Coral reef ecosystems of St. John, U.S. Virgin Islands:
Spatial and temporal patterns in fish and benthic communities (2001-2009)

Hawaii Cooperative Fishery Rearch Unit

Alan M. Friedlander

NOAA National Centers for Coastal Ocean Science
Center for Coastal Monitoring and Assessment

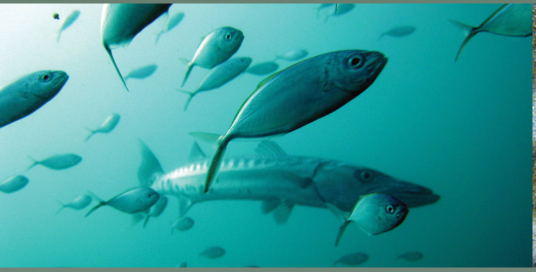
Christopher F.G. Jeffrey

Sarah D. Hile

Simon J. Pittman

Mark E. Monaco

Chris Caldow




August 2013

NOAA TECHNICAL MEMORANDUM NOS NCCOS 152

NOAA NCCOS Center for Coastal Monitoring and Assessment

Acknowledgements

This report was made possible through the gallant efforts of the Caribbean Coral Reef Ecosystem Project Team (see Appendix F). Their efforts with project implementation, data collection, field support and report production are highly appreciated. Specific recognition goes to Rafe Boulon, Daniel Dorfman, Michael Feeley, Jeff Miller, Matt Patterson and Caroline Rogers for critically reviewing the report; to Jamie Higgins for assistance in report production; and to Kevin McMachon for copy editing the document. Additional logistic support during field missions came from Low Key Watersports, Cruz Bay, St. John and the Red Hook Dive Center, Red Hook, St. Thomas, Donald Day Junior (Battery Hill Condominium and Suites St. John).

This report was a cooperative investigation between NOAA, the National Park Service, U.S. Geological Survey, University of Hawaii and the University of the Virgin Islands. Financial support was primarily provided by NOAA's Coral Reef Conservation Program, National Park Service and NOAA's National Centers for Coastal Ocean Science.

The front covers for this document were created by Gini Kennedy and layout completed by Sarah Hile. Cover photos were provided by the NCCOS/CCMA Biogeography Branch. Government contract labor was provided by CSS-Dynamac of Fairfax, Virginia under NOAA contract number DG133C11CQ0019.

Coral reef ecosystems of St. John, U.S. Virgin Islands: Spatial and temporal patterns in fish and benthic communities (2001-2009)

Prepared by the
Biogeography Branch (BB)
Center for Coastal Monitoring and Assessment (CCMA)
National Centers for Coastal Ocean Science (NCCOS)
National Ocean Service (NOS)
National Oceanic and Atmospheric Administration (NOAA)
1305 East West Highway (SSMC-IV, N/SCI-1)
Silver Spring, MD 20910
USA

August 2013

Editors
Alan M. Friedlander
Christopher F.G. Jeffrey
Sarah D. Hile
Simon J. Pittman
Mark E. Monaco
Chris Caldow

Citation

Full report citation:

Friedlander, A.M., C.F.G. Jeffrey, S.D. Hile, S.J. Pittman, M.E. Monaco and C. Caldow (eds.). 2013. Coral reef ecosystems of St. John, U.S. Virgin Islands: Spatial and temporal patterns in fish and benthic communities (2001-2009). NOAA Technical Memorandum 152. Silver Spring, MD. 150 pp.

Citations for individual chapters (example Chapter 1):

Friedlander, A.M., S.D. Hile, and S.J. Pittman. 2013. Chapter 1: Introduction. pp. 1-18. In: A.M. Friedlander, C.F.G. Jeffrey, S.D. Hile, S.J. Pittman, M.E. Monaco and C. Caldow (eds.). Coral reef ecosystems of St. John, U.S. Virgin Islands: Spatial and temporal patterns in fish and benthic communities (2001-2009). NOAA Technical Memorandum 152. Silver Spring, MD. 150 pp.

ABOUT THIS DOCUMENT

This Technical Memorandum is part of a series of three reports that provide a quantitative spatial and temporal characterization of marine biological communities associated with marine protected areas in the U.S. Caribbean. This work was conducted as part of NOAA's Coral Reef Conservation Program (CRCP) Caribbean Coral Reef Ecosystem Monitoring (CREM) project; a partnership effort between NOAA's National Ocean Service (NOS), National Centers for Coastal Ocean Science (NCCOS), Center for Coastal Monitoring and Assessment Biogeography Branch (CCMA-BB), U.S. Virgin Islands Department of Planning and Natural Resources – Division of Fish and Wildlife, U.S. Geological Survey (USGS), National Park Service (NPS), the University of the Virgin Islands (UVI), and the University of Hawaii (UH). The integration of NOAA/NPS led efforts with data generated by VI-DPNR provide spatial and temporal patterns in fish and benthic communities to characterize St. John coral reef ecosystems. The data and analyses in this report are intended to provide essential baseline biological information to support management decision making. This project was funded by CRCP, NOAA's NCCOS, and the NPS Natural Resource Preservation Program (NRPP) at Virgin Islands National Park (VIIS) and NPS's South Florida/Caribbean Inventory and Monitoring Program (SFCN).

Related projects and publications include:
Caribbean Coral Reef Ecosystem Monitoring
http://coastalscience.noaa.gov/projects/detail?key=57 (Accessed 14 may 2013)

Development of Reef Fish Monitoring Protocols to Support the National Park Service Inventory and Monitoring Program
http://coastalscience.noaa.gov/projects/detail?key=72

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

Benthic Habitat Mapping of Puerto Rico and the U.S. Virgin Islands
http://coastalscience.noaa.gov/projects/detail?key=33

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

Acoustic Tracking of Fish Movements in Coral Reef Ecosystems
http://ccma.nos.noaa.gov/ecosystems/coralreef/acoustic_tracking/

Pittman, S.J., S.D. Hile, C.F.G. Jeffrey, C. Caldow, M.S. Kendall, M.E. Monaco, and Z. Hillis-Starr. 2008. Fish assemblages and benthic habitats of Buck Island Reef National Monument (St. Croix, U.S. Virgin Islands) and the surrounding seascape: A characterization of spatial and temporal patterns. NOAA Technical Memorandum NOS NCCOS 71. Silver Spring, MD. 96 pp. Online: http://ccma.nos.noaa.gov/publications/biogeography/STC_LTR_Final_0309.pdf (Accessed 14 May 2014)

Pittman, S.J., S.D. Hile, C.F.G. Jeffrey, R. Clark, K. Woody, B.D. Herlach, C. Caldow, M.E. Monaco, R. Appeldoorn. 2010. Coral reef ecosystems of Reserva Natural La Parguera (Puerto Rico): Spatial and temporal patterns in fish and benthic communities (2001-2007). NOAA Technical Memorandum NOS NCCOS 107. Silver Spring, MD. 202 pp. Online: http://ccma.nos.noaa.gov/publications/nccostechmemo107.pdf (Accessed 14 May 2013)

For more information on this project direct questions and comments to:
Chris Caldow
NCCOS/CCMA Biogeography Branch Chief
National Oceanic and Atmospheric Administration
301-713-3028 x164
Chris.Caldow@noaa.gov

Table of Contents

List of Tables

List of Figures

Chapter 1

Chapter 4

Appendices

EXECUTIVE SUMMARY

BACKGROUND

Scientific and anecdotal observations during recent decades have suggested that the structure and function of the coral reef ecosystems around St. John, U.S. Virgin Islands have been impacted adversely by a wide range of environmental stressors. Major stressors included the mass die-off of the long-spined sea urchin (Diadema antillarum) in the early 1980s, a series of hurricanes (David and Frederick in 1979, and Hugo in 1989), overfishing, mass mortality of Acropora species and other reef-building corals due to disease and several coral bleaching events.

In response to these adverse impacts, the National Centers for Coastal Ocean Science (NCCOS), Center for Coastal Monitoring and Assessment, Biogeography Branch (CCMA-BB) collaborated with federal and territorial partners to characterize, monitor, and assess the status of the marine environment around the island from 2001 to 2012. This 13-year monitoring effort, known as the Caribbean Coral Reef Ecosystem Monitoring Project (CREM), was supported by the NOAA Coral Reef Conservation Program as part of their National Coral Reef Ecosystem Monitoring Program.

This technical memorandum contains analysis of nine years of data (2001-2009) from in situ fish belt transect and benthic habitat quadrat surveys conducted in and around the Virgin Islands National Park (VIIS) and the Virgin Islands Coral Reef National Monument (VICR). The purpose of this document is to:

1) Quantify spatial patterns and temporal trends in (i) benthic habitat composition and (ii) fish species abundance, size structure, biomass, and diversity;
2) Provide maps showing the locations of biological surveys and broad-scale distributions of key fish and benthic species and assemblages; and
3) Compare benthic habitat composition and reef fish assemblages in areas under NPS jurisdiction with those in similar areas not managed by NPS (i.e., outside of the VIIS and VICR boundaries).

This report provides key information to help the St. John management community and others understand the impacts of natural and man-made perturbations on coral reef and near-shore ecosystems. It also supports ecosystem-based management efforts to conserve the region's coral reef and related fauna while maintaining the many goods and ecological services that they offer to society.

Funding was provided by the Coral Reef Conservation Program and NCCOS, NPS' Natural Resource Preservation Program (NRPP) at VIIS, and NPS' South Florida/Caribbean Network (SFCN) Inventory and Monitoring Program. Data collection partners include the National Park Service (NPS), the Virgin Islands Department of Planning and Natural Resources (VI-DPNR), the U.S. Geological Survey (USGS), University of the Virgin Islands (UVI), and University of Hawaii (UH).

METHODS

Since 2001, benthic and reef fish surveys were randomly conducted annually on hard and soft bottom substrates throughout the entire marine seascape to characterize the floral and faunal assemblages of the area (Menza et al., 2006). All sites were visited during July of each year to minimize inherent seasonal variation in sample estimates.

Divers conducted detailed (full-scale) benthic surveys by estimating the percent cover of abiotic and biotic components of the substrate at five randomly placed 1-m^2 quadrats along a 25 x 4 m belt transect (Appendix A). Comparative analyses to describe benthic composition inside and outside VIIS boundaries were based solely on percent cover data estimated from these full-scale surveys and included 324 hardbottom and 403 soft bottom sites sampled between July 2001 and July 2009 (total n=727; Chapter 3, Table 3.3). Benthic composition of the VICR was also estimated with a "rapid habitat assessment (RHA) protocol at 641 sites, (Appendix B), but percent cover composition from those surveys were analyzed and reported previously by Monaco et al. (2007, 2009), Boulon et al. (2008).

Divers also identified fish to the lowest possible taxonomic level in each belt transect for 15 minutes during daylight hours and recorded their abundance in 5-cm size-class increments. A total of 1,048 fish transects conducted between July 2001 and July 2009 was used for comparative analyses of fish communities inside and outside VIIS and VICR boundaries. These analsyes were based on two fish variables (abundance and size) from which metrics such as biomass and diversity were derived to describe temporal and spatial trends for individual species, family, trophic, and assemblage groupings. Comparisons of fish metrics inside VIIS with outside VIIS used 677 surveys, with 379 sites occurring inside the VIIS (171 on hardbottom and 208 on softbottom), and 298 sites located outside the VIIS and VICR (129 on hardbottom and 169 on softbottom). Comparisons of fish metrics inside with outside the VICR were based on 371 hardbottom surveys conducted within Coral Bay; 222 of those sites were located inside the VICR and 149 sites occurred outside both the VICR and VIIS (see Chapter 4, Table 4.1).

Geographical Information System (GIS) tools were used to quantify the seascape surrounding each transect using habitat distributions represented in NOAA's benthic habitat maps (e.g., amount of seagrass, number of habitat types, etc.). Density maps from point estimates and kriged coverages of reef fish and benthic composition metrics were created through Inverse Distance Weighting (IDW, ArcGIS Version 9.x) to describe broad-scale spatial patterns benthic composition as well as juvenile and adult fish species and assemblage metrics.

MAJOR FINDINGS

Diversity hotspots

- Based on full-scale benthic surveys, the highest generic stony (Scleractinian) coral richness and cover occurred at the mouth of Coral Bay, St. John, along the north shore between Haulover and Newfound Bays, and along the south shore between Lameshur and Salt Pond Bays.
- Fish species richness and diversity were also highest where coral was most diverse and most abundant. In addition, the area on the north shore around Johnson's Reef, and Trunk and Cinnamon Bays also supported high fish diversity.

Benthic habitat

- The dominant benthic habitat types observed during full-scale hardbottom surveys were colonized pavement (33% of sampled), followed by linear reef (27%). Soft bottom sites consisted of 34% sand, 34% seagrass, and 20% patchy macroalgae (10-50% cover).
- Mean stony coral cover across all habitats was 2.19% (± 4.89 standard deviation [SD]). Mean coral cover on hard bottom sites was 4.88% (± 6.37 SD). Coral cover on hard bottom in VIIS was 4.27% (± 0.4 standard error [SE]) and 4.95% (± 0.6 SE) outside VIIS (but excluding the VICR). In Coral Bay, live coral cover averaged 7.90 % ± 2.31 SE within the VICR and 7.5% ± 1.96 SE in adjacent areas outside. An interpolated surface of live coral cover indicated that areas with higher live coral cover were more extensive in several of the southeastern locations of St. John, particularly in Coral Bay.
- Live stony coral cover included 26 coral genera. The three most commonly observed coral genera based on percent coral cover were *Montastraea* spp. (1.41%), *Siderastrea* spp. (0.68%)

and *Porites* spp. (0.70%). *Montastraea* spp. cover was highest on patch reef, linear reef, and pavement; *Siderastrea* spp. and *Porites* spp. cover were highest on linear reef and patch reef habitats.

- There was a chronic and significant average decline in live stony coral cover over the entire study area beginning from 2001 through 2008. Additionally, in October 2005, a mass coral bleaching event was recorded in the region; this may have further exacerbated the decline in coral cover inside VIIS, which dropped significantly from an average of 5.5% before the mass bleaching event to less than 3% afterwards. Similarly, coral cover outside VIIS was 5.8% prior to mass bleaching and 3.5% afterwards. Much of the observed coral decline was due to mortality that was primarily associated with disease rather than bleaching.
- Turf and macroalgae comprised the major components of the biotic benthic cover on hard bottom in both VIIS and adjacent sites. Turf algae accounted for 33% of the cover inside VIIS and 31% outside, and this difference was not statistically significant. Macroalgal cover was higher (20%) outside VIIS compared to inside (13%).
- On average, total seagrass cover on seagrass beds was fairly low (32%), with *Thalassia testudinum* and *Syringodium filiforme* being the most frequently observed species. Many seagrass beds had diverse assemblages that contained macro algae (16%), sponges, gorgonians, as well as living corals and other benthic invertebrates.

Fish assemblages

- Within VIIS and around St. John, 227 fish taxa from 56 families were recorded. The most frequently observed species was the Slippery Dick (*Halichoeres bivittatus*), which occurred in 55% of all transects, followed by the Ocean Surgeonfish (*Acanthurus bahianus*) in 47% of all transects.
- Fish species richness on hardbottom habitats averaged 22.6 species per transect with no significant difference between VIIS and outside areas. Numerical abundance averaged 1.7 individuals/m^2 on hardbottom and estimated biomass averaged 55 grams/m^2. Neither of these metrics showed any significant difference between VIIS and outside areas. Shannon's diversity index (H') on hardbottom was significantly higher inside VIIS (H'=2.35) compared with outside (H'=2.24; p=0.02).
- Invertivorous fishes accounted for 50% of the total fish biomass, followed by herbivores (30%), piscivores (11%), and planktivores (9%). On hard bottom habitats, invertivores accounted for 43% of the total biomass, followed by herbivores (41%), planktivores (9%), and piscivores (7%). Most of the invertivore biomass consisted of small wrasses.
- There was a significant and positive correlation between the Threespot Damselfish (*Stegastes planifrons*) and living cover of *Montastraea* spp. Threespot Damselfish therefore may be a good indicator of areas with relatively high coral cover because *Montastraea* spp. were the most commonly occurring corals during this study.
- Large-bodied groupers were extremely rare during the study. Only one Yellowmouth Grouper (*Mycteroperca interstitialis*), one Red Grouper (*Epinephelus morio*) and ten Nassau Groupers (*Epinephelus striatus*) were observed outside VIIS; nine of the Nassau Grouper and the one Red Grouper occurred in Coral Bay.

Macroinvertebrates

- A total of 228 queen conch (*Strombus gigas*) were observed in St. John (166 inside VIIS and 62 outside VIIS). Fifty percent of queen conch inside VIIS were juveniles (i.e., their shells had no lip) whereas 69% of queen conch outside VIIS were juveniles. Only 13 spiny lobsters were observed during surveys (two inside VIIS and eleven outside VIIS).

SUMMARY AND RECOMMENDATIONS

The biogeographic approach taken in this study is similar to ones currently conducted in the Buck Island National Monument and the surrounding waters of St. Croix, as well as the coral reef ecosystems of Reserva Natural de La Parguera, Puerto Rico (Pittman et al., 2008, 2010). The integration of geospatial information across the range of habitats present within the seascape has allowed for robust assessment and monitoring of the marine ecosystem within VIIS, VICR, and their surrounding waters. This information establishes a comprehensive baseline for the entire marine ecosystem surrounding the island of St. John and is useful to the NPS and the USVI territorial government to help guide future management decisions.

This report supports VIIS and VICR Management plans for boundary modification. For example, data collected during this study are being used as scientific justification for modifying the boundaries of the VICR to include additional high quality coral reef habitat (Boulon et al., 2008; Monaco et al., 2009). Based on CCMA-BB mapping and monitoring data, NPS Management is proposing to swap a territorially owned area outside VICR that has significantly more hard corals; greater habitat complexity; and greater richness, abundance and biomass of reef fishes, with an equally sized area within VICR that has less coral, less habitat complexity, and less species richness and diversity of reef fishes. By increasing the proportion of complex reef habitat and the amount of coral under protection within the VICR, the proposed swap will hopefully ensure long term sustainability of coral and fish assemblages in St. John and potentially increase the flow of ecological benefits to nearby unprotected areas.

This report also provides information that is critical to “vital signs” monitoring, which is being conducted by NPS SFCN (http://science.nature.nps.gov/im/units/sfcn/vs_plan/SFCN_VS_Plan.pdf). The Vital Signs Monitoring Plan describes a process for developing the infrastructure to monitor the overall condition of selected natural resources so that early detection of negative trends in resource condition is possible. NPS defines a vital sign “as a subset of physical, chemical, and biological elements and processes of park ecosystems that are selected to represent the overall health or condition of park resources, known or hypothesized effects of stressors, or elements that have important human values” (http://science.nature.nps.gov/im/monitor/glossary.cfm). Vital signs were ranked in order of importance to park management, and marine fish communities were ranked 2nd among 44 vital signs selected by SFCN for monitoring. Specific questions being addressed by SFCN Vital Signs monitoring include: “What are the status, trends, and variability in exploited fish assemblages (e.g., grouper/snapper/parrotfish/surgeonfish), reef fish communities, and nearshore and estuarine (bay) fish communities? Are there differences among areas with different management regimes?” To help address these questions at VICR and VIIS, CCMA-BB collected data between 2001 and 2009 to provide spatially explicit estimates for fish community taxonomic composition, species richness, abundance, biomass, and size structure of targeted species (e.g., grouper, snapper, parrotfish, and surgeonfish), as well as to describe spatial and temporal distribution patterns of fish assemblages in several different habitats.

In addition, CCMA-BB mapping and monitoring data are being used to support the development of NPS’ Standard Operating Procedures and Protocols (SOPs) as well as NOAA’s National Coral Reef Monitoring Plan (NCRMP) for monitoring of reef fish populations in reef and hardbottom habitats within the VICR and VIIS (Bryan et al., in press), and in other jurisdictions with coral reef ecosystem. Dramatic declines in reef fish populations during the past four decades along with the failure of reef fish populations to rebound under current management regimes, as well as the recent implementation of Allowable Catch Limits (ACLs) by NOAA's National Marine Fisheries Service (NMFS), has prompted NPS SFCN to develop these SOPs to increase the precision and accuracy of annual population estimates for exploited fish stocks, which are required by law under the re-authorization of the Magnuson-Stevens Act. These SFCN SOPs are a refinement of the existing CCMA-BB monitoring program and are being designed to improve the power for detecting changes in exploited fish stocks within VICR and VIIS as well as to provide annual USVI-wide population estimates for coral reef fishes (Menza et al., 2006; Bryan et al., in press). Through NCRMP, NOAA CRCP is also developing new protocols to holistically monitor and track changes in floral and faunal assemblages in coral reef ecosystems throughout all U.S. jurisdictions.

Long-term monitoring is necessary to determine the magnitude of the apparent declines for some species and also to track the trajectory of recovery for other species that exhibited an increase in density after several years of decline. This is critical given the inherent natural variability documented during this study and in similar ecosystems around the world. Continued annual monitoring is needed to determine the direction of change for several species that were highly variable from year to year. The stratified random survey design used in this study helped to understand the habitat use patterns of the many coral reef organisms that utilize multiple habitat types, as well as for recording the distribution of widespread species such as the invasive lionfish (*Pterois voltans*). Given that the NPS SOPs are being optimized for tracking population status and trends for select economically and ecologically important fish species on reef and hard bottom habitats, it is recommended that NPS develop additional programs or expand on existing efforts to track the status and trends of other coral reef ecosystem components such as lobster, conch, urchins, and seagrass beds which provide important ecological goods and services.

In addition, acoustic tracking studies may reveal the mechanisms underlying some of the observed temporal changes in fish communities and will determine connectivity between lagoons and coral reefs offshore (Monaco et al., 2008). Tracking will also provide important information on the time that individual fish spend inside and outside the boundaries of protected areas.

Benthic habitat maps should be periodically updated due to the dynamic nature of coral reef ecosystems. This is particularly important when linking fish seascape structure and when assessing seascape change such as quantifying gain or loss of major habitat types. Also, additional mapping, inventory and monitoring efforts are required to explore the deeper water ecosystems around VIIS and VICR boundaries that exist outside NOAA's current benthic habitat maps, and to quantify effects of changes in seascape (i.e., gain or loss of major habitat types) on fish assemblages.

This body of work will contribute greatly to Coastal and Marine Spatial Planning (CMSP) in the U.S. Caribbean. CMSP is a comprehensive, integrated national policy for the stewardship of the U.S. oceans, coasts, and Great Lakes that was established by Presidential Executive Order (E.O.) 13547. CMSP is intended to improve existing governmental decision-making and planning processes to enable an integrated, comprehensive, and adaptive approach for sustainable use and conservation of the nation's oceans, coasts, and Great Lakes. A stated intent of this policy is to achieve "an America whose stewardship ensures that the ocean, our coasts, and the Great lakes are healthy and resilient, safe and productive, and understood and treasured, so as to promote the well-being, prosperity, and security of present and future generations" (http://www.whitehouse.gov/files/documents/2010stewardship-eo.pdf). The status and trends of resources described in this report provide critical information and relevant baselines upon which territorial and federal management agencies can use CMSP activities to improve sustainability and conservation of marine resources in the USVI. The biological data and habitat maps synthesized in this report encompass the most spatially intensive characterization of any region in the Caribbean. The data will be invaluable for mapping the seafloor structure and associated biological communities to support marine management decision making. The report highlights some of the ways that information on shallow water coral reef ecosystems can be made spatially-explicit. CMSP efforts through the regional ocean partnership will require such information to effectively manage ocean uses, avoid conflicts between conservation interests and industry and other human uses.

Where management targets include protection of marine biodiversity, these data can be used to highlight diversity hotspots and cold spots and prioritize areas based on known sensitivity and vulnerability to human activities. In understanding resiliency, these data can be used to identify areas with biophysical properties that result in high or low resilience to disturbance. To examine land-sea connectivity, these data can be aligned with indices of terrestrial landscape condition to identify potential impact areas or threats from land requiring priority attention from managers and the broader island community.

RECOMMENDATIONS FOR ADDITIONAL ANALYSES

- The data within this report highlights diversity hotspots and species-habitat associations that can support decision making in prioritization of management actions and for targeted risk assessments. For example, if an area that supports high faunal diversity is proximal to a stressor then the adaptive management system should be activated and the sensitivity and vulnerability assessed.
- The faunal data has additional utility in understanding geographical and habitat/seascape preferences for species. This information can be combined with habitat maps and high resolution bathymetry to predict habitat suitability for particular species and to examine the impact of changes in habitat on habitat suitability.
- Determine if trends detected by the NPS SFCN monitoring at specific permanent monitoring stations also occur at the island scale (e.g., as may be detected by NCRMP) or within NPS waters versus outside NPS jurisdiction.
- Use the diversity hotspot information to understand the environmental drivers for these hotspots at specific sites since this information can inform conservation strategies, help anticipate impacts from environmental changes, guide restoration activities and the setting of ecologically realistic targets for recovery.
- Use the information herein to rank sites and prioritize actions that are local and involve local outreach to raise community awareness.
- More targeted surveys should be designed and implemented to obtain information on cryptic species and nocturnally active species such as lobster, *Diadema*, some fish, and macroinvertebrates.
- Addressing the rarity of large-bodied groupers requires a targeted effort to understand sources and sinks, juvenile settlement habitat, and identify any limits to population recovery. Connectivity to local shelf edge spawning aggregations such as Hind Bank and Grammanik Bank has now been confirmed through acoustic tracking, but the importance of these sites to populations within the NPS marine protected areas (MPAs) requires further investigation. Identification of key spawning aggregations and co-management of these critical sites along with Territorial agencies and the fishing community should be considered if rebuilding trophic structure of coral reef ecosystems is a desired management objective.

REFERENCES

Beets, J. 1997. Can coral reef fish assemblages be sustained as fishing intensity increases? pp. 2009-2014. In: H.A. Lessios and I.G. Macintyre (eds.), Proceedings of the 8th International Coral Reef Symposium, Vol. 2. Panama City, Panama. 2118 pp.

Boulon, R.H., M.E. Monaco, A.M. Friedlander, C. Caldow, J. Christensen, C. Rogers, J. Beets, J. Miller, and S.D. Hile. 2008. An ecological correction to marine reserve boundaries in the U.S. Virgin Islands. pp. 1082-1085. In: B.M Riegl and R.E. Dodge (eds.), Proceedings of the 11th International Coral Reef Symposium, Vol. 2. Ft. Lauderdale, FL. 742 pp.

Bryan, D.R., A.J. Atkinson, J.S. Ault, M.E. Brandt, J.A. Bohnsack, M.W. Feeley, M.E. Patterson, B.I. Ruttenberg, and S.G. Smith. In Press. A cooperative multiagency reef fish monitoring protocol for the U.S. Virgin Islands coral reef ecosystem. Natural Resource Report NPS/SFCN/NRR—2013/XXX. National Park Service, Fort Collins, Colorado.

Friedlander, A.M. and J. Beets. 2008. Temporal Trends in Reef Fish Assemblages inside Virgin Islands National Park and around St. John, U.S. Virgin Islands, 1988-2006. NOAA Technical Memorandum NOS NCCOS 70. Silver Spring, MD. 60 pp. Online: http://www.ccma.nos.noaa.gov/publications/VITempTrends.pdf (Accessed 14 May 2013)

Kendall, M.S., C.R. Kruer, K.R. Buja, J.D. Christensen, M. Finkbeiner, R.A. Warner, and M.E. Monaco. 2001. Methods used to map the benthic habitats of Puerto Rico and the U.S. Virgin Islands. NOAA Technical Memorandum NOS NCCOS 152. Silver Spring, MD. 45 pp. Online: http://ccma.nos.noaa.gov/products/biogeography/usvi_pr_mapping/manual.pdf (Accessed 14 May 2013)

Menza, C., J. Ault, J. Beets, C. Bohnsack, C. Caldow, J. Christensen, A. Friedlander, C. Jeffrey, M. Kendall, J. Luo, M.E. Monaco, S. Smith, and K. Woody. 2006. A guide to monitoring reef fish in the National Park Service's South Florida/Caribbean Network. NOAA Technical Memorandum NOS NCCOS 39. Silver Spring, MD. 166 pp. Online: http://www.ccma.nos.noaa.gov/publications/Reefmonitoringguide.pdf (Accessed 14 May 2013)

Monaco, M.E., A.M. Friedlander, C. Caldow, J.D. Christensen, J. Beets, J. Miller, C. Rogers, and R. Boulon. 2007. Characterizing reef fish populations and habitats within and outside the U.S. Virgin Islands Coral Reef National Monument: a lesson in MPA design. Fish. Manage. Ecol. 14:33-40.

Monaco, M. E., A. M. Friedlander, C. Caldow, S.D. Hile, C. Menza, and R. Boulon. 2009. Long-term monitoring of habitats and reef fish found inside and outside the U.S. Virgin Islands Coral Reef National Monument: A comparative assessment. Carib. J. Sci. 45(2-3): 338-347.

Pittman, S.J., S.D. Hile, C.F.G. Jeffrey, C. Caldow, M.S. Kendall, M.E. Monaco, and Z. Hillis-Starr. 2008. Fish assemblages and benthic habitats of Buck Island Reef National Monument (St. Croix, U.S. Virgin Islands) and the surrounding seascape: A characterization of spatial and temporal patterns. NOAA Technical Memorandum NOS NCCOS 71. Silver Spring, MD. 96 pp. Online: http://ccma.nos.noaa.gov/publications/biogeography/STC_LTR_Final_0309.pdf (Accessed 14 May 2014)

Pittman, S.J., S.D. Hile, C.F.G. Jeffrey, R. Clark, K. Woody, B.D. Herlach, C. Caldow, M.E. Monaco, and R. Appeldoorn. 2010. Coral reef ecosystems of Reserva Natural La Parguera (Puerto Rico): Spatial and temporal patterns in fish and benthic communities (2001-2007). NOAA Technical Memorandum NOS NCCOS 107. Silver Spring, MD. 202 pp. Online: http://ccma.nos.noaa.gov/publications/nccostechmemo107.pdf (Accessed 14 May 2013)

Rogers, C.S., J. Miller, E.M. Muller, P.J. Edmunds, R.S. Nemeth, J. Beets, A. Friedlander, T.B. Smith, R. Boulon, C.F.G. Jeffrey, C. Menza, C. Caldow, N. Idrisi, B. Kojis, M. Monaco, A. Spitzack, E. Gladfelter, J. Ogden, Z. Hillis-Starr, I. Lundgren, W.B. Schill, I. Kuffner, L.L. Richardson, B. Devine, and J. Voss. 2008. Ecology of coral reefs in the US Virgin Islands. pp. 303-373. In: B.M. Riegl and R.E. Dodge (eds.), Coral Reefs of the USA. Coral Reefs of the World, Volume 1. Springer, Berlin. 806 pp.

Chapter 1: Introduction

Alan M. Friedlander[1] , Sarah D. Hile[2,3] and Simon J. Pittman[2,3,4]

1.1. BACKGROUND

Figure 1.1. Photo of Trunk Bay on the north shore of St. John, U.S. Virgin Islands (USVI). Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.

NOAA's National Ocean Service (NOS), National Centers for Coastal Ocean Science (NCCOS), 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 St. John, U.S. Virgin Islands (USVI) since 1999 (Figure 1.1). This effort is part of the broader NOAA Coral Reef Conservation Program's (CRCP) National Coral Reef Ecosystem Monitoring Program (NCREMP) that evaluates the status and trends of coral reefs in U.S. waters. With support from CRCP, CCMA-BB conducts the Caribbean Coral Reef Ecosystem Monitoring (CREM) project, the goals of which are 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 also contributed to the Coral Reef Ecosystem Studies (CRES) program in Puerto Rico and the U.S. Virgin Islands. CRES was a 5-year research program funded through NOAA's NCCOS Center for Sponsored Coastal Ocean Research (CSCOR) and coordinated by the University of Puerto Rico's (UPR) Department of Marine Sciences to define and understand causes and effects of reef degradation, and provide managers information and tools to aid in reversing the degradation of U.S. Caribbean reef ecosystems. The St. John component of CRES focused on understanding how habitat utilization by fish and macroinvertebrate species varied inside and outside of two marine protected areas (MPAs). Basic premises of CRES were that cross-habitat movements of fishes and macroinvertebrates play vital roles in the health, structure, and function of coral reef systems; and that the movement and spatial distributions of these organisms are determined by the types, amounts and distribution of different habitats. Fishes and macro-invertebrates are important vectors for the non-random transport of nutrients, organic matter and energy among habitats. Furthermore, their activities often result in hotspots of productivity that are temporally and spatially predictable.

The intent of this report is to provide a comprehensive spatial and temporal characterization of the coral reef ecosystems around St. John, as well as, to evaluate the efficacy of MPAs as a resource

[1] U.S. Geological Survey
[2] Center for Coastal Monitoring and Assessment, National Centers for Coastal Ocean Science, National Ocean Service, National Oceanic and Atmospheric Administration
[3] Consolidated Safety Services, Inc.
[4] University of the Virgin Islands

management tool. Cross-shelf movements of fish and invertebrate species can have far-reaching consequences for regional fisheries and the performance of MPAs. Coral reef degradation is likely to affect the strength of ecological linkages and how species utilize particular habitats. By the same token, the breakdown of these ecological linkages through overfishing and alteration of community structure, will lead to a decline in coral reef productivity and health. The information provided in this report will provide resource managers with a better understanding of coral reef ecosystems in St. John and will improve human stewardship of goods and services obtained from these ecosystems.

1.2. INTRODUCTION TO THE STUDY REGION

1.2.1. Region background

The USVI is located in the Lesser Antilles and consist of three main islands: St. Croix, St. Thomas and St. John (Figure 1.2a). St. John is the smallest of the three main islands (52 km^2) and is located on the Puerto Rican shelf in the Eastern Caribbean (Figure 1.2b). Its topography primarily consists of steep slopes with outcroppings and exposed cliffs that shape numerous bays around the island (Rankin, 2002; Figure 1.3). Because of its location on the Puerto Rican Shelf, the ecology and geology of St. John resemble those of the islands of Puerto Rico and St. Thomas more closely than its sister isle St. Croix, which is further south (Hubbard et al., 2008).

The island's climate is warm and tropical with average monthly temperature ranging from 77°F during December through May to 83°F during June through November (NOAA SERO, 2007). Rainfall averages about 45.3 inches per year; the rainiest months are September through November, but torrential showers frequently occur throughout the year (NOAA SERO, 2007). Prevailing winds are easterly but vary both in intensity and direction, with maximum winds occurring during winter months and minimal airflow during fall. These winds along with tidal variation drive coastal surface currents that range from 0-40 cm/s (Halliwell and Mayer, 1996; Rogers et al., 2008). Occasional freshwater

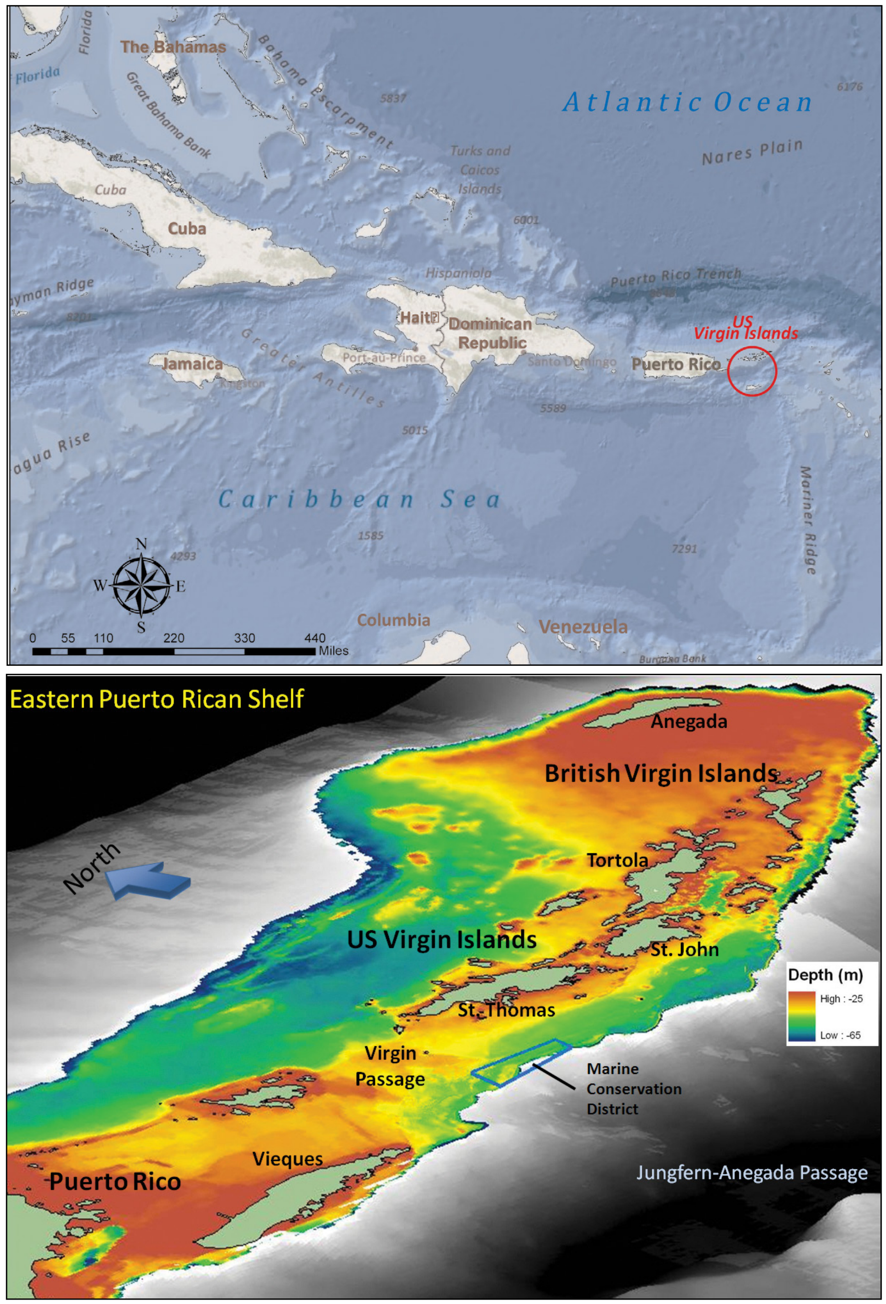


Figure 1.2. a) The three main islands of the USVI (St. Thomas, St. John and St. Croix) are part of the Lesser Antilles in the eastern Caribbean. b) The islands are located on a shallow insular shelf, the Puerto Rican Shelf, forming the boundary between the Caribbean continental plate and the North American plate. The shelf edges descend to the deep abyssal waters of the Puerto Rico Trench (max depth 8,800 m) on the north side and Jungfern-Anegada Passage (max depth 1,915 m) to the south. Source: a) W. Sautter (NOAA/NOS/NCCOS/CCMA/Biogeography Branch); b) Adapted from map produced by Tyler Smith (University of the Virgin Islands).

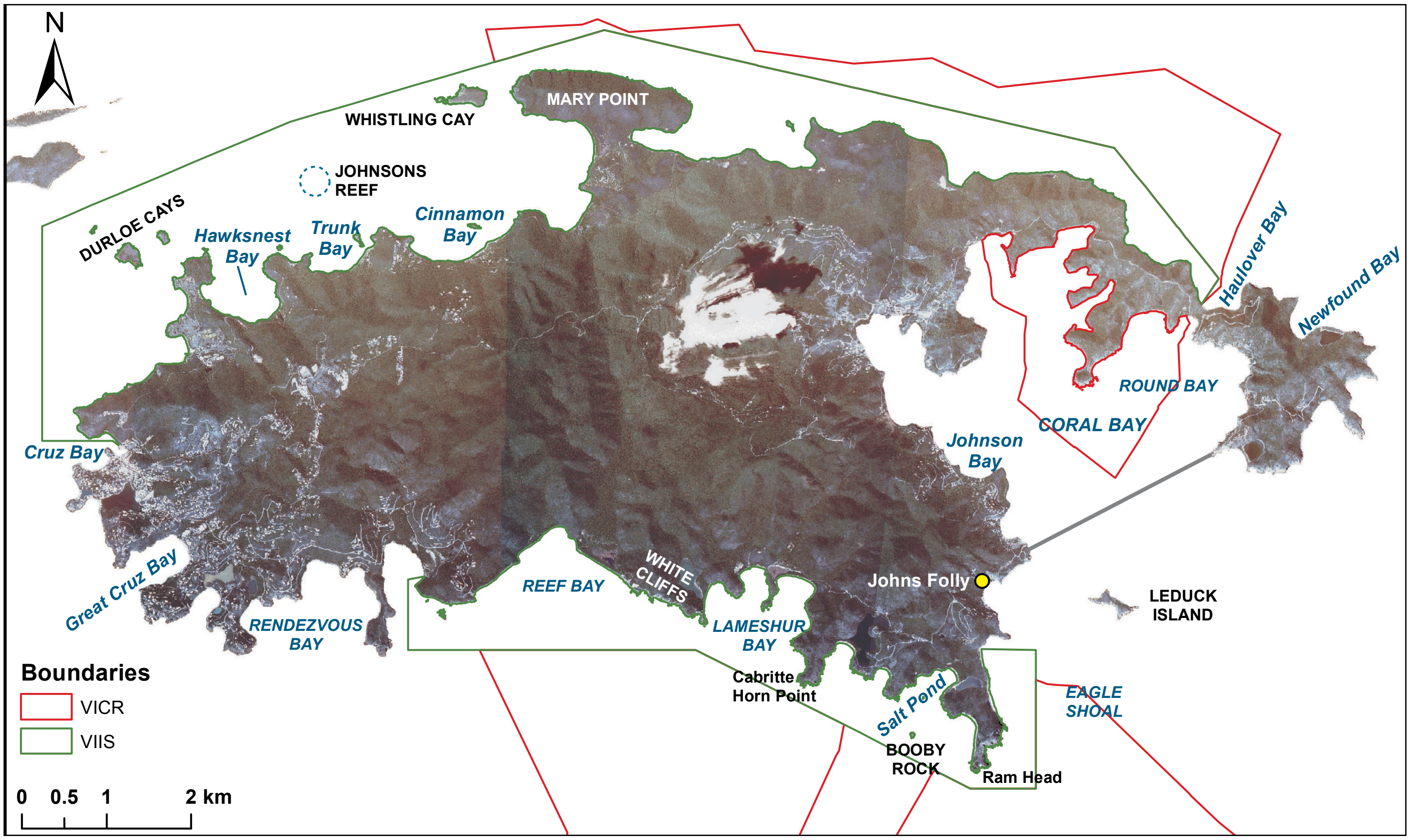


Figure 1.3. Locations of selected bays and geological features around St. John, USVI. Source: S.D. Hile (NOAA/NOS/NCCOS/CCMA/Biogeography Branch).

lenses migrating from the Amazon and Orinoco rivers have lowered salinity and increased nutrients around coral reefs in St. John during summer months (Hu et al., 2004; Rogers et al., 2008). Spatial variation in sea water temperature is minimal, although the average sea water temperature at Greater Lameshur Bay has increased by 0.6°C per decade since 1989 (Edmunds, 2004). Hurricanes, a major environmental stressor to coral reefs, occur most often during June through November, with the number of storms peaking during August and September (NOAA SERO, 2007).

The island of St. John is surrounded by a mosaic of linear, pavement, and patch coral reefs; seagrass beds and algal plains; along with fields of rhodolith rubble and sand sediments (Kendall et al., 2001; Menza et al., 2008; Pittman et al., 2007; Figure 1.4). The most developed reefs tend to occur along the northeastern and windward end of the island. The coral seascape generally is dominated by *Acropora* species in shallower, nearshore habitats (depth <6 m) and by the frame building coral *Montastraea* species in deeper reef habitats (~30-50 m; Rogers et al., 2008). The coral structure of nearshore habitats are primarily aggregated patch reefs within a matrix of sand and seagrass habitat. In deeper water (30 m), a linear reef complex known as the Mid-Shelf Reef is located about 2-8 km south of St. Thomas and St. John and ranges in depth from 10-70 m (Menza et al., 2008; Monaco et al., 2008; Figure 1.4).

