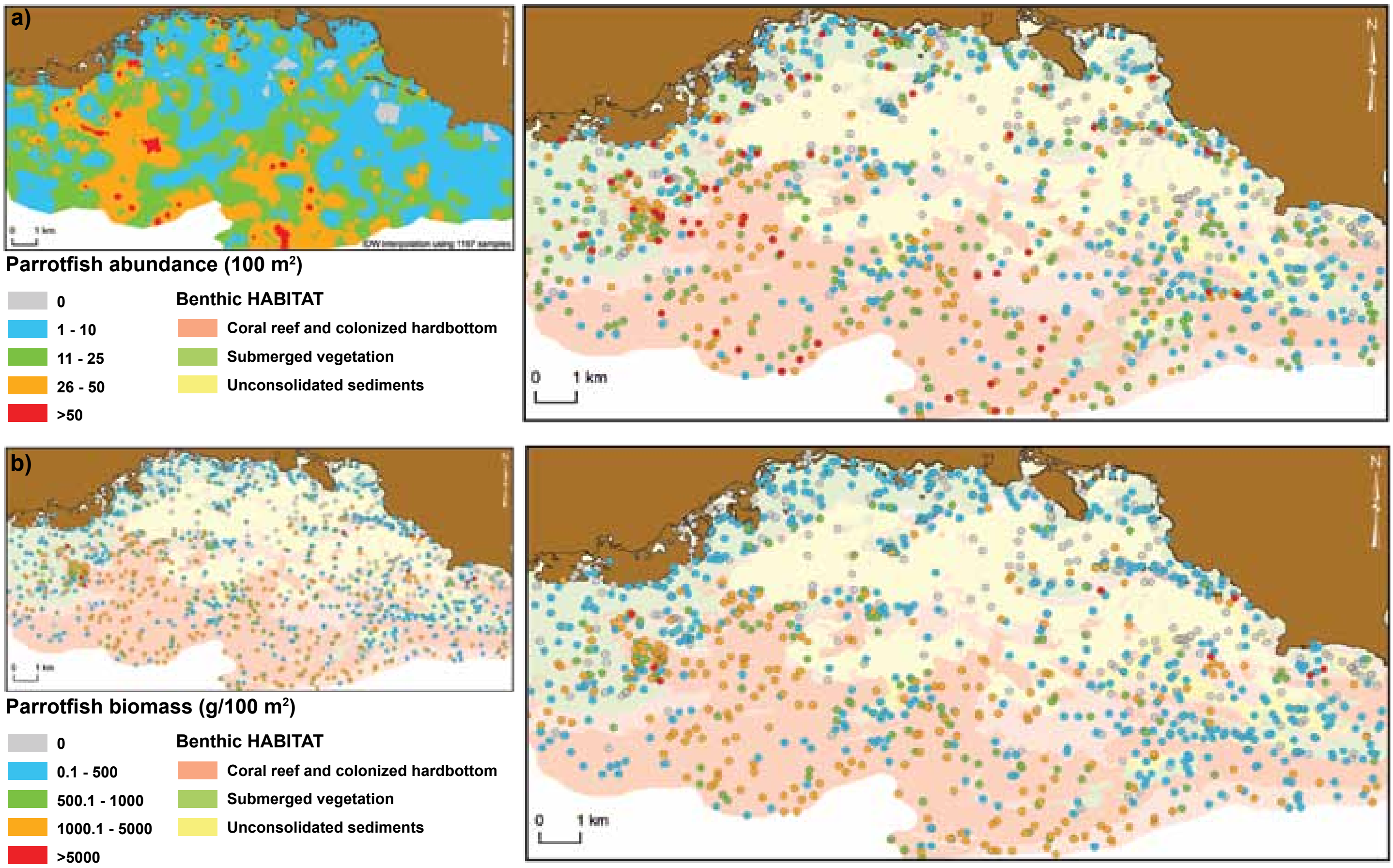


Figure 3.42. Maps of the interpolated (left map) and spatial (right map) distributions for parrotfish (Scaridae): (a) abundance and (b) biomass.

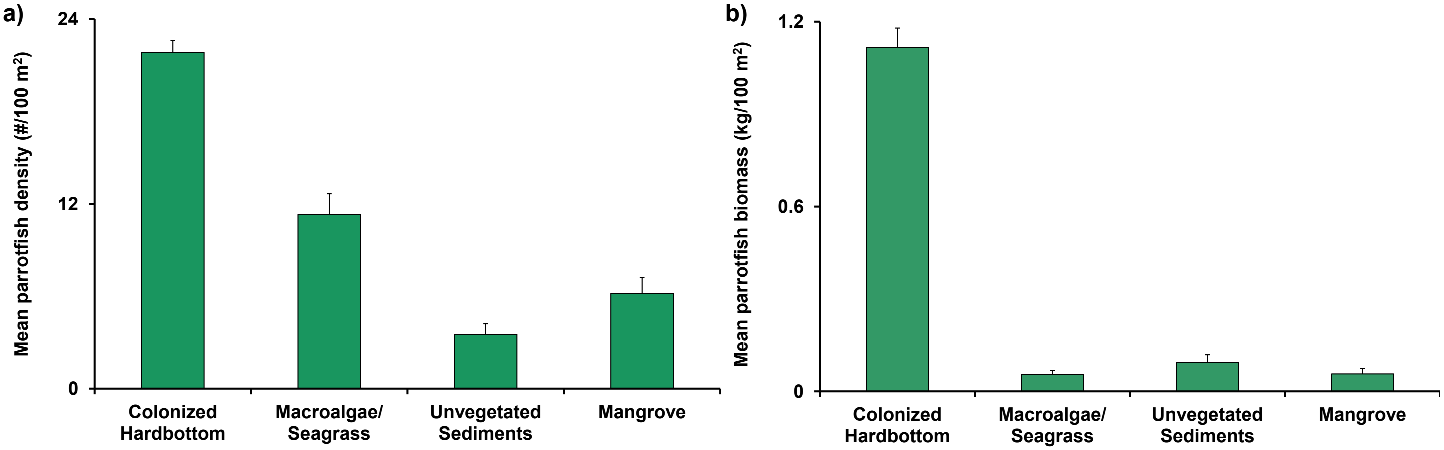


Figure 3.43. Comparison of mean (± SE) density and biomass by habitat type in the southwest Puerto Rico study area for parrotfish species (Scaridae).

parrotfish (*Scarus guacamaia*), two individuals observed. Midnight parrotfish (*Scarus coelestinus*) and blue parrotfish (*Scarus coeruleus*) were absent during the 2001-2007 sampling period.

Approximately 40% of all striped parrotfish (*Scarus iseri*) were the smallest juveniles (<5 cm; Figure 3.44a). In contrast, for princess parrotfish (*Scarus taeniopterus*) adults were more frequently seen than juveniles (Figure 3.44b). The largest *S. iseri, S. taeniopterus* and redband parrotfish (*Sparisoma aurofrenatum*) of more than 30 cm FL were very rarely seen (<3%; Figure 3.44). Yellowtail parrotfish (*Sparisoma rubripinne*) and stoplight parrotfish (*Sparisoma viride*), were the only species with individuals more than 35 cm (Figures 3.44d and 3.44e), albeit a very small proportion of the total (2 and 4%, respectively). The largest individual *S. rubripinne* in the study area was estimated at 40 cm FL (Figure 3.44d) and the maximum recorded for the species is 47.8 cm TL (Table 3.4). The largest individual *S. viride* in the study area was estimated at 50 cm FL (Figure 3.44e) and the maximum recorded for the species is 64 cm TL (Table 3.4).

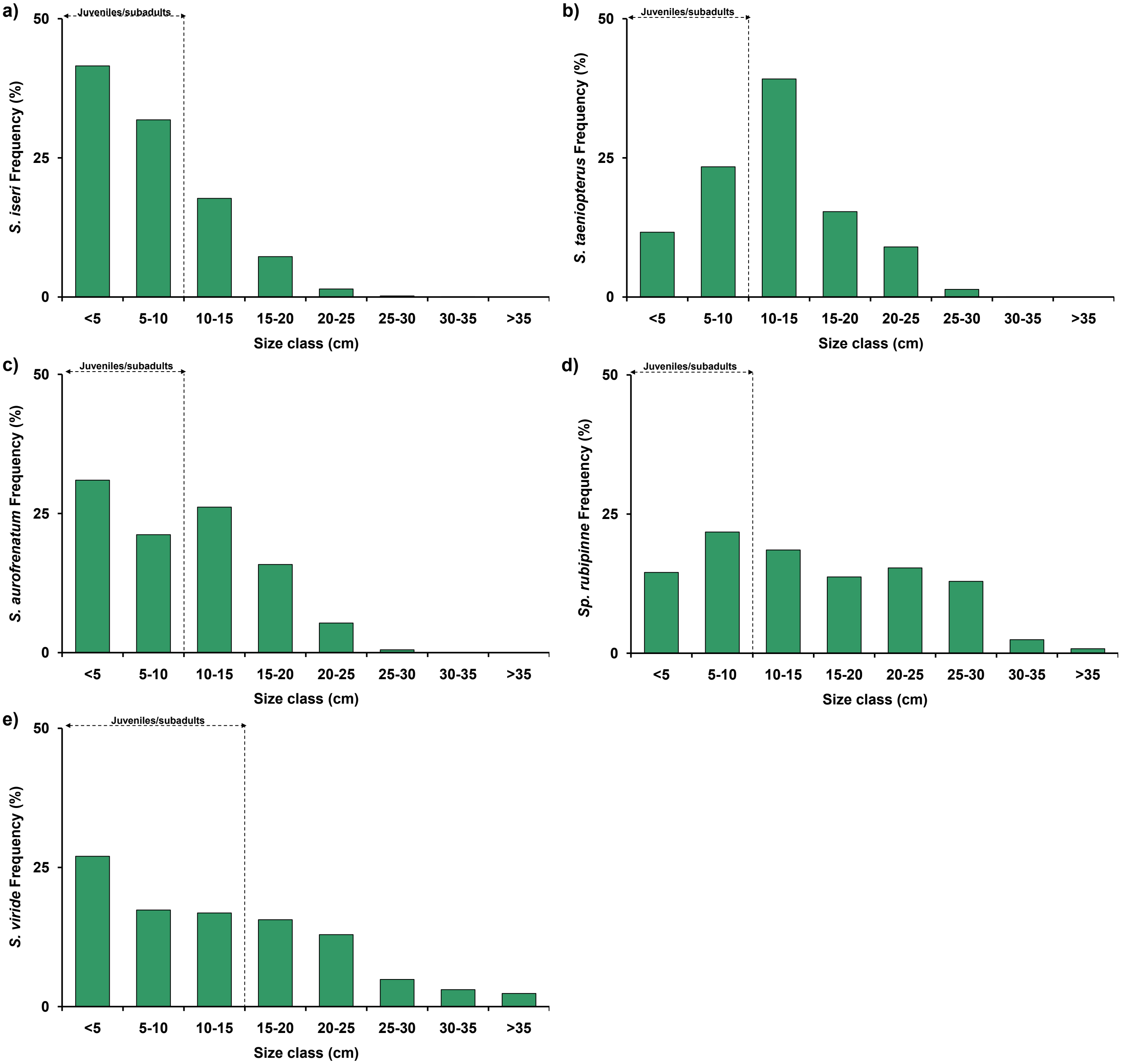
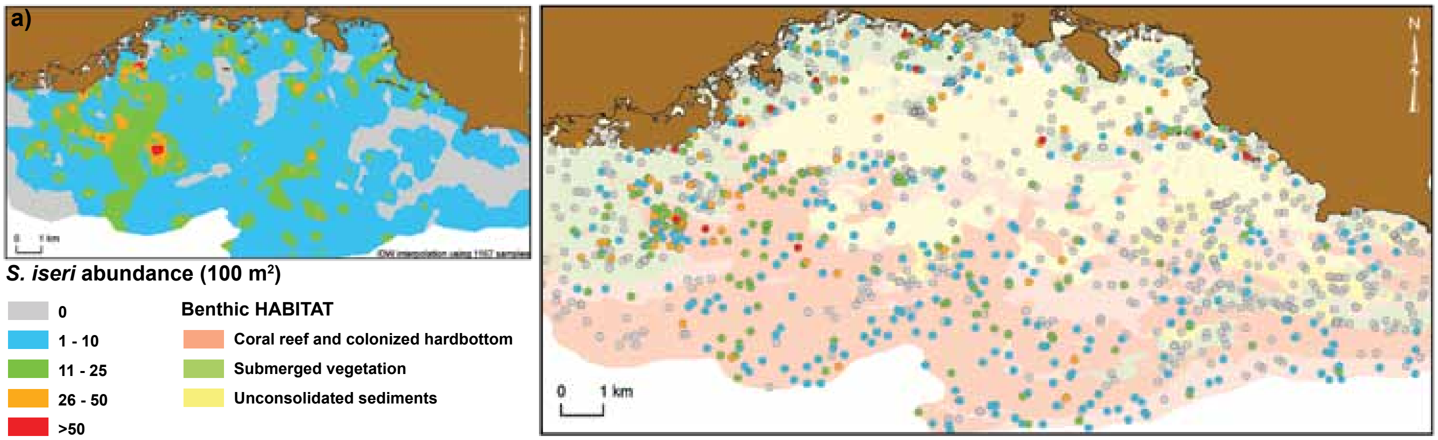


*Figure 3.44. Size frequency histogram for select parrotfish (Scaridae) in the southwest Puerto Rico study area. (a) striped parrotfish (*S. iseri*), (b) princess parrotfish (*S. taeniopterus*), (c) redband parrotfish (*S. aurofrenatum*), (d) yellowtail parrotfish (*S. rubripinne*) and (e) stoplight parrotfish (*S. viride*).*

Striped parrotfish (*Scarus iseri*)

S. iseri is the most frequently observed parrotfish across the study area found in all major habitat types and distributed widely across lagoonal, mid-shelf and outer shelf zones (Figure 3.45). Abundance and biomass is higher on the western side of the study area, particularly over highly rugose colonized hardbottom areas (Figure 3.45). Mean abundance was relatively high in colonized hardbottom, seagrasses and mangroves, but biomass was markedly higher in colonized hardbottom than any other habitat type (Figure 3.46).

Terminal (left) and juvenile/initial (right) phase *S. iseri*.



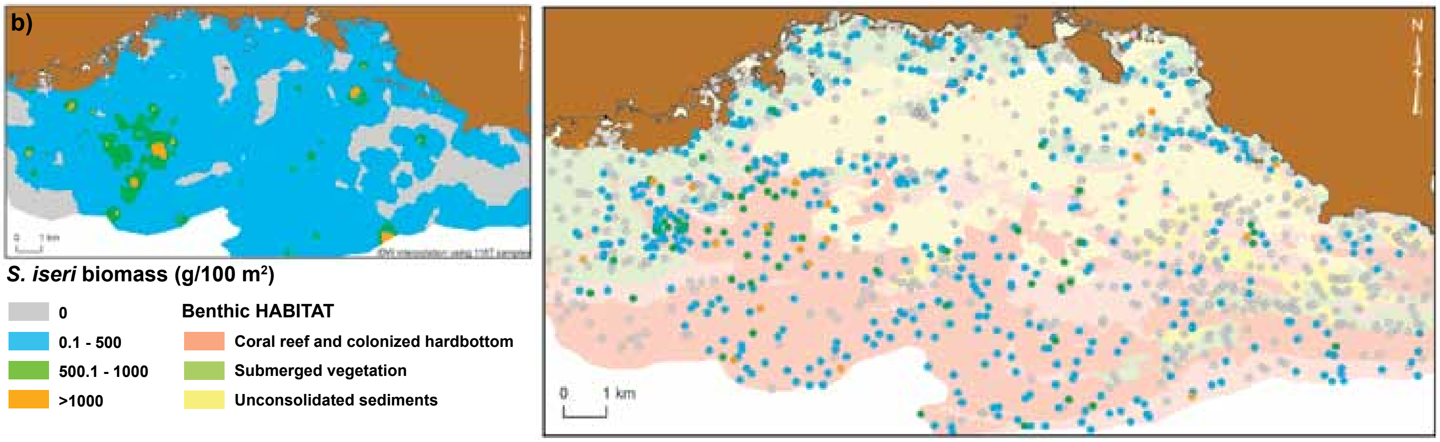


*Figure 3.45. Maps of the interpolated (left map) and spatial (right map) distributions for striped parrotfish (*S. iseri*): (a) abundance and (b) biomass.*

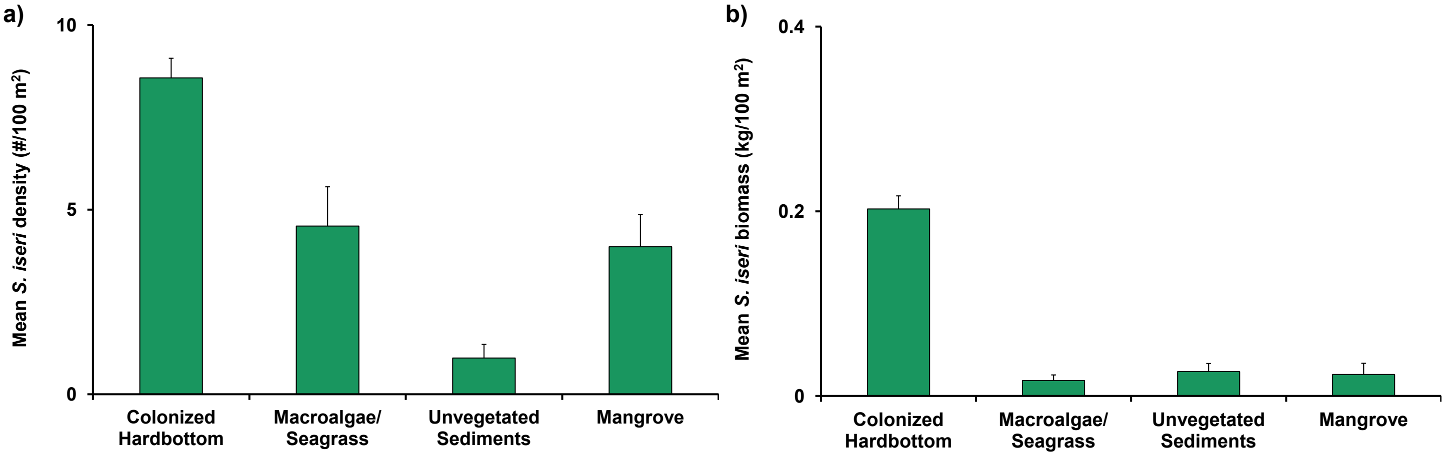


*Figure 3.46. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for striped parrotfish (*S. iseri*).*

S. iseri juveniles and adults co-occurred in all zones and in all abundant habitat types throughout the study area, although adults used inshore mangroves and seagrasses less frequently than did juveniles (Figure 3.47). Mean densities of juveniles and adults were higher in the colonized hardbottom habitat types than softbottom habitat types such as macroalgae, seagrasses and sand. Highest densities of adults were calculated for colonized pavement with sand channels, colonized pavements and patch reefs (Figure 3.47). Highest mean densities of juveniles were calculated for patch reefs, linear reefs and seagrasses. High densities at colonized bedrock sites were due to relatively high abundance at only two sites (Figure 3.47).

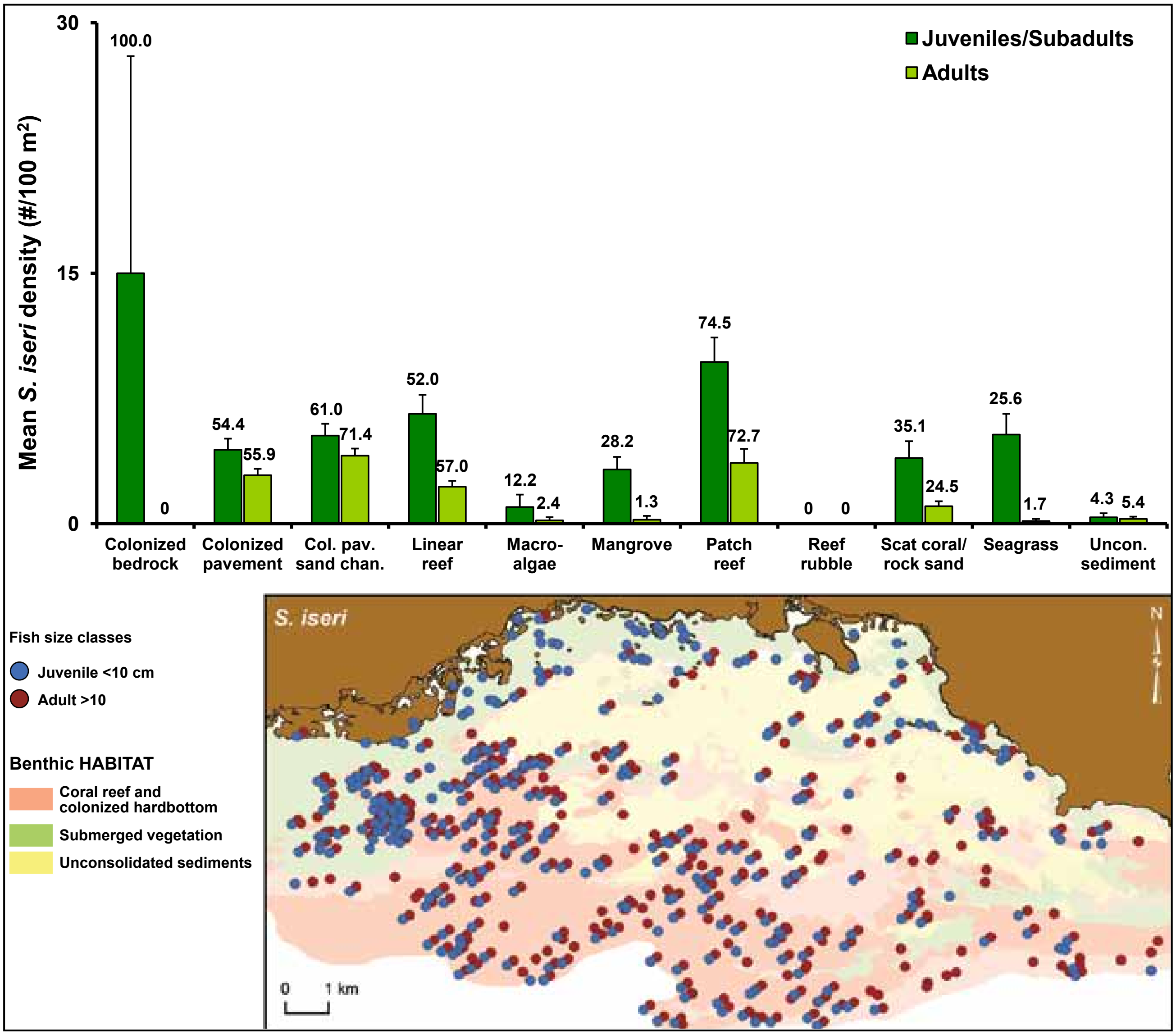
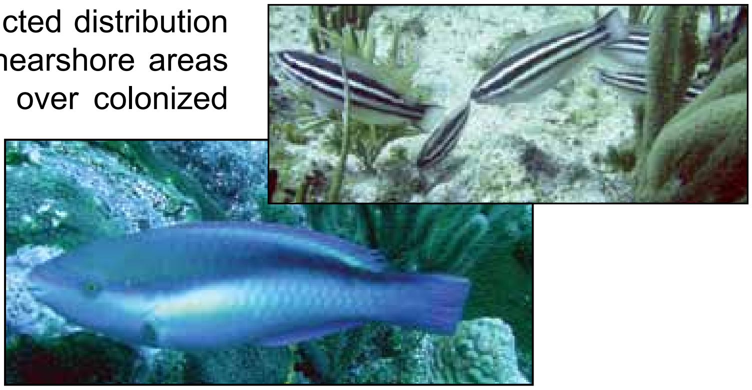


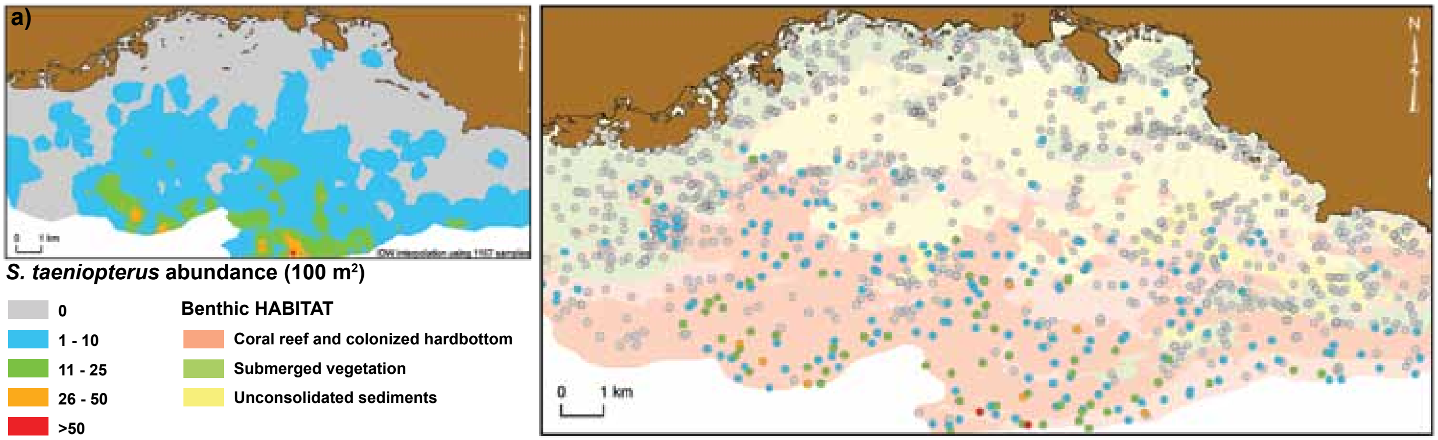
*Figure 3.47. Mean density (± SE) for juvenile/subadult and adult by mapped habitat type for striped parrotfish (*S. iseri*). Number above error bar indicates percent occurrence. Bottom: Spatial distribution of juvenile and adult for striped parrotfish (*S. iseri*) in the southwest Puerto Rico study area.*

Princess parrotfish (*Scarus taeniopterus*)

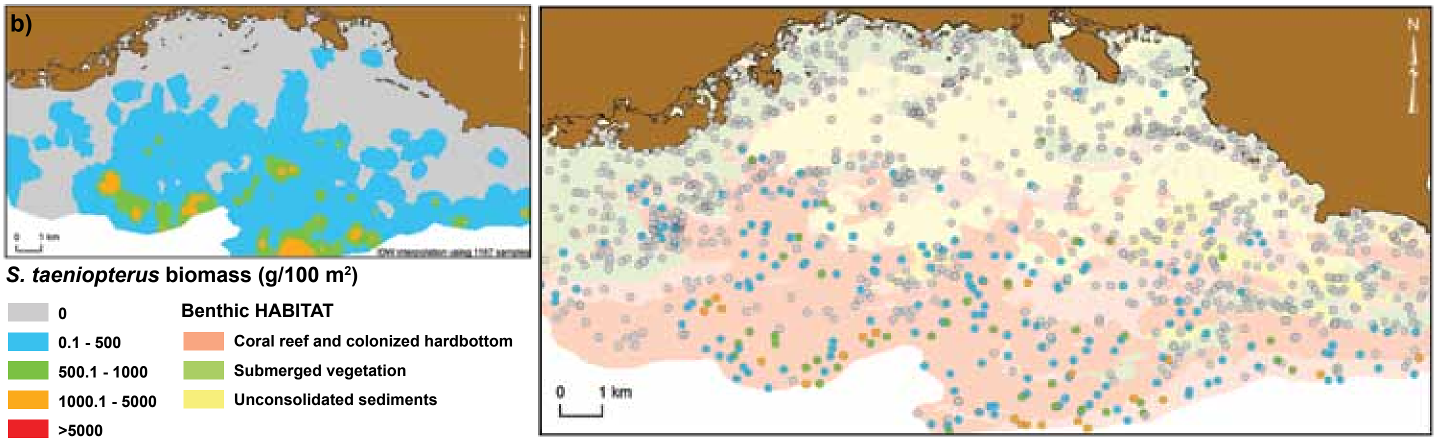
S. taeniopterus has a more geographically restricted distribution than *S. iseri*, with it being almost absent from nearshore areas and highest abundance and biomass observed over colonized hardbottom along the outer shelf zone (Figure 3.48). *S. taeniopterus* was more frequently observed over areas of highly contiguous colonized hardbottom. It was absent from mangroves and only found in very low abundance over seagrasses and unvegetated sediments (Figure 3.49).



Terminal (bottom) and juvenile/initial (top) phase princess parrotfish (*Scarus taeniopterus*)



***S. taeniopterus* abundance (100 m^2)**



***S. taeniopterus* biomass (g/100 m^2)**

*Figure 3.48. Maps of the interpolated (left map) and spatial (right map) distributions for princess parrotfish (*S. taeniopterus*): (a) abundance and (b) biomass.*

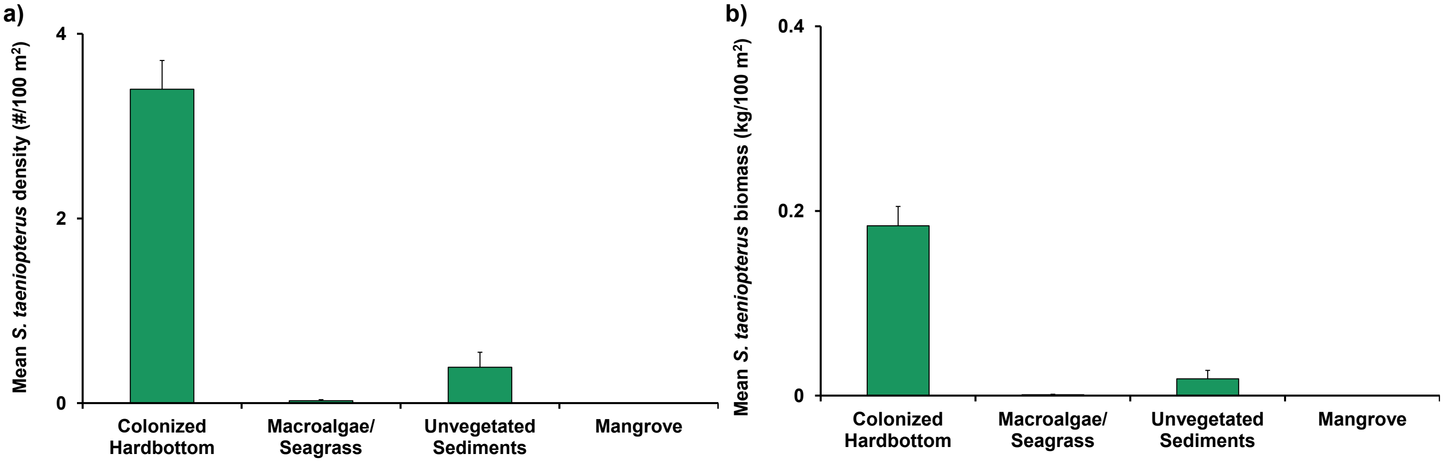


*Figure 3.49. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for princess parrotfish (*S. taeniopterus*).*

Juvenile and adult *S. taeniopterus* co-occurred across colonized hardbottom habitat types of the mid and outer shelf zones, but in contrast to *S. iseri* were rarely encountered in nearshore and lagoonal environments (Figures 3.47 and 3.50). Highest densities of juveniles and adults were calculated for the most topographically complex areas of colonized pavement, colonized pavement with sand channels, linear reefs and patch reefs. Juveniles and adults were absent from mangroves and reef rubble and occurred only once in seagrass beds (Figure 3.50).

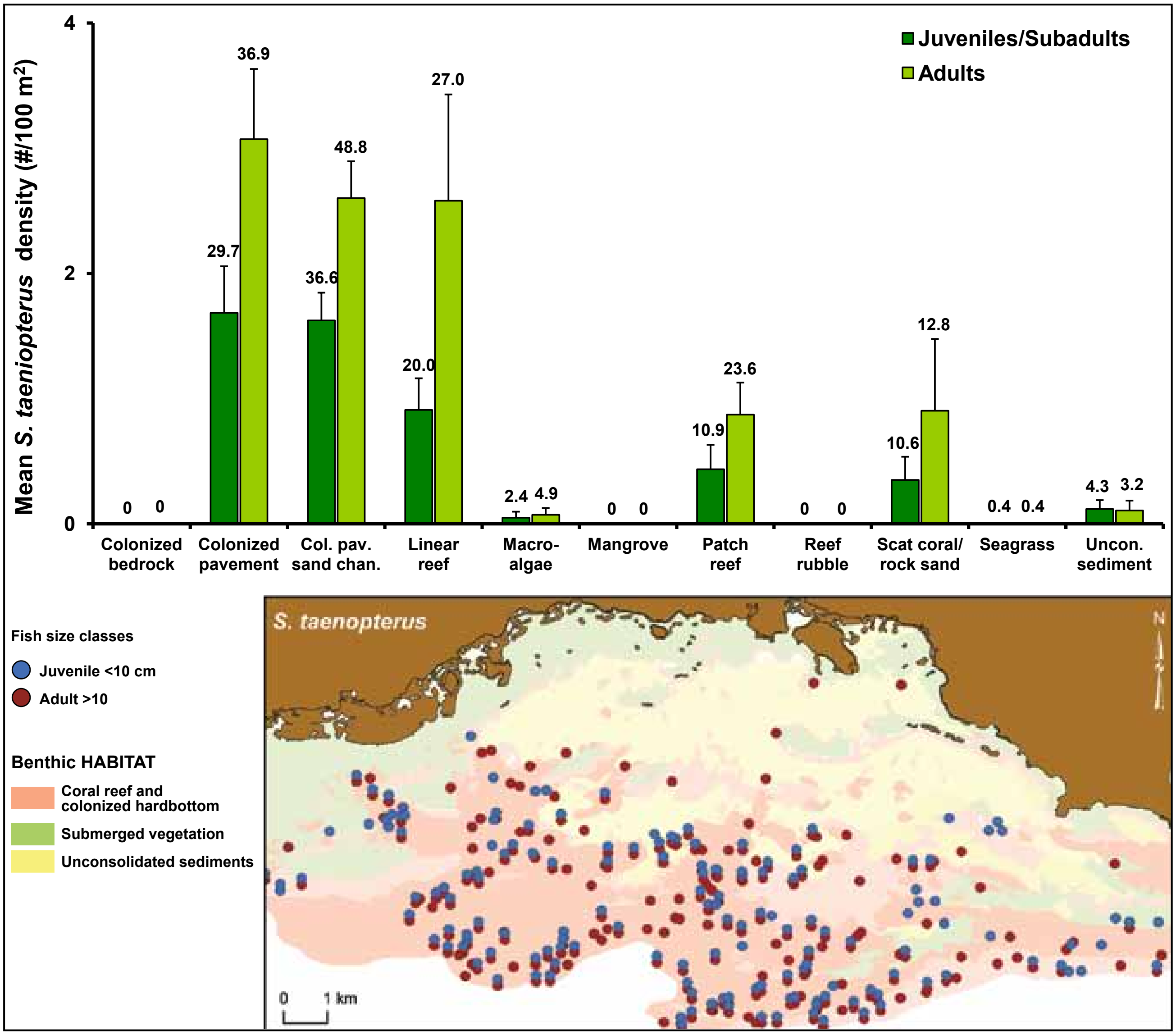


*Figure 3.50. Mean density (± SE) for juvenile/subadult and adult by mapped habitat type for princess parrotfish (*S. taeniopterus*). Number above error bar indicates percent occurrence. Bottom: Spatial distribution of juvenile and adult for princess parrotfish (*S. taeniopterus*) in the southwest Puerto Rico study area.*

Redband parrotfish (*Sparisoma aurofrenatum*)

S. aurofrenatum exhibited a close affinity with colonized hardbottom, with more than 90% of all sightings occurring over colonized hardbottom habitat types (Figures 3.51 and 3.52). Abundance and biomass were generally highest in the mid to outer shelf zones (Figure 3.51).

Terminal (bottom) and initial (top) phase (*S. aurofrenatum*.)

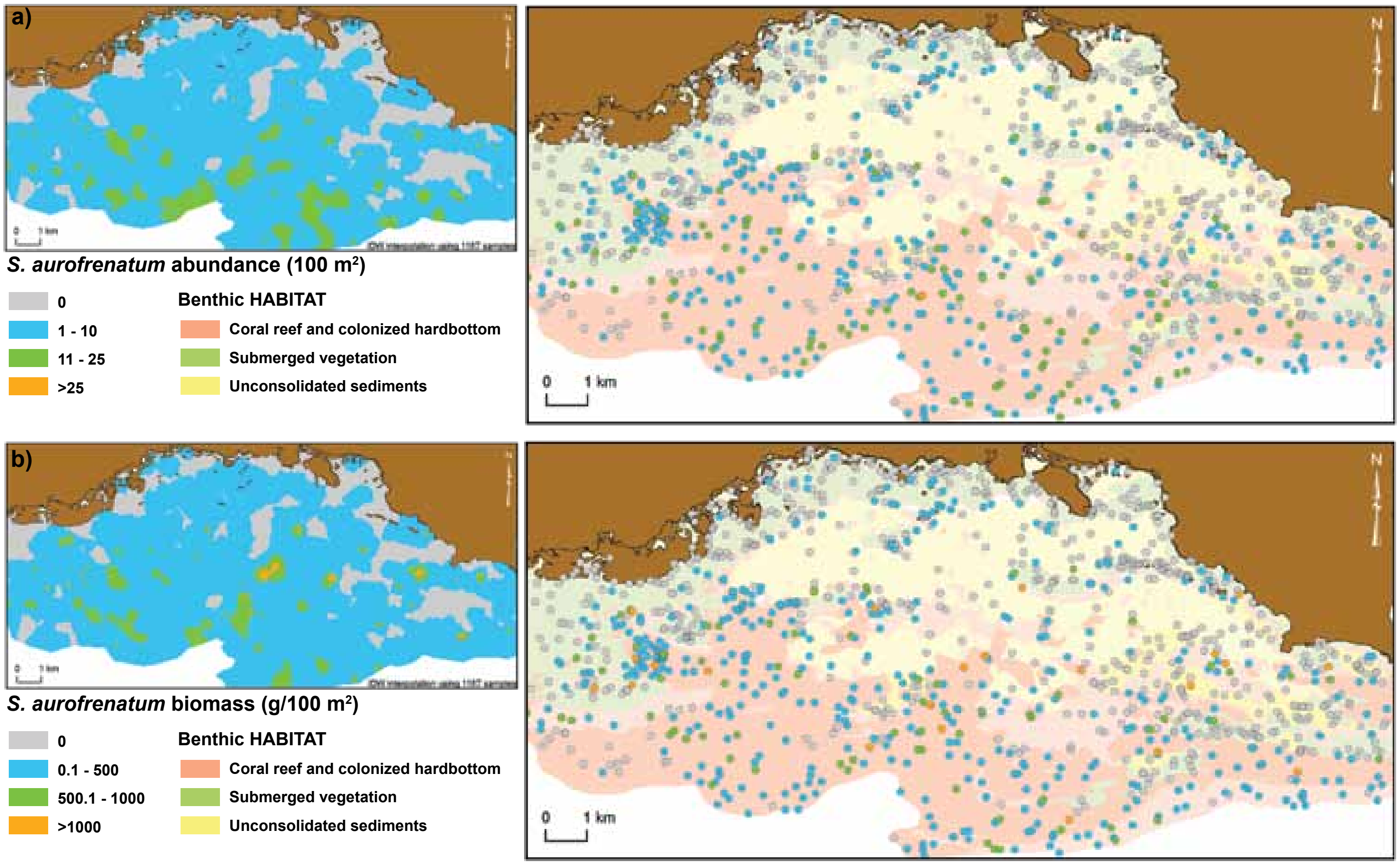


Figure 3.51. Maps of the interpolated (left map) and spatial (right map) distributions for redband parrotfish (S. aurofrenatum): (a) abundance and (b) biomass.

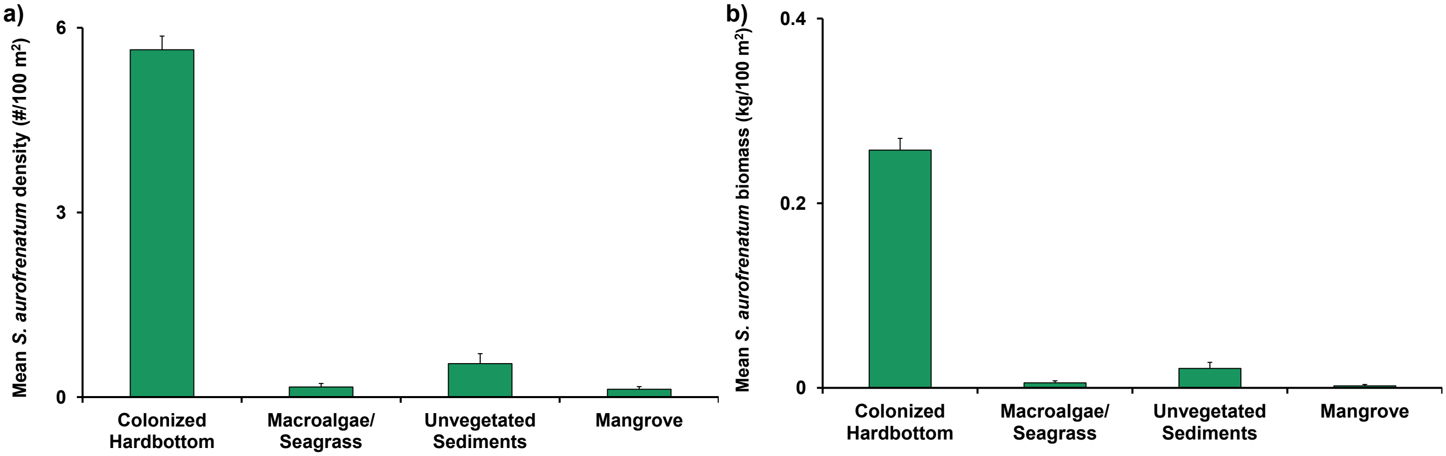


Figure 3.52. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for redband parrotfish (S. aurofrenatum).

Juvenile and adult *S. aurofrenatum* co-occurred throughout the study area and in all zones and in all habitat types except colonized bedrock (Figure 3.53). Density of juvenile and adult *S. aurofrenatum* was markedly higher over colonized hardbottom habitat types than softbottom, particularly colonized hardbottom with sand channels, patch reefs, linear reefs and colonized pavement. Softbottom habitat types and mangroves supported relatively low densities of both juveniles and adults (Figure 3.53).

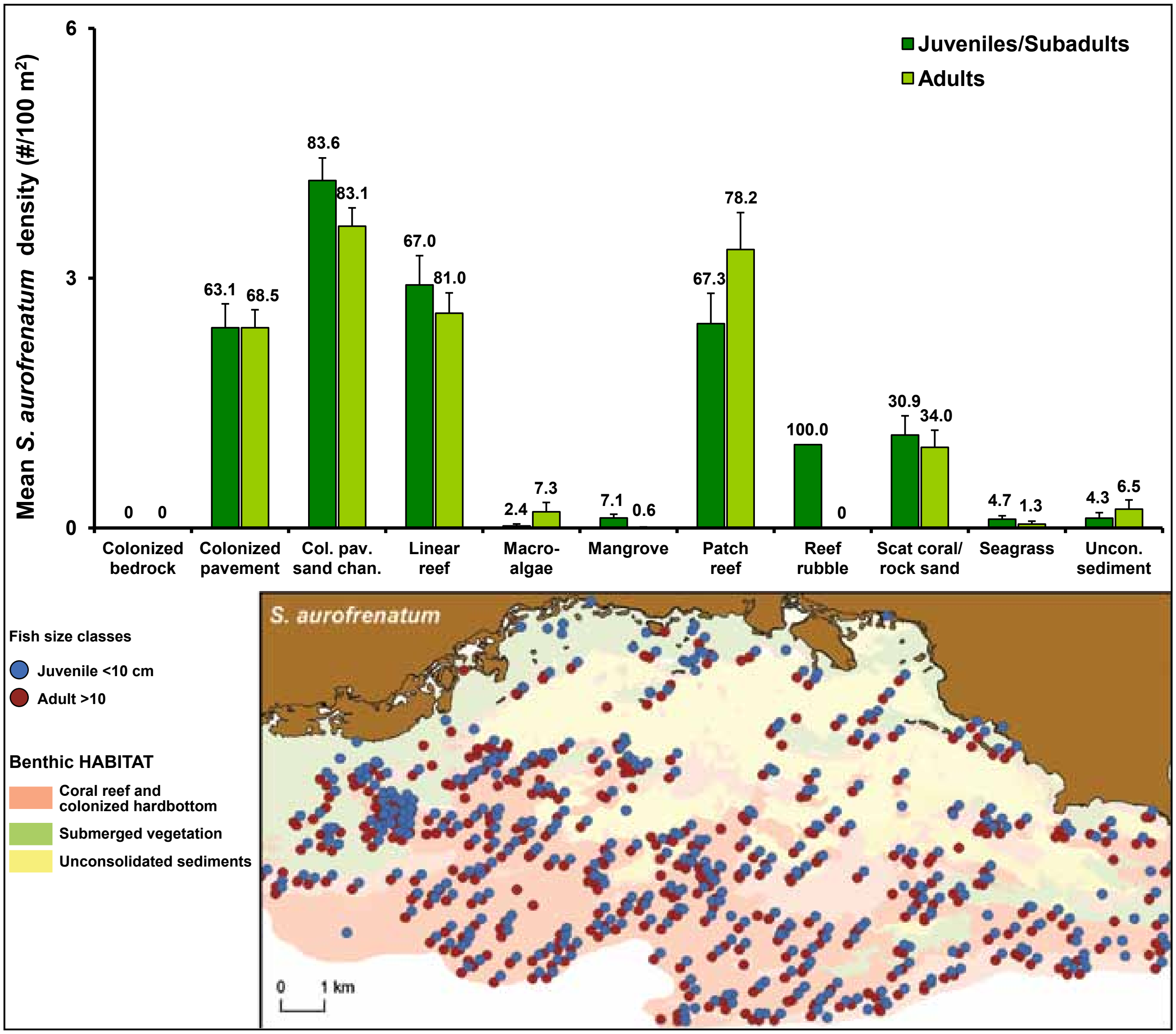


*Figure 3.53. Mean density (± SE) for juvenile/subadult and adult by mapped habitat type for redband parrotfish (*S. aurofrenatum*). Number above error bar indicates percent occurrence. Bottom: Spatial distribution of juvenile and adult for redband parrotfish (*S. aurofrenatum*) in the southwest Puerto Rico study area.*

Bucktooth parrotfish (*Sparisoma radians*)

S. radians is a widespread resident species in seagrass beds and is also found at the interface between seagrasses and mangroves and coral reefs and seagrasses (Figure 3.54). High abundance was recorded in the large continuous seagrass beds on the western side of the study area and on some surveys on the mid- and outer-shelf within patches of seagrasses surrounded by colonized hardbottom (Figure 3.55). Lowest mean abundance and biomass were recorded for colonized hardbottom and unvegetated sediments (Figure 3.55).

Bucktooth parrotfish (*Sparisoma radians*)

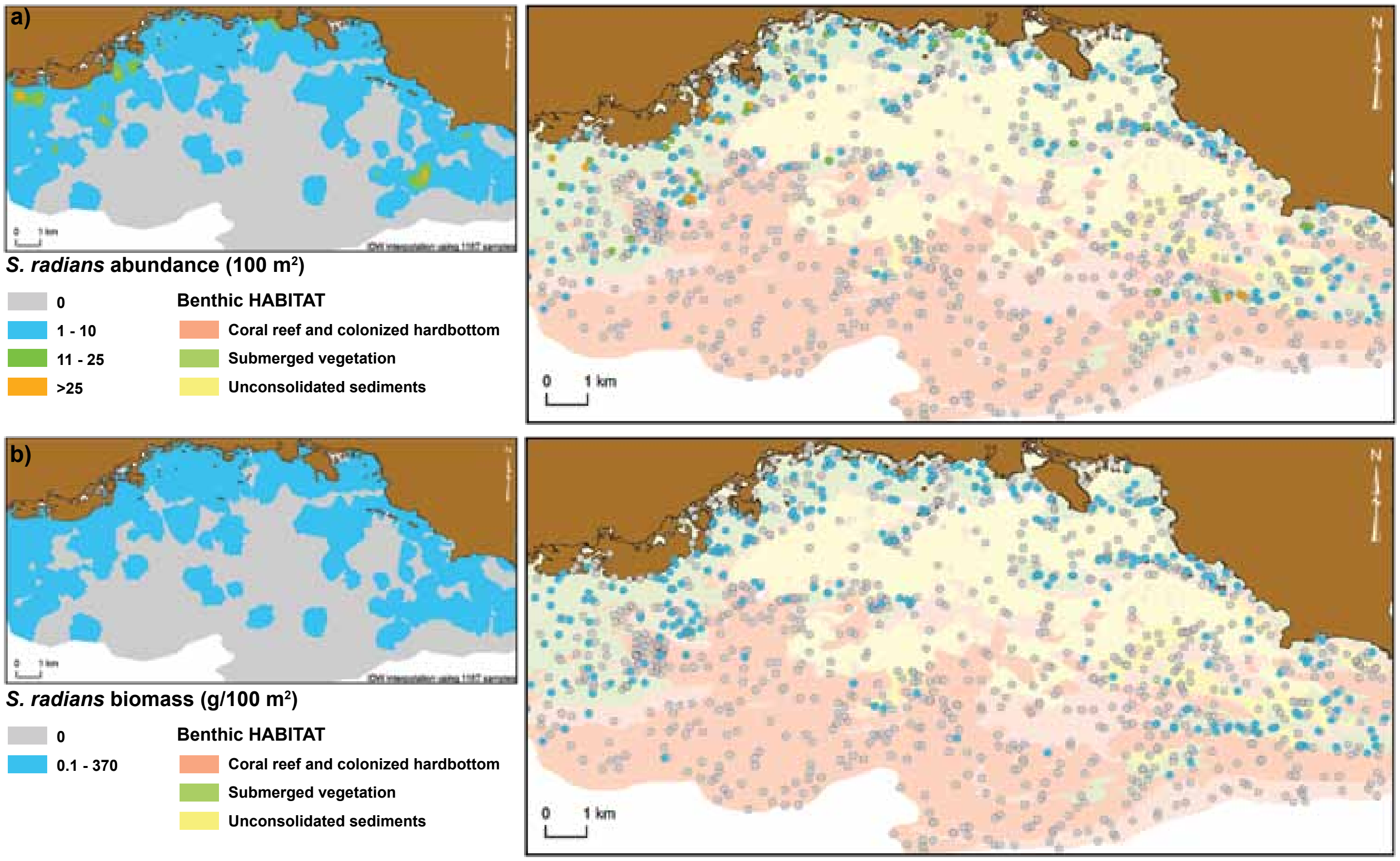


*Figure 3.54. Maps of the interpolated (left map) and spatial (right map) distributions for bucktooth parrotfish (*S. radians*): (a) abundance and (b) biomass.*

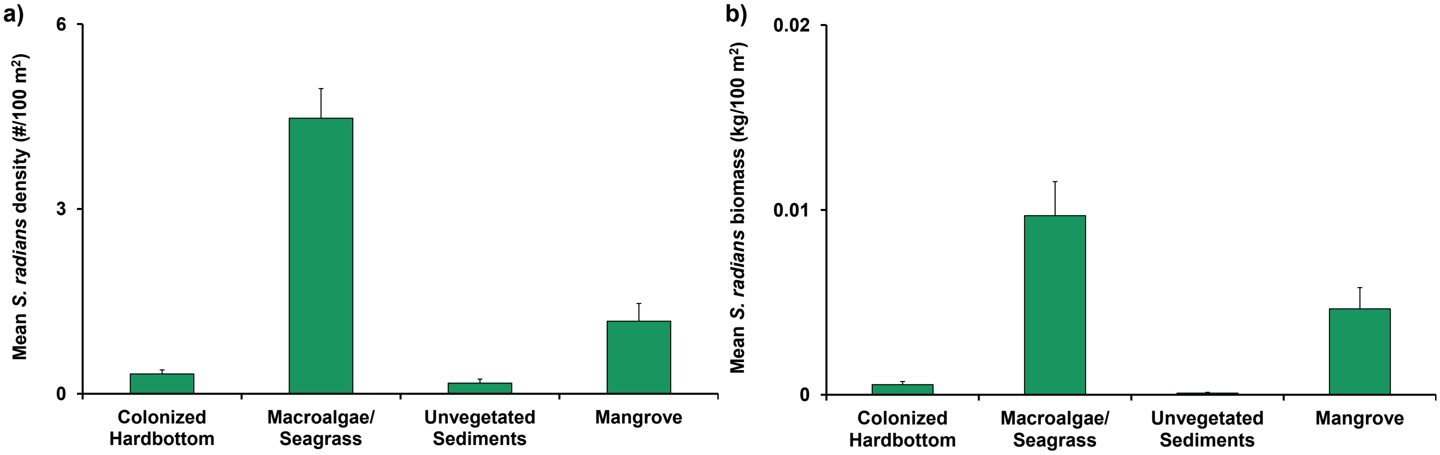


*Figure 3.55. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for bucktooth parrotfish (*S. radians*).*

Yellowtail parrotfish (*Sparisoma rubripinne*)

S. rubripinne was sighted in all major habitat types, but was more frequently sighted in colonized hardbottom habitat types than any others (Figure 3.56). The majority of *S. rubripinne* were from nearshore and mid-shelf zones (Figure 3.57).

Terminal phase yellowtail parrotfish (*Sparisoma rubripinne*) in St. Croix, USVI.

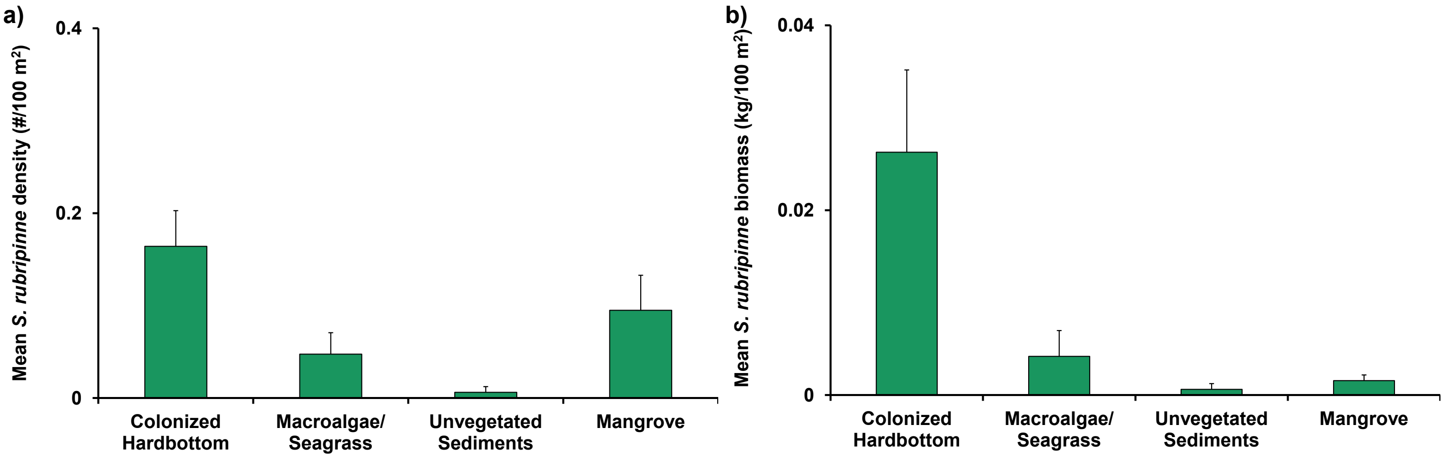


*Figure 3.56. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for yellowtail parrotfish (*S. rubripinne*).*

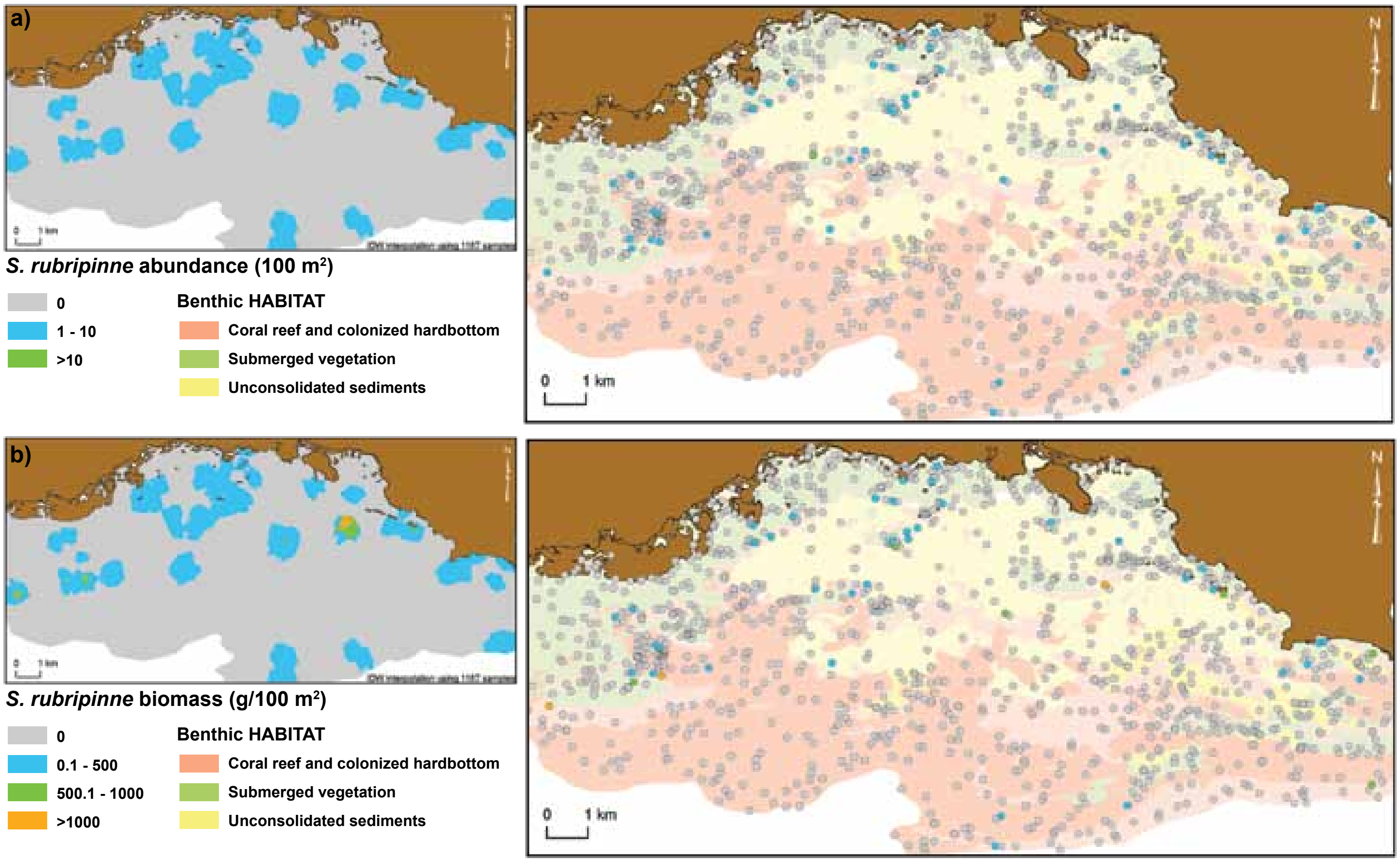
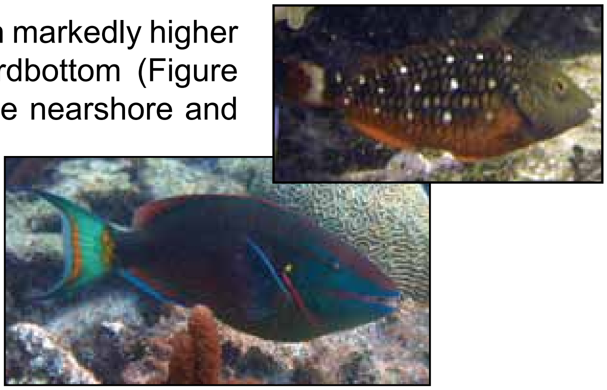


*Figure 3.57. Maps of the interpolated (left) and spatial (right) distributions for yellowtail parrotfish (*S. rubripinne*): (a) abundance and (b) biomass.*

Stoplight parrotfish (*Sparisoma viride*)

S. viride was associated with all major habitat types, but with markedly higher mean abundance and mean biomass over colonized hardbottom (Figure 3.59). Mid- and outer-shelf zones were used more than the nearshore and lagoonal zone (Figure 3.58). High biomass was recorded for the high rugosity coral reefs in the vicinity of El Palo and Margarita reefs and several of the nearshore reefs on the northeastern side of the study area (Figure 3.58b).