1.2.2. St. John and its marine protected areas

Virgin Islands National Park (VIIS) was established in 1956 to protect the ecosystems on the island of St. John, with marine portions added in 1962. The park consists of 2,947 hectares of land (about 56% of the 48-km^2 island) and 2,287 ha of surrounding waters (Figure 1.3) under the jurisdiction of the National Park Service (NPS). Within the park, taking of fishes or other marine life is prohibited except with rod and line or traps of ‘conventional Virgin Islands design’ and small seine nets. Trunk Bay (21 ha) is technically a no-take area where all fishing is prohibited. When the park was first established, fishers usually set only a few, small traps, but with the advent of outboard motors, line

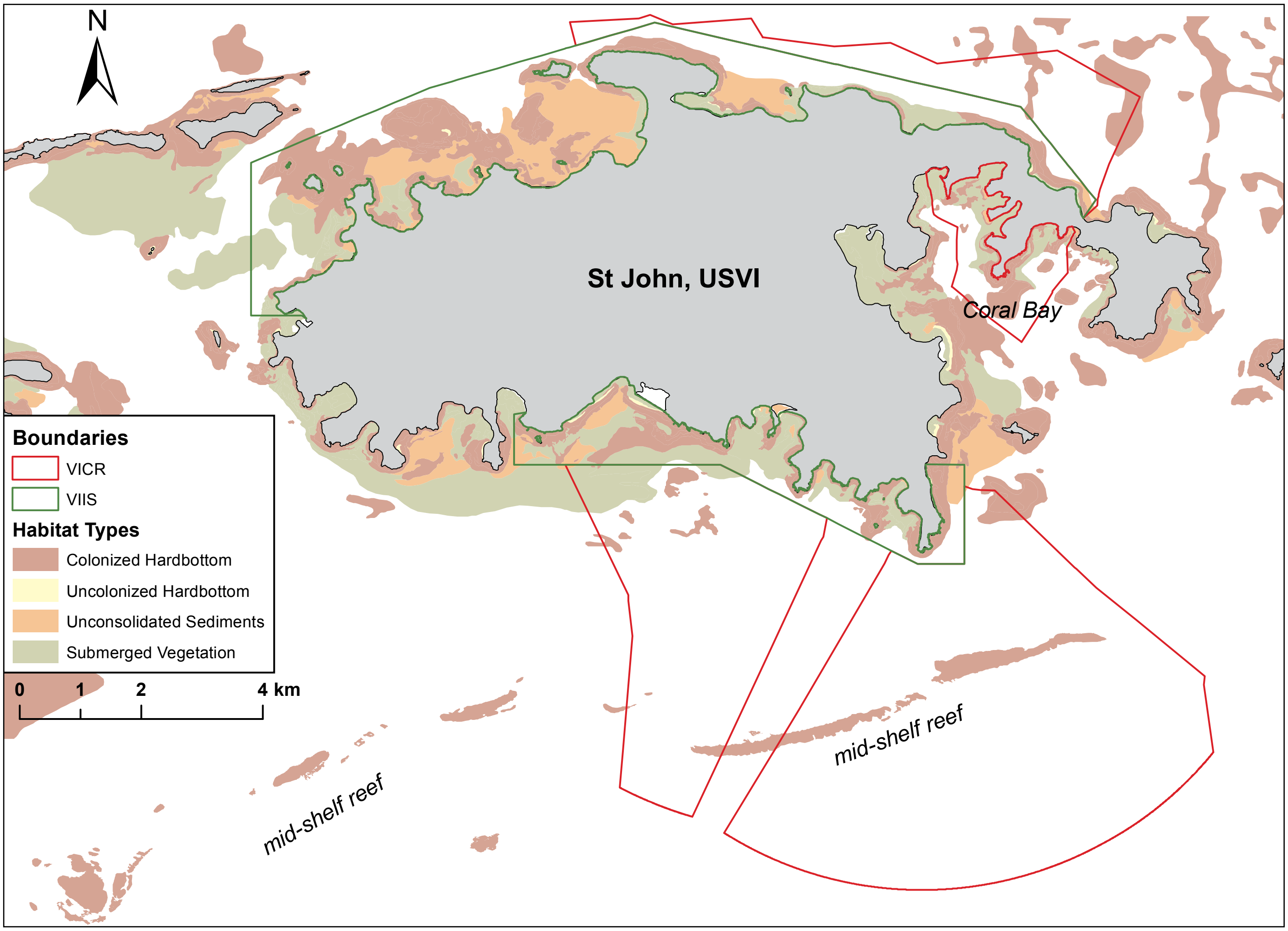


Figure 1.4. Nearshore benthic habitat of St. John, USVI. Source: S.D. Hile (NOAA/NOS/NCCOS/CCMA/Biogeography Branch).

hauls, and larger fiberglass boats, fishermen now fish further offshore with a larger number of traps (Beets, 1997; Garrison et al., 1998).The nature of these regulations means that fishing still persists within VIIS, with fisheries resources and the marine environment as a whole having shown dramatic declines in recent decades.

Owing to these declines, Virgin Islands Coral Reef National Monument (VICR) was established by Presidential Proclamation 7399 in 2001. This new monument added ca. 5,143 ha of marine habitat off the island of St. John, greatly increasing the NPS jurisdiction in USVI waters (Figure 1.4). Provisions within the Presidential Proclamation prohibit all extractive uses with the exception of fishing for a coastal pelagic species, Blue Runner (*Caranx crysos*) south of St. John and bait fishing in a small area within the Coral Bay component of VICR. In addition, boat anchoring is prohibited in VICR, except for emergency or authorized administrative purposes. The administrative (political) process used to establish VICR did not allow a robust ecological characterization of the area to determine the boundaries of the MPA. Analyses of surveys of habitat and fishes inside and outside of VICR that were conducted along the mid-shelf reef in 2002-07 have revealed that areas outside VICR (i.e. deeper areas beyond the monument boundary and the area called the "wedge") had significantly more hard corals; greater habitat complexity; and greater richness, abundance and biomass of reef fishes than areas within VICR (Monaco et al., 2007). Efforts are underway to increase amounts of complex reef habitat within VICR by swapping a part of VICR that has little coral reef habitat for a Territorially-owned area within VICR that contains a coral reef with higher coral cover (Boulon et al., 2008).

1.2.3. Coral Bay

At 13.3 km^2, Coral Bay represents the largest land surface area draining into an individual bay on St. John (Figure 1.3). Coral Bay encompasses over 16 km of shoreline, including some of St. John's largest salt ponds, extensive mangrove habitat, sea grass beds and fringing reefs. The bay includes portions of the VICR and supports protected *Acropora* corals and sea turtle nesting areas.

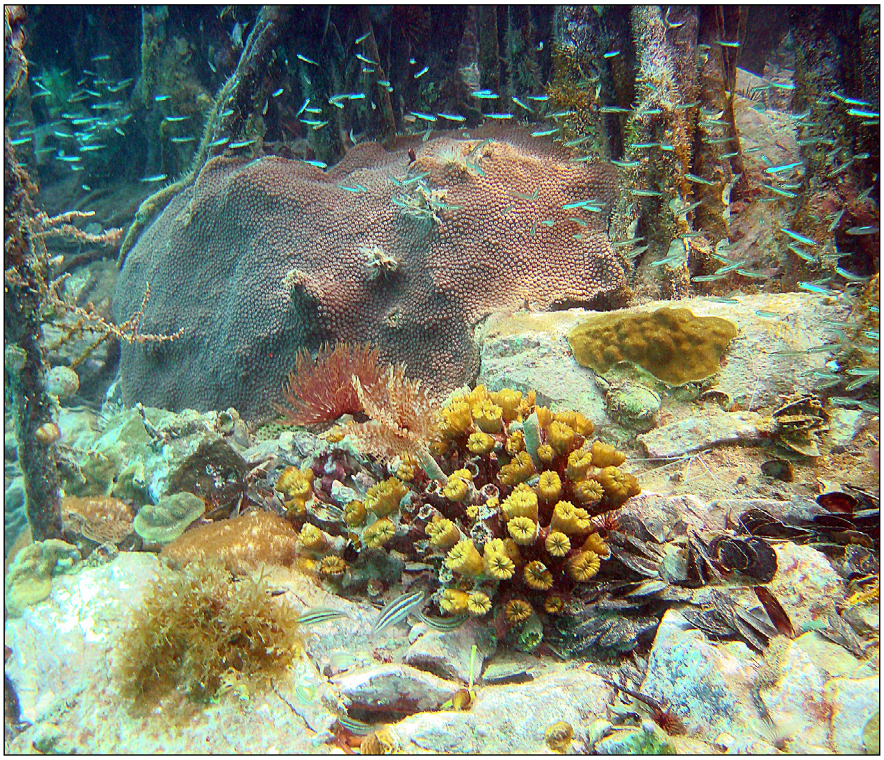


Figure 1.5. Coral rich communities amongst the red mangrove roots of Hurricane Hole, Coral Bay. Credit: Dr. Caroline Rogers (U.S. Geological Survey).

Several of the small mangrove-lined bays within the Coral Bay portion of VICR support diverse coral communities. In St. John, mangrove habitats are the most extensive, best developed, and least disturbed within the large embayment in Coral Bay, known as Hurricane Hole (Figure 1.5). Mangroves in that area function as a nursery habitat for juvenile fish, spiny lobsters, and queen conch (*Strombus gigas*); in fact, Hurricane Hole was included in the VICR, partly to protect the mangroves there (U.S. Presidential Proclamation 7399, 2001). As many as 28 species of scleractinian coral have been identified from initial surveys in 2008 (Rogers, 2009). Furthermore, many of the coral colonies within Hurricane Hole mangroves generally appear healthier than those on coral reefs around St. John (Rogers, 2011). U.S. Geological Survey (USGS) and NPS scientists are monitoring and studying these colonies as case-study examples of resilience because they survived the mass bleaching event in 2005 and 2006. The unusual mangrove communities of Hurricane Hole are documented in detail elsewhere (Rogers, 2011).

The watershed is characterized by steep slopes (averaging 18%, with a large percentage over 35%), highly erodible soils, and high runoff volumes associated with average rain events. These factors, combined with a large percentage of dirt roads, active construction, and no existing storm water management, have been shown to contribute to excessive sediment loading to the bay (Devine et al., 2003; Brooks et al., 2004; Ramos-Scharron and MacDonald, 2005). In addition, the watershed experienced an approximate 80% population increase between 1990 and 2000, making it the fastest growing area in the USVI.

The Coral Bay Community Council, Inc. (CBCC), a local nonprofit watershed management association, identified erosion and bay sedimentation as priority issues threatening both marine ecosystem health and the community's quality of life. Through the American Reinvestment and Recovery Act (ARRA), over $2.7 million was awarded to the Virgin Islands Resource Conservation and Development Council (V.I. RC&D), to implement the USVI Coastal Habitat Restoration through Watershed Stabilization Project. This work complements CBCC's Coral Bay Watershed Management Project and utilizes designs developed under CBCC's Environmental Protection Agency's (EPA) Community Action for a Renewed Environment (CARE) grant. The overarching themes of these projects are to improve coastal ecosystem condition in Coral Bay, St. John through an immediate and long-term reduction in sediment loading to the bay, and to stimulate the local economy through the creation of jobs and infrastructure improvements. Results from this report can be used as a baseline to help monitor potential changes in the coral reef community in Coral Bay as a result of changes in land-based sedimentation.

1.2.4. Environmental monitoring and ecosystem changes in St. John, USVI

Impacts from Multiple Stressors

The health, abundance and structural integrity of Caribbean coral reef ecosystems, including the USVI, are declining and continue to be threatened by multiple stressors (Bellwood et al., 2004; Wilkinson, 2004; Pandolfi et al., 2005; Rogers and Beets, 2001; Rogers et al., 2008; Rothenberger et al., 2008). The collapse of many Caribbean coral reefs has been attributed to increased euthrophication, sediment runoff, thermal stress, and is thought to be exacerbated by dwindling fish populations, particularly herbivorous fish (Hughes, 1994; Jackson et al., 2001; Mumby et al., 2006; Newman et al., 2006). Several major stressors have affected coral reefs in the USVI, including St. John. These include the mass die-off of the long-spined sea urchin (*Diadema antillarum*) in the early 1980s; several hurricanes beginning with Hurricane Hugo in 1989; and mass mortality of *Acropora* and other reef-building corals from disease and coral bleaching events, the most recent of which occurred in 2005 (Rogers and Beets, 2001; Rogers and Miller, 2001; Miller et al., 2006; Clark et al., 2009; Figures 1.6 and 1.7). Chronic overfishing during the past few decades have reduced populations of food fish and have also contributed to the overall degradation of coral reefs in St. John (Beets, 1997; Beets and Friedlander, 1999; Rogers and Beets, 2001; Rogers et al., 2008). Urbanization of the watersheds outside of VIIS has led to increased sediment runoff in to coastal waters. American Recovery and Reinvestment Act (ARRA) funded projects have been completed in some areas to control runoff and watershed management plans are being developed and updated. Comparison of St. John land cover changes over a 60 year period reveal the extent of change from large areas of grazing land, small settlements and few roads in 1947 (Figure 1.8) to the more recent land use patterns showing extensive regrowth of grazed land in the Park, but with intensive housing development in pockets of land outside NPS jurisdiction.

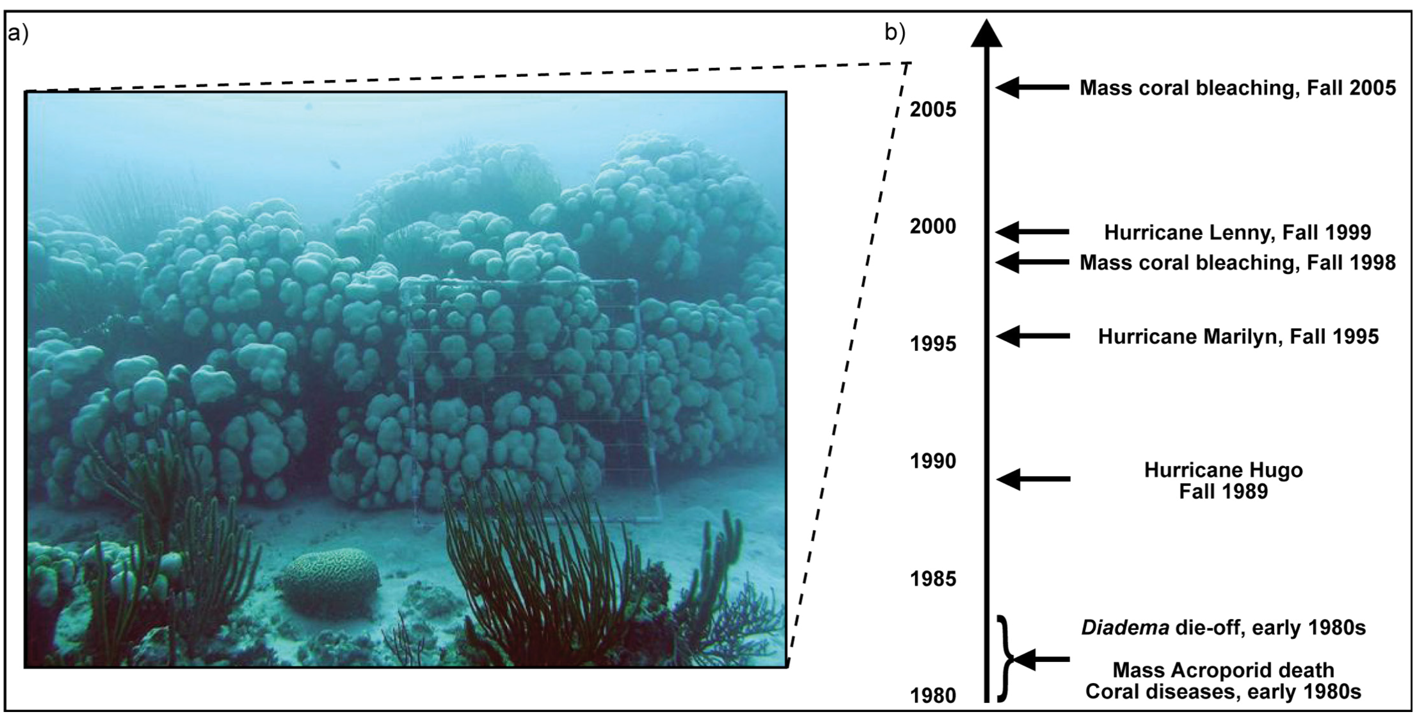


Figure 1.6. Chronology of major catastrophic environmental events impacting the structure and function of coral reef ecosystems in the USVI. Photograph shows bleached corals. Source: Pittman et al. (2008).

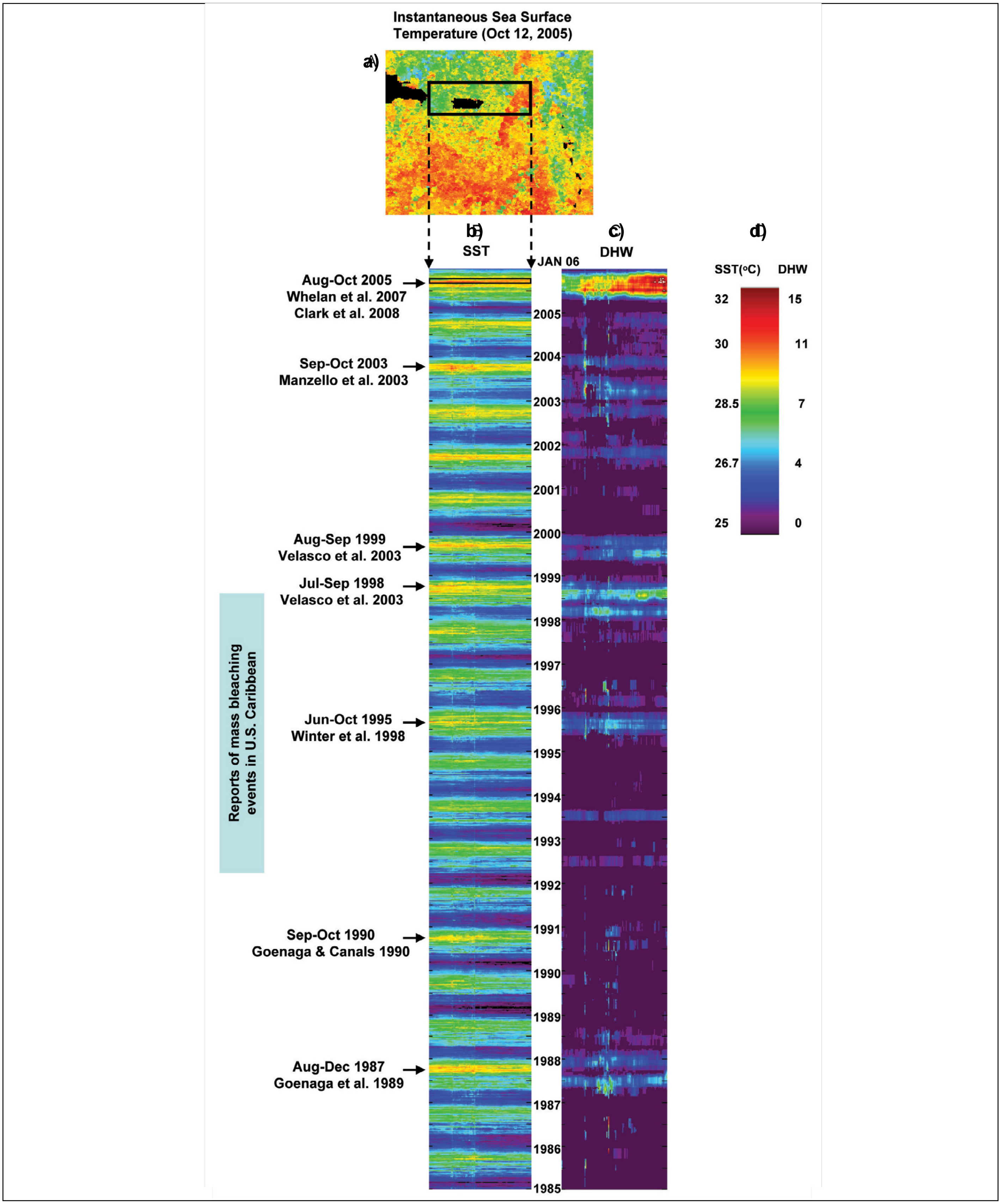


Figure 1.7. a) Instantaneous sea surface temperature (SST) during one of the days that coincided with the 2005 mass bleaching event recorded in the U.S. Caribbean (Clark et al., 2009). The SST data were recorded by the Advanced Very High Resolution Radiometer (AVHRR). b) A space-time chart showing the 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 (NOAA/NOS/NCCOS/CCMA/COAST).

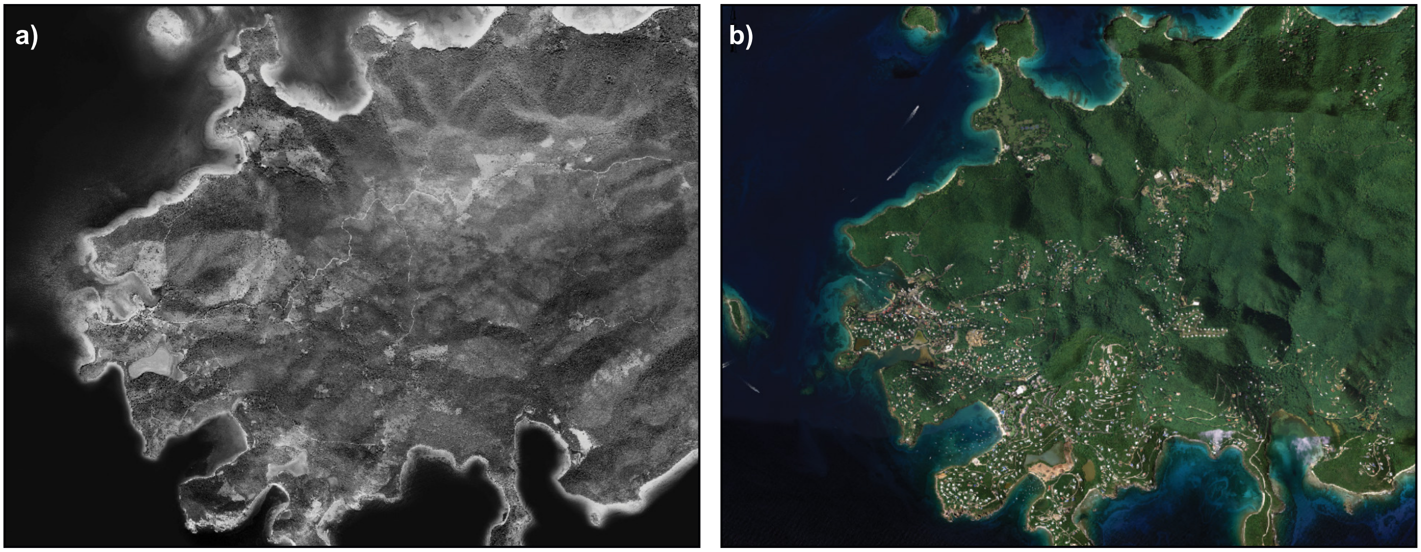


Figure 1.8. Sixty years of land use change on St. John can be examined and quantified through change analyses conducted on historical aerial photography such as this photo mosaic from: a) 1947 (panchromatic) and b) 2007 (color). Source: U.S. Army Corps of Engineers (USACE) and NOAA/NOS/NCCOS/CCMA Biogeography Branch.

Coral cover around St. John has declined over time due to hurricanes (Rogers et al., 1997), anchor damage (Rogers and Garrison, 2001), disease (Miller et al., 2003; 2009), and loss of herbivores due to overfishing and disease (Rogers et al., 2008). Although all of the diseases currently identified in the Caribbean are found in the USVI, white plague and band disease have had the most severe impact on the coral community (Rogers et al., 2008; Woody et al., 2008; Miller et al., 2009). In 2005, a massive coral bleaching event in the northeast Caribbean and a subsequent severe disease outbreak caused a 60% decline in corals in the USVI (Miller et al., 2009; Figure 1.9). Black band disease has been known to exist in the USVI since the 1980s (Edmunds, 1991), but has affected far fewer species than white plague (Rogers et al., 2008).

One of the greatest justifications for consistent and regular annual monitoring is to document the effects of natural events, such as the impact of hurricanes, bleaching and disease prevalence and to attempt to differentiate natural fluctuations from anthropogenic stressors. The magnitude and periodicity of disturbances greatly affects the spatio-temporal patterns observed on coral reefs (Done et al., 1991; Connell, 1997). The trajectories of these trends are determined by the synergistic effects of local and regional processes (Connell, 1997; Bythell et al., 2000), therefore monitoring needs to be conducted over time scales commensurate with the periodicity of these disturbance events.

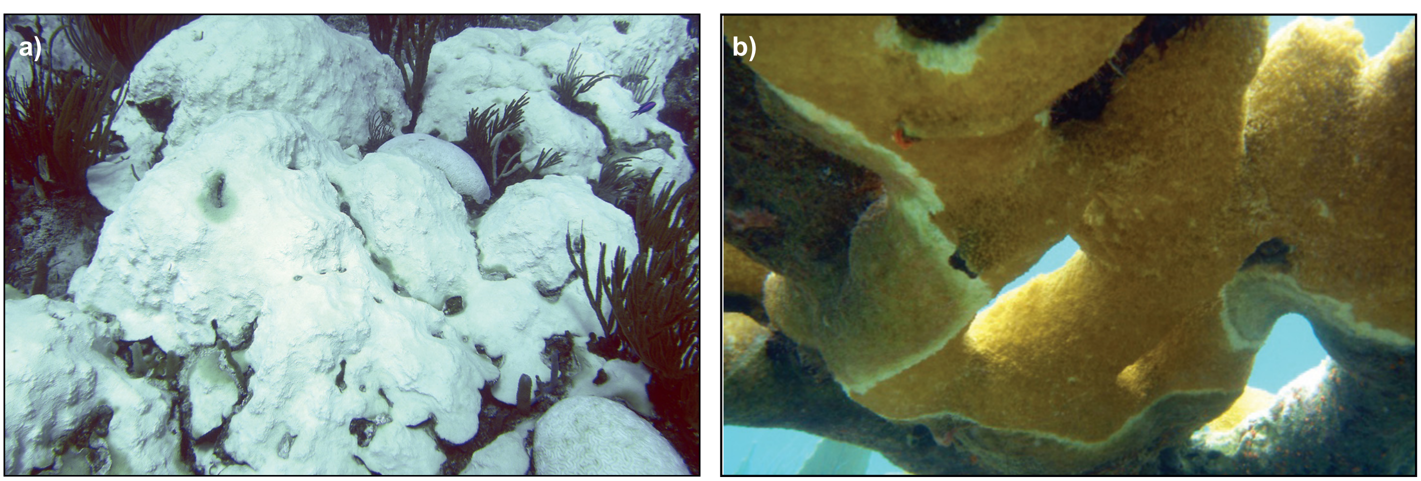


Figure 1.9. a) Bleached Montastraea annularis *complex coral in the USVI (October 2005) and b) diseased* Acropora palmata. *Sources: NOAA/NOS/NCCOS/CCMA/Biogeography Branch and J. Miller (National Park Service).*

Hurricanes and tropical storms

Numerous storms affected the community structure of reefs around St. John during the monitoring period covered in this report, with some storms having large effects (Rogers and Beets, 2001; Figure 1.10). The two largest storms passing St. John, Hurricane Hugo (1989) and Hurricane Marilyn (1995), devastated some reefs and had less influence on others (Rogers et al., 1991, 1997). Long-term and consistent monitoring data allowed for more critical assessment of these large disturbances and the differential effects that storms had on fish assemblage structure in addition to the impacts from fishing and other anthropogenic stressors.

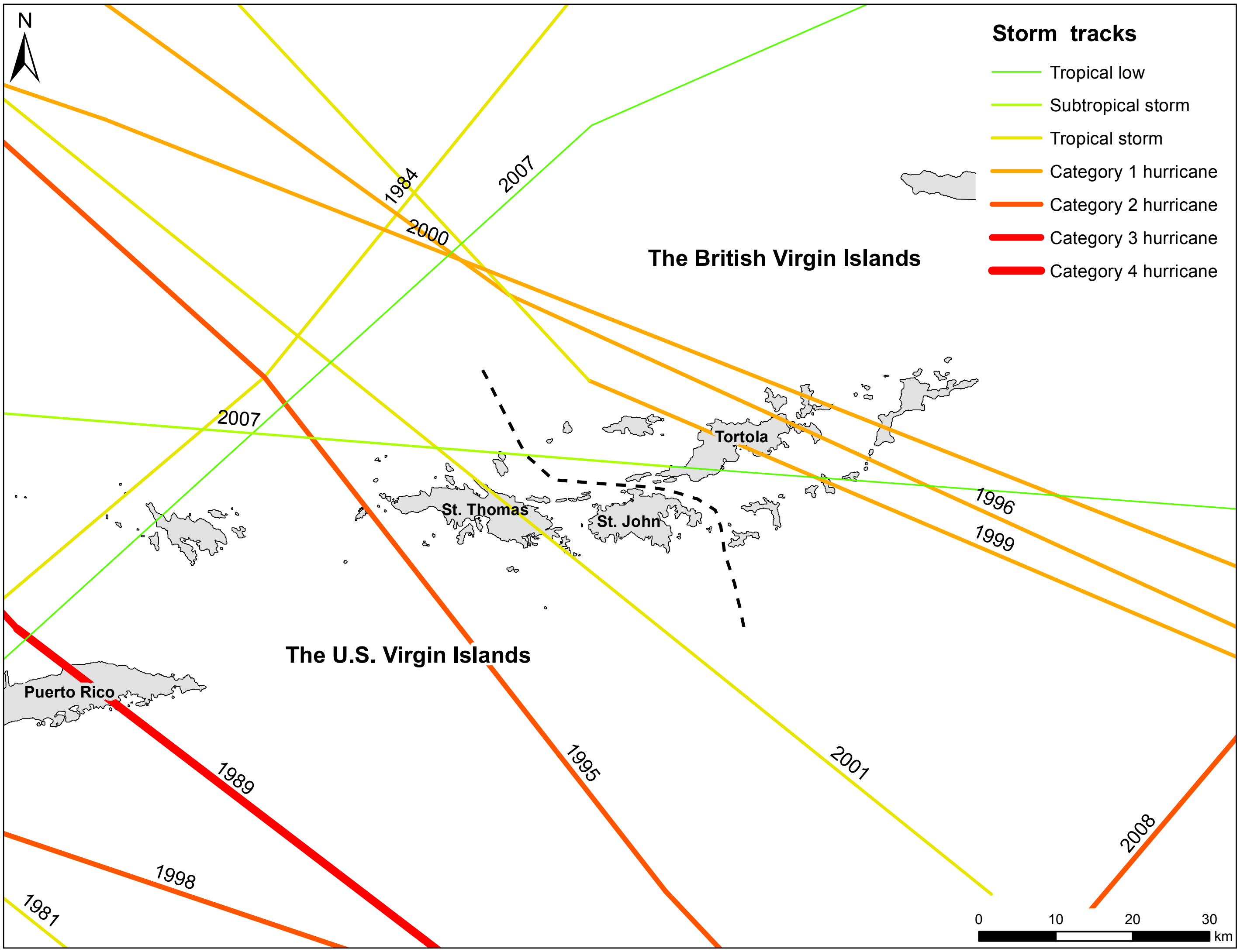


Figure 1.10. Tracks of major storms influencing marine habitats of the U.S. Caribbean (1979-2008). Source: K. Stamoulis (University of Hawaii).

Fishing

About 180 species of reef fishes are harvested in the USVI (Caribbean Fisheries Management Council, 1985), with the primary fishing gear being traps, followed by hook and line, and nets. Fishery resources throughout the USVI (Figure 1.11), including those within VIIS, have declined dramatically over the last 30-40 years in spite of federal and territorial government regulations designed to protect them (Beets, 1997; Friedlander and Beets, 2008; Rogers et al., 2008). As far back at the late 1950s, Randall (1963) noted that the limited fringing reef area around the USVI received nearly all of the fishing effort, and as a consequence the effects of overfishing were evident. Large predatory fishes such as groupers and snappers are now far less abundant, the relative abundance of herbivorous fishes has increased, individuals of many fish species are smaller, and some once heavily fished spawning aggregations have disappeared (Beets and Friedlander, 1992, 1999; Beets, 1997). However, a large multi-species

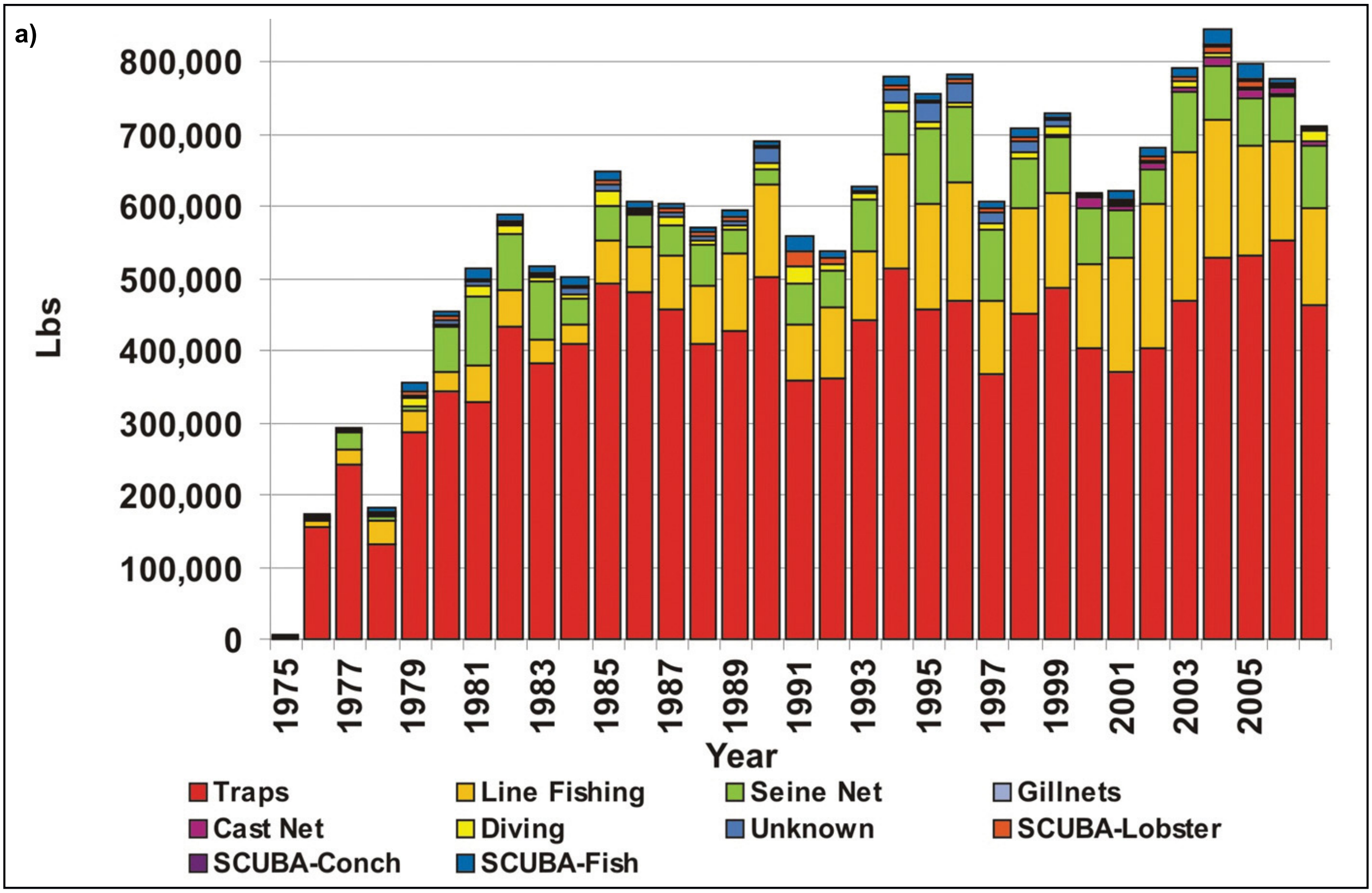


Figure 1.11. a) Thirty years of fish landings by weight (lbs) for the St. Thomas-St. John fishery (1975-2005) showing highest catch by traps followed by line fishing; b-c) USVI has a long history of trap fishing with skills passed down through generations of fishing families. Source: Dr. David Olsen (St. Thomas Fishermen's Association).

spawning aggregation at the shelf edge, Hind Bank, south of St. Thomas (Marine Conservation District [see Figure 1.2b]) closed to fishing since 1999, has shown marked recovery in snapper and grouper aggregations, particularly in populations of red hind (Nemeth, 2005). Furthermore, recent acoustic tracking studies have revealed that several species of large-bodied grouper and snapper migrate to the shelf edge spawning site from the nearshore waters of the national park (Pittman and Legare, 2010). This ecological connectivity will be critical to the recovery of vulnerable fished species in the national park and monument.

In the 1960s, groupers and snappers dominated the landings in the USVI fishery, but following the increased demand for fish with the tourism boom and technological changes in the fishery (larger boats, engines, and improved gear), fishers began to set more traps and target species like groupers and snappers, including fishing at spawning aggregations (Olsen and LaPlace, 1979; Beets and

Friedlander, 1992). Nassau Grouper (*Epinephelus striatus*) forms spawning aggregations around the full moon during winter months and was one of the dominant species in the fishery until the 1970s when the aggregation off St. Thomas was fished to collapse (Olsen and LaPlace, 1979; Beets and Friedlander, 1992). Following the decline of Nassau Grouper and other large grouper species, fishers targeted smaller groupers such as Red Hind (*Epinephelus guttatus*) and Coney (*Cephalopholis fulva*), which began to decline in landings as well over time (Beets and Friedlander, 1992, 1999; Beets et al., 1994). Long-term monitoring of spawning aggregations closed to fishing have reported a significant increase in grouper abundance at spawning aggregations and local fishermen now report increased body length in species such as Red Hind (Nemeth et al., 2005).

The level of fishing effort varies greatly among locations in the USVI, with some fishers using a limited amount of gear nearshore and others setting long trap lines on the insular shelf, including within VIIS boundaries (Garrison et al., 1998). Fisheries landings within the boundaries of the national parks are not recorded separately from the remainder of the territory. Annual visual sampling of traps from 1992 to 1994 provided minimum catch estimates of >5,000 kg/yr inside VIIS (Garrison et al., 1998). Landings from around the island by local St. John fishers are as high as 78,634 kg/yr (Beets, 1996), but do not include catch by non-local fishers.

Lionfish Invasion

Indo-Pacific lionfish (*Pterois volitans*) are rapidly invading the waters of the Caribbean and tropical Atlantic. Due to their population explosion and aggressive behavior, lionfish have the potential to have a dramatic impact on Caribbean fish populations since they have virtually no natural predators and prey species are unaccustomed to and not wary of this voracious predator (Albins and Hixon, 2008). During the 2010 NOAA field mission, scientific divers monitoring the coral reef ecosystems off the coasts of St. John, USVI identified and killed a sub-adult lion fish (15 cm total length) in Fish Bay. The invasive fish was first spotted July 15, 2010 and captured the following day within 10 meters of the original sighting (Figure 1.12).

*Figure 1.12. Adult lionfish (*Pterois volitans*) photographed in Fish Bay, St. John in July 2010. This fish was subsequently speared and removed from the ecosystem by scientific divers from NOAA's National Ocean Service (NOS), National Centers for Coastal Ocean Science (NCCOS), Center for Coastal Monitoring and Assessment Biogeography Branch (CCMA-BB) at the request of the National Park Service (NPS). Credit: NOAA/NOS/NCCOS/CCMA/ Biogeography Branch.*

Anecdotal information suggests that lionfish populations have exploded in the USVI since they were first sighted in the Territory in 2008. The St. Thomas Fishermen's Association reported over 34,000 lionfish being caught in fish traps set at depths greater than 100 ft (Rafe Boulon, pers. comm.). Furthermore, William Coles of USVI DPNR also reported that thousands of lionfish have been caught in the Territory by trap fishermen (McCoy, 2011). At the time of writing this report, more than 300 lionfish have been recorded around St. John including the VICR and VIIS. In response to the perceived threat, the National Park Service produced a lionfish response plan (McCreedy et al. 2012) and NOAA and partners produced a guide to control and management (Morris, 2012).

1.3. INTRODUCTION TO THE STUDY DESIGN

1.3.1. History of sampling around St. John

Many marine ecological studies have been carried out in the Virgin Islands with long-term monitoring studies first conducted by scientists collaborating with the National Park. NPS supported reef fish research in St. John starting with the seminal fish ecology work of Dr. John Randall from 1958-1961 which included investigations on: fisheries resources (Idyll and Randall, 1959), fish movements (Randall, 1962), population structure (Randall, 1963), fish grazing (Randall, 1965), food habits (Randall, 1967), and taxonomy (Randall, 1968). The Tektite Program in 1969 and 1970 involved scientists living in a saturation diving habitat at a depth of 17 m in Lameshur Bay, St. John for weeks at a time to conduct a wide variety of research including marine biology (Collette, 1996). In 1983, the Virgin Islands Resource Management Cooperative, supported primarily by NPS and under the direction of Island Resources Foundation, produced a series of reports from 1986-1988 that provided maps and data that are the basis of many ongoing projects in VIIS. Subsequent investigations of fish resources and fisheries investigations have been conducted around St. John, ranging from fisheries assessments, reef fish monitoring, sedimentation, hurricane impacts and coral bleaching and disease, and have yielded significant insights on the impacts of stressors on coral reefs around St. John.

1.3.2. Caribbean Coral Reef Ecosystem Monitoring (CREM) in St. John

After developing detailed benthic maps for the U.S. Caribbean (Kendall et al., 2001), CCMA-BB began *in-situ* sampling around St. John in 2001 in response to NPS' need for (i) a broad-scale characterization of coral reef resources within its parks and (ii) long-term data that could be used to determine efficacy of the parks' no-take marine reserves. This ecosystem characterization effort was also initiated to develop spatially-explicit estimates of reef fish habitat utilization patterns and help define essential fish habitats for the Caribbean Fisheries Management Council (CFMC). Between 2002 and 2007, sampling provided data to support NCCOS' CSCOR CRES project with a focus on understanding how habitat utilization by fish and macro invertebrate species varied inside and outside MPAs. Since 2007 and through funding from CRCP, sampling was expanded to include the VICR (mid-shelf reef and Coral Bay) to provide data to describe broad-scale patterns in habitat use by fishes and invertebrates, connectivity among fish assemblages in various habitats, long-term trends in fish populations, and MPA efficacy in protecting reef fish assemblages.

Sampling Design

Data presented in the Benthic Characterization and Fish Chapters of this report were collected by CCMA-BB, NPS, University of Hawaii-Hilo, Oceanic Institute, USGS and other partners using a stratified random sampling design (Menza et al., 2006). In 2001, two types of data collection methods were used: (1) belt-transect fish census and (2) fine-scale benthic composition census (Appendix A). The following year the survey area was expanded to the mid-shelf reef, south of St. John. From 2003 onward, the VICR area of Coral Bay used as a control for comparisons for the mid-shelf reef to determine the efficacy of resource protection on reef fish assemblages (Figure 1.13; see Chapters 3 and 4 for more details).

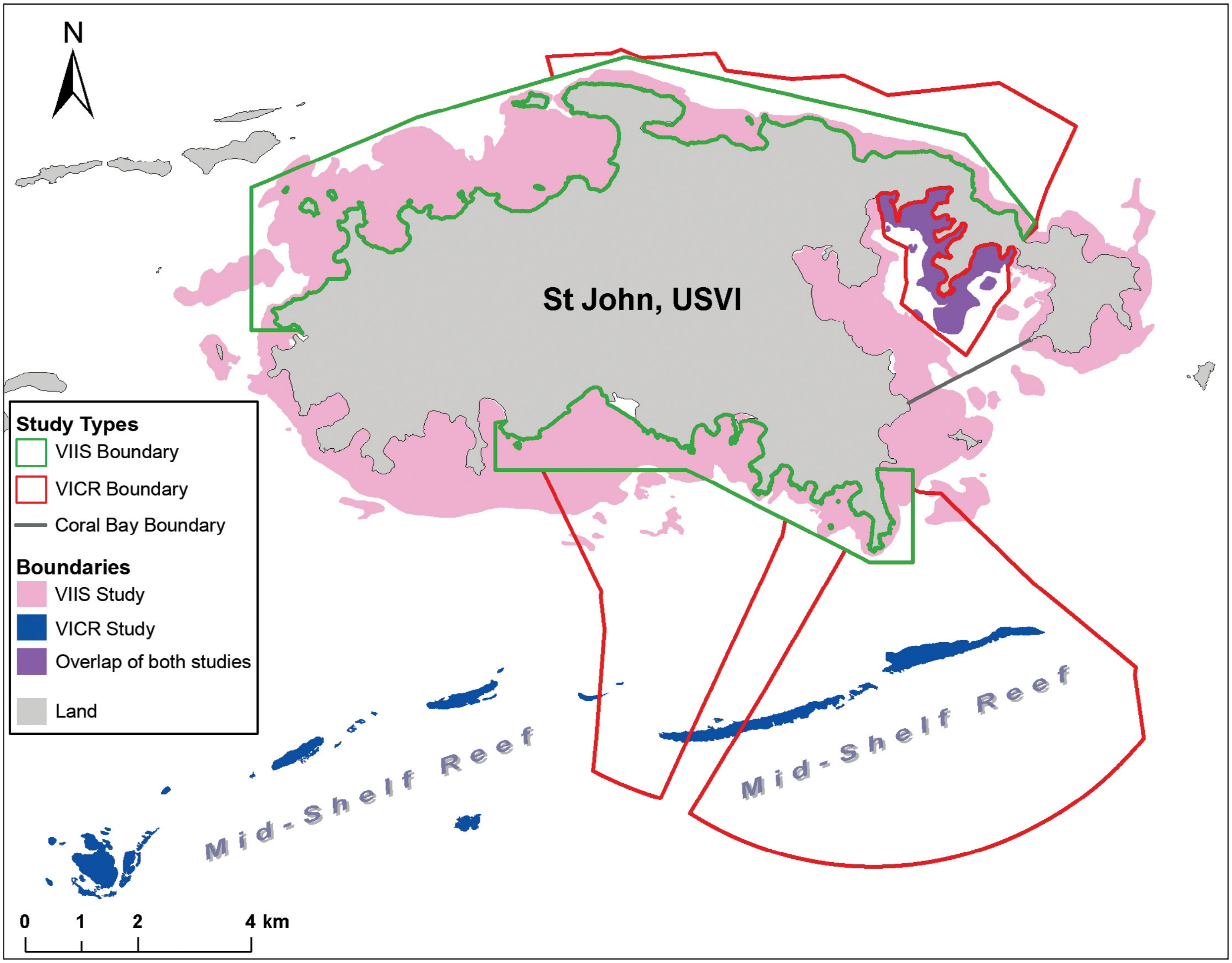


Figure 1.13. Two types of sampling studies conducted by CCMA-BB, NPS and U.S. Geological Survey (USGS) for the CCMA-BB's Caribbean Coral Reef Ecosystem Monitoring (CREM) project. Source: S.D. Hile (NOAA/NOS/NCCOS/CCMA/ Biogeography Branch).