Terminal (bottom; St. Croix) and initial (top; PR) phase *S. viride*

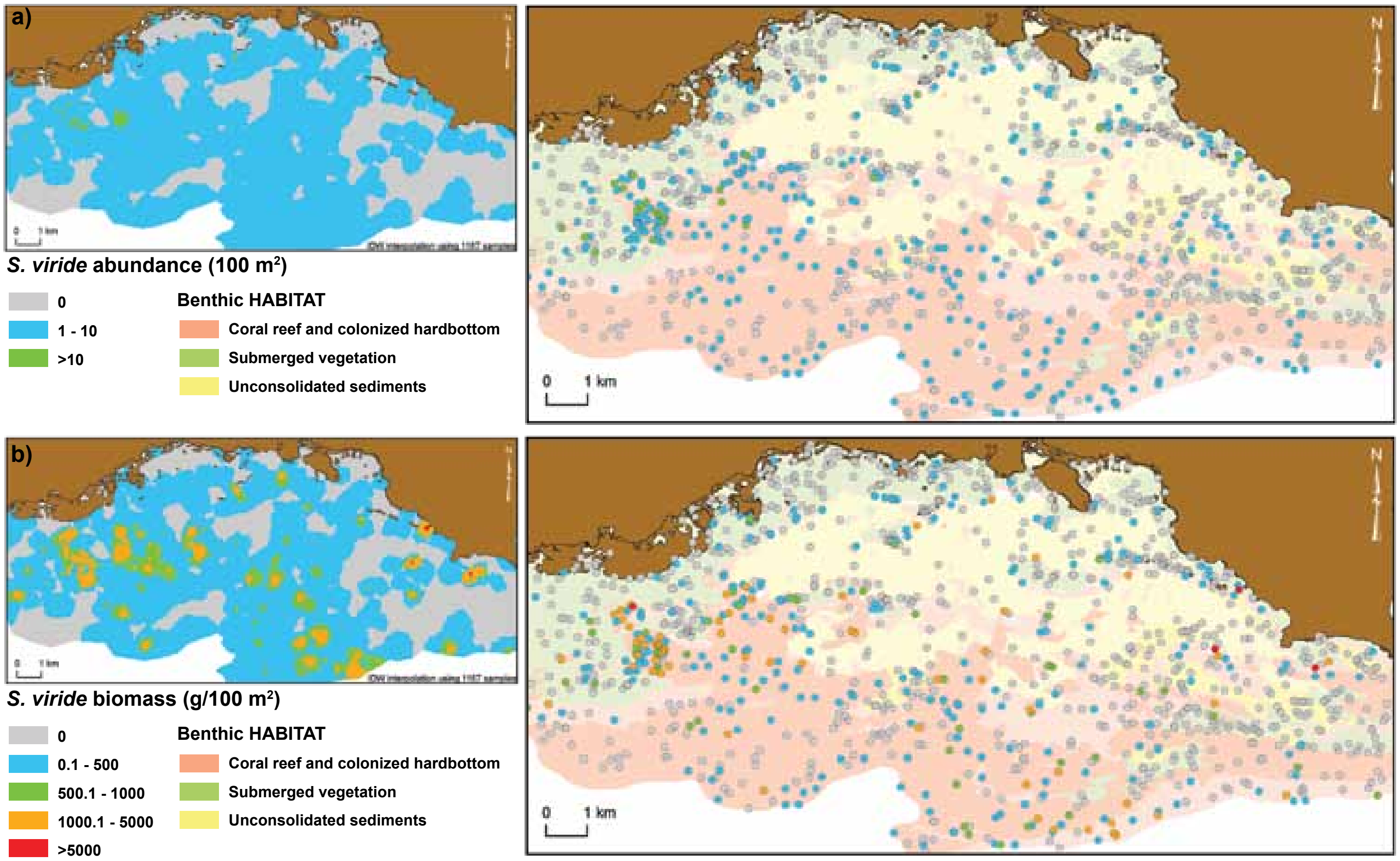


*Figure 3.58. Maps of the interpolated (left map) and spatial (right map) distributions for stoplight parrotfish (*S. viride*): (a) abundance and (b) biomass.*

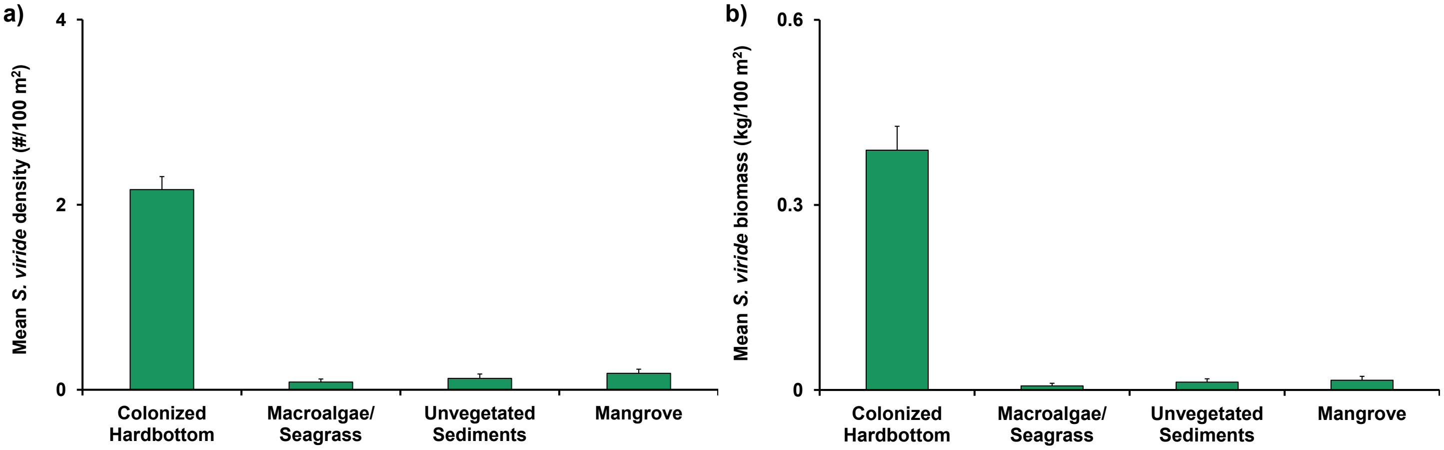


*Figure 3.59. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for stoplight parrotfish (*S. viride*).*

Like other common parrotfish species, life stages of *S. viride* co-occurred across the study area showing no recognizable spatial segregation (Figure 3.60). Both juveniles and adults were found in all of the most abundant habitat types, with highest densities over colonized hardbottom including patch reefs, colonized pavement with sand channels, colonized pavement and linear reefs (Figure 3.60). Lowest densities were recorded for unconsolidated sediments, seagrasses and mangroves. Both juveniles and adults were recorded in low abundance from seagrasses and mangroves. Similar to other Scaridae species, presence across the study area was higher in the most topographically complex hardbottom areas (Figure 3.60).

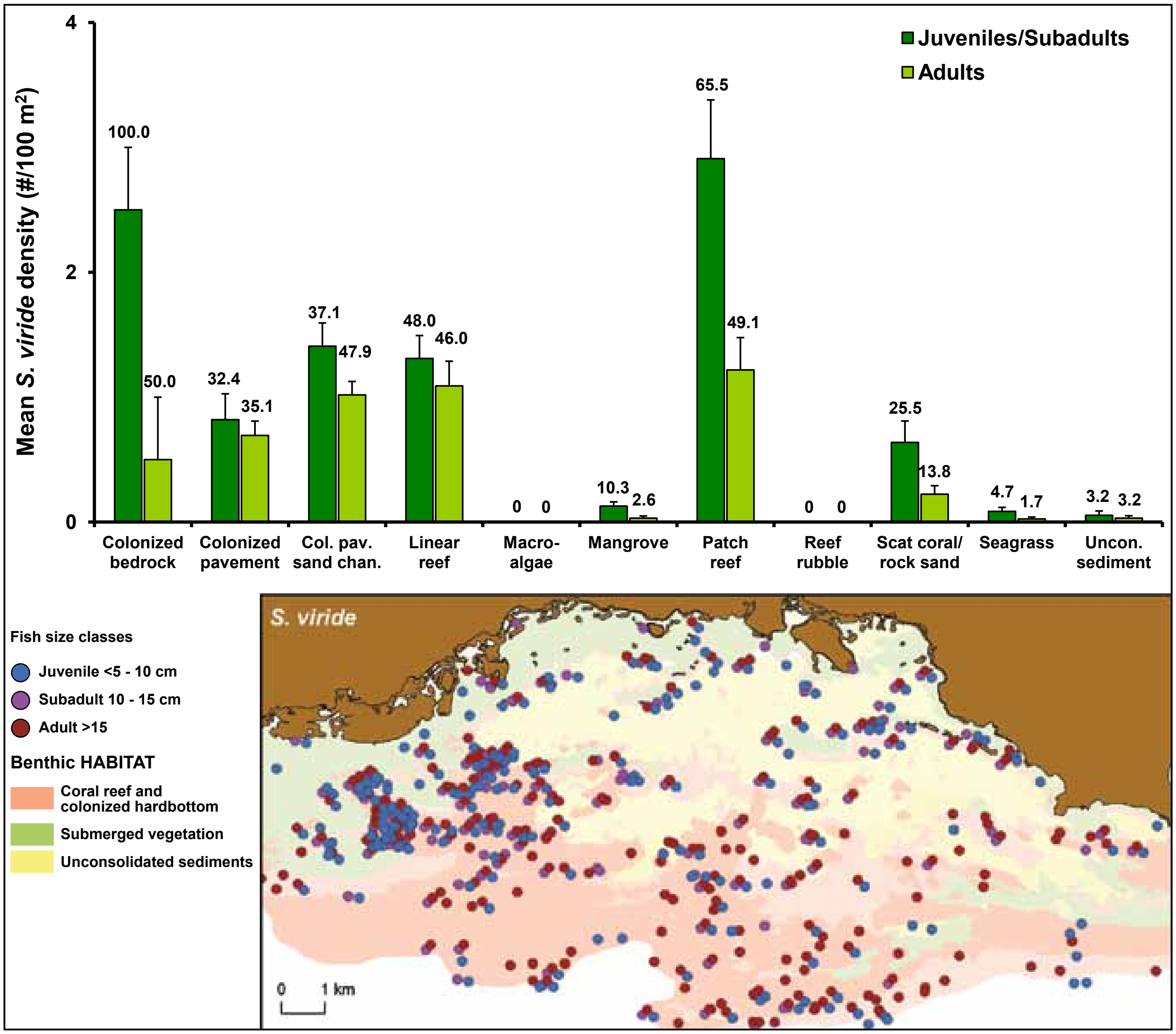


*Figure 3.60. Mean density (± SE) for juvenile/subadult and adult by mapped habitat type for stoplight parrotfish (*S. viride*). Number above error bar indicates percent occurrence. Bottom: Spatial distribution of juvenile, subadult and adult for stoplight parrotfish (*S. viride*) in the southwest Puerto Rico study area.*

Grunts (Haemulidae)

Assemblage of tomtates (*Haemulon aurolineatum*).

Haemulidae abundance was highest in the nearshore and midshelf mangroves, particularly around the mangrove islands in close proximity to coral reefs and seagrass beds and on shallow water patch reefs surrounded by seagrasses in the lagoonal and backreef zones (Figures 3.61a and 3.62a). Haemulidae abundance was also high in several locations over colonized pavement with sand channels in the outer shelf zone. Highest mean Haemulidae biomass was also found in mangroves (Figure 3.62b), with highest biomass calculated for several sites along the fringing mangroves and mangrove islands and associated backreef zones on the midshelf zone (Figure 3.61b). Several sites on colonized hardbottom sites (colonized pavement with sand channels and patch reefs) in close proximity to seagrasses and sand patches also exhibited high haemulidae biomass.

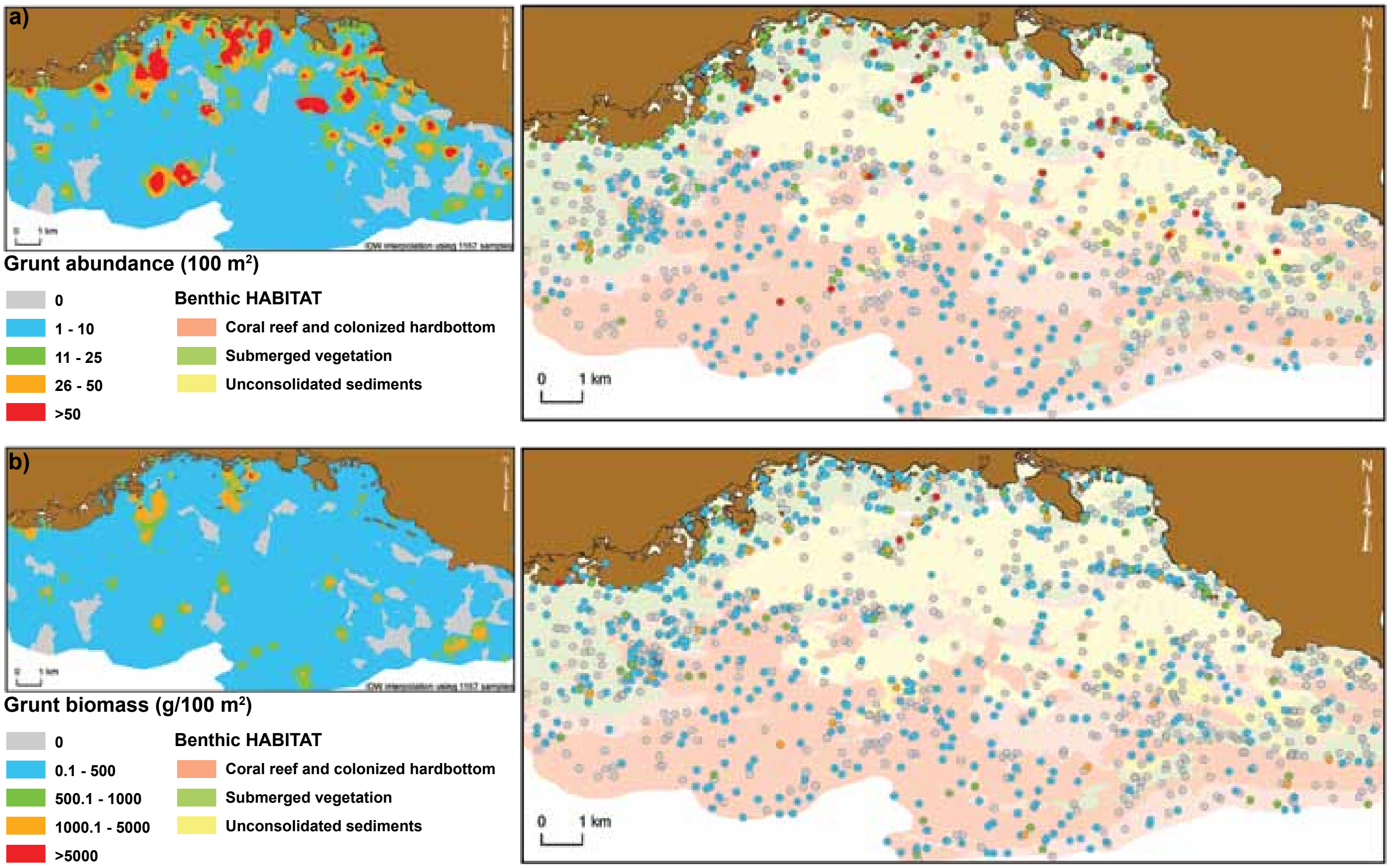


Figure 3.61. Maps of the interpolated (left map) and spatial (right map) distributions for grunt (Haemulidae): (a) abundance and (b) biomass.

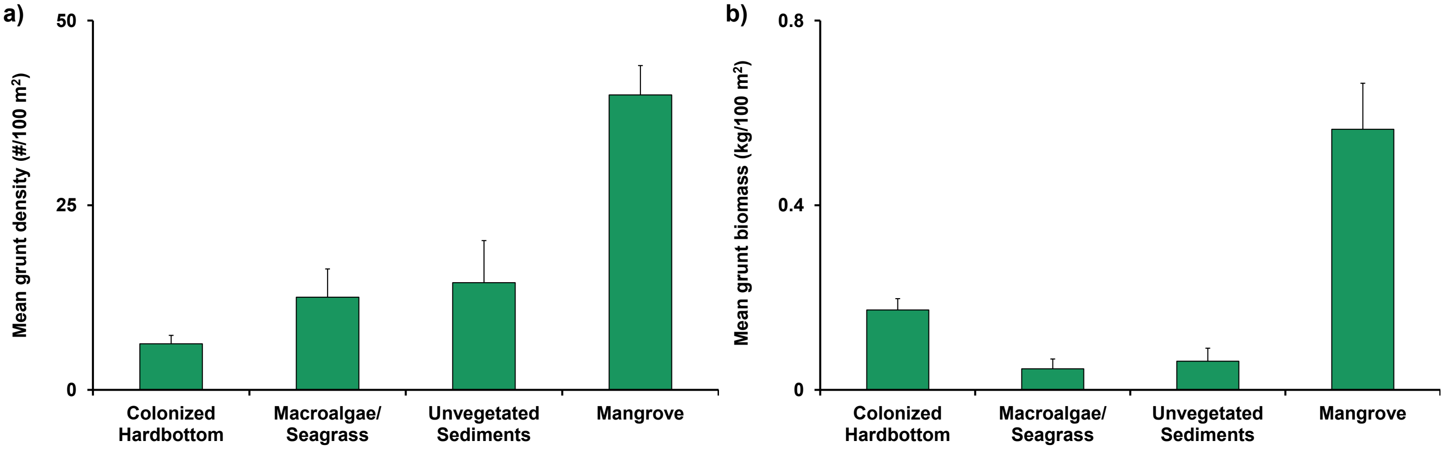


Figure 3.62. Comparison of mean (±̲ SE) density and biomass by habitat type in the southwest Puerto Rico study area for grunt species (Haemulidae).

The largest proportion of haemulids in the study area were represented by individuals of the smallest size class (<5 cm FL; Figure 3.63). This is likely to be an underestimate since some observers classify small grunts as unknown haemulids due to the difficulty of identifying the juveniles to species level. Approximately 95% of *Haemulon aurolineatum* (tomtate) and *Haemulon flavolineatum* (French grunt) were juveniles and subadults (Figures 3.63a and 3.63b) and approximately 85% of *Haemulon sciurus* (bluestriped grunt) and 80% of *Haemulon plumierii* (white grunt; Figures 3.63d and 3.63c). No individuals greater than 20 cm FL were recorded for *H. flavolineatum*; 25 cm FL for *H. aurolineatum*; 30 cm FL for *H. sciurus* and 30 cm FL for *H. plumierii* (Figure 3.63). None of the common haemulid individuals reached the maximum known for the species (Table 3.4).

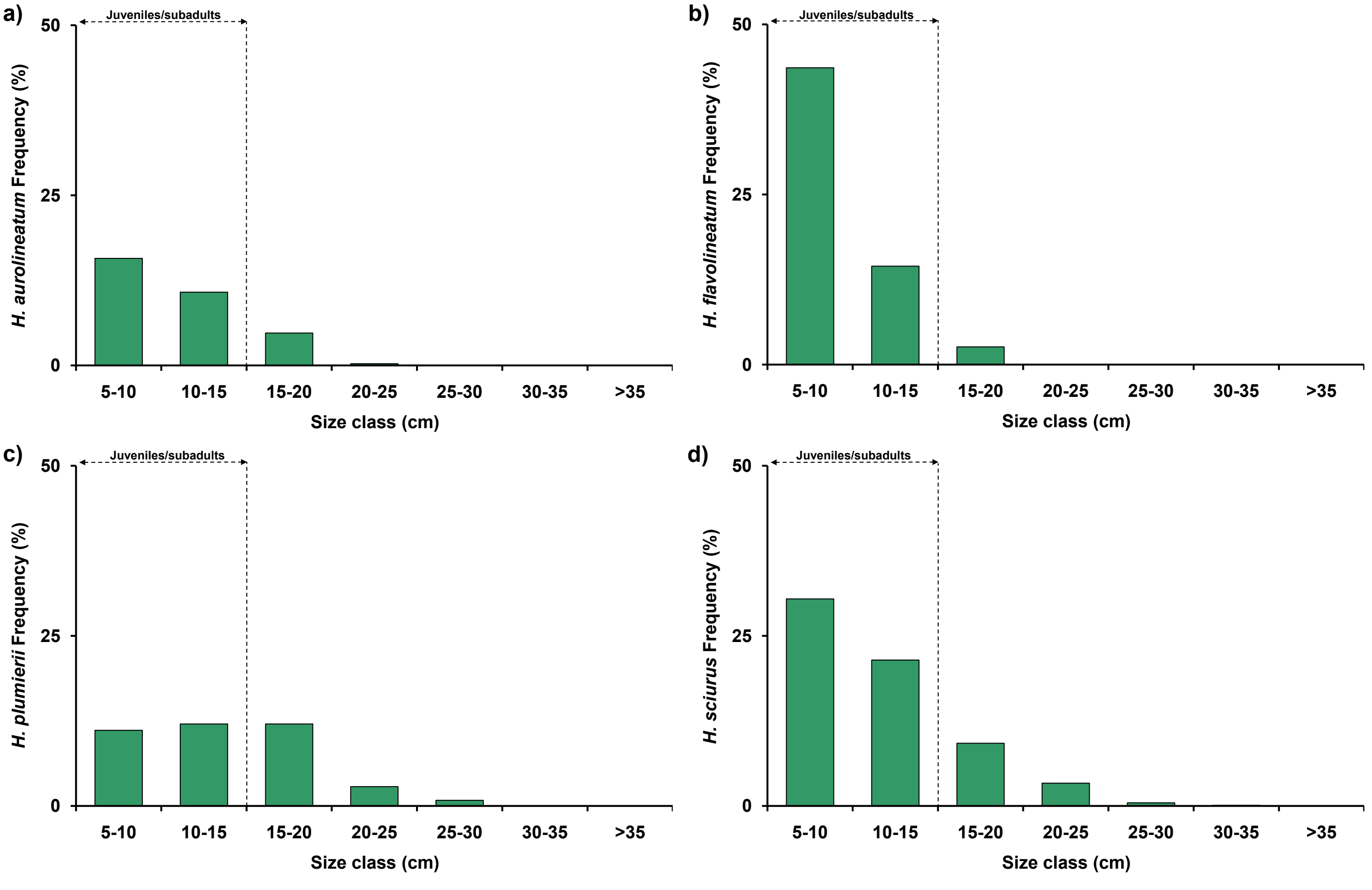


*Figure 3.63. Size frequency histogram for select grunts (Haemulidae) in the southwest Puerto Rico study area. (a) tomtate (*H. aurolineatum*), (b) French grunt (*H. flavolineatum*), (c) white grunt (*H. plumierii*) and (d) bluestriped grunt (*H. sciurus*).*

Tomtate (*Haemulon aurolineatum*)

H. aurolineatum utilized all major habitat types, with highest abundance recorded on nearshore patch reefs in close proximity to extensive softbottom habitat types (seagrasses and unvegetated sediments; Figure 3.64). Although the benthic habitat map indicates highest mean abundance and biomass over unvegetated sediments it is likely that some finer-scale unmapped hardbottom structure also occurred along these sand dominated sample sites (Figures 3.64 and 3.65). Abundance and biomass were lowest on colonized hardbottom areas of the outer shelf.

H. aurolineatum

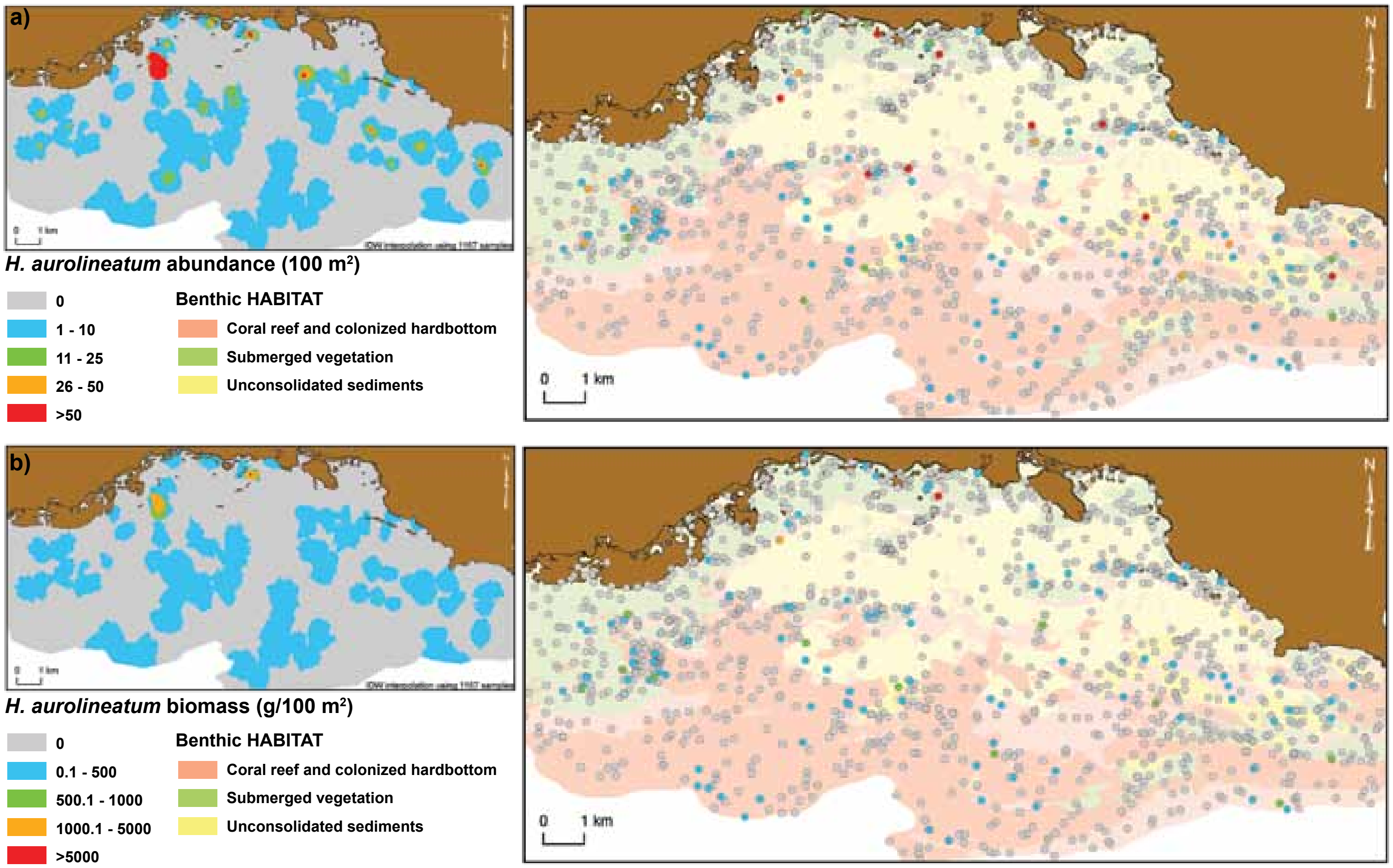


*Figure 3.64. Maps of the interpolated (left map) and spatial (right map) distributions for tomtate (*H. aurolineatum*): (a) abundance and (b) biomass.*

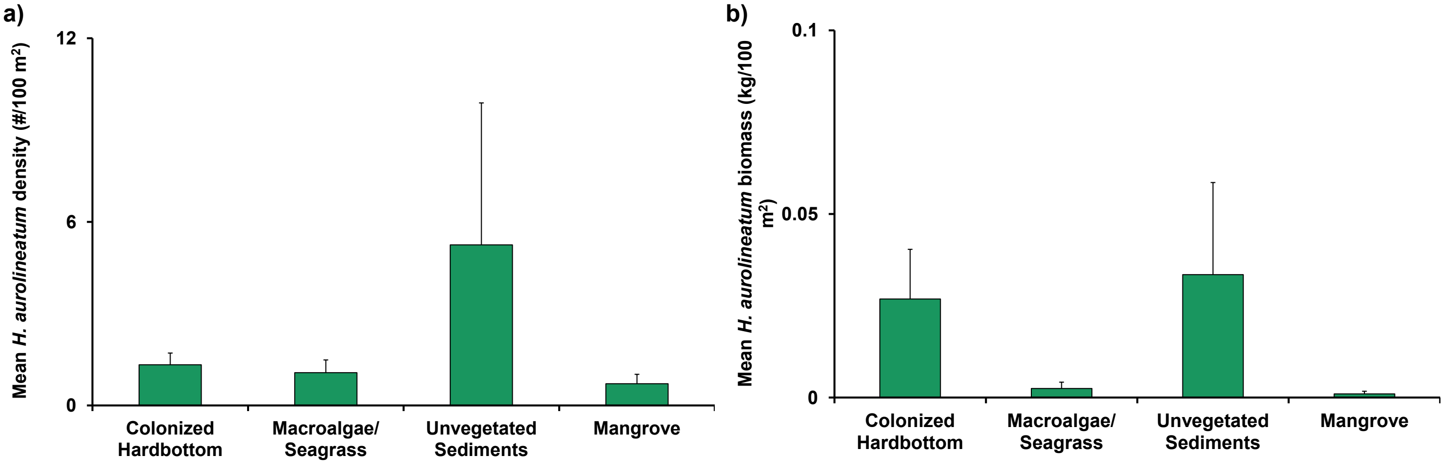


*Figure 3.65. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for tomtate (*H. aurolineatum*).*

H. aurolineatum juveniles, subadults and adults showed a less distinct pattern of spatial segregation across the shelf compared to other Haemulid species, although similarly, few adults were found amongst mangroves and in shallow nearshore vegetated habitat types and few juveniles were found on the outer shelf (Figure 3.66). Although high geographical overlap across lifestages was observed, highest mean densities of juveniles/subadults were recorded over scattered coral in sand, macroalgal beds, patch reefs and linear reefs, seagrasses and mangroves of the mid and nearshore zones, while highest densities of adults were recorded over scattered coral and patch reefs in mid- and outer-shelf zones (Figure 3.66).

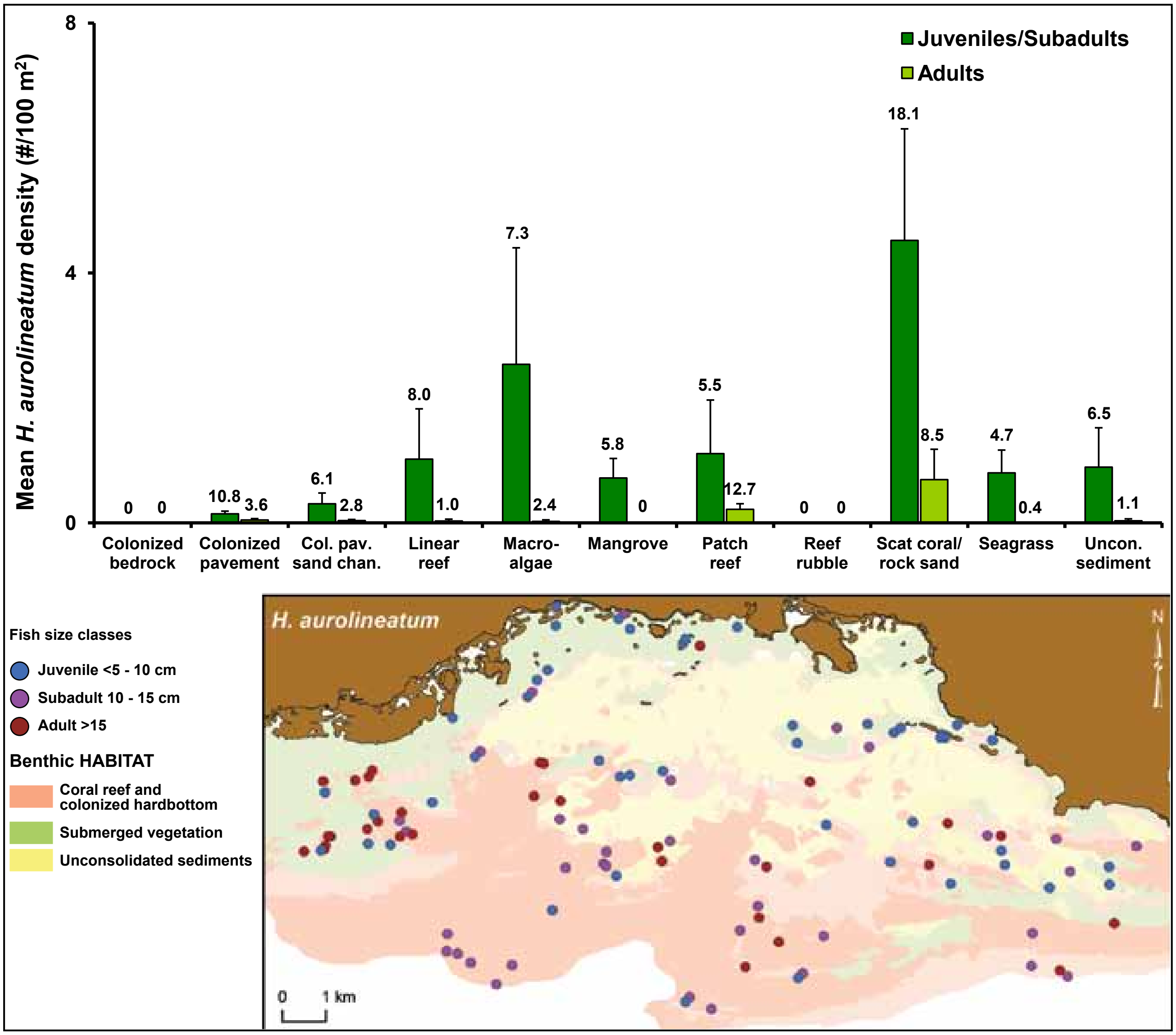
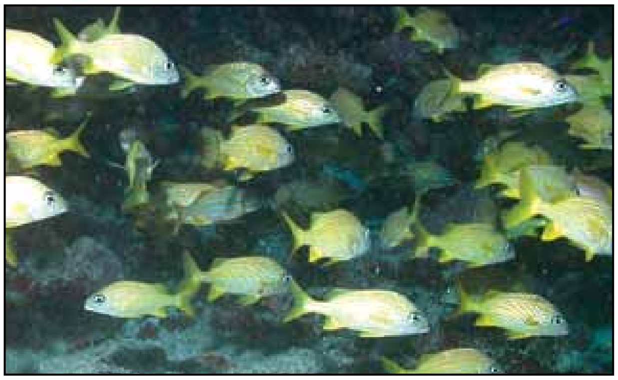


*Figure 3.66. Mean density (±̲ SE) for juvenile/subadult and adult by mapped habitat type for tomtate (*H. aurolineatum*). Number above error bar indicates percent occurrence. Bottom: Spatial distribution of juvenile, subadult and adult for tomtate (*H. aurolineatum*) in the southwest Puerto Rico study area.*

French grunt (*Haemulon flavolineatum*)

H. flavolineatum were widely distributed throughout the study area, with highest mean abundance and biomass in mangroves, particularly the mangrove cays around Isla Magueyes and Enrique reefs (Figure 3.67). Mean biomass was higher in colonized hardbottom than seagrasses and unvegetated sediments (Figure 3.68b).



French grunt (*Haemulon flavolineatum*)

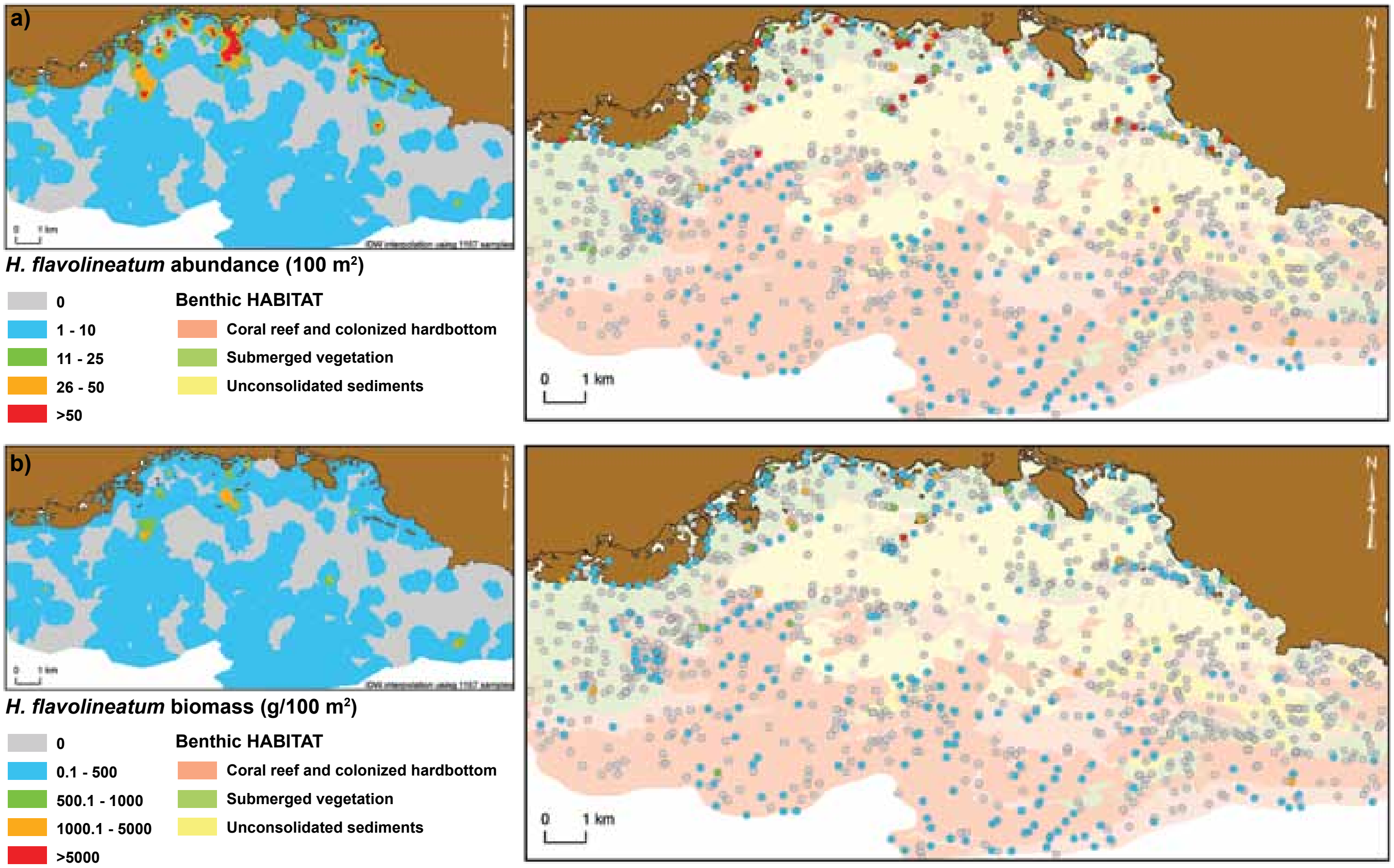


*Figure 3.67. Maps of the interpolated (left map) and spatial (right map) distributions for French grunt (*H. flavolineatum*): (a) abundance and (b) biomass.*

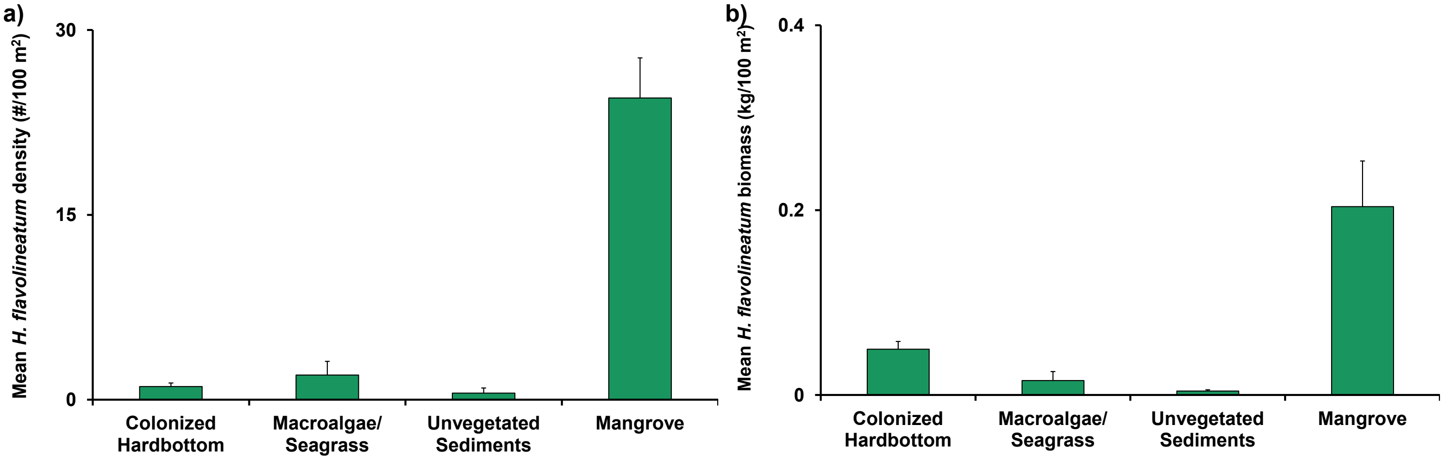


*Figure 3.68. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for French grunt (*H. flavolineatum*).*

H. flavolineatum juveniles were distributed primarily in fringing mangroves and mangrove cays, but were also observed co-occurring with adults and subadults at several hardbottom sites on the mid and outer shelf (Figure 3.69). Subadult, and particularly adults, occurred in lower densities than juveniles overall and exhibited a distinctly more offshore distribution, with highest densities on patch reefs, colonized pavement with sand channels and linear coral reefs. Subadults, and occasionally adults, were also observed using inshore habitat types (Figure 3.69).

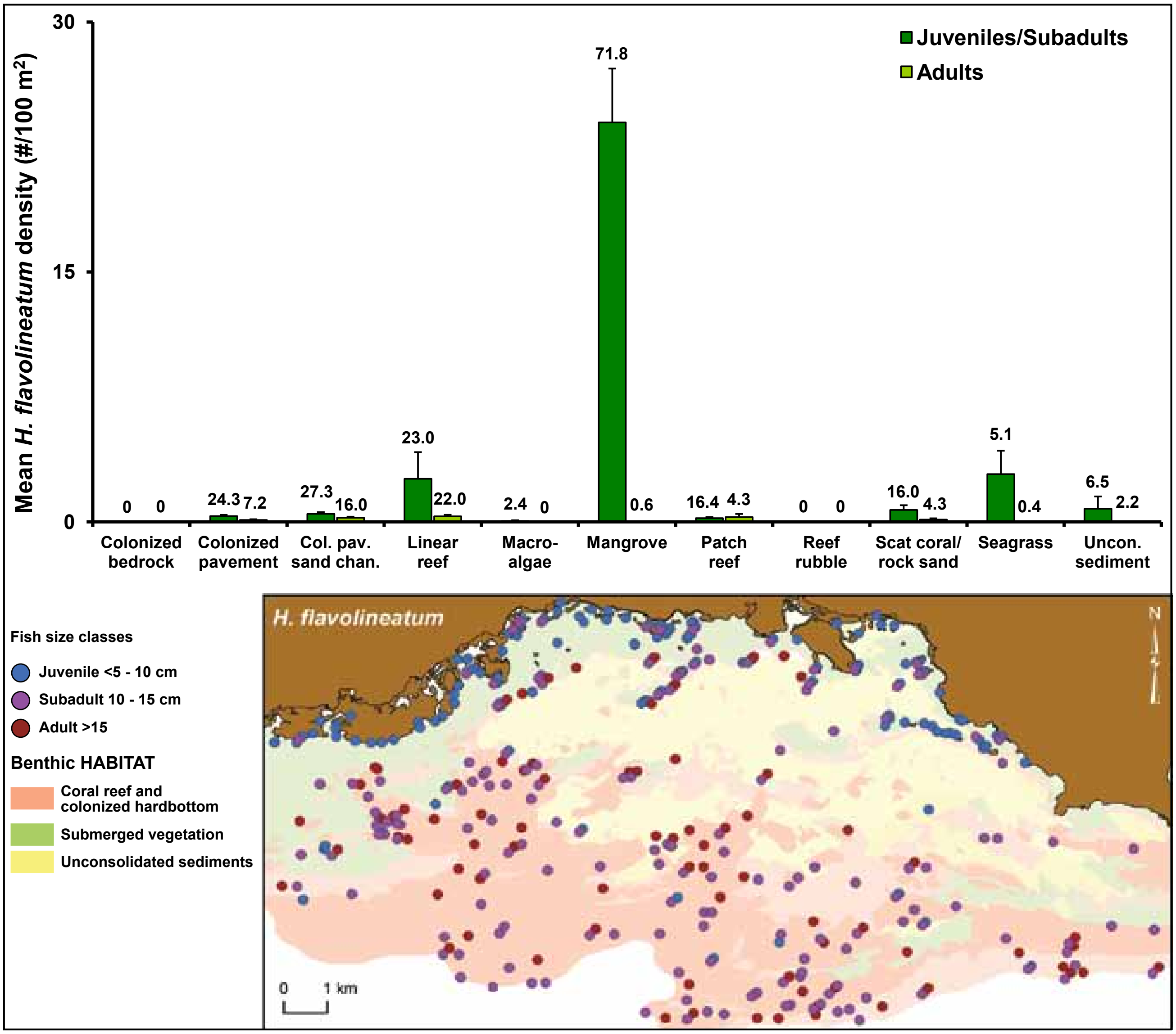


*Figure 3.69. Mean density (+ SE) for juvenile/subadult and adult by mapped habitat type for French grunt (*H. flavolineatum*). Number above error bar indicates percent occurrence. Bottom: Spatial distribution of juvenile, subadult and adult for French grunt (*H. flavolineatum*) in the southwest Puerto Rico study area.*

White grunt (*Haemulon plumierii*)

H. plumierii were widely distributed in the nearshore lagoonal zone and across all major habitat types (Figure 3.70). Mean density was highest in neashore seagrasses and mangroves and mean biomass was highest in colonized hardbottom habitat types (Figures 3.70 and 3.71). Several high biomass sites existed over colonized hardbottom areas along the mid to outer shelf zone.

White grunt (*Haemulon plumierii*)

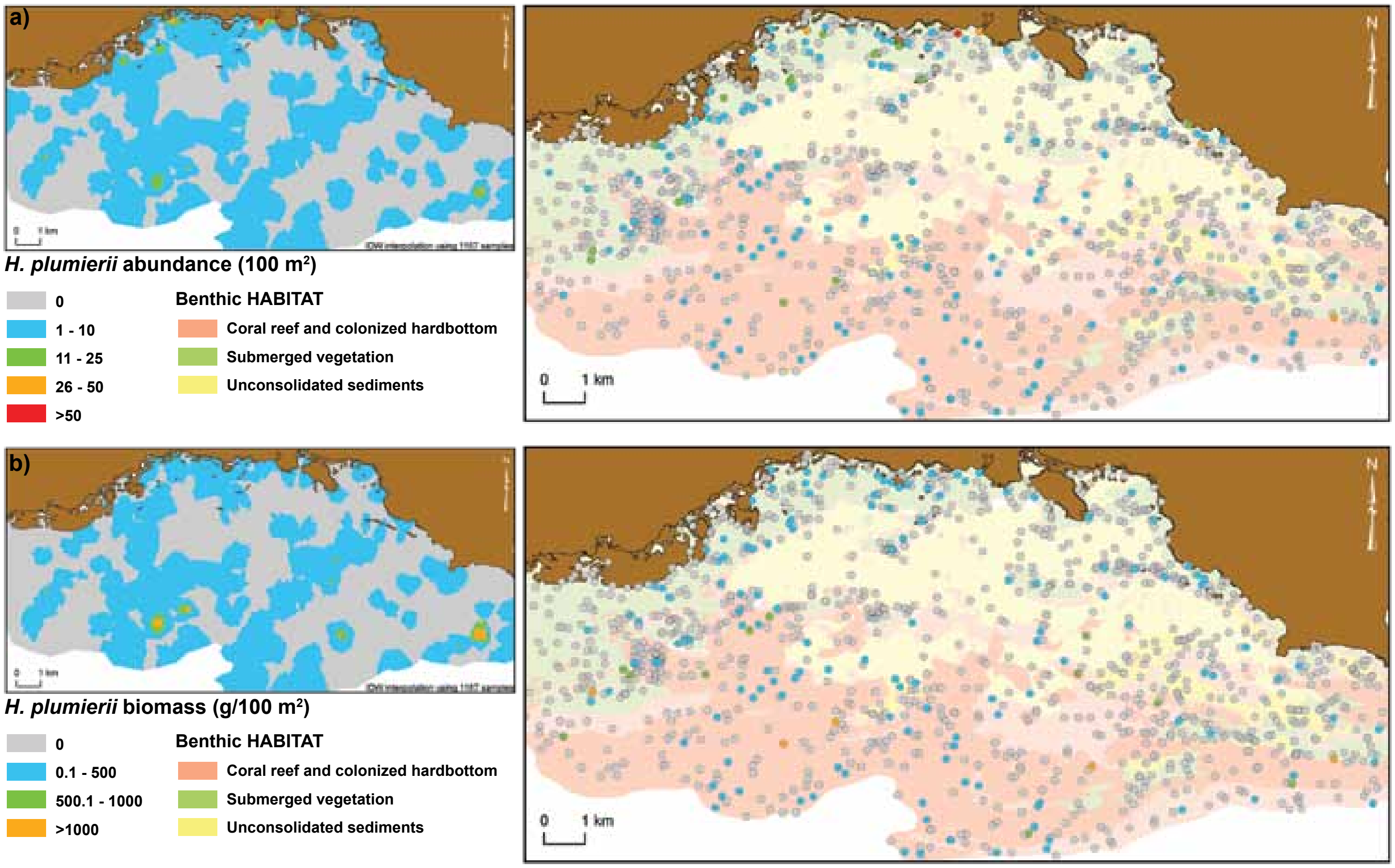


*Figure 3.70. Maps of the interpolated (left map) and spatial (right map) distributions for white grunt (*H. plumierii*): (a) abundance and (b) biomass.*

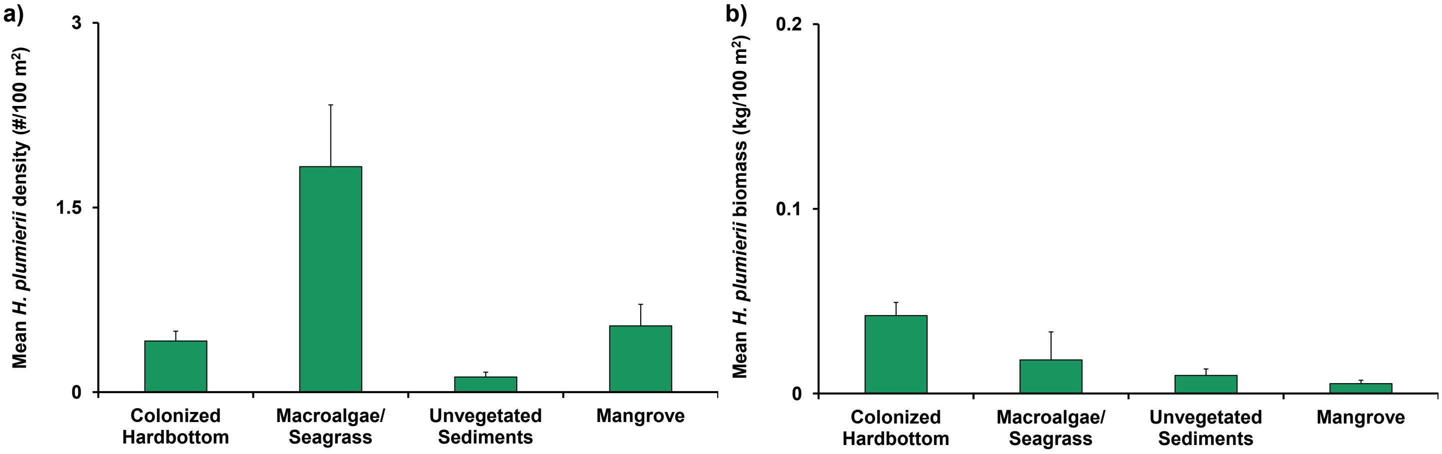


*Figure 3.71. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for white grunt (*H. plumierii*).*

H. plumierii juveniles were more frequently distributed in mangroves and seagrasses in lagoonal and nearshore zones (Figure 3.72), but like *H. flavolineatum*, juvenile *H. plumierii* were also observed in mid and outer shelf zones usually on or very near to seagrasses (Figures 3.69 and 3.72). Subadults and adults also inhabited nearshore habitat types, but were more frequently observed over colonized hardbottom sites on the mid and outer shelf zones (Figure 3.72). Highest densities of juveniles/subadults were recorded for macroalgal and seagrass habitat types and highest densities for adults were recorded for patch reefs and colonized pavement with sand channels.

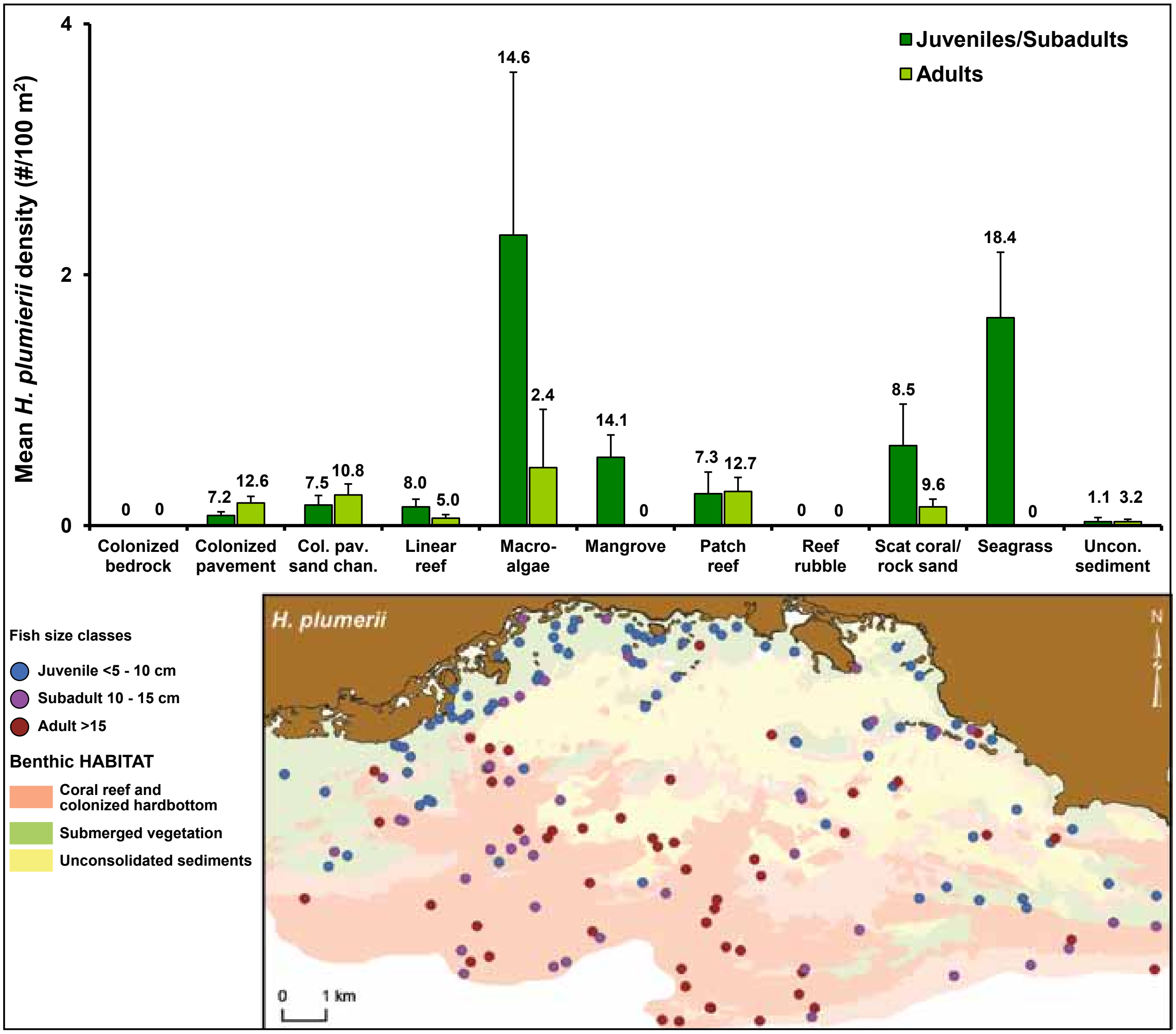


*Figure 3.72. Mean density (± SE) for juvenile/subadult and adult by mapped habitat type for white grunt (*H. plumierii*). Number above error bar indicates percent occurrence. Bottom: Spatial distribution of juvenile, subadult and adult for white grunt (*H. plumierii*) in the southwest Puerto Rico study area.*

Bluestriped grunt (*Haemulon sciurus*)

Although *H. sciurus* were observed in all major habitat types, both the abundance and biomass of *H. sciurus* was highest in the mangroves of the nearshore lagoonal zone and around many of the island cays, particularly on the western side of the study area (Figure 3.73). Lowest abundance was recorded for unvegetated sediments and lowest biomass for seagrasses and macroalgal beds (Figure 3.74).

Bluestriped grunt (*Haemulon sciurus*)

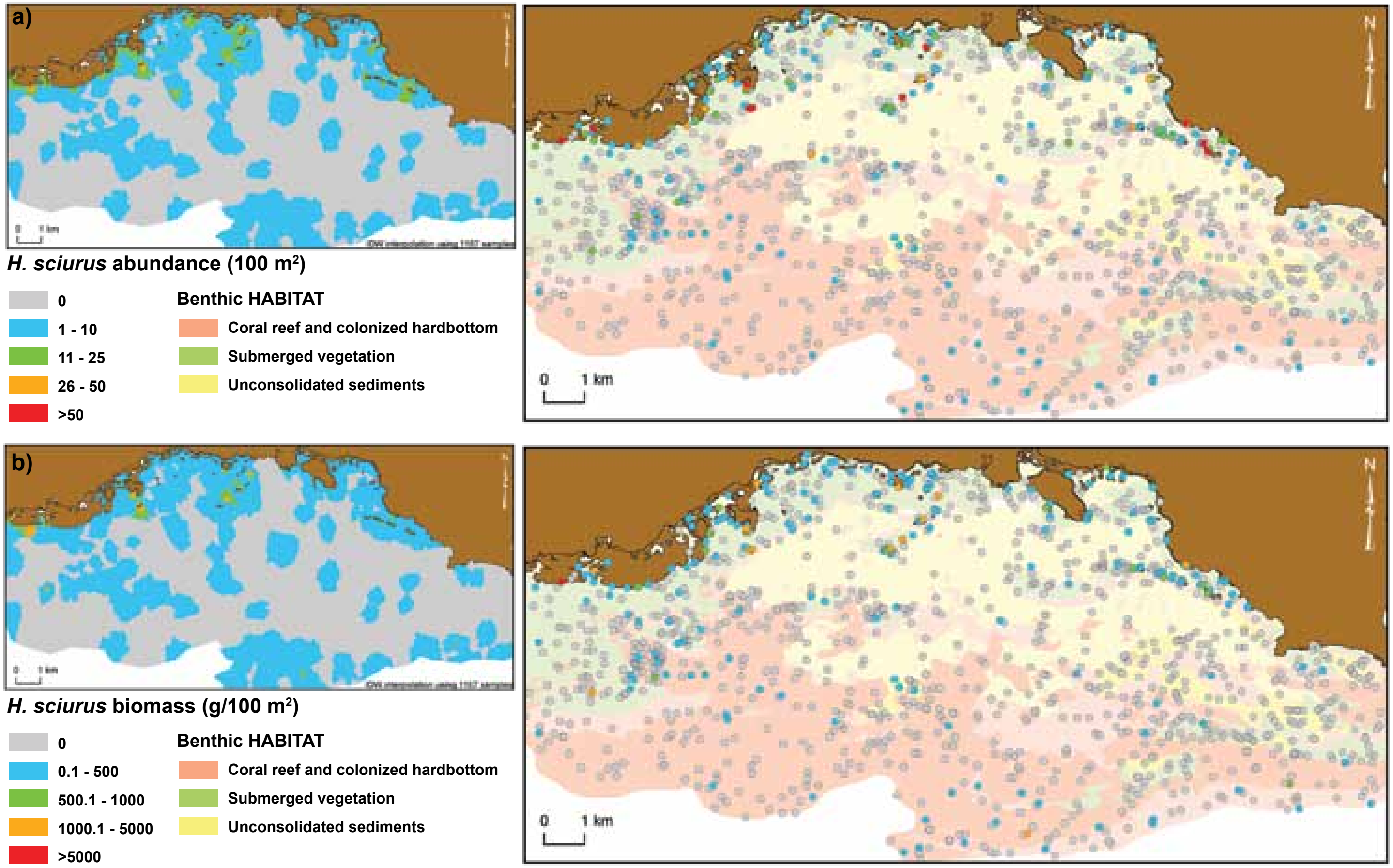


*Figure 3.73. Maps of the interpolated (left map) and spatial (right map) distributions for bluestriped grunt (*H. sciurus*): (a) abundance and (b) biomass.*

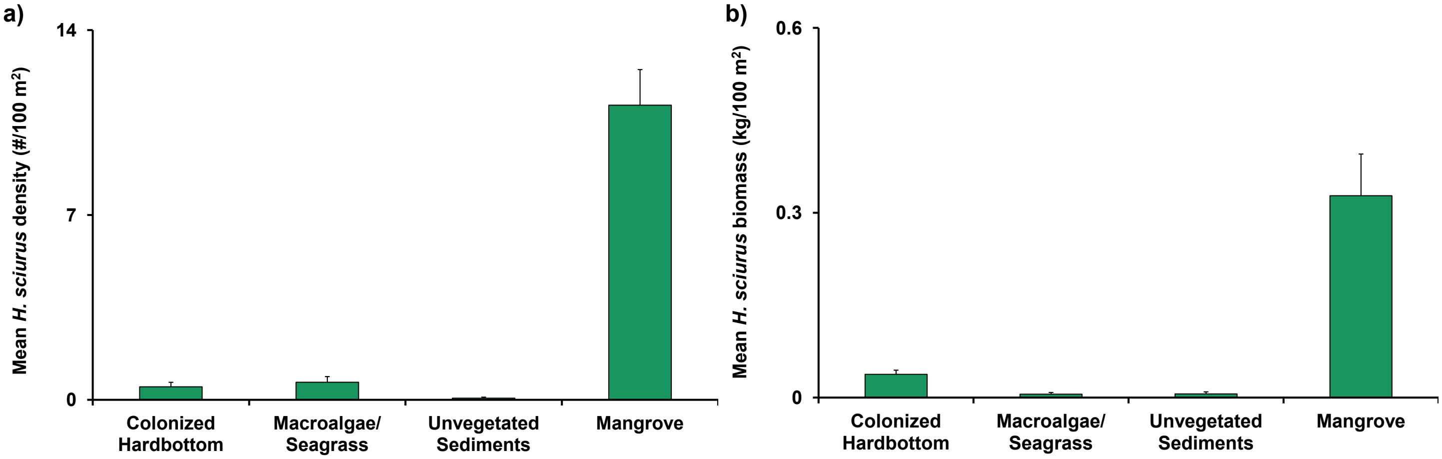


*Figure 3.74. Comparison of mean (+ SE) density and biomass by substrate type in the southwest Puerto Rico study area for bluestriped grunt (*H. sciurus*).*

H. sciurus juveniles were most frequently sighted in mangroves (fringing and cays), but also utilized several other habitat types including seagrasses, macroalgal beds, scattered coral in sand, patch reefs, linear reefs and colonized bedrock (Figure 3.75). In contrast to other haemulids, subadult *H. sciurus* were more frequently seen in nearshore and lagoonal habitat types than on the mid and outer shelf suggesting an offshore ontogenetic shift at a later stage than appears evident for other common haemulids (Figure 3.75). Subadults appeared to co-occur more often with juveniles (very occasionally with adults) and were frequently observed using mangroves, but adults were more frequently sighted on mid and outer shelf zones (Figure 3.75). Highest densities of all life stages were recorded for mangroves.

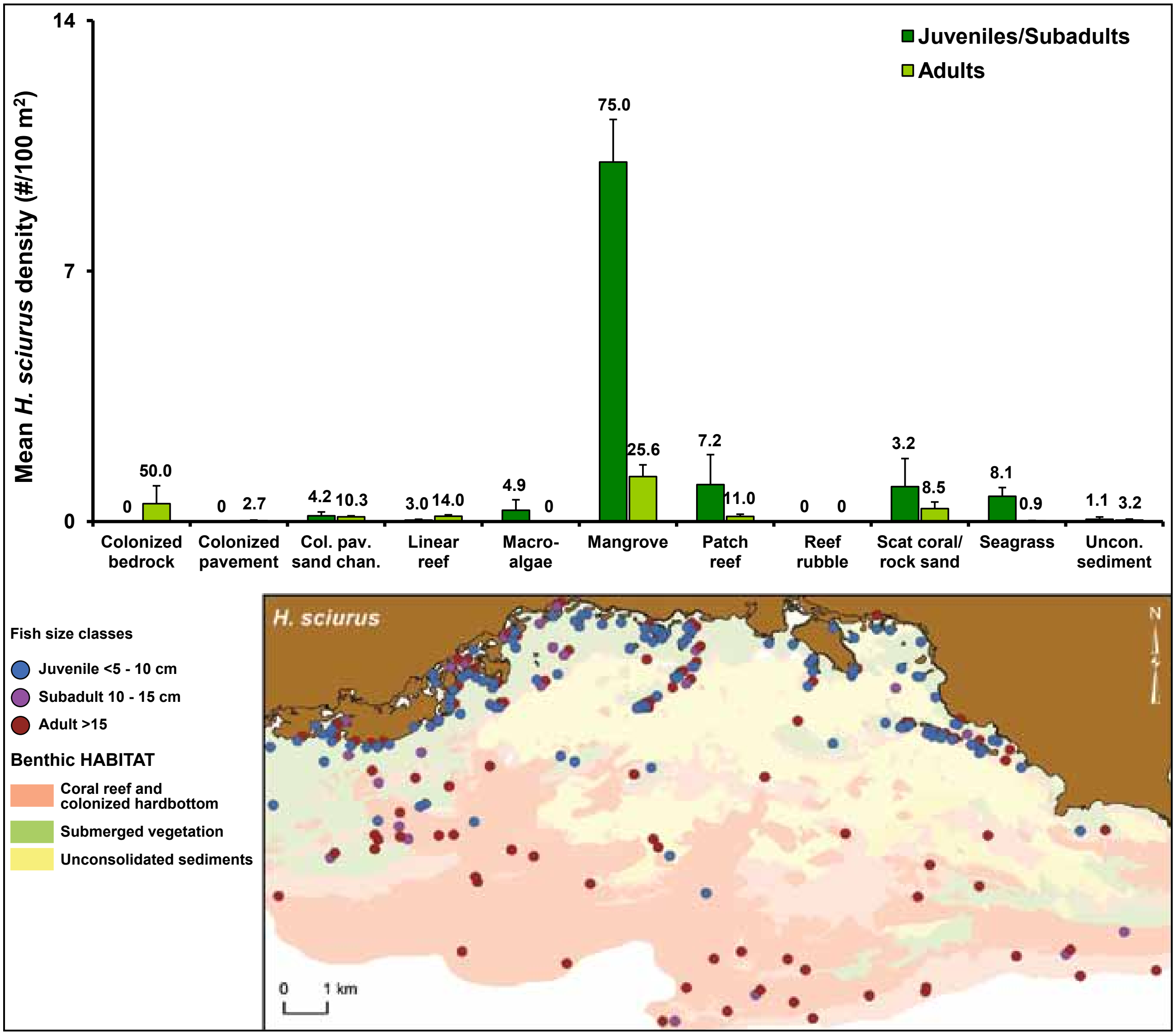


*Figure 3.75. Mean density (+ SE) for juvenile/subadult and adult by mapped habitat type for bluestriped grunt (*H. sciurus*). Number above error bar indicates percent occurrence. Bottom: Spatial distribution of juvenile, subadult and adult for bluestriped grunt (*H. sciurus*) in the southwest Puerto Rico study area.*

Surgeonfish (Acanthuridae)

Acanthurids, a major family of herbivorous fish, were markedly more abundant across colonized hardbottom habitat types, with abundance and biomass hotspots at several locations around El Palo and Margarita Reefs and across colonized hardbottom of the outer shelf (Figures 3.76 and 3.77). Lowest abundance and biomass were observed in the shallow lagoonal habitat types including mangroves closest to the shoreline (Figure 3.76).

Assemblage of blue tangs (*Acanthurus coeruleus*).

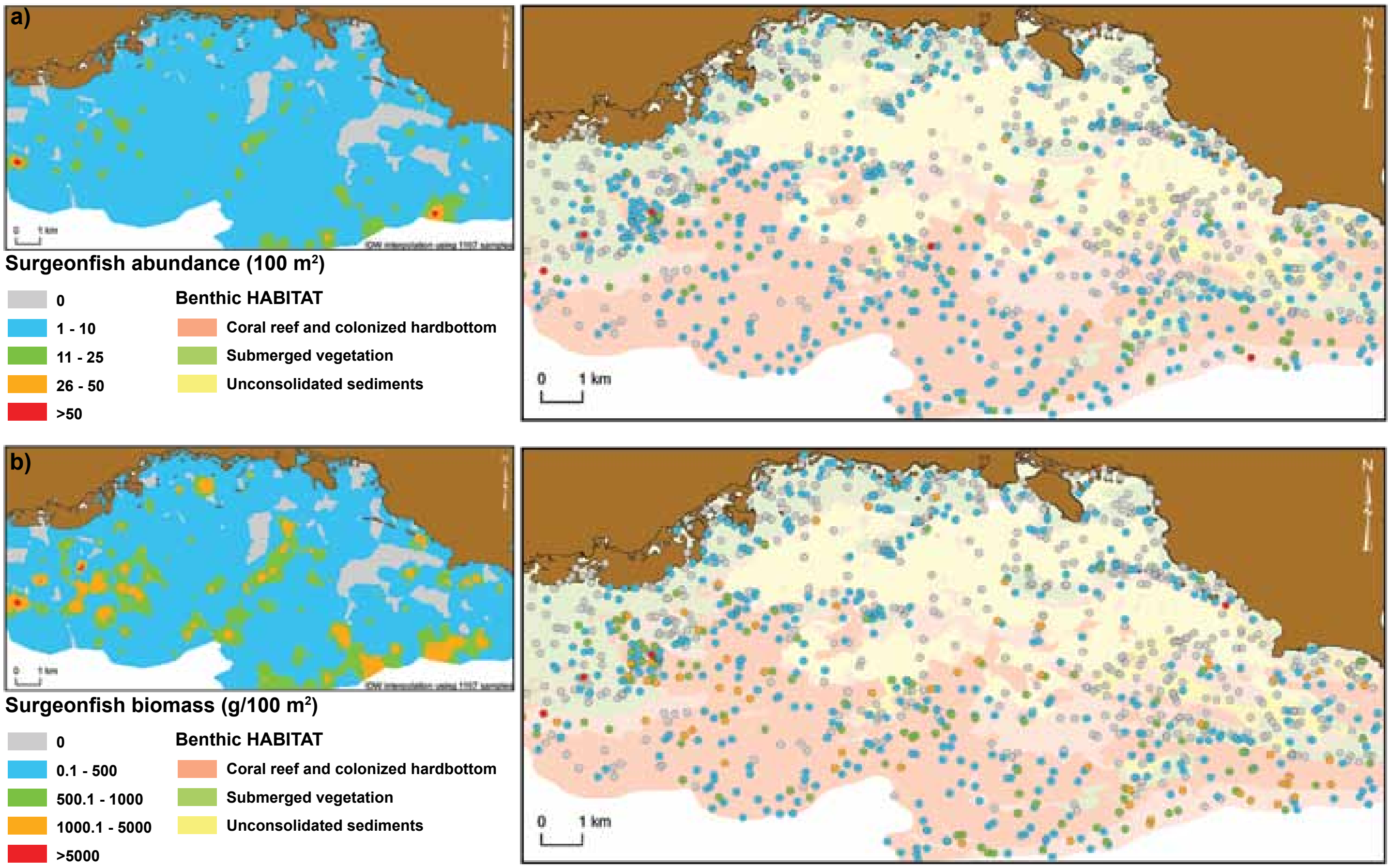


Figure 3.76. Maps of the interpolated (left map) and spatial (right map) distributions for surgeonfish (Acanthuridae): (a) abundance and (b) biomass.

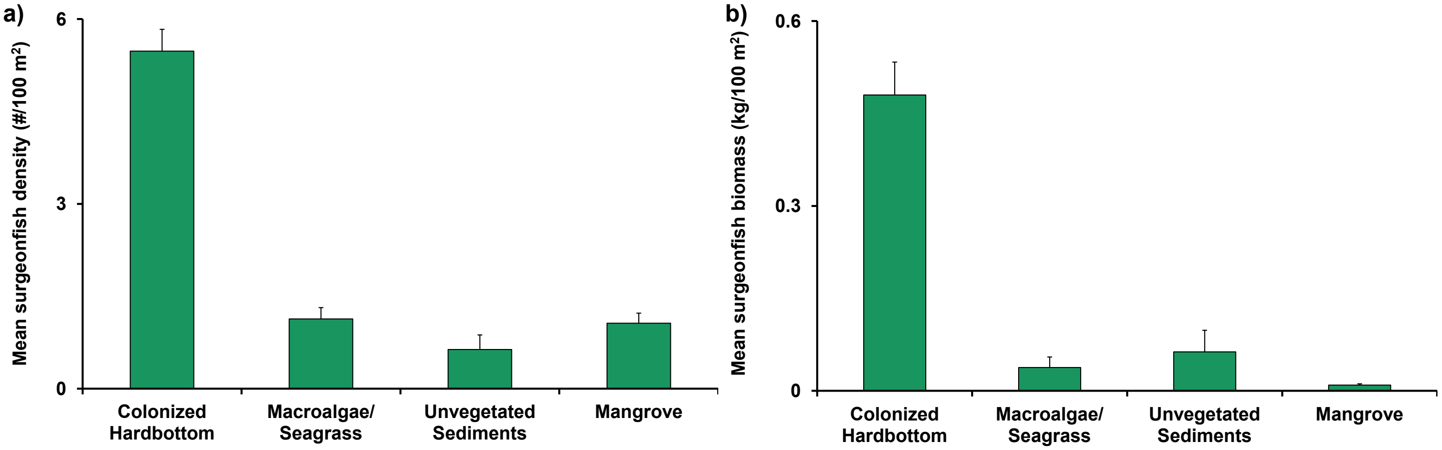


Figure 3.77. Comparison of mean (± SE) density and biomass by habitat type in the southwest Puerto Rico study area for surgeonfish (Acanthuridae).

Less than 35% of the surgeonfish population sampled in the La Parguera study area were classified as juveniles and subadults (≤10 cm; Figure 3.78). The majority of *Acanthurus bahianus* (ocean surgeon) and *Acanthurus coeruleus* (blue tang) were between 10 and 20 cm FL (Figures 3.78a and 3.78c) and between 10 and 25 cm for *Acanthurus chirurgus* (doctorfish; Figure 3.78b). Less than 5% of surgeonfish were larger than 25 cm FL. No individuals were seen above 25 cm FL for *A. bahianus*, 30 cm FL for *A. coeruleus* and 35 cm FL for *A. chirurgus* and the maximum sizes known for these species were not attained in the study area (Table 3.4).

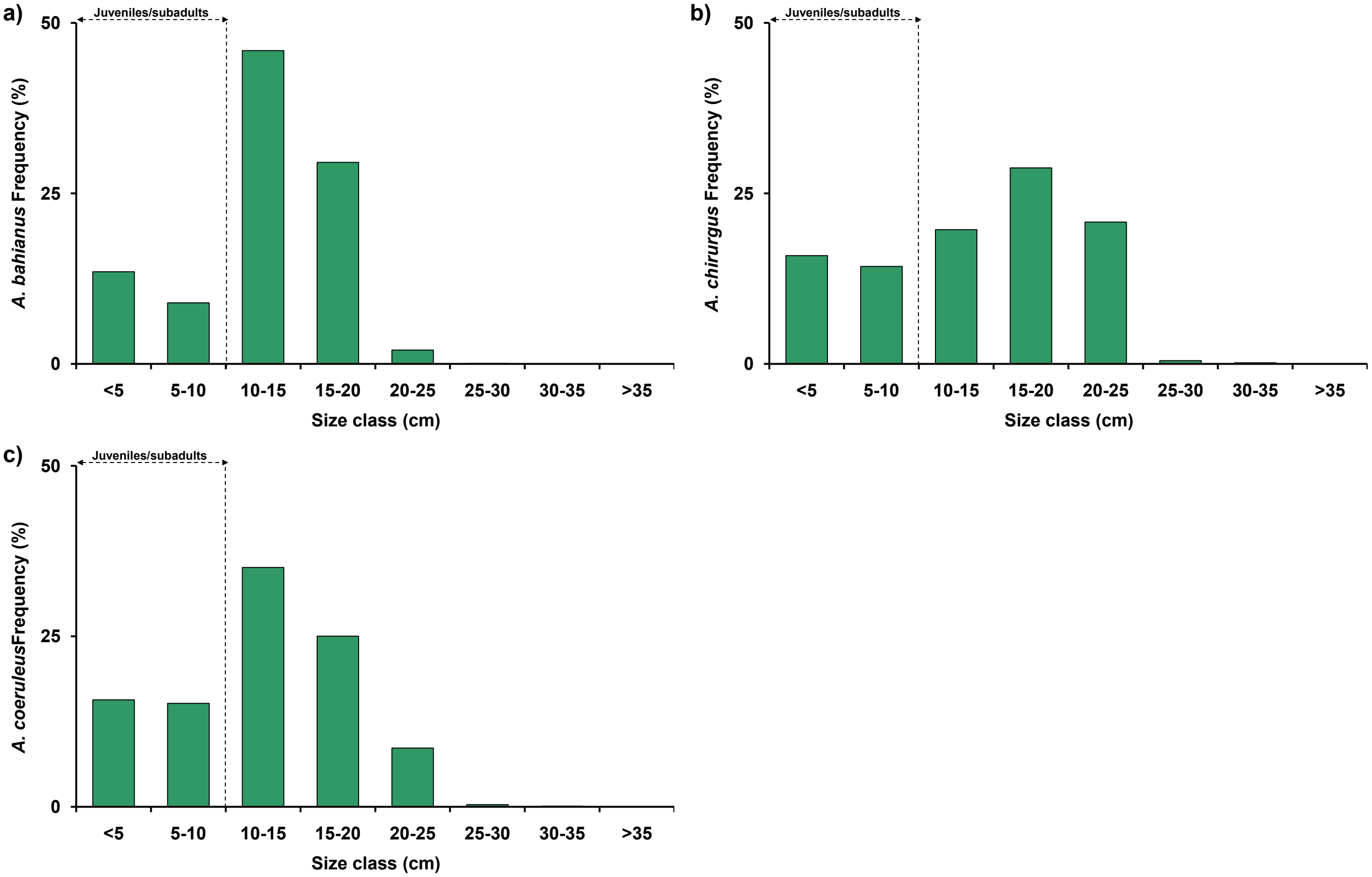


*Figure 3.78. Size frequency histogram for surgeonfish (Acanthuridae) in the southwest Puerto Rico study area. (a) ocean surgeon (*A. bahianus*), (b) doctorfish (*A. chirurgus*) and (c) blue tang (*A. coeruleus*).*

Ocean surgeonfish (*Acanthurus bahianus*)

The most frequently occurring of the surgeonfish species, *A. bahianus*, exhibited highest abundance and biomass over colonized hardbottom habitat types of the mid and outer shelf zones (Figures 3.79 and 3.80). Lowest mean abundance and biomass were observed in mangroves and over unvegetated sediments (Figure 3.80), particularly in the very nearshore zones (Figure 3.79).

Ocean surgeonfish (*Acanthurus bahianus;* top) and spotted goatfish (*Pseudupeneus maculatus*).

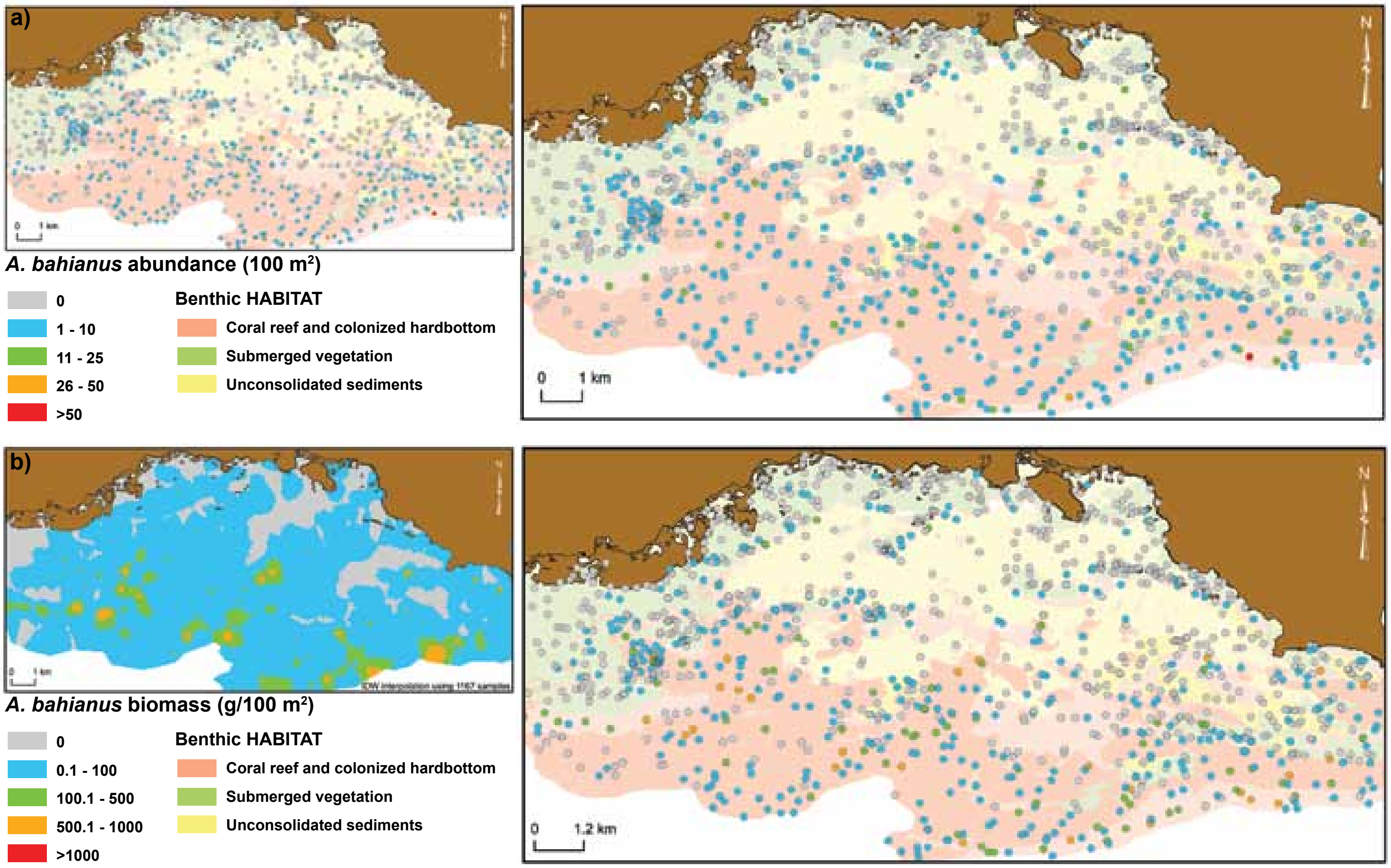


*Figure 3.79. Maps of the interpolated (left map) and spatial (right map) distributions for ocean surgeonfish (*A. bahianus*): (a) abundance and (b) biomass.*

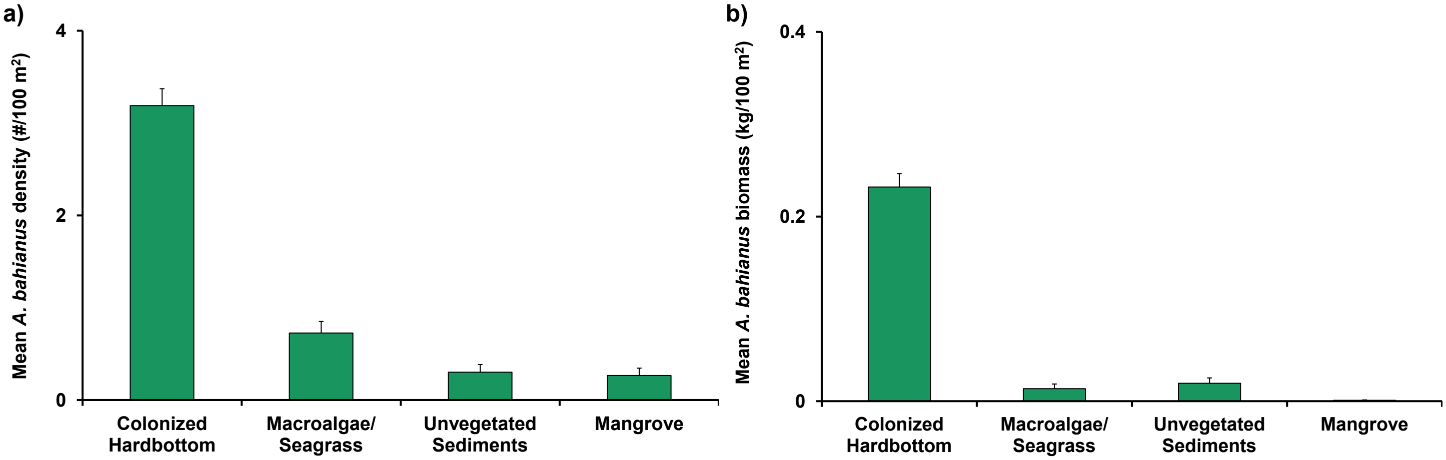


*Figure 3.80. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for ocean surgeonfish (*A. bahianus*).*

Doctorfish (*Acanthurus chirurgus*)

A. chirurgus is the least widely distributed of Acanthuridae species, with lowest abundance and biomass. An area of high abundance and biomass occured in close proximity to the coral reefs in the El Palo and Margarita Reef areas, as well as several locations along the linear reef on the outer shelf (Figure 3.81). In contrast to other Acanthuridae species, a large proportion of the total abundance is observed within mangroves and seagrasses, although biomass was relatively low (Figure 3.82).

Doctorfish (*Acanthurus chirurgus*)

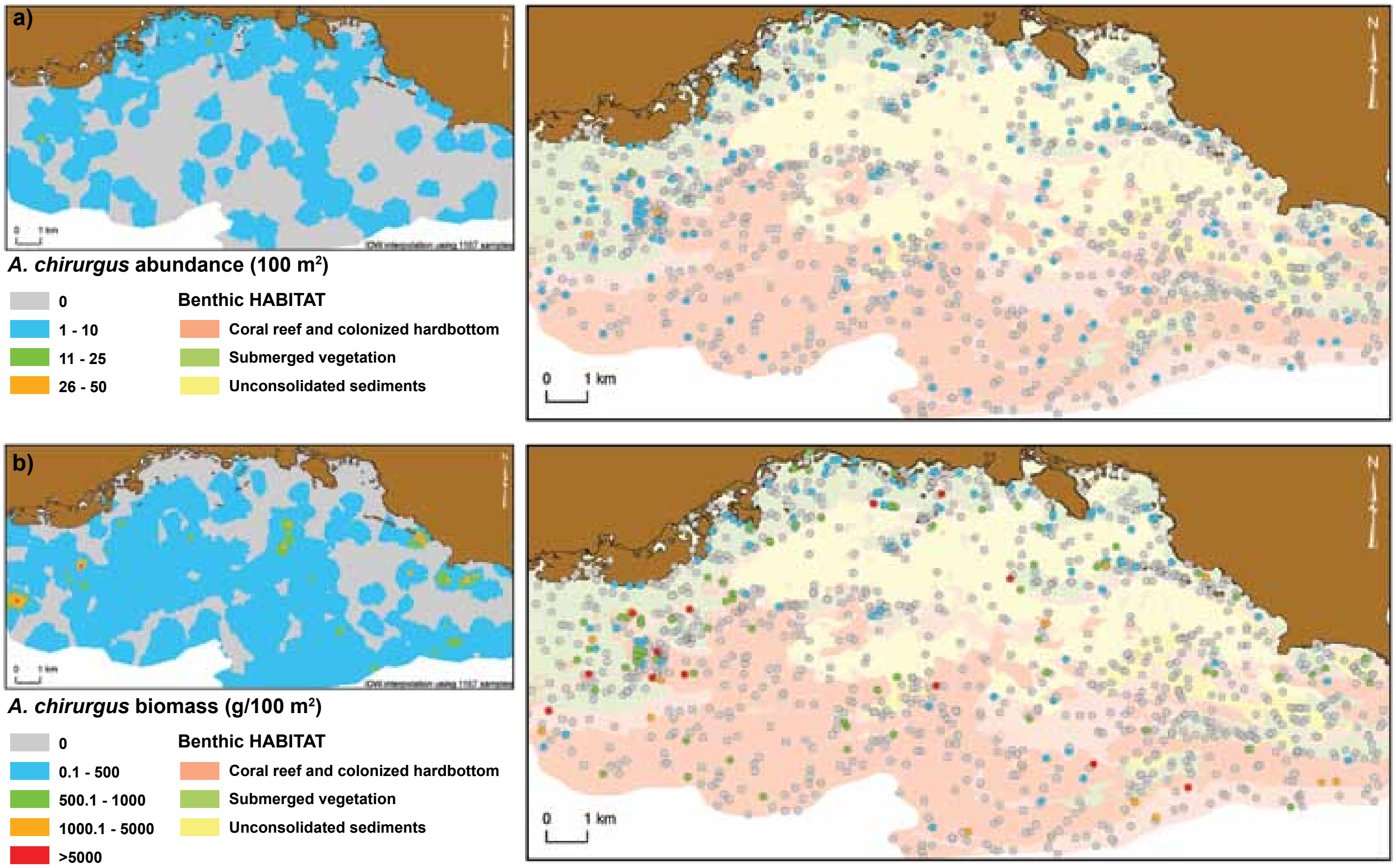


*Figure 3.81. Maps of the interpolated (left map) and spatial (right map) distributions for doctorfish (*A. chirurgus*): (a) abundance and (b) biomass.*

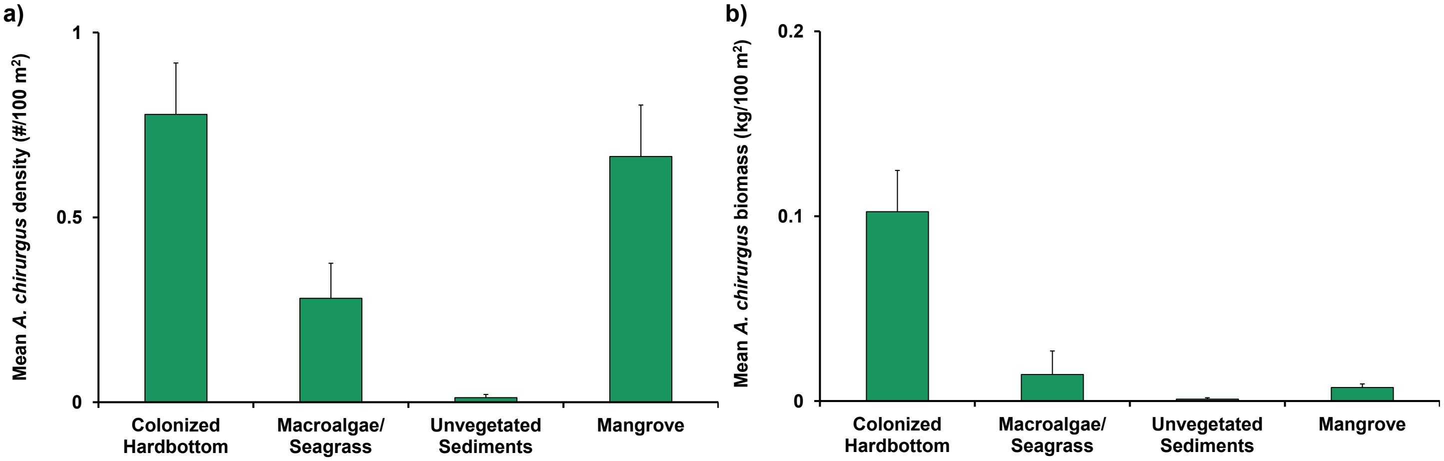


*Figure 3.82. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for doctorfish (*A. chirurgus*).*

Blue tang (*Acanthurus coeruleus*)

A. coeruleus distribution closely follows the distribution of colonized hardbottom, with hotspots of abundance and biomass located on Margarita Reef and around El Palo reef (Figure 3.83). Very few blue tang were observed in nearshore mangroves and seagrasses (Figure 3.84).

A. coeruleus

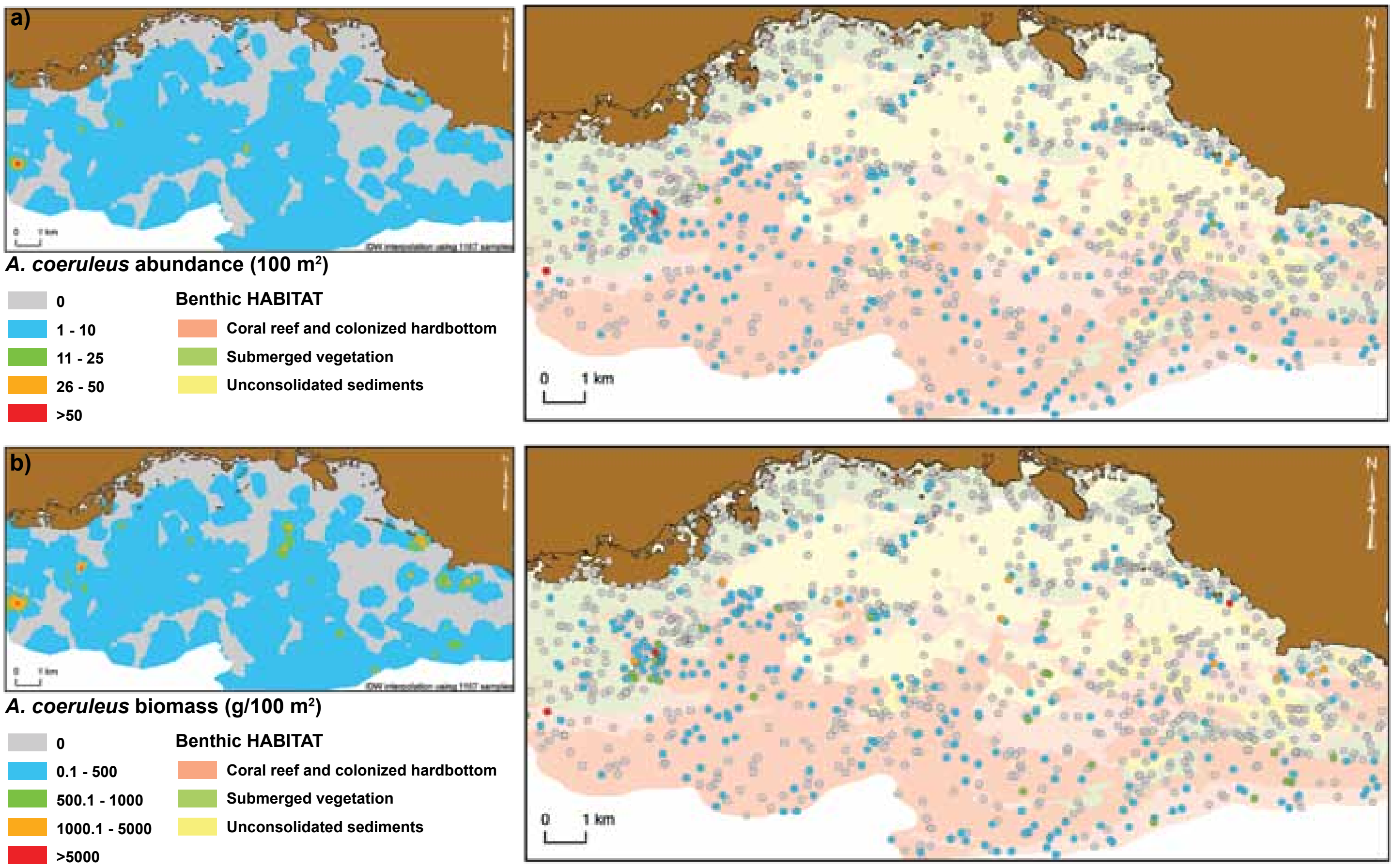


*Figure 3.83. Maps of the interpolated (left map) and spatial (right map) distributions for blue tang (*A. coeruleus*): (a) abundance and (b) biomass.*

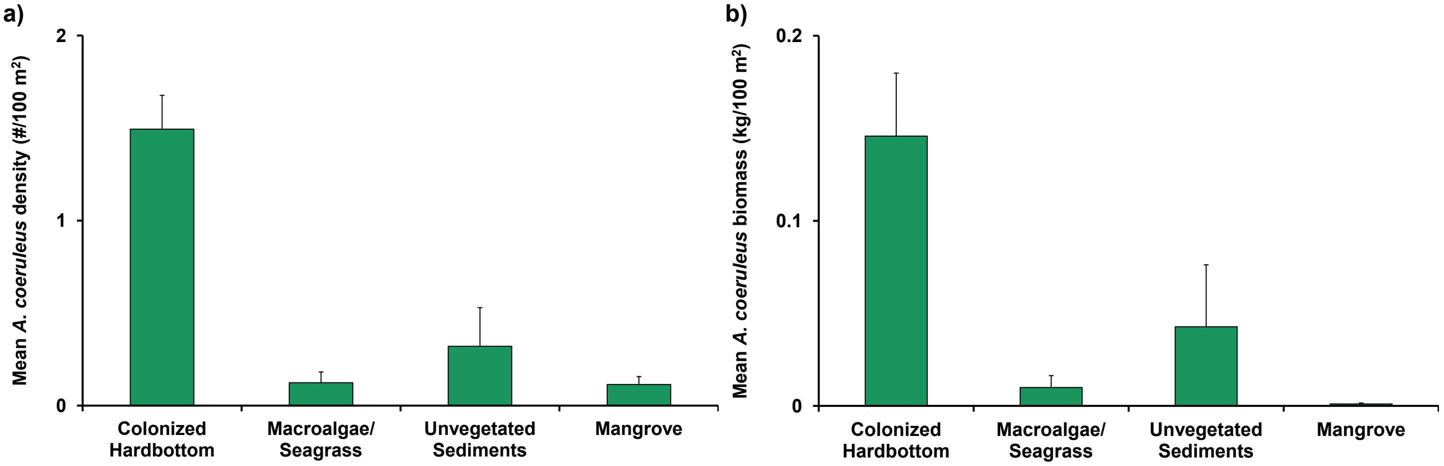


*Figure 3.84. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for blue tang (*A. coeruleus*).*

Goatfishes (Mullidae)

The smallest juveniles (<5 cm) represented a very small proportion (<6% of all *Pseudupeneus maculatus* [spotted goatfish]) of all Mullidae (Figure 3.85). None of the smallest *Mulloidichthys martinicus* (yellow goatfish) were seen in the study area (Figure 3.85a). The largest proportion of all Mullidae were large juveniles and subadults, with a marked decline in the proportion of adults, with a very small proportion of the largest adults (>30 cm FL= 1% of the total). No *P. maculatus* larger than 30 cm FL were recorded and no *M. martinicus* larger than 35 cm FL were recorded in the study area (Figure 3.85). The maximum known for the species is 30 cm for *P. maculatus* and 39.4 for *M. martinicus* (Table 3.4).

P. maculatus

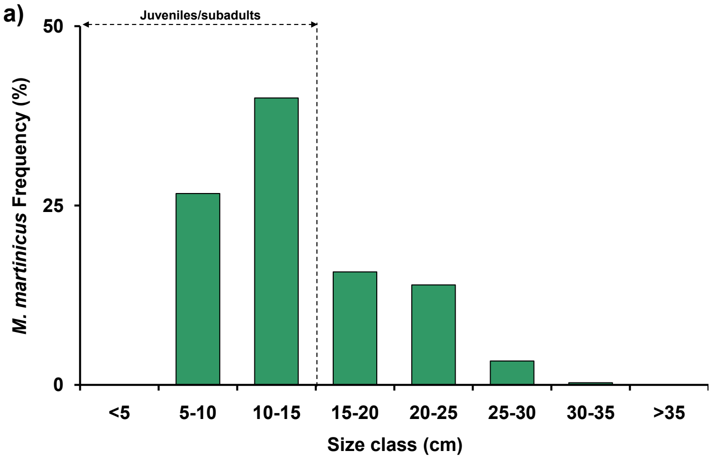
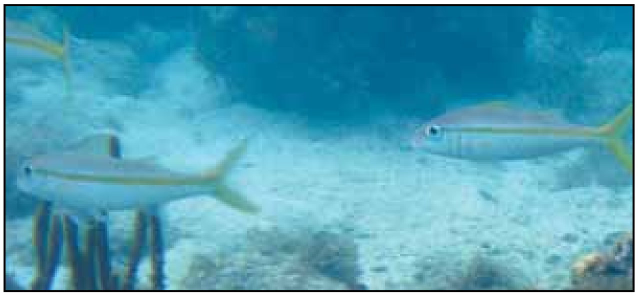


*Figure 3.85. Size frequency histogram for goatfish (Mullidae) in the southwest Puerto Rico study area. (a) yellow goatfish (*M. martinicus*) and (b) spotted goatfish (*P. maculatus*).*

Yellow goatfish (*Mulloidichthys martinicus*)

Although *M. martinicus* was observed using all major habitat types it was less widespread and abundant than *P. maculatus* (Figures 3.86 and 3.87), with highest mean abundance over unvegetated sediments and colonized hardbottom (Figure 3.87a). Mean biomass was highest for colonized hardbottom. *M. martinicus* also utilized mangroves (Figure 3.87b).



Yellow goatfish (*Mulloidichthys martinicus*) in St. John, USVI

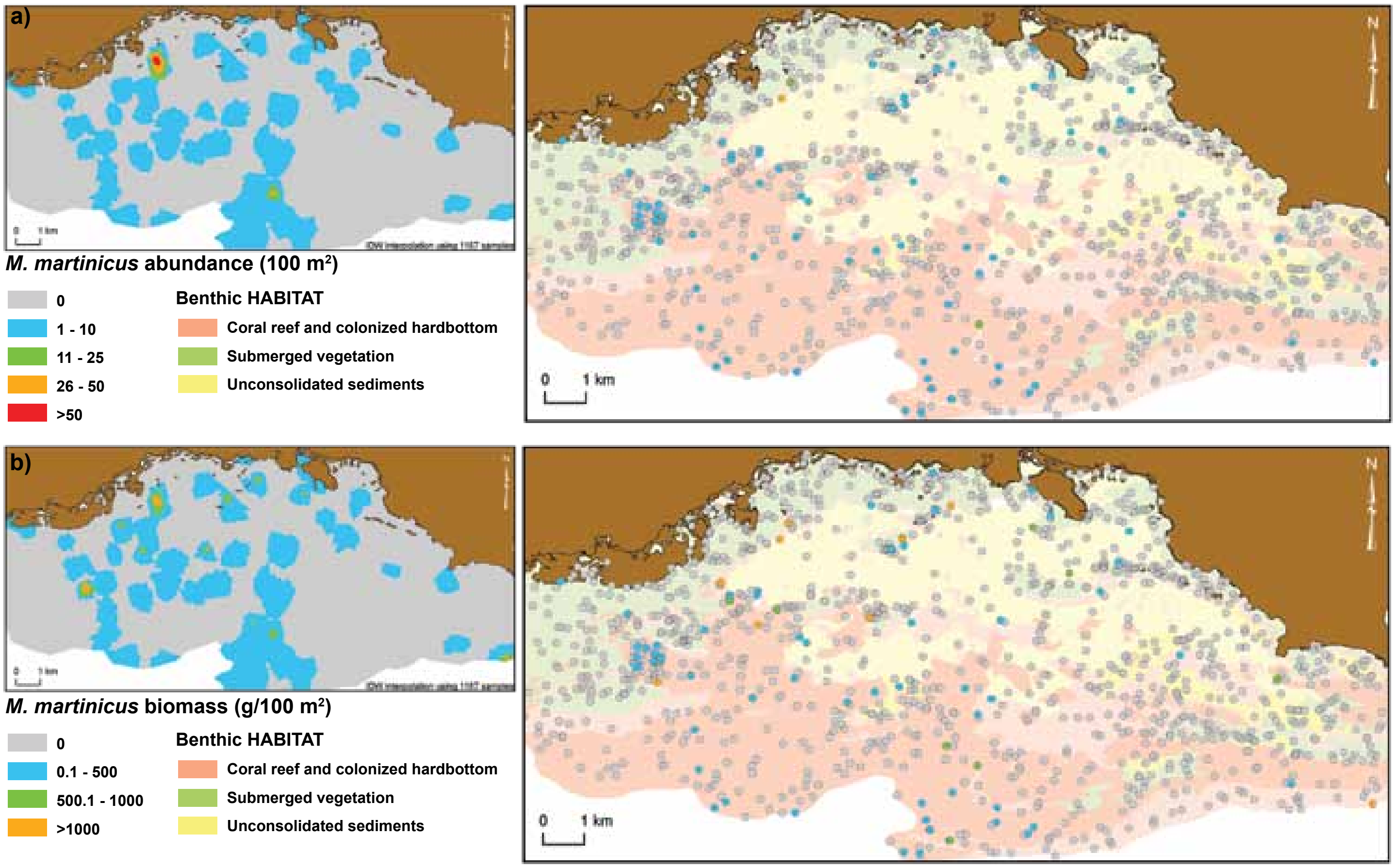


*Figure 3.86. Maps of the interpolated (left map) and spatial (right map) distributions for yellow goatfish (*M. martinicus*): (a) abundance and (b) biomass.*

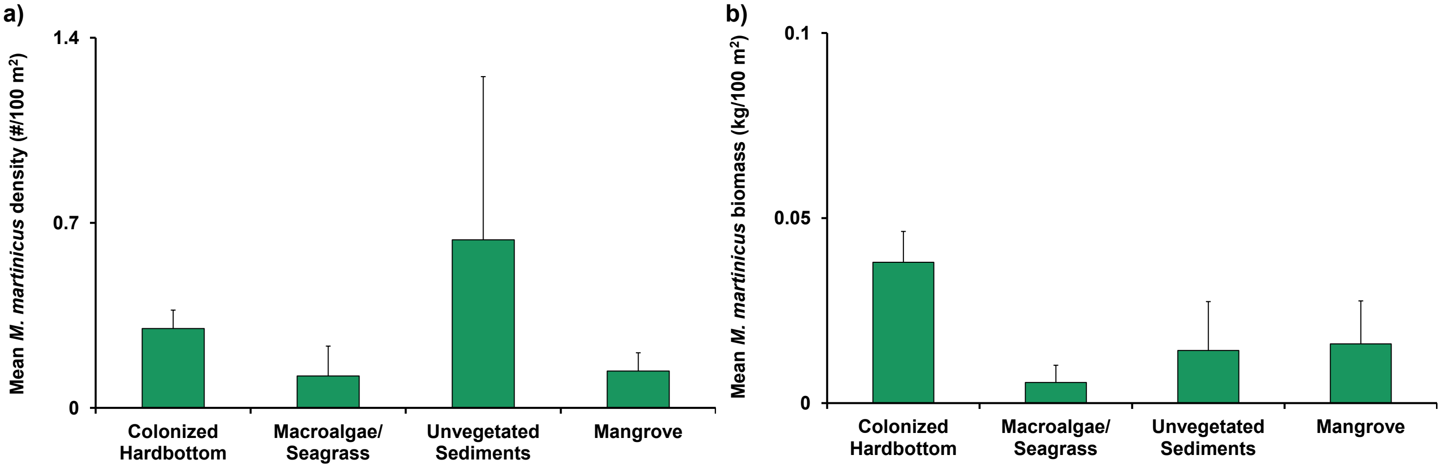


*Figure 3.87. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for yellow goatfish (*M. martinicus*).*

Spotted goatfish (*Pseudupeneus maculatus*)

P. maculatus was found in all major habitat types except mangroves and in all zones across the shelf (Figure 3.88). No distinct geographical hotspots were evident within the study area. Mean abundance was highest over seagrasses and then colonized hardbottom, while the reverse was true for mean biomass (Figure 3.89).

P. maculatus

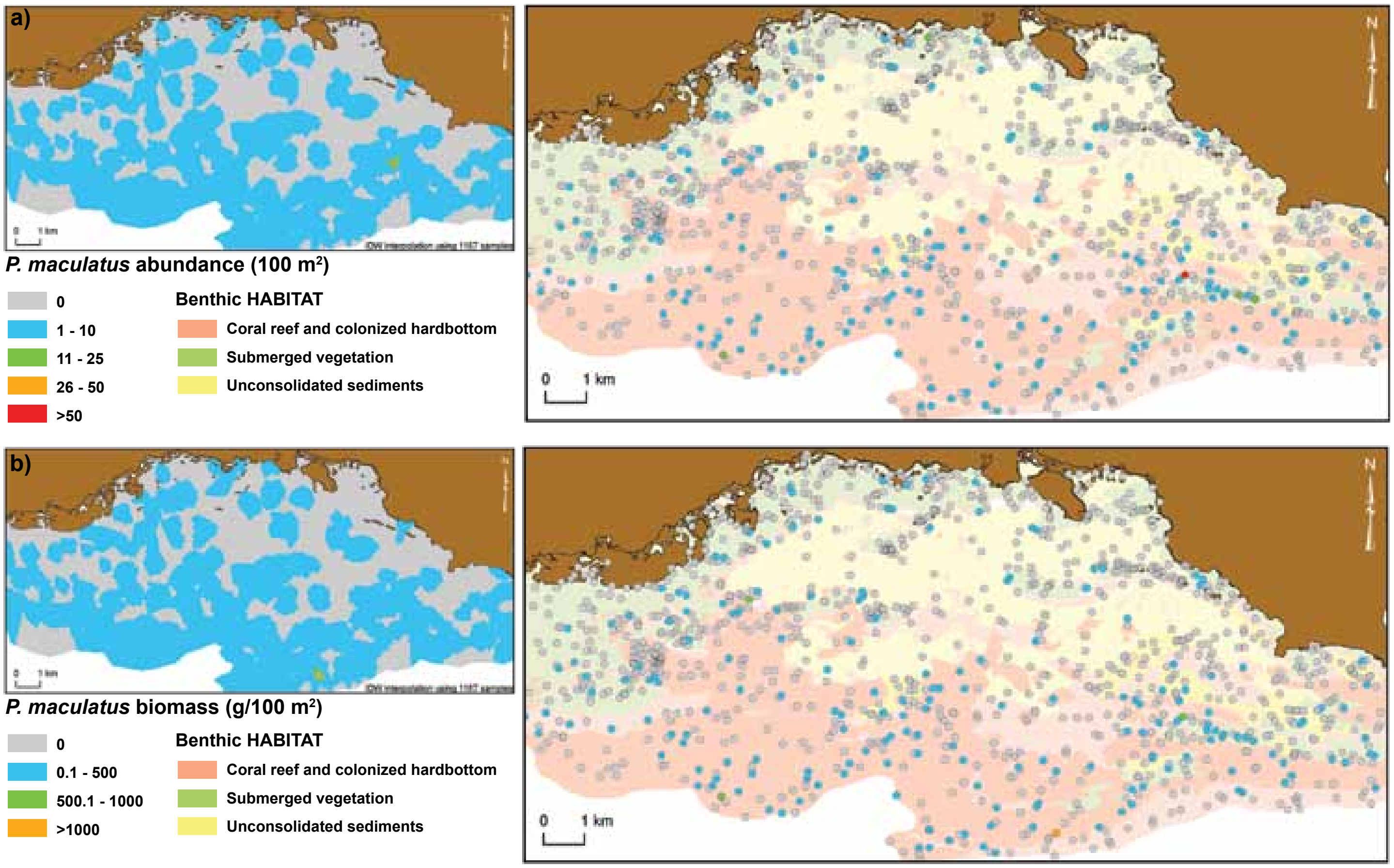


*Figure 3.88. Maps of the interpolated (left map) and spatial (right map) distributions for spotted goatfish (*P. maculatus*): (a) abundance and (b) biomass.*

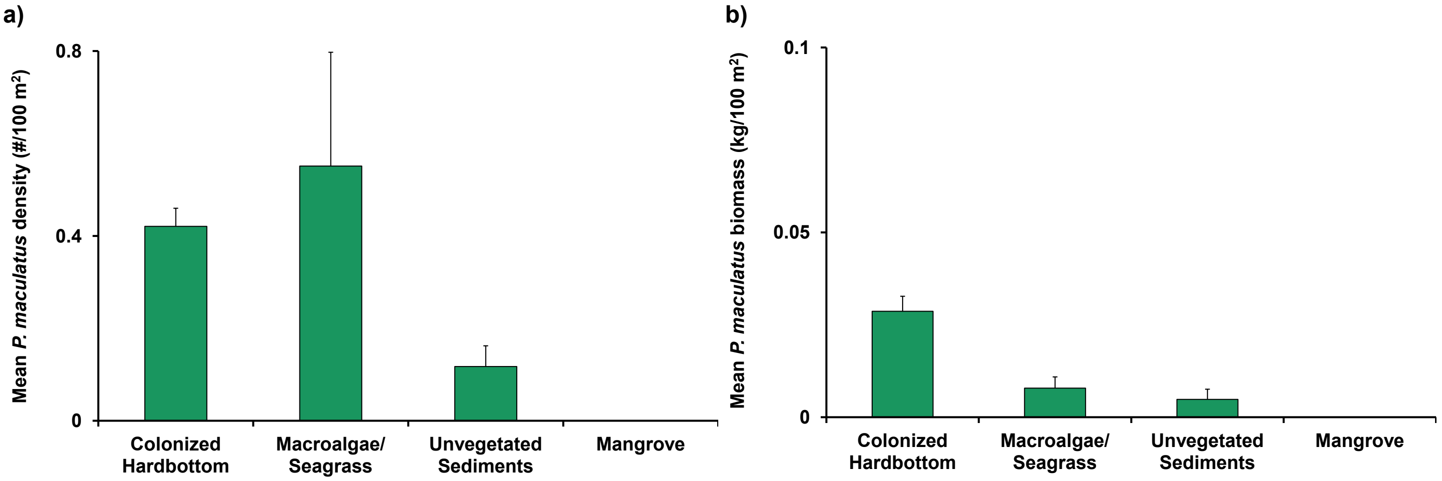


*Figure 3.89. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for spotted goatfish (*P. maculatus*).*

Jacks (Carangidae)

Highest mean Carangidae abundance was associated with colonized hardbottom habitat types (Figure 3.90a), with highest abundance found on or around the extensive area of colonized hardbottom on the mid and outer shelf zones (Figure 3.90a). The high biomass associated with larger schools of carangids and larger-bodied carangids were found at several locations over softbottom habitat types (seagrass and sand; Figure 3.90b) close to the edges of colonized hardbottom (Figure 3.90b). Very few carangids were observed in fringing mangroves and from nearshore environments in general (Figure 3.91).