1.4. REFERENCES

Albins, M.A. and M.A. Hixon. 2008. Invasive Indo-Pacific lionfish *Pterois volitans* reduce recruitment of Atlantic coral-reef fishes. Mar. Ecol. Prog. Ser. 367: 233-238

Beets, J. and A.M. Friedlander. 1992. Stock analysis and management strategies for red hind, *Epinephelus guttatus*, in the U.S. Virgin Islands. Proc. Gulf Carib. Fish Inst. 42:66-79.

Beets, J., A. Friedlander, and W. Tobias. 1994. Stock assessment of the coney, *Epinephelus fulvus*, in the U.S. Virgin Islands. Annu. Proc. Gulf Carib. Fish. Instit. 43:395-408.

Beets, J. 1996. The effects of fishing and fish traps on fish assemblages within Virgin Islands National Park and Buck Island Reef National Monument. U.S. National Park Service Technical Report - VINP 5/96, May 1996, St. John. 44 pp.

Beets, J. 1997. Can coral reef fish assemblages be sustained as fishing intensity increases? pp. 2009-2014. In: H.A. Lessios and I.G. Macintyre (eds.), Proceedings of the 8th International Coral Reef Symposium, Vol. 2. Panama City, Panama. 2118 pp.

Beets, J. and A. Friedlander. 1999. Evaluation of a conservation strategy: a spawning aggregation closure for red hind, *Epinephelus guttatus*, in the Virgin Islands. Environ. Biol. Fish. 55:91-98.

Bellwood, D.R., T.P. Hughes, C. Folke, and M. Nystrom. 2004. Confronting the coral reef crisis. Nature 429: 827-833.

Boulon, R.H., M.E. Monaco, A.M. Friedlander, C. Caldow, J. Christensen, C. Rogers, J. Beets, J. Miller, and S.D. Hile. 2008. An ecological correction to marine reserve boundaries in the U.S. Virgin Islands. pp. 1082-1085. In: B.M Riegl and R.E. Dodge (eds.), Proceedings of the 11th International Coral Reef Symposium, Vol. 2. Ft. Lauderdale, FL. 742 pp.

Brooks, G., R. Larson, and B. Devine. 2004. A depositional framework for St. Thomas and St. John, USVI. Virgin Islands Wetlands and Riparian Areas Inventory. Island Resource Foundation, Washington D.C., 81 pp.

Bythell, J.C., Z.M. Hillis-Starr, and C.S. Rogers. 2000. Local variability but landscape stability in coral reef communities following repeated hurricane impacts. Mar. Ecol. Prog. Ser. 204:93–100.

Clark, R., C. Jeffrey, K. Woody, Z. Hillis-Starr, and M. Monaco. 2009. Spatial and temporal patterns of coral bleaching around Buck Island Reef National Monument, St. Croix, U.S Virgin Islands. Bull. Mar. Sci. 84:167-182.

Collette, B.B. 1996. Results of the Tektite Program: Ecology of Coral-Reef Fishes. pp. 83-87. In: M.A. Lang and C.C. Baldwin (eds.), Methods and Techniques of Underwater Research. Proceedings of the American Academy of Underwater Sciences Sixteenth Annual Scientific Diving Symposium. American Academy of Underwater Sciences. Nahant, Massachusetts. 236 pp.

Connell, J.H. 1997. Disturbance and recovery of coral assemblages. Coral Reefs 16(5):101-113.

Devine, B., G. Brooks, and R. Nemeth. 2003. Coral Bay sediment deposition and reef assessment study. State of the Bay Final Project Report, submitted to the V.I. Department of Planning and Natural Resources, Division of Environmental Protection. Non-Point Source Pollution Grant Program MOA# NPS -01801. University of Virgin Islands, St. Thomas, U.S. Virgin Islands.

Done, T.J., P.K. Dayton, A.E. Dayton, and R. Steger. 1991. Regional and local variability in recovery of shallow coral communities: Moorea, French Polynesia and central Great Barrier Reef. Coral Reefs 9:183-192.

Edmunds, P.J. 1991. Extent and effect of black band disease on a Caribbean reef. Coral Reefs 10(3):161-165

Edmunds, P.J. 2004. Juvenile coral population dynamics track rising seawater temperature on a Caribbean reef. Mar. Ecol. Prog. Ser. 269:111-119.

Friedlander, A.M., and J. Beets. 2008. Temporal trends in reef fish assemblages inside Virgin Islands National Park and around St. John, U.S. Virgin islands, 1988-2006. NOAA Technical Memorandum NOS NCCOS 70. Silver Spring, MD. 50 pp. Online: http://ccma.nos.noaa.gov/publications/VITempTrends.pdf (Accessed 14 May 2013)

Garrison, V.H., C.S. Rogers, and J. Beets. 1998. Of reef fishes, overfishing and *in situ* observations of fish traps in St. John, U.S. Virgin Islands. Rev. Biol. Trop. 46 Supl. 5:41-59.

Halliwell, G. and D. Mayer. 1996. Frequency response properties of forced climate SST anomaly variability. J. Clim. 9(12):3575-3587.

Hu, C.M., E.T. Montgomery, R.W. Schmitt, and F.E. Muller-Karger. 2004. The dispersal of the Amazon and Orinoco River water in the tropical Atlantic and Caribbean Sea: Observation from space and S-PALACE floats. Deep-Sea Res. Part II. 51(10-11):1151-1171.

Hubbard, D.K., R.B. Burke, I.P. Gill, W.R. Ramirez, and C. Sherman. 2008. Coral-reef Geology: Puerto Rico and the U.S. Virgin Islands. pp. 263-301. In: B. Riegl and R.E. Dodge (eds.), Coral Reefs of the USA. Coral Reefs of the World, Volume 1. Springer. 806 pp.

Hughes, T.P. 1994. Catastrophes, phase shifts and large-scale degradation of a Caribbean coral reef. Science 301:929-933.

Idyll, C.P., and J.E. Randall. 1959. Sport and commercial fisheries potential of St. John, Virgin Islands. 4th International Gamefish Conference. 10 pp.

Jackson, J.B.C., M.X. Kirby, W.H. Berger, K.A. Bjorndal, L.W. Botsford, B.J. Bourque, R.H. Bradbury, R. Coke, J. Erlandson, J.A.Estes, T.P. Hughes, S. Kidwell, C.B. Lange, H.S. Lenihan, J.M. Pandolfi, C.H. Peterson, R.S. Steneck, M.J. Tegner, and R.R. Warner. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science 293:629-638.

Kendall, M.S., C.R. Kruer, K.R. Buja, J.D. Christensen, M. Finkbeiner, R.A. Warner, and M.E. Monaco. 2001. Methods used to map the benthic habitats of Puerto Rico and the U.S. Virgin Islands. NOAA Technical Memorandum NOS NCCOS 152. Silver Spring, MD. 45 pp. Online: http://ccma.nos.noaa.gov/products/biogeography/usvi_pr_mapping/manual.pdf (Accessed 14 May 2013)

McCoy, S. 2011. Thousands of lionfish now in the V.I. Virgin Islands Daily News. 1 February 2011. Online: http://virginislandsdailynews.com/news/thousands-of-lionfish-now-in-v-i-1.1098210 (Accessed 14 May 2013)

McCreedy, C., C.A. Toline, and V. McDonough. 2012. Lionfish response plan: A systematic approach to managing impacts from the lionfish, an invasive species, in units of the National Park System. Natural Resource Report NPS/NRSS/WRD/NRR—2012/497. National Park Service, Fort Collins, Colorado. Online: http://www.nature.nps.gov/water/marineinvasives/assets/documents/Lionfish_Response_Plan_%20final_small.pdf (Accessed 14 May 2013)

Menza, C., J. Ault, J. Beets, C. Bohnsack, C. Caldow, J. Christensen, A. Friedlander, C. Jeffrey, M. Kendall, J. Luo, M.E. Monaco, S. Smith, and K. Woody. 2006. A guide to monitoring reef fish in the National Park Service's South Florida/Caribbean Network. NOAA Technical Memorandum NOS NCCOS 39. Silver Spring, MD. 166 pp. Online: http://www.ccma.nos.noaa.gov/publications/Reefmonitoringguide.pdf (Accessed 14 May 2014)

Menza, C., M. Kendall, and S.D. Hile. 2008. The deeper we go the less we know. Rev. Biol. Trop. 56:11-24.

Miller, J., C. Rogers, and R. Waara. 2003. Monitoring the coral disease, plague type II, on coral reefs in St. John, U.S., Virgin Islands. Rev. Biol. Trop. 51:47-55.

Miller J., R. Waara, E. Muller, and C. Rogers. 2006. Coral bleaching and disease combine to cause extensive mortality on reefs in U.S. Virgin Islands. Coral Reefs 25:418.

Miller J., E. Muller, C.S. Rogers, R. Waara, A. Atkinson, K.R.T. Whelan, M. Patterson, and B. Witcher. 2009. Coral disease following massive bleaching in 2005 causes 60% decline in coral cover on reefs in the U.S. Virgin Islands. Coral Reefs 28:925-937.

Monaco, M.E., A.M. Friedlander, C. Caldow, J.D. Christensen, J. Beets, J. Miller, C. Rogers, and R. Boulon. 2007. Characterizing reef fish populations and habitats within and outside the U.S. Virgin Islands Coral Reef National Monument: a lesson in MPA design. Fish. Manage. Ecol. 14:33-40.

Monaco M.E., J. Waddell, A. Clarke, C. Caldow, C.F.G. Jeffrey, and S.J. Pittman. 2008. Status of the Coral Reef Ecosystems in the U.S. Caribbean, and Gulf of Mexico: Florida, Flower Garden Banks, Puerto Rico and Navassa. pp. 225-238. In: C.R. Wilkinson (ed.). Status of Coral Reefs of the World. Global Coral Reef Monitoring Network. 269 pp.

Morris, J.A., Jr. (Ed.). 2012. Invasive Lionfish: A Guide to Control and Management. Gulf and Caribbean Fisheries Institute Special Publication Series Number 1, Marathon, Florida, USA. 113 pp. Online: http://lionfish.gcfi.org/manual/ (Accessed 14 May 2012)

Mumby, P.J., C.P. Dahlgren, A.R. Harborne, C.V. Kappel, F. Micheli, D.R. Brumbaugh, K.E. Holmes, J.M. Mendes, K. Broad, J.N. Sanchirico, K. Buch, S. Box, R.W. Stoffle, and A.B. Gill. 2006. Fishing, trophic cascades, and the process of grazing on coral reefs. Science 311:98-101.

Nemeth, R.S. 2005. Population characteristics of a recovering U.S. Virgin Islands red hind spawning aggregation following protection. Mar. Ecol. Prog. Ser. 286:81-97.

National Oceanic and Atmospheric Administration (NOAA) - Southeast Regional Climate Office. 2007. Historical Climate Summaries for Puerto Rico and the U.S. Virgin Islands. The South East Regional Climate Center. The University of North Carolina, Chapel Hill, NC. Online: http://www.sercc.com/climateinfo/historical/historical_pr.html (Accessed 14 May 2014)

Newman, M.J.H., G.A. Paredes, E. Sala, and J.B.C. Jackson. 2006. Structure of Caribbean coral reef communities across a large gradient of fish biomass. Ecol. Lett. 9(11):1216-1227.

Olsen, D.A. and J.A. LaPlace. 1979. A study of a Virgin Islands grouper fishery based on a breeding aggregation. Annu. Proc. Gulf Carib. Fish. Inst. 31:130-144.

Pandolfi, J.M., J.B.C. Jackson, N. Baron, R.H. Bradbury, H.M. Guzman, T.P. Hughes, C.V. Kappel, F. Micheli, J.C. Ogden, H.P. Possingham, and E. Sala. 2005. Are U.S. coral reefs on the slippery slope to slime? Science 307:1725-1726.

Pittman, S.J., J. Christensen, C. Caldow, C. Menza, and M. Monaco. 2007. Predictive mapping of fish species richness across shallow-water seascapes of the U.S. Caribbean. Ecol. Model. 204:9-21

Pittman, S.J., S.D. Hile, C.F.G. Jeffrey, C. Caldow, M.S. Kendall, M.E. Monaco, and Z. Hillis-Starr. 2008. Fish assemblages and benthic habitats of Buck Island Reef National Monument (St. Croix, US Virgin Islands) and the surrounding seascape: A characterization of spatial and temporal patterns. NOAA Technical Memorandum 71. Silver Spring, MD. 96 pp. Online: http://ccma.nos.noaa.gov/publications/biogeography/STC_LTR_Final_0309.pdf (Accessed 14 May 2014)

Pittman, S.J. and B. Legare. 2010. Preliminary results from the Caribbean Acoustic Tracking Network (CATn): A data sharing partnership for acoustic tracking and movement ecology of marine animals in the Caribbean Sea. NOAA National Ocean Service, National Centers for Coastal Ocean Science, Center for Coastal Monitoring and Assessment, Biogeography Branch and the University of the Virgin Islands. Unpublished Report. 7 pp.

Ramos-Scharrón, C.E. and L.H. MacDonald. 2005. Measurement and prediction of sediment production from unpaved roads, St. John, U.S. Virgin Islands. Earth Surf. Process. Landf. 30(10):1283-1304.

Randall, J.E. 1962. Tagging reef fishes in the Virgin Islands. Annu. Proc. Gulf Carib. Fish. Inst. 14: 201-241.

Randall, J.E. 1963. An analysis of the fish populations of artificial and natural reefs in the Virgin Islands. Carib. J. Sci. 3:31-47.

Randall, J.E. 1965. Grazing effect on sea grasses by herbivorous reef fishes in the West Indies. Ecology 46:255-260.

Randall J.E. 1967. Food habits of reef fishes of the West Indies. Stud. Trap. Oceanogr. 5:665-847.

Randall, J.E. 1968. Caribbean Reef Fishes. T.F.H. Publications, Hong Kong. 319 pp.

Rankin, D.W. 2002. Geology of St. John, U.S. Virgin Islands. US Geological Survey Professional Paper 1631.

Rogers, C.S., L.N. McClain, and C.R. Tobias. 1991. Effects of Hurricane Hugo (1989) on a coral reef in St. John, USVI. Mar. Ecol. Prog. Ser. 78:189-199.

Rogers, C., V. Garrison, and R. Grober-Dunsmore. 1997. A fishy story about hurricanes and herbivory: seven years of research on a reef in St. John, US Virgin Islands. pp. 555–560. In: H.A. Lessios and I.G. Macintyre (eds.), Proceedings of the 8th International Coral Reef Symposium, Vol. 1. 1040 pp.

Rogers, C.S. and J. Beets. 2001. Degradation of marine ecosystems and decline of fishery resources in marine protected areas in the US Virgin Islands. Environ. Conserv. 28:312–322.

Rogers, C.S. and G. Garrison. 2001. Ten years after the crime: lasting effects of damage from a cruise ship anchor on a coral reef in St. John. Bull. Mar. Sci. 69:793-803.

Rogers, C.S. and J. Miller. 2001. Coral bleaching, hurricane damage, and benthic cover on coral reefs in St. John, US Virgin Islands: a comparison of results from the chain transect method and videography. Bull. Mar. Sci. 69:459-470.

Rogers, C.S., J. Miller, E.M. Muller, P.J. Edmunds, R.S. Nemeth, J. Beets, A. Friedlander, T.B. Smith, R. Boulon, C.F.G. Jeffrey, C. Menza, C. Caldow, N. Idrisi, B. Kojis, M. Monaco, A. Spitzack, E. Gladfelter, J. Ogden, Z. Hillis-Starr, I. Lundgren, W.B. Schill, I. Kuffner, L.L. Richardson, B. Devine, and J. Voss. 2008. Ecology of coral reefs in the US Virgin Islands. pp. 303-373. In: B.M. Riegl and R.E. Dodge (eds.), Coral Reefs of the USA. Coral Reefs of the World, Volume 1. Springer, Berlin. 806 pp.

Rogers, C.S. 2009. High diversity and abundance of scleractinian corals growing on and near mangrove prop roots, St. John, U.S. Virgin Islands. Coral Reefs 28:909.

Rogers, C.S. 2011. The mysterious, magical mangroves of St. John, U.S. Virgin Islands. Self Published. 128 pp.

Rothenberger, P., J. Blondeau, C. Cox, S. Curtis, W.S. Fisher, V. Garrison, Z. Hillis-Starr, C.F.G. Jeffrey, E. Kadison, I. Lundgren, W.J. Miller, E. Muller, R. Nemeth, S. Paterson, C. Rogers, T. Smith, A. Spitzack, M. Taylor, W. Toller, J. Wright, D. Wusinich-Mendez, and J. Waddell. 2008. The State of Coral Reef Ecosystems of the U.S. Virgin Islands. pp. 29-73. In: J.E. Waddell and A.M. Clarke (eds.), The state of Coral Reef Ecosystems of the United States and Pacific Freely Associated States: 2008. NOAA Technical Memorandum NOS NCCOS 73. Silver Spring, MD. 569 pp. Online: http://ccma.nos.noaa.gov/ecosystems/coralreef/coral2008/ (Accessed 14 May 2014)

U.S. Presidential Proclamation. 2001. Establishment of the Virgin Islands Coral Reef National Monument. Federal Register 66, no.14 (22 January 2001)

Wilkinson, C.R. (ed.). 2004. Status of coral reefs of the world: 2004. Volume 2. Australian Institute of Marine Science, Townsville, Australia. 380 pp.

Woody K., A. Atkinson, R. Clark, C. Jeffrey, I. Lundgren, J. Miller, M. Monaco, E. Muller, M. Patterson, C. Rogers, T. Smith, T. Spitzak, R. Waara, K. Whelan, B. Witcher and A. Wright. 2008. Coral bleaching in the US Virgin Islands in 2005 and 2006. pp. 68-72. In: C. Wilkinson and D. Souter (eds.), Status of Caribbean coral reefs after bleaching and hurricanes in 2005. Global Coral Reef Monitoring Network, Townsville, Australia. 148 pp.

Chapter 2: Benthic Habitat Mapping Around St. John

Simon J. Pittman[1,2,3], Bryan Costa[1,2] and Christopher F.G. Jeffrey[1,2]

2.1. HISTORICAL BENTHIC MAPPING

A diversity of habitat types are found around St. John, and these have been classified, described, and mapped (Kumpf and Randall, 1961; Beets et al., 1986; Kendall et al., 2001). The first map of nearshore marine ecosystems around St. John, U.S. Virgin Islands (USVI) was produced by the University of Miami in 1958 using towed diver surveys in combination with aerial photos (Figure 2.1; Kumpf and Randall, 1961). The objective of these surveys was to characterize the marine habitat in and around the recently designated Virgin Islands National Park (VIIS).

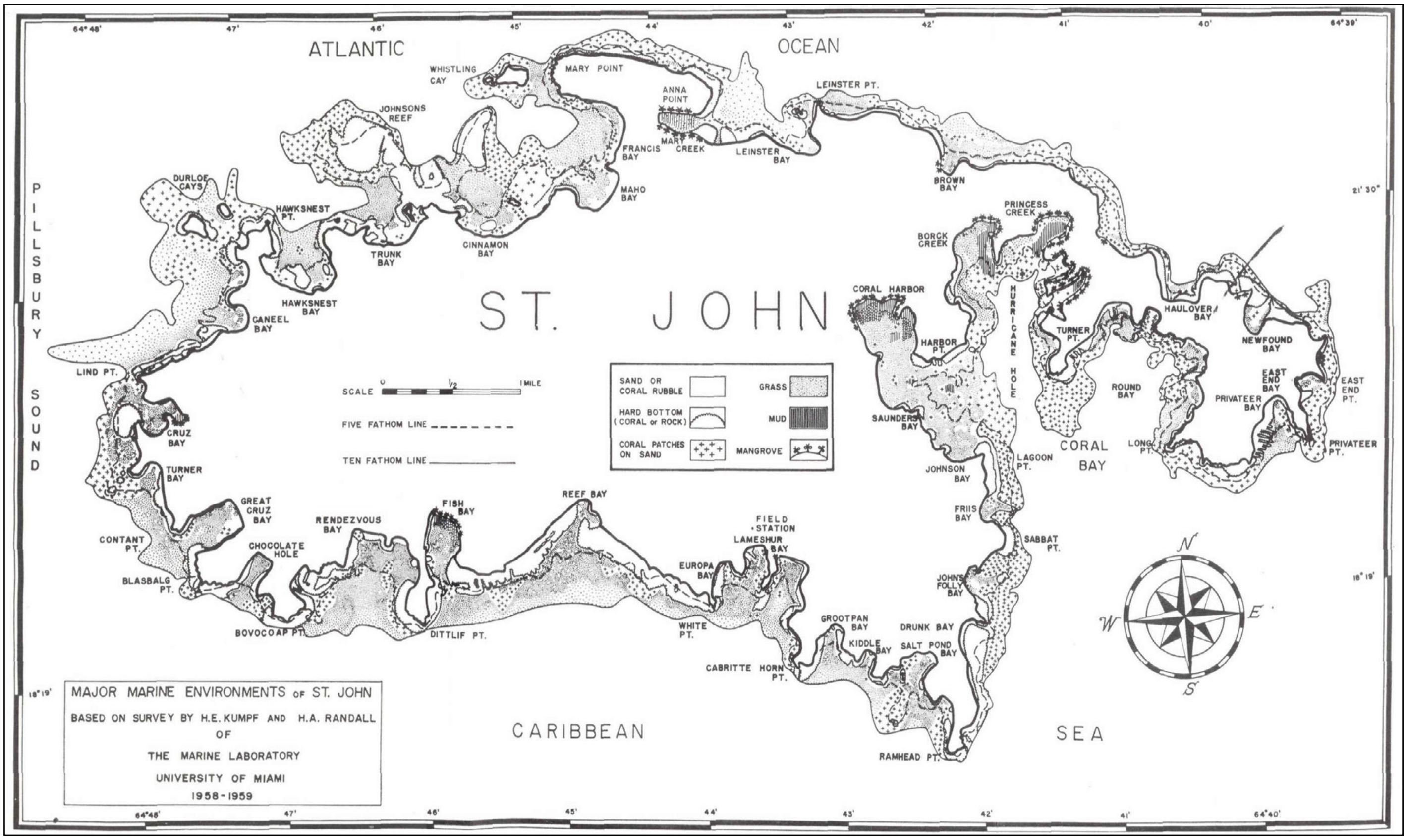


Figure 2.1. The first marine benthic habitat map of St. John, U.S. Virgin Islands (USVI). Source: Kumpf and Randall (1961).

In 1986 the National Park Service (NPS) mapped benthic habitats around St. John with the goal of preparing ecological community maps to support science and management efforts in the National Park (Beets et al., 1986; Boulon, 1986a; Figure 2.2). These habitat maps were created utilizing information from existing maps created by the University of Miami, aerial photographs and *in-situ* field work. Aerial photographs from 1983 (scale of 1: 5,300) were utilized to inform draft maps to a depth of approximately 20 m before field ground-truthing efforts took place. The methods for delineating the benthic habitats from aerial imagery involved overlaying drafting acetate over the photographs on a light table, with all benthic features distinguished with ink on acetate.

[1] Center for Coastal Monitoring and Assessment, National Centers for Coastal Ocean Science, National Ocean Service, National Oceanic and Atmospheric Administration
[2] Consolidated Safety Services, Inc.
[3] University of the Virgin Islands

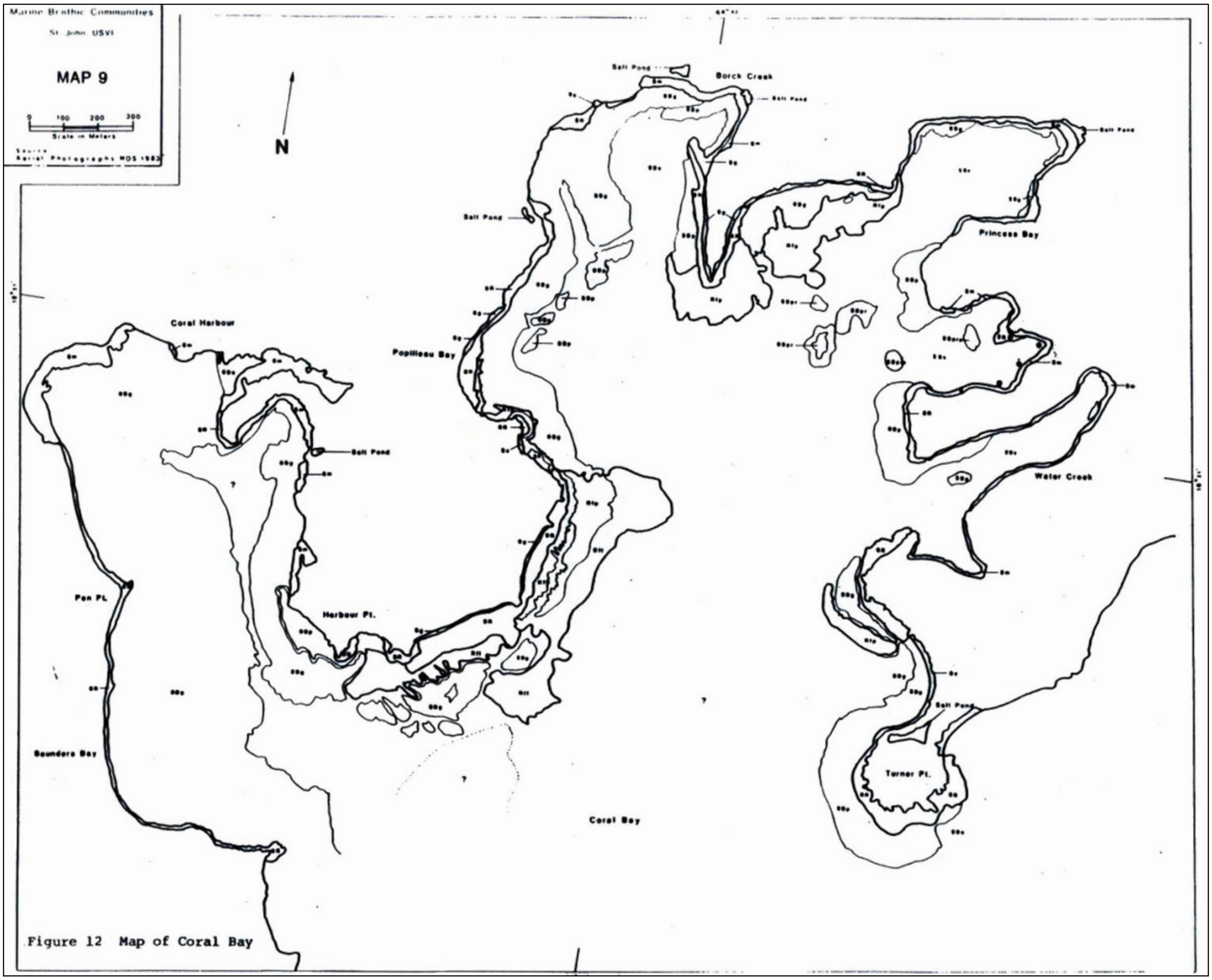


Figure 2.2. Marine benthic habitat map of Coral Bay, St. John from the Virgin Islands Resource Management Cooperative (VIRMAC) program. Source: Beets et al. (1986).

2.2. RECENT BENTHIC HABITAT MAPPING

In 1999, NOAA's National Ocean Service (NOS) acquired aerial photographs in order to create benthic habitat maps in response to a need to identify Essential Fish Habitat (EFH) in the U.S. Caribbean and better understand the distribution of coral reefs and associated habitat types. NOAA's NOS National Centers for Coastal Ocean Science (NCCOS), Center for Coastal Monitoring and Assessment Biogeography Branch (CCMA-BB) digitized benthic habitat for a 490 km^2 area of nearshore coral reef ecosystems in the USVI using a 1 acre (approximately 4,047 m^2) minimum mapping unit (MMU, Figure 2.3). Thematic accuracy around the test area of VIIS boundaries were assessed using 120 stratified-random benthic surveys resulting in an overall map accuracy of 93.6%, with 100% users accuracy for submerged vegetation, 97.2% for hard bottom habitat types and 86.1% for sand (Kendall et al., 2001).

A decade later in 2009, CCMA-BB released a revised finer scale nearshore benthic habitat map for St. John, as well as, a habitat map of deeper offshore benthic habitats using a new semi-autonomous classification technique applied to seafloor bathymetry (Figure 2.4). Combined, this effort mapped approximately 93% of the Virgin Islands Coral Reef National Monument (VICR) and 92% of the Virgin Islands National Park (VIIS) for NPS's Inventory and Monitoring Program (I&M). A total of 53.4 km^2 of shallow-water habitats around St. John and 90.2 km^2 of moderate-depth habitats south of St. John

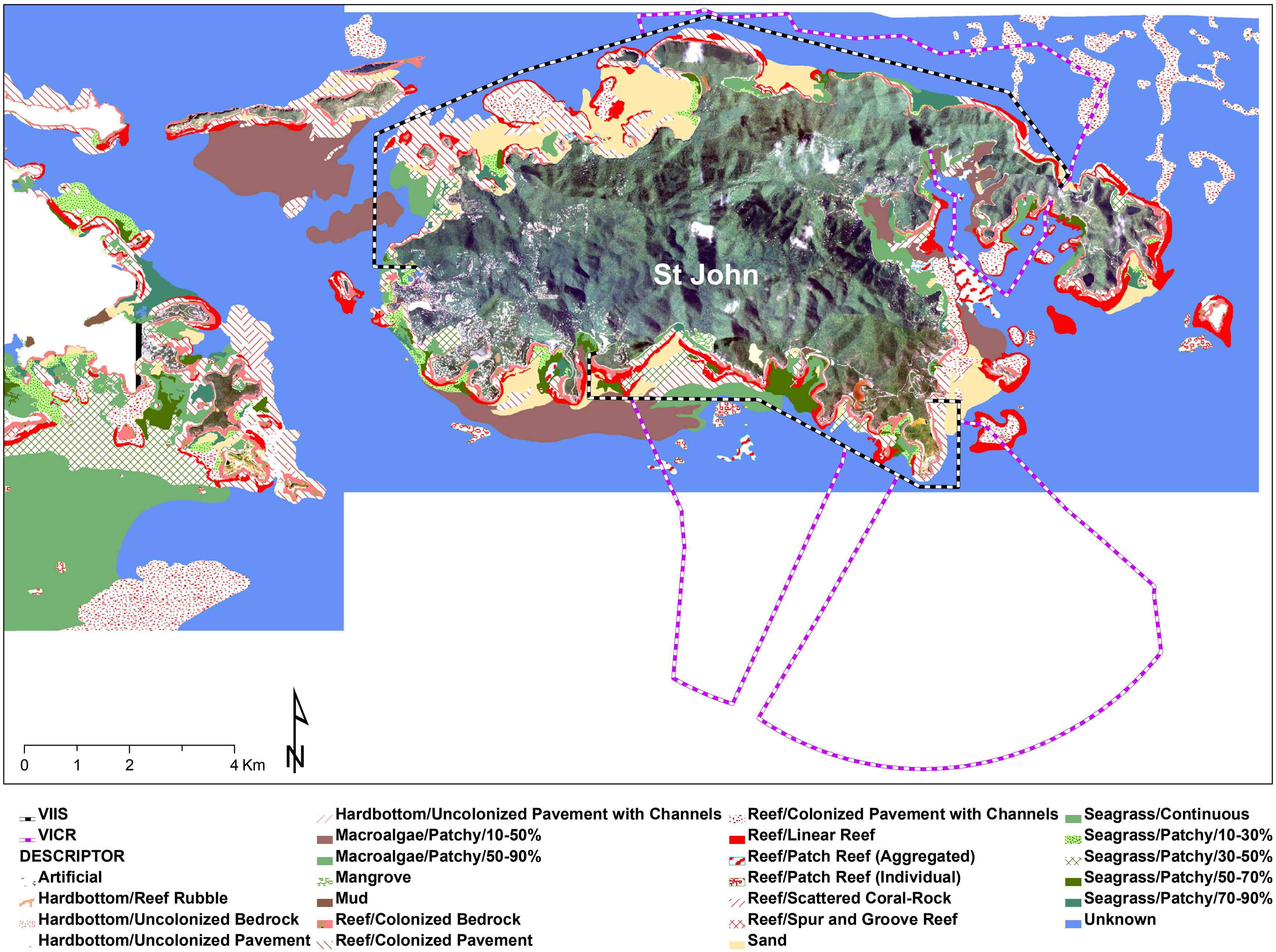


Figure 2.3. NOAA's National Ocean Service (NOS), National Centers for Coastal Ocean Science (NCCOS), Center for Coastal Monitoring and Assessment Biogeography Branch (CCMA-BB) 2001 marine benthic habitat map of St. John, USVI. Purple boundary= Virgin Islands Coral Reef National Monument (VICR), black boundary= Virgin Islands National Park (VIIS). Source: C.F.G. Jeffrey (NOAA/NOS/NCCOS/CCMA/Biogeography Branch).

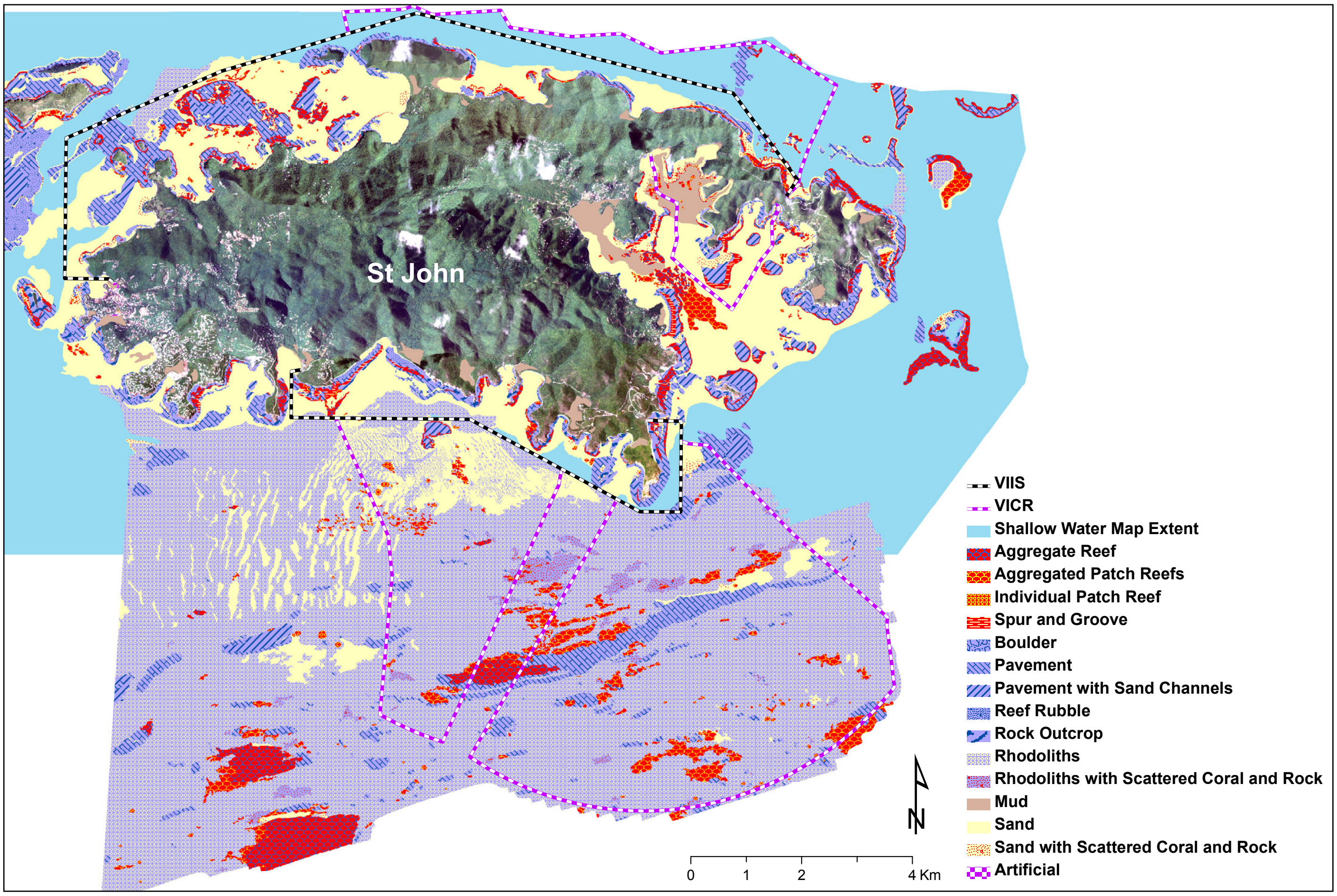


Figure 2.4. CCMA-BB 2009 fine-scale shallow water and moderate depth benthic habitat maps for St. John, USVI. Source: C.F.G. Jeffrey (NOAA/NOS/NCCOS/CCMA/Biogeography Branch).

were mapped (Figure 2.4). Thirty-two distinct benthic habitat types (i.e., three major and 16 detailed geomorphological structure classes; six major and three detailed biological cover types; and four live coral cover classes) within 12 zones were digitally mapped using a combination of heads-up visual interpretation of aerial imagery and semi-automated classification of acoustic imagery. Data available online: http://ccma.nos.noaa.gov/ecosystems/coralreef/benthic/ or via the custom designed map server available online at: http://ccma.nos.noaa.gov/explorer/biomapper/biomapper.html?id=StJohn (Accessed 14 May 2013).

2.2.1. LiDAR shallow-water bathymetric mapping

In 2011, high resolution seafloor bathymetry was mapped for nearshore areas of St. John (Figure 2.5 a, b). This product was primarily utilized to update navigational charts, but also provides detailed spatial information on the distribution of seafloor structure including three-dimensional surface complexity. The bathymetry can be used to create derivative products that will help predict the distribution and diversity patterns of marine organisms and better understand the coral reef ecosystems surrounding St. John and neighboring areas (Pittman et al., 2009; Pittman and Brown, 2011). For example, the structural complexity of northwest St. John appears very important for reef fishes and is an area with relatively high fish diversity (Figure 2.5b). Given that structural complexity is a main driver of spatial patterns in reef fish assemblages, understanding and identifying such patterns will provide a cost-effective and informative mechanism to better manage extractive effort on reef fish and macroinvertebrate populations thereby ensuring their long-term sustainability.

2.2.2. Deep-water bathymetric mapping

A number of recent seafloor mapping missions have been conducted by CCMA-BB in and around

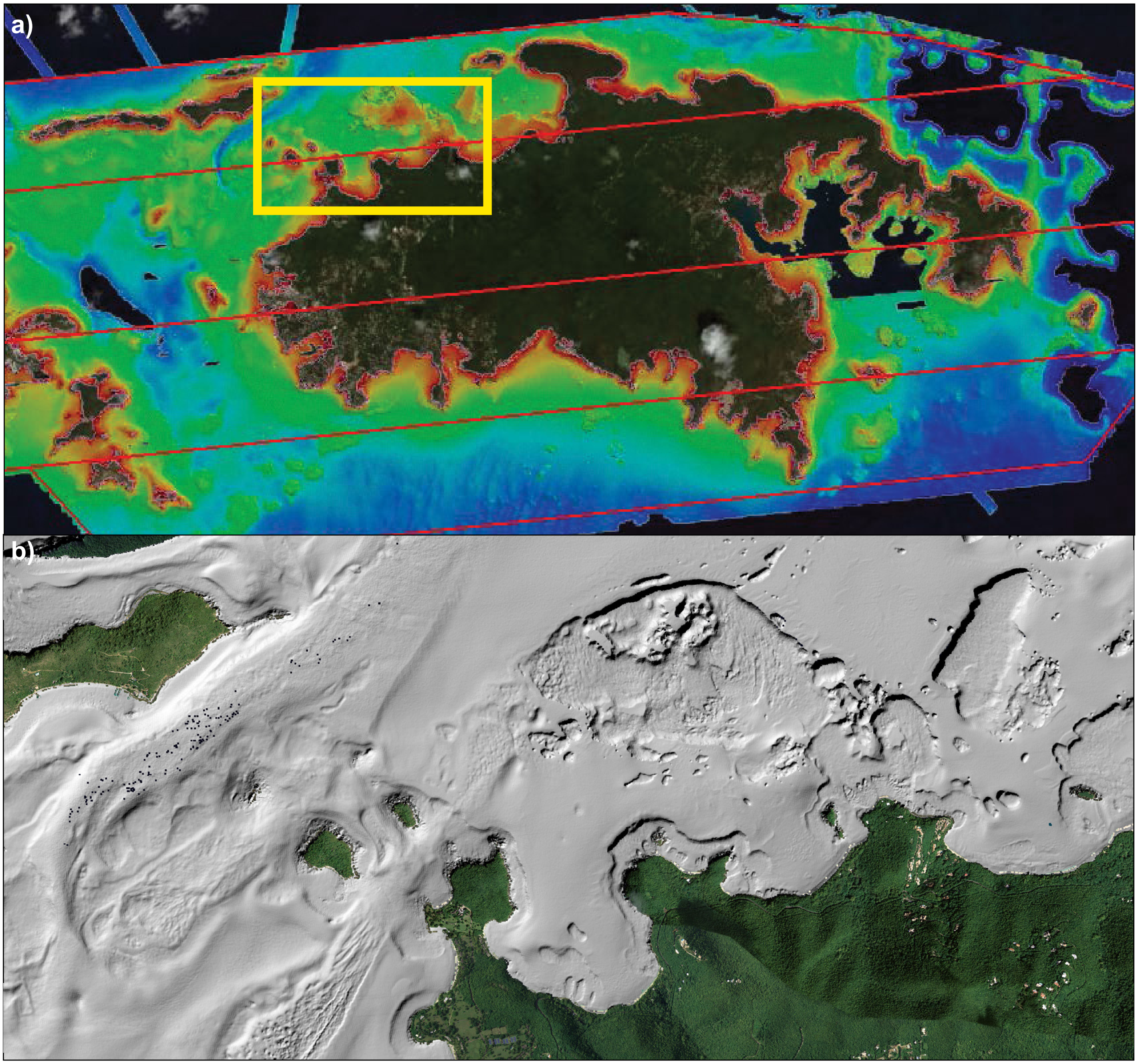


Figure 2.5. Maps showing structural complexity of the seafloor in St. John, USVI for the entire island (top) and East End St. Thomas and northwest St. John including Hawksnest, Trunk and Cinnamon Bays (bottom). High resolution bathymetry was acquired by Fugro LADS for NOAA in February 2011 using airborne hydrographic LiDAR (light detection and ranging) technology. Source: S.J. Pittman (NOAA/NOS/NCCOS/CCMA/Biogeography Branch).

VIIS and the USVI. The primary objective of the seafloor mapping project is to integrate abiotic data collected from acoustic sonar systems with biotic information obtained by underwater imagery systems (Remotely and Autonomously Operated Vehicles and Drop/drift camera systems) and SCUBA divers to create accurate benthic habitat maps. Other project objectives were to develop data acquisition standards, signal processing techniques, and mapping and sampling design protocols for acoustic data collection; as well as to evaluate the utility of these new technologies. The overall mission's intent was to develop a more complete understanding of the marine resources within the surveyed areas, information that will ultimately contribute to the development of detailed species utilization models linking physical habitats and biological information.

In 2004, 2005 and 2006 CCMA-BB and NOAA's Office of Coast Survey, in collaboration with NPS, USVI Territory and private sector partners, used multibeam sonar and underwater video to map

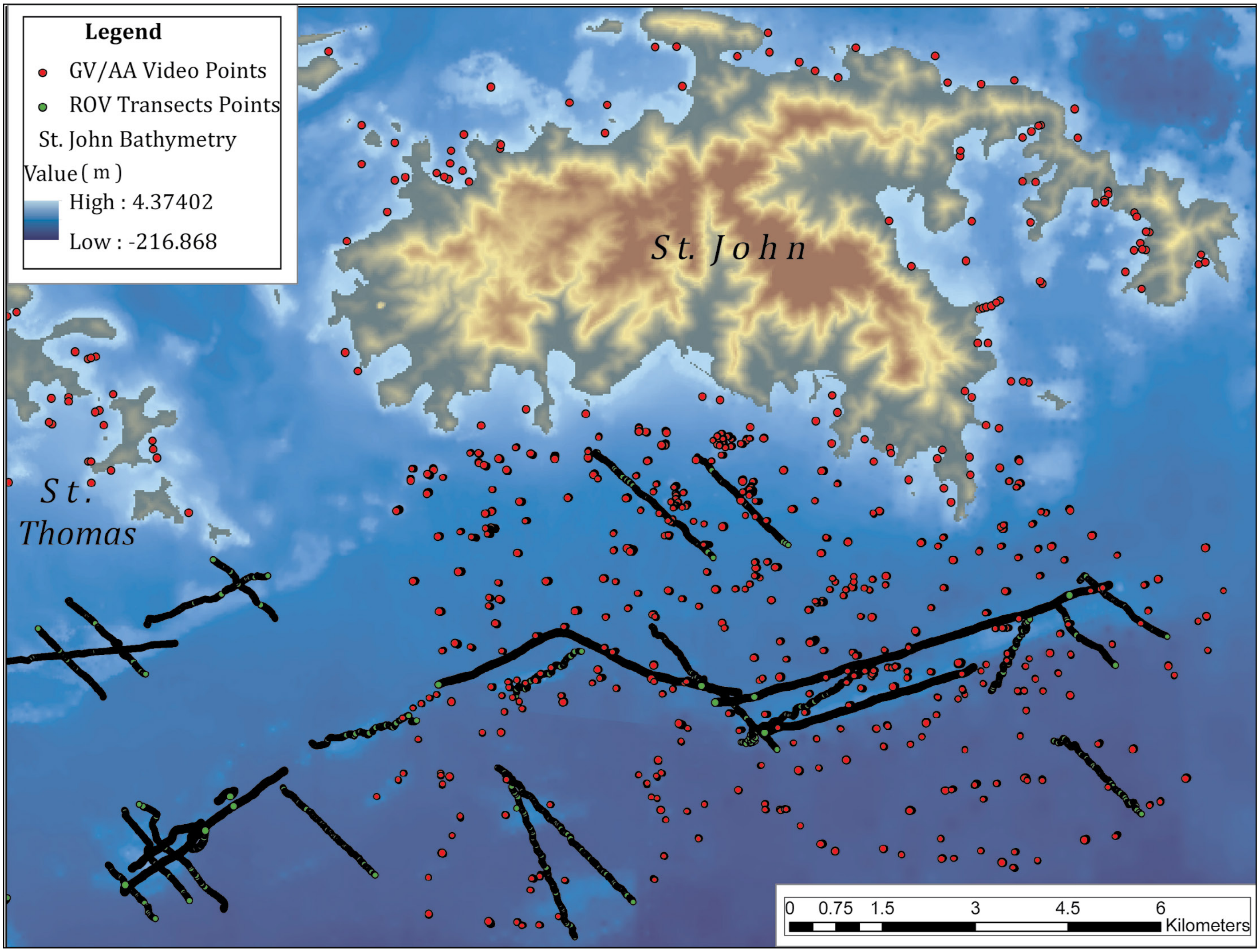


Figure 2.6. Bathymetric data from NOAA's acoustic multibeam seafloor mapping activities within and around VICR and VIIS. Source: W. Sautter (NOAA/NOS/NCCOS/CCMA/Biogeography Branch).

bottom features (>20 m depth) and characterize nearshore benthic structure around VIIS (Figure 2.6). These data are a component of the Seafloor Characterization of the Caribbean project supported by NOAA's Coral Reef Conservation Program (CRCP) and are available online at: http://ccma.nos.noaa.gov/ecosystems/coralreef/usvi_nps.aspx (Accessed 14 May 2013). Data products include 5-m raster grids, digital terrain models and mosaics of the acoustic backscatter.

2.3. APPLICATIONS FOR BENTHIC HABITAT MAPS

Sampling and monitoring strategies for reef fishes have been developing over the past 20 years for VIIS. The primary methods to sample reef fish populations have been visual census of fishes along transects or stationary point counts. Initial efforts to document reef fish conditions were conducted within selected habitats based on maps developed in the 1980s (Beets et al., 1986; Boulon, 1986a, 1986b, 1987). In 1989, annual monitoring of reef fish populations and coral assemblages were initiated at 18 permanent reference reef sites around St. John; and this monitoring is ongoing at present (Friedlander and Beets, 2008). Beginning in 2001, a comprehensive spatial island-wide assessment of reef fish and benthic habitats was implemented with the development of new GIS-based habitat maps by the NOAA-NOS Biogeography Program (later became CCMA-BB) in 2001 (Kendall et al., 2001; Monaco et al., 2001).

Understanding the ecological relationships among coastal living resources and the dynamic attributes of their habitats is critical to the successful management and conservation of coral reefs.

CCMA-BB approaches the assessment of management and conservation strategies such as marine protected areas (MPAs), boundary delineation, and defining species habitat utilization patterns via three integrated activities: 1) map the distribution and characteristics (quality) of benthic habitats; 2) inventory and map the distribution of macro-invertebrates and fishes; and 3) define species habitat relationships in space and time. These three components are integrated using a suite of analytical techniques and GIS tools to quantitatively define species habitat utilization patterns within and outside MPAs. This approach results in hypothesis-driven studies that address many aspects of evaluating MPA delineation, use, function, and effectiveness in protecting marine resources.

CCMA-BB has developed digital benthic habitat maps for the U.S. Caribbean, Florida and the U.S. Pacific Islands to support the National Coral Reef Ecosystem Monitoring Program, as directed by the U.S. Coral Reef Task Force (Monaco et al., 2012). These key maps products are being used widely to design many ecological studies that assess marine animal populations (Friedlander et al., 2007), species richness and diversity (Pittman et al., 2007), effects of pollutants on reefs (Pait et al., 2009), overall coral reef ecosystem condition (Whitall et al., 2011), and the efficacy of reef restoration efforts (Zitello et al., 2008; Whitall et al. 2011). CCMA-BB's integrated mapping and monitoring approach for assessing coral reef ecosystems and reef fish habitat utilization patterns are designed to aid resource managers in making informed decisions about conserving living marine resources. For example, this integrated mapping and monitoring approach is being used to support the designation of essential fish habitat (EFH) areas, delineation and modification of MPA boundaries, and the evaluation of the effectiveness of MPAs. Furthermore, the benthic habitat maps have been combined with surface rugosity information and bathymetry to predict the spatial patterns of fish richness around St. John yielding an overall map accuracy of 70.5 % and 86.5 % for the high (15-25) fish species richness class (Pittman et al., 2007; Figure 2.7). Information from these recently developed fine-scaled benthic habitat maps (2009) and newly available Light Detection and Ranging mapping technology (LiDAR; 2011/2012) will help improve analyses of current and future natural resources datasets, which ultimately would lead to even more robust predictive models and evaluations of management-induced changes in reef ecosystems.

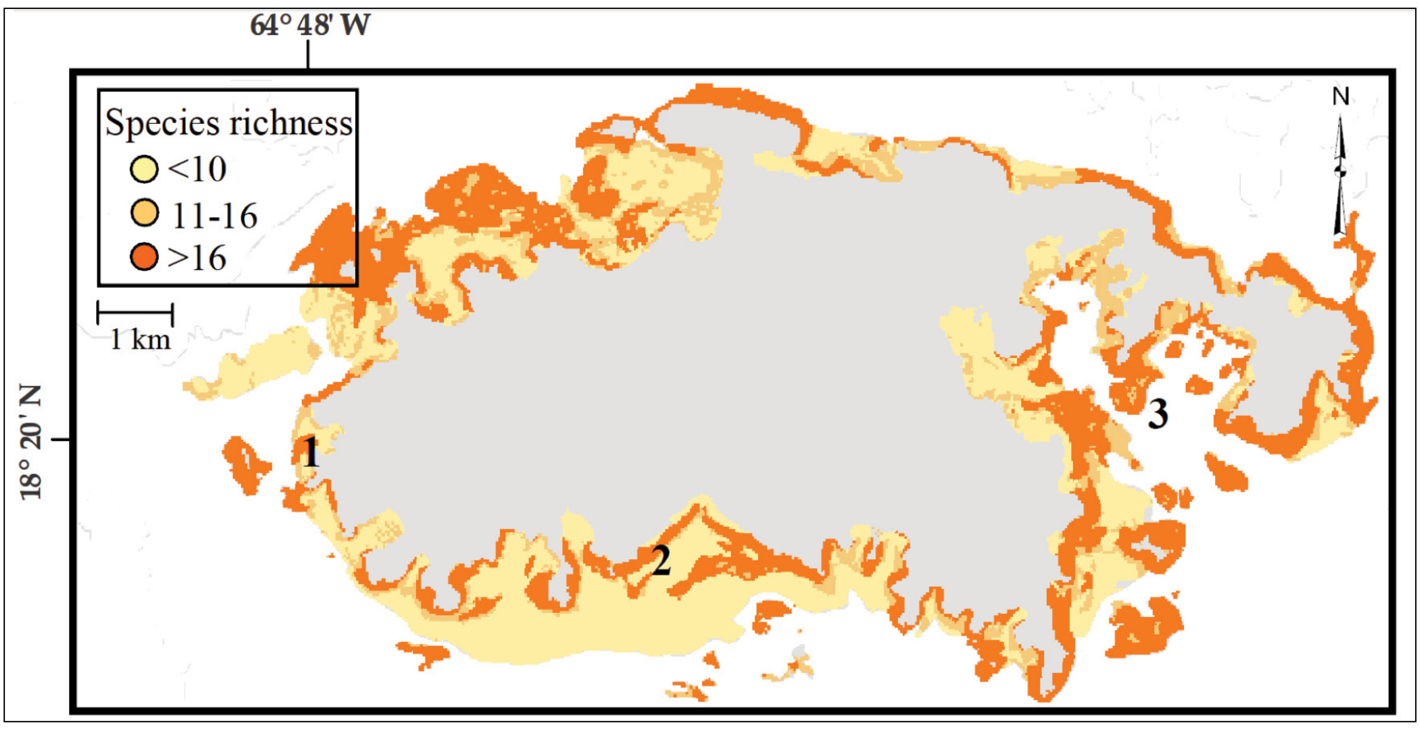


Figure 2.7. Predictive map of fish species richness (high, medium, low) around St. John based on a regression tree model of the relationship between the number of fish at 423 stratified random survey sites and the surrounding benthic structure (benthic habitat and associated rugosity combined with bathymetric variability): 1) Cruz Bay, 2) Reef Bay, and 3) Coral Bay are marked. Source: Pittman et al. (2007).

2.4. REFERENCES

Beets, J., L. Lewand, and E. Zullo. 1986. Marine community descriptions and maps of bays within the Virgin Islands National Park/Biosphere Reserve. Virgin Islands Resource Management Cooperative. Biosphere Reserve Research Reports 2. Virgin Islands National Park, St. John, U.S. Virgin Islands. 118 pp.

Boulon, R. 1986a. Map of fishery habitats within the Virgin Islands Biosphere Reserve. Virgin Islands Resource Management Cooperative. Biosphere Reserve Research Reports 8. Virgin Islands National Park, St. John, U.S. Virgin Islands. 70 pp.

Boulon, R. 1986b. Fisheries habitat of the Virgin Islands Region of Ecological Importance to the fishery resources of the Virgin Islands Biosphere Reserve. Virgin Islands Resource Management Cooperative. Biosphere Reserve Research Reports 9. Virgin Islands National Park, St. John, U.S. Virgin Islands. 22 pp.

Boulon, R. 1987. A basis for long-term monitoring of fish and shellfish species in the Virgin Islands National Park. Virgin Islands Resource Management Cooperative. Biosphere Reserve Research Reports 22. Virgin Islands National Park, St. John, U.S. Virgin Islands. 66 pp.

Friedlander, A.M. and J. Beets. 2008. Temporal trends in reef fish assemblages inside Virgin Islands National Park and around St. John, U.S. Virgin islands, 1988-2006. NOAA Technical Memorandum NOS NCCOS 70. Silver Spring, MD. 50 pp. Online: http://ccma.nos.noaa.gov/publications/VITempTrends.pdf (Accessed 14 May 2013)

Friedlander, A.M., E.K. Brown, and M.E. Monaco. 2007. Coupling ecology and GIS to evaluate efficacy of marine protected areas in Hawaii. Ecol. Appl. 17:715-730.

Kendall, M.S., C.R. Kruer, K.R. Buja, J.D. Christensen, M. Finkbeiner, R.A. Warner, and M.E. Monaco. 2001. Methods used to map the benthic habitats of Puerto Rico and the U.S. Virgin Islands. NOAA Technical Memorandum NOS NCCOS 152. Silver Spring, MD. 45 pp. Online: http://ccma.nos.noaa.gov/products/biogeography/usvi_pr_mapping/manual.pdf (Accessed 14 May 2013)

Kumpf, H.E. and H.A. Randall. 1961. Charting the marine environments of St. John, U.S. Virgin Islands. Bull. Mar. Sci. 11(4):543-551.

Monaco, M.E., J.D. Christensen, and S.O. Rohmann. 2001. Mapping and Monitoring of U.S. Coral Reef Ecosystems: The Coupling of Ecology, Remote Sensing and GIS Technology. Earth System Monitor 12(1):1-2, 4-7, 16.

Monaco, M.E., S.M. Anderson, T.A. Battista, M.S. Kendall, S.O. ohmann, L.M. Wedding and A.M. Clarke. 2012. National Summary of NOAA's Shallow-water Benthic Habitat Mapping of U.S. Coral Reef Ecosystems. NOAA Technical Memorandum NOS NCCOS 122. Perpared by the NCCOS Center for Coastal Monitoring and Assessment Biogeography Branch. Silver Spring, MD. 83 pp.

Pait, A.S., C.F.G. Jeffrey, C. Caldow, D.R. Whitall, S.I. Hartwell, A.L. Mason, and J.D. Christensen. 2009. Chemical contamination in southwest Puerto Rico: A survey of contaminants in the coral *Porites astreoides*. Caribb. J. Sci. 45(2-3):191-203.

Pittman, S.J., J. Christensen, C. Caldow, C. Menza, and M. Monaco. 2007. Predictive mapping of fish species richness across shallow-water seascapes of the U.S. Caribbean. Ecol. Model. 204:9-21.

Pittman, S.J., B. Costa, and T. Battista. 2009. Using Lidar bathymetry and boosted regression trees to predict the diversity and abundance of fish and corals. J. Coast. Res. Spec. Issue 53:27-38.

Pittman, S.J. and K.A. Brown. 2011. Multi-Scale Approach for Predicting Fish Species Distributions across Coral Reef Seascapes. PLoS ONE 6(5): e20583. Online: http://www.plosone.org/article/info%3Adoi%2F10.1371%2Fjournal.pone.0020583 (Accessed 14 May 2013).

Whitall, D.R., B.M. Costa, L.J. Bauer, A. Dieppa, and S.D. Hile (eds.). 2011. A Baseline Assessment of the Ecological Resources of Jobos Bay, Puerto Rico. NOAA Technical Memorandum NOS NCCOS 133. Silver Spring, MD. 188 pp.

Zitello, A.G., D.R. Whitall, A. Dieppa, J.D. Christensen, M.E. Monaco and S.O. Rohmann. 2008. Characterizing Jobos Bay, Puerto Rico: A Watershed Modeling Analysis and monitoring Plan. NOAA Technical Memorandum NOS NCCOS 76. 81 pp.

Chapter 3: Composition of Benthic Communities Around St. John

Alan Friedlander[1], Christopher F. G. Jeffrey[2,3], Jeff Miller[4], Kimberly W. Roberson[2] and Caroline Rogers[1]

3.1. INTRODUCTION

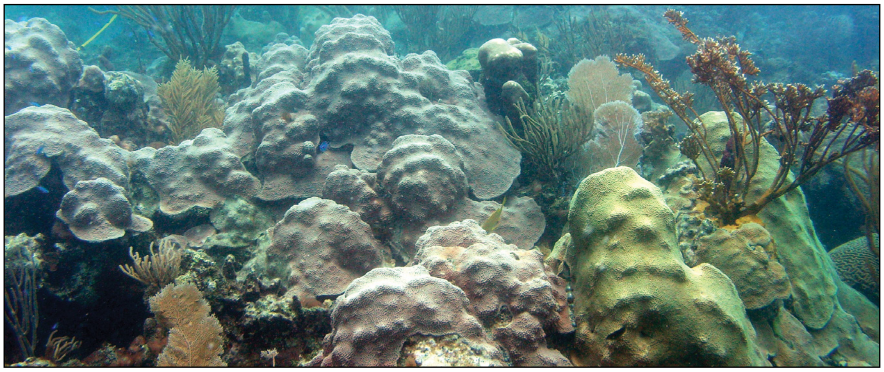


Figure 3.1. Montastraea annularis *complex in St. John, U.S. Virgin Islands (USVI). Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

The Virgin Islands National Park (VIIS) and Virgin Islands Coral Reef National Monument (VICR), in St. John together includes a combined total of 18,358 acres of submerged coastal habitats, which occur within 3 miles of the island's coastline and managed by the National Park Service (NPS; Kumpf and Randall 1961, Beets et al. 1986, Kendall et al., 2001, Zitello et al., 2009; Costa et al., 2009). These coastal habitats comprise an interconnected mosaic that includes intertidal mangrove forests, lagoonal seagrass and algal beds, shallow and deep coral reefs, and unconsolidated sediments that provide shelter and sustenance to fishes and invertebrates which form the basis of important fisheries in the region (Figure 3.1). A major goal of NOAA National Centers for Coastal Ocean Science (NCCOS), Center for Coastal Monitoring and Assessment, Biogeography Branch's (CCMA-BB) 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 for coral reef and hardbottom, seagrass, macroalgae, and unconsolidated habitats as defined by Kendall et al. (2001) in and around the VIIS and VICR (Coral Bay region). More specifically, data are presented to characterize the types, distributions and percent cover of benthic flora and fauna within these mapped substrates.

3.2. METHODS

3.2.1. Field methods for collection of survey data

Figure 3.2. Diver recording benthic habitat composition within the randomly placed 1 m^2 quadrat along the belt transect. Credit: NOAA/NOS/NCCOS/CCMA/ Biogeography Branch.

Underwater visual surveys were conducted to collect benthic composition data along a 25 m belt transect used for fish census within the St. John study area (inside and outside VIIS and VICR) from 2001 to 2009 (Appendix A; Figure 3.2). Benthic data were collected with a detailed and full-scale belt-transect survey (hereafter full-scale habitat,

---

[1] U.S. Geological Survey

[2] Center for Coastal Monitoring and Assessment, National Centers for Coastal Ocean Science, National Ocean Service, National Oceanic and Atmospheric Administration

[3] Consolidated Safety Services, Inc.

[4] South Florida / Caribbean Network, National Park Service

Table 3.1). Within the VICR, benthic data also were collected with a modified rapid habitat assessment survey (RHA) because of logistical constraints on bottom time caused by depth. RHA data were analyzed previously by Monaco et al. (2007 and 2009) and Boulon et al. (2008), and therefore were not analyzed in this report; however, the abstracts and citation from these publications are provided in Appendix B. In addition to data on abiotic and biotic benthic composition, information was collected on queen conch (*Strombus gigas*), long-spined sea urchin (*Diadema antillarum*), Caribbean spiny lobster (*Panulirus argus*) and marine debris. Collection of abundance and distribution data on these selected macroinvertebrates began in 2004, with their abundance being recorded only if individual(s) were observed within 4 x 25 m transects (Appendices A and C). Marine debris data collection began in January 2007 (Figure 3.3). Between 2001 and 2004, benthic habitats were categorized for sampling allocation and design according to the habitat types defined by (Kendall et al., 2001; Figure 3.4). Between 2004 and 2007 however, habitats were re-categorized into three habitat types (hard, soft and mangrove) for sampling and data collection because 1) existing benthic maps were most accurate at the courser habitat classes, 2) high variability in the way scuba divers interpreted and defined fine-scale habitat classes in-situ, and 3) analyses of data from the first year of sampling showed no significant loss in the precision of estimates when aggregated to the coarse-scale habitat classes.

Table 3.1. Abiotic and biotic variables measured to characterize benthic assemblages along fish transects in the St. John, U.S. Virgin Islands (USVI) study region.

	Measurements		
Benthic Variables	**Cover (%)**	**Height (cm)**	**Abundance (#)**
Abiotic			
Hardbottom	x	x	
Sand	x		
Rubble	x		
Fine sediment	x		
Biotic			
Corals (by species)	x		
Macroalgae	x	x	
Seagrass (by species)	x	x	
Gorgonians			
Sea rod, whips, and plumes	x	x	x
Sea fans	x	x	x
Encrusting form	x		
Sponges			
Barrel, tubes, rope, vase morphology	x	x	x
Encrusting morphology	x		
Other benthic macrofauna			
Anemones and hydroids			x
Tunicates and zooanthids			x
Macroinvertebrates			
Queen conch (by sexual maturity)			x
Lobster			x
Long-spined sea urchin			x

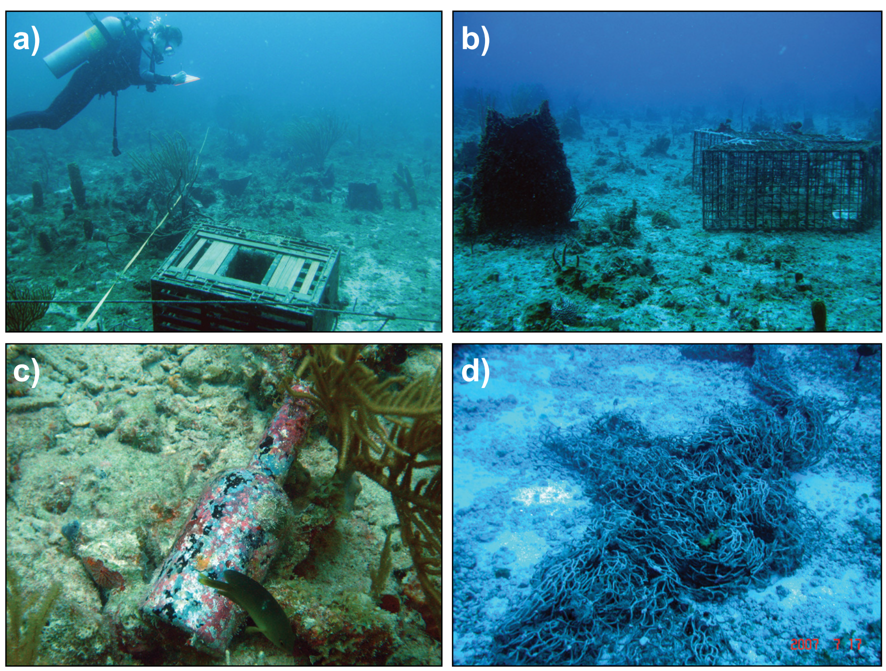


Figure 3.3. Marine debris observed and recorded by benthic divers at sites in St. John: a) diver and lobster trap, b) derelict fish trap, c) encrusted bottle and d) discarded fishing net. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.

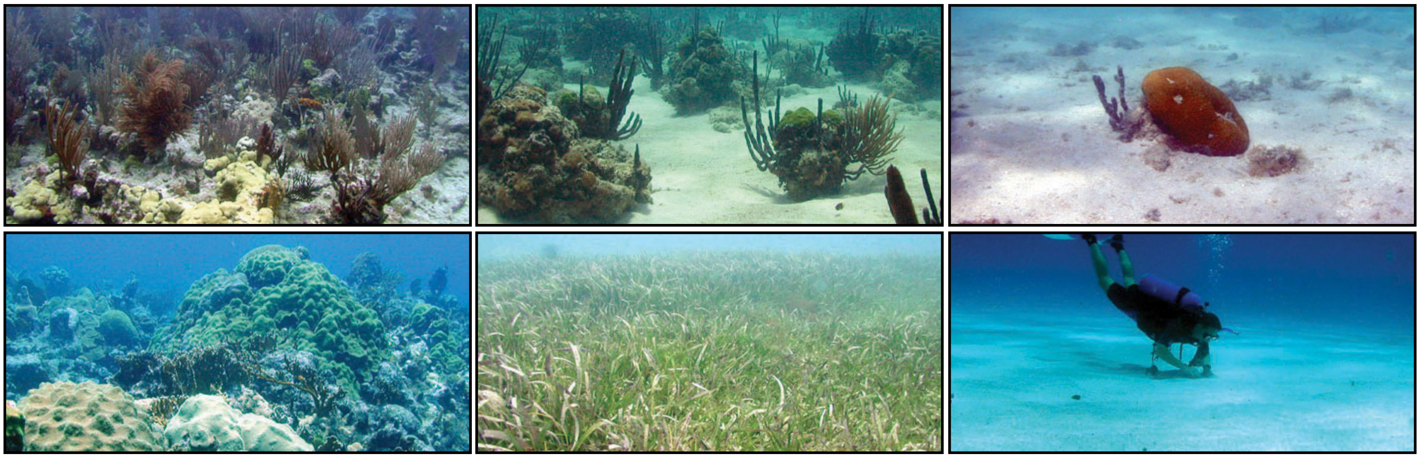


Figure 3.4. A selection of habitat types designated in the hierarchical classification scheme of NOAA's benthic habitat map (Kendall et al., 2001) for the U.S. Caribbean (clockwise from left to right): colonized pavement, patch reef, scattered coral and/or rock, linear reef, seagrass and sand. Source: Kendall et al.(2001): NOAA/NOS/NCCOS/CCMA/Biogeography Branch.

3.2.2. Analytical methods

In situ data on the cover of benthic biota were summarized from 677 full-scale belt surveys during 2001 to 2009 (Table 3.2; Figure 3.5). Within management units, 379 full-scale surveys were conducted inside VIIS and VICR whereas 297 full-scale surveys occurred outside VIIS and VICR (St. John Other [STJ]; Table 3.2). Full-scale surveys were conducted within three broad thematic habitat types in the study area: colonized coral reef and hardbottom areas (hereafter hardbottom habitats; n= 300), seagrass, algal communities (hereafter submerged aquatic vegetation or SAV; n= 242), and unconsolidated sediments (sand and mud habitats; n= 168, Table 3.3).

Table 3.2. Number of benthic habitat sites surveyed by full-scale habitat methods and by year inside and outside National Park Service (NPS) parks in St. John, USVI. VIIS = Virgin Islands National Park, VICR = Virgin Islands Coral Reef National Monument; STJ = St. John Other, outside VIIS and VICR.

Fish Sampling	Management	2001	2002	2003	2004	2005	2006	2007	2008	2009	Grand Total
VIIS study	Inside VIIS	24	45	50	54	45	41	37	34	50	380
	Outside VIIS & VICR	31	45	21	36	28	33	36	40	27	297
VIIS Subtotal		*55*	*90*	*71*	*90*	*73*	*74*	*73*	*74*	*77*	*677*
Coral Bay	Inside VICR	3		19	24	38	33	36	36	33	222
	STJ			18	6	25	25	25	25	25	149
Coral Bay Subtotal		*3*	*0*	*37*	*30*	*63*	*58*	*61*	*61*	*58*	*371*
Grand Total		**58**	**90**	**108**	**120**	**136**	**132**	**134**	**135**	**135**	**1048**

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 pavement and patch reefs (Table 3.4). 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. Although many benthic variables were measured during the surveys, 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 3.1. 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, macroalgae and seagrasses; and 3) temporal trends in live scleractinian (hard) coral and algal cover.

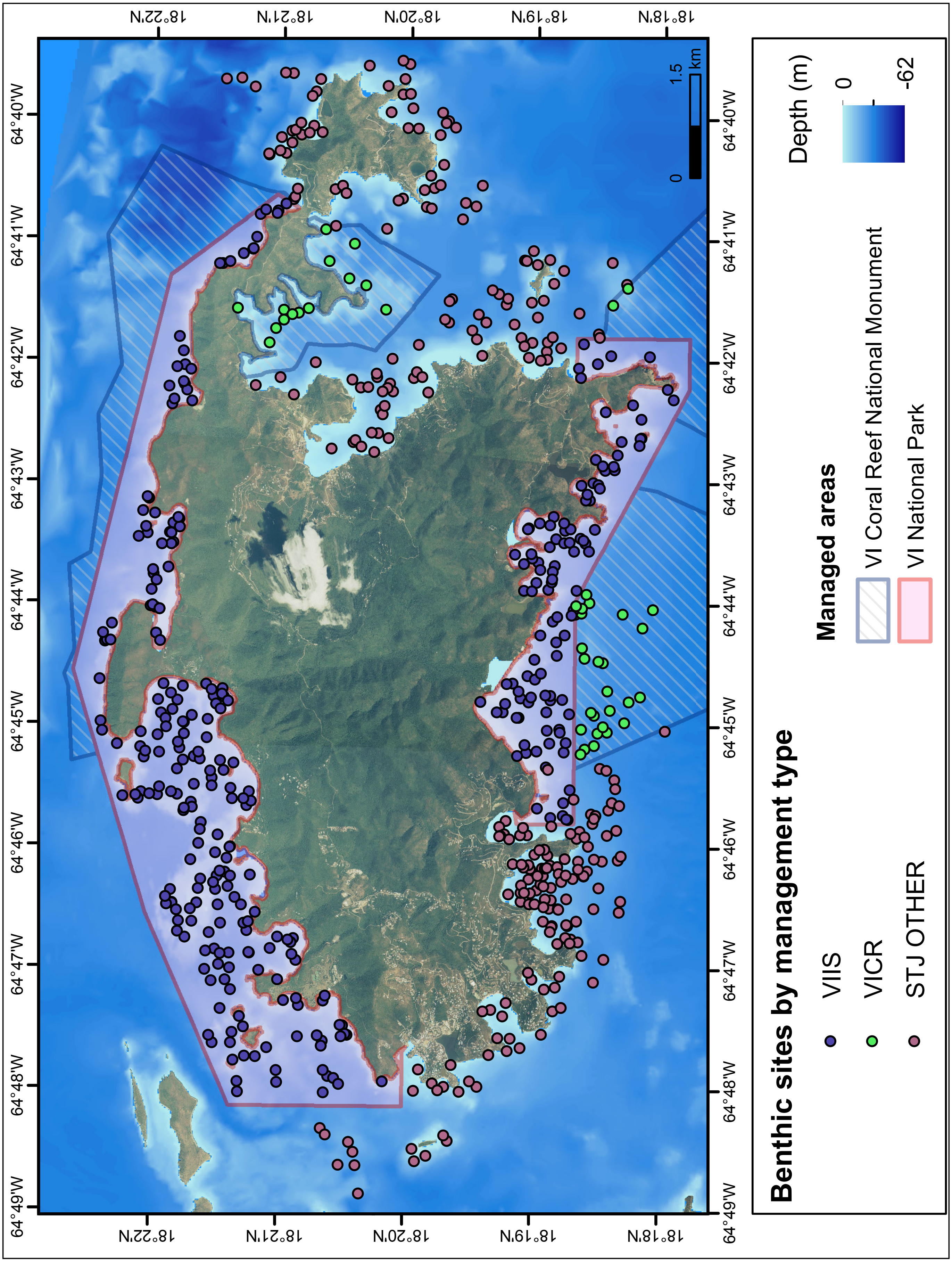


Figure 3.5. NOAA's National Centers for Coastal Ocean Science (NCCOS) Center for Coastal Monitoring and Assessment Biogeography Branch (CCMA-BB) Coral Reef Ecosystem (CREM) project survey sites around St. John and the mid-shelf reef conducted from 2001-2009. Source: K. Stamoulis (University of Hawaii).

Table 3.3. Number of full-scale benthic surveys conducted inside and outside of Virgin Islands National Park (VIIS) and VICR (Coral Bay) by habitat type between 2001 and 2009.

Hardbottom Substrates	Outside VIIS	Inside VIIS	Grand Total
Hardbottom/Reef Rubble		1	1
Hardbottom/Uncolonized Bedrock	3	2	5
Macroalgae/Patchy/10-50%	5	3	8
Macroalgae/Patchy/50-90%		3	3
Reef/Colonized Bedrock	6	15	21
Reef/Colonized Pavement	28	78	106
Reef/Colonized Pavement with Channels	13	17	30
Reef/Linear Reef	47	36	83
Reef/Patch Reef (Aggregated)	8	2	10
Reef/Patch Reef (Individual)	3	1	4
Reef/Scattered Coral-Rock	9	3	12
Sand	2	4	6
Seagrass/Patchy/10-30%		2	2
Seagrass/Patchy/30-50%	2	1	3
Seagrass/Patchy/50-70%	2		2
Seagrass/Patchy/70-90%		2	2
Unknown	1	1	2
Subtotal	*129*	*171*	*300*
Softbottom Substrates			
Macroalgae/Patchy/10-50%	61	4	65
Macroalgae/Patchy/50-90%		17	17
Reef/Colonized Pavement	1	13	14
Reef/Linear Reef		2	2
Reef/Patch Reef (Aggregated)	3		3
Reef/Scattered Coral-Rock	2	1	3
Seagrass/Continuous	17	12	29
Seagrass/Patchy/10-30%	12	7	19
Seagrass/Patchy/30-50%	10	15	25
Seagrass/Patchy/50-70%	13	23	36
Seagrass/Patchy/70-90%	10	19	29
Sand	39	96	135
Subtotal	*168*	*209*	*377*
Grand Total	**297**	**380**	**677**

Table 3.4. Number of full-scale benthic surveys by benthic habitat types and year. VIIS = Virgin Islands National Park, STJ = St. John Other, outside VIIS and VICR.

Management	Year	CB	CP	CPSC	LR	MA	PRA	PRI	RR	Sand	SCRUS	SG	UB	UNK	Total
STJ OTHER	2001		3	3	3	1				11		10			31
STJ OTHER	2002		8		9	2	3			5		15	3		45
STJ OTHER	2003		4	1	3	3		1		3	2	4			21
STJ OTHER	2004	1	3	3	9	10	1			3	2	4			36
STJ OTHER	2005		3	2	5	10				4	1	2		1	28
STJ OTHER	2006	1	1		5	11	1			4	1	9			33
STJ OTHER	2007		2	2	4	11	3	1		4	1	8			36
STJ OTHER	2008	4	2		7	8	3	1		4	2	9			40
STJ OTHER	2009		3	2	2	10				3	2	5			27
Sub total		*6*	*29*	*13*	*47*	*66*	*11*	*3*	*0*	*41*	*11*	*66*	*3*	*1*	*297*
VIIS	2001		6		5	1				6		6			24
VIIS	2002	4	9		10	1	1	1		8	1	10			45
VIIS	2003	2	15	3	3	2				11	1	13			50
VIIS	2004		11	5	4	7			1	9	1	15	1		54
VIIS	2005	5	10	2	2	3				15		7		1	45
VIIS	2006	1	12	1	4	5				8	1	8	1		41
VIIS	2007	2	11		3	1	1			11		8			37
VIIS	2008		7	1	1	4				16		5			34
VIIS	2009	1	10	5	6	3				16		9			50
Sub total		*15*	*91*	*17*	*38*	*27*	*2*	*1*	*1*	*100*	*4*	*81*	*2*	*1*	*380*

CB=colonized bedrock, CP= colonized pavement, CPSC= colonized pavement with sand channels, LR= linear reef, MA= macroalgae, PRA= patch reef (aggregate), PRI= patch reef (individual), RR= reef rubble, SCRUS= scatter coral/rock in unconsolidated sediment, SG= seagrass, UB= uncolonized bedrock, UNK= unknown.

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:

$$\Sigma(Q_i - n) / n \qquad \text{(Equation 1)}$$

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 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).

To assess differences in benthic composition inside versus outside VIIS, the parametric Tukey's HSD (Honestly Significant Difference) pairwise comparison was used for normally distributed data and the non-parametric Wilcoxon test was used for non-normally distributed data (Zar, 1999).

Characterizing seascape spatial patterns

Site values of benthic community metrics averaged from quadrat data were interpolated using inverse distance weighting (IDW) and resulting maps were visually interpreted to understand spatial patterns in benthic variables within the broader seascape around St. John. Simple deterministic interpolations for abundance of corals, benthic cover, and other site specific data were accomplished using the ArcGIS 9.3 Spatial Analyst extension, interpolation tool with Inverse Distance Weighted (IDW). To predict a value for any unmeasured location, IDW uses the measured values surrounding the prediction location. This method assumes that each measured point has a local influence that diminishes with distance and weights the points closer to the prediction location greater than those farther away. “Power” was set to 2 using a variable search radius and the output cell size was 30 m^2. Barrier polylines representing the boundaries of benthic habitat polygons derived from the CCMA-BB habitat maps for St. John were used to limit the interpolations to areas where monitoring was conducted. 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.

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 2009. Average site values were 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. Residual plots of live coral and algae percent cover determined that variance estimates were not homogeneous. 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 non-parametric 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 (Hollander and Wolfe, 1999):

$$U\mu v = \sum_{i=1}^{n\mu} \sum_{j=1}^{nv} \phi(Xiu, Xjv), 1 \leq \mu < v \leq k \qquad \text{(Equation 2)}$$

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 (Jα = 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.

3.3. RESULTS

3.3.1. Benthic habitat composition

Estimates of percent cover (mean ± standard error [SE]) of selected benthic organisms are reported for three habitat types observed in the study area: hardbottom habitats, SAV communities, and sand and mud habitats. 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 3.4). Comparisons presented here are intended to characterize broad differences in benthic composition among the four habitat types inside and outside management zones.

Characterization of hardbottom types

Turf algae – defined as a multispecific assemblage of small filamentous algae less than 1 cm high – was the most prevalent key benthic component accounting for 33% of the total biotic cover in hardbottom habitats (Figure 3.6). Cover of turf algae did not differ significantly inside versus outside VIIS (U=0.42, p=0.68). Macroalgae was the next most abundant benthic cover (15%) with cover 73% higher outside VIIS compared with inside (U=4.89, p<0.001). Live coral was the next most abundant biotic benthic component (4.6%) did not differ significantly inside versus outside VIIS (U=1.19, p=0.23). Although coral cover did not differ between management regimes, the average number of coral genera per transect was significantly greater (t=3.9, p<0.001) outside VIIS (5.7 ± 2.6) compared to inside (4.5 ± 2.4). The cover of gorgonians was similar to coral (4.4%) with significantly higher cover observed inside VIIS (U=3.07, p=0.002). All other benthic components comprised <8% of total biotic cover in total and none differed significantly inside versus outside VIIS.

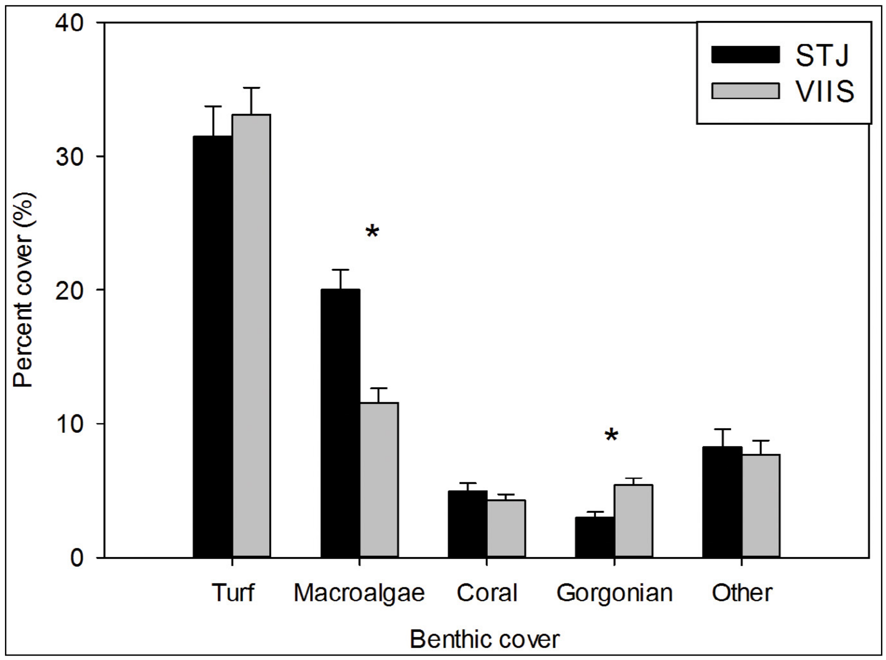


Figure 3.6. Mean (+SE) percent cover for key benthic components across hardbottom sites inside Virgin Islands National Park (VIIS; n=171) and outside VIIS (n=129) in the study region between 2001 and 2009.

Colonized pavement was the most spatially extensive habitat type (35% of the study area), followed by linear reef (28%), colonized pavement (10%), colonized bedrock (7%), scattered coral and rock (4%), and aggregated patch reefs (3%;Table 3.4). Turf algae was most abundant on colonized bedrock (46.3% ± 28.8%) and colonized pavement with sand channels (45.2% ± 19.7%;Figure 3.7). Macroalgae cover was highest on linear reefs (21.0% ± 17.1%) and aggregated patch reefs (19.8% ± 8.7%). Coral cover was highest on linear reefs (7.3% ± 7.1%) and colonized pavements (4.4% ± 7.3%). The cover of gorgonians was highest in colonized pavement with sand channels (6.5 ± 5.5) and linear reef (4.8% ± 6.3%). Low overall sample size in scattered coral and rock (n=12) and aggregated patch reef (n=8) habitats and unbalanced sample allocation between management strata from these habitat strata means that results from the areas should be viewed with caution.

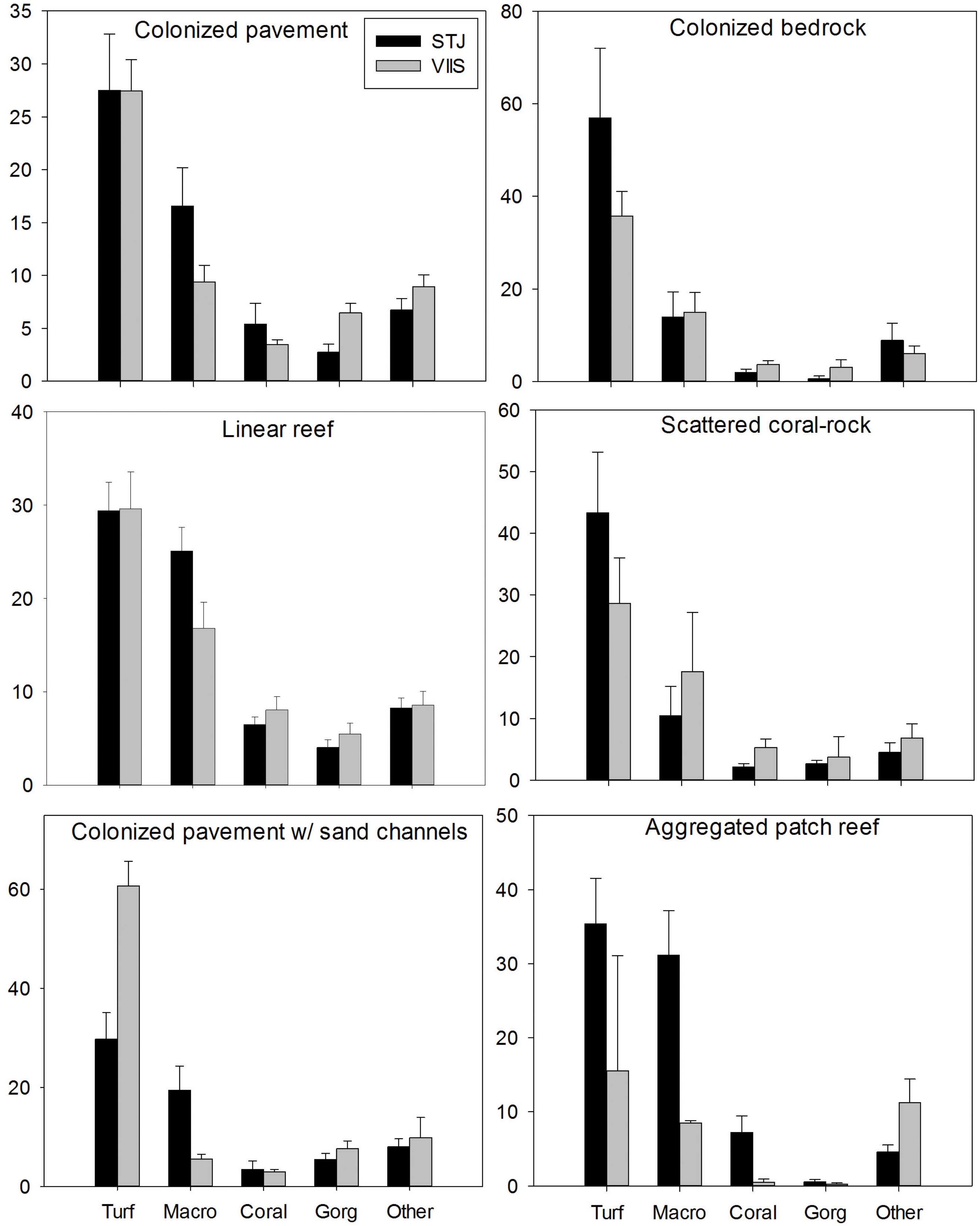


Figure 3.7. Mean (+SE) percent cover of key benthic components inside VIIS (n=171) and outside (n=129) among major habitat types within hardbottom.

Live scleractinian coral cover included 26 coral genera of which 24 were found on hardbottom habitats within the management units of interest (Figure 3.8). The three most abundant coral genera were *Montastraea* (1.80% ± 0.24%), *Porites* (0.97% ± 0.11%), and *Siderastrea* (0.76% ± 0.09%; Figure 3.9). *Montastraea* cover was highest on patch reef (3.9% ± 5.9%), linear reef (3.1% ± 4.3%), and pavement (1.5% ± 5.2%) habitats (Figure 3.10). *Porites* cover was highest on linear reef (1.4% ± 3.3%), colonized pavement (0.9% ± 1.3%), and colonized pavement with sand channels (0.8% ± 1.2%). The highest cover of *Siderastrea* was observed on linear reefs (1.2% ± 2.6%), aggregated patch reef (1.0% ± 1.4%), and colonized bedrock(0.8% ± 1.0%).