Yellow jack (*Carangoides bartholomaei*)

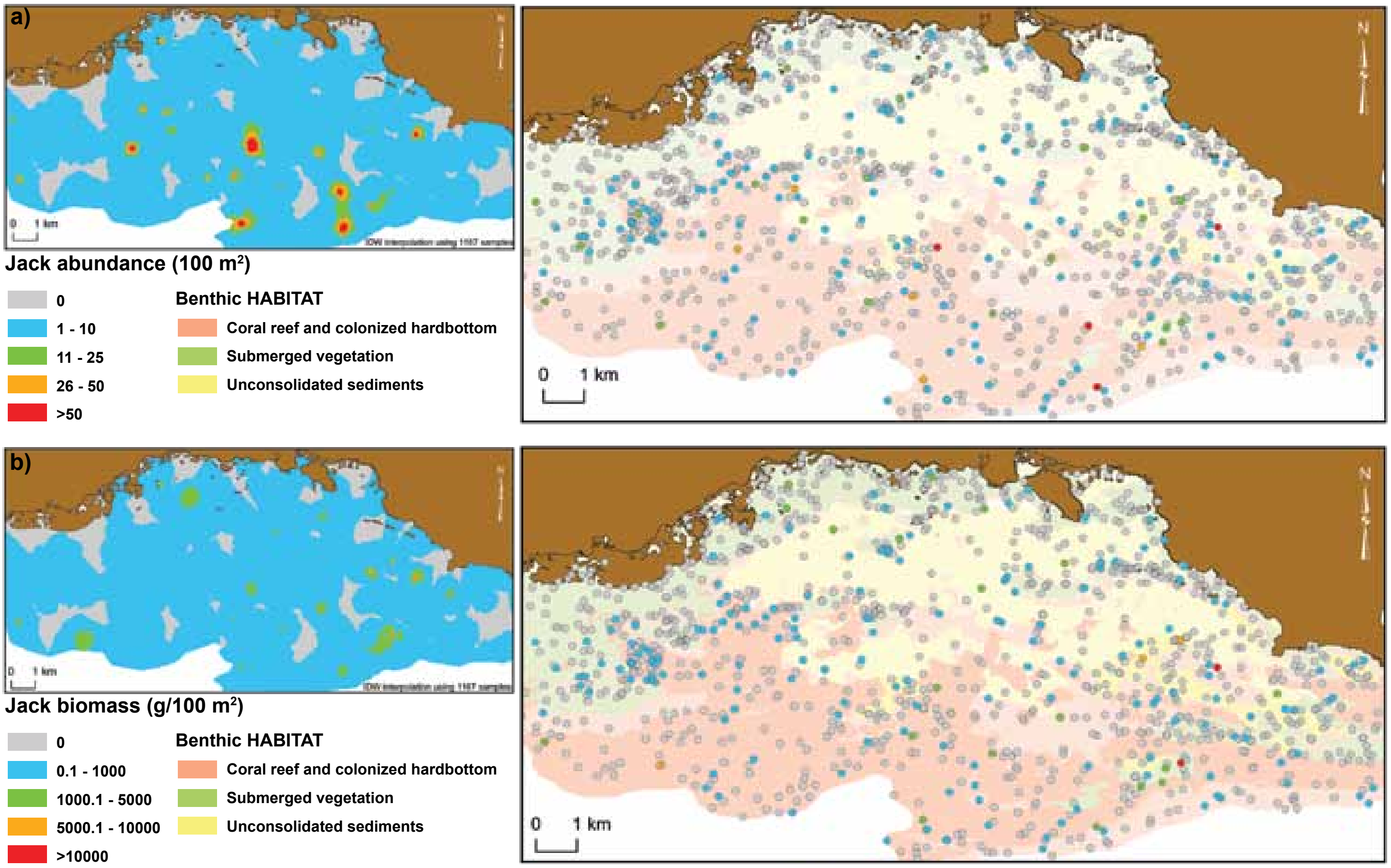


Figure 3.90. Maps of the interpolated (left map) and spatial (right map) distributions for jack (Carangidae): (a) abundance and (b) biomass.

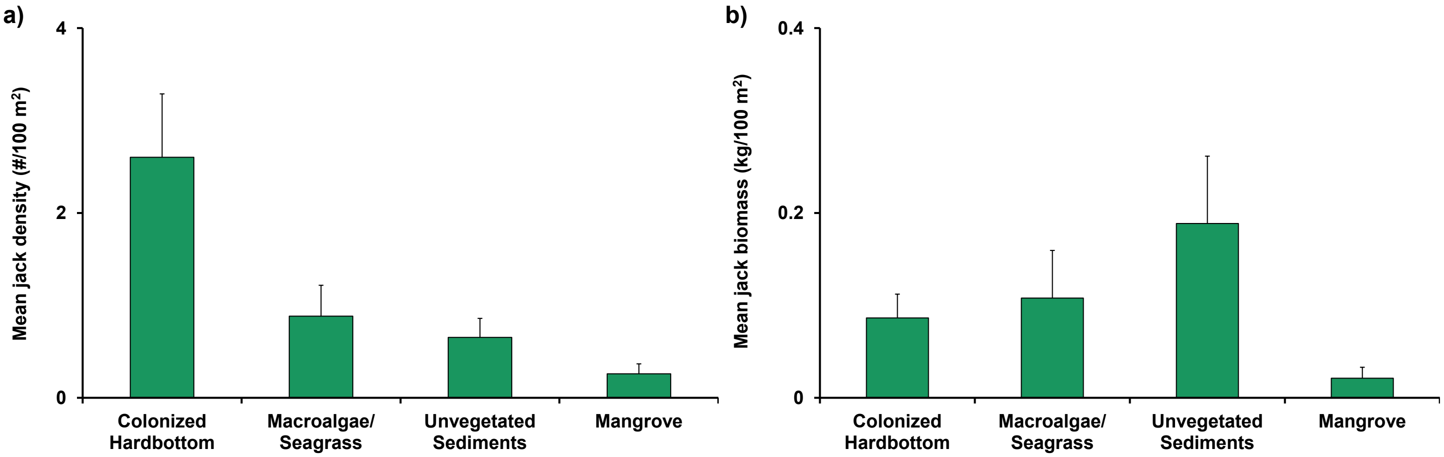


Figure 3.91. Comparison of mean (± SE) density and biomass by habitat type in the southwest Puerto Rico study area for jack (Carangidae).

Blue runner (*Caranx crysos*)

Bar jack (*Carangoides ruber*)

Two of the most common Carangidae species were *Caranx crysos* (blue runner) and *Carangoides ruber* (bar jack). *C. crysos* density and biomass were highest around unvegetated sediments. Very few were observed in colonized hardbottom and macroalgae/seagrass habitats, and absent in mangroves (Figure 3.92). *C. ruber* density and biomass were highest in colonized hardbottom habitats and very few recorded in softbottom habitats (sand and seagrasses; Figure 3.93). As with *C. crysos*, *C. ruber* were also absent from mangroves (Figure 3.93).

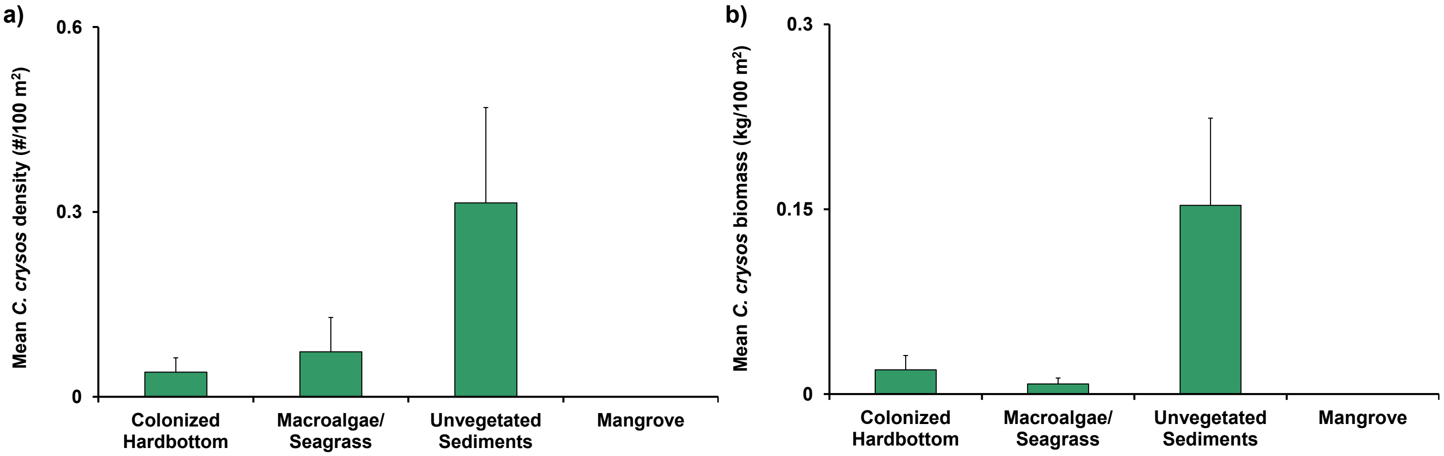


*Figure 3.92. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for blue runner (*C. crysos*).*

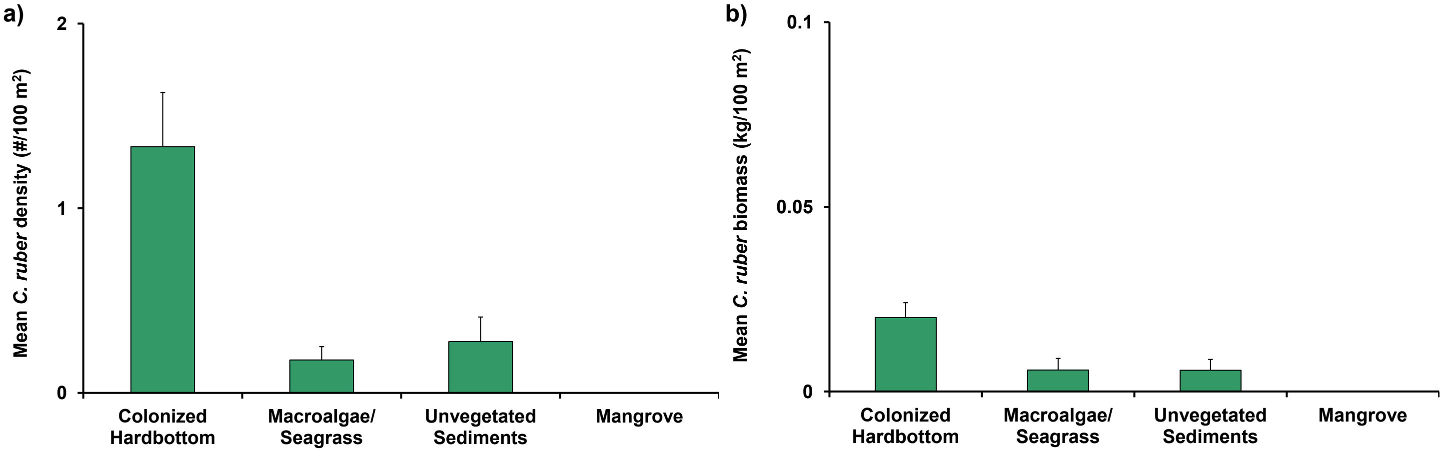


*Figure 3.93. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for bar jack (*C. ruber*).*

Damselfish (Pomacentridae)

Yellowtail damselfish (*Microspathodon chrysurus*)

Like most damselfish, *M. chrysurus* were very closely associated with colonized hardbottom habitat types, but exhibited a distribution primarily within the mid shelf zone (Figure 3.94). Mangroves were not used by this species and nearshore and lagoonal colonized hardbottom areas were rarely used. Mean abundance and biomass were very low over seagrasses and unvegetated sediments (Figure 3.95).

Yellowtail damselfish (*Microspathodon chrysurus*)

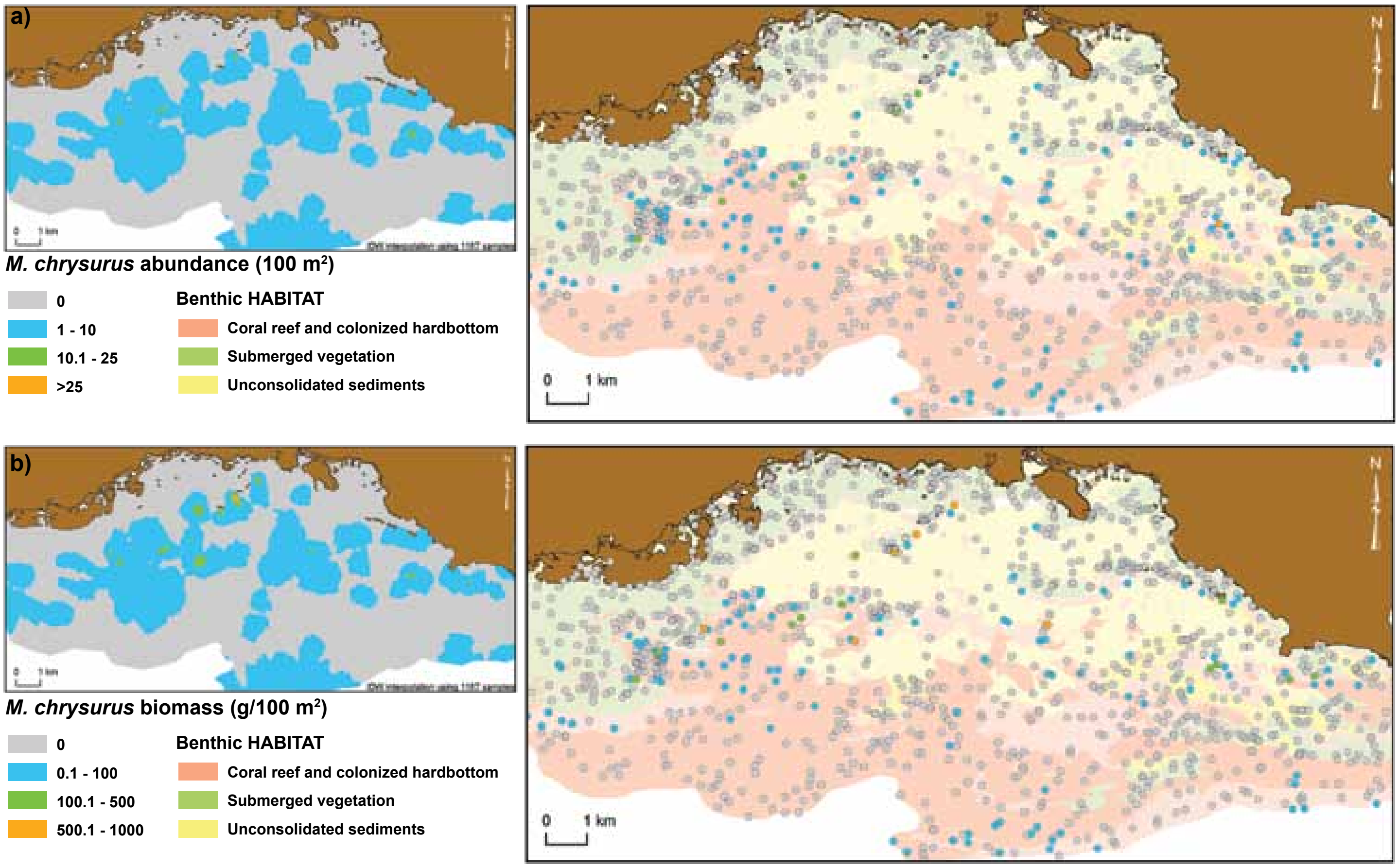


*Figure 3.94. Maps of the interpolated (left map) and spatial (right map) distributions for yellowtail damselfish (*M. chrysurus*): (a) abundance and (b) biomass.*

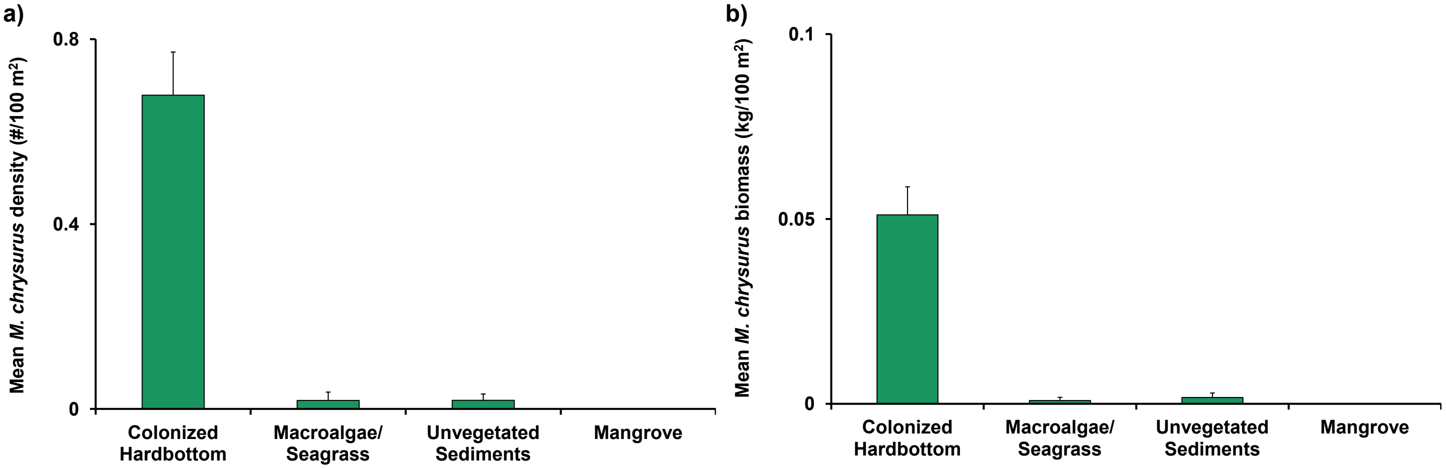


*Figure 3.95. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for yellowtail damselfish (*M. chrysurus*).*

Dusky damselfish (*Stegastes adustus*)

S. adustus were seen in all major habitat types, but were mostly observed over colonized hardbottom, with highest abundance found on the shallow water forereefs, reefcrests and patch reefs of the mid shelf and lagoonal zones (Figure 3.96). *S. adustus* were rarely seen on the deeper colonized hardbottom areas of the outer shelf zone. Lowest abundance and biomass were recorded for mangroves and unvegetated sediments (Figure 3.97).

Dusky damselfish (*Stegastes adustus*) in St. Croix, USVI

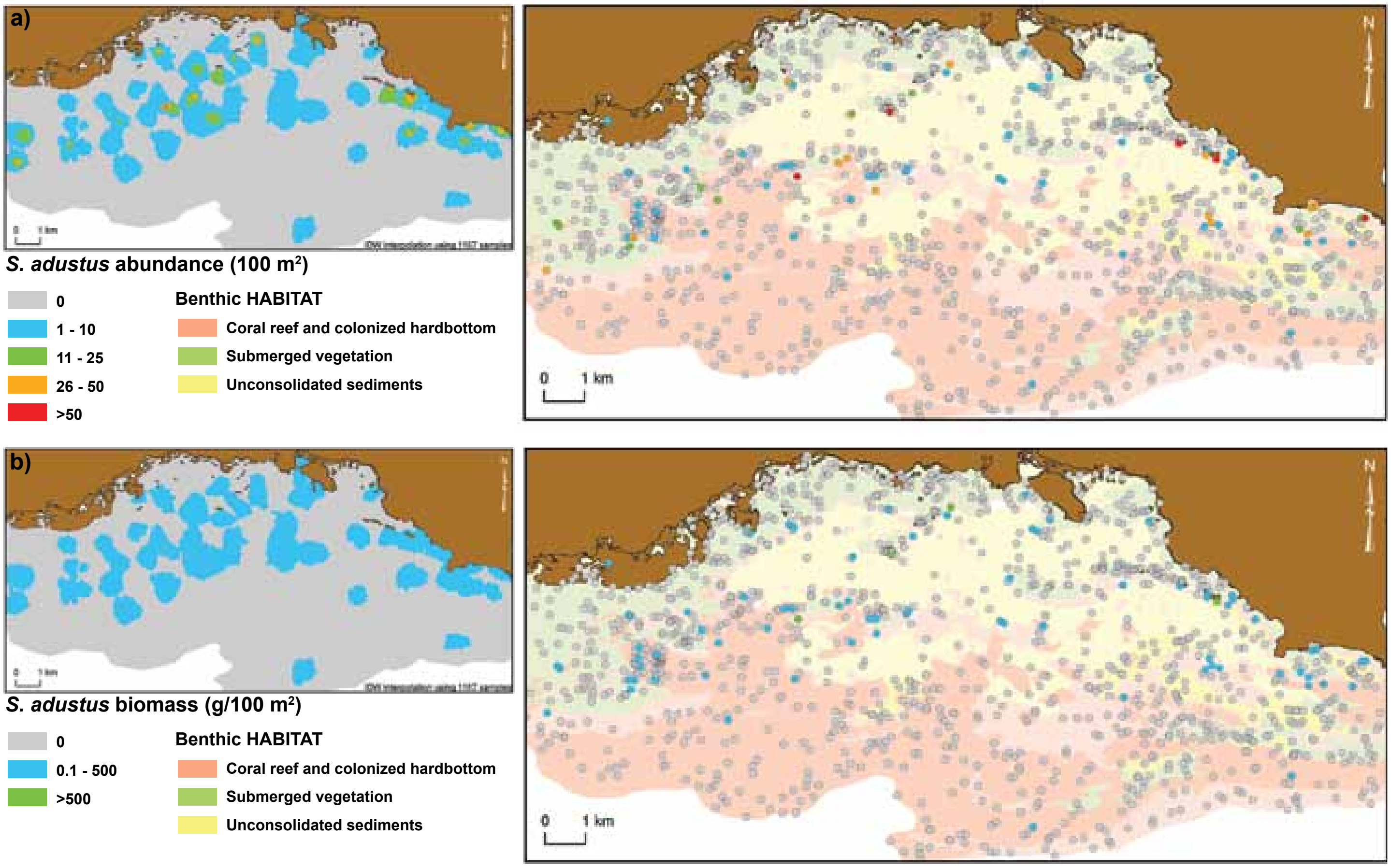


*Figure 3.96. Maps of the interpolated (left map) and spatial (right map) distributions for dusky damselfish (*S. adustus*): (a) abundance and (b) biomass.*

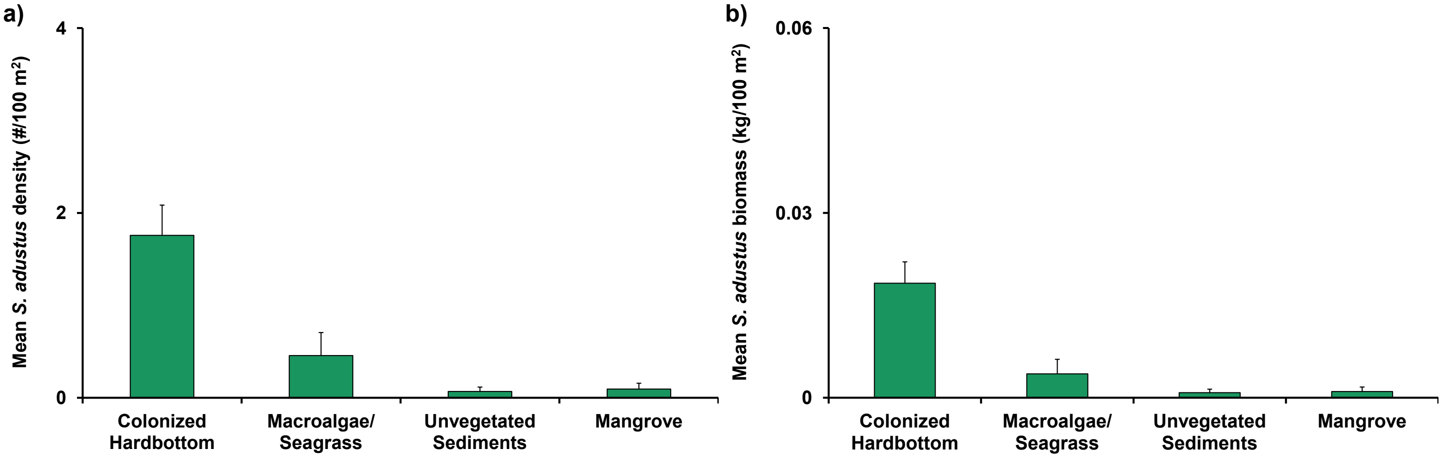


*Figure 3.97. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for dusky damselfish (*S. adustus*).*

Beaugregory (*Stegastes leucostictus*)

S. leucostictus was observed in all major habitat types and was widely distributed in nearshore and lagoonal zones, with highest mean abundance and biomass in both fringing mangroves and mangrove cays (Figure 3.98). Comparatively few *S. leucostictus* were observed over deeper water colonized hardbottom on the mid and outer shelf zone. Lowest abundance and biomass were recorded for unvegetated sediments (Figure 3.99).

Beaugregory (*Stegastes leucostictus*) juvenile in St. Croix, USVI

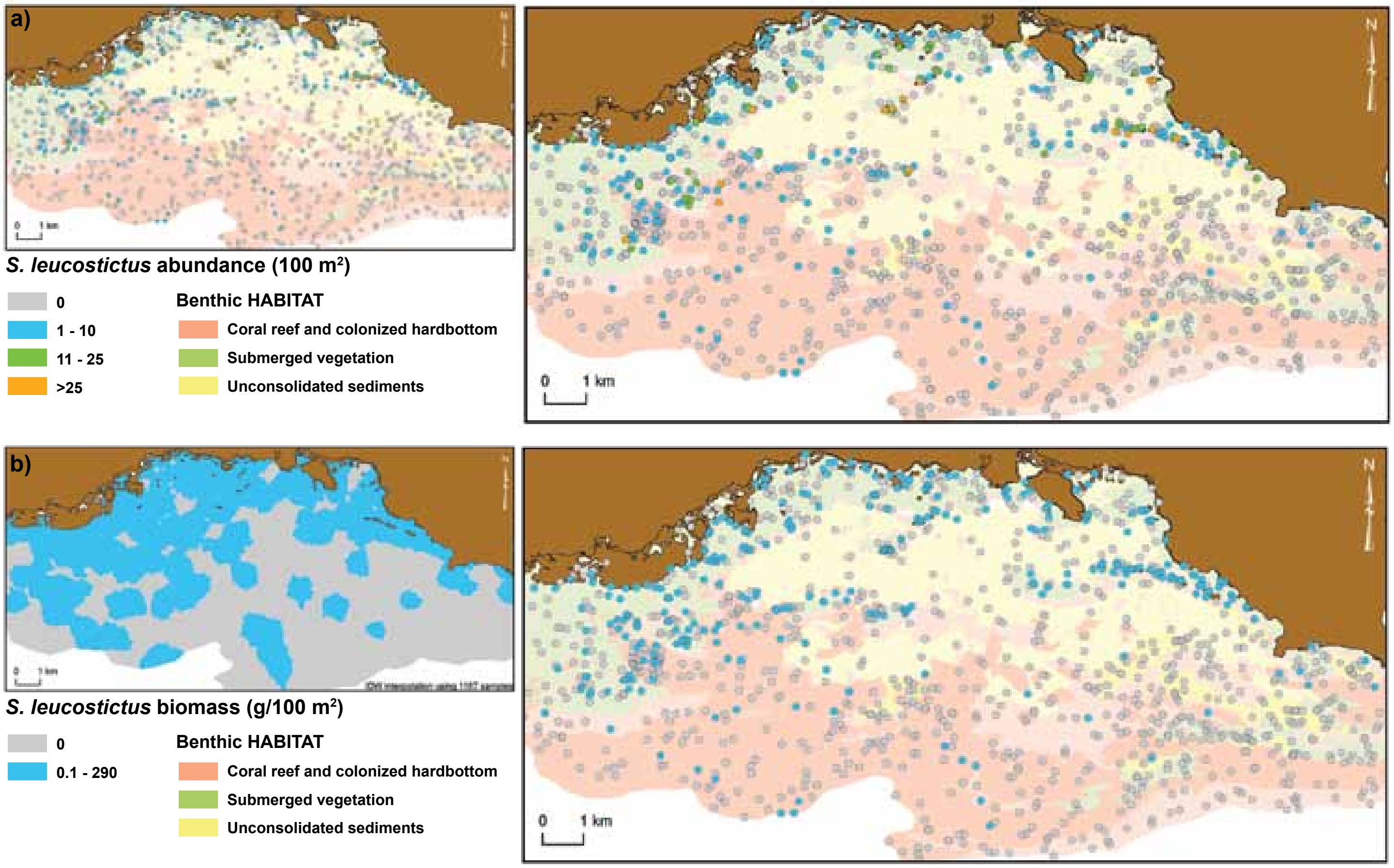


*Figure 3.98. Maps of the interpolated (left map) and spatial (right map) distributions for beaugregory (*S. leucostictus*): (a) abundance and (b) biomass.*

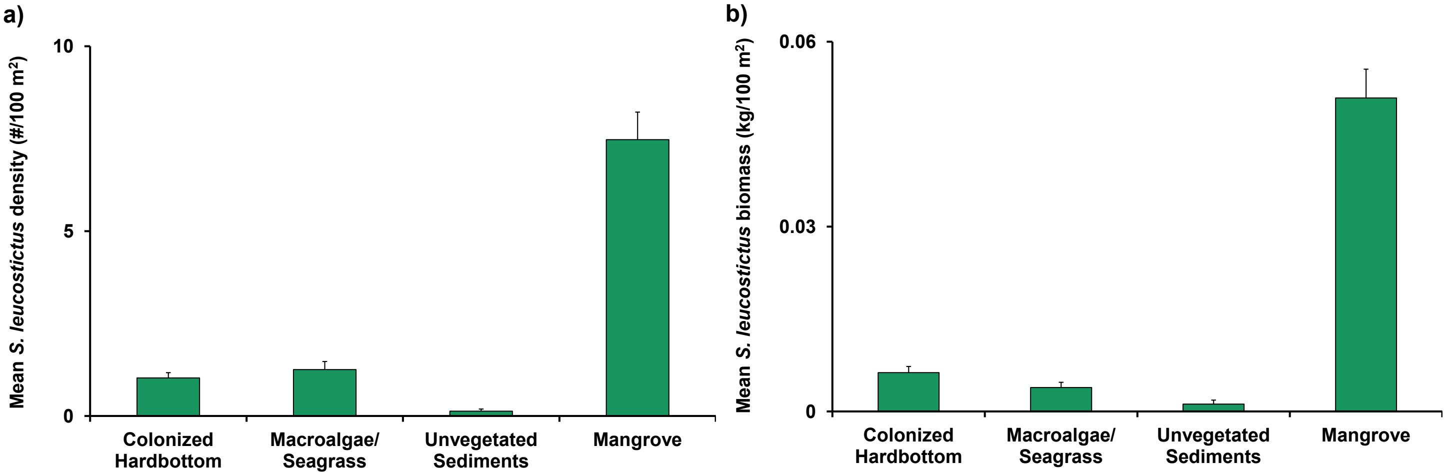


*Figure 3.99. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for beaugregory (*S. leucostictus*).*

Bicolor damselfish (*Stegastes partitus*)

S. partitus is found at approximately 39% of sites across the study area, with highest mean abundance and biomass associated with colonized hardbottom habitat types. Geographically, abundance is markedly higher on the colonized hardbottom sites located on the outer shelf zone and lowest for the very nearshore and lagoonal zones (Figure 3.100). Lowest abundance and biomass were recorded for mangroves and unvegetated sediments (Figure 3.101).

Bicolor damselfish (*Stegastes partitus*)

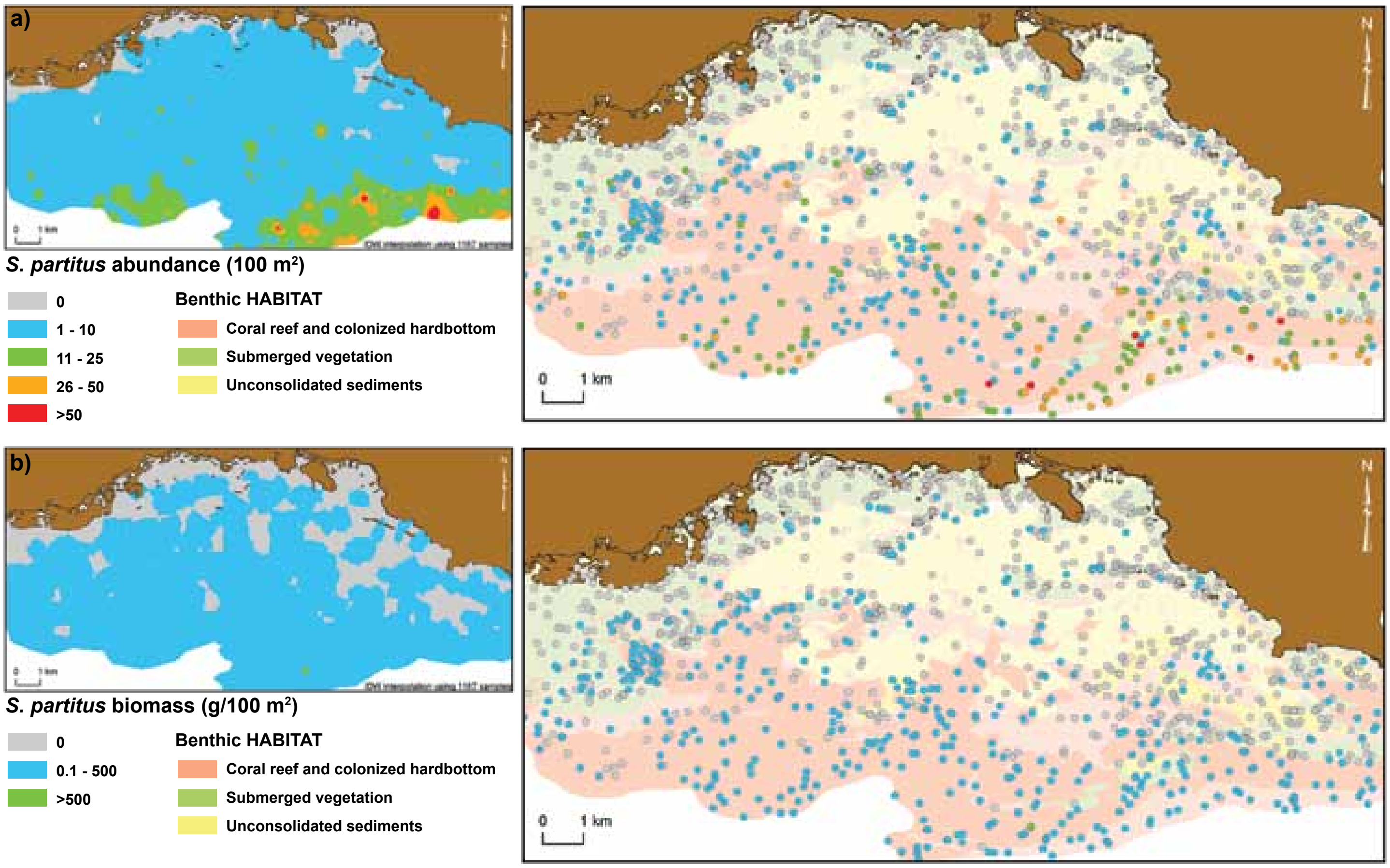


*Figure 3.100. Maps of the interpolated (left map) and spatial (right map) distributions for bicolor damselfish (*S. partitus*): (a) abundance and (b) biomass.*

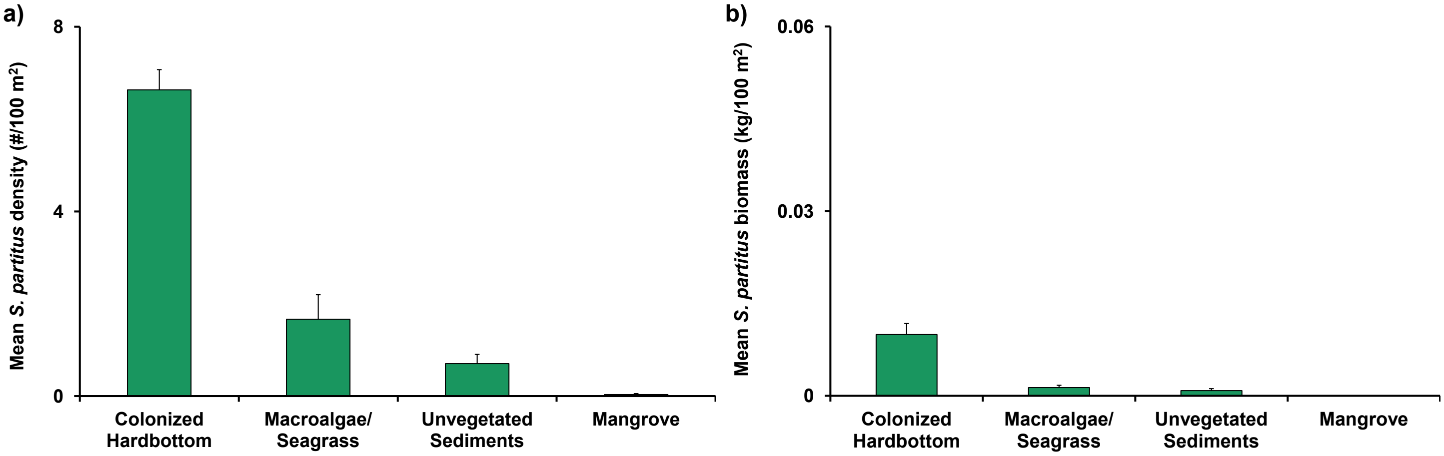


*Figure 3.101. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for bicolor damselfish (*S. partitus*).*

Threespot damselfish (*Stegastes planifrons*)

S. planifrons was well distributed across the colonized hardbottom habitat types of the mid shelf zone predominantly occurring in forereefs and reefcrest zones and in the most topographically complex areas of colonized pavement with sand channels on the mid and outer shelf zone (Figure 3.102). Lowest mean abundance and biomass of *S. planifrons* was recorded for mangroves (Figure 3.103).

Adult (left) and juvenile (right) threespot damselfish (*Stegastes planifrons*).

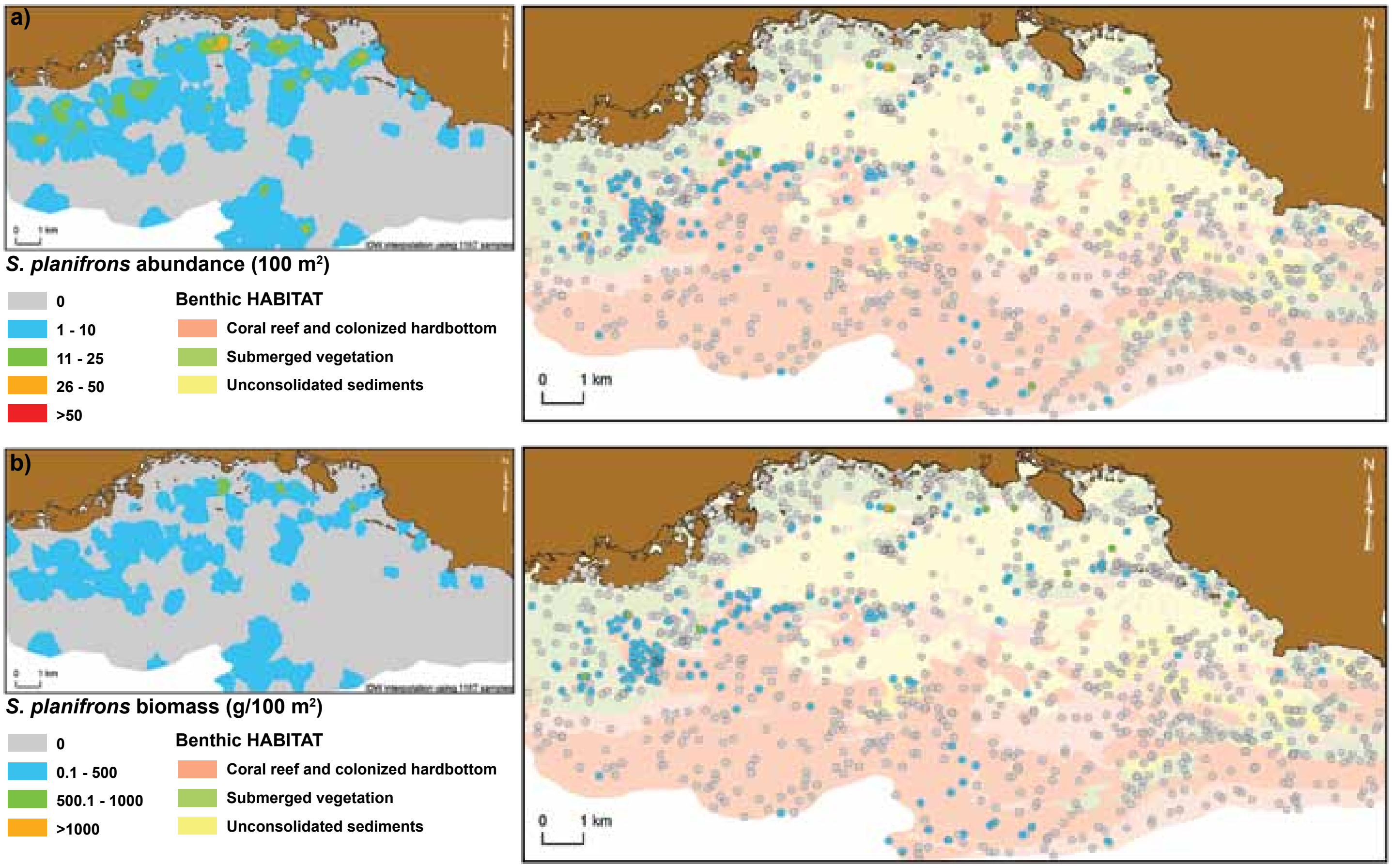


*Figure 3.102. Maps of the interpolated (left map) and spatial (right map) distributions for threespot damselfish (*S. planifrons*): (a) abundance and (b) biomass.*

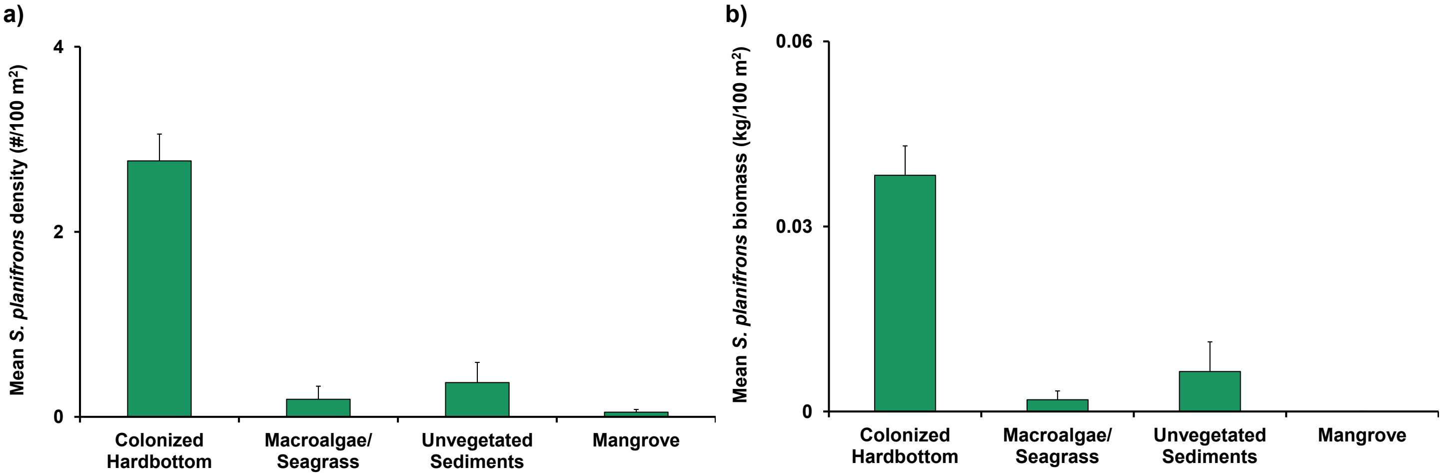
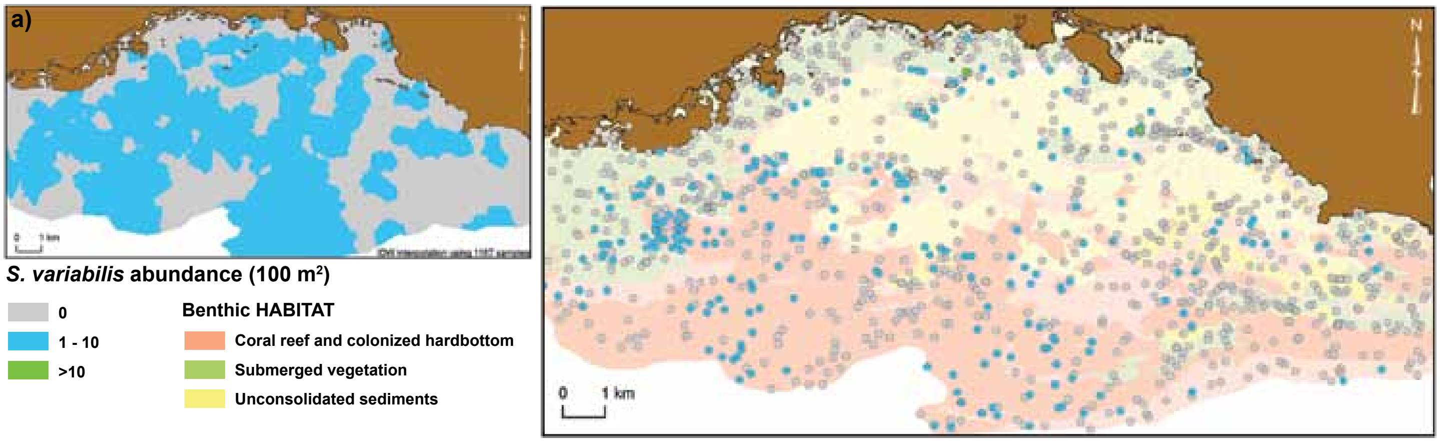


*Figure 3.103. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for threespot damselfish (*S. planifrons*).*

Cocoa damselfish (*Stegastes variabilis*)

S. variabilis exhibited a similar distribution as S. planifrons associated with topographically complex hardbottom, albeit observed at lower densities (Figures 3.104). Mean abundance and biomass were highest in colonized hardbottom and lowest in seagrasses. Of note was that *S. variabilis* abundance was higher in mangroves than seagrasses (Figure 3.105).

Cocoa damselfish (*Stegastes variabilis*), St. John, USVI.



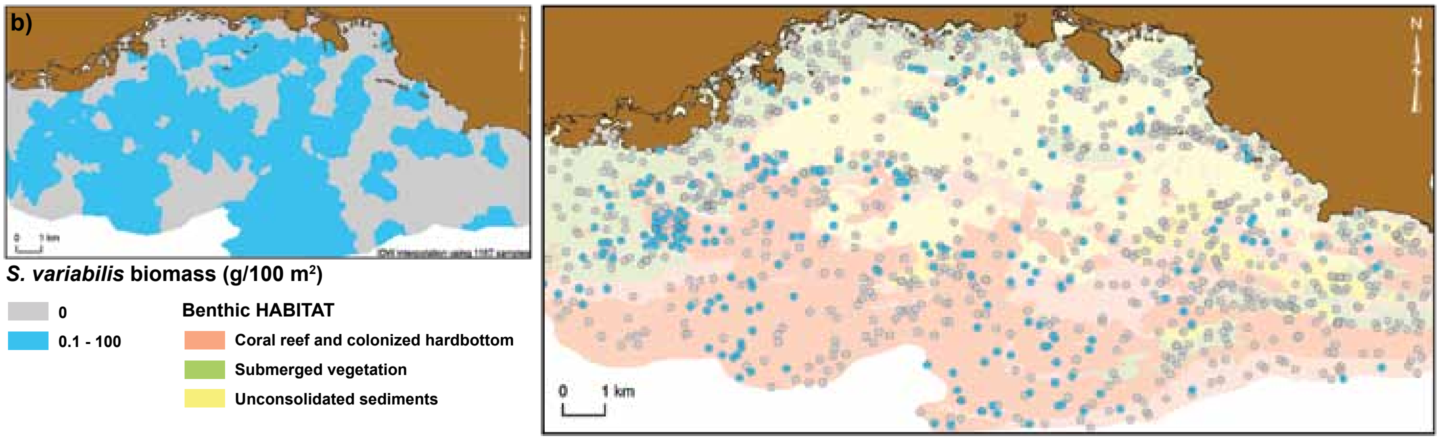


*Figure 3.104. Maps of the interpolated (left map) and spatial (right map) distributions for cocoa damselfish (*S. variabilis*): (a) abundance and (b) biomass.*

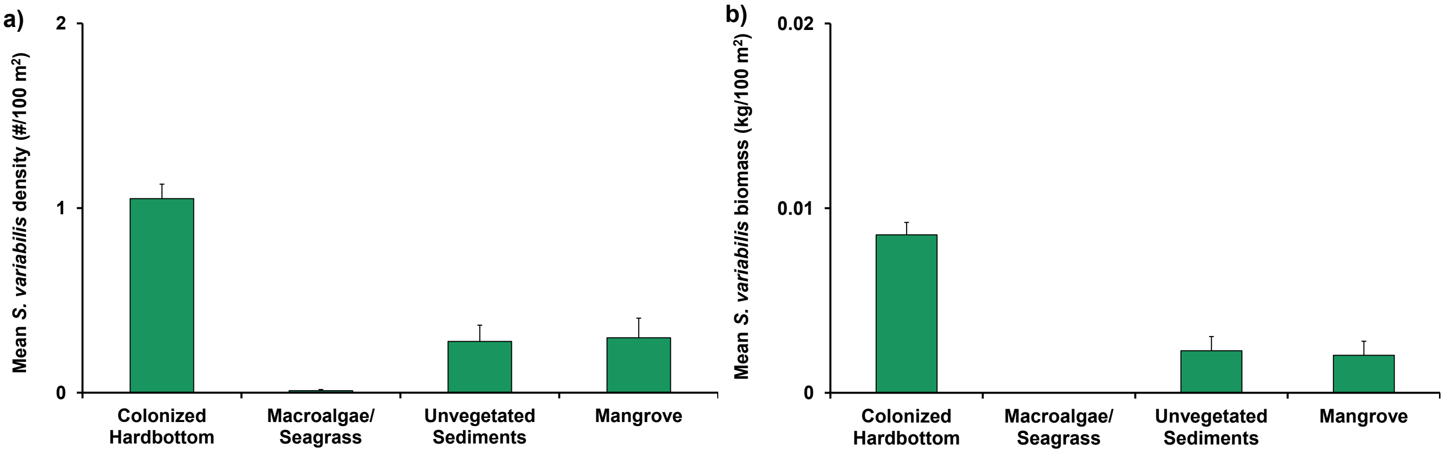


*Figure 3.105. Comparison of mean (± SE) density and biomass by substrate type in southwest Puerto Rico for cocoa damselfish (*S. variabilis*).*

Wrasses (Labridae)

Highest Labridae density and biomass were observed on the colonized hardbottom sites across the outer shelf zone, with several isolated abundance hotspots across colonized hardbottom sites in the midshelf zone (Figures 3.106 and 3.107).

Bluehead wrasse (*Thalassoma bifasciatum*)

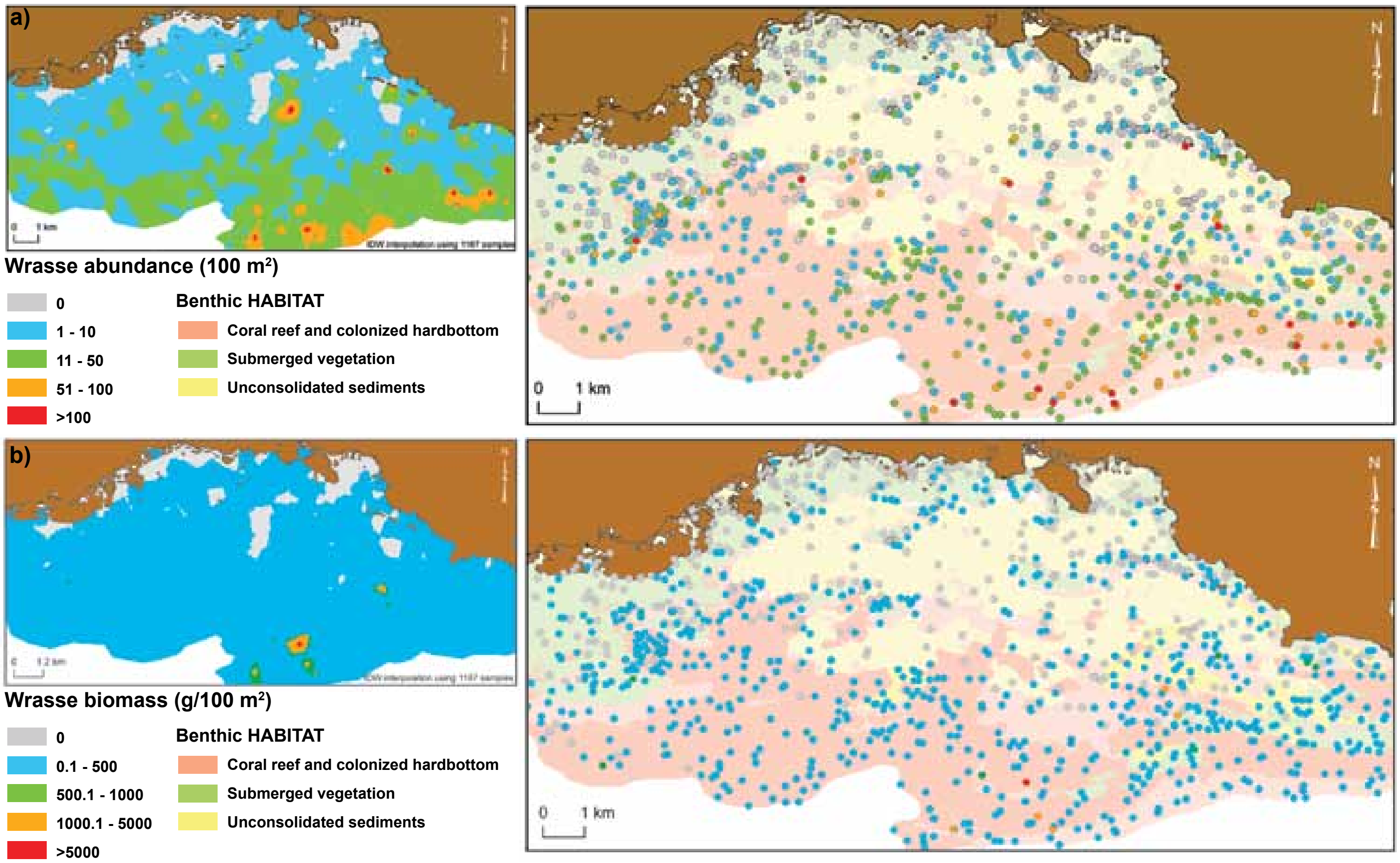


Figure 3.106. Maps of the interpolated (left map) and spatial (right map) distributions for wrasse (Labridae): (a) abundance and (b) biomass.

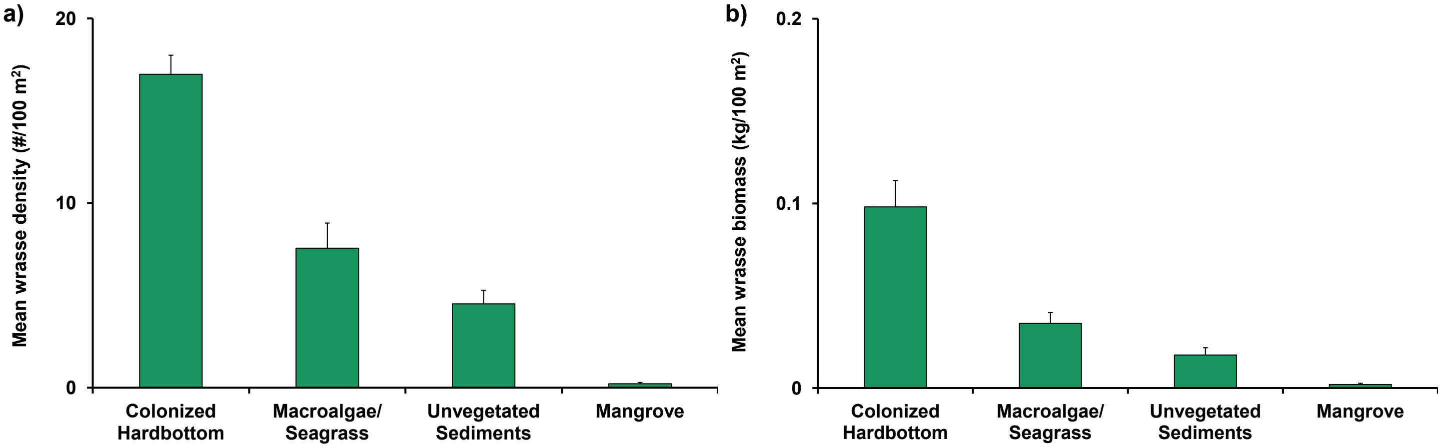


Figure 3.107. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for wrasse (Labridae).

Porgies (Sparidae)

Sparidae species were most abundant in fringing mangrove and mangrove islands, but were also found over colonized hardbottom habitat types across the mid- and outer shelf zones (Figures 3.108 and 3.109).

Saucereye porgy (*Calamus calamus*) in St. John, USVI

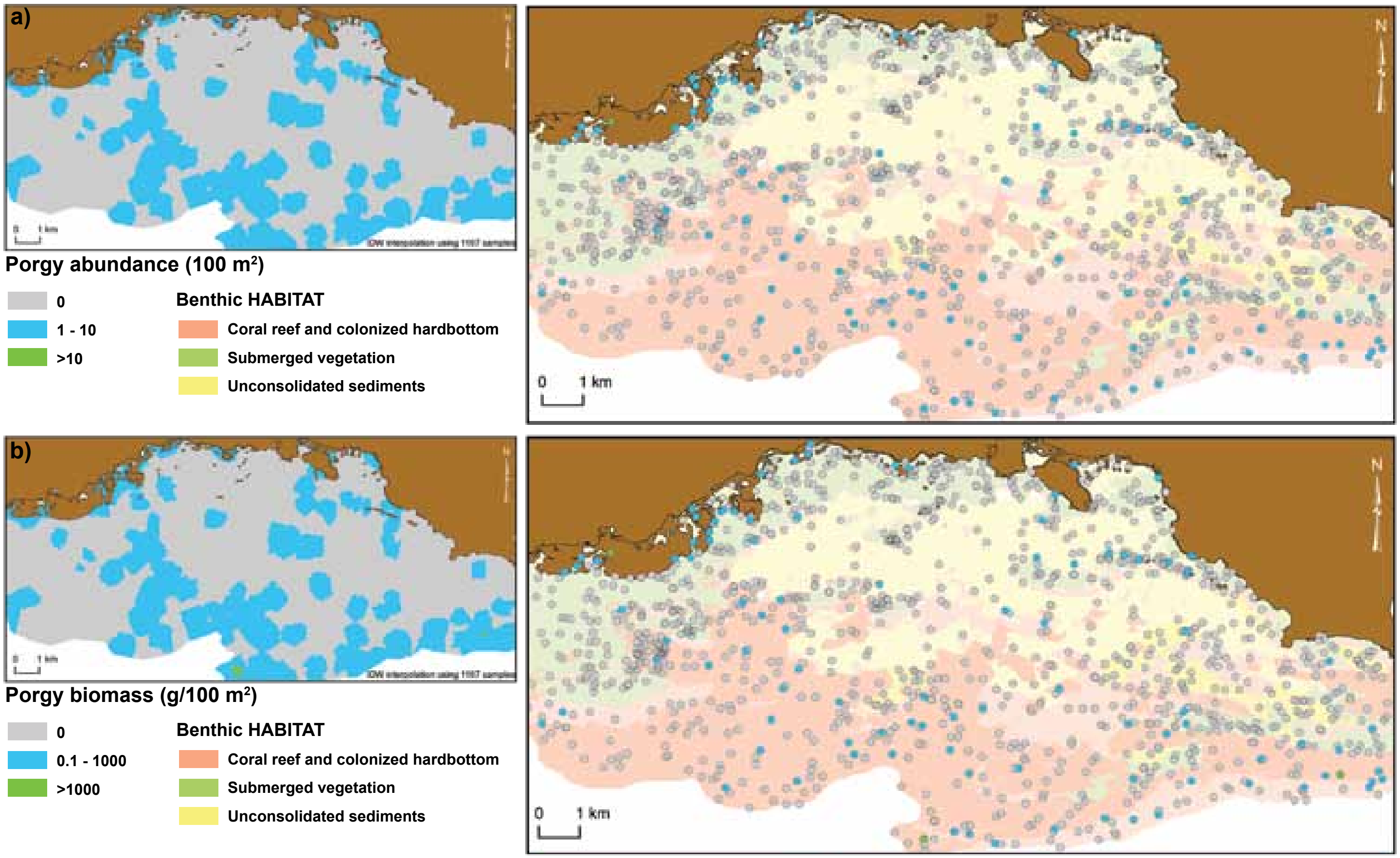


Figure 3.108. Maps of the interpolated (left map) and spatial (right map) distributions for porgy (Sparidae): (a) abundance and (b) biomass.

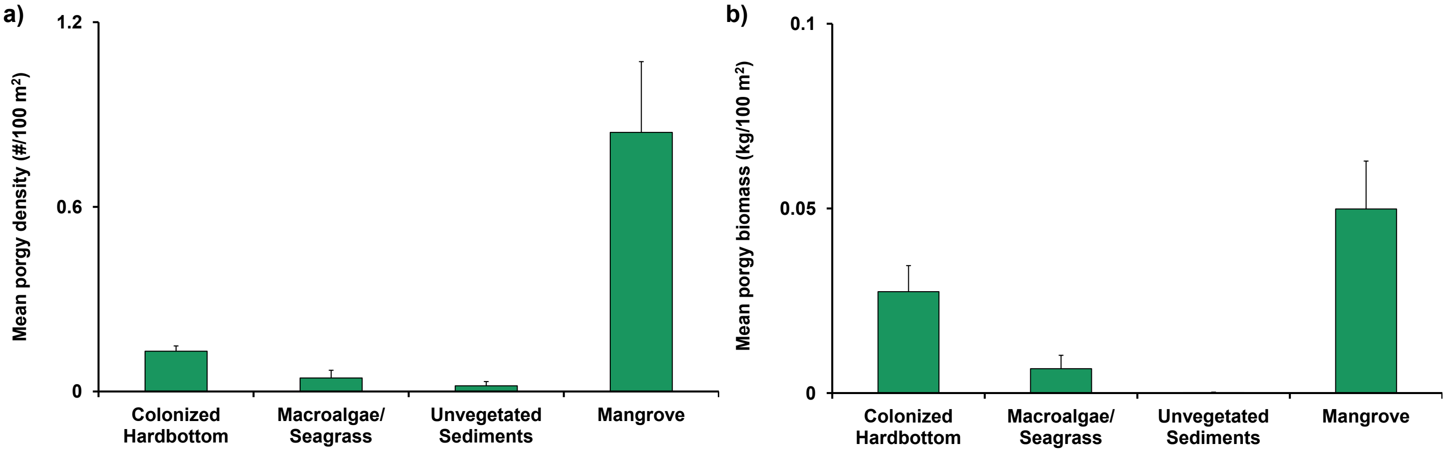


Figure 3.109. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for porgy (Labridae).

Other species

Sharks and Rays

Sharks and rays were comprised of four species including three *Dasyatis americana* (southern stingray), four *Ginglymostoma cirratum* (nurse shark) and one *Galeocerdo cuvier* (tiger shark).

Sightings occurred in all major habitat types, either in or close proximity to seagrass beds across the shelf (Figure 3.110 and 3.111). Highest mean biomass was recorded for seagrasses/macroalgal beds and was lowest over unvegetated sediments. Relatively high abundance, but low biomass for mangroves indicates that most sharks and rays in mangroves were smaller individuals than those sighted over seagrasses and colonized hardbottom. Insufficient sightings precluded a description of spatial distributions (Figure 3.110).

Nurse shark (*Ginglymostoma cirratum*)

Southern stingray (*Dasyatis americana*)

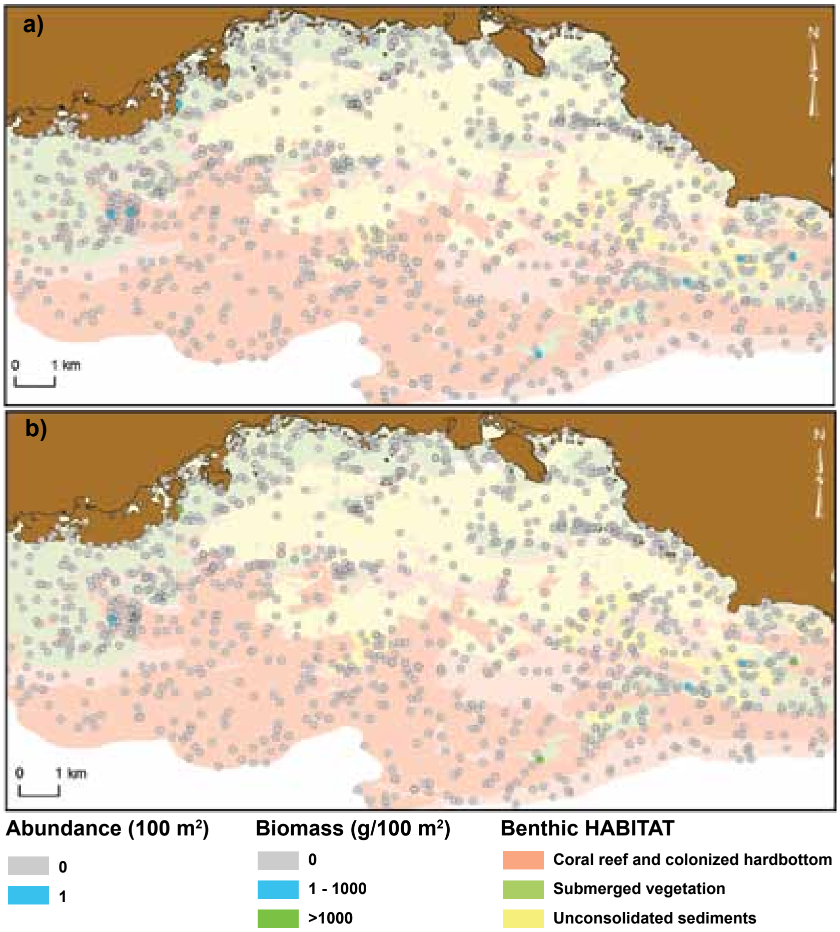


Figure 3.110. Map of the spatial distributions for sharks and rays: (a) abundance and (b) biomass.

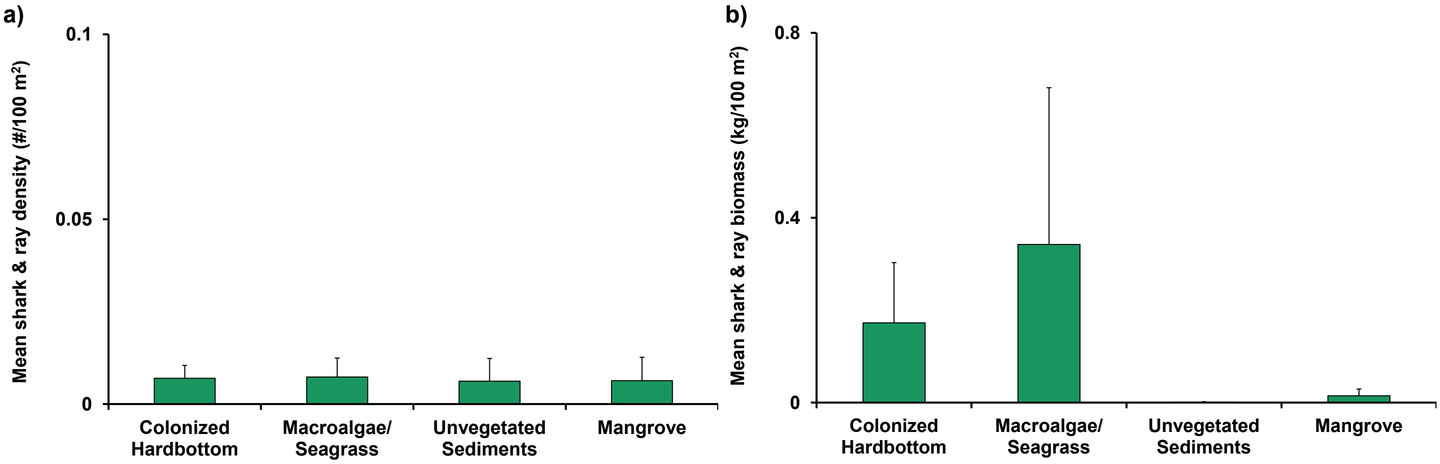


Figure 3.111. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for sharks and rays.

Queen triggerfish (*Balistes vetula*)

B. vetula was absent from nearshore and lagoonal areas, but was broadly distributed across colonized hardbottom habitat types of the mid and outer shelf zones (Figure 3.112). Highest abundance and biomass were recorded for topographically complex colonized pavement with sand channels. *B. vetula* was absent from mangroves and very few were observed over unvegetated sediments across the study area (Figure 3.113).

Queen triggerfish (*Balistes vetula*)

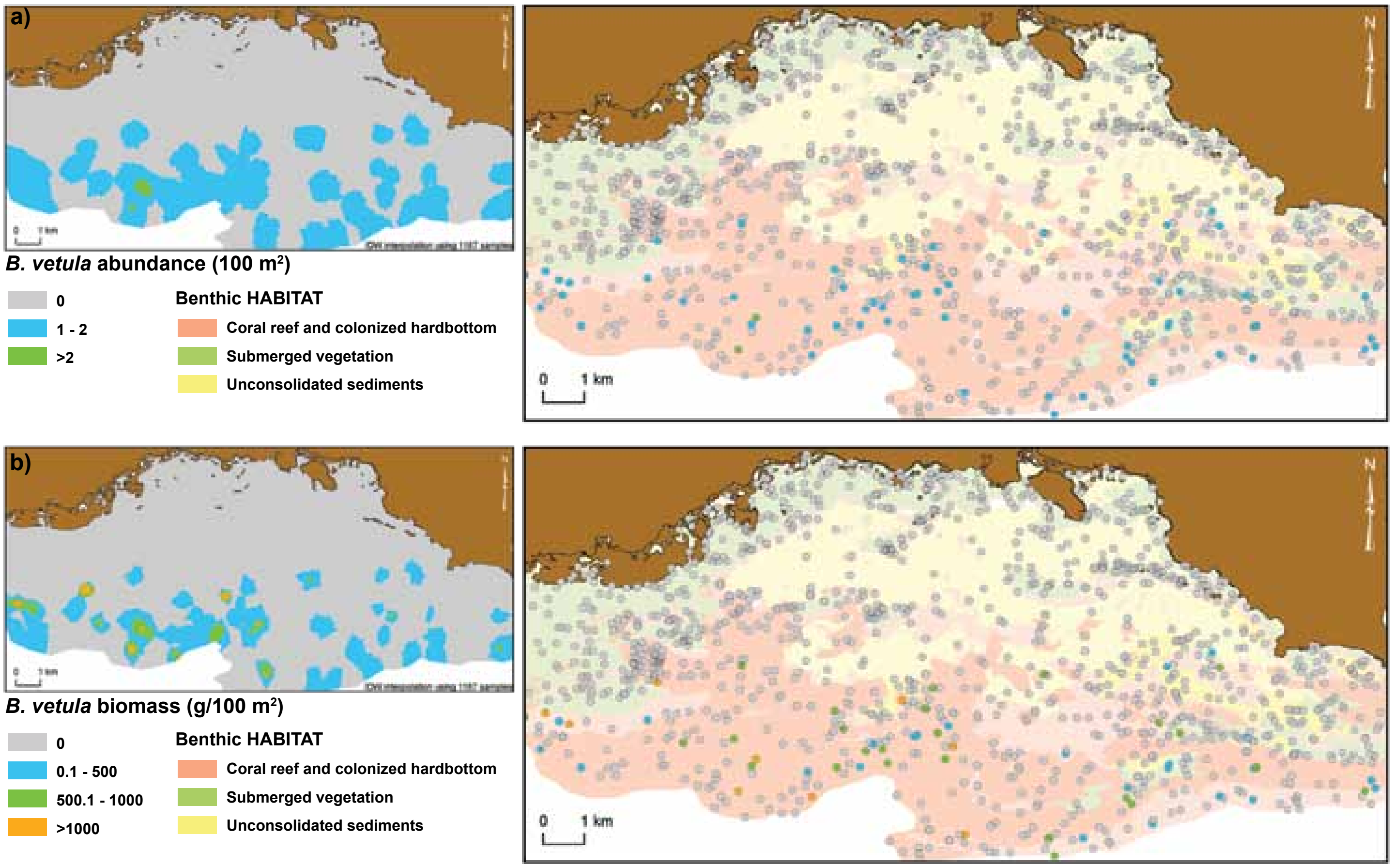


*Figure 3.112. Maps of the interpolated (left map) and spatial (right map) distributions for queen triggerfish (*B. vetula*): (a) abundance and (b) biomass.*

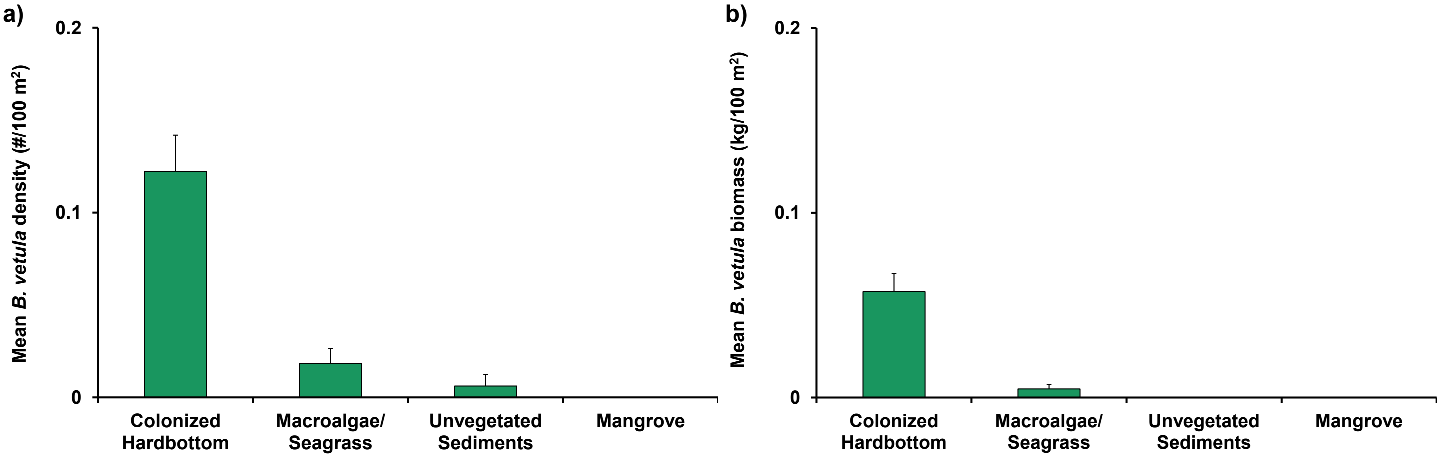


*Figure 3.113. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for queen triggerfish (*B. vetula*).*

Great barracuda (*Sphyraena barracuda*)

Highest mean abundance of *S. barracuda* was observed within both fringing mangroves of the mainland and mangrove cays (Figures 3.114a and 3.115a). Highest biomass, however, were calculated for colonized hardbottom sites in the mid and outer shelf zones and softbottom sites in close proximity to colonized hardbottom (Figures 3.114b and 3.115b).

Great barracuda (*Sphyraena barracuda*)

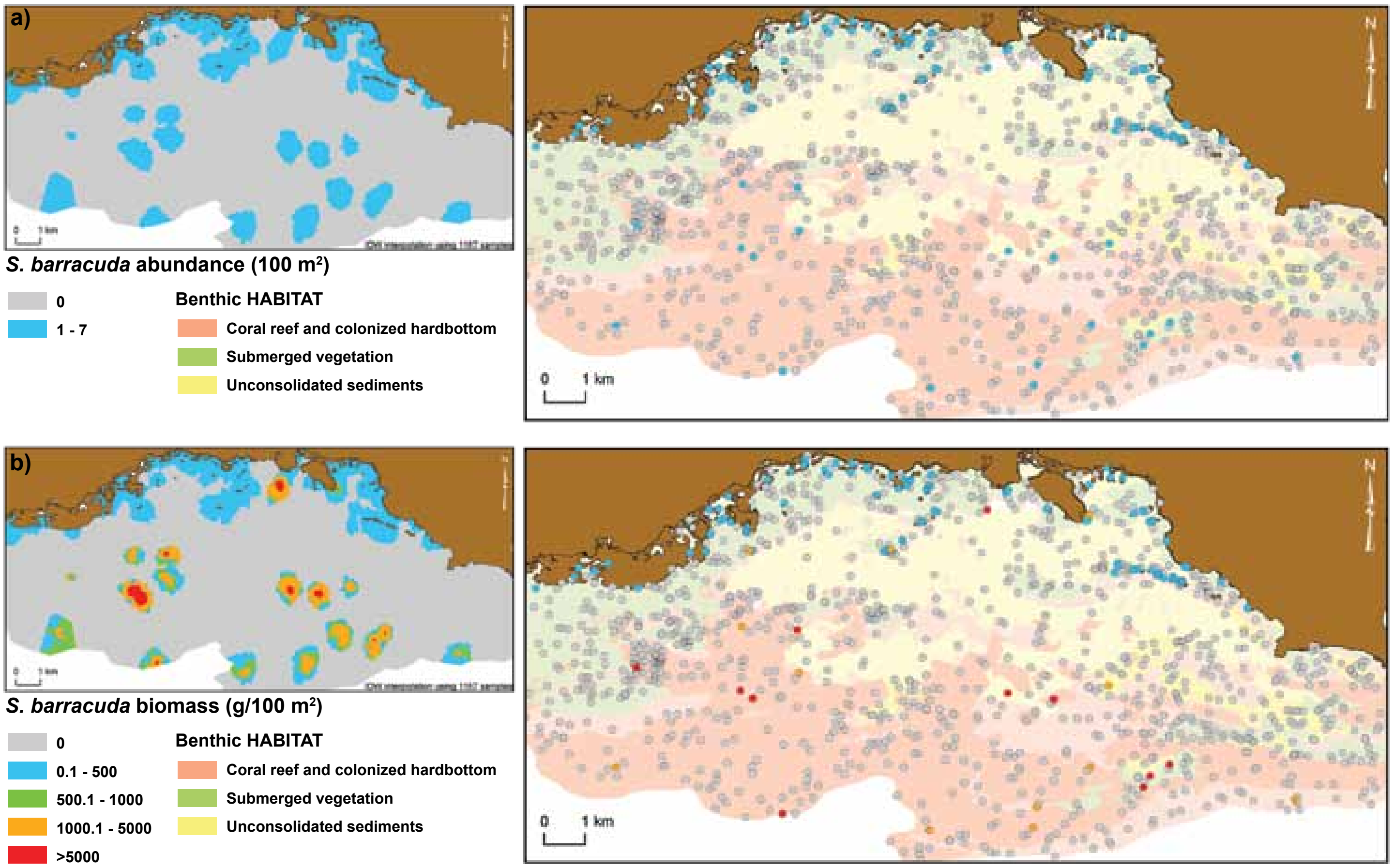


*Figure 3.114. Maps of the interpolated (left map) and spatial (right map) distributions for great barracuda (*S. barracuda*): (a) abundance and (b) biomass.*

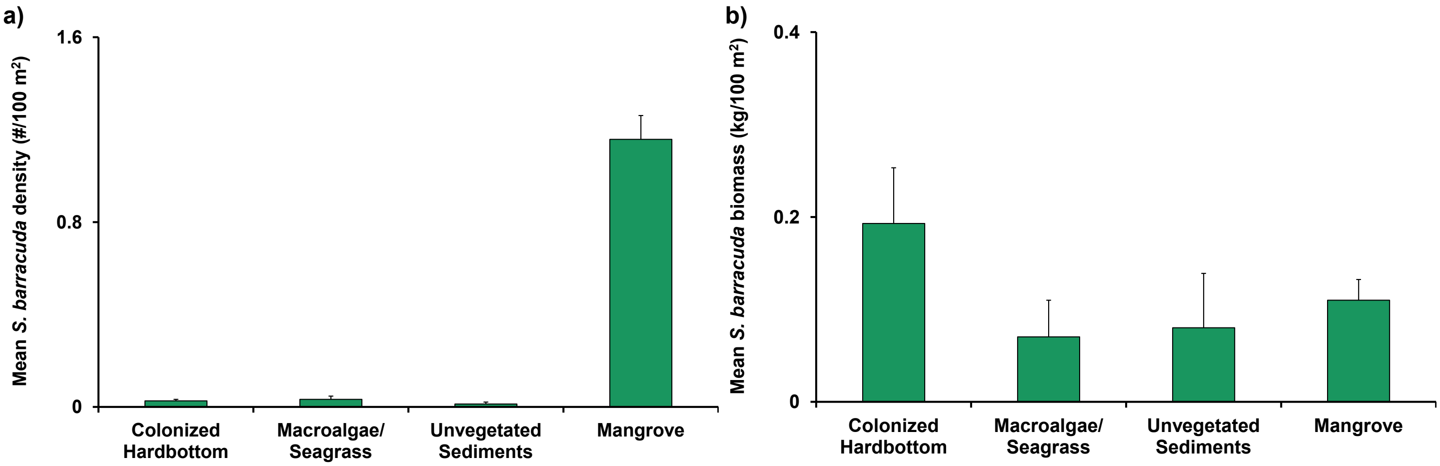


*Figure 3.115. Comparison of mean (± SE) density and biomass by substrate type in the southwest Puerto Rico study area for great barracuda (*S. barracuda*).*

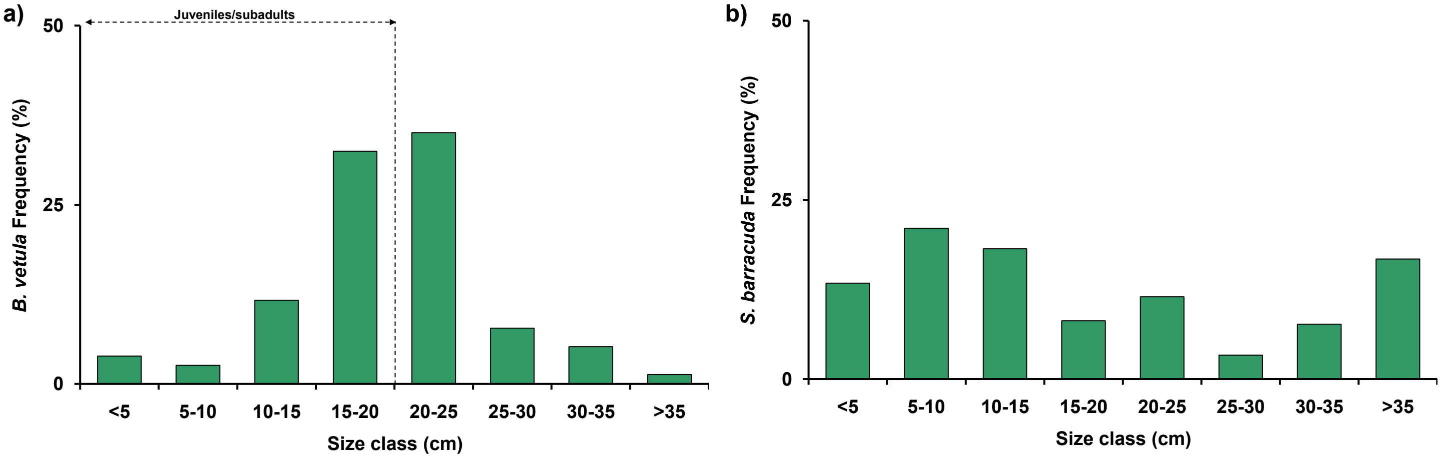


*Figure 3.116. Size frequency histogram for (a) queen triggerfish (B. vetula) and (b) great barracuda (*S. barracuda*) in the southwest Puerto Rico study area.*

The largest proportion of the *B. vetula* population in the study area were subadults and small adults, with only a small proportion (5%) of the total comprising small juveniles and large adults were very infrequent (Figure 3.116a). The largest individual was estimated at 40 cm FL.

S. barracuda reach sexual maturity at a minimum size of 66 cm FL, therefore juveniles represented the majority of the population surveyed in the study (Figure 3.116b). The maximum known size for the species is 200 cm TL and the largest in the study area was 150 cm FL.

3.4. Inter-annual trend in fish metrics (2001-2007)

Presented in this section is a synoptic overview of inter-annual changes in summary statistics (mean ± SE) for 81 fish metrics at the level of species, family, trophic and community using data collected from the southwest Puerto Rico study area from 2001-2007. Between-year comparisons were tested for statistical significance using a non-parametric test (see Methods). The following results are based on values obtained from Table 3.5, where density values are reported as # individuals/100m^2, biomass reported in grams/100m^2, and richness (community metrics section only) reported as # species/100m^2.

Table 3.5. Density and biomass (mean ± SE) for selected fish species and families (2001-2007) for the southwest Puerto Rico study area. Colored text indicate significant (p<0.05) directional change from previous year. BLUE= increasing; RED= decreasing. Asterisk (*) indicate there was a significant difference when years were compared.

	2001	2002	2003	2004	2005	2006	2007
Fish variable	**Mean (SE)**	**Mean (SE)**	**Mean (SE)**	**Mean (SE)**	**Mean (SE)**	**Mean (SE)**	**Mean (SE)**
Community metrics							
Number of species *	10.41 (0.77)	10.64 (0.55)	13.60 (0.58)	12.27 (0.53)	12.91 (0.58)	13.02 (0.61)	13.48 (0.62)
Total density (100 m^2) *	234.46 (46.68)	452.14 (122.44)	262.71 (77.61)	130.39 (29.20)	117.63 (19.02)	181.20 (61.01)	90.24 (9.07)
Biomass (g/100 m^2) *	1566.43 (246.60)	2850.21 (586.55)	2379.88 (275.44)	1661.84 (162.41)	2668.70 (347.93)	2690.22 (404.56)	2394.64 (232.87)
Herbivore density *	28.51 (3.16)	26.31 (2.28)	38.57 (2.64)	27.99 (2.05)	34.15 (2.71)	35.05 (2.73)	36.92 (2.87)
Herbivore biomass *	531.85 (110.30)	785.43 (122.25)	1091.05 (198.65)	830.05 (102.76)	1157.45 (158.35)	1125.46 (136.84)	1219.35 (172.45)
Piscivore density	3.37 (0.57)	4.63 (0.90)	4.30 (0.57)	3.15 (0.43)	4.37 (0.84)	6.72 (1.62)	3.97 (0.78)
Piscivore biomass	348.71 (85.19)	1065.54 (540.54)	546.22 (113.99)	327.79 (62.15)	760.95 (251.86)	527.03 (161.21)	399.50 (92.68)
Planktivore density	177.42 (44.63)	399.60 (121.44)	197.40 (76.74)	83.04 (27.93)	61.91 (17.73)	121.32 (59.80)	31.31 (8.24)
Planktivore biomass	153.94 (39.52)	215.01 (34.93)	210.35 (51.25)	139.65 (30.09)	258.23 (58.91)	188.76 (32.02)	171.17 (40.05)
Groupers - Serranidae							
Grouper density *	0.14 (0.05)	0.19 (0.05)	0.32 (0.06)	0.31 (0.06)	0.34 (0.07)	0.44 (0.08)	0.46 (0.08)
Grouper biomass *	45.97 (25.44)	33.62 (11.62)	36.82 (10.30)	59.75 (14.27)	42.59 (10.31)	30.35 (6.43)	44.89 (11.15)
Graysby (*C. cruentata*) density *	0.07 (0.03)	0.10 (0.03)	0.23 (0.04)	0.18 (0.04)	0.22 (0.05)	0.31 (0.06)	0.31 (0.06)
Graysby (*C. cruentata*) biomass *	4.04 (2.69)	8.73 (3.15)	15.45 (3.72)	19.13 (5.25)	11.33 (2.88)	13.38 (3.44)	16.40 (4.18)
Coney (*C. fulva*) density	0.01 (0.01)	0.06 (0.04)	0.07 (0.03)	0.07 (0.03)	0.08 (0.03)	0.08 (0.03)	0.10 (0.05)
Coney (*C. fulva*) biomass	0.91 (0.91)	11.33 (7.31)	8.50 (4.10)	11.52 (4.84)	16.19 (6.67)	9.42 (3.62)	11.76 (5.29)
Red hind (*E. guttatus*) density	0.04 (0.02)	0.04 (0.01)	0.02 (0.01)	0.05 (0.02)	0.05 (0.02)	0.02 (0.01)	0.03 (0.01)
Red hind (*E. guttatus*) biomass	14.11 (7.46)	13.56 (5.60)	12.87 (7.68)	28.64 (11.03)	15.07 (6.27)	5.20 (2.96)	11.11 (4.92)
Snappers - Lutjanidae							
Snapper density	2.41 (0.46)	3.38 (0.57)	2.85 (0.39)	2.24 (0.37)	2.00 (0.41)	5.32 (1.52)	3.39 (0.77)
Snapper biomass	162.97 (38.34)	286.98 (51.73)	212.25 (36.24)	145.93 (23.10)	195.71 (45.38)	245.74 (64.60)	209.61 (50.25)
Schoolmaster (*L. apodus*) density	1.31 (0.34)	2.25 (0.43)	1.80 (0.34)	1.22 (0.27)	1.12 (0.32)	2.22 (0.75)	1.73 (0.67)
Schoolmaster (*L. apodus*) biomass	90.80 (28.92)	155.65 (37.19)	77.99 (17.70)	59.13 (15.35)	71.26 (26.66)	85.08 (26.12)	82.01 (43.96)
Gray (*L. griseus*) density	0.27 (0.10)	0.46 (0.19)	0.22 (0.08)	0.49 (0.15)	0.18 (0.09)	1.88 (0.92)	0.76 (0.25)
Gray (*L. griseus*) biomass	19.51 (7.79)	46.62 (20.43)	40.27 (23.70)	33.29 (11.34)	46.68 (24.74)	92.97 (44.22)	46.26 (17.40)
Dog (*L. jocu*) density *	0.00 (0.00)	0.00 (0.00)	0.02 (0.01)	0.00 (0.00)	0.01 (0.01)	0.01 (0.01)	0.04 (0.02)
Dog (*L. jocu*) biomass *	0.00 (0.00)	0.00 (0.00)	6.91 (4.89)	0.00 (0.00)	2.02 (2.02)	1.92 (1.92)	12.87 (6.29)
Mahogany (*L. mahogoni*) density	0.02 (0.02)	0.04 (0.02)	0.02 (0.01)	0.01 (0.01)	0.14 (0.06)	0.04 (0.02)	0.04 (0.03)
Mahogany (*L. mahogoni*) biomass	0.20 (0.20)	1.11 (0.68)	5.01 (2.95)	0.80 (0.62)	3.21 (1.24)	3.21 (1.61)	4.64 (3.20)
Lane (*L. synagris*) density *	0.22 (0.08)	0.09 (0.03)	0.09 (0.03)	0.02 (0.01)	0.04 (0.02)	0.02 (0.01)	0.13 (0.07)
Lane (*L. synagris*) biomass *	6.57 (4.21)	3.66 (1.50)	7.02 (3.05)	1.49 (1.48)	3.69 (2.92)	0.64 (0.57)	2.65 (1.39)
Yellowtail (*O. chrysurus*) density *	0.58 (0.16)	0.54 (0.13)	0.66 (0.11)	0.48 (0.09)	0.51 (0.09)	0.97 (0.16)	0.66 (0.12)
Yellowtail (*O. chrysurus*) biomass *	45.89 (12.72)	79.94 (22.30)	56.42 (13.14)	49.03 (11.74)	68.66 (15.04)	61.56 (10.94)	61.12 (11.04)
Jacks - Carangidae							
Jack density	0.88 (0.30)	1.23 (0.71)	1.71 (0.66)	0.75 (0.27)	3.32 (1.77)	2.86 (1.13)	0.28 (0.08)
Jack biomass	68.41 (43.43)	146.55 (75.20)	147.43 (79.48)	92.85 (45.91)	67.72 (28.87)	99.78 (36.61)	41.97 (28.16)
Blue runner (*C. crysos*) density	0.15 (0.15)	0.10 (0.07)	0.11 (0.08)	0.15 (0.11)	0.01 (0.01)	0.02 (0.01)	0.05 (0.04)
Blue runner (*C. crysos*) biomass	3.45 (3.45)	43.41 (33.41)	50.78 (39.12)	65.42 (43.22)	2.29 (2.29)	21.24 (17.93)	29.34 (27.25)
Horse-eye (*C. latus*) density *	0.25 (0.16)	0.01 (0.01)	0.02 (0.02)	0.03 (0.02)	0.02 (0.01)	0.01 (0.01)	0.00 (0.00)
Horse-eye (*C. latus*) biomass *	18.63 (15.34)	0.43 (0.43)	0.84 (0.66)	0.15 (0.14)	1.05 (0.86)	0.21 (0.21)	0.00 (0.00)
Bar (*C. ruber*) density	0.27 (0.09)	0.30 (0.12)	0.92 (0.38)	0.34 (0.12)	1.63 (0.66)	1.32 (0.59)	0.21 (0.06)
Bar (*C. ruber*)) biomass	4.69 (1.58)	11.65 (7.47)	10.78 (2.46)	9.40 (3.73)	24.16 (9.35)	15.40 (6.03)	5.37 (2.76)

Table 3.5. Continued...

Fish variable	2001 Mean (SE)	2002 Mean (SE)	2003 Mean (SE)	2004 Mean (SE)	2005 Mean (SE)	2006 Mean (SE)	2007 Mean (SE)
Parrotfish - Scaridae							
Parrotfish density *	11.54 (1.58)	12.22 (1.26)	19.56 (1.55)	12.40 (1.09)	12.59 (1.76)	15.51 (1.36)	17.47 (1.70)
Parrotfish biomass *	232.37 (50.12)	461.23 (78.82)	568.80 (79.61)	543.23 (71.00)	692.15 (99.02)	679.57 (82.55)	747.38 (129.55)
Striped (*S. iseri*) density	6.25 (1.20)	5.71 (0.83)	7.27 (0.96)	4.69 (0.63)	5.02 (1.35)	5.98 (0.77)	6.79 (1.34)
Striped (*S. iseri*) biomass	70.23 (15.88)	95.85 (23.39)	98.67 (17.14)	84.67 (12.00)	101.00 (14.69)	153.73 (25.73)	149.00 (26.30)
Princess (*S. taeniopterus*) density *	0.19 (0.11)	0.49 (0.16)	1.97 (0.65)	1.45 (0.32)	1.96 (0.41)	2.48 (0.41)	2.89 (0.46)
Princess (*S. taeniopterus*) biomass *	9.38 (6.85)	31.69 (7.85)	138.51 (53.87)	78.66 (20.46)	111.14 (20.29)	131.64 (24.48)	113.80 (19.11)
Greenblotch (*S. atomarium*) density *	0.27 (0.10)	0.72 (0.24)	0.73 (0.17)	0.54 (0.09)	0.13 (0.05)	0.30 (0.09)	0.23 (0.09)
Greenblotch (*S. atomarium*) biomass *	0.24 (0.11)	4.07 (2.48)	0.54 (0.15)	0.56 (0.14)	0.37 (0.31)	0.33 (0.11)	0.36 (0.23)
Redband (*S. aurofrenatum*) density *	0.90 (0.23)	2.10 (0.32)	2.46 (0.30)	2.65 (0.30)	2.97 (0.32)	4.27 (0.43)	4.08 (0.43)
Redband (*S. aurofrenatum*) biomass *	44.18 (9.63)	125.50 (22.47)	115.91 (17.71)	151.28 (20.03)	147.26 (19.24)	163.11 (20.92)	132.79 (15.49)
Bucktooth (*S. radians*) density *	1.90 (0.41)	1.24 (0.22)	3.45 (0.63)	1.29 (0.30)	0.64 (0.19)	0.38 (0.12)	0.99 (0.29)
Bucktooth (*S. radians*) biomass *	3.44 (0.98)	3.97 (1.00)	8.93 (2.46)	2.74 (1.17)	1.36 (0.71)	0.67 (0.26)	1.16 (0.42)
Stoplight (*S. viride*) density	1.02 (0.27)	0.99 (0.18)	1.50 (0.25)	1.11 (0.21)	1.03 (0.15)	0.99 (0.17)	1.18 (0.19)
Stoplight (*S. viride*) biomass	86.59 (28.51)	173.97 (42.13)	160.52 (34.59)	213.44 (43.77)	265.56 (63.36)	148.07 (28.99)	282.30 (83.74)
Grunts - Haemulidae							
Grunt density	15.91 (4.03)	14.65 (2.98)	11.70 (2.07)	12.88 (2.98)	13.88 (3.62)	13.34 (5.42)	12.82 (4.75)
Grunt biomass	192.59 (86.01)	285.97 (72.06)	203.07 (62.32)	89.92 (16.28)	150.64 (37.44)	164.42 (40.39)	184.09 (43.56)
Tomtate (*H. aurolineatum*) density *	0.65 (0.60)	0.09 (0.05)	2.34 (0.90)	2.65 (0.91)	0.18 (0.10)	0.51 (0.29)	5.08 (4.19)
Tomtate (*H. aurolineatum*) biomass *	2.28 (1.46)	5.33 (2.40)	58.20 (42.48)	10.67 (4.54)	4.73 (2.28)	9.90 (3.52)	30.26 (22.40)
French (*H. flavolineatum*) density	9.26 (2.95)	7.64 (2.26)	4.18 (1.40)	3.46 (1.31)	4.17 (1.53)	2.11 (0.67)	1.98 (0.57)
French (*H. flavolineatum*) biomass	123.68 (61.36)	98.45 (29.42)	40.83 (10.95)	36.24 (10.39)	56.95 (22.64)	31.97 (8.51)	34.96 (11.71)
White (*H. plumierii*) density *	0.35 (0.20)	0.20 (0.06)	1.64 (0.61)	0.65 (0.30)	0.59 (0.17)	0.99 (0.39)	0.44 (0.14)
White (*H. plumierii*) biomass *	8.86 (3.99)	9.11 (3.83)	32.18 (10.02)	5.50 (2.07)	28.40 (9.87)	67.90 (28.49)	29.22 (7.30)
Bluestriped (*H. sciurus*) density *	3.33 (0.96)	4.63 (1.08)	2.34 (0.56)	0.83 (0.20)	0.94 (0.37)	0.97 (0.39)	0.99 (0.43)
Bluestriped (*H. sciurus*) biomass *	52.42 (25.91)	159.56 (52.29)	59.22 (16.26)	28.36 (6.36)	43.58 (19.29)	40.76 (12.72)	66.43 (27.09)
Surgeonfish - Acanthuridae							
Surgeonfish density *	2.46 (0.48)	1.91 (0.32)	4.39 (0.71)	2.44 (0.28)	3.13 (0.36)	4.13 (0.66)	3.52 (0.51)
Surgeonfish biomass *	174.42 (61.08)	153.40 (35.74)	369.61 (141.82)	161.71 (26.79)	265.21 (40.17)	309.24 (56.32)	311.40 (64.73)
Ocean surgeon (*A. bahianus*) density *	1.33 (0.26)	1.09 (0.19)	2.29 (0.27)	1.25 (0.19)	1.89 (0.25)	2.79 (0.41)	1.83 (0.24)
Ocean surgeon (*A. bahianus*) biomass *	76.42 (23.07)	76.32 (18.87)	121.62 (20.14)	78.49 (13.03)	139.33 (22.76)	201.58 (27.99)	126.72 (17.10)
Doctorfish (*A. chirurgus*) density *	0.25 (0.09)	0.36 (0.13)	1.01 (0.31)	0.55 (0.12)	0.49 (0.11)	0.25 (0.08)	0.73 (0.29)
Doctorfish (*A. chirurgus*) biomass *	13.82 (5.88)	39.81 (20.94)	99.84 (39.18)	42.92 (18.93)	38.65 (12.71)	25.15 (10.72)	103.89 (53.90)
Blue tang (*A. coeruleus*) density *	0.88 (0.36)	0.45 (0.14)	1.05 (0.31)	0.64 (0.13)	0.75 (0.16)	1.07 (0.37)	0.93 (0.25)
Blue tang (*A. coeruleus*) biomass *	84.17 (53.65)	37.28 (12.44)	148.15 (97.84)	40.29 (8.67)	87.24 (20.37)	82.50 (34.03)	80.80 (23.90)
Other Families							
Squirrelfish (Holocentridae) density *	1.85 (0.81)	0.76 (0.18)	0.72 (0.12)	0.54 (0.10)	0.65 (0.09)	1.18 (0.17)	0.97 (0.15)
Squirrelfish (Holocentridae) biomass *	107.47 (50.94)	79.52 (20.56)	68.96 (13.26)	59.90 (10.48)	65.30 (10.20)	99.11 (14.52)	85.95 (18.27)
Wrasse (Labridae) density *	8.87 (3.12)	8.84 (1.50)	11.66 (1.61)	8.18 (1.20)	10.56 (1.65)	13.69 (1.86)	12.67 (1.36)
Wrasse (Labridae) biomass *	50.02 (12.42)	59.96 (10.87)	63.92 (13.48)	28.30 (4.36)	98.59 (44.60)	47.17 (5.83)	64.04 (7.44)
Other species							
Queen triggerfish (*B. vetula*) density *	0.03 (0.02)	0.05 (0.02)	0.02 (0.01)	0.02 (0.01)	0.03 (0.01)	0.12 (0.03)	0.18 (0.05)
Queen triggerfish (*B. vetula*) biomass *	18.46 (10.98)	26.60 (14.48)	7.62 (5.21)	4.18 (2.40)	22.05 (15.27)	48.81 (13.38)	71.73 (18.08)
Yellow goatfish (*M. martinicus*) density	0.54 (0.33)	0.14 (0.05)	0.13 (0.05)	0.08 (0.03)	0.19 (0.08)	0.30 (0.18)	0.71 (0.56)
Yellow goatfish (*M. martinicus*) biomass	40.64 (22.33)	19.28 (7.33)	28.48 (13.10)	7.48 (4.19)	31.44 (19.12)	17.20 (6.91)	31.72 (15.21)
Spotted goatfish (*P. maculatus*) density	0.28 (0.07)	0.74 (0.37)	0.36 (0.12)	0.13 (0.03)	0.31 (0.08)	0.33 (0.06)	0.29 (0.05)
Spotted goatfish (*P. maculatus*) biomass	13.55 (4.05)	29.82 (10.99)	18.29 (5.49)	5.57 (1.77)	17.23 (4.85)	13.72 (2.98)	16.89 (3.76)

Overall Comparisons

None of the fish metrics decreased for more than three consecutive years within the seven year monitoring period (2001-2007). Fifteen fish metrics (approximately 19%) exhibited at least three consecutive years of decline and 44 fish metrics (approximately 54%) exhibited at least two consecutive years of decline. All but one of the 15 metrics exhibiting a three year consecutive decline occurred prior to 2006. In contrast, 13 of the 15 metrics exhibiting a three year consecutive increase occurred from 2004-2007. Two fish metrics (*C. fulva* density and *S. aurofrenatum* density) exhibited more than three years of consecutive increase. Mean *C. fulva* density increased very slightly every year from 2001-2007, but was not statistically significant. Mean *S. aurofrenatum* density increased for five consecutive years from 2001-2006 and although not greater in 2007 than 2006, mean density remained significantly higher than it was in years 2001 and 2002.

A total of 65 (approximately 80%) metrics decreased from 2003 to 2004, five of which decreased significantly ($p<0.05$); followed by 70% of the metrics increasing the following year (2004-2005), with no significant increase. There were 108 (approximately 62%) significant ($p<0.05$) increases as time progressed and 66 (approximately 38%) significant decreases. The highest number of significant differences (n=19) was found when comparing fish metrics between the first two years and the last year of the monitoring period between 2001 and 2007 and 2002 and 2007. Fourteen metrics were significantly higher in 2007 than 2001 and five were significantly lower; and 16 metrics were significantly higher in 2007 than 2002 and three were significantly lower. A summarized subset of metrics from Table 3.5 with significant differences in 2001/2007 and 2002/2007 comparisons are shown in Tables 3.6 and 3.7.

S. radians, *A. chirurgus*, *H. flavolineatum* and squirrelfish (Holocentridae) were the only metrics to have three consecutive years of decline for both metrics of mean density and biomass. *S. radians* and *A. chirurgus* declined from 2003-2006 and *H. flavolineatum* and squirrelfish from 2001-2003. The other metrics that exhibited three years of consecutive decline include; total fish, planktivore, total snapper and *L. apodus* mean density from 2002-2005; and *S. radians* density and biomass, *S. viride* density and *A. chirurgus* mean biomass from 2003-2006.

Table 3.6. Inter-annual metrics with significant differences ($p<0.05$) in 2001 and 2007 comparisons.

Significantly higher in 2001	Significantly higher in 2007
Grouper biomass	Species richness
Sparisoma radians abundance	Total biomass
Sparisoma radians biomass	Herbivore biomass
Caranx latus abundance	Grouper abundance
Caranx latus biomass	*Cephalopholis cruentata* abundance
	Cephalopholis cruentata biomass
	Wrasse abundance
	Parrotfish abundance
	Parrotfish biomass
	Scarus taeniopterus biomass
	Sparisoma aurofrenatum abundance
	Sparisoma aurofrenatum biomass
	Balistes vetula abundance
	Balistes vetula biomass

Table 3.7. Inter-annual metrics with significant differences ($p<0.05$) in 2002 and 2007 comparisons.

Significantly higher in 2002	Significantly higher in 2007
Haemulon sciurus abundance	Total species richness
Haemulon sciurus biomass	Grouper abundance
Sparisoma radians biomass	*Cephalopholis cruentata* abundance
	Cephalopholis cruentata biomass
	Lutjanus jocu abundance
	Lutjanus jocu biomass
	Scarus taeniopterus abundance
	Scarus taeniopterus biomass
	Sparisoma aurofrenatum abundance
	Sparisoma aurofrenatum biomass
	Surgeonfish abundance
	Surgeonfish biomass
	Acanthurus bahianus biomass
	Wrasse abundance
	Balistes vetula abundance
	Balistes vetula biomass

3.4.1. Fish community metrics

A summarized subset of community metrics from Table 3.5 is presented in Table 3.8.

Table 3.8. Density and biomass (mean ± SE) for community metrics (2001-2007) for the southwest Puerto Rico study area. Asterisks (*) indicate there was a significant difference when years were compared. Values have been rounded to one decimal place up from Table 3.5 where appropriate. Bold indicates either the lowest or highest mean for that metric.

Community variable	2001 Mean (SE)	2002 Mean (SE)	2003 Mean (SE)	2004 Mean (SE)	2005 Mean (SE)	2006 Mean (SE)	2007 Mean (SE)
Number of species *	**10.4 (0.8)**	10.6 (0.6)	**13.6 (0.6)**	12.3 (0.5)	12.9 (0.6)	13.0 (0.6)	13.5 (0.6)
Total density (100 m^2) *	234.5 (46.7)	**452.1 (122.4)**	262.7 (77.6)	130.4 (29.2)	117.6 (19.0)	181.2 (61.0)	**90.2 (9.1)**
Biomass (g/100 m^2) *	**1,566.4 (246.6)**	**2,850.2 (586.6)**	2,379.9 (275.4)	1,661.8 (162.4)	2,668.7 (347.9)	2,690.2 (404.6)	2,394.6 (232.9)
Herbivore density *	28.5 (3.2)	**26.3 (2.3)**	**38.6 (2.6)**	28.0 (2.1)	34.1 (2.7)	35.1 (2.7)	36.9 (2.9)
Herbivore biomass *	**531.9 (110.3)**	785.4 (122.3)	1,091.0 (198.6)	830.0 (102.8)	1,157.5 (158.3)	1,125.5 (136.8)	**1,219.3 (172.5)**
Piscivore density	3.4 (0.6)	4.6 (0.9)	4.3 (0.6)	**3.1 (0.4)**	4.4 (0.8)	**6.7 (1.6)**	4.0 (0.8)
Piscivore biomass	348.7 (85.2)	**1,065.5 (540.5)**	546.2 (114.0)	**327.8 (62.2)**	760.9 (251.9)	527.0 (161.2)	399.5 (92.7)
Planktivore density	177.4 (44.6)	**399.6 (121.4)**	197.4 (76.7)	83.0 (27.9)	61.9 (17.7)	121.3 (59.8)	**31.3 (8.2)**
Planktivore biomass	153.9 (39.5)	215.0 (34.9)	210.4 (51.3)	**139.6 (30.1)**	**258.2 (58.9)**	188.8 (32.0)	171.2 (40.0)

Mean species richness increased from 2001-2003 followed by a decline in 2003-2004 and then increased for a period of three consecutive years (2004-2007). Mean species richness was significantly ($p=0.0003$) lower in 2001 than 2003 and 2007, and in 2002 than 2007. Mean fish density (all species combined) was highest in 2002 and lowest in 2007 and average total fish biomass was lowest in 2001 and highest the following year in 2002. Neither total fish density nor biomass had a significant difference between years of high and low means.

Mean density of herbivores exhibited three consecutive years of increase (2004-2007) and mean biomass of herbivores increased over two consecutive years (2001-2002) and then exhibited alternating directional change from 2002-2007. Mean herbivore density recorded from 2001-2007 was lowest in 2002 (26.31 ± 2.28) and highest the following year in 2003 (38.57 ± 2.64). Mean biomass was lowest in 2001 (531.85 ± 110.30) and highest in 2007 (1219.35 ± 172.45). The high and low means for both herbivore metrics were significantly different (p<0.01).

Both mean piscivore density and biomass recorded from 2001-2007 was lowest in 2004, whereas mean density was highest in 2006 and mean biomass in 2002. Mean planktivore density was highest in 2002 and lowest in 2007 and mean biomass was lowest in 2004 and highest the following year in 2005. Neither piscivore nor planktivore metrics were significantly different between years of highest and lowest.


Checkered puffer (*Sphoeroides testudineus*)


Squirrelfish (*Holocentrus adscensionis*)


Blackbar soldierfish (*Myripristis jacobus*)

3.4.2. Taxonomic groups

Large-body Groupers (Serranidae)

The species in this section include all large-bodied species from the Serranidae Family, Subfamily Epinephelinae (*Cephalopholis*, *Epinephelus*, *Mycteroperca* genus) that were recorded in the southwest Puerto Rico study area from 2001-2007. Grouper species included in the 'total grouper' metrics include *C. cruentata*, *C. fulva*, *Epinephelus adscensionis* (rock hind), *E. guttatus* and *Epinephelus striatus* (Nassau grouper). The mean density and biomass for the most common grouper species recorded/ observed in the study area are summarized in Table 3.5 and subset Table 3.9.

Table 3.9. Density and biomass (mean ± SE) for selected grouper species (2001-2007) for the southwest Puerto Rico study area. Asterisks (*) indicate there was a significant difference when years were compared. Values have been rounded to one decimal place up from Table 3.5 where appropriate. Bold indicates either the lowest or highest mean for that metric.

Grouper variable	2001 Mean (SE)	2002 Mean (SE)	2003 Mean (SE)	2004 Mean (SE)	2005 Mean (SE)	2006 Mean (SE)	2007 Mean (SE)
Grouper (Serranidae) density *	**0.1 (0.05)**	0.2 (0.05)	0.3 (0.1)	0.3 (0.1)	0.3 (0.1)	0.4 (0.1)	**0.5 (0.1)**
Grouper (Serranidae) biomass *	46.0 (25.4)	33.6 (11.6)	36.8 (10.3)	**59.8 (14.3)**	42.6 (10.3)	**30.4 (6.4)**	44.9 (11.2)
Graysby (*C. cruentata*) density *	**0.1 (0.03)**	0.1 (0.03)	0.2 (0.04)	0.2 (0.04)	0.2 (0.05)	**0.3 (0.1)**	0.3 (0.1)
Graysby (*C. cruentata*) biomass *	**4.0 (2.7)**	8.7 (3.2)	15.5 (3.7)	**19.1 (5.3)**	11.3 (2.9)	13.4 (3.4)	16.4 (4.2)
Coney (*C. fulva*) density	**<0.01 (<0.01)**	0.1 (0.04)	0.1 (0.03)	0.1 (0.03)	0.1 (0.03)	0.1 (0.03)	**0.1 (0.05)**
Coney (*C. fulva*) biomass	**0.9 (0.9)**	11.3 (7.3)	8.5 (4.1)	11.5 (4.8)	**16.2 (6.7)**	9.4 (3.6)	11.8 (5.3)
Red hind (*E. guttatus*) density	0.04 (0.02)	0.04 (0.01)	**0.02 (0.01)**	**0.05 (0.02)**	0.05 (0.02)	0.02 (0.01)	0.03 (0.01)
Red hind (*E. guttatus*) biomass	14.1 (7.5)	13.6 (5.6)	12.9 (7.7)	**28.6 (11.0)**	15.1 (6.3)	**5.2 (3.0)**	11.1 (3.9)

Only mean density and mean biomass for all groupers and *C. cruentata* exhibited significant differences between years. Mean density and biomass for *C. cruentata* were significantly (p<0.01) lower in 2001 and 2002 than 2007. Mean density of total groupers was significantly lower (p=0.0046) in 2001 (0.14 ± 0.05) than 2007 (0.46 ± 0.08), whereas total grouper biomass was significantly (p<0.05) higher in 2001 than 2007. All of the grouper metrics (except *C. cruentata* density) increased from 2006-2007, although not statistically significant. Four grouper metrics (*C. cruentata* and *C. fulva* density and biomass, and total grouper density) were lowest in 2001; and four grouper metrics (total grouper and *C. cruentata* biomass, and *E. guttatus* density and biomass) had their highest means in 2004.