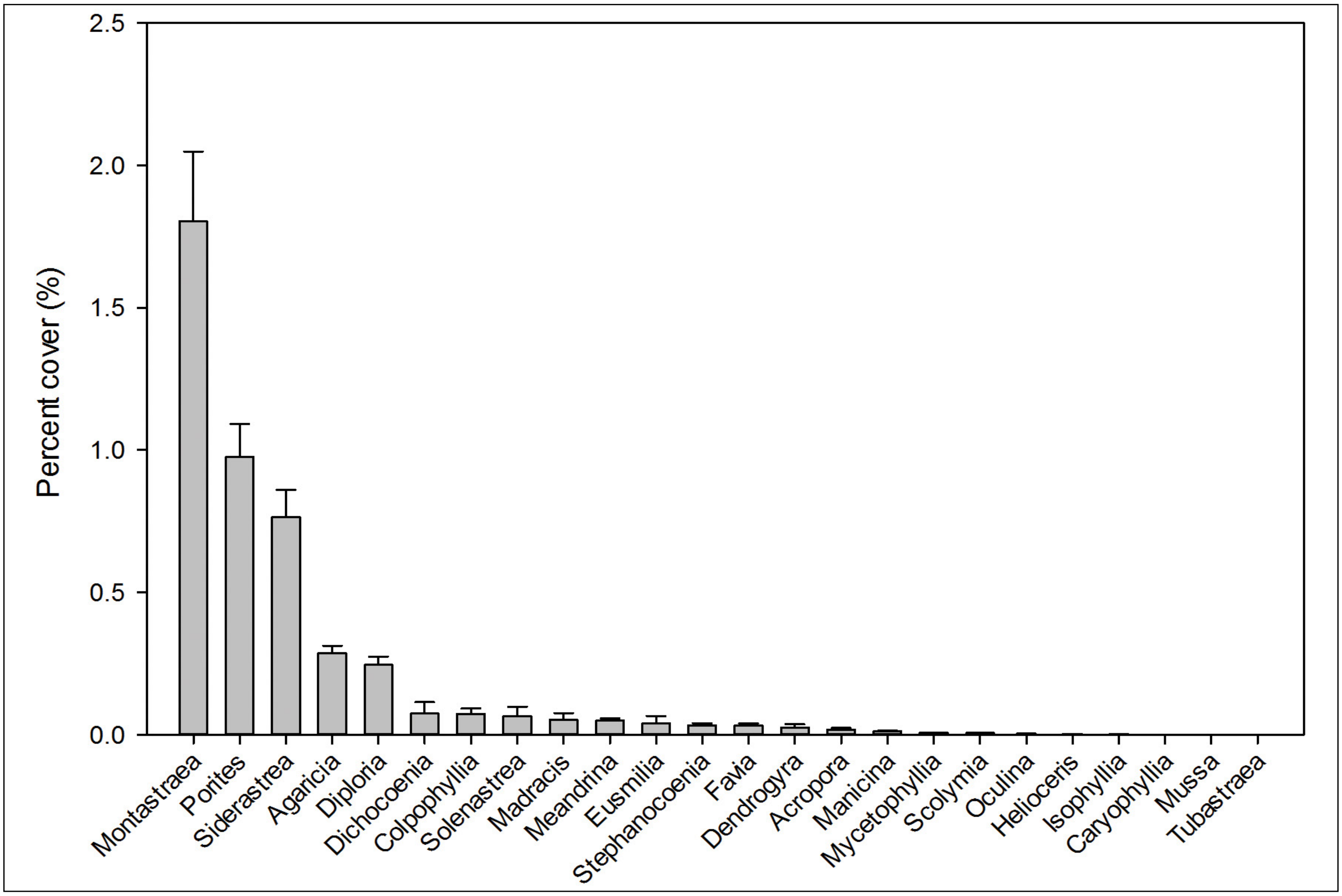


Figure 3.8. Abundance (+SE) of coral genera found across hardbottom sites in the study region between 2001 and 2009.

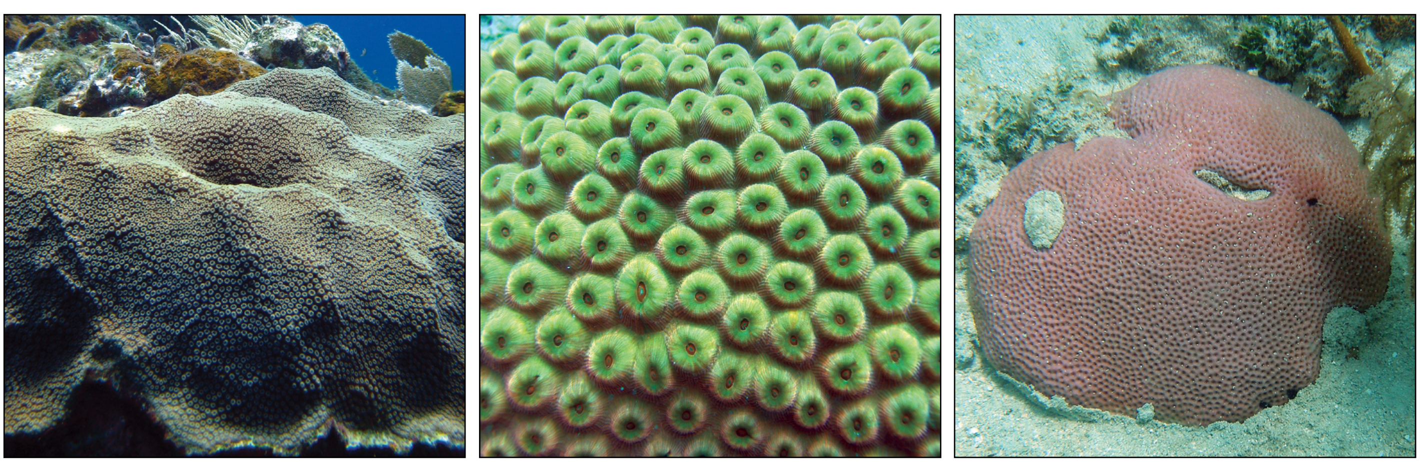


Figure 3.9. Photos of Montastraea annularis *complex (left),* Montastraea cavernosa *(middle) and* Siderastrea sidera *(right). Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

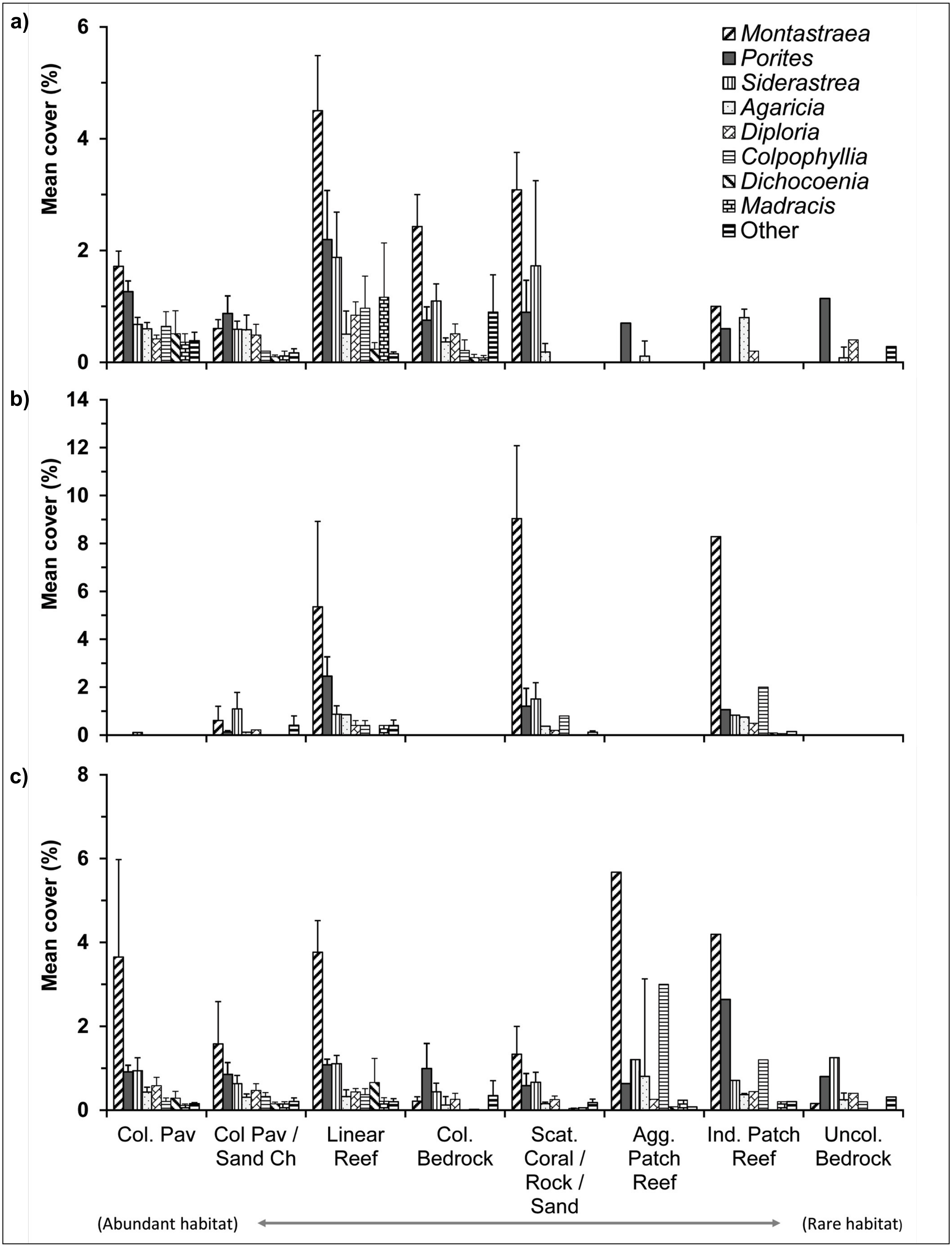


Figure 3.10. Mean (+SE) percent cover of coral genera by hardbottom habitat type in St. John study region from 2001-2009 for: a) VIIS, b) Virgin Islands Coral Reef National Monument (VICR), and c) outside all managed areas.

Characterization of submerged aquatic vegetation (SAV) habitat types

Total cover recorded in SAV habitats was 31.8% ± 1.4%. Seagrasses had the highest mean cover (22.2% ± 1.6%) in habitats classified as SAV followed by macroalgae (16.2% ± 1.2%; Figure 3.11; Figure 3.12). Four seagrass species were observed, of which *Thalassia testudinum* (turtle grass) had the most cover (21.2% ± 1.7%; Figure 3.13). Cyanobacteria and unidentified filamentous algae (5.5% ± 0.9%) were also observed colonizing seagrasses, macroalgae, and patches of hardbottom substrates encountered in SAV habitats. Other organisms found inhabiting SAV habitats included sponges, gorgonians, hard corals, crustose coralline algae (CCA), tunicates, and hydroids such as fire corals. These organisms were rare with mean estimates of cover less than 0.7% ± 0.1%.

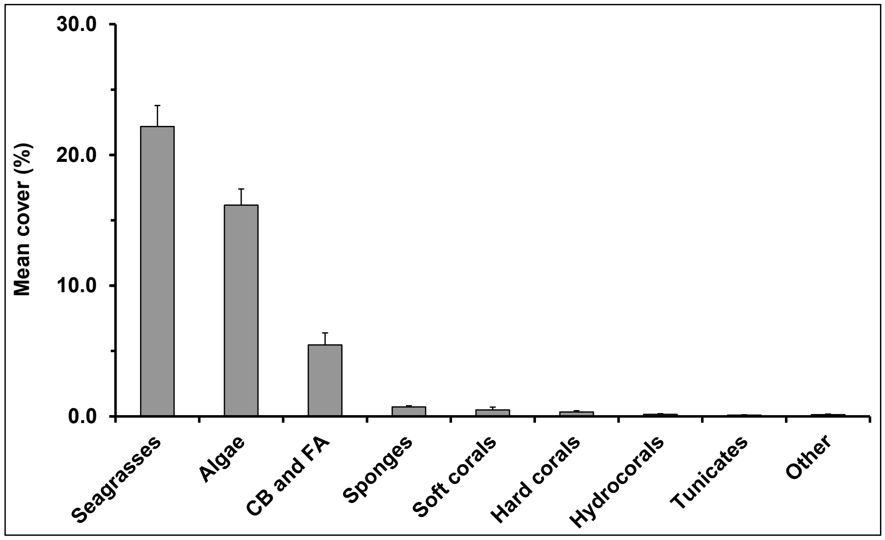


Figure 3.11. Mean (+SE) percent cover of key benthic components on submerged aquatic vegetation (SAV) sites (n=398) in the study region from 2001-2009. CB and FA=cyanobacteria and filamentous algae.

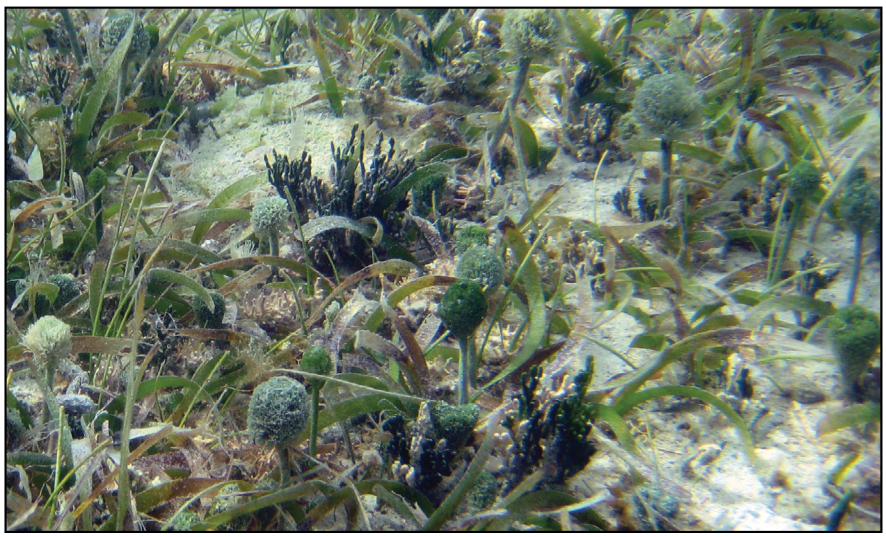


*Figure 3.12. Photo of multiple benthic components observed on unconsolidated sediments such as seagrasses (*Thalassia testudinum *and* Syringodium filiforme*) and macroalgae (primarily* Penicillus *and* Halimeda *genera). Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

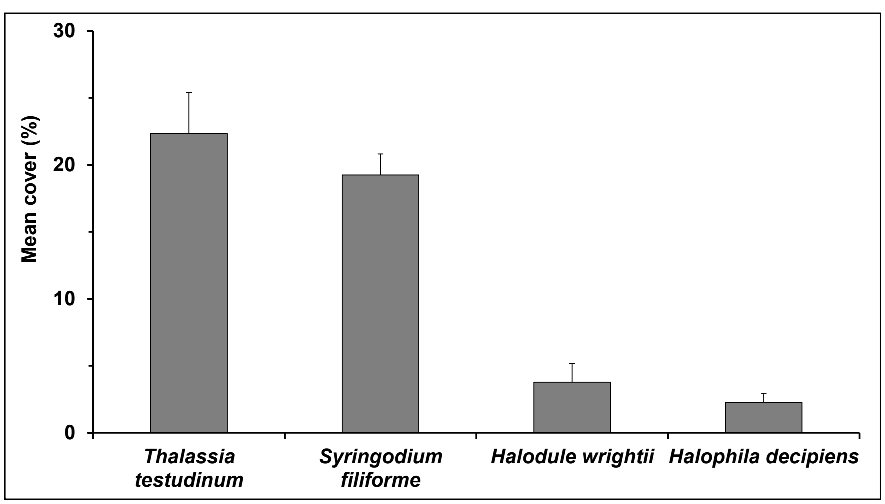


Figure 3.13. Mean (+ SE) percent cover of seagrass species observed on SAV sites (n=272) in the St. John study region from 2001-2009.

Characterization of unconsolidated sediment types

Overall, the total benthic cover on unconsolidated sediments was low (10.9% ± 0.7%). Most of the cover observed on this habitat type was macroalgae (18.8% ± 5.8%), followed by seagrass (10.6% ± 5.6%), and sponges (0.9% ± 0.7%; Figures 3.14 and Figure 3.15). Hard and soft corals, sponges, cyanobacteria, filamentous algae, and tunicates were also observed, but their mean cover was less than 1.2% ± 0.2%. 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* (manatee grass) was the most dominant of the seagrass species in unconsolidated sediments habitats, with a mean cover of 16.0% ± 4.7% (Figure 3.16).

3.3.2. Spatial 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.

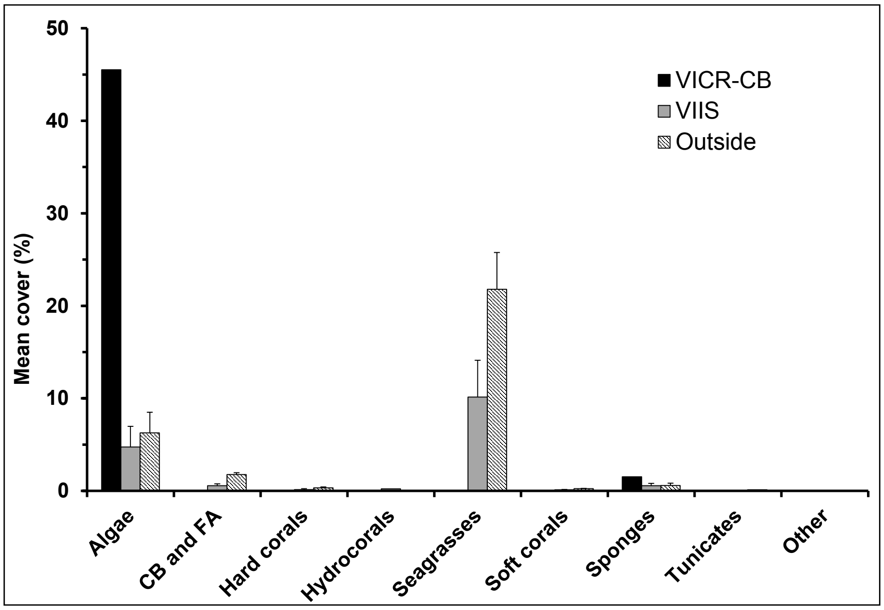


Figure 3.14. Mean (+SE) percent cover of key benthic components on unconsolidated sediment sites (n=86) in the study region from 2001-2009. VICR-CB= VICR within Coral Bay; Outside= outside any management area; CB and FA=cyanobacteria and filamentous algae; Other= other invertebrates.

Figure 3.15. Photo of Penicillus *algae observed and recorded in sand habitats in the St. John study region. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

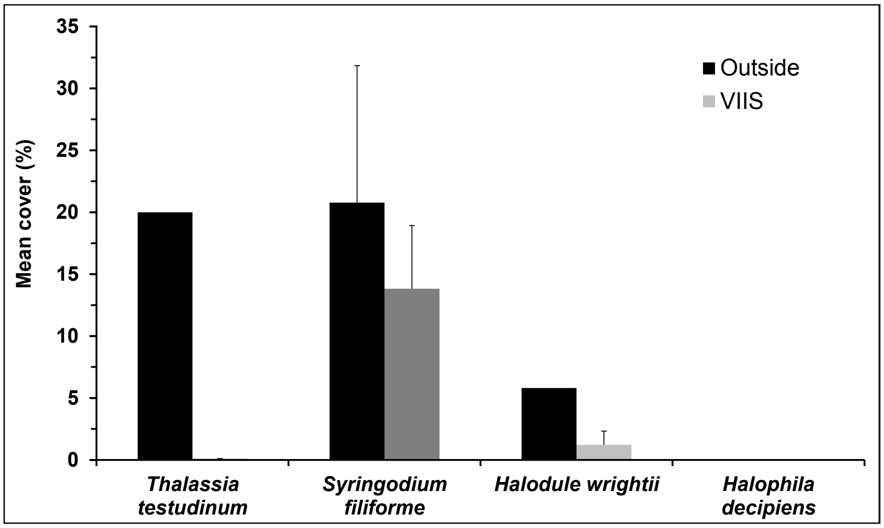


Figure 3.16. Mean (+ SE) percent cover of seagrass species observed on unconsolidated sediment sites (n=86) in the St. John study region from 2001-2009. Outside= outside VIIS management area.

Spatial patterns in coral cover

Examination of an interpolated surface of live coral cover indicated that areas with higher live coral cover were more extensive within Coral Bay and along the northeastern portion of St. John from Haulover Bay to Newfound Bay (Figure 3.17a). These areas also had the highest number of coral genera with additional areas off of Leduck Island (also known and referred to as LeDuc) and Eagle Shoal to the southeast and Johnston Reef in the northwest also having higher numbers of coral genera (Figure 3.17b). Most of the locations that demonstrated greater numbers of coral genera and higher percent coral cover, were also the most topographically complex as reflected in the high indices of rugosity in these areas (Figure 3.17c). A number of the areas with high coral cover and generic richness were outside VIIS, particularly around the eastern portion of the island.

Interpolations of cover for the six most abundant coral species showed patchy and uneven distributions (Figures 3.18 and 3.19). *Montastraea annularis* complex was more common in Coral Bay, along the northeast near Newfound Bay, and at scattered locations along the southeast shore from White Cliffs to Booby Rock (Figure 3.18a). Cover of *Porites astreoides* was higher in a few locations in Coral Bay including Johnson's Bay, in Haulover Bay, and at a few locations along the northwest shore including Hawksnest Bay, Johnson's Reef, and Cinnamon Bay (Figure 3.18b). *Siderastrea siderea* had higher cover off Leduck Island, off Whistling Cay, and scattered locations along the southshore (Figure 3.18c). *Montastraea cavernosa* demonstrated fairly patch distribution and overall low percent cover inside and outside VIIS, with a few distinct areas of high percent cover off the eastern end of St. John and off Mary's Point (Figure 3.19a). *Siderastrea radians* had areas of higher cover between Cabritte Horn Point and Ram Head (Figure 3.19b). Cover of *Porites porites* showed higher concentrations around Durloe Cays and Johnson's Reef in the northwest, in Round Bay inside Coral Bay, and between Haulover and Newfound Bay to the northeast (Figure 3.19c).

Spatial patterns in macroalgal cover

Intermediate to high levels of macroalgal cover (17-90%) were evident over a broad expanse of hardbottom area, particularly along the southeast of St. John from Great Cruz Bay to Rendezvous Bay (Figure 3.20a). Additional areas of high macroalgae cover were observed in Coral Bay and along the eastern shore near John's Folly Bay. Low levels (4%-17%) of macroalgal cover were present across the north shores of St. John. Algal turf was broadly distributed around St. John, both within and outside VIIS, with most of the algal turf on the south and east shores occurring in shallow, nearshore locations (Figure 3.20b). High percentages of algal turf cover were also observed on Johnson's Reef and off Cinnamon Bay off the northwestern shore. Very few areas had CCA exceeding 31% cover (Figure 3.20c). Areas of high CCA cover were relatively localized and occurred is a few deep water sites outside the southern VIIS boundary (Figure 3.20c).

Spatial patterns in seagrass cover

Seagrasses were common on the softbottom habitats in the study area (Figure 3.21). Seagrass cover was most extensive close to shore within intermediate to high percent cover within Coral, Lameshur, Reef, and Rendezvous Bays (Figure 3.21a). Large continuous areas of seagrass are present nearshore, adjacent to mangroves in the Coral Bay and inside the fringing reefs. In this zone, seagrass was predominantly *T. testudinum* that was abundant to depths of approximately 16 m (Figure 3.21b). *T. testudinum* cover was highly variable, ranged from 0.02 to 78.6%, and occurred primarily in Coral Bay where several sites had greater than 30% cover and Rendezvous Bay which also had some sites with greater than 30% cover (Figure 3.21b). *S. filiforme* was commonly observed at several bays with intermediate (>30%) and high (> 60 %) cover Hawknest, Lameshur, Rendezvous, Reef, and Trunk Bays, and also at one site around Ram's Head Point (Figure 3.21c). Similar to that of *T. testudinum*, percent cover of *S. filiforme* was highly variable and ranged from 0.02 to 87.0% when observed.

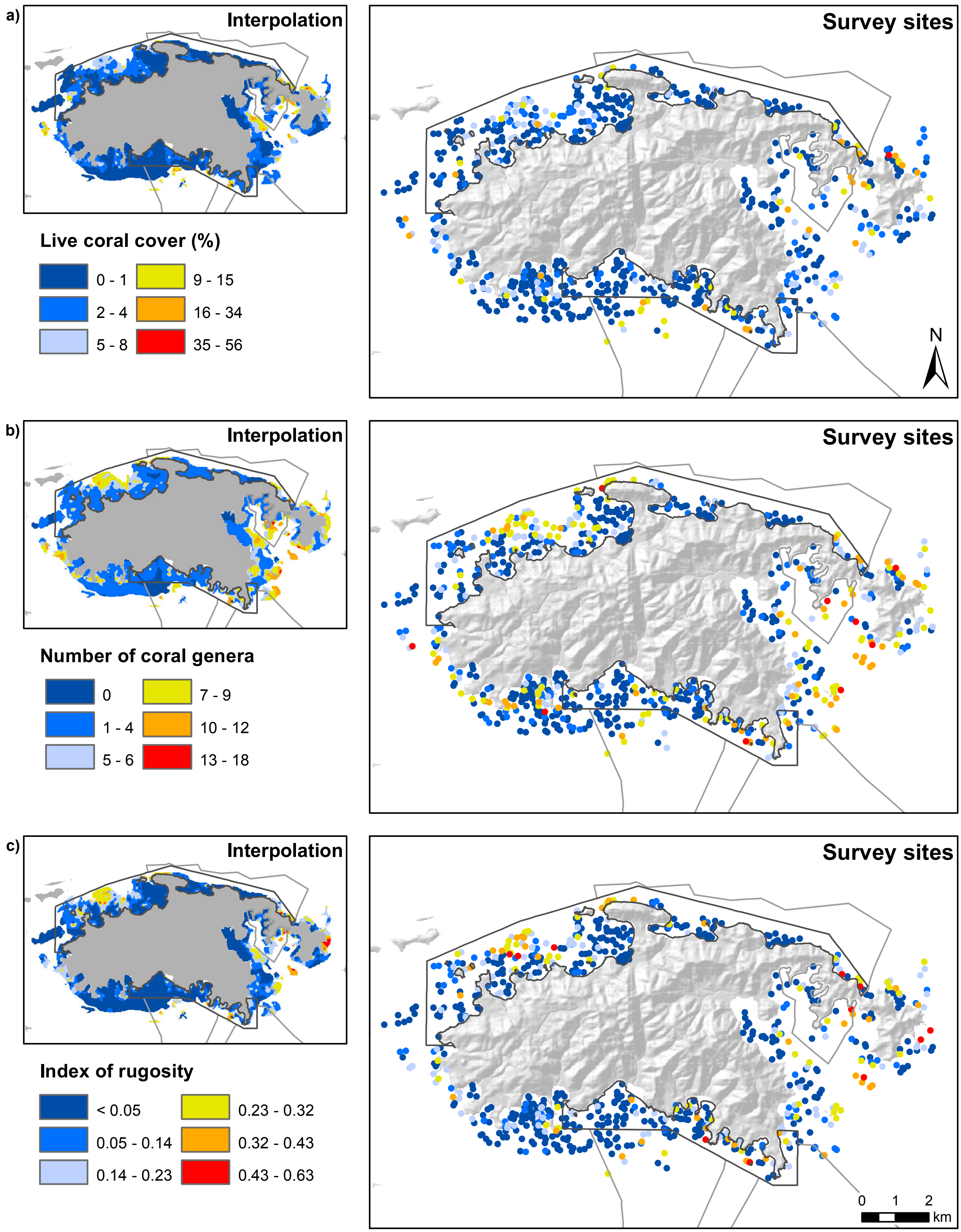


Figure 3.17. Spatial distributions of benthic components at all transects in the study region between 2001 and 2009. (a) Percentage live coral cover (hard coral including fire coral); (b) number of coral species/groups; and (c) rugosity. Source: K. Stamoulis (University of Hawaii).

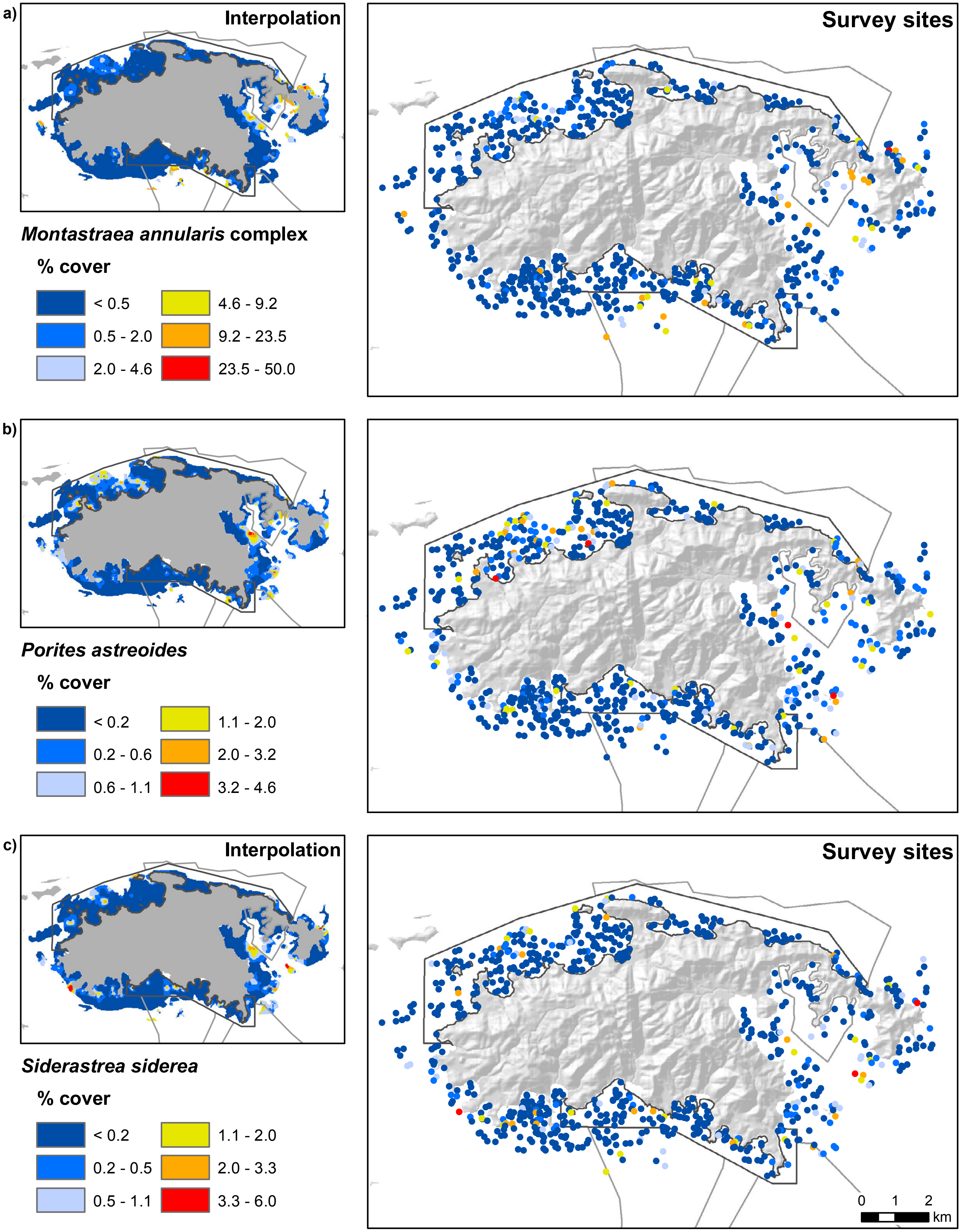


Figure 3.18. Spatial distributions of coral cover for individual coral species at all transects in the study region between 2001 and 2009. (a) Montastraea annularis *complex, (b)* Porites astreoides*, and (c)* Siderastrea siderea*. Source: K. Stamoulis (University of Hawaii).*

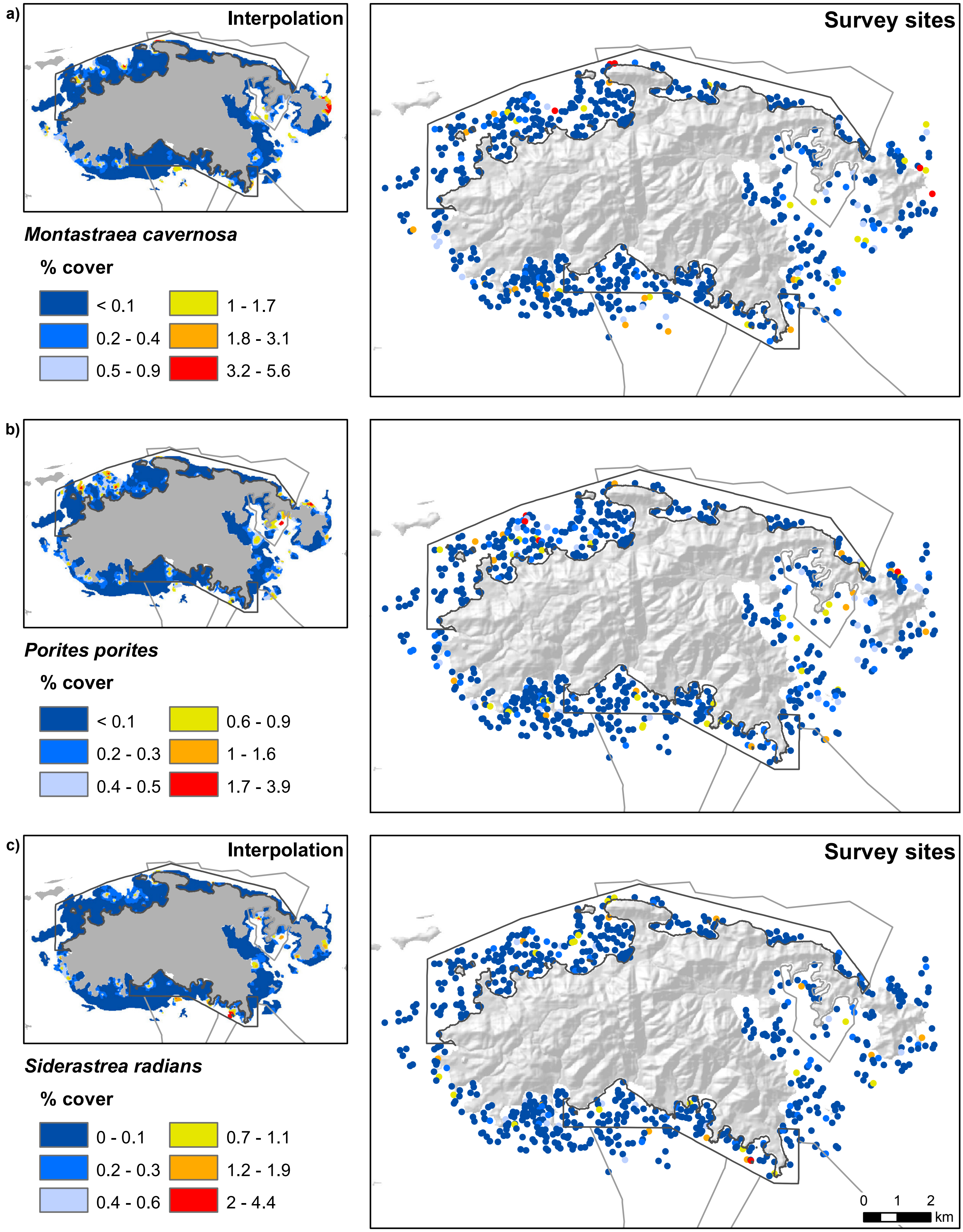


Figure 3.19. Spatial distributions of coral cover for individual coral species at all transects in the study region between 2001 and 2009. (a) Montastraea cavernosa*, (b)* Porites porites *and (c)* Siderastrea radians*. Source: K. Stamoulis (University of Hawaii).*

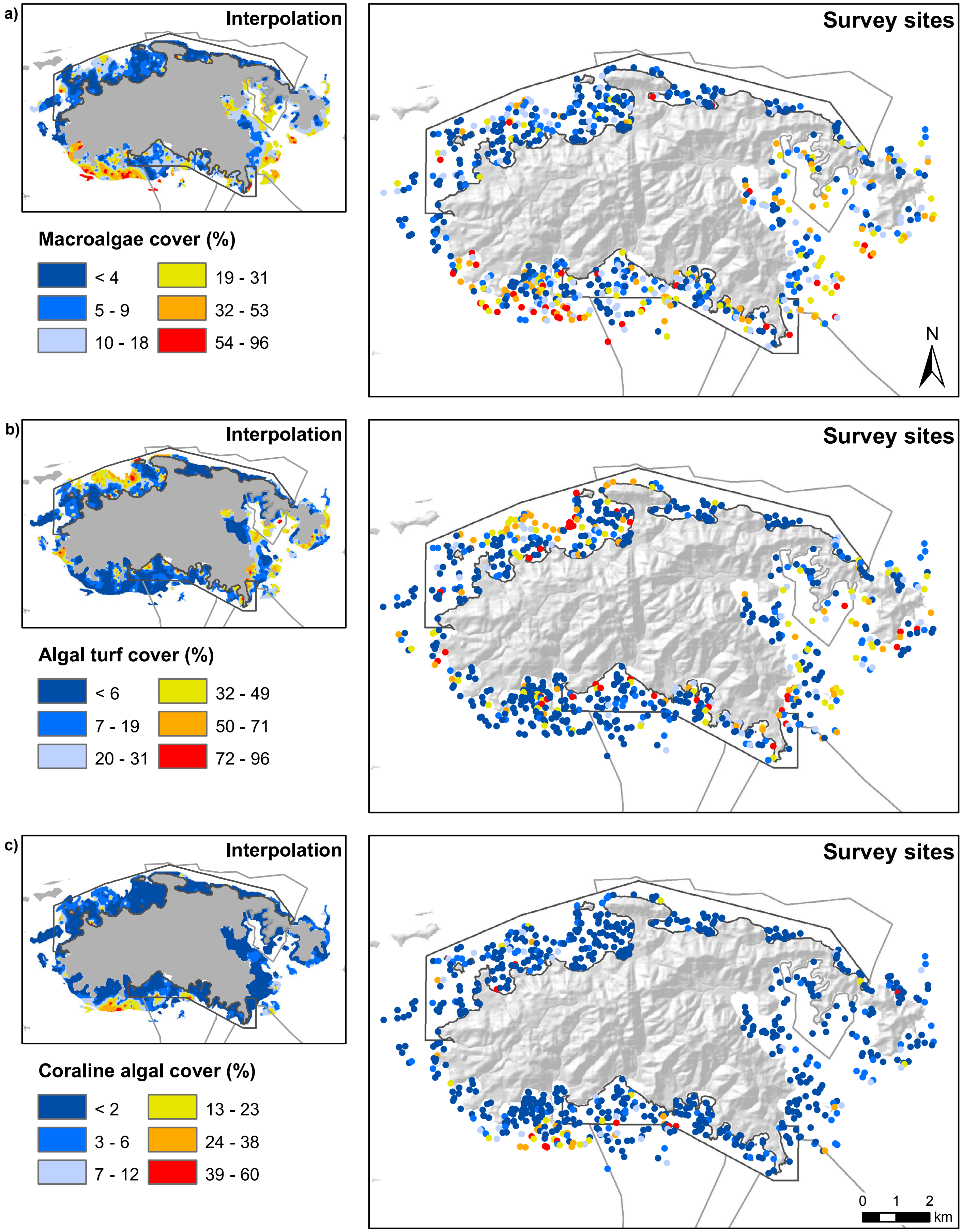


Figure 3.20. Spatial distributions of benthic components at all transects in the study region between 2001 and 2009. (a) Macroalgal cover (including filamentous algae/cyanobacteria); (b) algal turf cover; and (c) crustose coralline algal cover. Source: K. Stamoulis (University of Hawaii).

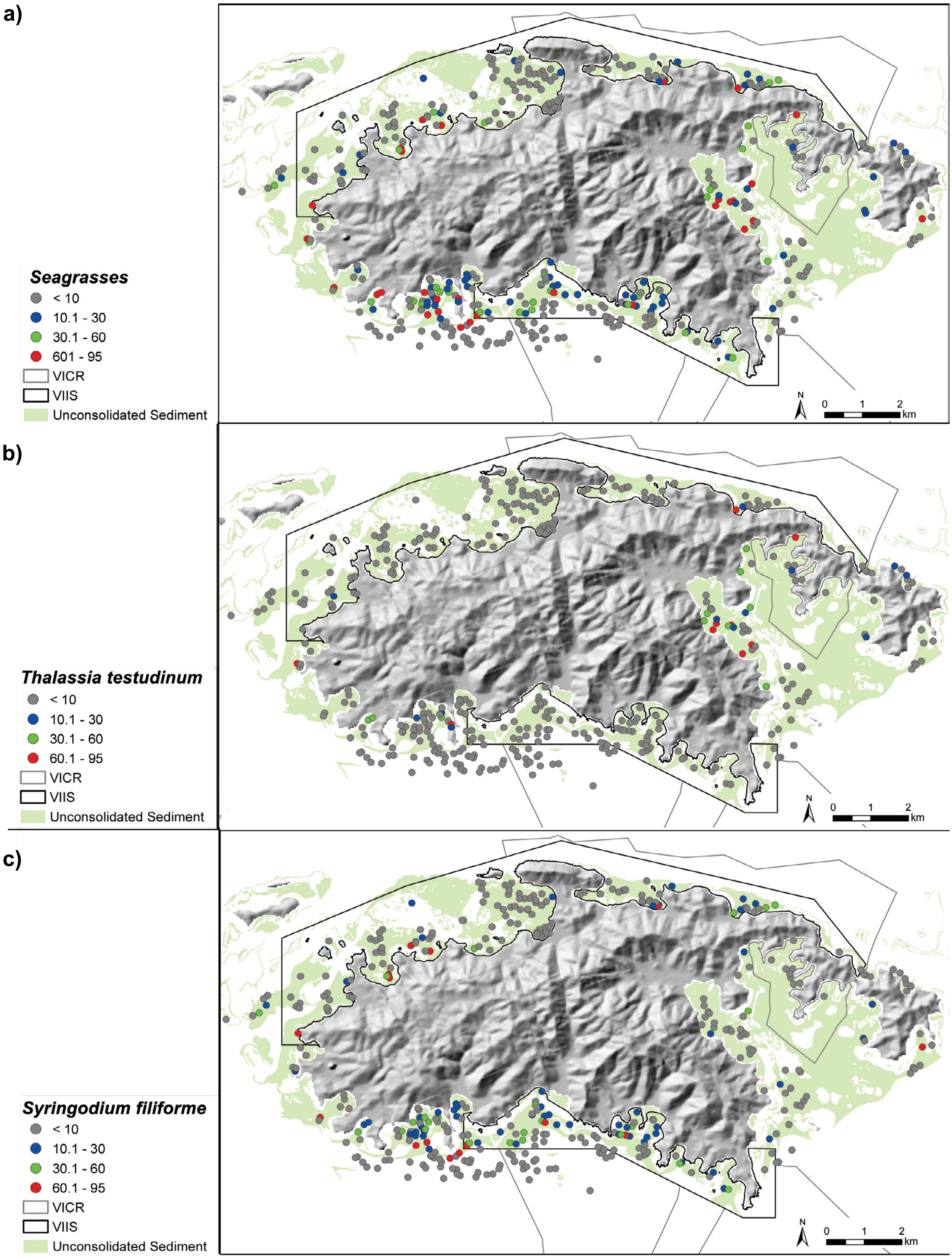


Figure 3.21. Spatial distributions of benthic components at all transects in the study region between 2001 and 2009. (a) all seagrasses, (b) Thalassia testudinum*; and (c)* Syringodium filiforme*. Source: C.F.G. Jeffrey (NOAA/NOS/NCCOS/CCMA/Biogeography Branch).*

3.3.3. Temporal patterns in benthic composition

Temporal trends in the percent cover of live coral and algae were based on multiple pair-wise comparisons among sampling periods as well as before and after a major bleaching event that occurred in 2005. 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.

Temporal patterns in coral cover

Mean live coral cover declined over the entire study area between 2001 and 2009 ($F_{1,299}$=15.8, p<0.001, r^2=0.05) and this decline was significant both inside (p=0.01) and outside (p=0.02) VIIS (Figure 3.22). Outside VIIS, no two years differed significantly from one another (p>0.05 for all pair-wise test). Inside VIIS, the highest coral cover was observed in 2003 (7.4% ± 6.3%) and cover for that year differed significantly from cover observed inside VIIS in 2004 (p<0.05) and 2008 (p<0.05).

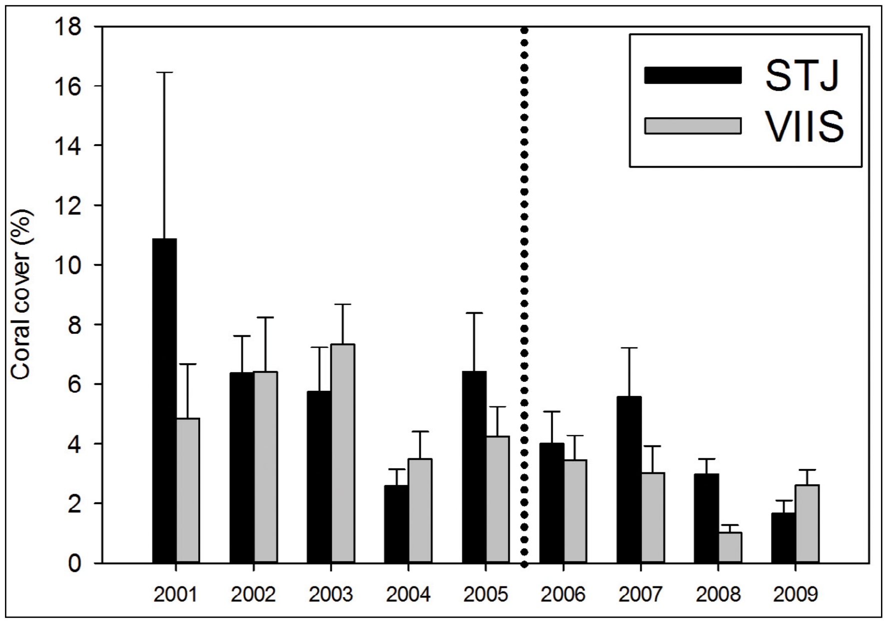


Figure 3.22. Inter-annual patterns of live coral cover inside and outside VIIS over a nine year sampling period. Means and error bars indicate (+SE). Dashed line indicates October 2005 bleaching event.

In September 2005 a mass coral bleaching event was recorded in the region (Figure 3.22). Comparisons of coral cover before (2001-2005) and after (2006-2009) the bleaching event showed significantly lower coral cover after the event (t=3.0, p=0.003). Coral cover declined significantly inside VIIS between these two time periods (p=0.005) but the decline outside was less dramatic and not significant (p>0.05).