E. guttatus

Mean grouper biomass was the only grouper metric that had consecutive years of increases and decreases and the only grouper metric to decrease significantly (p=0.0164) when comparing the start (2001) and finish of the monitoring data set (2007). Mean density and biomass of *C. cruentata* were significantly (p<0.01) higher in 2006 and 2007 than in 2001; and significantly (p<0.01) higher in 2007 than 2002.

C. fulva

Mean density of *C. fulva* increased across all years from 2001-2007, although the consecutive change through years were not significant. The average *C. fulva* density recorded from 2001-2007 was lowest in 2001 and the highest in 2007, and the average biomass was lowest in 2001 and the highest in 2005. Neither metric had a significant difference between years of mean high or low.

Mean density and biomass of *E. guttatus* did not exhibit any consecutive years of increase between 2001 and 2007, but did show periods with two consecutive years of decrease. Mean density was lowest in 2003 and highest in 2003. Mean biomass was highest in 2004 and lowest in 2006. Neither metric had a significant difference between years of mean high or low.

Snappers (Lutjanidae)

Snapper metrics included all species from the Lutjanidae family that were recorded in the southwest Puerto Rico study area from 2001-2007. The snapper species included in the 'total snapper' metrics were mutton snapper (*Lutjanus analis*), *L. apodus*, blackfin snapper (*Lutjanus buccanella*), *L. griseus*, dog snapper (*Lutjanus jocu*), *L. mahogoni*, *L. synagris*, *O. chrysurus* and unidentified *Lutjanus* species. The mean density and biomass for the most common snapper species recorded/observed in the study area are summarized in Table 3.5 and subset Table 3.10.

Table 3.10. Density and biomass (mean ± SE) for selected snapper species (2001-2007) for the southwest Puerto Rico study area. Asterisks (*) indicate there was a significant difference when years were compared. Values have been rounded to one decimal place up from Table 3.5 where appropriate. Bold indicates either the lowest or highest mean for that metric.

Snapper variable	2001 Mean (SE)	2002 Mean (SE)	2003 Mean (SE)	2004 Mean (SE)	2005 Mean (SE)	2006 Mean (SE)	2007 Mean (SE)
Snapper (Lutjanidae) density	2.4 (0.5)	3.4 (0.6)	2.8 (0.4)	2.2 (0.4)	**2.0 (0.4)**	**5.3 (1.5)**	3.4 (0.8)
Snapper (Lutjanidae) biomass	163.0 (38.3)	**287.0 (51.7)**	212.3 (36.2)	**145.9 (23.1)**	195.7 (45.4)	245.7 (64.6)	209.6 (50.2)
Lutjanus apodus density	1.3 (0.3)	**2.3 (0.4)**	1.8 (0.3)	1.2 (0.3)	**1.1 (0.3)**	2.2 (0.8)	1.7 (0.7)
Lutjanus apodus biomass	90.8 (28.9)	**155.7 (37.2)**	78.0 (17.7)	**59.1 (15.4)**	71.3 (26.7)	85.1 (26.1)	82.0 (44.0)
Lutjanus griseus density	0.3 (0.1)	0.5 (0.2)	0.2 (0.1)	0.5 (0.2)	**0.2 (0.1)**	**1.9 (0.9)**	0.8 (0.2)
Lutjanus griseus biomass	**19.5 (7.8)**	46.6 (20.4)	40.3 (23.7)	33.3 (11.3)	46.7 (24.7)	**93.0 (44.2)**	46.3 (17.4)
Lutjanus jocu density *	**0.0 (0.0)**	**0.0 (0.0)**	0.0 (0.0)	**0.0 (0.0)**	<0.01(<0.01)	<0.01(<0.01)	**0.04 (0.02)**
Lutjanus jocu biomass *	**0.0 (0.0)**	**0.0 (0.0)**	6.9 (4.9)	**0.0 (0.0)**	2.0 (2.0)	1.9 (1.9)	**12.9 (6.3)**
Lutjanus mahogoni density	0.02 (0.02)	0.04 (0.02)	0.02 (0.01)	**0.01 (<0.01)**	**0.1 (0.1)**	0.04 (0.02)	0.04 (0.03)
Lutjanus mahogoni biomass	**0.2 (0.2)**	1.1 (0.7)	**5.0 (3.0)**	0.8 (0.6)	3.2 (1.2)	3.2 (1.6)	4.6 (3.2)
Lutjanus synagris density *	**0.2 (0.1)**	0.1 (0.03)	0.1 (0.03)	0.02 (0.01)	0.04 (0.02)	**0.02 (0.01)**	0.1 (0.1)
Lutjanus synagris biomass *	6.6 (4.2)	3.7 (1.5)	**7.0 (3.0)**	1.5 (1.5)	3.7 (2.9)	**0.6 (0.6)**	2.7 (1.4)
Ocyurus chrysurus density *	0.6 (0.2)	0.5 (0.1)	0.7 (0.1)	**0.5 (0.1)**	0.5 (0.1)	**1.0 (0.2)**	0.7 (0.1)
Ocyurus chrysurus biomass *	**45.9 (12.7)**	**79.9 (22.3)**	56.4 (13.1)	49.0 (11.7)	68.7 (15.0)	61.6 (10.9)	61.1 (11.0)

L. mahogoni in St. Croix, USVI

The only snapper metric to exhibit three consecutive years of increase was *L. mahogoni* biomass from 2004-2007. Mean total snapper density was highest in 2006 and lowest in 2005. Mean biomass for total snapper was highest in 2002 and lowest in 2003. Mean density of *L. apodus* was highest in 2002 and lowest in 2005 and mean biomass was highest in 2002 and lowest in 2003. Mean density of *L. griseus* was highest in 2006 and lowest in 2005 and mean biomass was highest in 2006 and lowest in 2001. Mean density of *L. mahogoni* was highest in 2005 and lowest in 2004 and the average biomass was lowest in 2001 and the highest in 2003. None of these metrics had a significant difference between years of mean high or low.

Dog snapper (*Lutjanus jocu*) in St. John, USVI

Mean density and biomass of *L. jocu* was lowest in 2001, 2002 and 2004 (0 ± 0 and 0 ± 0, respectively) and significantly highest in 2007 (0.04 ± 0.02 for density and 12.87 ± 6.289 for biomass). Mean density and biomass of *L. synagris* were significantly ($p<0.01$) higher in 2001 than 2004 and 2006. Mean *L. synagris* density was highest in 2001 (0.22 ± 0.08) and lowest in 2006 (0.02 ± 0.01) and mean biomass was highest in 2003 (7.02 ± 3.05) and lowest in 2006 (0.64 ± 0.57). Both *L. synagris* and *L. jocu* metrics were significantly different ($p<0.01$) between years of high and low.

O. chrysurus

Mean biomass of *O. chrysurus* decreased from 2002-2004 and again from 2005-2007 after a brief increase from 2004-2005. Mean density of *O. chrysurus* was significantly ($p<0.05$) lower in 2002 than 2006; whereas, mean biomass was lower in 2002 than 2006 although not significant. Mean density of *O. chrysurus* was lowest in 2004 and highest in 2006; and mean biomass was lowest in 2001 and highest the following year in 2002. None of these metrics had a significant difference between years of mean high or low.

Jacks (Carangidae)

Jacks include all species from the Carangidae family that were recorded in the southwest Puerto Rico study area from 2001-2007. The jack species included in the 'total jack' metrics are yellow jack (*Carangoides bartholomaei*), *C. ruber*, *C. crysos*, horse-eye jack (*Caranx latus*), black jack (*Caranx lugubris*), and unidentified *Carangoides/Caranx* species. The mean density and biomass for the most common jack species recorded/observed in the study area are summarized in Table 3.5 and subset Table 3.11.

Table 3.11. Density and biomass (mean ± SE) for selected jack species (2001-2007) for the southwest Puerto Rico study area. Asterisks (*) indicate there was a significant difference when years were compared. Values have been rounded to one decimal place up from Table 3.5 where appropriate. Bold indicates either the lowest or highest mean for that metric.

Jack variable	2001 Mean (SE)	2002 Mean (SE)	2003 Mean (SE)	2004 Mean (SE)	2005 Mean (SE)	2006 Mean (SE)	2007 Mean (SE)
Jack (Carangidae) density	0.9 (0.3)	1.2 (0.7)	1.7 (0.7)	0.8 (0.3)	**3.3 (1.8)**	2.9 (1.1)	**0.3 (0.1)**
Jack (Carangidae) biomass	68.4 (43.4)	146.6 (75.2)	**147.4 (79.5)**	92.9 (45.9)	67.7 (28.9)	99.8 (36.6)	**42.0 (28.2)**
Caranx crysos density	0.1 (0.1)	0.1 (0.1)	0.1 (0.1)	**0.2 (0.1)**	**<0.01 (<0.01)**	0.0 (0.0)	0.1 (0.0)
Caranx crysos biomass	3.5 (3.5)	43.4 (33.4)	50.8 (39.1)	**65.4 (43.2)**	**2.3 (2.3)**	21.2 (17.9)	29.3 (27.2)
Caranx latus density *	**0.2 (0.2)**	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	**0.0 (0.0)**
Caranx latus biomass *	**18.6 (15.3)**	0.4 (0.4)	0.8 (0.7)	0.1 (0.1)	1.1 (0.9)	0.2 (0.2)	**0.0 (0.0)**
Carangoides ruber density	0.3 (0.1)	0.3 (0.1)	0.9 (0.4)	0.3 (0.1)	**1.6 (0.7)**	1.3 (0.6)	**0.2 (0.1)**
Carangoides ruber biomass	**3.7 (1.6)**	11.6 (7.5)	10.8 (2.5)	9.4 (3.7)	**23.2 (9.4)**	15.4 (6.0)	5.4 (2.8)

Mean density of total jacks was highest in 2005 and lowest in 2007; and the average biomass was highest in 2003 and the lowest in 2007. Mean density of *C. ruber* was highest in 2005 and lowest in 2007 and mean biomass was highest in 2003 and lowest in 2002. None of these metrics had a significant difference between years of mean high or low.


Fish diver and *C. ruber*

Mean density and biomass of *C. crysos* were the only jack metrics to increase from 2006-2007 and all others decreased. These two metrics were also the only jack metrics that did not have consecutive years of decrease, with mean biomass of *C. crysos* showing three years of consecutive increase (2001-2004). Mean density and biomass of *C. crysos* was highest in 2004 and lowest the following year in 2005. Neither metric had a significant difference between years of mean high or low.

Mean density of *C. latus* exhibited three years of consecutive decrease (2004-2007). *C. latus* density and biomass were significantly higher in 2001 than 2002, 2006 and 2007. Mean density and biomass were highest in 2001 (0.25 ± 0.16 and 18.63 ± 15.34, respectively) and the lowest in 2007 (0 ± 0 and 0 ± 0, respectively). Both metrics had a significant difference (p=0.0157) between years of mean high or low.

Parrotfishes (Scaridae)

Parrotfish metrics included all species from the Scaridae family that were recorded in the southwest Puerto Rico study area from 2001 to 2007. The parrotfish species included in the 'total parrotfish' metrics were bluelip parrotfish (*Cryptotomus roseus*), rainbow parrotfish (*Scarus guacamaia*), *S. iseri*, *S. taeniopterus*, queen parrotfish (*Scarus vetula*), greenblotch parrotfish (*Sparisoma atomarium*), *S. aurofrenatum*, redtail parrotfish (*Sparisoma chrysopterum*), *S. radians, S. rubripinne*, *S. viride*, and unidentified *Scarus* and *Sparisoma* species. The mean density and biomass for the most common parrotfish species recorded/observed in the study area are summarized in Table 3.5 and subset Table 3.12.

Table 3.12. Density and biomass (mean ± SE) for selected parrotfish species (2001-2007) for the southwest Puerto Rico study area. Asterisks (*) indicate there was a significant difference when years were compared. Values have been rounded to one decimal place up from Table 3.5 where appropriate. Bold indicates either the lowest or highest mean for that metric.

Parrotfish variable	2001 Mean (SE)	2002 Mean (SE)	2003 Mean (SE)	2004 Mean (SE)	2005 Mean (SE)	2006 Mean (SE)	2007 Mean (SE)
Parrotfish (Scaridae) density *	**11.5 (1.6)**	12.2 (1.3)	**19.6 (1.5)**	12.4 (1.1)	12.6 (1.8)	15.5 (1.4)	17.5 (1.7)
Parrotfish (Scaridae) biomass *	**232.4 (50.1)**	461.2 (78.8)	568.8 (79.6)	543.2 (71.0)	692.1 (99.0)	679.6 (82.6)	747.4 (129.5)
Scarus iseri density	6.3 (1.2)	5.7 (0.8)	**7.3 (1.0)**	**4.7 (0.6)**	5.0 (1.4)	6.0 (0.8)	6.8 (1.3)
Scarus iseri biomass	**70.2 (15.9)**	95.8 (23.4)	98.7 (17.1)	84.7 (12.0)	101.0 (14.7)	**153.7 (25.7)**	149.0 (26.3)
Scarus taeniopterus density *	**0.2 (0.1)**	0.5 (0.2)	2.0 (0.6)	1.4 (0.3)	2.0 (0.4)	2.5 (0.4)	2.9 (0.5)
Scarus taeniopterus biomass *	**9.4 (6.8)**	31.7 (7.9)	**138.5 (53.9)**	78.7 (20.5)	111.1 (20.3)	131.6 (24.5)	113.8 (19.1)
Sparisoma atomarium density *	0.3 (0.1)	0.7 (0.2)	**0.7 (0.2)**	0.5 (0.1)	**0.1 (0.1)**	0.3 (0.1)	0.2 (0.1)
Sparisoma atomarium biomass *	**0.2 (0.1)**	**4.1 (2.5)**	0.5 (0.2)	0.6 (0.1)	0.4 (0.3)	0.3 (0.1)	0.4 (0.2)
Sparisoma aurofrenatum density *	**0.9 (0.2)**	2.1 (0.3)	2.5 (0.3)	2.7 (0.3)	3.0 (0.3)	**4.3 (0.4)**	4.1 (0.4)
Sparisoma aurofrenatum biomass *	**44.2 (9.6)**	125.5 (22.5)	115.9 (17.7)	151.3 (20.0)	147.3 (19.2)	**163.1 (20.9)**	132.8 (15.5)
Sparisoma radians density *	1.9 (0.4)	1.2 (0.2)	**3.5 (0.6)**	1.3 (0.3)	0.6 (0.2)	**0.4 (0.1)**	1.0 (0.3)
Sparisoma radians biomass *	3.4 (1.0)	4.0 (1.0)	**8.9 (2.5)**	2.7 (1.2)	1.4 (0.7)	**0.7 (0.3)**	1.2 (0.4)
Sparisoma viride density	1.0 (0.3)	**1.0 (0.2)**	**1.5 (0.2)**	1.1 (0.2)	1.0 (0.2)	1.0 (0.2)	1.2 (0.2)
Sparisoma viride biomass	**86.6 (28.5)**	174.0 (42.1)	160.5 (34.6)	213.4 (43.8)	265.6 (63.4)	148.1 (29.0)	**282.3 (83.7)**

Of all metrics analyzed, the parrotfish species exhibited the most significant difference for between year comparisons. The majority of the lowest means for parrotfish metrics occurred in 2001; whereas the majority of the highest means occurred in 2003. All parrotfish metrics,

Juvenile phase *S. taeniopterus*

except mean density of *S. iseri* and *S. viride* increased from 2001-2002. Mean density of total parrotfish increased from 2001-2003 followed by a significant (p<0.0001) decrease in 2004, and then increased consecutively until 2007, albeit not statistically significant. It is worthy to note that the mean density for total parrotfish in 2007 was lower than the mean density in 2003, before the significant decrease in 2003. Mean biomass was significantly (p<0.0001) lower in 2001 than all year from 2003-2007; and lower in 2002 than 2003. Mean density of total parrotfish was lowest in 2001 (11.54 ± 1.58) and the highest in 2003 (19.56 ± 1.55); and mean biomass was lowest in 2001 (232.37 ± 50.12) and highest in 2007 (747.38 ± 129.55). Both metrics had a significant difference (p<0.0001) between years of mean high or low.

Juvenile phase *S. iseri*

Mean density of *S. iseri* exhibited three years of consecutive increase from 2004 to 2007, with highest density in 2003 and lowest the following year in 2003. Mean biomass was lowest in 2001 and the highest in 2006. Neither metric had a significant difference between years of mean high or low. Mean density of *S. taeniopterus* exhibited three consecutive years of increase (2004 to 2007) and one of the highest number (n=10) of significantly different (p<0.0001) year comparisons indicative of high variability. Mean density of *S. taeniopterus* was highest in 2007 (2.89 ± 0.46) and significantly (p<0.0001) lowest in 2001 (0.19 ± 0.11). Mean biomass of *S. taeniopterus* exhibited two consecutive years of increase, although neither were significant; however, mean biomass *S. taeniopterus* also had one of the highest number (n=11) of significantly different (p<0.0001) year comparisons. Mean biomass was lowest in 2001 and highest in 2003 and not significantly different. Mean density and biomass of *S. taeniopterus* was significantly (p<0.0001) higher between 2005 and 2007 than 2001 and 2002, but mean biomass was significantly (p<0.0001) higher in 2003 than all years from 2005-2007.

Mean density of *S. atomarium* was the only parrotfish metric that had two years of consecutive increase (2001-2003), followed by two years of decrease (2003-2005). Mean density and biomass of *S. atomarium* was significantly (p<0.0001) higher in 2004 than 2001, 2006, and 2007; and with 2003 significantly higher than 2005 and 2007. Mean density of *S. atomarium* was highest in 2003 (0.73 ± 0.17) and significantly (p<0.0001) lowest in 2005 (0.13 ± 0.05). Mean biomass was lowest in 2001 and highest the following year in 2002, and not significantly different.

Mean density of *S. aurofrenatum* increased from 2001-2006, but was not significantly different between years, whereas no consecutive years of change were found for mean biomass of *S. aurofrenatum*. Mean *S. aurofrenatum* density and biomass were highest in 2006 (3.27 ± 0.43 and 163.11 ± 20.92, respectively) and significantly (p=<0.0001) lowest in 2001 (0.90 ± 0.23 and 43.18 ± 9.63, respectively). Mean density and biomass of *S. aurofrenatum* were significantly higher from 2003 to 2007 than in 2001 and lower in 2002 than 2006 and 2007.

Terminal phase *S. aurofrenatum*

Mean density and biomass of *S. radians* decreased over three consecutive years from 2003-2006, with 2003-2004 exhibiting the only significant difference (p<0.0001). Mean biomass of *S. radians* exhibited the highest number (n=12) of significantly different (p<0.0001) year comparisons. *S. radians* density and biomass were significantly higher in 2001 and 2003 than all years from 2004-2007, with highest mean density and biomass recorded in 2003 (3.45 ± 0.63 and 8.93 ± 2.46, respectively) and significantly lowest in 2006 (0.38 ± 0.12 and 0.67 ± 0.26, respectively).

Mean density of *S. viride* decreased over three consecutive years from 2003-2006 although the difference was not significant. From 2001-2005, mean density and biomass of *S. viride* showed an opposite directional change (i.e., when density increased, biomass decreased and vice versa). Mean density of *S. viride* was highest in 2003 and lowest in 2002 and mean biomass was highest in 2007 and lowest in 2001. Neither metric had a significant difference between years of mean high or low.

Haemulidae (Grunts)

Grunt metrics include all species from the Haemulidae family that were recorded in the southwest Puerto Rico study area from 2001-2007. The grunt species included in the 'total grunt' metrics were porkfish (*Anisotremus virginicus*), *H. aurolineatum*, caesar grunt (*Haemulon carbonarium*), smallmouth grunt (*Haemulon chrysargyreum*), *H. flavolineatum*, cottonwick (*Haemulon melanurum*), sailors choice (*Haemulon parra*), *H. plumierii*, *H. sciurus* and unidentified *Haemulon* species (most of which are juveniles). The mean density and biomass for the most common grunt species recorded/observed in the study area are summarized in Table 3.5 and subset Table 3.13.

Table 3.13. Density and biomass (mean ± SE) for selected grunt species (2001-2007) for the southwest Puerto Rico study area. Asterisks (*) indicate there was a significant difference when years were compared. Values have been rounded to one decimal place up from Table 3.13 where appropriate. Bold indicates either the lowest or highest mean for that metric.

Grunt variable	2001 Mean (SE)	2002 Mean (SE)	2003 Mean (SE)	2004 Mean (SE)	2005 Mean (SE)	2006 Mean (SE)	2007 Mean (SE)
Grunt (Haemulidae) density	**15.9 (4.0)**	14.6 (3.0)	**11.7 (2.1)**	12.9 (3.0)	13.9 (3.6)	13.3 (5.4)	12.8 (4.7)
Grunt (Haemulidae) biomass	192.6 (86.0)	**286.0 (72.1)**	203.1 (62.3)	**89.9 (16.3)**	150.6 (37.4)	164.4 (40.4)	184.1 (43.6)
Haemulon aurolineatum density *	0.6 (0.6)	**0.1 (0.0)**	2.3 (0.9)	2.7 (0.9)	0.2 (0.1)	0.5 (0.3)	**5.1 (4.2)**
Haemulon aurolineatum biomass *	**2.3 (1.5)**	5.3 (2.4)	**58.2 (42.5)**	10.7 (4.5)	4.7 (2.3)	9.9 (3.5)	30.3 (22.4)
Haemulon flavolineatum density	**9.3 (2.9)**	7.6 (2.3)	4.2 (1.4)	3.5 (1.3)	4.2 (1.5)	2.1 (0.7)	**2.0 (0.6)**
Haemulon flavolineatum biomass	**123.7 (61.4)**	98.4 (29.4)	40.8 (10.9)	36.2 (10.4)	56.9 (22.6)	**32.0 (8.5)**	35.0 (11.7)
Haemulon plumierii density *	0.4 (0.2)	**0.2 (0.1)**	**1.6 (0.6)**	0.7 (0.3)	0.6 (0.2)	1.0 (0.4)	0.4 (0.1)
Haemulon plumierii biomass *	8.9 (4.0)	9.1 (3.8)	32.2 (10.0)	**5.5 (2.1)**	28.4 (9.9)	**67.9 (28.5)**	29.2 (7.3)
Haemulon sciurus density *	3.3 (1.0)	**4.6 (1.1)**	2.3 (0.6)	**0.8 (0.2)**	0.9 (0.4)	1.0 (0.4)	1.0 (0.4)
Haemulon sciurus biomass *	52.4 (25.9)	**159.6 (52.3)**	59.2 (16.3)	**28.4 (6.4)**	43.6 (19.3)	40.8 (12.7)	66.4 (27.1)

Of the selected grunt metrics, all of the individual species means, except total grunt metrics, exhibited a significant difference between years. Mean *H. aurolineatum* density and *H. plumierii* biomass were the only grunt metrics that did not have any consecutive years of decline. Overall, 2004 had the lowest means. Six of the metrics that decreased from 2003-2004, increased the following year (2004-2005).

Total grunt mean density was highly variable with several periods of two consecutive years of unidirectional change, which decreased from 2001-2003, increased from 2003-2005, then decreased from 2005-2007. None of these changes in the mean were statistically significant. Mean density of total grunt was highest in 2001 and lowest in 2003; and mean biomass was highest in 2002 and lowest in 2003. Neither metric had a significant difference between years of mean high or low. The only times where mean total grunt density and biomass had the same directional change were from 2002-2003 and 2004-2005. Total grunt biomass had three consecutive years of increasing means (2004-2007), although the changes were not significant.

H. aurolineatum

Smallmouth grunt (*Haemulon chysargyreum*)

Juvenile grunts

H. aurolineatum

Mean biomass of *H. aurolineatum* exhibited three periods of two consecutive years of change, which increased from 2001-2003, decreased from 2003-2005, then increased again from 2005-2007. Only the change from 2002-2003 was significant (p=0.0024). Additionally, mean *H. aurolineatum* density increased significantly (p=0.0011) from 2002-2003 and was significantly higher in 2003 than 2005. Mean density of *H. aurolineatum* was highest in 2007 and lowest in 2002 and mean biomass was lowest in 2001 and the highest in 2003. Mean density and biomass of *H. flavolineatum* was the only grunt metrics to have three consecutive years of decline (2001-2004). Mean density of *H. flavolineatum* was highest in 2001 and lowest in 2007; and the average biomass was highest in 2001 and the lowest in 2006. Neither *H. aurolineatum* and *H. flavolineatum* metrics have significant differences between years of mean high or low.

Mean density of *H. plumierii* was lowest in 2002 (0.20 ± 0.06) and significantly (p=0.0423) highest the following year in 2003 (1.64 ± 0.61). Mean biomass of *H. plumierii* was lowest in 2004 and highest in 2006. Mean density of *H. sciurus* exhibited three years of consecutive increases (2004-2007), although they were not significant. Mean density of *H. sciurus* and biomass were significantly (p<0.05) higher in 2002 than all years from 2005-2007; and mean density was significantly (p=0.0002) higher in 2001 than 2005. Mean density of *H. sciurus* was highest in 2002 and lowest in 2004; and the average biomass was highest in 2002 and lowest in 2003.

Surgeonfishes(Acanthuridae)

Surgeonfish metrics include all species from the Acanthuridae family that were recorded in the southwest Puerto Rico study area. The surgeonfish species included in the 'total surgeonfish' metrics were *A. bahianus*, *A. chirurgus*, *A. coeruleus* and unidentified *Acanthurus* species. The mean density and biomass for the most common surgeonfish species recorded/observed in the study area are summarized in Table 3.5 and subset Table 3.14.

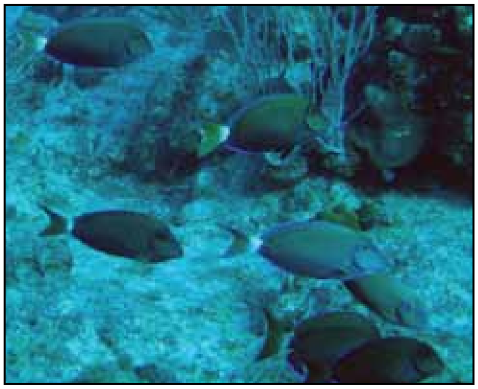
Table 3.14. Density and biomass (mean ± SE) for selected surgeonfish species (2001-2007) for the southwest Puerto Rico study area. Asterisks (*) indicate there was a significant difference when years were compared. Values have been rounded to one decimal place up from Table 3.14 where appropriate. Bold indicates either the lowest or highest mean for that metric.

Surgeonfish variable	2001 Mean (SE)	2002 Mean (SE)	2003 Mean (SE)	2004 Mean (SE)	2005 Mean (SE)	2006 Mean (SE)	2007 Mean (SE)
Surgeonfish (Acanthuridae) density *	2.5 (0.5)	**1.9 (0.3)**	**4.4 (0.7)**	2.4 (0.3)	3.1 (0.4)	4.1 (0.7)	3.5 (0.5)
Surgeonfish (Acanthuridae) biomass *	174.4 (61.1)	**153.4 (35.7)**	**369.6 (141.8)**	161.7 (26.8)	265.2 (40.2)	309.2 (56.3)	311.4 (64.7)
Acanthurus bahianus density *	1.3 (0.3)	**1.1 (0.2)**	2.3 (0.3)	1.3 (0.2)	1.9 (0.3)	**2.8 (0.4)**	1.8 (0.2)
Acanthurus bahianus biomass *	76.4 (23.1)	**76.3 (18.9)**	121.6 (20.1)	78.5 (13.0)	139.3 (22.8)	**201.6 (28.0)**	126.7 (17.1)
Acanthurus chirurgus density *	**0.2 (0.1)**	0.4 (0.1)	**1.0 (0.3)**	0.5 (0.1)	0.5 (0.1)	0.3 (0.1)	0.7 (0.3)
Acanthurus chirurgus biomass *	**13.8 (5.9)**	39.8 (20.9)	99.8 (39.2)	42.9 (18.9)	38.6 (12.7)	25.1 (10.7)	**103.9 (53.9)**
Acanthurus coeruleus density *	0.9 (0.4)	**0.5 (0.1)**	1.0 (0.3)	0.6 (0.1)	0.7 (0.2)	**1.1 (0.4)**	0.9 (0.3)
Acanthurus coeruleus biomass *	84.2 (53.6)	**37.3 (12.4)**	**148.2 (97.8)**	40.3 (8.7)	87.2 (20.4)	82.5 (34.0)	80.8 (23.9)

Juvenile *A. coeruleus*

The means for all surgeonfish metrics increased from 2002-2003 then decreased the following year from 2003-2003. The highest means for all of the surgeonfish metrics occurred in 2003, 2006 and 2007. The lowest means for all of the surgeonfish metrics occurred in 2001 and 2002. Density and biomass of all surgeonfish and *A. coeruleus*, as well as, mean density of *A. chirurgus* was significantly (p<0.05) lower in 2002 than 2006. Mean density of total surgeonfish was the only surgeonfish metric to increase over three consecutive years. Mean density and biomass of total surgeonfish was lowest in 2002 (1.91 ± 0.32 and 153.40 ± 35.74, respectively) and significantly (p<0.0001) higher the following year in 2003 (4.39 ± 0.71 and 369.61 ± 141.82, respectively). Surgeonfish density and biomass means were significantly (p<0.0001) lower in 2002 than in 2003, 2005, 2006 and 2007.

Mean density and biomass of *A. bahianus* was highest in 2006 (2.79 ± 0.41 and 201.58 ± 27.99, respectively) and lowest in 2002 (1.09 ± 0.19 and 76.32 ± 18.87, respectively); however only biomass had a significant difference between years of mean high and low. Mean density of *A. bahianus* was significantly lower in 2004 than 2003 and 2006 and significantly (p=0.0001) lower in 2002 than 2003 and 2003. Mean biomass of *A. bahianus* was significantly lower in 2002 than 2003, 2006 and 2007; and significantly higher in 2006 than 2003.



A. bahianus

A. chirurgus

Mean density and biomass of *A. chirurgus* exhibited three consecutive years of decline. Mean density and biomass were significantly (p<0.05) higher in 2003 than 2006. Mean density was highest in 2003 and lowest in 2001 and mean biomass was lowest in 2001 (13.82 ± 5.88) and highest in 2007. Neither metric had a significant difference between years of mean high or low. Mean biomass of *A. coeruleus* was the only surgeonfish metric that didn’t have any consecutive years of increase. Mean density and biomass were significantly higher (p<0.05) in 2006 than 2002. Mean density was highest in 2006 (1.07 ± 0.37) and significantly (p=0.0261) lowest in 2002 (0.45 ± 0.14). Mean biomass was also lowest in 2002, but was highest the following year in 2003, although not statistically significant.

Other Families

Squirrelfish (Holocentridae) and wrasse (Labridae) were selected because they are locally abundant and geographically widespread families in the southwest Puerto Rico study area. The wrasses also make up a large part of the herbivore group. The species included in the ‘total squirrelfish’ metrics are squirrelfish (*Holocentrus adscensionis*), longspine squirrelfish (*Holocentrus rufus*), blackbar soldierfish (*Myripristis jacobus*), longjaw squirrelfish (*Neoniphon marianus*), reef squirrelfish (*Sargocentron coruscum*), and dusky squirrelfish (*Sargocentron vexillarium*). The species included in the ‘total wrasse’ metrics are Spanish hogfish (*Bodianus rufus*), *C. parrae*, dwarf wrasse (*Doratonotus megalepis*), slippery dick (*Halichoeres bivittatus*), yellowcheek wrasse (*Halichoeres cyanocephalus*), yellowhead wrasse (*Halichoeres garnoti*), clown wrasse (*Halichoeres maculipinna*), rainbow wrasse (*Halichoeres pictus*), blackear wrasse (*Halichoeres poeyi*), puddingwife (*Halichoeres radiatus*), hogfish (*Lachnolaimus maximus*), bluehead wrasse (*Thalassoma bifasciatum*), rosy razorfish (*Xyrichtys martinicensis*), pearly razorfish (*Xyrichtys novacula*) and green razorfish (*Xyrichtys splendens*). The mean density and biomass for these two common families recorded/observed in the study area are summarized in Table 3.5 and subset Table 3.15.

Table 3.15. Density and biomass (mean ± SE) for additional selected families (2001-2007) for the southwest Puerto Rico study area. Asterisks (*) indicate there was a significant difference when years were compared. Values have been rounded to one decimal place up from Table 3.5 where appropriate. Bold indicates either the lowest or highest mean for that metric.

Other Families variable	2001 Mean (SE)	2002 Mean (SE)	2003 Mean (SE)	2004 Mean (SE)	2005 Mean (SE)	2006 Mean (SE)	2007 Mean (SE)
Squirrelfish (Holocentridae) density *	**1.9 (0.8)**	0.8 (0.2)	0.7 (0.1)	**0.5 (0.1)**	0.7 (0.1)	1.2 (0.2)	1.0 (0.2)
Squirrelfish (Holocentridae) biomass *	**107.5 (50.9)**	79.5 (20.6)	69.0 (13.3)	**59.9 (10.5)**	65.3 (10.2)	99.1 (14.5)	86.0 (18.3)
Wrasse (Labridae) density *	8.9 (3.1)	8.8 (1.5)	11.7 (1.6)	**8.2 (1.2)**	10.6 (1.7)	**13.7 (1.9)**	12.7 (1.4)
Wrasse (Labridae) biomass *	50.0 (12.4)	60.0 (10.9)	63.9 (13.5)	**28.3 (4.4)**	**98.6 (44.6)**	47.2 (5.8)	64.0 (7.4)

Mean density and biomass of total squirrelfish was highest in 2001 followed by three consecutive years of decrease (2001-2004) resulting in the lowest means in 2003. Neither metric had a significant difference between years of mean high or low. Squirrelfish density and biomass means were significantly (p<0.01) higher in 2006 than 2002 and 2003.

Mean density of total wrasse was highest in 2006 and lowest in 2004 and mean biomass was highest in 2005 and lowest in 2003. Neither metric had a significant difference between years of mean high and low. When comparing first and last years, mean density of wrasse was significantly (p=0.0014) higher in 2007 than 2001 and 2002.

Other Species

B. vetula, *M. martinicus* and *P. maculatus* were included in the summary metrics because they are commercially important fish species and exhibited significant inter-annual differences. The mean density and biomass for these species recorded/observed in the study area are summarized in Table 3.5 and subset Table 3.16.

Table 3.16. Density and biomass (mean ± SE) for additional selected species (2001-2007) for the southwest Puerto Rico study area. Asterisks (*) indicate there was a significant difference when years were compared. Values have been rounded to one decimal place up from Table 3.5 where appropriate. Bold indicates either the lowest or highest mean for that metric.

Other species variable	**2001 Mean (SE)**	**2002 Mean (SE)**	**2003 Mean (SE)**	**2004 Mean (SE)**	**2005 Mean (SE)**	**2006 Mean (SE)**	**2007 Mean (SE)**
Balistes vetula density *	0.03 (0.02)	0.05 (0.02)	0.02 (0.01)	**0.02 (0.01)**	0.03 (0.01)	0.1 (0.03)	**0.2 (0.1)**
Balistes vetula biomass *	18.5 (11.0)	26.6 (14.5)	7.6 (5.2)	**4.2 (2.4)**	22.1 (15.3)	48.8 (13.4)	**71.7 (18.1)**
Mulloidichthys martinicus density	0.5 (0.3)	0.1 (0.05)	0.1 (0.1)	**0.1 (0.03)**	0.2 (0.1)	0.3 (0.2)	**0.7 (0.6)**
Mulloidichthys martinicus biomass	**40.6 (22.3)**	19.3 (7.3)	28.5 (13.1)	**7.5 (4.2)**	31.4 (19.1)	17.2 (6.9)	31.7 (15.2)
Pseudupeneus maculatus density	0.3 (0.1)	**0.7 (0.4)**	0.4 (0.1)	**0.1 (0.03)**	0.3 (0.1)	0.3 (0.1)	0.3 (0.1)
Pseudupeneus maculatus biomass	13.6 (4.1)	**29.8 (11.0)**	18.3 (5.5)	**5.6 (1.8)**	17.2 (4.9)	13.7 (3.0)	16.9 (3.8)

Mean density and biomass of *B. vetula* was highest in 2007 (0.18 ± 0.05 and 71.73 ± 18.08, respectively) and lowest in 2004 (0.02 ± 0.01 and 4.18 ± 2.40, respectively). Both metrics had a significant difference (p<0.0001) between years of mean high or low. Furthermore, mean density and biomass of *B. vetula* was significantly (p<0.0001) higher in 2007 than all years from 2001-2005 and were significantly higher in 2006 than 2003 and 2003.

B. vetula

Like many other metrics, both goatfish species (*M. martinicus* and *P. maculatus*) had their lowest means in 2004, although no inter-year differences were statistically significant. Mean *M. martinicus* density was highest in 2007 and lowest in 2004 and mean biomass was highest in 2001 and lowest in 2003. Both mean density and biomass of *P. maculatus* was highest in 2002 and the lowest in 2004; and mean biomass was highest in 2002 and lowest in 2003. Neither *M. martinicus* nor *P. maculatus* metrics had a significant difference between years of mean high or low.

Other species, families and groups (such as sharks/rays, porgies, angelfish, butterflyfish, damselfish) were not included because mean density or biomass values were too low (i.e., small sizes or infrequent observations) to illustrate in the summary table.

Foureye butterflyfish (*Chaetodon capistratus*)

3.5. Seasonal patterns in fish metrics (2004-2007)

Inter-seasonal comparisons of fish metrics were based on data from 2004-2007 only when semi-annual surveys were standardized to just two specific periods representing winter (December-March) and summer (August-September). Mangroves were not surveyed during the August 2005 mission, thus the summer 2005 data are presented without mangrove data. Table 3.17 shows the winter and summer densities for the 20 most abundant fish species.

Table 3.17. Winter and summer total and mean (± SE) density for the 20 most abundant fish species (2004-2007) for southwest Puerto Rico study area. N=708 surveys (per 100 m^2).

Species	Common name	Total density	Winter density	Winter mean density	Summer density	Summer mean density
Thalassoma bifasciatum	bluehead wrasse	4068	1532	3.3 (0.9)	2536	7.3 (0.6)
Scarus iseri	striped parrotfish	3989	2330	6.5 (0.4)	1659	3.8 (1.0)
Stegastes partitus	bicolor damselfish	3376	1520	3.2 (0.6)	1856	5.3 (0.5)
Sparisoma aurofrenatum	redband parrotfish	2489	1035	2.9 (0.3)	1454	3.2 (0.2)
Haemulon flavolineatum	french grunt	2065	1200	3.3 (0.7)	865	2.5 (0.8)
Halichoeres garnoti	yellowhead wrasse	1679	741	2.1 (0.3)	938	2.7 (0.3)
Scarus taeniopterus	princess parrotfish	1573	595	1.7 (0.3)	978	2.8 (0.2)
Haemulon aurolineatum	tomtate	1511	1047	2.9 (0.4)	464	1.3 (2.1)
Acanthurus bahianus	ocean surgeon	1378	669	1.9 (0.2)	709	2.0 (0.2)
Coryphopterus personatus/ hyalinus	glass/masked goby	1118	340	0.9 (0.6)	778	2.2 (0.3)
Lutjanus apodus	schoolmaster	1118	606	1.7 (0.4)	512	1.5 (0.4)
Halichoeres bivittatus	slipper dick	1086	603	1.7 (0.3)	483	1.4 (0.3)
Chaetodon capistratus	foureye butterflyfish	1070	487	1.4 (0.1)	583	1.7 (0.1)
Nes longus	orangespotted goby	1066	515	1.4 (0.3)	551	1.6 (0.3)
Coryphopterus glaucofraenum	bridled goby	1005	526	1.5 (0.2)	479	1.4 (0.3)
Stegastes leucostictus	beaugregory	979	516	1.4 (0.2)	463	1.3 (0.2)
Stegastes planifrons	threespot damselfish	835	410	1.1 (0.3)	425	1.2 (0.2)
Sparisoma viride	stoplight parrotfish	763	365	1.0 (0.1)	398	1.1 (0.1)
Gnatholepis thompsoni	goldspot goby	751	329	0.9 (0.2)	422	1.2 (0.2)
Chromis cyanea	blue chromis	742	163	0.5 (0.5)	579	1.7 (0.1)

Mean densities in summer were higher for 13 of the 20 most abundant fish. Not all means were very different, but greatest increase from winter to summer densities was recorded for *T. bifasciatum, S. partitus, S. taeniopterus; C. personatus/hyalinus* and *C. cyanea*. In contrast, seven species exhibited higher mean densities in winter, with highest difference recorded for *S. iseri, H. aurolineatum* and *H. flavolineatum* (Table 3.17).

3.5.1. Community

Overall, nine of the 12 community metrics (total and trophic groups) had lowest means in the winter sampling season. Five of those metrics occurred during the 2006 winter (total, herbivore and piscivore richness, and herbivore and planktivore biomass) and four occurred during the 2004 winter (total fish density, piscivore and planktivore density, and planktivore richness). Two metrics (mean fish biomass and piscivore biomass), however, had lowest means in summer 2007. Summer mean total fish density decreased consecutively from 2004-2007, while winter mean total density increased from 2004-2007 (Figure 2.117a). Total mean biomass was more highly variable alternating between winter and summer highs (Figure 2.117b). Mean species richness was higher during the summer months than winter for all years (Figure 2.117c).


Planktivorous fish blue chromis (*Chromis cyanea*)

Herbivore means in 2004 and 2006 exhibited a similar pattern; winter means were higher in 2004 and summer means were higher in 2006 for all herbivore metrics (Figure 3.118). Winter mean herbivore density increased from 2005-2007; however, were higher in the summer during that same time period (Figure 3.118a). Summer biomass exhibited consecutive years of decrease from 2004-2007 and from 2004-2006 during the winter sampling period (Figure 3.118b). Winter mean herbivore biomass was higher in all years except for 2006. Mean herbivore richness also declined consecutively in the winter from 2004-2006 (Figure 3.118c). Winter herbivore richness means were higher from 2004-2005 and summer exhibited the highest means from 2006-2007 (Figure 3.118c).

Piscivore mean density and richness were highest during the summer sampling period and biomass was highest during the winter. Piscivore density was at least two times higher during the summer than winter

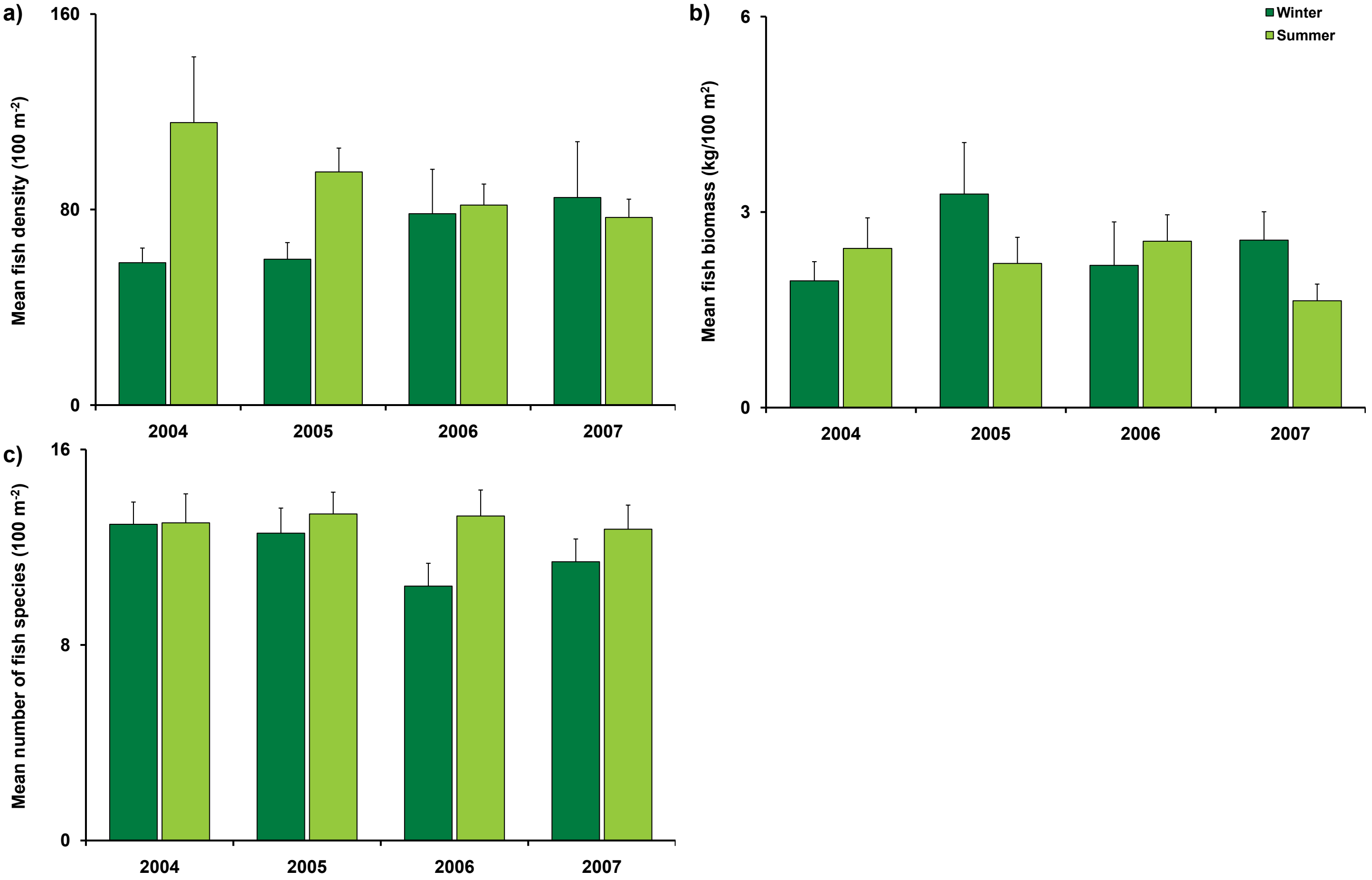


Figure 3.117. Seasonal change in mean (± SE) fish: (a) density, (b) biomass and (c) species richness in the southwest Puerto Rico study area.

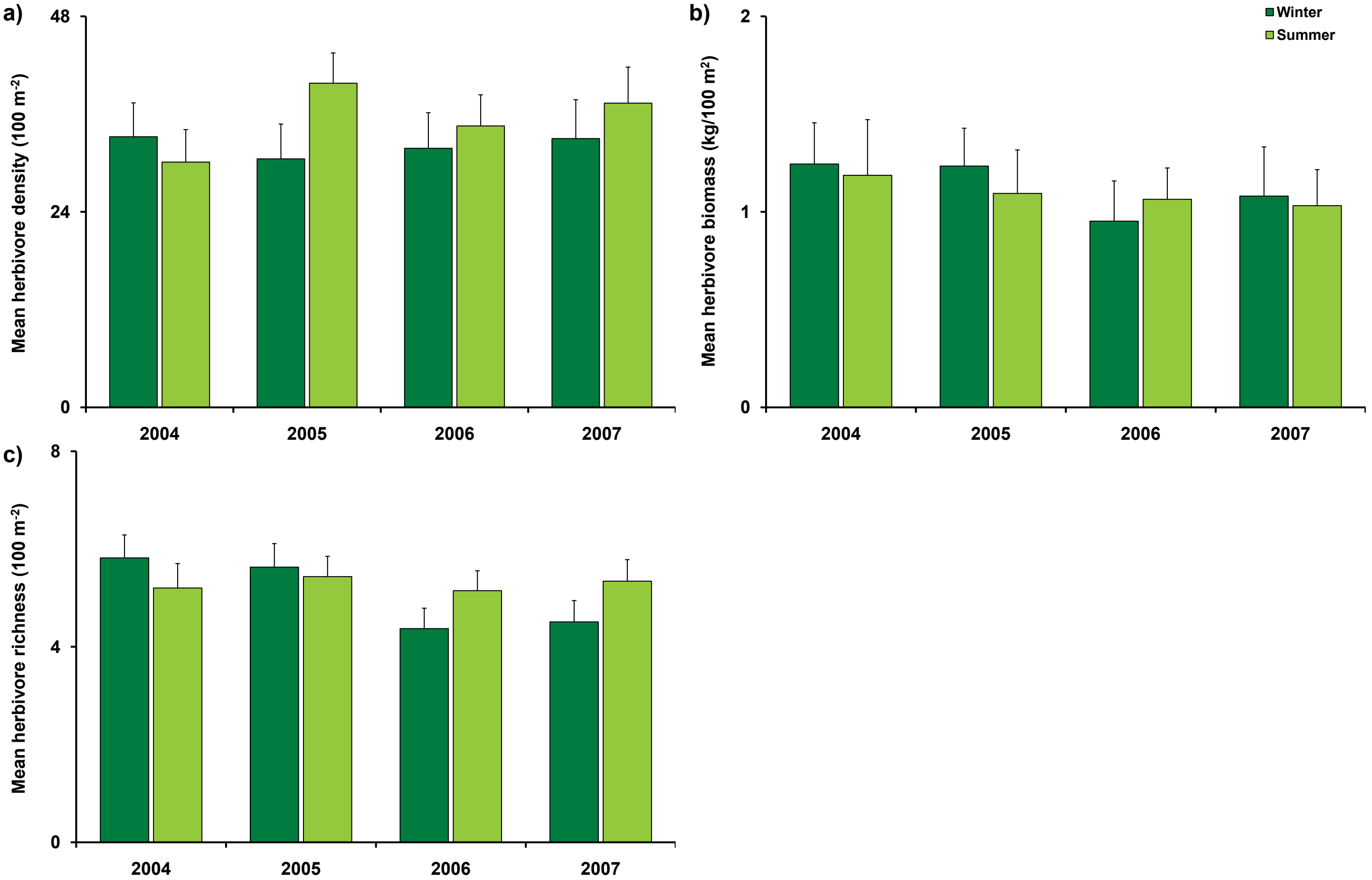


Figure 3.118. Seasonal change in mean (± SE) herbivore: (a) density, (b) biomass and (c) richness in the southwest Puerto Rico study area.

from 2004-2006 (Figure 3.119a). Both winter and summer sampling periods increased from 2004-2005 followed by decreasing means from 2005-2006, with summer continuing to decline from 2006-2007 (Figure 3.119a). Piscivore mean biomass decreased consecutively from 2005-2007, whereas winter sampling periods were variable with peaks in 2005 and 2007 (Figure 3.119b). Mean piscivore species richness was higher in summer than winter for all years. Richness decreased consecutively during the winter from 2004-2006 and during the summer from 2005-2007 (Figure 3.119c).

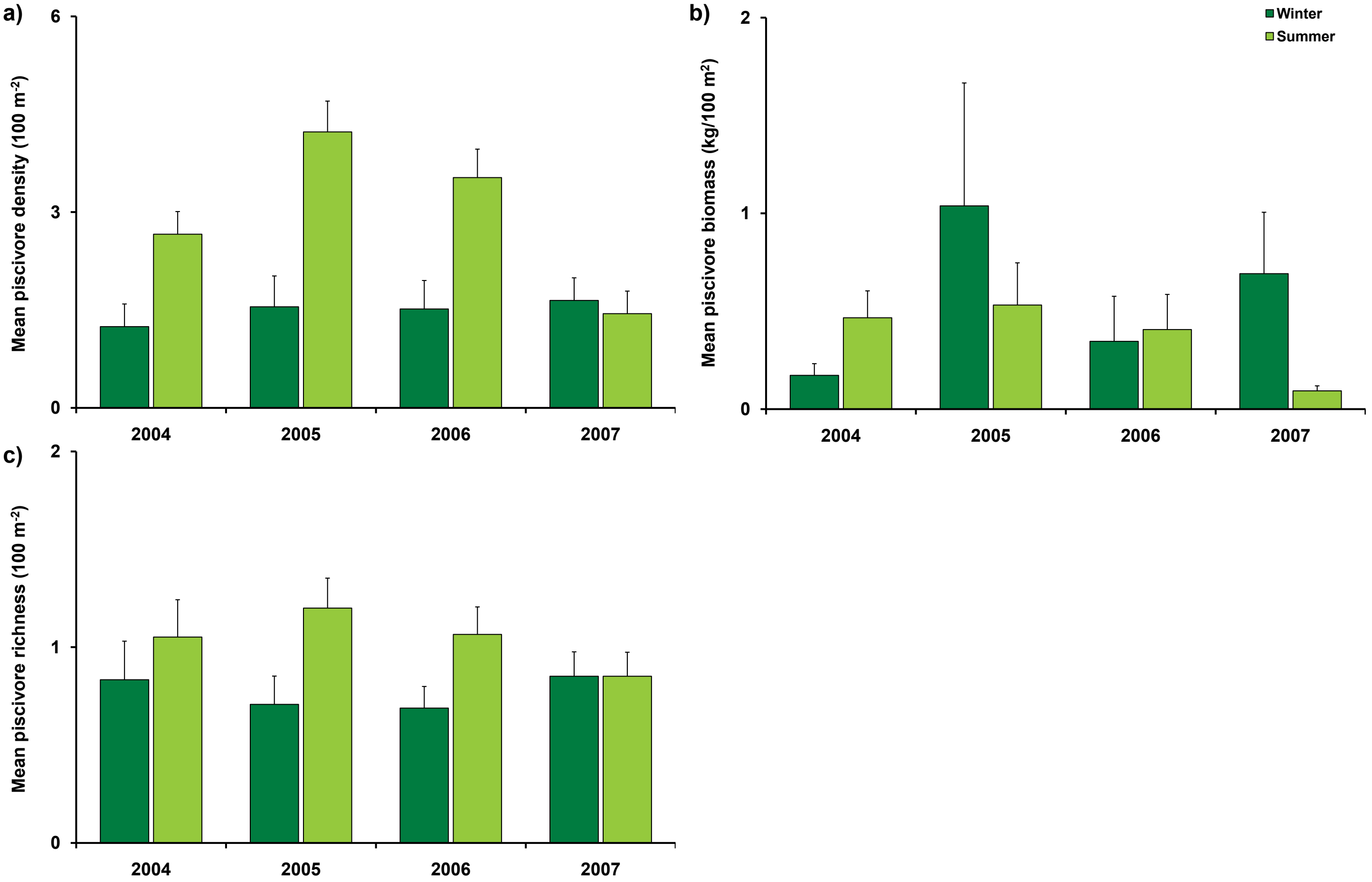


Figure 3.119. Seasonal change in mean (± SE) piscivore: (a) density, (b) biomass and (c) richness in the southwest Puerto Rico study area.

Comparison between trophic groups

Mean densities of herbivorous and planktivorous fish for winter and summer were a maximum of more than 30 times greater than the mean density of piscivores (Figures 3.118a, 3.120a and 3.119a). Herbivorous fish comprised approximately half of the mean total fish biomass for both winter and summer sampling periods (Figures 3.118b and 3.117b), although piscivore biomass appeared to follow the trend in total mean biomass more closely, with alternating years of increasing and decreasing of winter and summer values between years and between seasons (Figures 3.119b and 3.117b). The 2005 winter sampling period had the highest values for all biomass metrics except for herbivore biomass. The 2006 winter sampling period had the lowest values for planktivore and herbivore biomass (Figures 3.120b and 3.118b) and summer of 2007 had the lowest for total biomass and piscivore biomass (Figures 3.117b and 3.119b).

Mean richness of piscivores and planktivores (Figures 3.119c and 3.120c) followed the total trend values (Figure 3.117c) with the highest means occurring during the summer sampling period. Herbivore richness values were five times higher than piscivore richness during both winter and summer sampling periods (Figures 3.118c and 3.119c). Planktivore richness was approximately two times higher than piscivore richness (Figures 3.120c and 3.119c) and herbivore richness was approximately two and a half times higher than planktivore richness (Figures 3.118c and 3.120c).

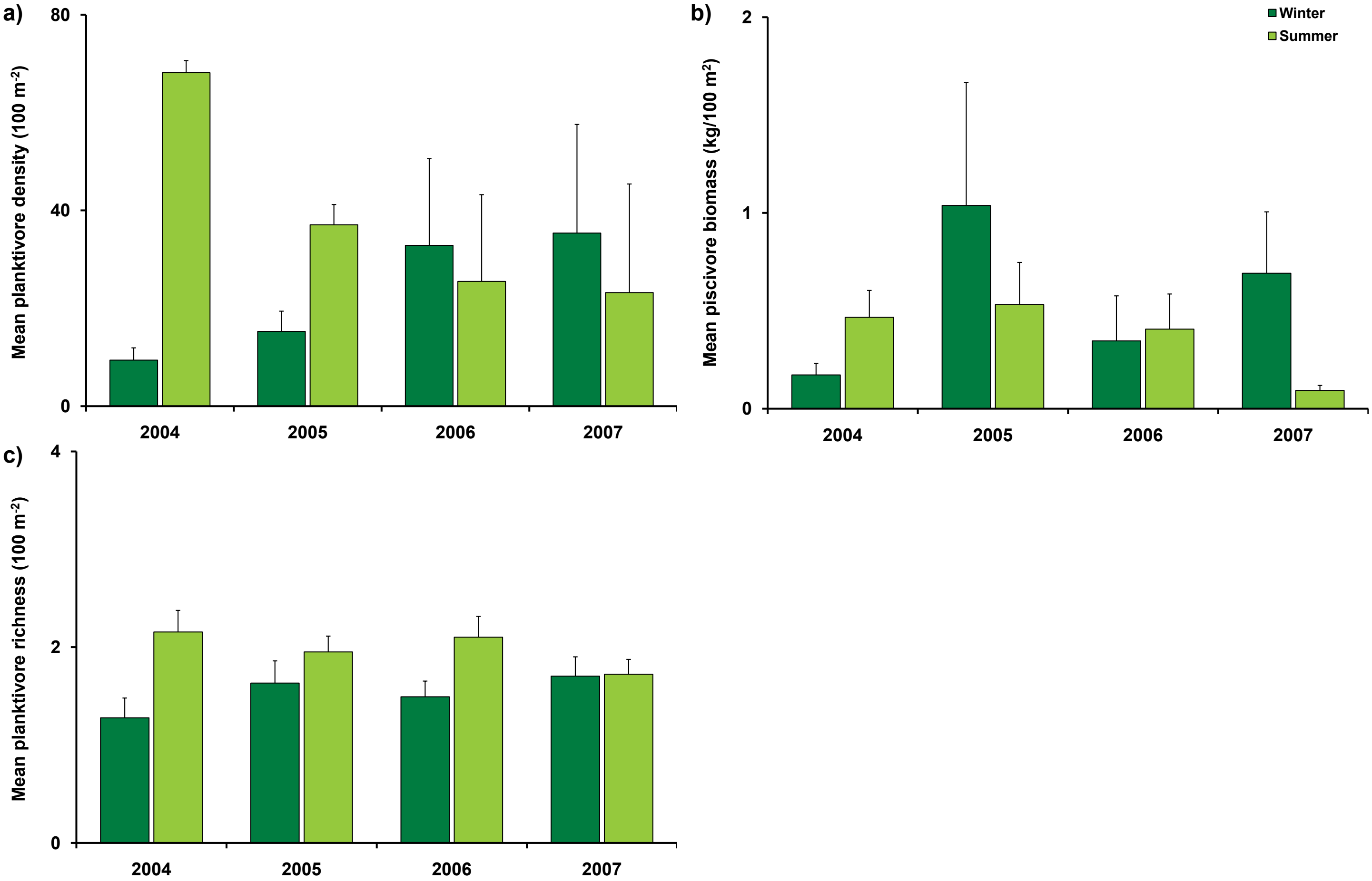


Figure 3.120. Seasonal change in mean (+ SE) planktivore: (a) density, (b) biomass and (c) richness in the southwest Puerto Rico study area.

3.5.2. Taxonomic groups

Overall family comparisons

The seasonal trends in biomass for four major fish families were examined including groupers, snappers, parrotfish and grunts.

Mean biomass for all four major families (grouper, snapper, parrotfish, grunts) was lower in the winter than summer, with lowest mean biomass for grouper, snapper and parrotfish during the 2006 winter sampling period (Figure 3.121). Mean biomass of grunts was highest during summer 2006 and lowest in winter 2005, although there was little difference between winter and summer in 2004, 2006 and 2007 (Figure 3.121d). Grouper biomass was lowest in both winter and summer of 2006 (Figure 3.121a). Mean biomass of parrotfish was higher in the winter months except in 2006 (Figure 3.121c). Winter biomass values decreased from 2004-2006, then increased in 2007, whereas summer months were variable. In contrast, snapper mean biomass values were higher in the summer months (except 2005; Figure 3.121b). The highest and lowest mean biomass snappers occurred in 2005 (i.e., highest in the winter, lowest in summer).

Large-body Groupers (Serranidae)

The lowest mean biomass values for the three selected groupers (*C. cruentata*, *C. fulva*, *E. guttatus*) occurred during 2006 (Figure 3.122). *C. cruentata* and *E. guttatus* exhibited lowest mean biomass in winter 2006 and *C. fulva* in summer 2006 (Figure 3.122). Low mean biomass values for *C. cruentata* and *E. guttatus* coincided during the same year and season (winter 2006) as mean biomass of total groupers (Figure 3.122a). The years of highest mean biomass, however, occurred at different times; *C. cruentata* in summer 2004, *C. fulva* in winter 2005 and *E. guttatus* in winter 2004 (Figure 3.122). Overall, there appeared to be little difference in mean biomass values for selected groupers throughout the years regardless of season.

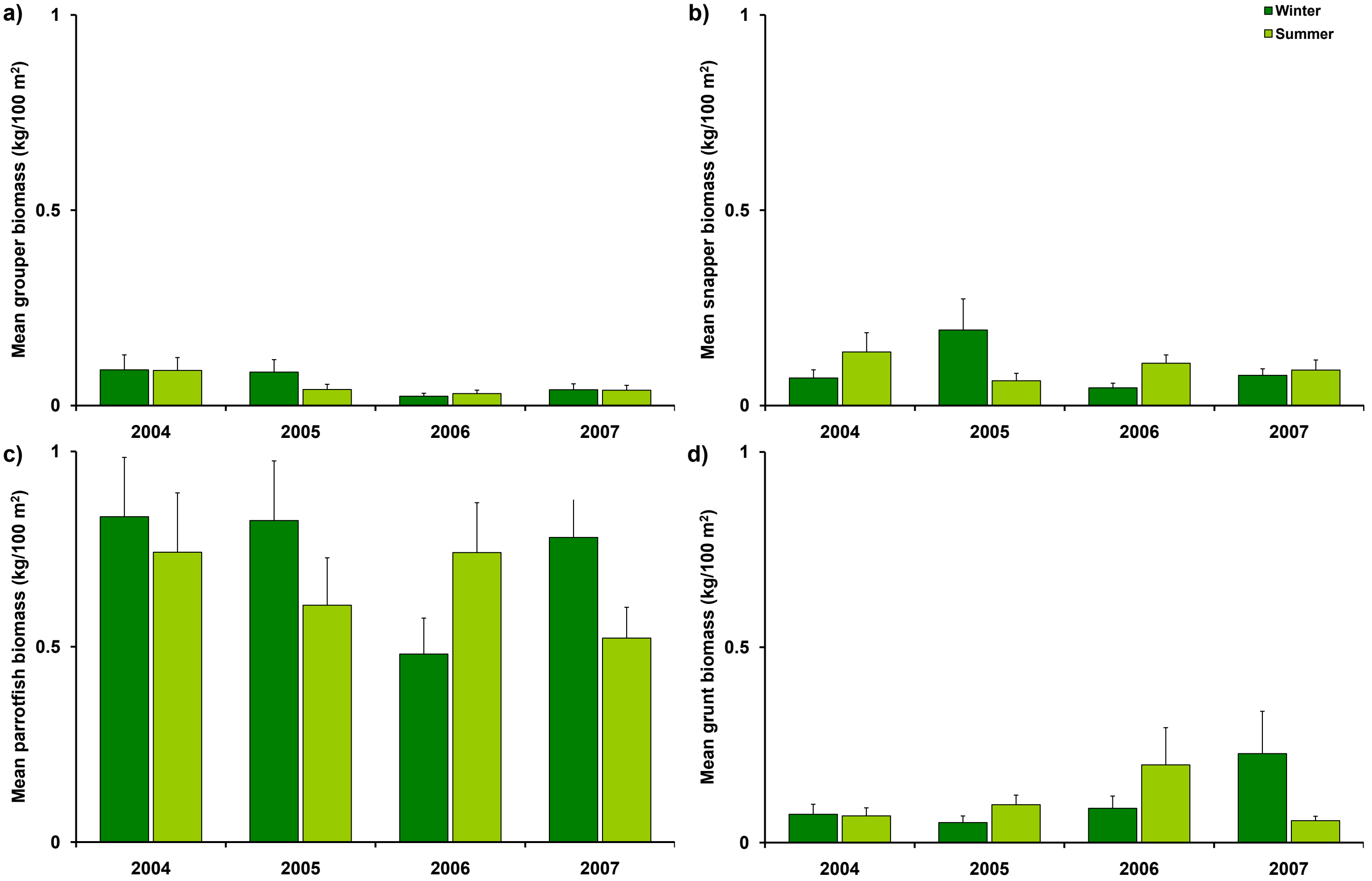


Figure 3.121. Seasonal change in mean (+ SE) fish biomass in the southwest Puerto Rico study area for: (a) groupers, (b) snappers, (c) parrotfish and (d) grunts.

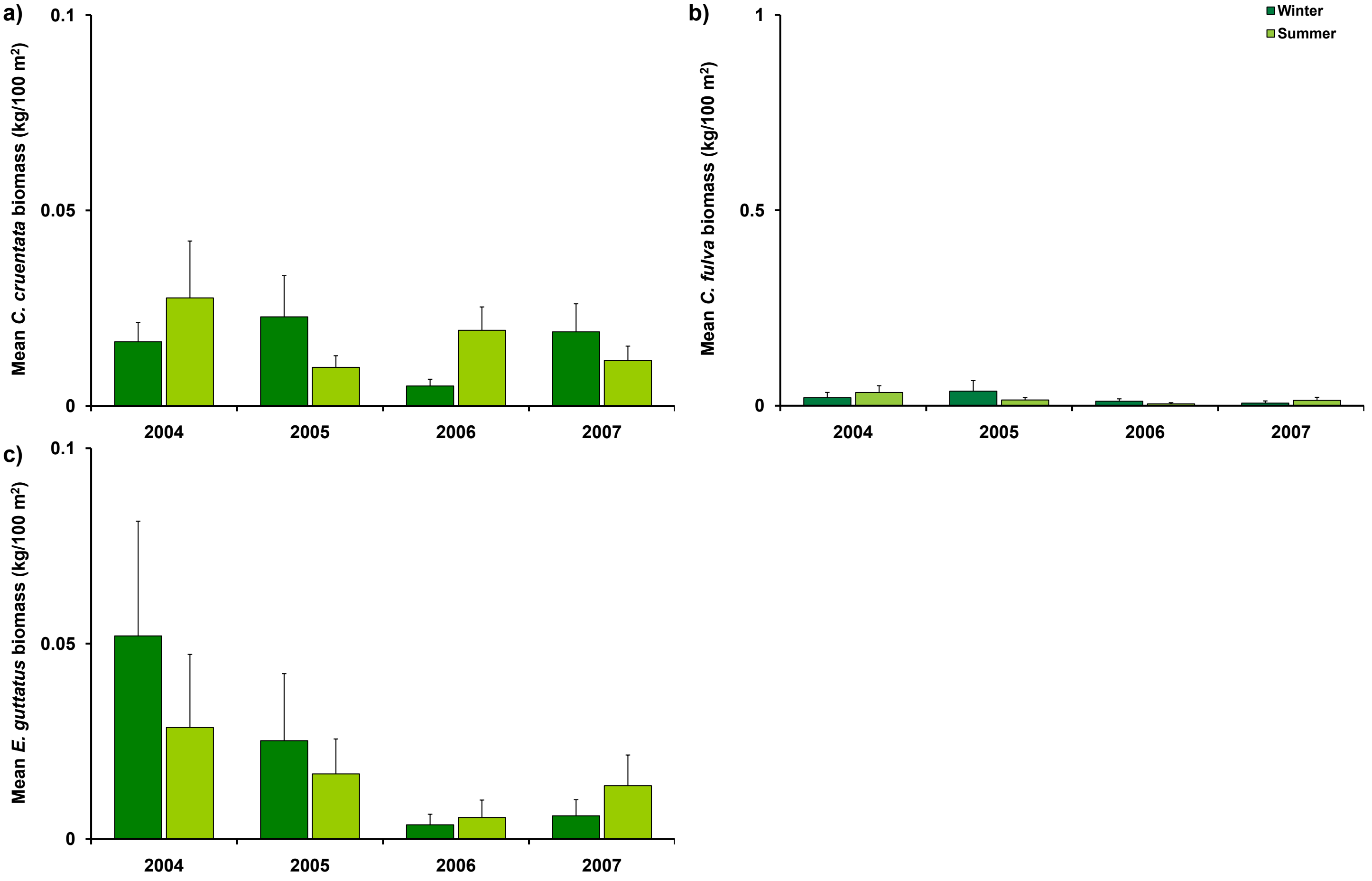


*Figure 3.122. Seasonal change in mean (± SE) grouper biomass in the southwest Puerto Rico study area for: (a) graysby (*C. cruentata*), (b) coney (*C. fulva*) and (c) red hind (*E. guttatus*).*

Snappers (Lutjanidae)

Highest mean biomass for *L. griseus* and *O. chrysurus* was recorded in winter 2005 and *L. apodus* had the highest mean biomass in summer 2004 (Figure 3.123). The lowest mean biomass for all three species occurred during the winter sampling period; *L. apodus* and *L. griseus* in winter 2004 and *O. chrysurus* in winter 2006. It is likely that had mangroves been sampled in summer 2005 that *L. griseus* biomass would be higher for that period.

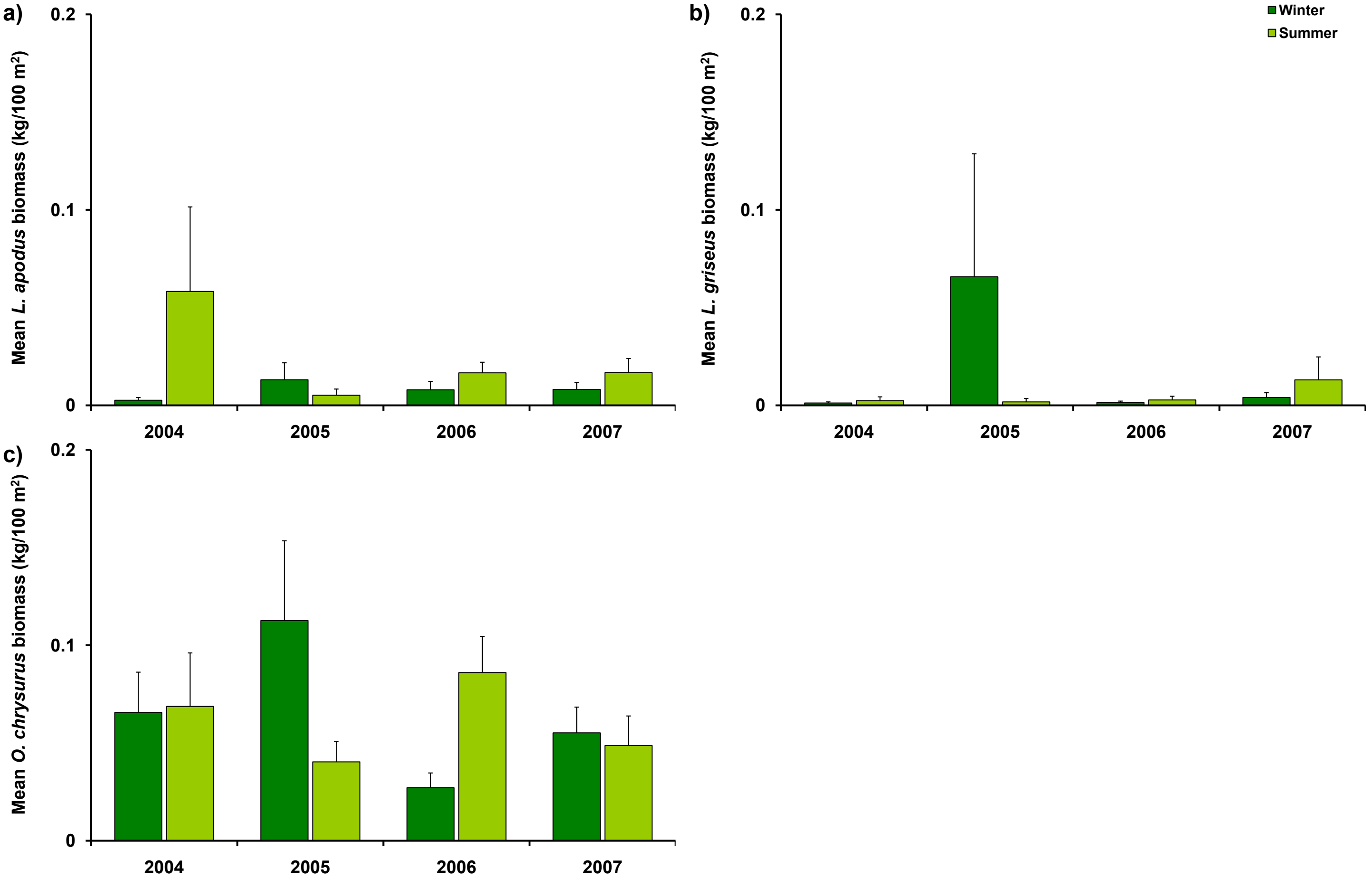


*Figure 3.123. Seasonal change in mean (± SE) snapper biomass in the southwest Puerto Rico study area for: (a) schoolmaster (*L. apodus*), (b) gray snapper (*L. griseus*) and (c) yellowtail snapper (*O. chrysurus*).*

Parrotfish (Scaridae)

All four species exhibited a comparatively low mean biomass during the summer 2007 sampling period (Figure 3.124). For *S. aurofrenatum* and *S. viride*, winter 2006 was the period with lowest mean biomass during the time series (Figure 3.124c,d). Mean biomass of *S. iseri* was the highest in winter 2007 and the lowest in summer 2004 (Figure 3.124a), whereas *S. taeniopterus* had the exact opposite trend, having the highest mean biomass in summer 2004 and the lowest mean in winter 2007 (Figure 3.124b). Overall, *S. iseri* biomass values were higher during the winter sampling periods than summer, except during 2006 (Figure 3.124a). *S. taeniopterus* had higher means in the summer sampling period, except in 2005. Mean biomass of *S. viride* was higher in the winter season from 2004-2007, except for 2006 (Figure 3.124d). In 2005, mean biomass for *S. viride* in 2005 differed very little between seasons. Mean biomass of *S. viride* during the winter sampling period decreased from 2004-2006, then increased in 2007; while values during the summer sampling period increased from 2004-2005 then decreased from 2005-2007 (Figure 3.124d).

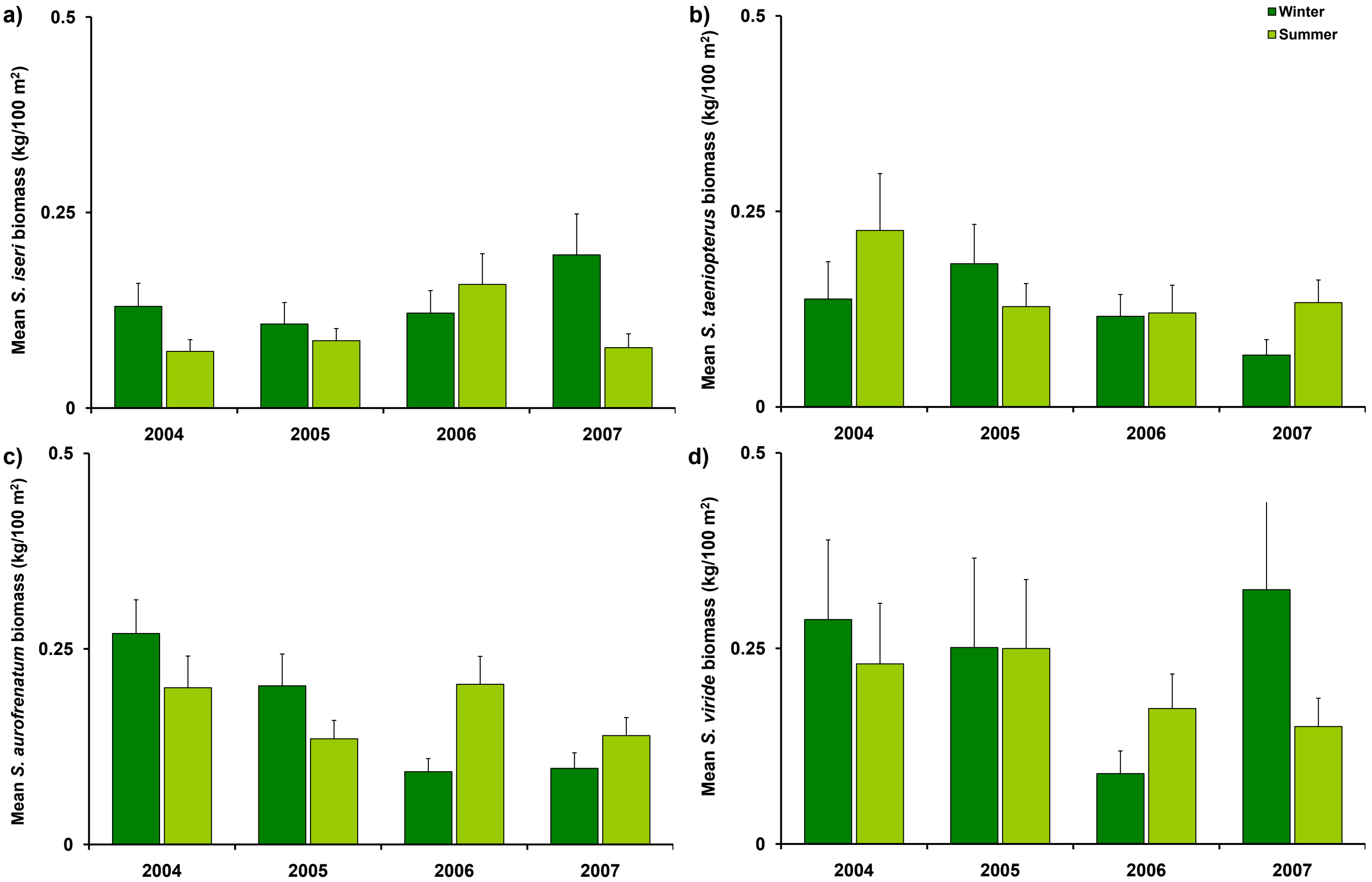


*Figure 3.124. Seasonal change in mean (± SE) parrotfish biomass in the southwest Puerto Rico study area for: (a) striped parrotfish (*Sc. iseri*), (b) princess parrotfish (*Sc. taeniopterus*), (c) redband parrotfish (*Sp. aurofrenatum*) and (d) stoplight parrotfish (*Sp. viride*).*

Grunts (Haemulidae)

Two of the three selected grunt species (*H. flavolineatum* and *H. plumierii*) had the highest biomass mean occur during the summer 2006 sampling period; whereas, the highest mean biomass for *H. sciurus* during the 2004 winter sampling period (Figure 3.125). None of the selected grunt species shared the same sampling period for lowest mean biomass. Summer biomass values for *H. flavolineatum* were higher than all winter values except for 2007 (Figure 3.125a). Additionally, the highest and lowest mean biomass values for *H. flavolineatum* occurred during the same year. From 2004-2006 *H. flavolineatum* winter biomass values were less than 0.02 kg/100m^2 and summer biomass values were greater than 0.02 kg/100m^2; the highest winter biomass and lowest summer biomass values occurred in 2007 (Figure 3.125a).

The highest *H. plumierii* biomass values for winter and summer sampling periods were in 2006, and the lowest biomass values were recorded in 2004; with summer 2004 having the lowest biomass values across all years (Figure 3.125b). *H. sciurus* mean biomass values were higher during the winter sampling periods for all years except for 2006 (Figure 3.125c). The lowest mean biomass for *H. sciurus* was during the summer 2005 sampling period.

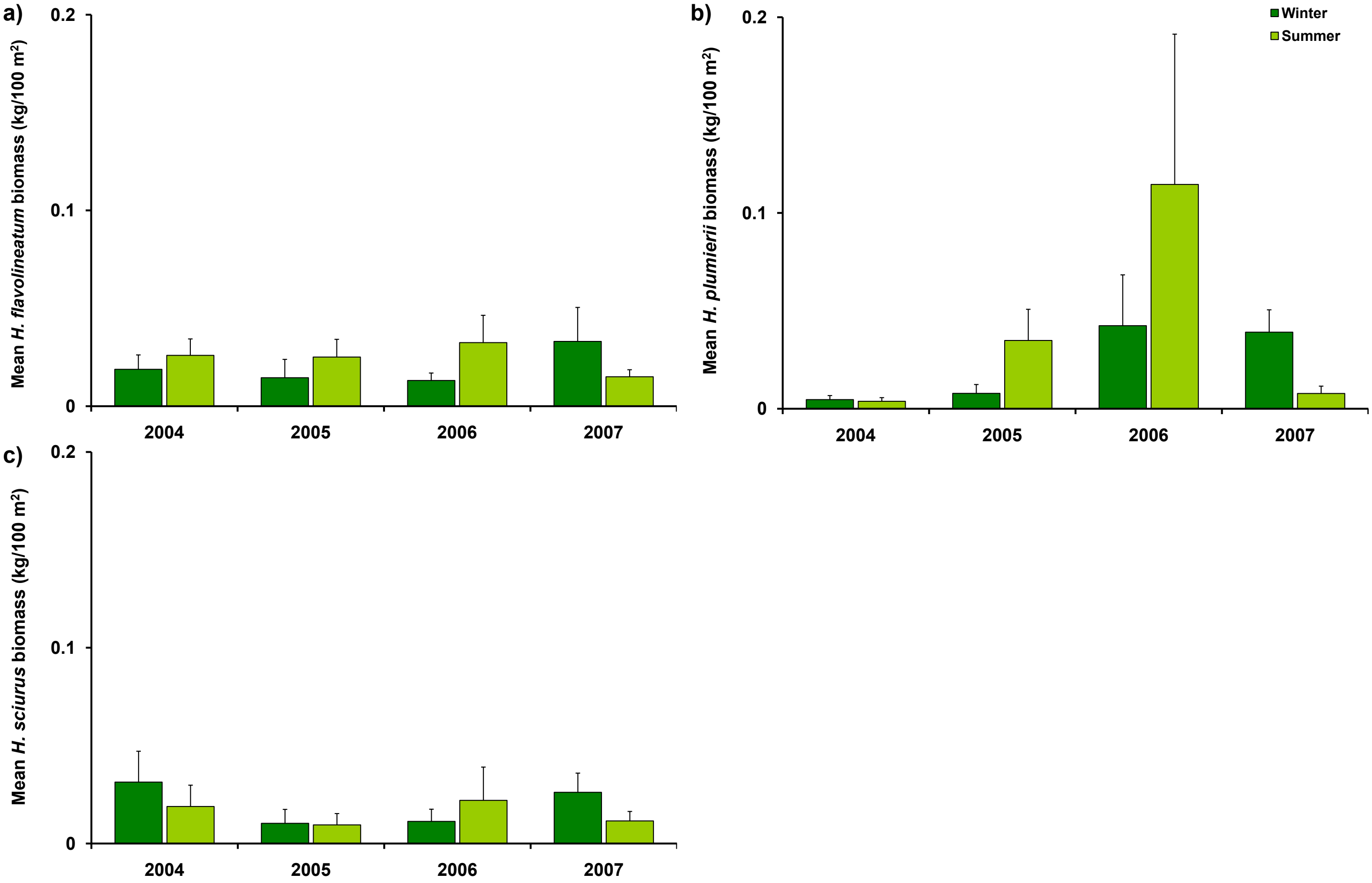


*Figure 3.125. Seasonal change in mean (± SE) grunt biomass in the southwest Puerto Rico study area for: (a) French grunt (*H. flavolineatum*), (b) white grunt (*H. plumierii*) and (c) bluestriped grunt (*H. sciurus*).*

Other species

The two goatfish (Mullidae) species recorded in the southwest Puerto Rico study area were *M. martinicus* and *P. maculatus*. There appeared to be little change in mean biomass from 2004-2007 for either goatfish species, with the exception of winter 2007 for *M. martinicus* (Figure 3.126a). The highest and lowest means for *M. martinicus* occurred during the winter sampling seasons, with the highest mean biomass recorded in winter 2007 and the lowest in winter 2004 (Figure 3.126). *P. maculatus* exhibited an opposite pattern, with the highest and lowest means occurring during the summer months, highest in summer 2006 and the lowest in summer 2004 (Figure 3.126b).

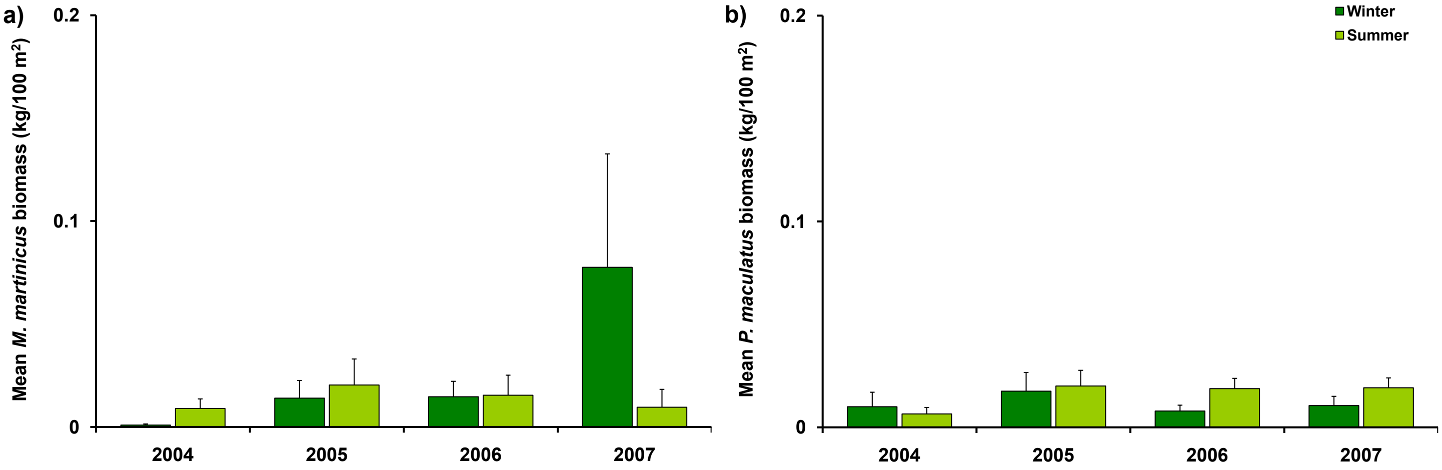


*Figure 3.126. Seasonal change in mean (± SE) goatfish biomass in the southwest Puerto Rico study area for: (a) yellow goatfish (*M. martinicus*) and (b) spotted goatfish (*P. maculatus*).*

Mean biomass for *B. vetula* was highest in summer 2007 and lowest in summer 2004 (Figure 3.127a) and appears to have increased gradually in both winter and summer seasons over the time series. Mean biomass for *S. barracuda* was highly variable between seasons and years, with very low values (<0.0001 kg/100m^2) in the winters of 2004 and 2006 and summer of 2007. The biomass means for the winter sampling period were variable with alternating increase and decrease, while biomass in the summer sampling period were more consistent from 2004-2006 (Figure 3.127b).

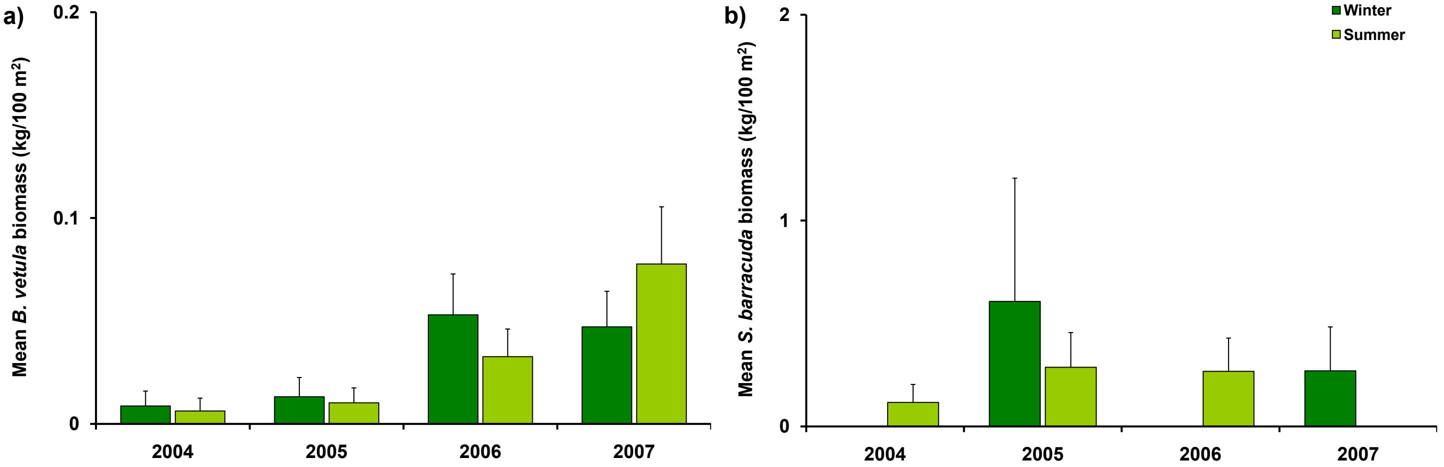


*Figure 3.127. Seasonal and inter-annual (2003-2006) change in mean (± SE) biomass in the southwest Puerto Rico study area for: (a) queen triggerfish (*B. vetula*) and (b) great barracuda (*S. barracuda*).*

3.6. Historical comparison of species occurrence between 1980-1981 and 2001-2007

Kimmel (1985) used timed underwater visual census techniques to survey the fish communities of the La Parguera region associated with 21 biotopes across the insular shelf (e.g., mangroves, seagrasses, high relief shoreward reefs, high relief shelf edge reefs, etc.). Kimmel identified over 250 species in 168 samples between 1980 and 1981. The visual fast count method was used of 50 minute timed surveys, with data recorded within 10 minute survey periods. Eight locations were surveyed within each biotope. These data do not provide comparable quantitative samples from which densities can be calculated and compared with CCMA-BB data, but if large differences occurred in the frequency of occurrence for particular species in 1980 compared with 2001-2007, then this is likely to be indicative of real change due to the spatially comprehensive across shelf sampling inherent in both studies.

In 1980-1981, rainbow parrotfish (*S. guacamaia)* was observed by Kimmel at 50% of samples from the eight mangrove sites surveyed (mean abundance 2.79); 25% of high relief shoreward coral reef samples (mean abundance 3.36); 38% of low relief coral reef (mean abundance 2.75); 31% of low relief shoreward reefs (mean abundance 6.10) and 6% of high relief shelf edge reefs (mean abundance 0.15). The data reveal that in the early 1980s, the species occurred widely across the shelf from mangroves to shelf edge habitat types. In contrast, only two individuals of *S. guacamaia* were observed between 2001 and 2007 from 1,167 samples in the La Parguera region, thus confirming the rarity of the species in the region and supporting its status as vulnerable according to the IUCN red list of threatened species (Figure 3.128).

*Figure 3.128. A spearfisherman with a 9.78 kg rainbow parrotfish (*Scarus guacamaia*) caught in Puerto Rico (exact location unknown) on January 29th 2005. Source: International Underwater Spearfishing Association.*

Midnight parrotfish (*S. coelestinus)* was less frequently observed in 1980 and 1981 than rainbow parrotfish with sightings at 19% of high relief shoreward reef samples (mean abundance 0.95); 19% low relief apron reef samples (mean abundance 0.52); 25% low relief shoreward reef samples (mean abundance 1.67); 19% high relief shelf edge samples (mean abundance 0.94). In contrast, between 2001 and 2007, no midnight parrotfish were observed in the study area even though the range of habitat types sampled included those where the species was observed in 1980 and 1981. The status of this species urgently requires evaluation in the region.


Midnight parrotfish (*Scarus coelestinus*) Source: Kevin Eddy

In 1980 and 1981, Nassau grouper (*E. striatus*) was observed at 44% of high relief outer reef samples (mean abundance 1.23); 25% high relief shoreward reef samples (mean abundance 0.57); 6% low relief apron samples (0.07); 25% low relief shoreward reef samples (mean abundance 0.89); 50% of high relief shelf edge samples (mean abundance 1.80). In contrast, between 2001 and 2007 only two individual Nassau grouper were observed in the La Parguera study area indicating that populations have been depleted since 1980-81 and are not showing signs of recovery even though the species has been legally protected as a “Species of Concern” since 1991 under the Endangered Species Act.

In 1980 and 1981, *M. tigris* was observed only at 17% of high relief shelf edge reef samples (mean abundance 0.73). Between 2001 and 2007, *M. tigris* were not recorded on any transects in the study area suggesting that the population of this species may have declined further since 1980-1981. Historical comparison of other grouper species suggest that *E. adscensionis* was more frequently sighted over coral reefs in 1980-1981 (17.9%) than in 2001-07 (0.4%); *C. fulva* at 36.4% in 1980-1981 versus 7.8% in 2001-2007; *E. guttatus* at 64.7% in 1980-1981 versus 7.5% in 2001-07; and *C. cruentata* at 58.8% in 1980-1981 versus 29% in 2001-2007.

In 1980 and 1981, *O. chrysurus* an important species to regional fisheries, was observed at 78.4% of samples across hard, softbottom and mangroves, with highest abundance over seagrasses and low relief shoreward reef sites. In contrast, although *O. chrysurus* were the most abundant snapper in the study area in 2001-2007, they were observed at only 27.4% of all sites sampled. A recent published analyses using a portion of the 2001-2007 data for southwest Puerto Rico has shown that yellowtail snapper is one of several overfished species based on the body-length distribution and spawning potential (Ault et al., 2008). Kimmel’s data do not provide size estimates for the 1980-1981 period, thus it is not possible to tell the proportion of adults and juveniles. Based on the most recent and best-available data, it appears that the population size may have declined in the past 25 years and thus this species should be carefully monitored and protected from fishing if it is to recover. In addition, declines were found for other snapper species. *L. jocu* was observed at 8.3% of all samples across the shelf in 1980-81 and at 0.8% in 2001-2007. *L. synagris*, a fished species, occurred at 12.2% of samples in 1980-1981 and 3.9% in 2001-2007. *L. mahogoni* was observed at 47.4% of the coral reef samples in 1980 versus 2.9% in 2001-2007.

In 1980 and 1981, *B. vetula*, a popular local food fish, was observed at 20% of coral reef samples, with highest abundance at high relief offshore reefs. Although similar habitat preferences were observed in 2001-2007, the percent occurrence had declined by half to approximately 10% occurrence across the shelf.

Furthermore, in 1980-1981, *S. planifrons* (an indicator species of complex and live coral cover) occurred at 46% of all coral reef samples across the shelf, with highest abundance and occurrence at high and low relief shoreward reefs (91% and 81% respectively) and high relief outer reefs (50%). During 2001-

2007, the percentage occurrence of *S. planifrons*, however, was lower, with *S. planifrons* present in 31% (n=167) of hardbottom samples (n=541). To determine whether these differences are an artifact of the different sampling designs or a real shift due to fishing and changes in the structural complexity of coral reefs and the decline in live coral cover will require further studies.

3.7. Summary of results

3.7.1. Fish assemblage composition

- When comparing fish assemblage composition between major mapped habitat types (softbottom, hardbottom and mangroves), greatest difference was found between mangroves and hardbottom habitat types. The higher abundance of Haemulidae (grunts), Lutjanidae (snappers) and small-bodied schooling fish (*Jenkinsia* spp. and *Atherinomorus* spp.) in mangroves contributed most to dissimilarities. Thus, the results indicated that benthic habitat maps classified at the coarse thematic resolution of mangroves, hard and softbottom are ecologically meaningful and can be used to determine spatial differences in fish community composition.
- At finer levels of the hierarchical classification scheme (i.e., finer thematic resolution such as linear reefs, patch reefs, etc.), fish communities were not easily separated by mapped habitat types. This suggests that other characteristics including topographic complexity may need to be incorporated into mapped classes to better predict the distribution of fish communities.
- At the finest thematic resolution where softbottom habitat types were classified by the amount of seagrass cover, fish assemblages associated with continuous seagrasses were most distinct when compared with assemblages associated with lower proportions of seagrass cover, with a possible threshold effect at the level of >90% seagrass cover.

3.7.2. Multi-habitat use

- Our multi-habitat surveys of fish revealed that 12 of the 30 most abundant fish species observed in mangroves have also been observed over seagrasses, coral reefs and unvegetated sediments. Twenty-five of 30 species were observed in both mangroves and coral reefs indicating a high level of multi-habitat use by common fish species in the La Parguera region.
- For approximately 50% of the 30 most abundant fish in mangroves, fish body length was markedly smaller in mangroves than individuals of the same species on coral reefs, indicative of size dependent ontogenetic habitat shifts, particularly for Haemulidae, Lutjanidae, Scaridae and Sphyraenidae (barracuda).
- Based on body size, mangroves appeared to function as an intermediate habitat type for some grunts and snappers, with smallest fish associated with seagrasses, larger fish in mangroves and the largest mean length recorded for fish on coral reefs.

3.7.3. Spatial patterns of fish diversity, biomass and abundance

- Across the La Parguera study area a total of 210 fish species were identified to species level, with at least another 14 fishes identified only to genera.
- Highest species richness of 41 fish per 100 m^2 area was recorded near the eastern shelf edge over colonized pavement with sand channels.
- Hotspots of high fish species richness, high fish biomass and high herbivore abundance and richness co-occurred along the shelf edge, particularly along the central to eastern edge of the surveyed area and around the complex of patch reefs between Margarita Reef and the El Palo area.
- Highest fish abundance was associated with mangroves, both inshore and around offshore mangrove cays, with the assemblages composed mostly of juveniles indicating that mangroves of the La Parguera region are an important resource for juvenile fish.

- Although *S. guacamaia* is thought to have a high dependence on mangroves and coral reefs, only two individuals were observed between 2001 and 2007 in the La Parguera region confirming the rarity of the species in the U.S. Caribbean and its status as vulnerable according to the IUCN red list of threatened species. No midnight or blue parrotfish were sighted within the surveyed areas between 2001 and 2007.
- The small to medium-bodied parrotfish species, (*S. iseri* and *S. aurofrenatum*) were the most commonly occurring fish across the seascapes at La Parguera.
- *O. chrysurus* was the most abundant and widespread snapper species and the 11th most commonly occurring fish species (738 individuals) in the La Parguera study area found at 27.4% (n=320) of survey sites.
- The largest snappers seen in the study area were markedly smaller than the maximum known size for the species, particularly *L. apodus*, *O. chrysurus* and *L. griseus*.
- Small-bodied groupers were more abundant than large-bodied grouper and included *C. cruentata* (n=246) and *C. fulva* (n=81). Abundance and biomass of small-bodied grouper species was highest along the shelf edge, with a maximum density of seven grouper recorded at one site (100 m2).
- None of the grouper species observed had attained the maximum known size for their species, with maximum length for Cephalopholis species in the La Parguera region estimated at 30 cm FL compared with a maximum known for *C. fulva* of 41 cm TL and 43 cm TL for *C. cruentata*. The largest *E. guttatus* was approximately 50% of the maximum recorded size.
- From a total of 1,167 surveys (572 from hardbottom) over seven years, no *M. tigris* or *M. venenosa* were observed and only two *E. striatus*, two *M. bonaci*, two *E. adscensionis* and 43 *E. guttatus* were observed, even though a known spawning aggregation for *E. guttatus*, *E. adscensionis, M. tigris* and *M. venenosa* exists along the shelf edge in southwest Puerto Rico.
- Five sharks (two species) and three stingrays were the only sharks and rays observed within transects at La Parguera between 2001 and 2007.

3.7.4. Habitat and ontogenetic space use patterns

- *O. chrysurus* juveniles occurred in 10 of the 11 habitat types, with highest abundance over colonized hardbottom. Juveniles showed no spatial segregation from adults, but adults were more frequently observed in the mid and outer shelf zones than in lagoonal areas.
- Parrotfish (Scaridae) exhibited high spatial heterogeneity in distributions across the shelf, with highest abundance over the most topographically complex hardbottom. Juveniles and adults co-occurred across the study area.
- *S. taeniopterus* exhibited a preference for mid and outer shelf zones with highly contiguous coral reef, while striped parrotfish occurred in all zones across the shelf.
- Grunt (Haemulidae) abundance and biomass was highest in the nearshore and offshore mangroves in close proximity to coral reefs and seagrass beds. All common species of grunt showed a strong across shelf size-dependent distribution, with the majority of juveniles in lagoonal nearshore areas and adults in deeper mid- and outer-shelf zones. Juveniles and adults did also co-occur at several sites across the shelf indicating some flexibility in the strategy for ontogenetic segregation.
- Damselfish (Pomacentridae) species *S. planifrons, M. chrysurus* and *S. variabilis* abundance is highest on topographically complex coral reefs and can be used as a good indicator of complex coral reefs.
- *B. vetula*, a fished species in the La Parguera region, exhibited highest abundance and biomass in the topographically complex colonized pavement with sand channels habitat type of the outer shelf zone.
- *S. barracuda* exhibited a strong size segregated spatial distribution, with juveniles confined to nearshore lagoonal areas and most adults found in the mid and outer shelf zones.

3.7.5. Temporal patterns in fish abundance and biomass

- None of the fish metrics decreased for more than three consecutive years within the seven year monitoring period (2001-2007), but 14 metrics decreased for at least three consecutive years prior to 2006.
- Grouper biomass in 2007 was significantly lower than in 2001.
- Only *C. fulva* and *S. aurofrenatum* increased in abundance consecutively for more than three years between 2001 and 2007.
- Total fish biomass, herbivore biomass, grouper abundance, parrotfish and wrasse abundance were significantly higher in 2007 than in 2001.
- Total snapper density and *L. apodus* density decreased from 2002 to 2005.
- The most striking inter-annual difference occurred between 2003 and 2004, whereby 65 metrics (approximately 80% of all metrics) decreased, with five decreasingly significantly; followed by 70% of metrics increasing the following year (2004-2005).
- Sixty-five percent of the 20 most abundant fish exhibited higher density in summer than in winter. *T. bifasciatum* and *C. cyanea* mean density was more than 50% higher in summer. However, *H. aurolineatum*, *H. flavolineatum* and *S. iseri* were more abundant in winter.
- Mean biomass for grouper, snapper, parrotfish and grunts was lower in the winter than summer.
- Planktivores and piscivores were more abundant in summer than winter.
- Comparison of the number of times a species was sighted across the shelf at La Parguera in 1980-81 versus 2001-07 indicated that the population sizes of several fished species have declined substantially over the past 25 years. These include *S. guacamaia, S. coelestinus, E. striatus, M. tigris, E. adscensionis, E. guttatus, C. fulva, E. cruentata, O. chrysurus, L. synagris, L. mahogoni, L. jocu and B. vetula*. These are all relatively large-bodied and important food fish for the region.

3.8. Discussion

3.8.1. Fish community composition and mapped habitat types

Benthic habitat maps are often produced for multiple purposes including characterizing the distribution, abundance and diversity of biological resources to support decision making in marine management. For instance, benthic maps often play a central role in site identification and prioritization for marine protected areas, delineating essential fish habitat in ecosystem-based management and zoning in marine spatial planning, but rarely are the classifications evaluated for their meaningfulness to key faunal groups. In Australia, Ward et al. (1999) examined the utility of habitat maps as surrogates for biodiversity and concluded that the mapped classes were good surrogates for biodiversity. This project evaluated the utility of the NOAA benthic habitat map classes for predicting differences in fish community composition across multiple habitat classes at different levels of a hierarchical map classification. It was expected that fish communities would differ significantly among habitat types due to the perceived structural differences between habitat types observed during aerial photo interpretation and via underwater observations, which originally resulted in the delineation and classification of distinct map classes.

In fact, our characterization and analysis revealed statistically significant compositional differences in the fish assemblages only when samples were grouped at the coarsest thematic resolution (softbottom, hardbottom and mangroves). Of all pairwise habitat comparisons, fish communities associated with mangroves and fish associated with hardbottom habitats showed highest dissimilarity (least overlap). This was primarily due to the higher abundances of grunts and of small-bodied schooling species such as Atherinids, Clupeids and Engraulids within mangroves compared with hardbottom areas. Similarly, Chittaro et al. (2005) in Belize found low similarity between the fish community composition of mangroves and coral reefs and the authors interpreted this as indicative of low connectivity between the two habitat types. Community similarity/dissimilarity, however, should not be used as a measure of relative connectivity since it focuses on assemblage level patterns and not individual species movements. For example, high exchange between habitat types can occur for specific species, yet the community compositions for the two habitat types can still be very dissimilar.

At finer thematic resolutions considerable overlap was detected in fish community composition, particularly within hardbottom habitat types, where high similarity occurred between geomorphologically different classes such as patch reefs, linear reefs and colonized pavement. Although differences sometimes occurred they were not statistically significant and thus we conclude that the assemblage composition did not respond strongly to the different biophysical structures observed in the aerial photography and subsequently delineated by the cartographer. With regard to the use of habitat maps for predicting diversity patterns, the coarsest levels of the classifications scheme also performed best. In an earlier study focusing on predicting fish species richness in southwest Puerto Rico, Pittman et al. (2007b) coupled Biogeography Branch fish survey data with seascape structure from the benthic habitat map to examine between-habitat differences in fish species richness. The results from a simple regression tree model revealed that the greatest difference in fish species richness was found between softbottom and hardbottom sites, with pairwise comparisons between habitat types within the softbottom grouping and between habitat types within the hardbottom grouping being mostly non-significant. Some of the overlap in community similarity may be explained by high within-habitat type variability making habitat types less distinct as has been described in similar studies elsewhere in the Caribbean (Harborne et al., 2008; Mumby et al., 2008). Recent work in St Croix (U.S. Virgin Islands) has demonstrated that both the spatial resolution of a benthic map and the thematic resolution selected for a study can influence the model of fish-habitat relationships (Kendall and Miller, 2008; 2009).


Baitfish in mangrove habitat

Interestingly though, at the finest thematic resolution, where softbottom habitat types were classified by the amount of seagrass cover our study did reveal some differences. Fish assemblages associated with continuous seagrasses were most distinct when compared with assemblages associated with samples from several sites with incrementally lower proportions of seagrass cover, suggesting that a threshold effect may be occurring at the level of >90% seagrass cover. Very few studies have examined fish communities across a gradient in seagrass cover to identify threshold effects in the faunal-seagrass relationship. In seagrass beds of Moreton Bay (Australia) a gradual decline in resident fish abundance was detected, along spatial gradients in seagrass cover, until approximately 15–20% seagrass cover, beyond which many abundant species were absent (Pittman et al., 2004). In the Caribbean (Virgin Islands, Florida Keys, and Turks and Caicos), an examination of fish communities on coral reefs along a spatial gradient in seagrass cover revealed that fish diversity and abundance increased from 0 to 20–30% seagrass coverage then plateaued at 40% indicating a threshold like response (Grober-Dunsmore, 2005). Critical thresholds imply that absence, loss, or degradation of habitat can have deleterious effects on the distribution and population dynamics of some species, and are therefore crucially important to studies predicting the impact of environmental change in tropical marine systems. Further studies are required to determine if a threshold effect occurs in the relationship between fish and seagrass cover since other factors could influence the patterns along a gradient including


Colonized hardbottom habitat

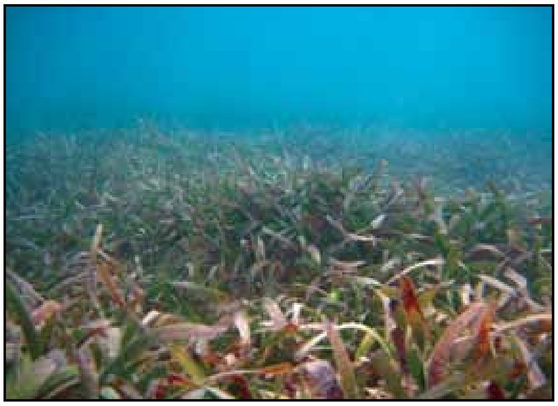
the surrounding habitat types (i.e., proximity to mangroves and coral reefs), water quality and the type of seagrasses. It is likely that the very sparse seagrasses function more like unvegetated sand in that they have a more similar structure and refuge function, sediment chemistry, food resources and other conditions than do highly vegetated areas with dense and long-leaved seagrasses and muddy organic rich sediments.

3.8.2. Multi-habitat utilization

Although direct evidence of habitat connectivity cannot be directly inferred from our underwater surveys, the current report does demonstrate that many fish species use multiple habitat types in the La Parguera region. By the term “use” we are referring to space use i.e., the fish were observed over specific habitat types and were thus associated with them. Our multi-habitat surveys of fish revealed that 12 of the 30 most abundant fish species observed in mangroves also used seagrasses, coral reefs and unvegetated sediments. As many as 25 of 30 species were observed in both mangroves and coral reefs, indicating a high level of multi-habitat use by many of the most common fish species. Additional analyses revealed a strong linear relationship (r^2=0.77) between the number of habitat types used by a species and its total abundance across the region. As such, fish species that have evolved to use all habitat types (seascape generalists) are also the most abundant species across the region. These seascape relationships require further study and need to be evaluated relative to the implications for resource management.


Mangrove habitat

For some species, distinctive size distribution patterns across habitat types were revealed suggesting that combinations of habitat types may provide a sequence of ontogenetic stepping stones from settlement habitat to adult habitat. Based on body size, mangroves appeared to function as an intermediate habitat type for some grunts and snappers, with smallest fish associated with seagrasses, larger fish in mangroves and the largest mean length recorded for fish on coral reefs. This size-specific habitat use pattern has also been observed for grunts and snappers in Florida (Serafy et al., 2007), Belize (Mumby et al., 2004) and Puerto Rico (Appeldoorn et al., 1997; Aguilar-Perera and Appeldoorn, 2007; 2008). The strong evidence for multiple habitat use in tropical marine fish requires that we move away from a single habitat approach when studying ecological relationships toward a seascape approach that considers mosaics of habitat types (Pittman and McAlpine, 2003). The problem is pervasive and in the scientific literature multi-habitat fish species are routinely assigned a single habitat descriptor based on a perceived habitat association regardless of their movements and resource requirements. For example, a multi-habitat species is often separately classified as a “seagrass fish species”, a “coral reef fish species”, and a “mangrove fish species” depending on the habitat surveyed and the focus of the research project. While single habitat descriptors may be appropriate for habitat specialists that are resident within a single habitat type, they are inappropriate for species that are capable of using more than one habitat type (habitat generalists) and particularly misleading for individuals that are critically dependent on multiple habitat types through daily home range movements and ontogenetic habitat shifts. Single habitat discriptors are also routinely misused when applied to assemblages and communities (i.e., coral reef fish community, mangrove fish community) since habitat association is typically based on only a snapshot of their entire space-use patterns captured during relatively fine-scale surveys. The fact that some species may have a critical


Seagrass habitat

dependence on one or a specific combination of habitat types, while others have adapted generalist strategies able to use a wide range of habitat types at all life stages adds significant complexity to the conventional view on fish-habitat relationships and strategies for delineation of essential fish habitat (EFH) and design of marine protected areas (MPAs).

3.8.3. Spatial patterns of fish diversity, biomass and abundance

Hotspots of high fish species richness, high fish biomass and high herbivore abundance and richness occurred along the shelf edge, particularly along the central to eastern edge of the surveyed area. Edges are often associated with elevated productivity and diversity since species from either side of the edge or from adjacent patch types intermingle. The shelf edge is the boundary between shallower shelf environments and the deeper water slope areas creating heterogeneous habitats. A higher diversity of planktivorous species is typical of shelf edges due to stronger currents and upwelling carrying plankton rich waters along and over the edge. In addition, the shelf edge is characterized by high topographic complexity of the seafloor, which provides higher surface area for habitat and more abundant refuge space for a larger number and diversity of species including large-bodied fish such as grouper. Additional diversity and abundance hotspots were identified around the complex of patch reefs between Margarita Reef and the El Palo area. The recent availability of LiDAR bathymetry (airborne laser altimetry) has allowed us to develop a 3-dimensional surface for the La Parguera seafloor. Examination of the 3-dimensional model of bathymetry derived from hydrographic LiDAR data reveals a complex aggregation of patch reefs over the seafloor around the El Palo reef. It is likely that this high structural complexity together with the geographical location on the shelf near extensive seagrasses and mangroves supports the diverse and productive fish assemblages observed. The shelf edge areas and the El Palo region also exhibited the highest species richness of herbivorous fish, an assemblage characteristic that can have beneficial effects on coral reef structure and function. Experimental studies by Burkepile and Hay (2008) demonstrated that multiple species of herbivorous fish lowered macroalgal abundance by 54–76%, enhanced cover of crustose coralline algae by 52–64%, increased coral cover by 22%, and prevented coral mortality.

The most abundant fish species were the small-bodied parrotfish species (*S. iseri* and *S. aurofrenatum*), confirming that herbivores numerically dominate the fish communities across the shelf at La Parguera. This could be indicative of a phase shift towards algae dominated seascapes. Further historical comparisons need to be carried out to determine if the abundance and distribution of these species has changed over the years and whether shifts in the numerically dominant species have occurred relative to environmental changes.


S. iseri

The highest abundance and biomass of parrotfish was recorded over the most topographically complex hardbottom across the study area, but species-specific distributions were evident too. For example, *S. taeniopterus* exhibited a preference for mid- and outer- shelf zones with highly contiguous coral reef, while striped parrotfish occurred in all zones across the shelf. In northeast St. Croix too, *S. taeniopterus* exhibited higher abundance across the deeper extensive colonized pavement areas seaward of Buck Island than did *S. iseri*. This appears to be related to a preference for structurally complex areas in deeper water.

3.8.4. Habitat and ontogenetic space use patterns

As discussed above, many species associated with coral reef ecosystems utilize multiple habitat types, often with very different biophysical structure (seagrasses, mangroves, coral reefs, etc.) and species composition. For some species the habitat types used can also be in geographically distinct zones across the shelf that may or may not correspond with changes in water depth. The patterns in across shelf distributions for specific species and the influence of size on these patterns was shown by Christensen et al. (2003) using the first year of data collected by the Biogeography Branch and was highlighted by Aguilar-Perera and Appeldoorn (2008). Christensen et al. (2003) indicated that habitat type was more important than cross-shelf location in determining the distributional patterns among fishes in the study area and Aguilar-Perera and Appeldoorn (2008) found a consistent spatial pattern with fish assemblages (based on the most abundant species) grouping into clusters associated with mangroves, seagrasses, shallow coral reefs and deeper water coral reefs, with densities and diversity highest over coral reefs than any other habitat type.

Building on this work through analysis of a greater number of samples with more extensive spatial coverage collected over seven years we show some distinctive spatial patterns both across the shelf and by habitat for juveniles and adults. Some key species exhibited spatial segregation between distribution patterns of juveniles and adults, while for other species juveniles and adults co-occurred at the same sites, habitat types and zones. Grunts (Haemulidae) showed a strong across shelf size-dependent distribution, with the majority of juveniles in lagoonal nearshore areas and adults in deeper mid- and outer-shelf zones. Juveniles and adults, however, also co-occured at several sites across the shelf indicating some flexibility in the dominant strategy of ontogenetic across-shelf segregation.

O. chrysurus juveniles occurred in 10 of the 11 habitat types, indicating that the juveniles are very generalist users of the seascape. Further research would be required to determine the relative importance of different habitat types on growth and survival. Although densities differ, adults and juveniles co-exist across the shelf at the same sites, habitats and zones and thus no geographically distinct nearshore nursery area was evident for *O. chrysurus* in southwest Puerto Rico as has been defined elsewhere in the Caribbean and referred to as “nursery species” by Nagelkerken (2000). Nagelkerken et al. (2000, 2001) defined “nursery species” as those species with high densities of juveniles in nearshore habitat types such as mangroves and seagrasses and with an almost complete absence of juveniles on the coral reefs where the adults were more abundant. Aguilar-Perera and Appeldoorn (2007) also showed that several species considered as nursery species by Nagelkerken in Curacao could not be considered as such in southwest Puerto Rico including *H. flavolineatum*, *H. sciurus*, H. plumerii, *L. apodus* and *O. chrysurus*. In the current study, juveniles of these species were recorded at higher densities in mangroves than elsewhere, but juveniles were also observed on mid and outer shelf edge zones. The geographical differences across the Caribbean may reflect real differences in habitat use strategies or may reflect differences in predation pressure or the amount of survey effort between studies. Furthermore, densities of fish in space are highly influenced by habitat structure and location and can be a misleading indicator of habitat quality and should not be used alone when assigning relative value to a specific habitat types (van Horne, 1983).