Bleaching and Coral Disease

During the Summer/Fall of 2005, anomalously warm water temperatures (Whelan et al., 2007) and doldrum-like conditions in the NE Caribbean created massive coral bleaching (Wilkinson and Souter, 2008). Prior to the 2005 Caribbean coral-bleaching event, long term monitoring around St. John and Buck Island had shown decreasing coral cover due to hurricanes (e.g., Hubbard et al., 1991; Rogers et al., 1997), anchor damage (Rogers and Garrison, 2001), disease (Miller et al., 2003), and overfishing (Rogers et al., 2008). Monitoring of 100 randomly chosen, permanent transects at five study sites in the USVI revealed over 90% of the scleractinian coral cover showed signs of thermal stress by paling or becoming completely white as a result of these anomalous environmental conditions (Miller et al., 2009; Figures 3.23 and 3.24). Lower water temperatures in October allowed some re-coloring of corals; however, a subsequent unprecedented regional outbreak of coral disease affected all sites. Isolated disease outbreaks have been documented before in the Virgin Islands, but never as widespread or devastating as the one that occurred after the 2005 Caribbean coral-bleaching event.

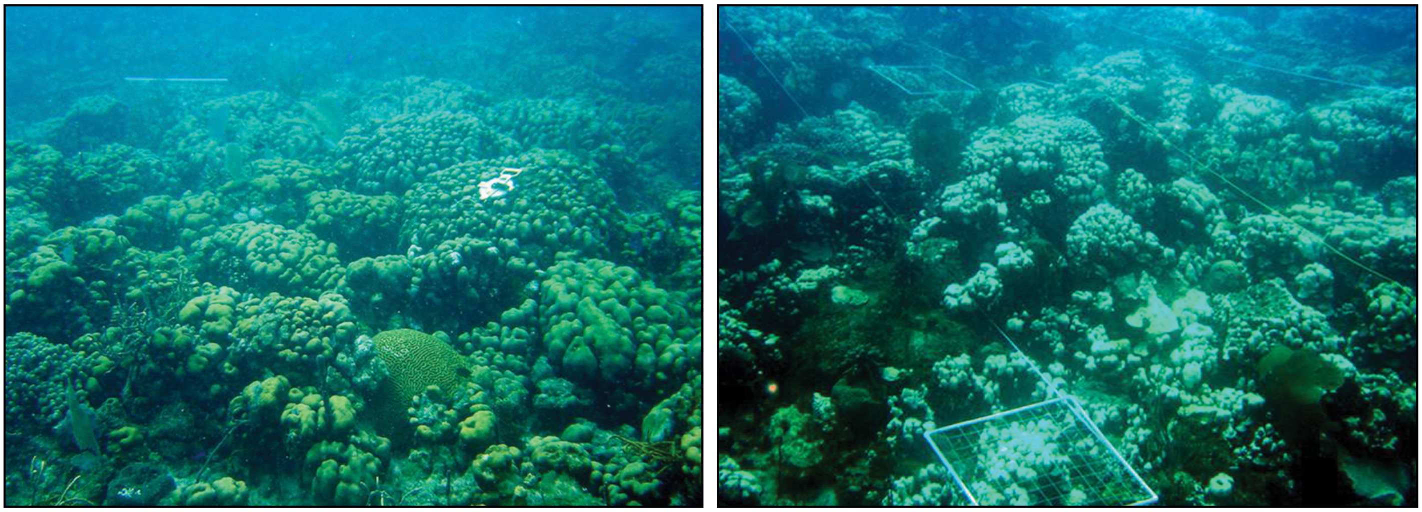


Figure 3.23. Tektite Reef before and after 2005 bleaching event. Credit: J. Miller (National Park Service).

Figure 3.24. Time series of identical video captures at Tektite Reef showing: a) bleached Montastraea annularis*, September 2005; b)* M. annularis *re-coloring and heavily affected by coral disease, November 2005; c) near-total mortality of* M. annularis *with surviving portion still pale, January 2006. Source: Miller et al. (2009).*

Temporal patterns in algal cover

Turf algae cover increased significantly over the course of the study period both inside and outside VIIS (p =0.002 and p <0.001, respectively; Figure 3.25a). Although the rate of increase was greater inside VIIS (slope=0.057) compared with outside (slope=0.036), these rates were not significantly different (p=0.73). Turf algae cover was significantly higher in the time period after the 2005 bleaching event (2006-2009) compared with before the event (2001-2005) for both inside (p<0.001) and outside (p=0.006) VIIS. There was no significant trend in macroalgae cover over the sampling period overall or within management strata (all p>0.05; Figure 3.25b). A sharp increase in macroalgae cover was documented between 2005 and 2006 (following the bleaching event), but the high variability in macroalgae cover resulted in non-significant differences before and after the event (p>0.05). By 2009, macroalgae cover declined to pre-bleaching event levels. Gorgonian cover declined significantly over the study period (p=0.02; Figure 3.25c). Within VIIS, gorgonians declined significantly (p=0.05) while the decline outside VIIS was not significant (p=0.13).

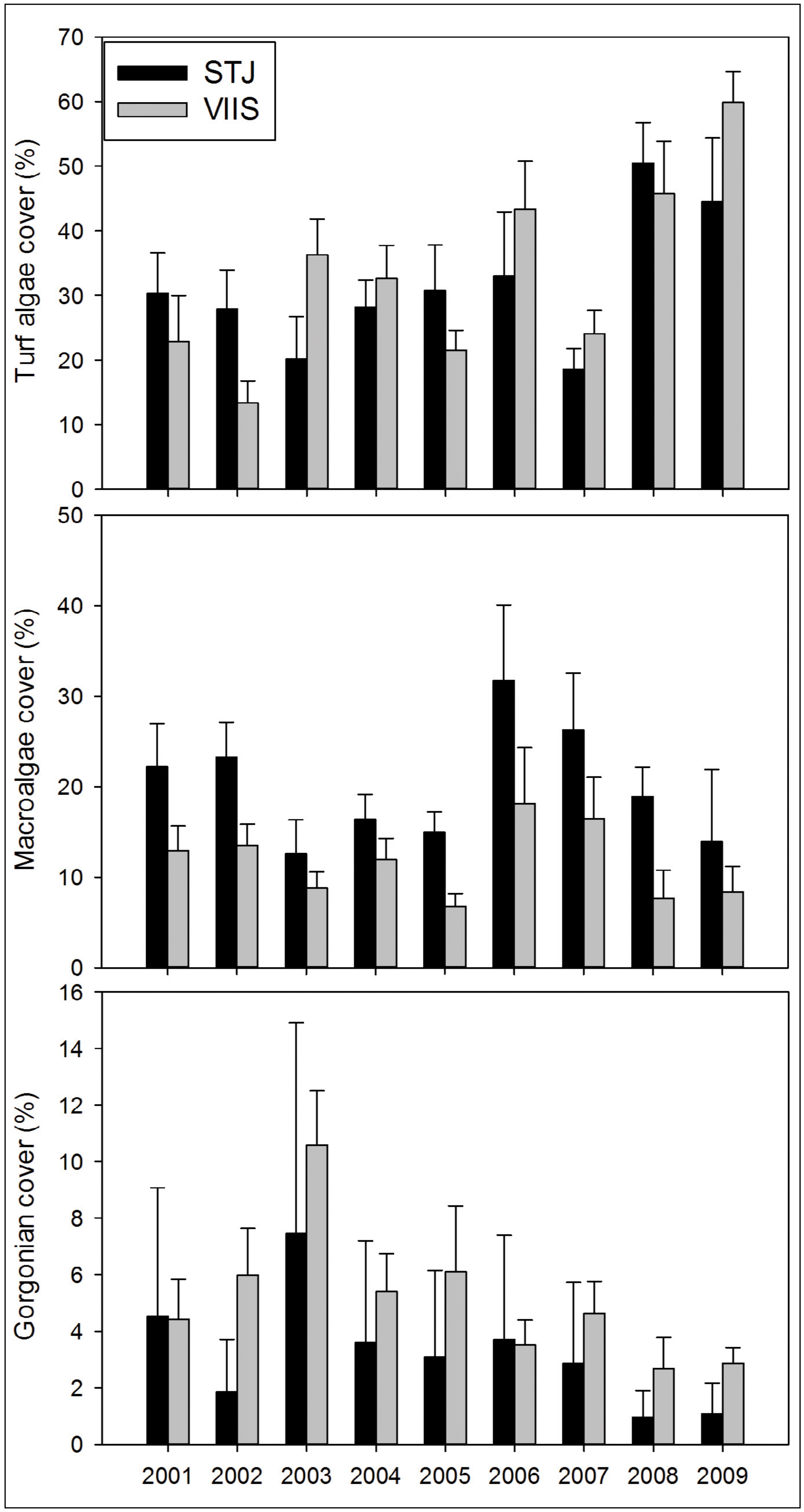


Figure 3.25. Inter-annual patterns in percent cover of (a) turf algae (b) macroalgae, and (c) gorgonians. Values are means and error bars indicate (+SE).

3.3.4. Benthic composition of Coral Bay

In the Coral Bay area, turf algae comprised the dominant biotic cover on hardbottom (34.5% ± 17.6%), followed by macroalgae (19.4% ± 11.8%), coral (7.7% ± 7.7%), gorgonians (4.0% ± 3.6%), and others (4.7% ± 3.4%; Figure 3.26). There were no differences in cover between management strata among these key benthic components (all p>0.05, Figure 3.27). Live coral cover was highest along Johnson Bay and inside Round Bay (Figure 3.28). Coral generic richness was also high along Johnson Bay, as well as off Turner Point and Long Point. Macroalgae was highest around the inner portions of Coral Bay and Hurricane Hole.

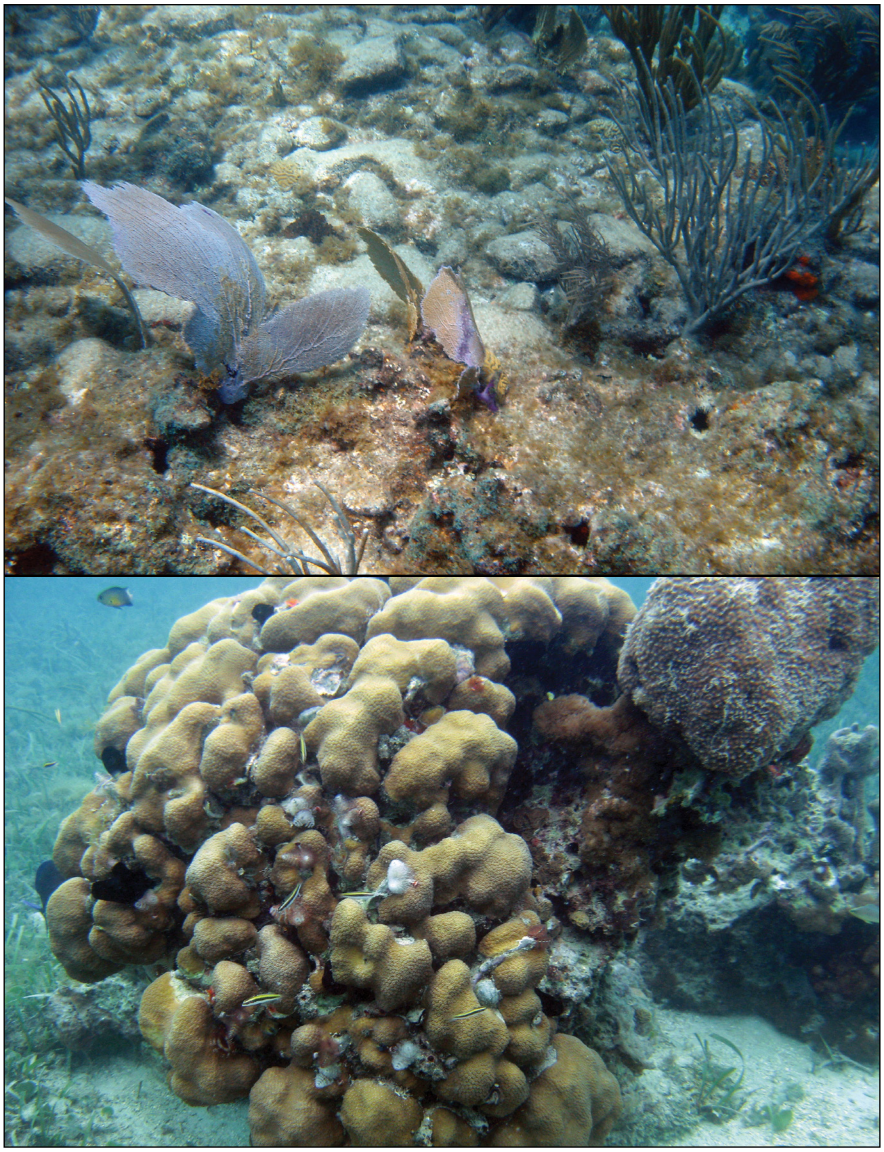


Figure 3.26. Photos of turf algae over hardbottom (top) and M. annularis *complex (bottom) in Coral Bay. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

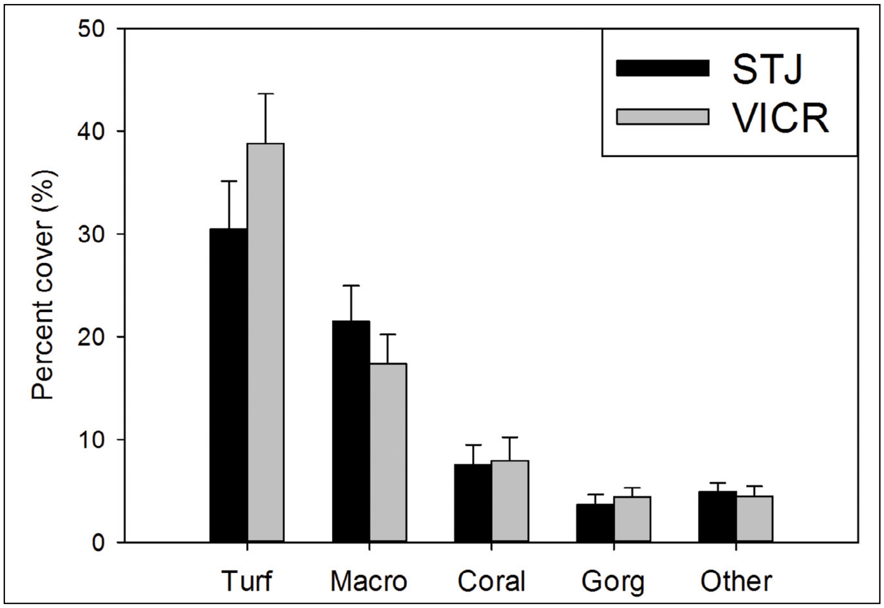


Figure 3.27. Percentage cover (+SE) for key benthic components across hardbottom sites inside Coral Bay VICR and adjacent areas between 2001 and 2009.

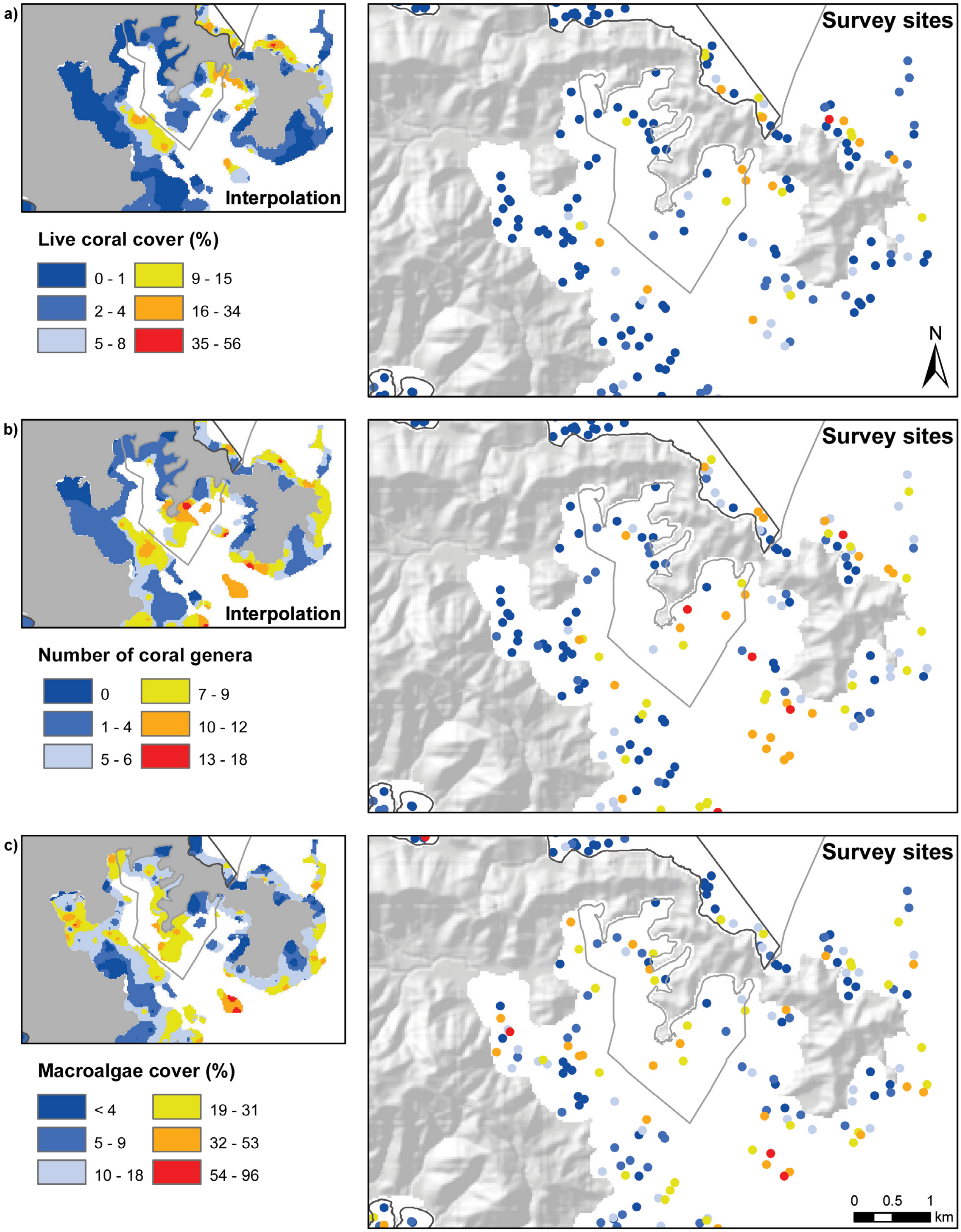


Figure 3.28. Spatial distributions of key benthic components at all quantitative benthic survey sites within Coral Bay between 2001 and 2009. (a) Percentage live coral cover; (b) number of coral species/groups; and (c) macroalgae cover. Source: K. Stamoulis (University of Hawaii).

3.4. DISCUSSION

This study resulted in a comprehensive examination of spatial and temporal patterns of benthic composition within three major habitat types around St. John, USVI. In general, fine-scale (100 m^2) *in situ* data sets on benthic composition are valuable complements to marine benthic maps of nearshore environments that are used for conservation and management of biological resources. Due to cost restraints, however, such maps are often 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. Kendall et al. (2001) for example, used a minimum mapping unit (MMU) of 0.4 ha (1 acre) to identify 26 distinct benthic habitat types in the USVI and Puerto Rico, and that information has been used repeatedly by local management agencies for broad-scale spatial planning and designing programs for *in situ* monitoring of biological resources. However, these maps and even those with a smaller MMU of 1000 m^2 and greater spatial resolution (e.g., Zitello et al. 2009; Costa et al., 2009) contain identifiable benthic features (such as sand halos and small patch reefs in unconsolidated substrates or sand patches within hardbottom habitats) that were not delineated as distinct features because they 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 0.5 ha (Helfman, 1978; Meyer et al., 1983; 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.

3.4.1. Colonized hardbottom habitats: benthic characterization and spatial patterns

Although the composition of benthic substrates varied spatially within and among habitat types in St. John, some general spatial patterns in occurrence and cover of benthic organisms were observed. Most coral reefs and hardbottom substrates in St. John including the VIIS and VICR appear to be dominated by some form of algae, with occasional patches of hard corals, gorgonians, sponges, and other encrusting invertebrates. For example, turf algae was the most extensively occurring benthic organism group within all hardbottom habitat types, followed by macroalgae (Figure 3.29). Another general pattern was the low average cover of live scleractinian coral (~5%) on coral reef and hardbottom areas. Such low coral cover is now typical of most reefs in the USVI and other parts of the Caribbean and has resulted from the synergy of natural and anthropogenic factors operating over the several decades (Gardener et al. 2003; Jeffrey et al., 2005; Rogers et al., 2008; Rothenberger et al., 2008).

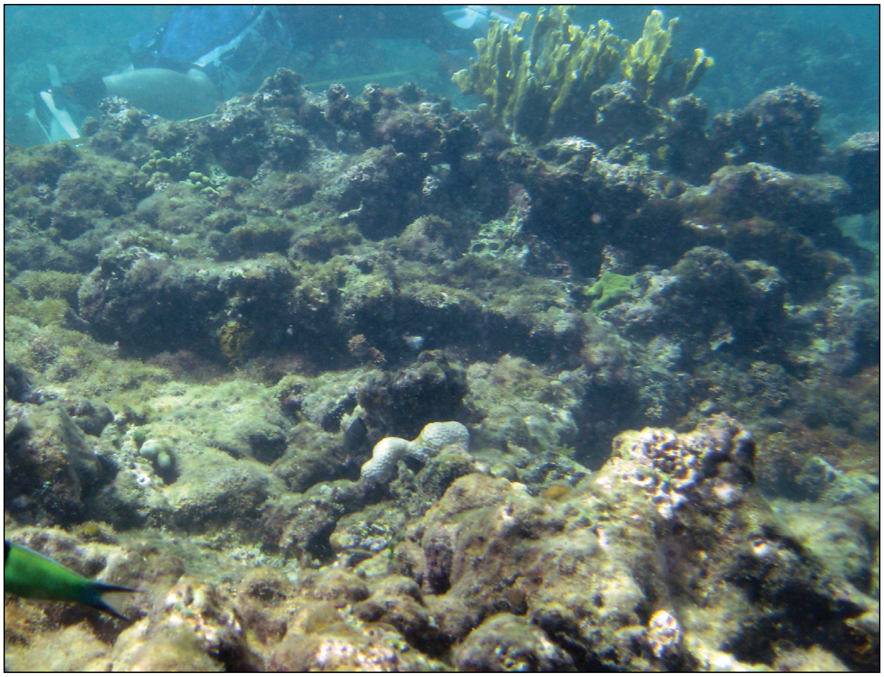


Figure 3.29. Hardbottom habitat dominated by turf algae and various species of macroalgae in St. John, USVI. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.

Interpolations of this study's synoptic estimates of live coral cover, which were summed across species, revealed a few hotspots of relatively high coral cover in southeastern St. John, particularly in Coral Bay (see Figure 3.17a). These hotspots may be refuge areas where demographic processes have resulted in coral populations that are resilient to multiple synergistic stressors (Pittman et al., 2010). 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 numbers of coral genera and high rugosity (Figure 3.17b,c). Protection of these hotspots may benefit ecosystem conservation, but several of these hotspots occur in areas outside of the VIIS. Interestingly, the five most dominant taxa in terms of coral cover around St. John were *M. annularis* complex (frame-building), *M. cavernosa* (mound-shaped) and three weedy species (*P. astreoides*, *P. porites*, and *S. siderea;* Figure 3.30).

Figure 3.30. Photo of Porites porites *in St. John, USVI. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

Current densities of *A. palmata* (elkhorn coral) are on reefs in St. John are nowhere as high as they were in the 1960s and 1970s (Rogers et al., 2002; Rogers et al., 2008). However, monitoring data from NPS, USGS, NFMS, and UVI indicate that several colonies remain extant, and that elkhorn-dominated fore-reef zones are still present on at least 13 bays around the island. These hotspots of elkhorn coral provide some promise for long-term resilience and persistence of coral reefs in St. John, if mortality from localized stressors (e.g., sedimentation and physical damage from snorkeling, boaters, and other forms of human use) are minimized. Further work is needed to understand the physical and oceanographic properties that correlate with their enhanced ecological features. At least nine of the 13 bays with elkhorn coral occur within VIIS and would benefit from increased management and protection from anthropogenic stressors. Reef frame builders such as *A. palmata* and *M. Annularis* complex 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). NPS resource managers should work with their Territorial counterparts to implement policies to reduce local stressors and to educate local users about minimizing damage to these elkhorn reefs.

3.4.2. Softbottom habitats: benthic characterization and spatial patterns

Analysis of the benthic maps used in this study showed that softbottom habitats comprised approximately 53% of the study area (20% macroalgae, 18% seagrass, and 16% unconsolidated sediments). As shown by the spatial interpolations of synoptic estimates from this study, softbottom areas in St. John 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 were also zoned. *T. testudinum* dominated nearshore areas up 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 macroalgae habitats. Calcareous macroalgae (e.g., *Halimeda* spp., *Udotea* spp., and *Penicillus* spp.; Figure 3.31) were commonly encountered on softbottom habitats, but their percent cover was low relative to those of seagrass and more foliose algae (such as *Lobophora*, *Dictyota*, and *Padina* spp.). Never-the-less 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).

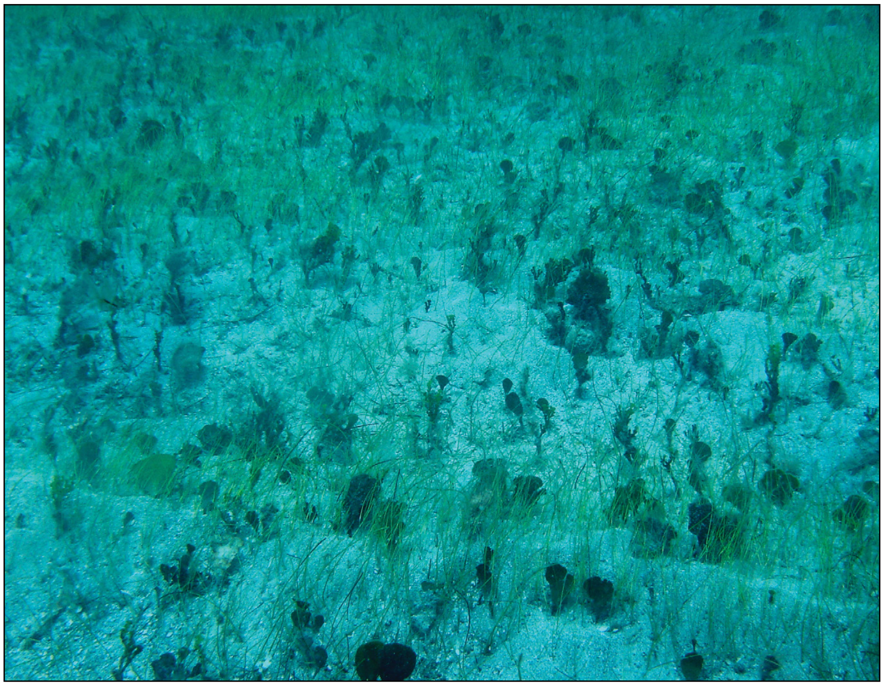


Figure 3.31. Photo of calcareous alga Udodea *and* Halimeda *spp. in* S. filiforme *seagrass in St. John, USVI. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

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 most likely was 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 hardbottom areas, forage on adjacent non coral reef habitats (sand, seagrasses, and 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., 2007; Pittman et al., 2010). 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.

3.4.3. Temporal trends in benthic composition on hardbottom habitats

Temporal analysis of data on percent cover revealed a general trend of a decrease in live coral in St. John, particularly on pavement habitats between fall of 2003 and summer of 2007. Our observations of a temporal decline in live coral cover around St. John are consistent with those of other long-term studies as reviewed in Rogers et al. (2008). Additionally, Miller et al. (2009) reported that average live coral cover at permanent sites in St. John declined from 21.4% in 2005 to 8% by October 2007 after the 2005 widespread event. Temporal declines ranging from 40-50% in live coral have been reported at Buck Island, St. Croix, around La Parguera Puerto Rico, off Isla Desecheo, Mayguez, Guanica, and Ponce, with most of the loss occurring after the 2005 bleaching event (Garcia-Sais et al., 2008; Pittman et al., 2010). 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). Much of the reported loss in live coral occurred in a few species, namely *M. annularis* complex, *Colpophyllia natans*, and *Agaricia agaricites* (St. John; Figure 3.32), *M. annularis* complex (Puerto Rico), and *M. annularis* and *Agaricia* spp. (Buck Island, St. Croix). After the drastic decline in acroporid corals, *Montastraea* remains one of the most abundant coral species in St. John (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).

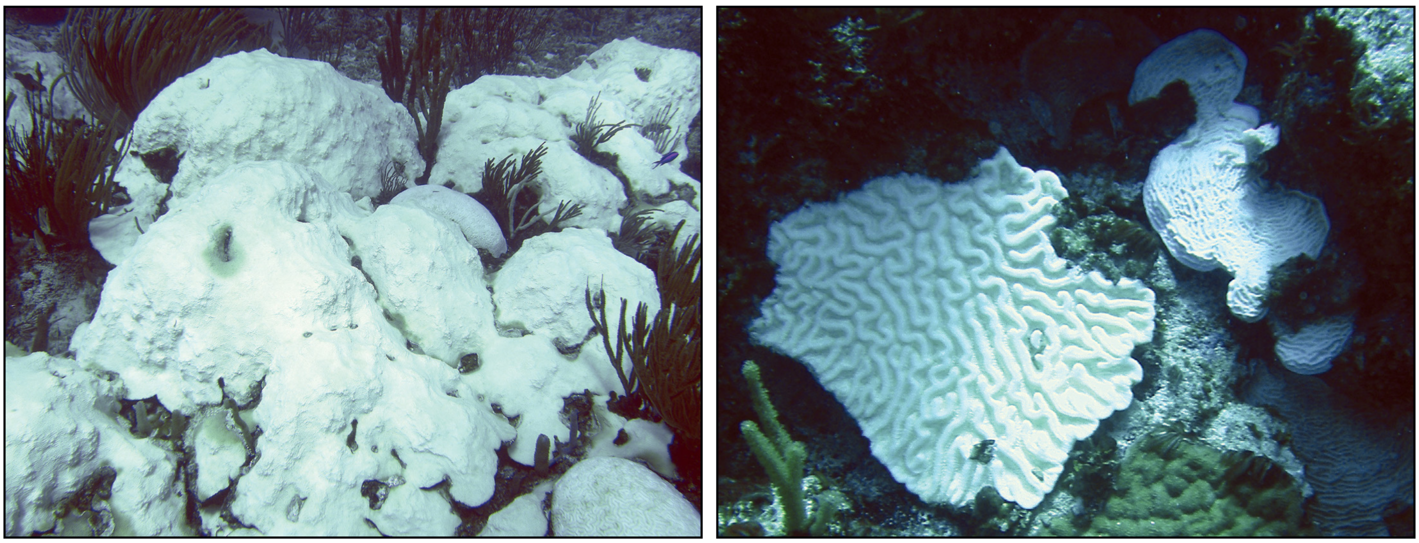


Figure 3.32. Photos of bleached M. annularis *complex (left) and* Agaricia *spp and* Colpophyllia natans *(right) in St. Croix, USVI. Photos were taken around Buck Island, St. Croix during the October 2005 bleaching event. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

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.

Such coral loss and reef degradation have 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 nearshore 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 inside and outside VIIS 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.

3.4.4. Summary

Diversity hotspots

- The highest generic coral richness and coral cover were found at the mouth of Coral Bay, along the north shore between Haulover and Newfound Bays, and along the south shore between Lameshur and Salt Pound Bays.

Benthic habitats

- The dominant benthic habitat types on hardbottom were colonized pavement (33% of sampled), followed by linear reef (27%). Softbottom sites consisted of 34% sand, 34% seagrass, and 20% patchy marcoalgae (10-50%) cover.
- Average percent hard coral cover pooled across all habitats around St. John was 2.19% (± 4.89% SD). Coral cover on hardbottom sites only was 4.88% (± 6.37% SD). Coral cover on hardbottom in VIIS was 4.27% and 4.95% around adjacent areas of St. John. Interpolated surface of live coral cover revealed several patches with relatively high cover (16-56%) of living stony coral inside VIIS (e.g., Hawksnest, west of Salt Pond, and west of Haulover Bay) and outside the VIIS (e.g., Haulover to Newfound Bay. Coral Bay).
- Live scleractinian coral cover included at least 26 coral genera. The three most abundant coral genera were *Montastraea* spp. (1.41%), *Porites* spp. (0.70%), and *Siderastrea* spp. (0.68%). *Montastraea* spp. cover was highest on patch reef, linear reef, and pavement; *Siderastrea* spp. and *Porites* spp. cover were highest on linear reef and patch reef.
- In October 2005 a mass coral bleaching event was recorded in the region. Coral cover inside VIIS dropped from an average of 5.5% from before the bleaching event to less than 3% afterwards. Similarly, coral cover outside VIIS was 5.8% prior to 2005 and declined to 3.5% afterwards.
- Turf and macroalgae comprised the major components of the biotic benthic cover on hardbottom in both VIIS and adjacent sites. Turfs accounted for 33% of the cover inside VIIS and 31% outside. Macroalgal cover was 54% higher outside VIIS (20%) compared to inside (13%).
- Among the most abundant algaes were *Dictyota* spp., Rhodophyta spp., *Lobophora* spp., and *Halimeda* spp.

3.5. REFERENCES

Adams, A.J., C.P. Dahlgren, G.T. Kellison, M.S. Kendall, C.A. Layman, J.A. Ley, I. Nagelkerken, and J.E. Serafy. 2006. Nursery function of tropical back-reef systems. Mar. Ecol. Prog. Ser. 318: 287-301.

Beets, J., L. Lewand, and E. Zullo. 1986. Marine community descriptions and maps of bays within the Virgin Islands National Park/Biosphere Reserve. Virgin Islands Resource Management Cooperative. Biosphere Reserve Research Reports 2. Virgin Islands National Park, St. John, U.S. Virgin Islands. 118 pp.

Burkepile, D.E. and M.E. Hay. 2008. Herbivore species richness and feeding complimentarity affect community structure and function on a coral reef. Proceedings of the National Academy of Sciences 105:16201-16206.

Chittaro, P.M. 2004. Fish-habitat associations across multiple spatial scales. Coral Reefs 23: 235-244.

Clark, R., C. Jeffrey, K. Woody, Z. Hillis-Starr, and M. Monaco. 2009. Spatial and temporal patterns of coral bleaching around Buck Island Reef National Monument, St. Croix, U.S. Virgin Islands. Bull. Mar. Sci. 84(2): 167-182.

Costa, B.M., L.J. Bauer, T.A. Battista, P.W. Mueller, and M.E. Monaco. 2009. Moderate-depth benthic habitats of St. John, U.S. Virgin Islands. NOAA Technical Memorandum NOS NCCOS 105. Silver Spring, MD. 57 pp. Online: http://www.ccma.nos.noaa.gov/ecosystems/coralreef/benthic/StJohnModDepthRPTsm.pdf (Accessed 14 May 2013)

Dahlgren, C. and J. Marr. 2004. Back reef systems: Important but overlooked components of tropical marine ecosystems. Bull. Mar. Sci. 75: 145-152.

Done, T. 1992. Phase shifts in coral reef communities and their ecological significance. Hydrobiologia 247: 121-132.

Drew, E.A. 1983. *Halimeda* biomass, growth rates and sediment generation on reefs in the central Great Barrier Reef Province. Coral Reefs 2: 101-110.

García-Sais, J., R. Appeldoorn, T. Battista, L. Bauer, A. Bruckner, C. Caldow, L. Carrubba, J. Corredor, E. Diaz, C. Lilyestrom, G. Garcia-Moliner, E. Hernández-Delgado, C. Menza, J. Morrell, A. Pait, J. Sabater, E. Weil, E. Williams, and S. Williams. 2008. The State of Coral Reef Ecosystems of Puerto Rico. pp. 75-116. In: J.E. Waddell and A.M. Clarke (eds.). The State of Coral Reef Ecosystems of the United States and Pacific Freely Associated States: 2008. NOAA Technical Memorandum NOS NCCOS 73. Silver Spring, MD. 569 pp.

Gardner, T., I.M. Cote, J.A. Gill, A. Grant, and A. Watkinson. 2003. Long-term region-wide declines in Caribbean corals. Science 2: 495-50.

Helfman, G.S. 1978. Patterns of community structure in fishes: summary and overview. Environ. Biol. Fish. 3:129-148.

Hoegh-Guldberg, O., P.J. Mumby, A.J. Hooten, R.S. Steneck, P. Greenfield, E. Gomez, C.D. Harvell, P.F. Sale, A.J. Edwards, K. Caldeira, N. Knowlton, C.M. Eakin, R. Iglesias-Prieto, N. Muthiga, R.H. Bradbury, A. Dubi, and M.E. Hatziolos. 2007. Coral reefs under rapid climate change and ocean acidification. Science 318: 1737-1742.

Hollander, M. and D.A. Wolfe. Nonparametric statistical methods. 2nd Edition. John Wiley and Sons, NY.

Hubbard, D.K., J.L. Sadd, A.I. Miller, L.P. Gill, and R.F. Dill. 1981. The production, transportation, and deposition of carbonate sediments on the insular shelf of St. Croix, U.S. Virgin Islands, West Indies. Laboratory Contrib. No. 76. 145 pp

Hubbard, D.K., K.M. Parsons, J.C. Bythell, and N.D. Walker. 1991. The effects of Hurricane Hugo on the reefs and associated environments of St. Croix, U.S. Virgin Islands - a preliminary assessment. J. Coast. Res. Spec. Issue 8:33-48.

Jeffrey, C.F.G. 2004. Benthic habitats, fish assemblages, and resource protection in Caribbean marine sanctuaries. Ph.D. Dissertation. The University of Georgia. Athens, GA.

Jeffrey, C.F.G., U. Anlauf, J. Beets, S. Caseau, W. Coles, A.M. Friedlander, S. Herzlieb, Z. Hillis-Starr, M. Kendall, V. Mayor, J. Miller, R. Nemeth, C. Rogers, and W. Toller. 2005. The State of Coral Reef Ecosystems of the U.S. Virgin Islands. pp 45-90. In: J. Waddell (ed.), The State of Coral Reef Ecosystems of the United States and Pacific Freely Associated States: 2005. NOAA Technical Memorandum NOS NCCOS 11. Silver Spring, MD. 522 pp.

Kendall, M.S., C.R. Kruer, K.R. Buja, J.D. Christensen, M. Finkbeiner, R.A. Warner, and M.E. Monaco. 2001. Methods used to map the benthic habitats of Puerto Rico and the U.S. Virgin Islands. NOAA Technical Memorandum NOS NCCOS 152. Silver Spring, MD. 45 pp. Online: http://ccma.nos.noaa.gov/products/biogeography/usvi_pr_mapping/manual.pdf (Accessed 14 May 2013)

Kendall, M.S., J.D. Christensen, and Z. Hillis-Starr. 2003. Multi-scale data used to analyze the spatial distribution of French grunts, *Haemulon flavolineatum*, relative to hard and softbottom in a benthic landscape. Environ. Biol. Fish. 66: 19-26.

Kumpf, H.E. and H.A. Randall. 1961. Charting the marine environments of St. John, U.S. Virgin Islands. Bull. Mar. Sci. 11(4):543-551.

McFarland, W.N., J.C. Ogden, and J.N. Lythgoe. 1979. The influence of light on the twilight migrations of grunts. Environ. Biol. Fish. 4:9-22.

Meyer, J.L., E.T. Schultz, and G.S. Helfman. 1983. Fish Schools: An Asset to Corals. Science 220: 1047-1049.

Miller, J., C. Rogers, and R. Waara. 2003. Monitoring the coral disease, plague type II, on coral reefs in St. John, U.S., Virgin Islands. Rev. Biol. Trop. 51:47-55.

Miller, J., E. Muller, C. Rogers, R. Waara, A. Atkinson, K.R.T. Whelan, M. Patterson, and B. Witcher. 2009. Coral disease following massive bleaching in 2005 causes 60% decline in coral cover on reefs in the US Virgin Islands. Coral Reefs 28(4):925-937.

Nagelkerken, I., G. van der Velde, M.W. Gorissen, G.J. Meijer, T. van't Hof, and C. den Hartog. 2000. Importance of mangroves, seagrass beds and the shallow coral reef as a nursery for important coral reef fishes, using a visual census technique. Est. Coast. Shelf Sci. 51: 31-44.

Parrish, J.D. 1989. Fish communities of interacting shallow-water habitats in tropical oceanic regions. Mar. Ecol. Prog. Ser. 58: 143-160.

Pittman, S.J., C. Caldow, S.D. Hile, and M.E. Monaco. 2007. Using seascape types to explain the spatial patterns of fish in the mangroves of SW Puerto Rico. Mar. Ecol. Prog. Ser. 348: 273-284.

Pittman, S.J., S.D. Hile, C.F.G. Jeffrey, C. Caldow, M.S. Kendall, M.E. Monaco, and Z. Hillis-Starr. 2008. Fish assemblages and benthic habitats of Buck Island Reef National Monument (St. Croix, US Virgin Islands) and the surrounding seascape: A characterization of spatial and temporal patterns. NOAA Technical Memorandum 71. Silver Spring, MD. 96 pp. Online: http://ccma.nos.noaa.gov/publications/biogeography/STC_LTR_Final_0309.pdf (Accessed 14 May 2014)

Pittman, S.J., S.D. Hile, C.F.G. Jeffrey, R. Clark, K. Woody, B.D. Herlach, C. Caldow, M.E. Monaco, R. Appeldoorn. 2010. Coral reef ecosystems of Reserva Natural La Parguera (Puerto Rico): Spatial and temporal patterns in fish and benthic communities (2001-2007). NOAA Technical Memorandum NOS NCCOS 107. Silver Spring, MD. 202 pp. Online: http://ccma.nos.noaa.gov/publications/nccostechmemo107.pdf (Accessed 14 May 2013)

Rogers, C.S., V. Garrison, and R. Grober-Dunsmore. 1997. A fishy story about hurricanes and herbivory: seven years of research on a reef in St. John, US Virgin Islands. pp. 555-560. In: H.A. Lessios and I.G. Macintyre (eds.), Proceedings of the 8th International Coral Reef Symposium, Vol. 1. Panama City, Panama. 1040 pp.

Rogers, C.S. and V.H. Garrison. 2001. Ten years after the crime: lasting effects of damage from a cruise ship anchor on a coral reef in St. John. Bull. Mar. Sci. 69:793-803.

Rogers, CS., W. Gladfelter, D. Hubbard, E. Gladfelter, J. Bythell, R. Dunsmore, C. Loomis, B. Devine, Z. Hillis-Starr, and B. Phillips. 2002. *Acropora* in the U.S. Virgin Islands: A Wake or an Awakening? pp. 99-122. In: A.W. Bruckner (ed.), Proceedings of the Caribbean *Acropora* Workshop: Potential Application of the US Endangered Species Act as a Conservation Strategy. NOAA Tech Memorandum NMFS OPR 24. Silver Spring, MD. 199 pp. Online: http://sero.nmfs.noaa.gov/pr/pdf/Bruckner2002.pdf (Accessed 14 May 2013)

Rogers, C.S., J. Miller, E.M. Muller, P.J. Edmunds, R.S. Nemeth, J. Beets, A. Friedlander, T.B. Smith, R. Boulon, C.F.G. Jeffrey, C. Menza, C. Caldow, N. Idrisi, B. Kojis, M. Monaco, A. Spitzack, E. Gladfelter, J. Ogden, Z. Hillis-Starr, I. Lundgren, W.B. Schill, I. Kuffner, L.L. Richardson, B. Devine, and J. Voss. 2008. Ecology of coral reefs in the US Virgin Islands. pp. 303-373. In: B.M. Riegl and R.E. Dodge (eds.), Coral Reefs of the USA. Coral Reefs of the World, Volume 1. Springer, Berlin. 806 pp.

Rothenberger, P., J. Blondeau, C. Cox, S. Curtis, W.S. Fisher, V. garrison, Z. Hillis-Starr, C.F.G. Jeffrey, E. Kadison, I. Lundgren, W.J. Miller, E. Muller, R. Nemeth, S. Paterson, C. Rogers, T. Smith, A. Spitzack, M. Taylor, W. Toller, J. Wright, D. Wusinich-Mendez and J. Waddell. 2008. The State of Coral Reef Ecosystems of the U.S. Virgin Islands. pp. 29-73. In: J.E. Waddell and A.M. Clarke (eds.), The State of Coral Reef Ecosystems of the United States and Pacific Freely Associated States: 2008. NOAA Technical Memorandum NOS NCCOS 73. Silver Spring. 569 pp.

Whelan, K.R.T., J. Miller, O. Sanchez, and M. Patterson. 2007. Impact of the 2005 coral bleaching event on *Porites porites* and *Colpophyllia natans* at Tektite Reef, US Virgin Islands. Coral Reefs 26:689-693.

Wilkinson, C. and D. Souter. 2008. Status of Caribbean coral reefs after bleaching and hurricanes in 2005. Global Coral Reef Monitoring Network, and Reef and Rainforest Research Centre. Townsville, Queensland, Australia. 152 pp.

Zar, J.H. 1999. Biostatistical Analysis, 4th edition. Prentice Hall. Englewood Cliffs, NJ. 929 pp.

Zitello, A.G., L.J. Bauer, T.A. Battista, P.W. Mueller, M.S. Kendall and M.E. Monaco. 2009. Shallow-Water Benthic Habitats of St. John, U.S. Virgin Islands. NOAA Technical Memorandum NOS NCCOS 96. Silver Spring, MD. 53 pp. Online: http://ccma.nos.noaa.gov/ecosystems/coralreef/benthic/ShallowBenHabStJohn_Zitello_etal_reduced_qual.pdf (Accessed 14 May 2013)

Chapter 4: Fish Communities Around St. John

Alan M. Friedlander[1], Lisa Wedding[2,3], Simon J. Pittman[4,5,6] and Chris F.G. Jeffrey[4,5]

4.1. INTRODUCTION

In the late 1950s Randall (1963) noted that the limited fringing reef area around the U.S. Virgin Islands (USVI) received nearly all of the fishing effort, and as a consequence the effects of overfishing were evident. Large predatory fishes such as groupers and snappers are now far less abundant, the relative abundance of herbivorous fishes has increased, individuals of many fish species are smaller, and some spawning aggregations have been decimated (Beets and Friedlander, 1992, 1999; Beets, 1997; Figure 4.1). In the 1960s, groupers and snappers dominated the landings in the USVI fishery but following the increased demand for fish with the tourism boom and technological changes in the fishery (larger boats, engines, and improved gear), fishers began to set more traps and target species like groupers and snappers, especially their spawning aggregations (Olsen and LaPlace, 1979; Beets and Friedlander, 1992). Although it is implied that commercial fishing is prohibited, Virgin Island National Park's (VIIS) enabling legislation allows for the "customary uses of or access" to park waters for fishing, including the use of traps of "conventional Virgin Islands design". When the park was first established, fishers usually set only a few, smaller traps but with the advent of outboard motors, line hauls, and larger fiberglass boats, fishermen now fish further offshore with a larger number of traps (Beets, 1997; Garrison et al., 1998).

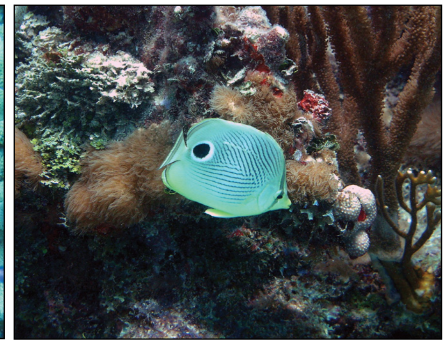


*Figure 4.1. Photos of (a) herbivorous fish Blue Tang (*Acanthurus coeruleus*) with* Porites porites *coral and* Lobophora *algae; and (b) Foureye Butterflyfish (*Chaetodon capistratus*), a common species observed in St. John, U.S. Virgin Islands (USVI). Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

The National Park Service (NPS) has supported reef fish research starting with the seminal work by John Randall from 1958-1961 which included: fisheries resources (Idyll and Randall, 1959), fish tagging (Randall 1962), population structure (Randall, 1963), fish grazing (Randall, 1965), food habits (Randall, 1967), and taxonomy (Randall, 1968). The Tektite Program in 1969 and 1970 involved scientists living in a saturation diving habitat at a depth of 17 m in Lameshur Bay, St. John for weeks at a time. Nine studies dealing with various aspects of the ecology of coral-reef fishes were carried out during Tektite I and II and examined activity patterns, behavior, bio-acoustics, and herbivory (Collette and Earle, 1972). In 1983, the Virgin Islands Resource Management Cooperative, supported primarily by the NPS and under the direction of Island Resources Foundation, produced a series of reports from 1986-1988 that provided maps and data that are the basis of many ongoing projects in VIIS. Subsequent investigations of fish resources and fisheries investigations have been conducted around St. John, ranging from fisheries assessments, reef fish monitoring, hurricane impacts, and declining resources (Friedlander and Beets, 2008). In 1992/1993 and 1999/2002, the NOAA Southeast Area Monitoring and Assessment Program for the Caribbean (SEAMAP-C) conducted a fishery independent

[1] U.S. Geological Survey
[2] University of California, Santa Cruz
[3] Fisheries Ecology Division, Southwest Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration
[4] Center for Coastal Monitoring and Assessment, National Centers for Coastal Ocean Science, National Ocean Service, National Oceanic and Atmospheric Administration
[5] Consolidated Safety Services, Inc.
[6] University of the Virgin Islands

survey of reef fish populations using traditional fishing gear (traps, and hook and line) to assess the status and change in marine resources. Sampling was carried out south of St. John covering 166 km^2 and extending 11 km from the shore out to the shelf edge (Whiteman et al., 2005). The shallow water reef fish fishery in the U.S. Caribbean primarily targets grunts (Haemulidae), groupers (large-bodied Serranidae), goatfish (Mullidae), parrotfish (Scaridae) and snappers (Lutjanidae). Triggerfish, squirrelfish, hogfish, porgies and trunkfish are also caught and represent approximately 15% of the total catch (CFMC, 1985; Appendix D; Figure 4.2). Declines in catch between 1992 and 2002 were evident for Yellowmouth Grouper (*Mycteroperca interstitialis*), White Grunt (*Haemulon plumierii*), Nassau Grouper (*Epinephelus striatus*), Dog Snapper (*Lutjanus jocu*), Schoolmaster Snapper (*Lutjanus apodus*) and Yellowtail Snapper (*Ocyurus chrysurus*; Appendix D). In contrast, increases in biomass were evident for Graysby (*Cephalopholis cruentata*), Coney (*Cephalopholis fulva*) and Queen Triggerfish (*Balistes vetula*).

*Figure 4.2. Blackbar Soldierfish (*Myripristis jacobus*) a species of the Squirrelfish Family in the St. John, USVI study area. Credit: NOAA//NOS/NCCOS/CCMA/ Biogeography Branch.*

Previous studies of fish distributions and diversity around St. John have either been limited to selected coral reef locations or specific habitat types. Such data do not provide sufficient spatial information to develop comprehensive management plans across the complex multiple-habitat seascapes that characterize VIIS. In response, NOAA's National Ocean Service (NOS), National Centers for Coastal Ocean Science (NCCOS), Center for Coastal Monitoring and Assessment Biogeography Branch (CCMA-BB) in partnership with NPS conducted a decade-long study to characterize spatial patterns and monitor temporal changes in fish assemblages inside and outside MPAs in St. John. This study (analyzed here) was designed to provide information on fish community structure and change across all components of the seascape and to examine inter-annual and multi-year trends for fish species and assemblage biomass, abundance and diversity.

This section of the report focuses on the spatial distribution of fish species and assemblage metrics (i.e., composition, species richness, biomass, abundance of assemblages) and the temporal patterns (2001-2009) across mosaics of habitat types in the study area around St. John, USVI. 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 assemblages are highly heterogeneous in time and space and can also function as indicators of ecosystem integrity and health (Mora et al., 2011). Examination of fish assemblage 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.

4.2. FIELD SURVEY METHODS

4.2.1. 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; Appendix A). 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 (see Chapter 3, Figure 3.5). Data from 1,048 sites collected between 2001 and 2009 were analyzed for this study (Table 4.1, Figure 4.3). Of these, 379 were conducted inside VIIS and 298 outside the park (Table 4.1, Figure 4.3). In addition, 371 surveys were conducted as part of a study to compare Virgin Islands Coral Reef National Monument (VICR) sites (n=222) in Coral Bay to adjacent sites in the territorial waters of Coral Bay (n=149). Additional fish surveys were conducted inside and outside of the VICR along the mid-shelf reef approximately 6 km offshore (Monaco et al., 2007, 2009; Boulon et al., 2008; see Appendix B). A complete list of species recorded on transects and their associated metrics can be found in Appendix E. For a detailed description of CCMA-BB's fish census survey methods see Appendix A.

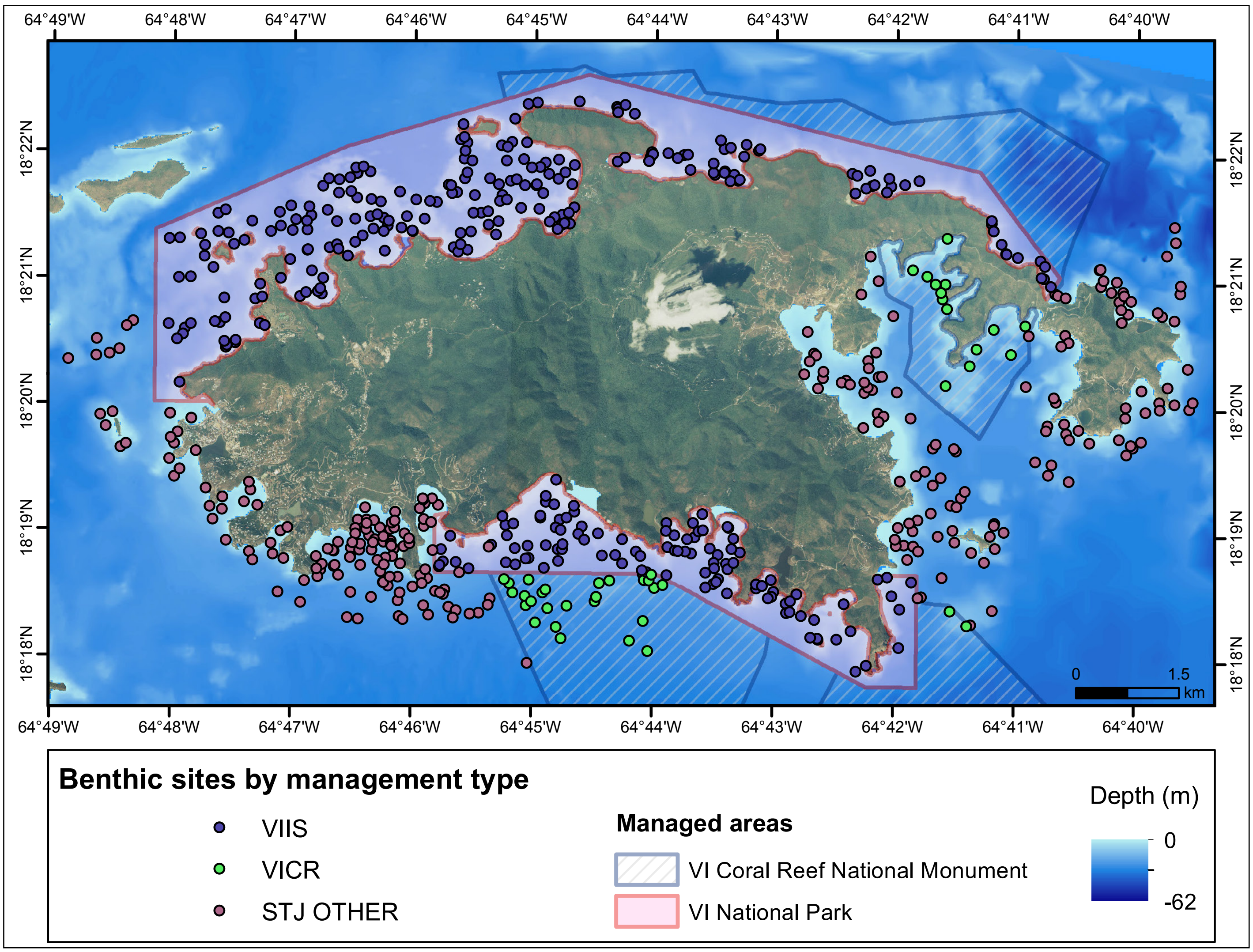


Figure 4.3. NOAA's National Centers for Coastal Ocean Science (NCCOS) Center for Coastal Monitoring and Assessment Biogeography Branch (CCMA-BB) Coral Reef Ecosystem (CREM) project survey sites around St. John conducted from 2001-2009. Survey sites were restricted to water less than 35 m depth. Source: K. Stamoulis (University of Hawaii).

Table 4.1. Number of sites surveyed for fish by year inside and outside National Park Service (NPS) parks in St. John, U.S. Virgin Islands (USVI). VIIS = Virgin Islands National Park; VICR = Virgin Islands Coral Reef National Monument.

Fish Sampling	Management	2001	2002	2003	2004	2005	2006	2007	2008	2009	Grand Total
VIIS study	Inside VIIS	24	45	50	53	45	41	37	34	50	379
	Outside VIIS & VICR	31	45	21	37	28	33	36	40	27	298
VIIS Subtotal		*55*	*90*	*71*	*90*	*73*	*74*	*73*	*74*	*77*	*677*
Coral Bay	Inside VICR	3		19	24	38	33	36	36	33	222
	Outside VICR & VIIS			18	6	25	25	25	25	25	149
Coral Bay Subtotal		*3*	*0*	*37*	*30*	*63*	*58*	*61*	*61*	*58*	*371*
Grand Total		**58**	**90**	**108**	**120**	**136**	**132**	**134**	**135**	**135**	**1048**

4.2.2. Data analysis

Assessing differences in univariate metrics inside versus outside VIIS

Differences in univariate community metrics, as well as individual species/group metrics inside versus outside VIIS were tested using parametric and non-parametric tests as appropriate. Designations of habitat type (colonized hardbottom, seagrasses and unvegetated sediments) were collected by benthic habitat observers**.** (Table 4.1)

Diversity was measured using the Shannon Index (H'; Equation 1). In this way, the diversity measure incorporates richness, commonness and rarity. Although, the Shannon Index has been shown to be an effective discriminator of community structure it is not independent of sample size (Magurran, 1988). Taxonomic indices, on the other hand are considered to be significantly less influenced by sample size than the conventional species richness, evenness and diversity indices (Warwick and Clarke, 1995) and, therefore, more appropriate for any comparative studies with unbalanced sampling effort (Clarke and Warwick, 1998).

$$H' = - S_i \, p_i \, (\log_e p_i) \qquad \text{(Equation 1)}$$

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

<u>Taxonomic Indices</u>

Samples may differ in the way assemblages are composed at the genus, family, order, class and phylum levels of the standard Linnean taxonomic hierarchy. For example, species diversity may be similar between two samples, yet one may support several species belonging to the same family, while the other may support several species, all belonging to different families and even different classes, orders, etc. Quantitative taxonomic diversity indices therefore provide an additional dimension of information that is likely to be more closely linked to functional diversity (Clarke and Warwick, 1999). The importance of this measure of diversity is that families, orders, etc. as opposed to species, represent a greater variety of fundamentally different body plans and life histories.

As such Taxonomic diversity (Δ; Warwick and Clarke, 1995; Equation 2) was measured for all samples. Fish were distinguished at four taxonomic levels: species, genus, family and class. Samples were grouped by habitat type as determined by benthic habitat observers and by management domain as determined by the mapped strata (e.g., inside and outside VIIS).

$$\Delta = \frac{\Sigma\Sigma_{i<j} w_{ij} x_i x_j + \Sigma_i 0.\, x_i (x_i - 1)/2}{\Sigma\Sigma_{i<j} x_i x_j + \Sigma_i x_i (x_i - 1)/2} \qquad \text{(Equation 2)}$$

Letting x_i denote the presence or absence of the *i*th species and the W_{ij} the "distinctness weight" given to the path length linking species *i* and *j* in the hierarchical classification, then taxonomic diversity (D) is defined simply as the average (weighted) path length between every pair of individuals. The null second term in the numerator has been included to emphasize that the weight for the path linking individuals of the same species is taken to be zero.

Assessing differences in community composition inside versus outside

Differences and similarities in the species composition of communities between samples (often referred to as assemblage or community structure) were examined using a species biomass by site data matrix. Biomass estimates were derived from calculated live wet weight. Live wet weight (W) was derived from the visually estimated mean fork length (FL) for each size class for each species using the relation W = a(FL)b. Values of the fitting parameters a and b for each species were derived from Bohnsack et al. (1986) and the FishBase web site (http://fishbase.org/). For species not in these databases, estimates from available literature on the species or congeners were used. Biomass of all fishes recorded in all censuses was obtained by multiplying the mean live wet weight for each size class for each species by the total number of individuals observed in that size class. Samples with zero fish were removed from the data matrix. Infrequently observed species, with extreme outlying biomass were removed, including small-bodied pelagic schooling fish (e.g., Clupeidae, Antherinidae, etc.) and large-bodied broad ranging species (e.g., sharks, rays, barracuda). All analyses of numerical abundance and biomass excluded the masked goby (*Coryphopterus personatus*) because they were ubiquitous and their large numbers in samples (1,000's) masked trends in the rest of the fish assemblage. The masked goby was included in calculations of species richness. Masked gobies are most abundant in reef structure with high topographic complexity and may be an important indicator of reef condition in the USVI, but this species contributes negligibly to biomass estimates because of its small size (<3 cm). The matrix was ln(x+1) transformed to ensure that intermediate biomass species, in addition to the high biomass 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 (Equation 3).

$$S'_{jk} = \left[1 - \frac{\sum_{i=1}^{n} X_{ij} - X_{ik}}{\sum_{i=1}^{n} X_{ij} + X_{ik}} \right] \qquad \text{(Equation 3)}$$

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. Sample sites were assigned a factor representing a dominant habitat type (e.g., colonized hardbottom, seagrasses or sand) and a management domain (e.g., inside or outside VIIS). Factors were used to identify pairs of treatments in order to test for significant differences using Analysis of Similarities (ANOSIM), a multivariate version of Analysis of Variance (ANOVA; Primer v5; Clarke and Warwick, 1994), and for visual examination of similarities in patterns between sites using a non-metric multidimensional scaling plot (nMDS). In addition, Similarity Percentages (SIMPER) were calculated and used to identify the species which contributed most to the differences between treatments (Primer v5; Clarke and Warwick, 1994).

Where species groups were used, herbivores included all species that were important consumers of marine algae; piscivores included all fish that were important predators of fish; snapper included all Lutjanidae species; groupers included all commercially harvested Serranidae species; and grunts included all Haemulidae. Fishes were categorized as juveniles/subadults based on length at maturity information provided by García-Cagide et al. (1994) and FishBase (http://www.fishbase.org, version 11/2007), whereby, juveniles/subadults were fish with lengths less than the mean length at maturity and the remainder were considered as adults (Table 4.2). If the mean length at maturity was 14 cm then size classes <5 and 5-10 cm were considered juvenile/subadult. Where length at maturity was unknown, 1/3 of maximum adult size was used to segregate juveniles/subadults from adults. For mapping of juvenile and adult distribution all samples were used from 2001 to 2009.

Table 4.2. Length at first maturity estimates used to determine approximate size classes for juvenile/subadult and adult fish. Estimates are derived from data held by FishBase (http://www.fishbase.org, version 11/2007).

Species	Mean length at first maturity, L_m (cm)	Juvenile/subadult size class (cm)
Acanthurus bahianus	15.5	≤15
Acanthurus coeruleus	unknown	≤10
Balistes vetula	25	≤20
Cephalopholis fulva	16	≤15
Epinephelus guttatus	25	≤20
Halichoeres bivittatus	unknown	≤10
Haemulon flavolineatum	16	≤15
Haemulon plumierii	19	≤15
Haemulon sciurus	18.5	≤15
Holacanthus tricolor	17.4	≤15
Lutjanus apodus	25	≤20
Lutjanus griseus	31	≤20
Ocyurus chrysurus	24.5	≤20
Scarus iseri	unknown	≤10
Sparisoma aurofrenatum	unknown	≤10
Sparisoma viride	16.3	≤15
Thalassoma bifasciatum	unknown	0-5

Species habitat associations

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.

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.

4.3. RESULTS

4.3.1. Spatial distribution patterns and species-habitat associations

Fish assemblages

Fish metrics were selected from several levels of biological organization to include measures of the entire assemblage, trophic groupings, key fish families and species. Samples were surveyed across a wide range of soft and hardbottom habitat types, but for some analyses variables were grouped and synthesized across only major habitat types (hardbottom [HB], submerged aquatic vegetation [SAV] and sand) both inside and outside of VIIS.

Fish assemblage metrics

Species richness averaged 21.9 (± 7.5 standard deviation [SD]) on hardbottom, 8.5 (± 5.3 SD) on SAV and 5.9 (± 5.0 SD) on sand (Figure 4.4a). Significant differences were found in species numbers among all three habitat types ($p<0.001$; HB>SAV>sand). Richness between inside VIIS and outside (STJ) was not significantly different on hardbottom ($p=0.4$) or SAV ($p=0.98$), but was significantly higher (37%) on sand (or unvegetative sediment) outside VIIS ($p=0.025$).

Biomass averaged 5.2 kg/100 m^2 on hardbottom, 1.9 kg/100 m^2 on SAV, and 0.6 kg/100 m^2 on sand ($p<0.001$; HB>SAV>sand; Figure 4.4b). Overall, fish biomass was slightly greater (by 16%) inside VIIS in hardbottom habitats, although this pattern was not significant ($p=0.53$). Although biomass in seagrass habitat outside VIIS was 3.5 times higher than inside, the large overall coefficient of variation (COV=4.6) within this habitat resulted in no significant difference in biomass between these management units ($p=0.83$). Biomass in unvegetated sediments was similar between management strata ($p = 0.67$;Figure 4.4b).

Numerical abundance averaged 1.6 individuals/m^2 on hardbottom habitats, 0.6 individuals/m^2 on SAV and 0.5 individuals/m^2 on sand ($p<0.001$; HB>SAV>sand; Figure 4.5a). Numerical abundance on hardbottom was significantly greater (by 17%) outside VIIS compared with inside ($p=0.05$; Figure 4.5a). The number of individuals on sand habitat was nearly twice as high outside versus inside VIIS ($p=0.026$). SAV showed no difference in number of individuals between management units ($p=0.43$).

Shannon Diversity within hardbottom habitats averaged 2.3 (± 0.4 SD) and was 60% higher than on SAV and more than two times higher than on sand habitats (Figure 4.5b; $p<0.001$; HB>SAV>sand). Diversity did not differ significantly between management units for any of the three major habitat types surveyed (all $p>0.05$).

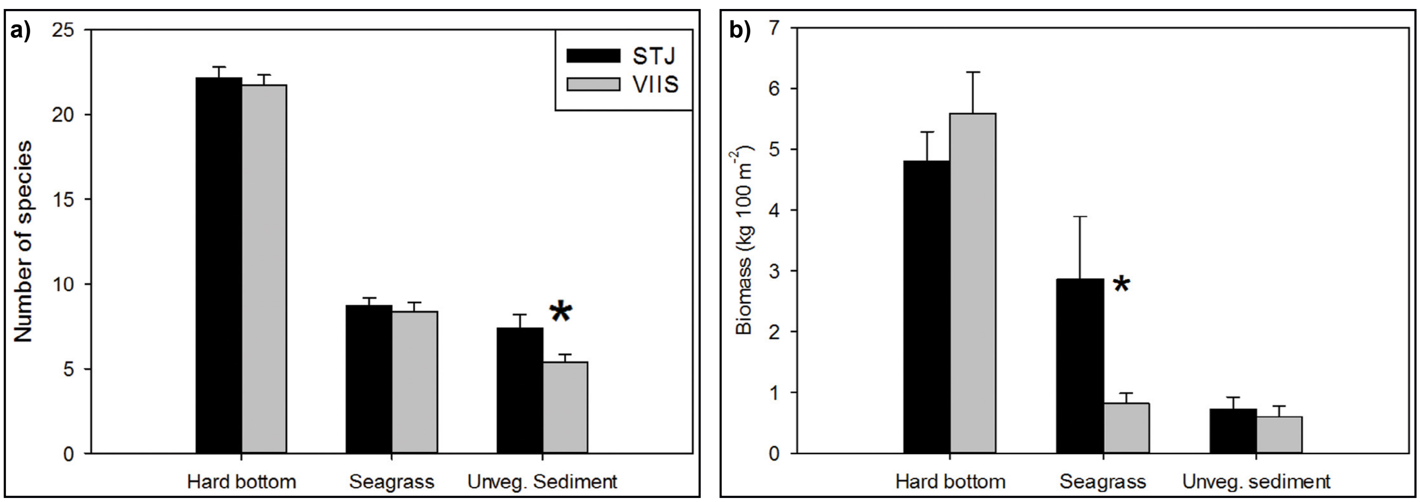


Figure 4.4. Fish assemblage characteristics among major habitat types and between management strata: a) species richness and b) biomass (kg/100 m^2). Asterisks (*) indicates a large coefficient of variation (COV) such that the differences between inside Virgin Islands National Park (VIIS) and outside VIIS (STJ) were not statistically different.

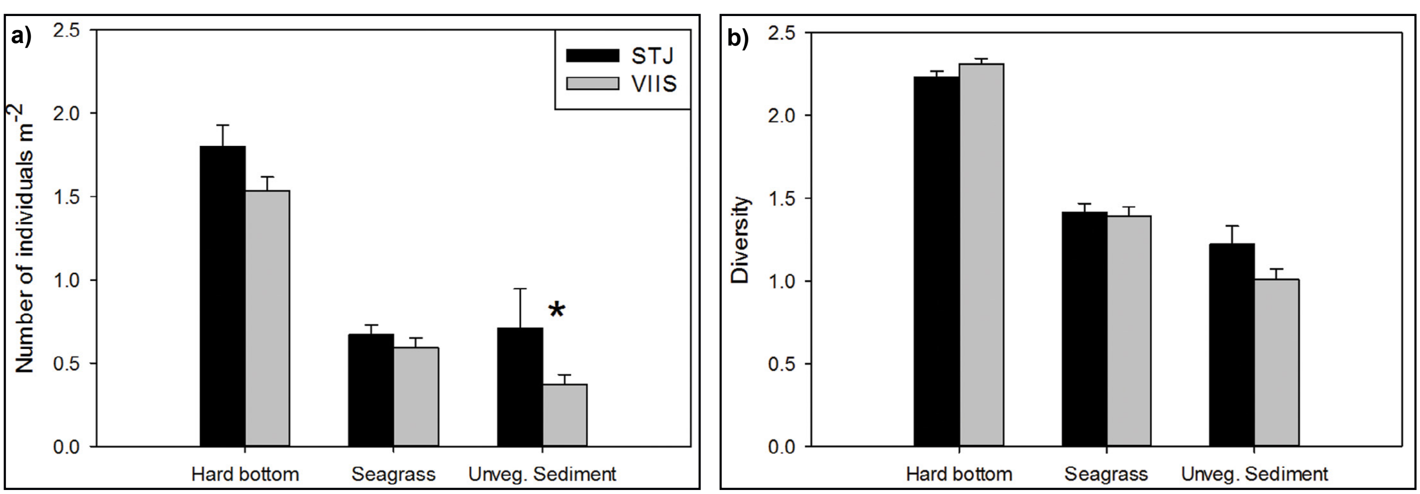


Figure 4.5. Fish assemblage characteristics among major habitat types and between management strata: a) number of individuals/100 m^2 and b) Shannon-Weiner Diversity. Asterisks (*) indicates a statistically significant difference between inside and outside VIIS.

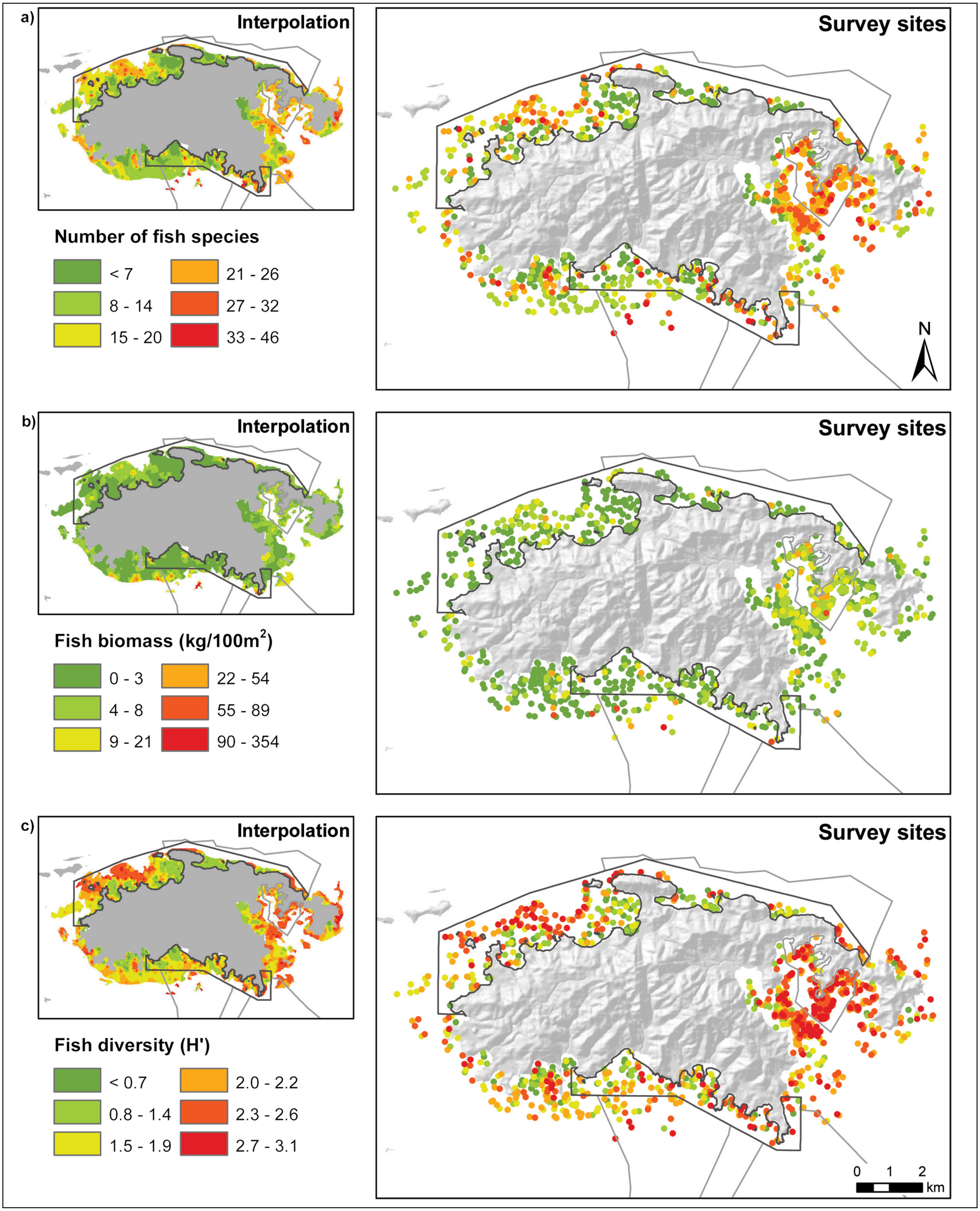


Figure 4.6. Spatial distribution of fish metrics around St. John, USVI: a) number of fish species, b) fish biomass and c) fish diversity.

Spatial patterns in fish assemblage metrics

A total of 227 unique fish taxa were recorded at 677 sites around St. John, excluding within VICR. Of the sites with high species richness (>30 species, n=46), 63% occurred within VIIS. Fish species richness and diversity were highest along the east shore, within Coral Bay and along the north shore between Mary's Point and the Durloes (Figure 4.6a,c). The largest continuous area of high fish species richness and high fish diversity occurred within Coral Bay (Figure 4.6a,c). Fish biomass was generally evenly distributed inside and outside of VIIS, with a small number of high biomass sites on hardbottom habitats on the patch reefs off of White Cliffs and in deeper areas off Reef Bay (Figure 4.6b).

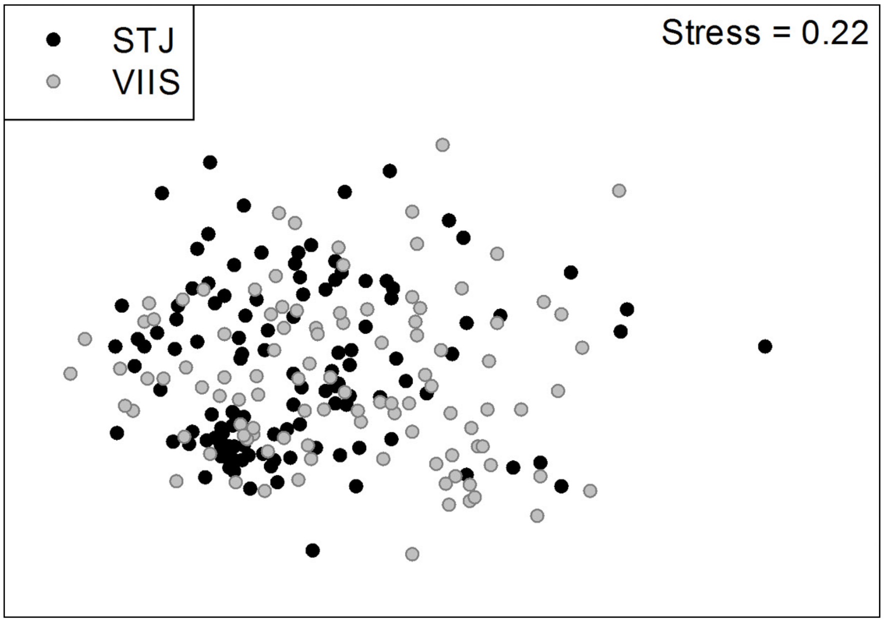


Figure 4.7. Non-metric multidimensional scaling (nMDS) ordination based on between site similarities in fish assemblage structure among habitat types. Results based on numerical abundance data.

Table 4.4. Pairwise Analysis of Similarities (ANOSIM) test comparisons between among habitat types. SAV = submerged aquatic vegetation, HB = hardbottom. The R statistic represents pairs of habitats that are well separated (R > 0.75), overlapping but clearly different (R > 0.5), or barely separable at all (R < 0.25). Stress = 0.19, Global R = 0.451.

Group	R	P	Interpretation
SAV, Sand	0.12	0.001	barely separable (R<0.25)
SAV, HB	0.49	0.001	overlapping but clearly different (R>0.5)
Sand, HB	0.64	0.001	overlapping but clearly different (R>0.5)

Fish assemblage composition

nMDS and ANOSIM tests indicated that fish assemblage structure between hard and soft habitat types showed moderate separation (Figure 4.7; Table 4.4). Hardbottom sites had high concordance while seagrass and sand sites were highly discordant and showed considerable overlap with each other (Figure 4.7; Table 4.4). Considerable overlap was found in fish assemblage structure based on numerical abundance between management regimes inside and outside VIIS. ANOSIM R values were very low (e.g., high similarity) between management regimes for all three habitat types (Table 4.5; Figure 4.8).

Table 4.5. Global ANOSIM test between inside and outside VIIS within each major habitat type.

Habitat	Stress	R	P	Interpretation
Hard bottom	0.20	0.015	0.090	barely separable (R<0.25)
SAV	0.19	0.041	0.002	barely separable (R<0.25)
Sand	0.19	0.097	0.001	barely separable (R<0.25)

Patterns based on biomass showed similar results to those for numerical abundance. Pairwise comparisons between hardbottom habitat types revealed that fish assemblages were barely separable, with substantial overlap among habitat types. (Table 4.6). However, the fish assemblage on uncolonized bedrock showed relatively high dissimilarity with assemblages on colonized pavement with sand channels and individual patch reefs (Table 4.6).

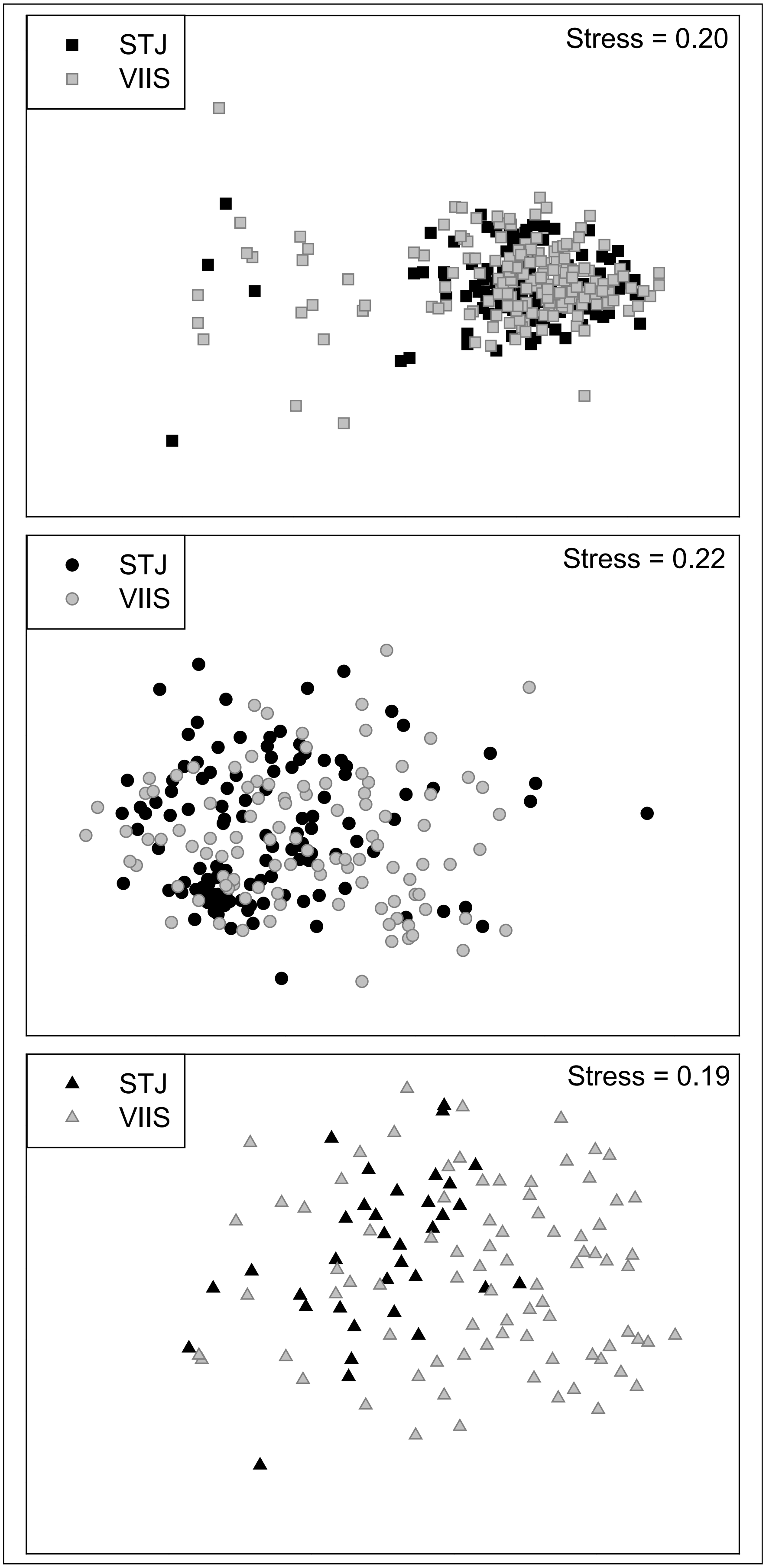


Figure 4.8. nMDS ordination based on between-site similarities in fish assemblage structure between management regimes. Results based on numerical abundance data and analyzed separately for a) hardbottom, b) SAV and c) sand. SAV = subaquatic vegetation.

Table 4.6. ANOSIM pairwise comparisons between hardbottom subhabitat types. Global R = 0.26, p = 0.046. The R statistic represents pairs of subhabitats that are well separated (R>0.75), overlapping but clearly different (R>0.5; dark grey), or barely separable at all (R<0.25; light grey).

Groups	Statistic	P
Colonized Pavement with Sand Channels, Uncolonized Bedrock	0.699	0.001
Patch Reef (Individual), Uncolonized Bedrock	0.613	0.008
Linear Reef, Uncolonized Bedrock	0.487	0.001
Colonized Pavement with Sand Channels, Patch Reef (Individual)	0.447	0.006
Colonized Bedrock, Patch Reef (Individual)	0.434	0.005
Colonized Pavement with Sand Channels, Colonized Bedrock	0.371	0.001
Colonized Pavement with Sand Channels, Patch Reef (Aggregated)	0.345	0.001
Colonized Bedrock, Scattered Coral/Rock in Unconsolidated	0.275	0.001
Linear Reef, Scattered Coral/Rock in Unconsolidated	0.258	0.005
Linear Reef, Patch Reef (Aggregated)	0.256	0.010
Patch Reef (Aggregated), Colonized Bedrock	0.247	0.003
Colonized Pavement with Sand Channels, Scattered Coral/Rock in Unconsolidated	0.239	0.003
Linear Reef, Patch Reef (Individual)	0.235	0.080
Linear Reef, Colonized Bedrock	0.194	0.001
Colonized Pavement, Uncolonized Bedrock	0.186	0.102
Colonized Pavement, Patch Reef (Aggregated)	0.110	0.130
Colonized Pavement, Scattered Coral/Rock in Unconsolidated Sediment	0.102	0.122
Scattered Coral/Rock in Unconsolidated, Uncolonized Bedrock	0.098	0.222
Colonized Pavement, Patch Reef (Individual)	0.071	0.300
Linear Reef, Colonized Pavement with Sand Channels	0.021	0.350
Colonized Pavement, Linear Reef	0.004	0.377
Patch Reef (Aggregated), Scattered Coral/Rock in Unconsolidated	0.004	0.373
Colonized Pavement, Colonized Bedrock	-0.005	0.497
Patch Reef (Aggregated), Uncolonized Bedrock	-0.005	0.405
Patch Reef (Individual), Scattered Coral/Rock in Unconsolidated	-0.013	0.466
Colonized Bedrock, Uncolonized Bedrock	-0.028	0.600
Patch Reef (Aggregated), Patch Reef (Individual)	-0.080	0.659
Colonized Pavement, Colonized Pavement with Sand Channels	-0.132	0.996

Fish assemblage metrics by trophic groupings

Mean herbivore biomass was highest on hardbottom habitats with no significant difference detected between inside and outside VIIS (Figure 4.9). Overall piscivore biomass was low throughout the study area, but was higher (but not significantly) inside VIIS than outside on hardbottom habitats. Mean piscivores biomass was higher outside VIIS than inside in SAV and sand habitats. Mean invertivore biomass was highest on hardbottom habitats, and was greater inside VIIS than outside in this habitat (Figure 4.9a). In contrast, invertivore biomass in SAV was significantly ($p<0.05$) higher outside VIIS compared to inside. Mean planktivore biomass was higher outside VIIS than inside on SAV but the overall biomass was low relative to hardbottom habitats (Figure 4.9).

The spatial distribution of herbivore biomass revealed areas of high biomass around Coral Bay, off Eagle Shoal, and along the north shore from Johnson's Reef to Cinnamon Bay (Figure 4.10a). Planktivore biomass was low overall, with a few high spots off the south shore along White Cliffs and the southeast portion of the island from Johns Folly to LeDuc (also referred to as Leduck; Figure 4.10b). Piscivore biomass was low overall and biased by a few high biomass hotspots created by large-bodied fish predators, primarily nurse sharks. No clear pattern of piscivore biomass appears (Figure 4.10c). Biomass of invertivores was highest in deeper water south of Rendezvous and Reef Bays along the south shore of St. John (Figure 4.10d).

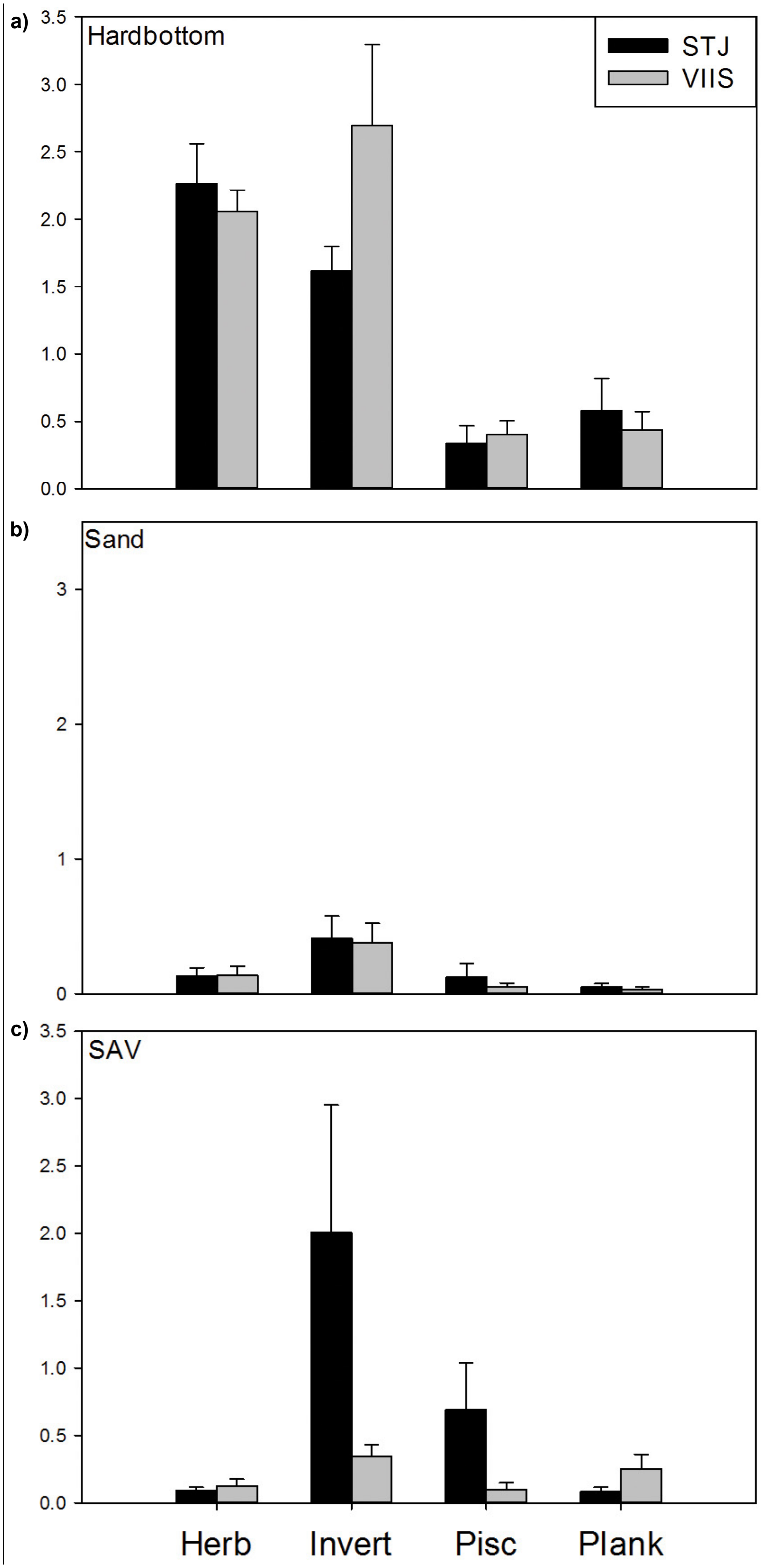


Figure 4.9. Trophic biomass (kg/100 m^2) within major habitat types and between management regimes, including values and means and standard error (SE): a) Hardbottom, b) sand and c) SAV. Herb=herbivore, Invert=invertivore, Pisc=piscivore, Plank=planktivore.