In addition, juvenile and adult parrotfish co-occurred across the study area. For these ubiquitous species where all life stages can co-occur, it is conceivable that although nearshore areas may be beneficial in supporting high abundance across the region, nearshore areas may not be critical to the maintenance of populations in the region. In contrast, some species showed geographically distinctive zones for juveniles and adults in line with the concepts defined by the nursery hypothesis. For example, *S. barracuda* exhibited a strong size segregated spatial distribution, with juveniles confined to nearshore lagoonal areas and most adults found in the mid and outer shelf zones. Other species, showed no segregation of juveniles and adults, but instead exhibited a strong apparent preference for the most topographically complex colonized hardbottom areas. For instance, damselfish (Pomacentridae) species *S. planifrons, M. chrysurus* and *S. variabilis* were distributed over the most topographically complex coral reefs and their presence and possibly abundance may be useful as a good indicator of complex coral reefs. *B. vetula* also exhibited highest

S. planifrons

abundance and biomass in the topographically complex colonized pavement with sand channels habitat type of the outer shelf zone.

3.8.5. Size structure and shifts in predators for vulnerable species

Declines in maximum body size and changes in the abundance of size classes over time can be indicative of fishing pressure. *O. chrysurus*, a highly prized food fish in the Caribbean was the 11th most commonly occurring fish species (738 individuals) in the La Parguera study area found at 27.4% of survey sites. The largest fish was estimated at 40 cm FL, therefore, none of these individuals had achieved anywhere near the maximum known size for the species (86.3 cm TL) and only 4% of all fish observed were more than 30 cm FL. For this species too, historical comparison is required to determine if changes in the size structure of the population have occurred. This is particularly important to determine whether yellowtail snapper should be listed as overfished by NOAA's National Marine Fisheries Service. Based on life history characteristics, *O. chrysurus* is classified as a species with relatively slow population doubling time (4.5-14 years) and low resilience to fishing (Froese and Pauly, 2010).


Nassau grouper (*Epinephelus striatus*)

Also of note was that small-bodied groupers were markedly more abundant than large-bodied groupers. With the exception of red hind, the large-bodied groupers were exceedingly rare across the shelf. From a total of 1,167 surveys over seven years, no *M. tigris* or *M. venenosa* were observed and only two *E. striatus* and two *M. bonaci* were observed. The largest *E. guttatus* was approximately 50% of the maximum recorded size and this species should receive close attention with regard to its population size structure and viability in the La Parguera region. In contrast, abundance of *C. cruentata* was an order of magnitude higher across the La Parguera study area than red hind and *C. fulva* were also recorded at higher densities than all of the larger-bodied grouper species combined. Furthermore, *C. cruentata* was the only grouper species that appeared to be increasing in density from 2002 to 2007.

Elsewhere in the Caribbean and Florida, shifts in the dominant predators have been attributed to fishing pressure, since large-bodied predators such as groupers are particularly vulnerable to fishing and when removed from the ecosystem are replaced with smaller, often less-targeted predators (Pauly et al., 1998). Studies by Chiappone in fished and unfished areas in Florida showed that the abundance and biomass of small-bodied groupers, including *C. fulva*, were higher in areas where fishing had reduced the large-bodied groupers (e.g., *Epinephelus* spp., *Mycteroperca* spp.). Our own studies in northeast St Croix, USVI. (Pittman et al., 2008) reported only four individuals of the very large-bodied and highly vulnerable grouper species, compared with 697 of the low to moderate vulnerability grouper (413 *C. fulva*, 231 *E. guttatus* and 53 *C. cruentata*). Comparison between fish monitoring surveys around Buck Island (St. Croix, U.S. Virgin Islands) conducted in 1979 (Gladfelter and Gladfelter, 1980) and a geographically similar subset of CCMA-BB monitoring surveys conducted between 2001 and 2006 reveal that while *E. striatus*, *M. venenosa* and *M. tigris* existed at low densities in 1979 they were absent between 2001 and 2006. In contrast, *E. guttatus* and *C. fulva* have increased in density significantly from 0.08 individuals/100 m^2 of red hind in 1979 to 0.18/100 m^2 in 2001-2006; and from 0.01/100 m^2 of *C. fulva* in 1979 to 0.30/100 m^2 in 2001-2006 (Table 12 in Pittman et al., 2008).

The consequences of such shifts may have cascading effects through the biological community. Studies by Stallings (2008, 2009) have shown that coney are more voracious predators of newly settled fish than are some of the larger-bodied species such as *E. striatus* and have been found to have a more significant impact on the recruitment to patch reefs for a wide range of common fish species. The ecological implication of an increase in small-bodied grouper on the fish community of the U.S.

Caribbean is currently unknown. Research is needed to determine if a shift in predators has occurred in the La Parguera region and then to determine what the ecological consequences are for the population and community dynamics of the region.

3.8.6. Temporal patterns in fish abundance and biomass

Our semi-annual long-term monitoring data revealed a distinctive seasonal pattern in the abundance of fish in southwest Puerto Rico with 65% of the 20 most abundant fish exhibiting higher densities in summer than winter. For two small bodied planktivorous fish species, *T. bifasciatum* and *C.cyanea*, the mean density was more than 50% higher in summer and this may reflect seasonal differences in food availability for planktivorous fish. Studies by Youngbluth (1980) in Jobos Bay, Puerto Rico found peaks in zooplankton biomass and copepod densities from August-October.

It is likely that the largest changes in abundance of large-bodied fish happened before the current CCMA-BB monitoring program (i.e., pre-2001). Synoptic analyses of trends over the entire sampling period showed that none of the fish metrics decreased for more than three consecutive years within the seven year monitoring period (2001-2007), but 14 species decreased for three consecutive years prior to 2006. The upturn from 2006-2007 could be an indication of a new trend and requires continuation of long-term monitoring to elucidate on future trends. This apparent increase at the end of the study period is also apparent among many other fish metrics with 13 of 15 metrics increasing consecutively over three years showing an increase in the latter half of the study period between 2004 and 2007. Total fish biomass, herbivore biomass, grouper abundance, parrotfish and wrasse abundance were significantly higher in 2007 than in 2001. *S. aurofrenatum* exhibited more than three consecutive years of increase. This pattern could benefit the health of coral reefs in the area since redband parrotfish are known to consume large quantities of several major macroalgal species from the genera *Dictyota, Halimeda, Lobophora, Sargassum, Haloplegma, Kallymenia* and *Codium* species (Burkepile and Hay, 2008). *Dictyota* and *Lobophera* spp. have been implicated in phase shifts from coral dominated reefs to macroalgal dominated reefs, thus an increase in a key consumer of *Dictyota* spp. and *Lobophora variegata* may play an important role in controlling macroalgal biomass. Furthermore, these fast growing macroalgal species also inhibit coral recruitment (Kuffner et al., 2006).

For the small-bodied grouper species, mean biomass was significantly higher in 2007 than it was in 2001. This increase could be due to increased food availability or due to the relatively low abundance of large piscivorous fish (i.e., low predation pressure). For example, the large-bodied groupers showed no increase over seven years and total snapper density and *L. apodus* density decreased from 2002-2005. Similarly, release of coney and graysby populations due to shifts in the dominant piscivorous fish (i.e. large groupers) on the reef have been documented elsewhere in the Caribbean (Stallings, 2008).

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Appendix A

Table A1. Finfish landings as a proportion of the total finfish landings reported for the U.S.Caribbean in 1980. Listed are the most commonly landed species and species groups. Data from the Caribbean Fisheries Management Council (CFMC, 1985).

Species/Species group	Fish Family	USVI % of total landings US Caribbean
Grunts	Haemulidae	0.47
Groupers	Serranidae	13.91
Goatfish	Mullidae	0.99
Parrotfish	Scaridae	5.83
Lane snapper (*L. synagris*)	Lutjanidae	0.03
Yellowtail snapper (*O.chrusurus*)	Lutjanidae	2.89
Triggerfishes	Balistidae	29.68
Squirrelfishes	Holocentridae	4.84
Mutton snapper (*L. analis*)	Lutjanidae	0.13
Other snappers	Lutjanidae	1.04
Hogfish	Labridae	1.06
Trunkfish	Ostraciidae	0.08

CFMC (Caribbean Fisheries Management Council). 1985. Fishery Management Plan, Final Environmental Impact Statement and Draft Regulatory Impact Review for the Shallow Water Reef Fish Fishery of Puerto Rico and the U.S. Virgin Islands. San Juan, PR. 179.pp. http://www.caribbeanfmc.com/

Appendix B

Table B1. Fish species list and summary data on occurrence, abundance and biomass (2001-2007) for the southwest La Parguera, Puerto Rico study region.

Family *Species name*	**Common name**	**% occurrence**	**Total occurrence**	**Total abundance**	**Mean abundance (± SE)**	**Total biomass, g**	**Mean biomass, g (± SE)**
Acanthuridae							
Acanthurus bahianus	ocean surgeonfish	41.9	489	2125	1.8 (0.10)	139867.3	119.9 (8.0)
Acanthurus chirurgus	doctorfish	15.6	182	630	0.54 (0.07)	63940.7	54.8 (11.5)
Acanthurus coeruleus	blue tang	24.9	291	963	0.83 (0.10)	93316.7	80.0 (17.5)
Acanthurus UNK	SURGEONFISH sp	0.2	2	3	<0.01 (<0.01)	2.3	(<0.01) (<0.01)
Apogonidae							
Apogon binotatus	barred cardinalfish	0.3	3	3	<0.01 (<0.01)	1.4	(<0.01) (<0.01)
Apogon maculatus	flamefish	0.3	4	9	(<0.01) (<0.01)	4.1	(<0.01) (<0.01)
Apogon quadrisquamatus	sawcheek cardinalfish	0.4	5	10	(<0.01) (<0.01)	4.6	(<0.01) (<0.01)
Apogon townsendi	belted cardinalfish	0.3	3	5	<0.01 (<0.01)	2.3	(<0.01) (<0.01)
Apogon UNK	CARDINALFISH sp	0.2	2	2	<0.01 (<0.01)	0.92	(<0.01) (<0.01)
Astrapogon stellatus	conchfish	0.1	1	2	<0.01 (<0.01)	1.0	(<0.01) (<0.01)
Atherinidae							
Atherinomorus UNK	SILVERSIDE sp	1.4	16	19533	16.7 (6.0)	5155.0	4.4 (1.6)
Aulostomidae							
Aulostomus maculatus	trumpetfish	4.0	47	48	0.04 (<0.01)	4554.5	3.9 (0.71)
Balistidae							
Balistes vetula	queen triggerfish	5.1	60	77	0.07 (0.01)	34089.5	29.2 (4.9)
Melichthys niger	black durgon	2.4	28	193	0.17 (0.05)	69209.4	59.3 (17.5)
Belonidae							
Belonidae UNK	NEEDLEFISH Family sp	0.2	2	5	<0.01 (<0.01)	84.7	0.07 (0.07)
Tylosurus crocodilus crocodilus	houndfish	0.1	1	4	<0.01 (<0.01)	164.7	0.14 (0.14)
Blenniidae							
Ophioblennius macclurei	redlip blenny	2.4	28	92	0.08 (0.02)	252.3	0.22 (0.05)
Bothidae							
Bothus lunatus	peacock flounder	0.3	3	3	<0.01 (<0.01)	393.4	0.34 (0.33)
Bothus ocellatus	eyed flounder	0.3	3	3	<0.01 (<0.01)	12.4	0.01 (<0.01)
Bothus UNK	LEFTEYE FLOUNDER sp	0.3	4	4	<0.01 (<0.01)	18.5	0.02 (<0.01)
Callionymidae							
Paradiplogrammus bairdi	lancer dragonet	1.5	17	21	0.02 (<0.01)	14.9	0.01 (<0.01)
Carangidae							
Carangoides bartholomaei	yellow jack	0.3	3	29	0.02 (0.02)	12943.0	11.1 (10.6)
Caranx crysos	blue runner	1.8	21	94	0.08 (0.03)	38365.9	32.9 (11.5)
Caranx latus	horse-eye jack	1.2	14	40	0.03 (0.01)	2371.5	2.0 (1.4)
Caranx lugubris	black jack	0.2	2	2	<0.01 (<0.01)	4059.9	3.5 (2.6)
Carangoides ruber	bar jack	12.3	144	861	0.74 (0.15)	13993.1	12.0 (2.2)
Caranx UNK	JACK sp	0.1	1	2	<0.01 (<0.01)	205.5	0.18 (0.18)
Decapterus macarellus	mackerel scad	0.9	11	428	0.37 (0.19)	13947.1	12.0 (4.9)
Decapterus UNK	SCAD sp	0.3	4	260	0.22 (0.12)	8884.3	7.6 (3.9)
Selar crumenophthalmus	bigeye scad	0.3	3	162	0.14 (0.11)	17212.0	14.7 (10.4)
Trachinotus goodei	palometa	0.1	1	2	<0.01 (<0.01)	1021.8	0.88 (0.88)
Carcharhinidae							
Galeocerdo cuvier	tiger shark	0.1	1	1	<0.01 (<0.01)	92976.9	79.7 (79.7)
Chaenopsidae							
Acanthemblemaria aspera	roughhead blenny	0.1	1	1	<0.01 (<0.01)	0.20	(<0.01) (<0.01)
Acanthemblemaria maria	secretary blenny	2.3	27	43	0.04 (<0.01)	8.6	(<0.01) (<0.01)
Acanthemblemaria spinosa	spinyhead blenny	0.5	6	6	(<0.01) (<0.01)	1.2	(<0.01) (<0.01)
Acanthemblemaria UNK	TUBE BLENNY sp	2.1	24	35	0.03 (<0.01)	7.0	(<0.01) (<0.01)
Chaenopsis limbaughi	yellowface pikeblenny	1.0	12	12	0.01 (<0.01)	13.6	0.01 (<0.01)
Chaenopsis ocellata	bluethroat pikeblenny	1.1	13	29	0.02 (0.01)	75.5	0.06 (0.03)
Chaenopsis UNK	PIKEBLENNY sp	0.4	5	6	(<0.01) (<0.01)	4.0	(<0.01) (<0.01)
Emblemaria pandionis	sailfin blenny	0.9	10	11	(<0.01) (<0.01)	2.2	(<0.01) (<0.01)
Emblemariopsis UNK	BLENNY sp	0.6	7	8	(<0.01) (<0.01)	1.6	(<0.01) (<0.01)
Chaetodontidae							
Chaetodon capistratus	foureye butterflyfish	42.2	493	1526	1.3 (0.06)	23585.6	20.2 (1.5)
Chaetodon ocellatus	spotfin butterflyfish	0.5	6	6	(<0.01) (<0.01)	159.1	0.14 (0.07)
Chaetodon sedentarius	reef butterflyfish	0.3	4	4	<0.01 (<0.01)	26.2	0.02 (0.01)
Chaetodon striatus	banded butterflyfish	5.4	63	93	0.08 (0.01)	2358.6	2.0 (0.37)
Prognathodes aculeatus	longsnout butterflyfish	0.2	2	2	<0.01 (<0.01)	13.8	0.01 (0.01)

Table B1 cont...

Family *Species name*	**Common name**	**% occurrence**	**Total occurrence**	**Total abundance**	**Mean abundance (± SE)**	**Total biomass, g**	**Mean biomass, g (± SE)**
Cirrhitidae							
Amblycirrhitus pinos	redspotted hawkfish	0.6	7	7	(<0.01) (<0.01)	5.7	(<0.01) (<0.01)
Clupeidae							
Clupeidae UNK	HERRING Family sp	2.2	26	5897	5.1 (2.04)	2084.2	1.8 (1.4)
Jenkinsia UNK	HERRING sp	6.9	80	130548	111.9 (23.5)	6268.8	5.4 (1.1)
Congridae							
Heteroconger longissimus	brown garden eel	1.2	14	342	0.29 (0.11)	12233.7	10.5 (4.6)
Dactylopteridae							
Dactylopterus volitans	flying gurnard	0.1	1	1	<0.01 (<0.01)	5.1	(<0.01) (<0.01)
Dasyatidae							
Dasyatis americana	southern stingray	0.3	3	3	<0.01 (<0.01)	1163.2	1.0 (0.70)
Diodontidae							
Diodon holocanthus	balloonfish	4.4	51	76	0.07 (0.01)	3691.2	3.2 (1.3)
Diodon hystrix	porcupinefish	2.0	23	28	0.02 (<0.01)	26105.0	22.4 (7.6)
Echeneidae							
Echeneis naucrates	sharksucker	1.1	13	18	0.02 (<0.01)	4559.2	3.9 (2.4)
Echeneis neucratoides	whitefin sharksucker	1.1	13	15	0.01 (<0.01)	1558.3	1.3 (0.42)
Engraulidae							
Engraulidae UNK	ANCHOVIES Family sp	0.2	2	1160	0.99 (0.87)	181.7	0.16 (0.14)
Ephippidae							
Chaetodipterus faber	atlantic spadefish	0.2	2	59	0.05 (0.05)	4301.6	3.7 (3.5)
Gerreidae							
Eucinostomus gula	silver jenny	3.8	44	109	0.09 (0.02)	1129.8	0.97 (0.20)
Eucinostomus melanopterus	flagfin mojarra	8.2	96	468	0.40 (0.07)	674.1	0.58 (0.15)
Eucinostomus UNK	MOJARRA sp	0.5	6	20	0.02 (<0.01)	37.8	0.03 (0.02)
Gerres cinereus	yellowfin mojarra	9.3	108	397	0.34 (0.05)	3095.8	2.7 (0.43)
Ginglymostomatidae							
Ginglymostoma cirratum	nurse shark	0.3	4	4	<0.01 (<0.01)	30007.4	25.7 (20.0)
Gobiidae							
Bathygobious soporator	frillfin goby	0.3	4	8	(<0.01) (<0.01)	36.5	0.03 (0.02)
Bollmannia boqueronensis	white-eye goby	0.8	9	14	0.01 (<0.01)	9.8	(<0.01) (<0.01)
Coryphopterus dicrus	colon goby	4.2	49	91	0.08 (0.01)	59.8	0.05 (<0.01)
Coryphopterus eidolon	pallid goby	1.3	15	24	0.02 (<0.01)	15.8	0.01 (<0.01)
Coryphopterus glaucofraenum	bridled goby	28.1	328	1758	1.5 (0.15)	1292.5	1.1 (0.13)
Coryphopterus lipernes	peppermint goby	1.8	21	91	0.08 (0.02)	59.7	0.05 (0.01)
Coryphopterus personatus/ hyalinus	masked/glass goby	6.0	70	1299	1.1 (0.24)	851.4	0.73 (0.16)
Microgobius UNK	GOBY sp	0.1	1	4	<0.01 (<0.01)	0.85	(<0.01) (<0.01)
Ctenogobius saepepallens	dash goby	9.4	110	1307	1.1 (0.24)	856.7	0.73 (0.16)
Ctenogobius stigmaticus	marked goby	0.3	3	26	0.02 (0.02)	17.0	0.01 (0.01)
Elacatinus chancei	shortstripe goby	1.8	21	32	0.03 (<0.01)	8.0	(<0.01) (<0.01)
Elacatinus dilepis	orangesided goby	2.0	23	59	0.05 (0.02)	14.8	0.01 (<0.01)
Elacatinus evelynae	sharknose goby	20.1	235	605	0.52 (0.05)	151.6	0.13 (0.01)
Elacatinus saucrus	leopard goby	0.2	2	8	(<0.01) (<0.01)	2.0	(<0.01) (<0.01)
Coryphopterus UNK	GOBY sp	0.1	1	2	<0.01 (<0.01)	1.3	(<0.01) (<0.01)
Gnatholepis thompsoni	goldspot goby	19.5	228	1068	0.92 (0.09)	388.3	0.33 (0.09)
Gobiidae UNK	GOBY Family sp	1.7	20	227	0.19 (0.08)	148.8	0.13 (0.05)
Gobiosoma grosvenori	rockcut goby	0.2	2	2	<0.01 (<0.01)	1.3	(<0.01) (<0.01)
Lophogobius cyprinoides	crested goby	0.7	8	21	0.02 (<0.01)	139.0	0.12 (0.10)
Microgobius carri	seminole goby	3.4	40	240	0.21 (0.06)	66.8	0.06 (0.02)
Microgobius signatus	microgobius signatus	0.1	1	1	<0.01 (<0.01)	0.21	(<0.01) (<0.01)
Elacatinus UNK	GOBY sp	0.1	1	1	<0.01 (<0.01)	0.25	(<0.01) (<0.01)
Nes longus	orangespotted goby	15.3	179	1502	1.3 (0.13)	4391.5	3.8 (0.48)
Oxyurichthys stigmalophius	spotfin goby	1.5	18	48	0.04 (0.01)	135.8	0.12 (0.04)
Priolepis hipoliti	rusty goby	0.4	5	5	<0.01 (<0.01)	1.9	(<0.01) (<0.01)
Grammatidae							
Gramma loreto	fairy basslet	7.5	88	503	0.43 (0.07)	251.8	0.22 (0.04)
Haemulidae							
Anisotremus virginicus	porkfish	6.5	76	121	0.10 (0.01)	7769.2	6.7 (1.3)
Haemulon aurolineatum	tomtate	9.1	106	2016	1.7 (0.68)	21637.9	18.5 (7.5)
Haemulon carbonarium	caesar grunt	1.8	21	46	0.04 (0.01)	2236.6	1.9 (0.50)
Haemulon chrysargyreum	smallmouth grunt	1.4	16	115	0.10 (0.05)	1865.6	1.6 (0.53)

Table B1 cont...

Family *Species name*	**Common name**	**% occurrence**	**Total occurrence**	**Total abundance**	**Mean abundance (± SE)**	**Total biomass, g**	**Mean biomass, g (± SE)**
Haemulon flavolineatum	French grunt	27.2	317	5114	4.4 (0.58)	65514.5	56.1 (8.4)
Haemulon macrostomum	Spanish grunt	0.5	6	6	(<0.01) (<0.01)	305.0	0.26 (0.14)
Haemulon parra	sailors choice	2.3	27	72	0.06 (0.02)	2186.9	1.9 (0.62)
Haemulon plumierii	white grunt	15.1	176	846	0.72 (0.13)	31558.1	27.0 (5.1)
Haemulon sciurus	bluestriped grunt	18.2	212	2238	1.9 (0.23)	75884.2	65.0 (10.2)
Haemulon UNK	GRUNT sp	10.0	117	5104	4.4 (1.07)	2045.1	1.8 (0.41)
Holocentridae							
Holocentrus adscensionis	squirrelfish	5.7	67	127	0.11 (0.02)	16165.8	13.9 (2.8)
Holocentrus rufus	longspine squirrelfish	28.1	328	784	0.67 (0.07)	65778.2	56.4 (5.1)
Myripristis jacobus	blackbar soldierfish	3.8	44	113	0.10 (0.02)	10040.1	8.6 (2.4)
Neoniphon marianus	longjaw squirrelfish	0.7	8	9	(<0.01) (<0.01)	223.3	0.19 (0.08)
Sargocentron coruscum	reef squirrelfish	0.2	2	3	<0.01 (<0.01)	76.0	0.07 (0.06)
Sargocentron vexillarium	dusky squirrelfish	0.7	8	10	(<0.01) (<0.01)	190.9	0.16 (0.08)
Kyphosidae							
Kyphosus sectatrix	chub (bermuda/yellow)	0.3	3	3	<0.01 (<0.01)	126.4	0.11 (0.10)
Labridae							
Bodianus rufus	Spanish hogfish	3.0	35	47	0.04 (<0.01)	4071.8	3.5 (0.88)
Clepticus parrae	creole wrasse	2.0	23	506	0.43 (0.17)	14832.7	12.7 (6.6)
Doratonotus megalepis	dwarf wrasse	0.3	3	4	<0.01 (<0.01)	0.92	(<0.01) (<0.01)
Halichoeres bivittatus	slippery dick	27.1	316	1989	1.7 (0.14)	12810.1	11.0 (1.2)
Halichoeres cyanocephalus	yellowcheek wrasse	0.1	1	1	<0.01 (<0.01)	0.30	(<0.01) (<0.01)
Halichoeres garnoti	yellowhead wrasse	27.7	323	2127	1.8 (0.14)	13213.9	11.3 (0.91)
Halichoeres maculipinna	clown wrasse	8.7	101	242	0.21 (0.03)	1504.9	1.3 (0.25)
Halichoeres pictus	rainbow wrasse	0.6	7	25	0.02 (0.01)	198.4	0.17 (0.09)
Halichoeres poeyi	blackear wrasse	14.7	171	547	0.47 (0.05)	2376.2	2.0 (0.24)
Halichoeres radiatus	puddingwife	3.4	40	57	0.05 (<0.01)	1096.4	0.94 (0.26)
Lachnolaimus maximus	hogfish	3.0	35	43	0.04 (<0.01)	3396.5	2.9 (0.69)
Thalassoma bifasciatum	bluehead	34.1	398	5995	5.1 (0.40)	12013.7	10.3 (1.1)
Xyrichtys martinicensis	rosy razorfish	3.7	43	586	0.50 (0.27)	1671.7	1.4 (0.89)
Xyrichtys novacula	pearly razorfish	0.3	3	6	(<0.01) (<0.01)	20.1	0.02 (0.01)
Xyrichtys splendens	green razorfish	8.3	97	388	0.33 (0.05)	1815.1	1.6 (0.32)
Labrisomidae							
Labrisomus nuchipinnis	hairy blenny	0.2	2	2	<0.01 (<0.01)	8.9	(<0.01) (<0.01)
Malacoctenus aurolineatus	goldline blenny	0.5	6	10	(<0.01) (<0.01)	6.8	(<0.01) (<0.01)
Malacoctenus boehlkei	diamond blenny	0.7	8	8	(<0.01) (<0.01)	1.9	(<0.01) (<0.01)
Malacoctenus macropus	rosy blenny	4.3	50	107	0.09 (0.02)	72.5	0.06 (0.01)
Malacoctenus triangulatus	saddled blenny	2.1	25	34	0.03 (<0.01)	11.7	0.01 (<0.01)
Malacoctenus UNK	SCALY BLENNY sp	0.3	4	4	<0.01 (<0.01)	1.4	(<0.01) (<0.01)
Malacoctenus versicolor	barfin blenny	0.1	1	2	<0.01 (<0.01)	0.48	(<0.01) (<0.01)
Lutjanidae							
Lutjanus analis	mutton snapper	0.3	4	4	<0.01 (<0.01)	825.3	0.71 (0.47)
Lutjanus apodus	schoolmaster	18.8	219	1974	1.7 (0.19)	103463.1	88.7 (11.3)
Lutjanus buccanella	blackfin snapper	0.1	1	2	<0.01 (<0.01)	3.0	(<0.01) (<0.01)
Lutjanus cyanopterus	cubera snapper	0.1	1	1	<0.01 (<0.01)	2814.4	2.4 (2.4)
Lutjanus griseus	gray snapper	8.1	94	741	0.63 (0.15)	56346.6	48.3 (9.6)
Lutjanus jocu	dog snapper	0.8	9	13	0.01 (<0.01)	4253.9	3.6 (1.3)
Lutjanus mahogoni	mahogany snapper	2.2	26	54	0.05 (0.01)	3221.6	2.8 (0.75)
Lutjanus synagris	lane snapper	3.9	46	89	0.08 (0.01)	4069.8	3.5 (0.83)
Lutjanus UNK	SNAPPER sp	1.0	12	43	0.04 (0.02)	228.7	0.20 (0.15)
Ocyurus chrysurus	yellowtail snapper	27.4	320	738	0.63 (0.05)	71432.6	61.2 (5.5)
Malacanthidae							
Malacanthus plumieri	sand tilefish	3.7	43	58	0.05 (<0.01)	11013.5	9.4 (2.4)
Microdesmidae							
Ptereleotris helenae	hovering goby	3.6	42	100	0.09 (0.02)	109.6	0.09 (0.04)
Monacanthidae							
Cantherhines macrocerus	american whitespotted filefish	0.2	2	7	(<0.01) (<0.01)	321.3	0.28 (0.21)
Cantherhines pullus	orangespotted filefish	0.9	10	10	(<0.01) (<0.01)	963.0	0.83 (0.31)
Monacanthus ciliatus	fringed filefish	1.2	14	14	0.01 (<0.01)	56.9	0.05 (0.02)
Monacanthus tuckeri	slender filefish	1.5	18	20	0.02 (<0.01)	58.6	0.05 (0.02)
Mugilidae							
Mugil cephalus	striped mullet	0.1	1	3	<0.01 (<0.01)	485.2	0.42 (0.42)

Table B1 cont...

Family *Species name*	**Common name**	**% occurrence**	**Total occurrence**	**Total abundance**	**Mean abundance (± SE)**	**Total biomass, g**	**Mean biomass, g (± SE)**
Mullidae							
Mulloidichthys martinicus	yellow goatfish	6.6	77	330	0.28 (0.10)	28166.3	24.1 (4.9)
Pseudupeneus maculatus	spotted goatfish	16.6	194	412	0.35 (0.06)	19362.4	16.6 (2.2)
Muraenidae							
Gymnothorax funebris	green moray	0.3	4	4	<0.01 (<0.01)	5129.5	4.4 (3.3)
Gymnothorax miliaris	goldentail moray	0.2	2	2	<0.01 (<0.01)	28.4	0.02 (0.02)
Gymnothorax moringa	spotted moray	0.7	8	8	(<0.01) (<0.01)	1593.1	1.4 (0.55)
Ophichthidae							
Myrichthys breviceps	sharptail eel	0.1	1	1	<0.01 (<0.01)	421.9	0.36 (0.36)
Myrichthys UNK	SNAKE EEL sp	0.2	2	2	<0.01 (<0.01)	75.4	0.06 (0.06)
Opistognathidae							
Lonchopisthus micrognathus	swordtail jawfish	1.6	19	47	0.04 (0.01)	187.1	0.16 (0.07)
Opistognathus aurifrons	yellowhead jawfish	4.9	57	138	0.12 (0.02)	508.6	0.44 (0.09)
Opistognathus macrognathus	banded jawfish	0.2	2	3	<0.01 (<0.01)	8.0	(<0.01) (<0.01)
Opistognathus UNK	JAWFISH sp	0.1	1	3	<0.01 (<0.01)	7.9	(<0.01) (<0.01)
Opistognathus whitehursti	dusky jawfish	0.1	1	1	<0.01 (<0.01)	3.9	(<0.01) (<0.01)
Ostraciidae							
Acanthostracion polygonius	honeycomb cowfish	0.1	1	1	<0.01 (<0.01)	0.48	(<0.01) (<0.01)
Acanthostracion quadricornis	scrawled cowfish	0.3	4	4	<0.01 (<0.01)	25.0	0.02 (0.02)
Lactophrys bicaudalis	spotted trunkfish	0.5	6	6	(<0.01) (<0.01)	3298.8	2.8 (2.3)
Lactophrys trigonus	trunkfish	0.1	1	1	<0.01 (<0.01)	75.4	0.06 (0.06)
Lactophrys triqueter	smooth trunkfish	1.3	15	15	0.01 (<0.01)	1112.6	0.95 (0.31)
Lactophrys UNK	TRUNKFISH sp	0.1	1	1	<0.01 (<0.01)	0.83	(<0.01) (<0.01)
Paralichthyidae							
Syacium UNK	SAND FLOUNDER sp	0.3	4	5	<0.01 (<0.01)	208.2	0.18 (0.18)
Pomacanthidae							
Holacanthus ciliaris	queen angelfish	2.8	33	41	0.04 (<0.01)	9250.8	7.9 (2.0)
Holacanthus tricolor	rock beauty	2.8	33	37	0.03 (<0.01)	3202.3	2.7 (0.64)
Pomacanthus arcuatus	gray angelfish	7.8	91	133	0.11 (0.01)	110334.9	94.5 (13.8)
Pomacanthus paru	French angelfish	1.5	17	24	0.02 (<0.01)	10979.9	9.4 (3.9)
Pomacentridae							
Abudefduf saxatilis	sergeant major	10.4	121	488	0.42 (0.05)	6720.3	5.8 (1.5)
Abudefduf taurus	night sergeant	0.3	3	3	<0.01 (<0.01)	218.4	0.19 (0.12)
Chromis cyanea	blue chromis	6.1	71	992	0.85 (0.16)	2994.2	2.6 (0.45)
Chromis multilineata	brown chromis	3.5	41	327	0.28 (0.08)	1773.2	1.5 (0.45)
Microspathodon chrysurus	yellowtail damselfish	10.8	126	398	0.34 (0.05)	29812.5	25.5 (3.8)
Stegastes adustus	dusky damselfish	6.9	81	1187	1.0 (0.17)	12003.6	10.3 (1.8)
Stegastes diencaeus	longfin damselfish	7.1	83	489	0.42 (0.08)	7110.6	6.1 (1.0)
Stegastes leucostictus	beaugregory	26.1	305	2135	1.8 (0.15)	12900.3	11.1 (0.95)
Stegastes partitus	bicolor damselfish	38.8	453	4378	3.8 (0.26)	6202.1	5.3 (0.90)
Stegastes planifrons	threespot damselfish	15.2	177	1718	1.5 (0.15)	23525.8	20.2 (2.5)
Stegastes variabilis	cocoa damselfish	19.7	230	699	0.60 (0.05)	5594.0	4.8 (0.38)
Priacanthidae							
Heteropriacanthus cruentatus	glasseye snapper	0.4	5	16	0.01 (0.01)	1326.0	1.1 (0.70)
Priacanthus arenatus	bigeye	0.1	1	2	<0.01 (<0.01)	245.1	0.21 (0.21)
Scaridae							
Cryptotomus roseus	bluelip parrotfish	19.6	229	948	0.81 (0.08)	4341.4	3.7 (0.49)
Scarus guacamaia	rainbow parrotfish	0.2	2	2	<0.01 (<0.01)	23.3	0.02 (0.01)
Scarus iseri	striped parrotfish	45.0	525	6977	6.0 (0.39)	128555.1	110.2 (7.9)
Scarus taeniopterus	princess parrotfish	21.1	246	2034	1.7 (0.16)	108626.2	93.1 (10.7)
Scarus UNK	PARROTFISH sp	0.3	4	5	<0.01 (<0.01)	297.9	0.26 (0.25)
Scarus vetula	queen parrotfish	1.1	13	27	0.02 (<0.01)	5005.8	4.3 (1.9)
Sparisoma atomarium	greenblotch parrotfish	13.6	159	504	0.43 (0.05)	1125.6	0.96 (0.38)
Sparisoma aurofrenatum	redband parrotfish	42.8	499	3408	2.9 (0.14)	152788.8	130.9 (7.3)
Sparisoma chrysopterum	redtail parrotfish	6.8	79	154	0.13 (0.02)	28634.8	24.5 (6.1)
Sparisoma radians	bucktooth parrotfish	23.8	278	1625	1.4 (0.13)	3716.3	3.2 (0.48)
Sparisoma rubripinne	yellowtail parrotfish	4.4	51	124	0.11 (0.02)	16550.8	14.2 (4.4)
Sparisoma UNK	PARROTFISH sp	1.5	18	30	0.03 (<0.01)	67.4	0.06 (0.02)
Sparisoma viride	stoplight parrotfish	30.1	351	1314	1.1 (0.08)	229172.5	196.4 (19.8)
Sciaenidae							
Equetus lanceolatus	jackknife fish	0.1	1	1	<0.01 (<0.01)	0.08	(<0.01) (<0.01)

Table B1 cont...

Family *Species name*	**Common name**	**% occurrence**	**Total occurrence**	**Total abundance**	**Mean abundance (± SE)**	**Total biomass, g**	**Mean biomass, g (± SE)**
Equetus punctatus	spotted drum	0.4	5	5	<0.01 (<0.01)	652.4	0.56 (0.27)
Odontoscion dentex	reef croaker	0.3	4	46	0.04 (0.03)	251.6	0.22 (0.14)
Pareques acuminatus	highhat	1.6	19	26	0.02 (<0.01)	276.0	0.24 (0.17)
Scombridae							
Scomberomorus regalis	cero	0.9	11	14	0.01 (<0.01)	23748.6	20.4 (7.5)
Scorpaenidae							
Scorpaena plumieri	spotted scorpionfish	0.6	7	7	(<0.01) (<0.01)	984.8	0.84 (0.44)
Scorpaena UNK	SCORPIONFISH sp	0.1	1	1	<0.01 (<0.01)	9.3	(<0.01) (<0.01)
Serranidae							
Alphestes afer	mutton hamlet	0.2	2	2	<0.01 (<0.01)	14.7	0.01 (<0.01)
Cephalopholis cruentata	graysby	13.8	161	246	0.21 (0.02)	15443.2	13.2 (1.5)
Cephalopholis fulva	coney	3.8	44	81	0.07 (0.01)	12244.5	10.5 (2.0)
Diplectrum bivittatum	dwarf sand perch	0.5	6	13	0.01 (<0.01)	17.8	0.02 (<0.01)
Diplectrum formosum	sand perch	0.1	1	1	<0.01 (<0.01)	0.34	(<0.01) (<0.01)
Epinephelus adscensionis	rock hind	0.2	2	2	<0.01 (<0.01)	12.9	0.01 (<0.01)
Epinephelus guttatus	red hind	3.6	42	43	0.04 (<0.01)	16774.9	14.4 (2.7)
Epinephelus striatus	Nassau grouper	0.2	2	2	<0.01 (<0.01)	1331.3	1.1 (0.91)
Hypoplectrus aberrans	yellowbelly hamlet	0.9	10	11	(<0.01) (<0.01)	56.8	0.05 (0.02)
Hypoplectrus chlorurus	yellowtail hamlet	9.9	115	157	0.13 (0.01)	1294.1	1.1 (0.14)
Hypoplectrus guttavarius	shy hamlet	0.9	11	12	0.01 (<0.01)	64.8	0.06 (0.02)
Hypoplectrus indigo	indigo hamlet	0.9	11	12	0.01 (<0.01)	226.0	0.19 (0.08)
Hypoplectrus nigricans	black hamlet	1.8	21	22	0.02 (<0.01)	163.4	0.14 (0.04)
Hypoplectrus puella	barred hamlet	8.9	104	141	0.12 (0.01)	553.2	0.47 (0.06)
Hypoplectrus unicolor	butter hamlet	5.5	64	75	0.06 (<0.01)	550.3	0.47 (0.08)
Hypoplectrus UNK	HAMLET sp	4.4	51	67	0.06 (<0.01)	101.9	0.09 (0.03)
Liopropoma rubre	peppermint basslet	0.1	1	1	<0.01 (<0.01)	5.8	(<0.01) (<0.01)
Mycteroperca bonaci	black grouper	0.2	2	2	<0.01 (<0.01)	2744.4	2.4 (2.1)
Rypticus bistrispinus	freckled soapfish	0.1	1	1	<0.01 (<0.01)	5.8	(<0.01) (<0.01)
Rypticus saponaceus	greater soapfish	0.2	2	2	<0.01 (<0.01)	164.0	0.14 (0.10)
Serraniculus pumilio	pygmy sea bass	0.3	3	3	<0.01 (<0.01)	1.1	(<0.01) (<0.01)
Serranus baldwini	lantern bass	7.1	83	253	0.22 (0.03)	216.4	0.19 (0.03)
Serranus tabacarius	tobaccofish	5.0	58	107	0.09 (0.02)	482.8	0.41 (0.18)
Serranus tigrinus	harlequin bass	8.7	101	150	0.13 (0.01)	726.2	0.62 (0.08)
Serranus tortugarum	chalk bass	3.9	45	325	0.28 (0.07)	138.7	0.12 (0.03)
Serranus UNK	SEABASS sp	1.2	14	26	0.02 (<0.01)	25.7	0.02 (0.01)
Sparidae							
Archosargus rhomboidalis	sea bream	3.3	39	135	0.12 (0.03)	7890.1	6.8 (1.8)
Calamus calamus	saucereye porgy	1.4	16	18	0.02 (<0.01)	2597.3	2.2 (0.73)
Calamus penna	sheepshead porgy	0.1	1	1	<0.01 (<0.01)	8.3	(<0.01) (<0.01)
Calamus pennatula	pluma	4.0	47	61	0.05 (<0.01)	14349.1	12.3 (3.5)
Calamus UNK	PORGY sp	0.2	2	3	<0.01 (<0.01)	77.7	0.07 (0.05)
Diplodus holbrooki	spottail pinfish	0.1	1	5	<0.01 (<0.01)	518.3	0.44 (0.44)
Sphyraenidae							
Sphyraena barracuda	great barracuda	10.4	121	209	0.18 (0.02)	160161.4	137.2 (32.3)
Sphyraena picudilla	southern sennet	0.2	2	54	0.05 (0.04)	39372.0	33.7 (28.0)
Stromateidae							
Stromateidae UNK	BUTTERFISH Family sp	0.5	6	16	0.01 (<0.01)	6.3	(<0.01) (<0.01)
Syngnathidae							
Cosmocampus elucens	shortfin pipefish	0.1	1	1	<0.01 (<0.01)	1.2	(<0.01) (<0.01)
Hippocampus reidi	longsnout seahorse	0.1	1	1	<0.01 (<0.01)	0.04	(<0.01) (<0.01)
Synodontidae							
Synodus intermedius	sand diver	5.2	61	63	0.05 (<0.01)	3476.1	3.0 (0.85)
Tetraodontidae							
Canthigaster rostrata	sharpnose puffer	25.0	292	484	0.41 (0.03)	1072.9	0.92 (0.10)
Sphoeroides spengleri	bandtail puffer	3.1	36	40	0.03 (<0.01)	993.5	0.85 (0.32)
Sphoeroides testudineus	checkered puffer	4.4	51	78	0.07 (0.01)	6249.7	5.4 (1.1)
Triglidae							
Triglidae UNK	searobin family	0.1	1	1	<0.01 (<0.01)	21.5	0.02 (0.02)
Tripterygiidae							
Enneanectes UNK	TRIPLEFIN sp	0.3	3	3	<0.01 (<0.01)	1.2	(<0.01) (<0.01)

Appendix C

*Table C1. Summary information on selected species from five key fish families showing maximum size observed in the SW Puerto Rico study region compared with maximum known size for the species and the proportion of juveniles found based on 1167 samples from 2001-2007. * Maximum known fish size and size at first maturity are from FishBase (http://www.fishbase.org). TL=total length; FL= fork length.*

Species	Common name	Approx. size class first maturity*	Juv./sub-adult size class (cm)	Max. known size*, TL	Max. size observed PR, FL
Serranids					
Cephalopholis cruentata	graysby	15-20	<15	42.6	30
Cephalopholis fulva	coney	15-20	<15	41	30
Epinephelus guttatus	red hind	20-25	<20	76	40
Lutjanids					
Lutjanus apodus	schoolmaster	20-25	<20	67.2	45
Lutjanus griseus	gray snapper	25-30	<25	89	65
Lutjanus mahogoni	mahogany snapper	15-20	<15	48	
Lutjanus synagris	lane snapper	20-25	<20	89	65
Ocyurus chrysurus	yellowtail snapper	20-25	<20	86.3	40
Haemulids					
Haemulon aurolineatum	tomtate	15-20	<15	25	25
Haemulon flavolineatum	French grunt	15-20	<15	30	30
Haemulon plumierii	white grunt	15-20	<15	53	30
Haemulon sciurus	bluestriped grunt	15-20	<15	46	35
Scarids					
Scarus iseri	striped parrotfish	10-15	<10	35	35
Scarus taeniopterus	princess parrotfish	10-15	<10	35	35
Sparisoma aurofrenatum	redband parrotfish	10-15	<10	28	35
Sparisoma viride	stoplight parrotfish	15-20	<15	64	50
Other species					
Mulloidichthys martinicus	yellow goatfish	15-20	<15	35.4	25
Pseudupeneus maculatus	spotted goatfish	15-20	<15	27.8	25
Balistes vetula	queen triggerfish	20-25	<20	47.1	30
Sphyraena barracuda	great barracuda	n/a	n/a	200	130

Appendix D

Methods

To assist in monitoring coral reef ecosystem resources and to achieve a better understanding of fish-habitat relationships in the U.S. Caribbean, CCMA-BB developed a fish and macro-invertebrate monitoring protocol to provide precise, fishery-independent and size-structured survey data, needed to comprehensively assess faunal populations and communities (Menza et al., 2006). In addition, a complementary benthic composition survey was also developed to support studies of fish-habitat relationships. These data collection activities and analytical products are core components of NOAA's Coral Reef Conservation Program (CRCP) implemented through CCMA-BB's CREM project. CREM protocols were created primarily to quantify long-term changes in fish species and assemblage diversity, abundance, biomass and size structure and to compare these metrics between areas inside and outside of Marine Protected Areas (MPAs). A stratified random sampling design was used to optimize the allocation of samples and allow rigorous inferences to the entire study area. Three strata were selected based upon: 1) the study objectives; 2) parsimony in the approach; and 3) results from statistical analyses of variance (Menza et al., 2006). The "hard" stratum comprised bedrock, pavement, rubble and coral reefs. The "soft" stratum comprised sand, seagrasses and macroalgal beds. The "mangrove" stratum comprised the seaward edge of mangrove habitat able to be surveyed with these underwater methods (Figure 1.8, pg. 8).

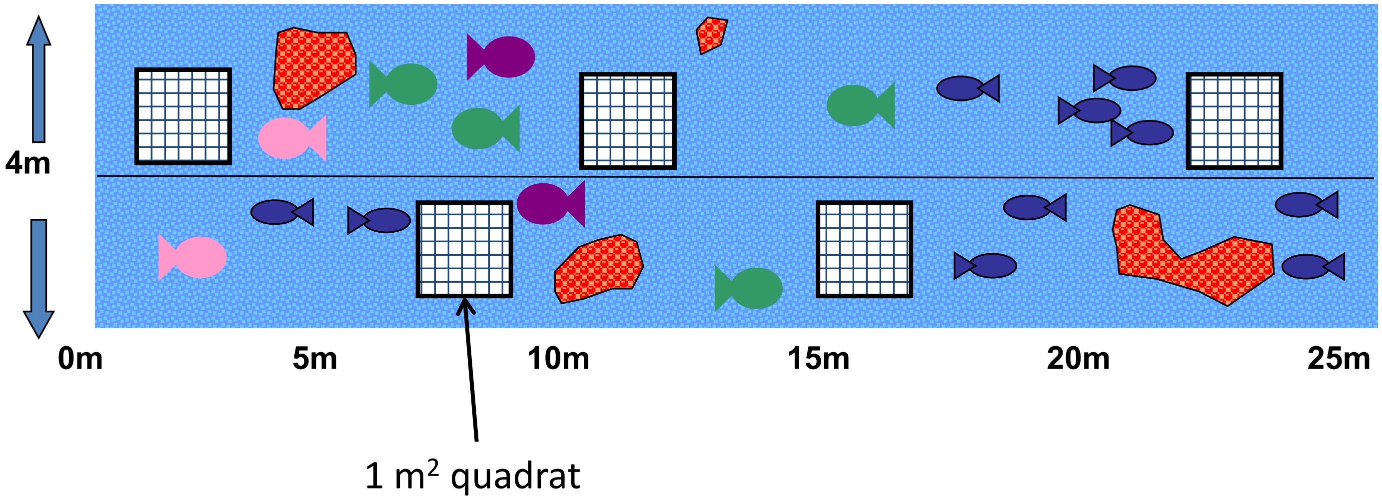
Field survey methods

This report uses underwater census data collected from 2001 to 2007. Survey missions occurred each year during all seasons and were standardized to Winter and Summer starting in 2005 (Table 2.2, pg. 14). This data set is part of a broader ongoing monitoring study that began in 2001, with over 1,000 transects surveyed thus far throughout the La Parguera study region. There are two complementary components to the biological field methods: (1) benthic habitat composition surveys and (2) fish surveys.

Benthic habitat composition surveys

To conduct benthic habitat surveys, an observer places a 1 m^2 quadrat divided into 100 (10 x 10 cm) smaller squares (1 square = 1% cover) at five randomly pre-selected locations along the transect, such that a quadrat is placed once somewhere within every 5 m interval along the transect (see schematic below). Percent cover is estimated within the quadrat in a two-dimensional plane perpendicular to the observer's line of vision.

Schematic of fish transect with random habitat quadrat placement.



Information recorded includes:

<u>Habitat structure</u> (e.g., colonized hardbottom, spur and groove, patch reef, pavement) - based on the habitat types used in the benthic habitat maps (Kendall et al., 2002; Figure 1.8 pg. 8), until 2004, after which habitat structure was classified only to hard, soft and mangrove.

<u>Abiotic footprint</u> - defined as the percent cover (to the nearest 1%) of sand, rubble, hardbottom, fine sediments and other non-living bottom types within a 1 m^2 quadrat.

Biotic footprint - defined as the percent cover to the nearest 1% of algae, seagrass, upright sponges, gorgonians and other biota and to the nearest 0.1% for live, bleached and recently dead/diseased coral within a 1 m^2 quadrat.

Transect depth profile - the depth at each quadrat position. Depth is measured with a digital depth gauge and rounded up or down to the nearest foot.

Maximum canopy height - for each biota type, height of soft structure (e.g., gorgonians, upright sponges, seagrass, algae) is recorded to the nearest 1 cm.

Hardbottom rugosity - measured by placing a 6-m chain at two randomly selected start positions ensuring no overlap along 25-m belt transect. The chain is placed such that it follows the relief along centerline of the belt transect. Two divers measure the straight-line horizontal distance covered by the chain.

Proximity of structure - on seagrass and sand sites, the habitat diver records the absence or presence of reef or hard structure within 3 m of the belt transect.

Table 2.1 (pg. 13) provides a list of measured variables. The habitat observer also counts queen conch (*Eustrombus gigas*), long-spined sea urchins (*Diadema antillarum*) and Caribbean spiny lobster (*Panulirus argus*).

Mangrove habitat data

At mangrove sites, an observer swims close to the prop roots and surveys as far into the mangroves as possible, up to 2 m and then out to the edge of the mangrove overhang such that the total area surveyed is still 100 m^2. In this case, some of the survey may necessarily fall on seagrass habitat. This is allowed as the mangrove habitat is defined as a transition zone habitat. In addition to the habitat data collected above, further mangrove data are collected including number of prop roots, number of prop roots colonized by algae,number of prop roots colonized by sponges and nujmber of prop roots colonized by other (tunicates, anemones, zooanthids, etc).

Fish surveys

Fish surveys were conducted along a 25 m long by 4 m wide belt (100 m^2) using a fixed survey duration of 15 minutes (Figure 3.1, pg. 56). The fixed duration of 15 minutes standardizes the samples collected to facilitate between-site comparisons. The number of individuals per species is recorded in 5 cm size class increments up to 35 cm using the visual estimation of fork length. Individuals greater than 35 cm are recorded as an estimate of the actual fork length to the nearest centimeter.

Macroinvertebrates counts

Queen conch

The abundance of immature and mature queen conch (*Eustrombus gigas*) was assessed and quantified within the 25 x 4 m belt transects used for fish surveys. The maturity of each conch was determined by the presence (mature) or absence (immature) of a flared lip (pg. 46). Conch were included in the survey protocol from August 2004 onward.

Caribbean spiny lobster

Abundance of Caribbean spiny lobsters (*Panulirus argus*) was reported for the period 2005 to 2007. Lobster sightings were recorded during fish and benthic composition surveys (i.e., within the 100 m^2 survey unit area). Lobsters were recorded if seen, but without active searches of holes or crevices.

Long-spined sea urchins

Long-spined sea urchins (*Diadema antillarum*) were counted within the 25 x 4 m belt transect during 2006 and 2007. No measurements of size or estimates of maturity were collected.

Marine debris data

Type of marine debris within 25 x 4 m belt transect was noted. The size of the marine debris and the area of affected habitat is also recorded along with a note identifying any flora or fauna that colonized the debris. Marine debris data collection began in 2007.

Observer training

Observers were trained and tested in the identification of species/groups for both fish and habitat surveys by pairing inexperienced and experienced observers in the water and comparing data. Fish size estimation training was carried out *in situ* by estimating lengths of model fish of various shapes and sizes.

Data management

All fish and benthic habitat survey data were quality assessed before storage on an online relational database. All survey data were stored with a unique identification number and a geographical coordinate to facilitate spatial analyses. The database (including metadata) that provides detailed field methods are available online: http://ccmaserver.nos.noaa.gov/ecosystems/coralreef/reef_fish/protocols.html.

Although the 1m-square-quadrat remained the basic method of choice for habitat data collection, overtime, changes in data collection methods were made for some habitat variables and several additional variables were added. These changes were deemed necessary to capture more precise information and as many variables as possible to explain better the observed variability in reef fish assemblage metrics. Detailed information on all changes to the protocols for collecting habitat data in Puerto Rico can be found at: http://ccmaserver.nos.noaa.gov/ecosystems/coralreef/reef_fish/protocols.html. A brief list is included below:

Over time, some changes were made to the stratified random site selection process as follows: 1) Habitat strata initially consisted of hard bottom, sand, seagrass and mangrove. Sand and seagrass strata were subsequently combined into one soft bottom strata. This action was taken after the February 2002 mission. 2) In addition to the habitat strata, Puerto Rico originally contained three strata representing levels of protection from waves and currents. These strata were the Bank Shelf, Outer Lagoon and Inner Lagoon. This was changed beginning with the December 2002 mission to simply Protected and Unprotected. After the January 2005 mission, strata of Protected and Unprotected was removed leaving only habitat strata. 3) A small subset of sites were resampled during each mission through June 2002. These station names contain the letter ‘P’ indicating they are permanent stations.

In 2007, algae data collection changed from identification of each alga to the genus level to grouping algae into six morphological groups: macro, turf, crustose, filamentous, rhodolith, and cyanobacteria for more efficient data collection.

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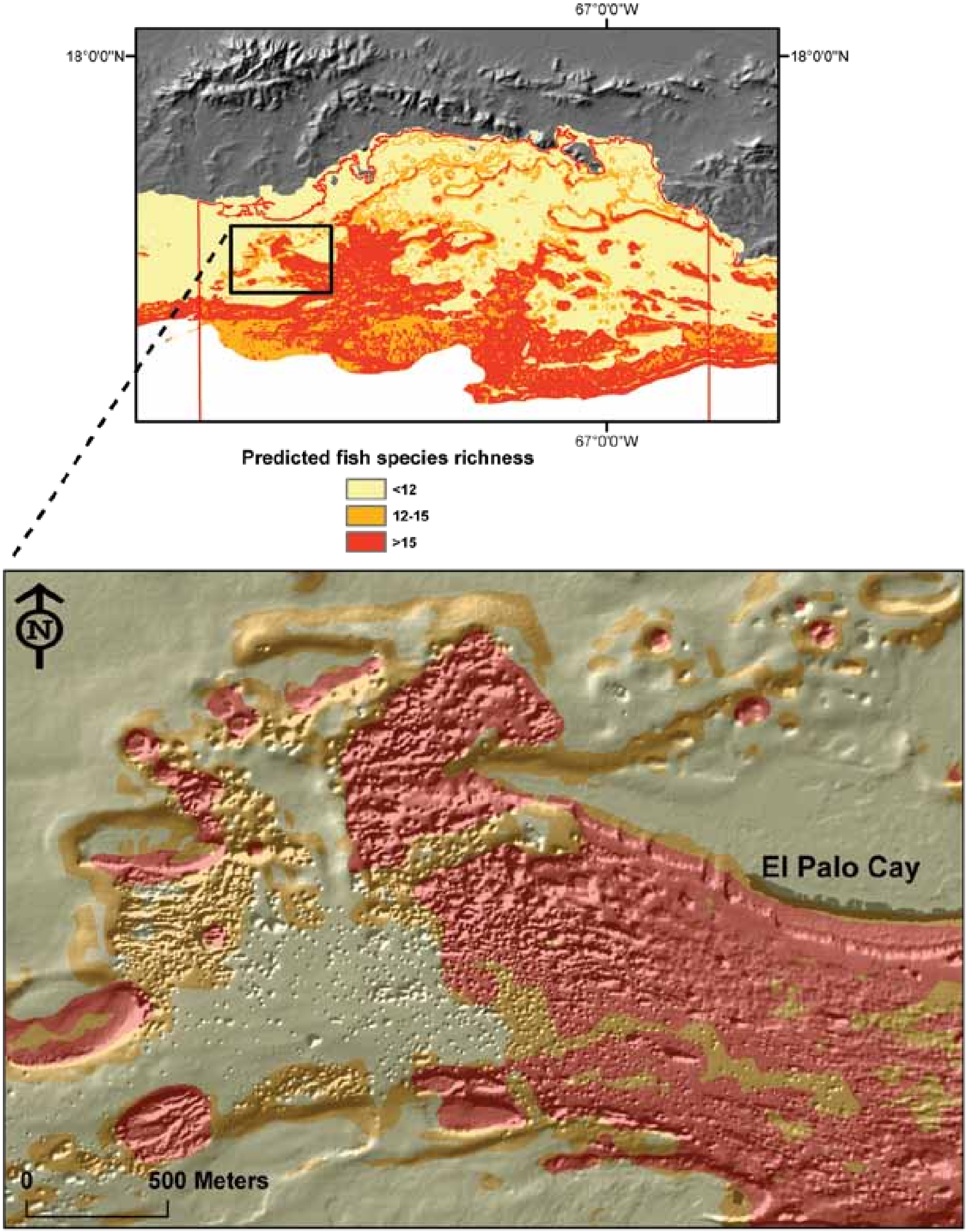
Appendix E



Adapted from Pittman et al. (2007a)

Figure E1. Spatial predictive map of fish species richness across the seascapes of La Parguera developed from the relationship between the number of species observed via underwater fish surveys and the underlying topographic complexity derived from chain rugosity measurements and historical depth soundings. The predictive map was then overlaid on the more recent LiDAR bathymetry. Modeling was performed using a regression tree as documented in Pittman et al. (2007a) with a 75% overall map accuracy.

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