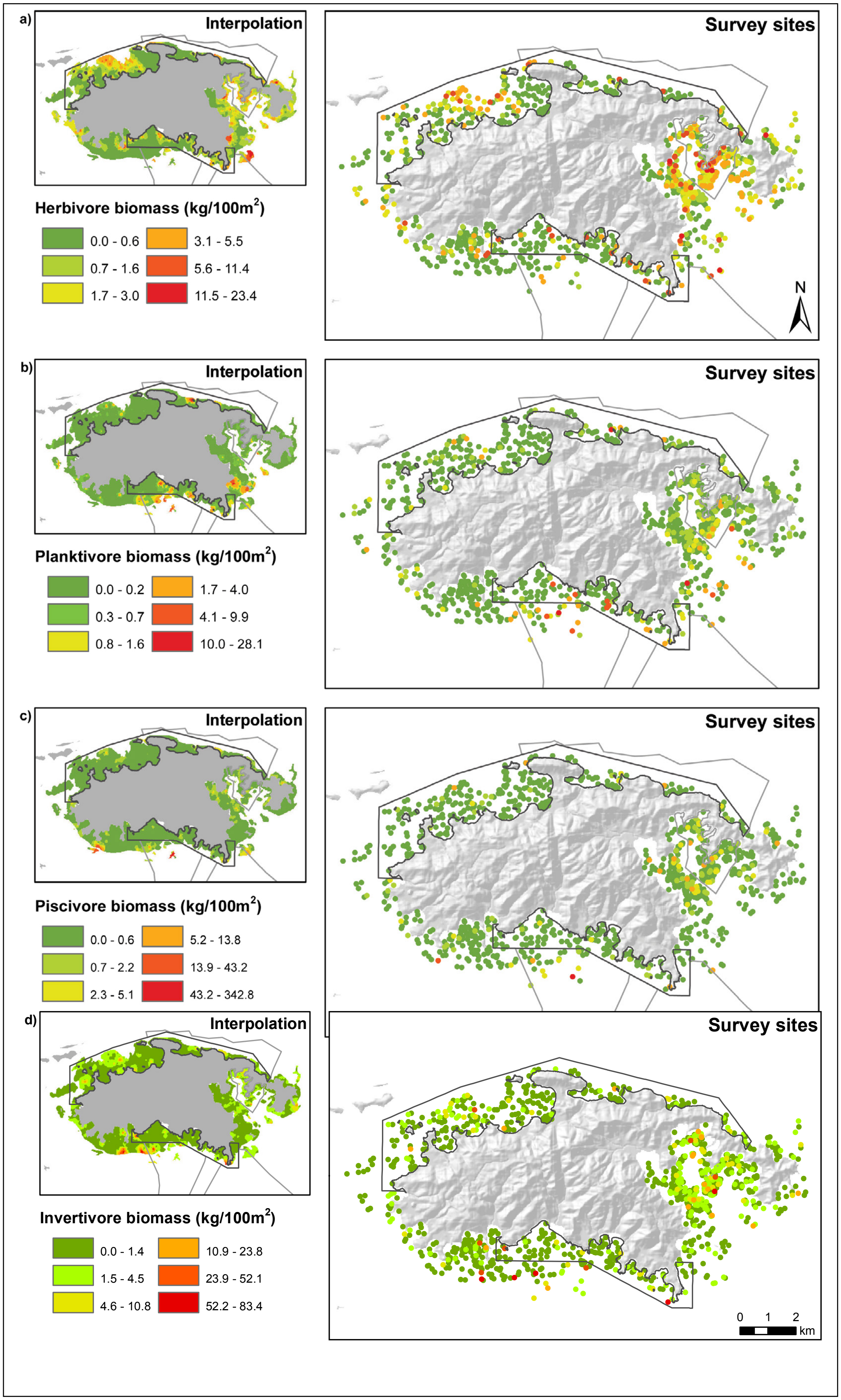


Figure 4.10. Spatial distributions of fish biomass (kg/100 m²) for: a) herbivores, b) planktivores, c) piscivores and d) invertivore.

Fish assemblage metrics by family

Wrasses (Family Labridae) were the most abundant and common family among all habitats, followed by parrotfishes (Scaridae), damselfishes (Pomacentridae), surgeonfishes (Acanthuridae) and grunts (Haemulidae; Table 4.7). Together these five families accounted for 74% of total fish biomass around St. John. Parrotfishes accounted for 17% of total assemblage biomass and occurred in 78% of all surveys. Stingrays (Dasyatidae) comprised 16% of the total biomass, but occurred on <2% of all surveys (Table 4.8).

Table 4.7. Fish numerical abundance (number of individuals/100 m^2) by family among all habitat types combined. COV= coefficient of variation.

Family	Abundance (no./m^2)	SD	N	COV	Frequency	Percent total
Labridae - wrasses	0.28	0.38	565	1.4	83.46%	26.88%
Scaridae - parrotfishes	0.16	0.23	530	1.4	78.29%	15.92%
Pomacentridae - damselfishes	0.16	0.36	378	2.3	55.83%	15.25%
Acanthuridae - surgeonfishes	0.09	0.21	353	2.4	52.14%	8.67%
Haemulidae - grunts	0.08	0.59	186	7.7	27.47%	7.45%
Serranidae – seabasses/groupers	0.07	0.17	393	2.5	58.05%	6.45%
Gobiidae - gobies	0.07	0.15	428	2.3	63.22%	6.40%
Carangidae - jacks	0.03	0.22	102	8.2	15.07%	2.59%
Lutjanidae - snappers	0.02	0.07	285	3.1	42.10%	2.12%
Chaetodontidae - butterflyfishes	0.01	0.02	209	2.1	30.87%	0.93%

Table 4.8. Fish biomass (kg/100 m^2) by family among all habitat types combined.

Family	Biomass (kg/100 m^2)	SD	N	COV	Frequency	Percent total
Scaridae - parrotfishes	0.52	1.02	530	2.0	78.29%	16.67%
Dasyatidae - stingrays	0.49	4.92	13	10.0	1.92%	15.84%
Acanthuridae - surgeonfishes	0.39	1.22	353	3.1	52.14%	12.45%
Lutjanidae - snappers	0.28	1.30	284	4.7	41.95%	8.89%
Ginglymostomatidae – nurse sharks	0.20	3.28	6	16.8	0.89%	6.29%
Carangidae - jacks	0.16	1.70	102	10.8	15.07%	5.06%
Serranidae – seabasses/groupers	0.14	0.33	393	2.3	58.05%	4.57%
Labridae - wrasses	0.14	0.28	565	2.1	83.46%	4.34%
Haemulidae - grunts	0.12	0.51	186	4.2	27.47%	3.87%
Sphyraenidae - barracudas	0.11	0.85	18	7.7	2.66%	3.55%

Wrasses were the most numerically dominant family in all habitat types (Figure 4.11a). Damselfishes were the next most common family on hardbottom followed by parrotfishes, and surgeonfishes. In SAV habitat, seabasses and groupers (Serranidae) were the second most numerous families in SAV, however nearly 75% of these individuals were composed of small *Serranus* and *Hypolpectrus* species. Parrotfishes were also common in SAV and consisted mostly of small juveniles (<10 cm). Small grunts and gobies (Gobiidae) were most commonly found in sand habitat.

Parrotfishes dominated hardbottom habitats by weight (21% of total), followed by surgeonfishes (16%), and snappers (10%, Figure 4.11b). Nurse sharks (Ginglymostomatidae) were the next most important by weight (9% of total) on hardbottom represented by 15 Nurse Sharks (*Ginglymostoma cirratum*) and as a result had high variance associated with the estimate. The Southern Stingray (*Dasyatis americana*) comprised 49% of the biomass in SAV, 29% in sand, and 5% in hardbottom habitats, however only 15 individuals were observed of which eight (53%) occurred in SAV.

Individual species

A total of 227 taxa from 55 families were identified during visual surveys around St. John. The Blue Tang (*Acanthurus coeruleus*) was the most dominant species based on the Index of Relative Dominance (frequency occurrence x relative abundance; IRD), ranking 2nd in total biomass, 9th in total numbers, and occurring on 37% of all transects. The Ocean Surgeonfish (*Acanthurus bahianus*) ranked 2nd in IRD, occurred in 47% of all transects and ranked 6th overall in both numbers and biomass (Table 4.9). The 3rd most dominant species based on IRD was the Redband Parrotfish (*Sparisoma aurofrenatum*) which ranked 5th in total biomass, 10th in total numbers, and occurred in 44% of all transects.

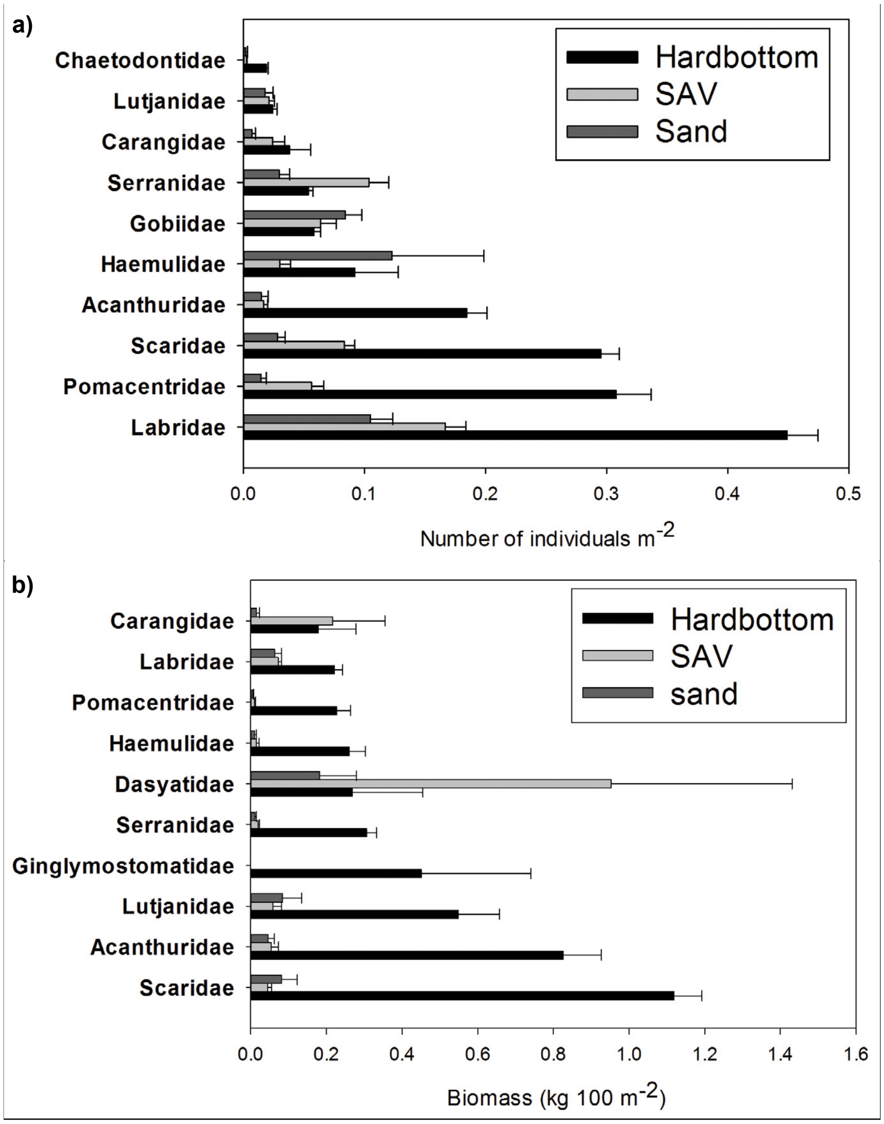


Figure 4.11. Density of dominant fish families by major habitat type: a) numerical abundance (number of individuals/100 m^2) and b) biomass (kg/100 m^2).

The most frequently observed species was the Slippery Dick (*Halichoeres bivittatus*) which occurred in 55% of all transects and ranked 2nd in numerical abundance (Table 4.9). Bluehead Wrasse (*Thalassoma bifasciatum*) were the most numerically dominant species and occurred in 40% of all transects (Figure 4.12). Striped Parrotfish (*Scarus iseri*) ranked 3rd in numerical abundance (6% of total) but ranked 9th and only accounted for 3% of total biomass as a result of the large schools of juveniles and subadults that dominate the distribution of this species around St. John.

*Figure 4.12. Photo of the numerically dominant fish species Bluehead Wrasse (*Thalassoma bifasciatum*), both terminal and juvenile/initial stages, in St. John, USVI. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

Select species of special interest to NPS, those that were potentially threatened by overfishing, and those that were dominant components of the fish community across the region were examined in further detail (Table 4.10). These included two groupers, four snappers, two grunts, two surgeonfishes, two parrotfishes, and one triggerfish representing a wide range of sizes and feeding groups.

Table 4.9. Numerical (number of individuals/100 m^2) and biomass (kg/100 m^2) densities of the top 20 species observed across all habitats and ordered by Index of Relative Dominance (IRD = % biomass x frequency of occurrence).

Species	Common Name	Numbers (#/100 m^2)	Number (%)	Number rank	Biomass (kg/100 m^2)	Biomass (%)	Biomass rank	N	Fre-quency	IRD
Acanthurus coeruleus	Blue Tang	4.03 (1.44)	3.87	9	0.21 (0.94)	6.60	2	252	37.22	246
Acanthurus bahianus	Ocean Surgeonfish	4.29 (0.93)	4.13	6	0.14 (0.37)	4.47	6	320	47.27	211
Sparisoma aurofrenatum	Redband Parrotfish	3.92 (0.68)	3.77	10	0.15 (0.31)	4.69	5	298	44.02	206
Sparisoma viride	Stoplight Parrotfish	1.51 (0.35)	1.46	17	0.16 (0.48)	5.14	4	205	30.28	156
Ocyurus chrysurus	Yellowtail Snapper	1.52 (0.57)	1.46	16	0.13 (0.93)	4.19	7	221	32.64	137
Scarus iseri	Striped Parrotfish	6.61 (1.57)	6.36	3	0.09 (0.23)	2.98	9	256	37.81	113
Epinephelus guttatus	Red Hind	0.36 (0.09)	0.35	49	0.09 (0.27)	2.85	10	142	20.97	60
Halichoeres bivittatus	Slippery Dick	7.55 (1.60)	7.26	2	0.03 (0.07)	1.04	24	375	55.39	58
Thalassoma bifasciatum	Bluehead Wrasse	11.17 (2.60)	10.74	1	0.03 (0.07)	0.85	30	268	39.59	34
Halichoeres garnoti	Yellowhead Wrasse	4.23 (0.75)	4.07	7	0.02 (0.06)	0.76	33	289	42.69	32
Dasyatis americana	Southern Stingray	0.02 (0.02)	0.02	122	0.49 (4.92)	15.84	1	13	1.92	30
Holocentrus rufus	Longspine Squirrelfish	0.49 (0.19)	0.47	42	0.04 (0.14)	1.34	20	138	20.38	27
Haemulon flavolineatum	French Grunt	0.90 (0.49)	0.87	29	0.04 (0.19)	1.41	17	119	17.58	25
Scarus taeniopterus	Princess Parrotfish	1.15 (0.35)	1.11	25	0.04 (0.15)	1.17	22	142	20.97	25
Acanthurus chirurgus	Doctorfish	0.60 (0.29)	0.58	36	0.04 (0.19)	1.37	19	92	13.59	19
Pseudupeneus maculatus	Spotted Goatfish	0.54 (0.17)	0.52	39	0.03 (0.11)	0.81	32	153	22.60	18
Chaetodon capistratus	Foureye Butterflyfish	0.82 (0.18)	0.78	30	0.02 (0.06)	0.62	34	187	27.62	17
Balistes vetula	Queen Triggerfish	0.08 (0.04)	0.08	89	0.06 (0.34)	2.04	12	45	6.65	14
Stegastes partitus	Bicolor Damselfish	4.84 (1.21)	4.66	5	0.01 (0.03)	0.30	50	266	39.29	12
Carangoides ruber	Bar Jack	1.19 (0.90)	1.15	23	0.03 (0.16)	0.87	29	89	13.15	11

Table 4.10. Select species of special interest to the National Park Service (NPS), threatened by overfishing, and dominant components of the fish assemblage around St. John that were examined in further detail. Species listed in phylogenetic order. Summary information showing maximum size observed in the study region compared with maximum known size for the species and the proportion of juveniles found inside and outside VIIS from 2001-2009. MI = mobile invertebrates, SI = sessile invertebrates

Family	Species	Common name	Trophic group	Size class (cm) at 1st sexual maturity	Max. size (TL cm) Known	Max. size (TL cm) STJ	Percent juveniles STJ	Percent juveniles VIIS
Serranidae - groupers & seabasses	*Cephalopholis fulva*	Coney	MI and fishes	15-20	41	30-35	33.3	21.8
	Epinephelus guttatus	Red Hind	MI and fishes	25-30	76	50	77.4	77.2
Lutjanidae	*Lutjanus apodus*	Schoolmaster Snapper	MI and fishes	25-30	67.2	50	93.1	47.8
	Lutjanus griseus	Gray Snapper	MI and fishes	25-30	89	30-35	20.0	96.3
	Lutjanus synagris	Lane Snapper	MI and fishes	20-25	60	35-40	92.3	93.5
	Ocyurus chrysurus	Yellowtail Snapper	MI and zooplankton	25-30	86.3	50	89.8	89.3
Haemulidae	*Haemulon flavolineatum*	French Grunt	MI and SI	15-20	30	25-30	59.6	81.2
	Haemulon sciurus	Bluestriped Grunt	MI	15-20	46	38	75.0	97.9
Acanthuridae	*Acanthurus bahianus*	Ocean Surgeonfish	Herbivore	15-20	38	30-35	91.6	92.5
	Acanthurus coeruleus	Blue Tang	Herbivore	10-15	39	20-25	87.5	90.9
Scaridae	*Scarus iseri*	Striped Parrotfish	Herbivore	15-20	35	30-35	94.2	93.8
	Sparisoma aurofrenatum	Redband Parrotfish	Herbivore	15-20	28	40	84.7	80.4
Balistidae	*Balistes vetula*	Queen Triggerfish	MI	25-30	60	50	53.6	53.6

4.3.2. Temporal patterns in fish assemblage metrics

Species richness varies over the nine year study period both inside and outside VIIS (Figure 4.13a). Although there was a slight decline in richness following the 2005 bleaching and disease event, particularly inside VIIS, there was no significant difference in richness before compared with after the event (both p>0.05). Numerical abundance was highly variable among years but showed no trend over time although abundance increased slightly outside VIIS after 2005 (Figure 4.13b). Total biomass also varied widely among years both inside and outside VIIS but this variability was not synchronous (Figure 4.13c). However, the coefficient of variation between management regimes was similar (STJ COV=1.1, VIIS COV = 1.2).

Total catch biomass and frequency in St. John (from both gear types) was dominated in all years by Red Hind (*Epinephelus guttatus*) and Queen Triggerfish which together formed 43-50% of the total catch biomass (Table 4.11). Smaller species including Coney, formed a larger proportion of the catch frequency while the larger Ocean Triggerfish (*Canthidermis sufflamen*) contributed more significantly to catch biomass. Yellowfin Grouper (*Mycteroperca venenosa*) and white grunt were common in the catch in 1992 but in low abundance in 2002.

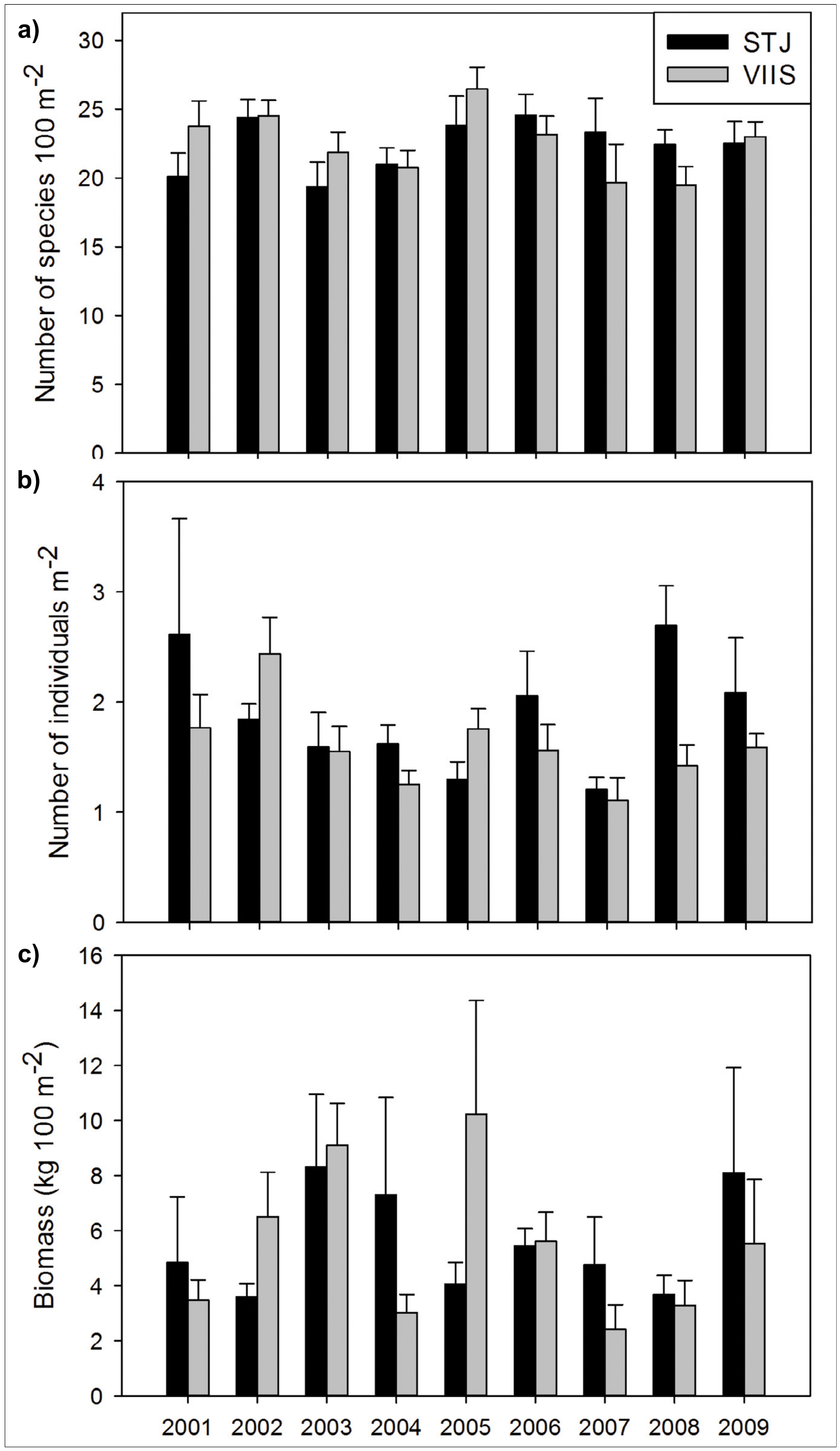


Figure 4.13. Temporal trends in fish assemblage metrics from 2001 to 2009: a) number of species, b) number of individuals and c) biomass.

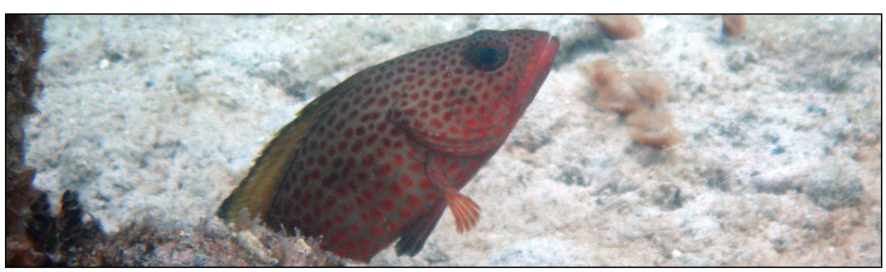


*Figure 4.14. Red Hind (*Epinephelus guttatus*) in the St. John, USVI study area. Credit: NOAA//NOS/NCCOS/CCMA/Biogeography Branch.*

Distribution data for selected species

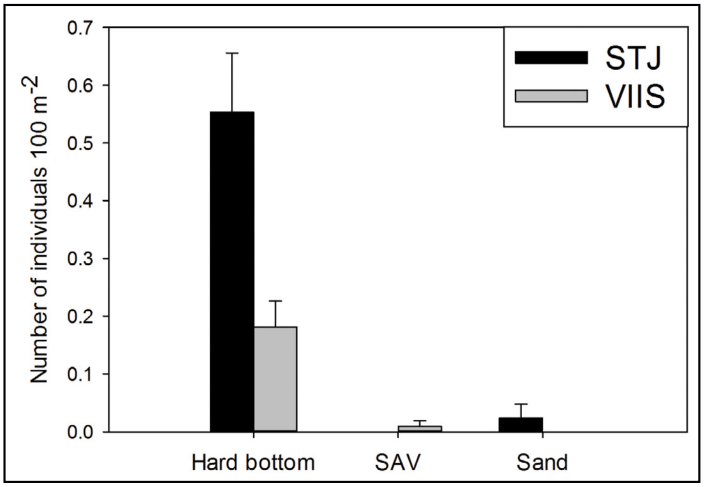
Coney (*Cephalopholis fulva*)

Coneys are small groupers (Serranidae) that feed mostly on small mobile invertebrates (crabs and shrimp) and small fishes and hunts diurnally (Randall, 1967; Figure 4.15). They are a common food fish in the Virgin Islands and were the second most common grouper found in traps around St. John (Garrison et al., 2004). Due to the decline in larger groupers, Coneys have become an increasingly important fisheries resource in the Virgin Islands (Beets et al., 1994).

*Figure 4.15. Coney (*Cephalopholis fulva*) in St. John, USVI. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

Coneys were found almost exclusively on the south coast hardbottom habitats (>98% of all individuals, >99% of total biomass [kg/100 m^2]; Figure 4.16). Within hardbottom habitats, numerical density (number of individuals/100 m^2) was nearly three times higher outside VIIS compared to inside (Figure 4.16). Although densities of Coney were much higher outside VIIS, the average size (±SE) of individuals was significantly greater (t=2.18, p=0.03) inside VIIS (X=19.8 ± 8.5) compared to outside (X=16.6 ± 5.5; Figure 4.17). The length at first maturity for Coney is 16 cm fork length (FL; Heemstra and Randall, 1993; Thompson and Munro, 1978) and based on the observed size distributions, 78% of the individuals inside VIIS were >15 cm while 67% were >15 cm outside VIIS.

Coney adults were most common in colonized bedrock, colonized pavement with sand channels, and linear reef. Together these three habitat harbored 87% of all adult Coney on hardbottom habitats (Figure 4.18). Juveniles and subadults were most commonly observed on aggregated patch reefs (35% of total), followed by linear reef (18%) and colonized bedrock (14%). Juvenile Coneys were most common in Coral Bay, Rendezvous Bay, and along the southeast portion of St. John between LeDuc Island and Ram Head. Adults were more widely distributed than juveniles but generally restricted to the southern and eastern portions of the island (Figure 4.19).



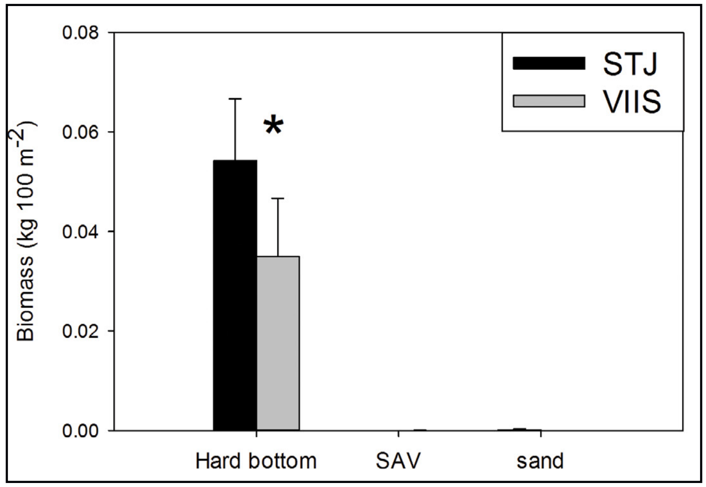


*Figure 4.16. a) Numerical density and b) biomass (+ standard error [SE]) of Coney (*C. fulva*) by major habitat type inside VIIS and adjacent areas around St. John (STJ).*

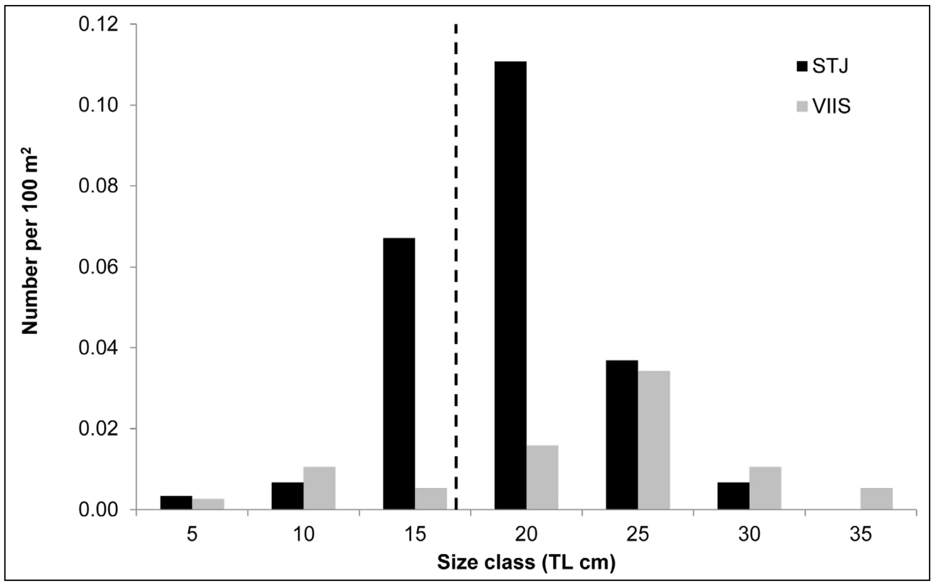


*Figure 4.17. Size frequency distribution of Coney (*C. fulva*) inside and outside VIIS. Dashed line represents size class at 1st sexual maturity.*

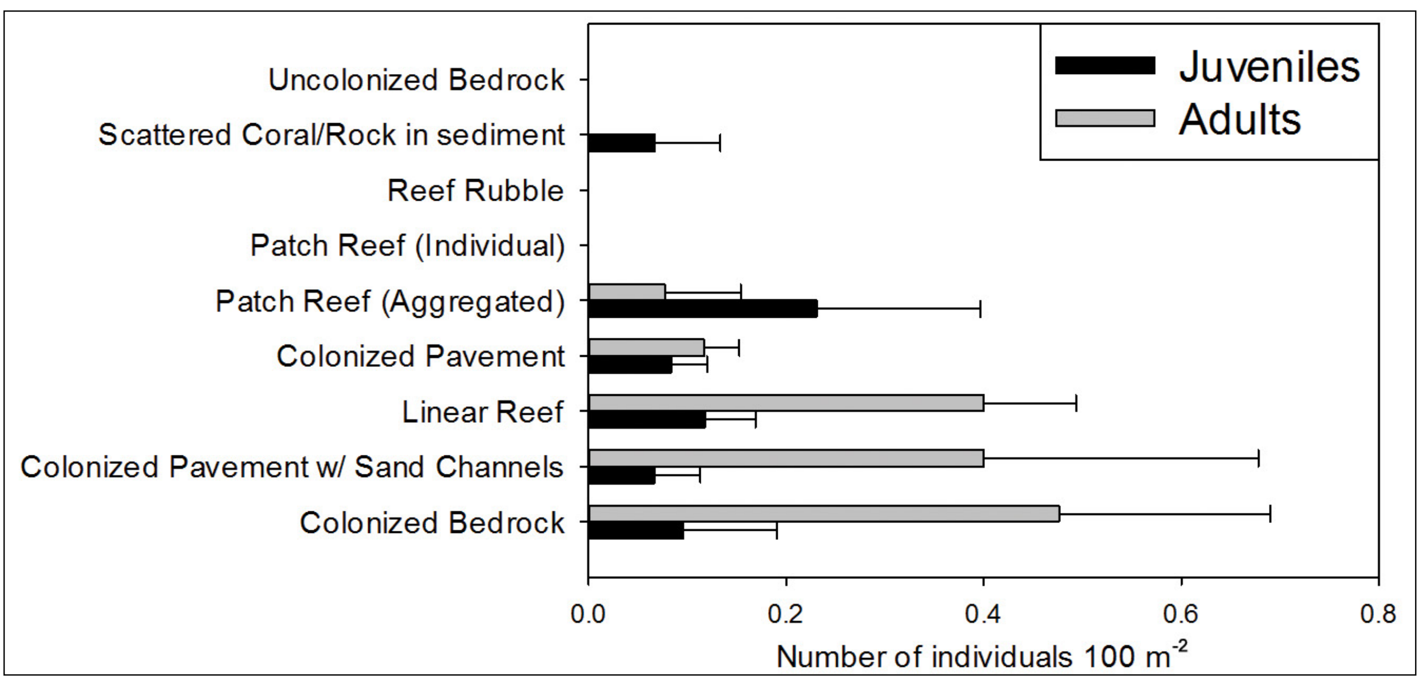


*Figure 4.18. Mean (+SE) density for juvenile/subadult and adult by observer habitat type for Coney (*C. fulva*).*

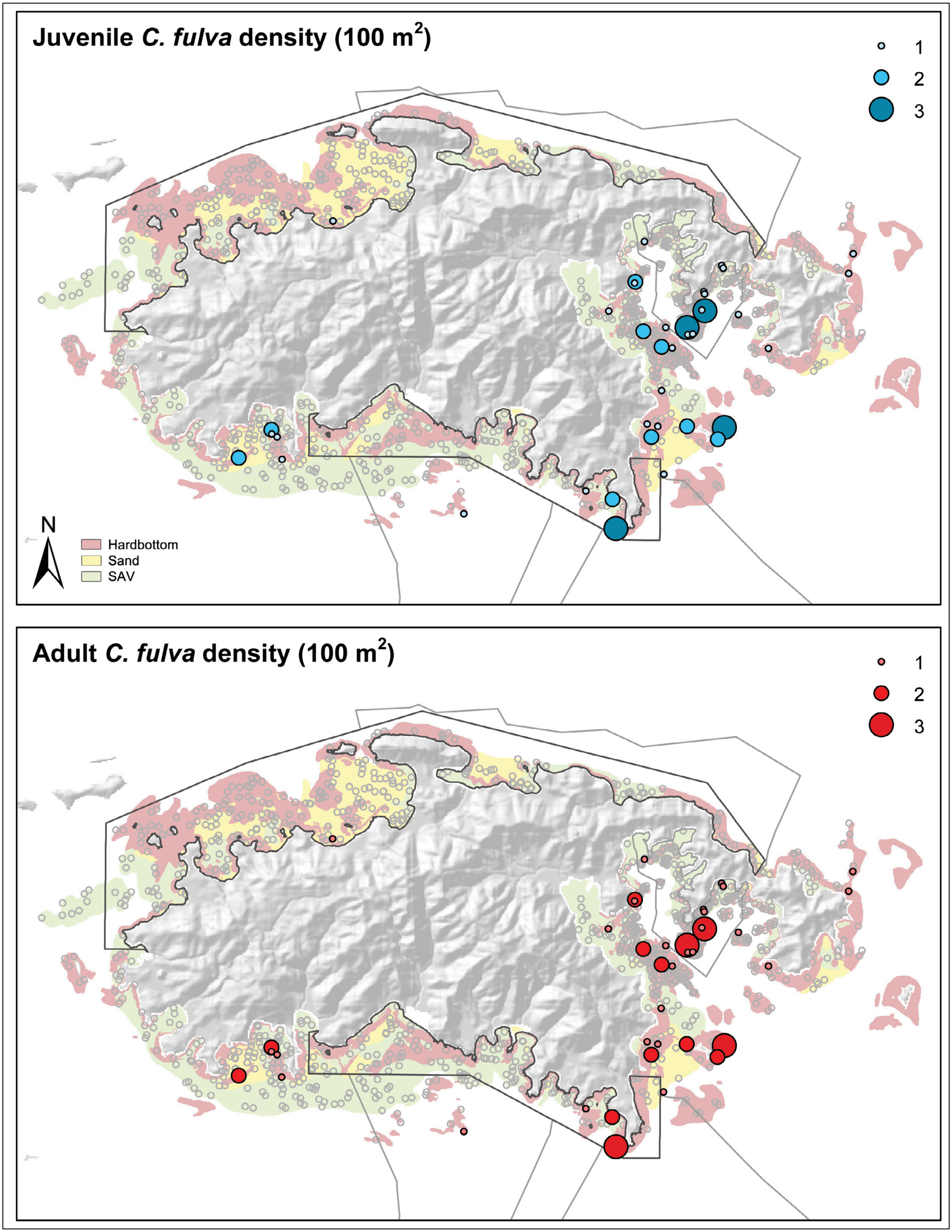


*Figure 4.19. Spatial distributions of juvenile and adult Coney (*C. fulva*) around St. John, USVI. Source: K. Stamoulis (University of Hawaii).*

Red Hind (*Epinephelus guttatus*)

The Red Hind is a protogynous grouper (Serranidae) that has historically been an important component of commercial fisheries catch in the Caribbean. Spawning aggregations of Red Hind have been protected since 1990 in St. Thomas, USVI owing to declining sizes and skewed sex ratios (Beets and Friedlander, 1999; Nemeth, 2005; Figure 4.20). Garrison et al. (2004) found Red Hinds in traps near coastal reefs around St. John, making up 68% of all serranids in the traps.

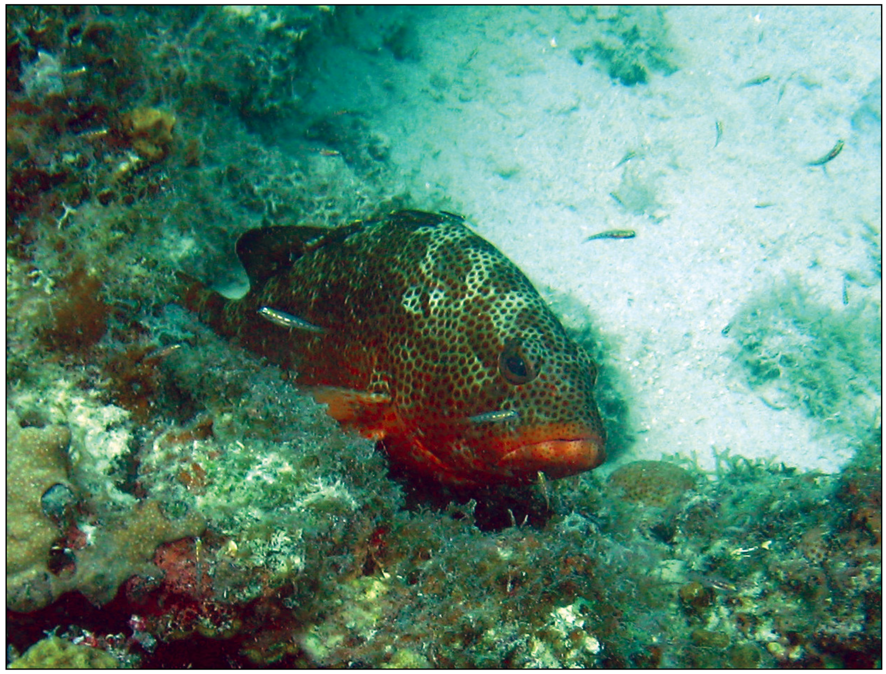


*Figure 4.20. Red Hind (*Epinephelus guttatus*) in St. John, USVI. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

Density and biomass were an order of magnitude higher on hardbottom habitats compared to SAV and sand sediments (Figure 4.21). Within hardbottom habitats, patterns across management regimes were similar with mean density and biomass higher inside VIIS, although not statistically (p= >0.05) significant. The average size (±SE) of Red Hind was not significantly different (t=0.11, p=0.91) inside VIIS (X=19.9 ± 8.5) versus outside VIIS (X=20.0 ± 7.8). The largest Red Hind observed inside VIIS was 50 cm while 40 cm was the largest individual observed outside VIIS. Nearly three-fourths of the Red Hind surveyed consisted of juveniles, below size at first maturity (>25 cm; VIIS=76%; STJ=73%; Figure 4.22).

Juvenile Red Hind were found in a variety of hardbottom habitats with colonized pavement with sand channels having the highest density, followed by scattered coral/rock in sediment, linear reef, colonized pavement, and patch reefs (Figure 4.23). In contrast, adult Red Hind were most common on individual patch reefs with lower densities in the same habitat types occupied by juveniles (Figure 4.23). Juvenile Red Hind were most common on the north shore between Mary's Point and the Durloe's as well as in the Coral Bay area (Figure 4.24). Adult Red Hind were widely distributed with highest concentrations in Coral Bay and eastern St. John.

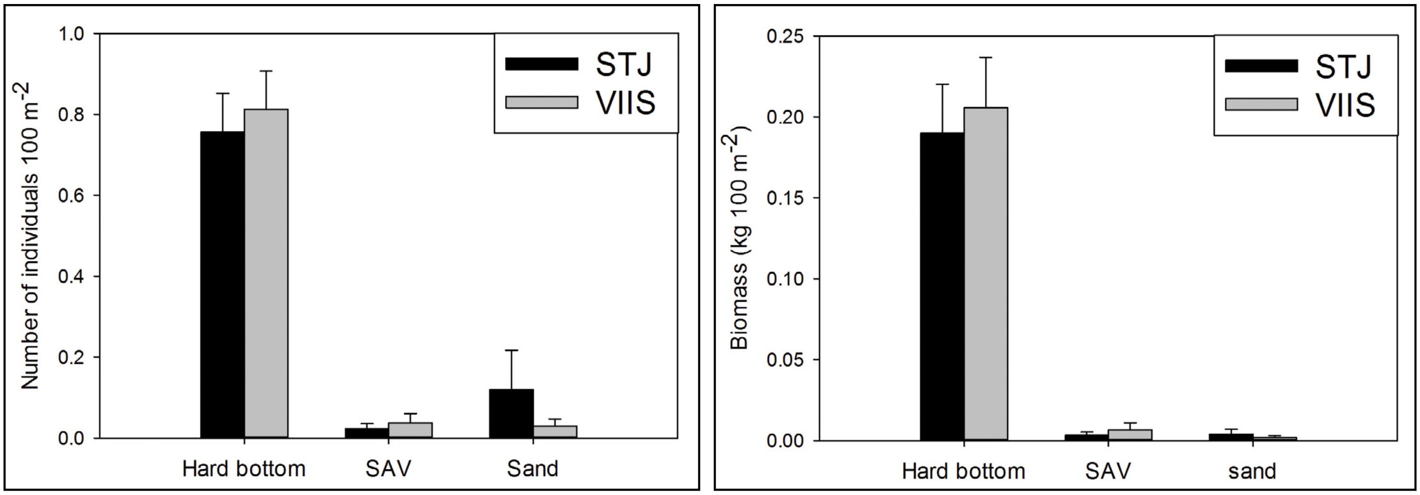


*Figure 4.21. Numerical density and biomass (+SE) of Red Hind (*E. guttatus*) by major habitat type inside VIIS and adjacent areas around St. John (STJ).*

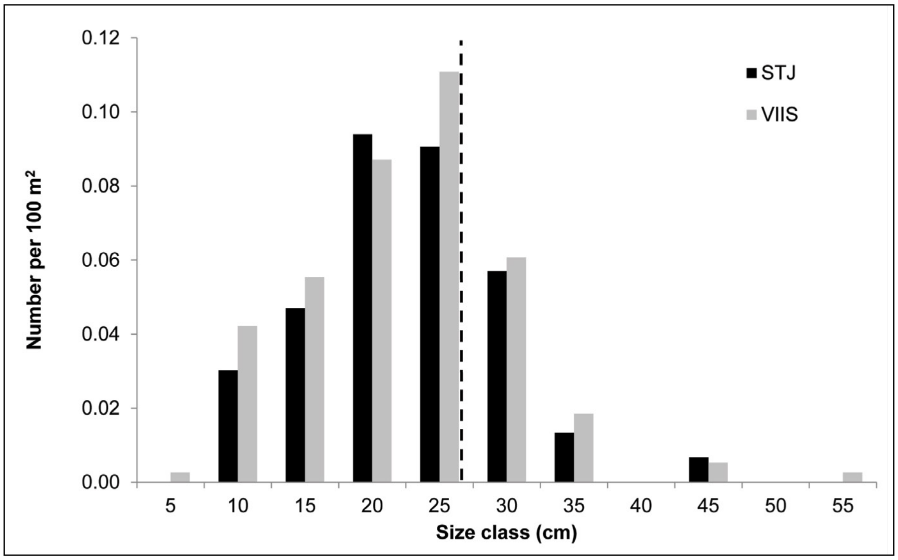


*Figure 4.22. Size frequency distribution for Red Hind (*E. guttatus*) inside and outside VIIS. Dashed line represents size class at 1st sexual maturity.*

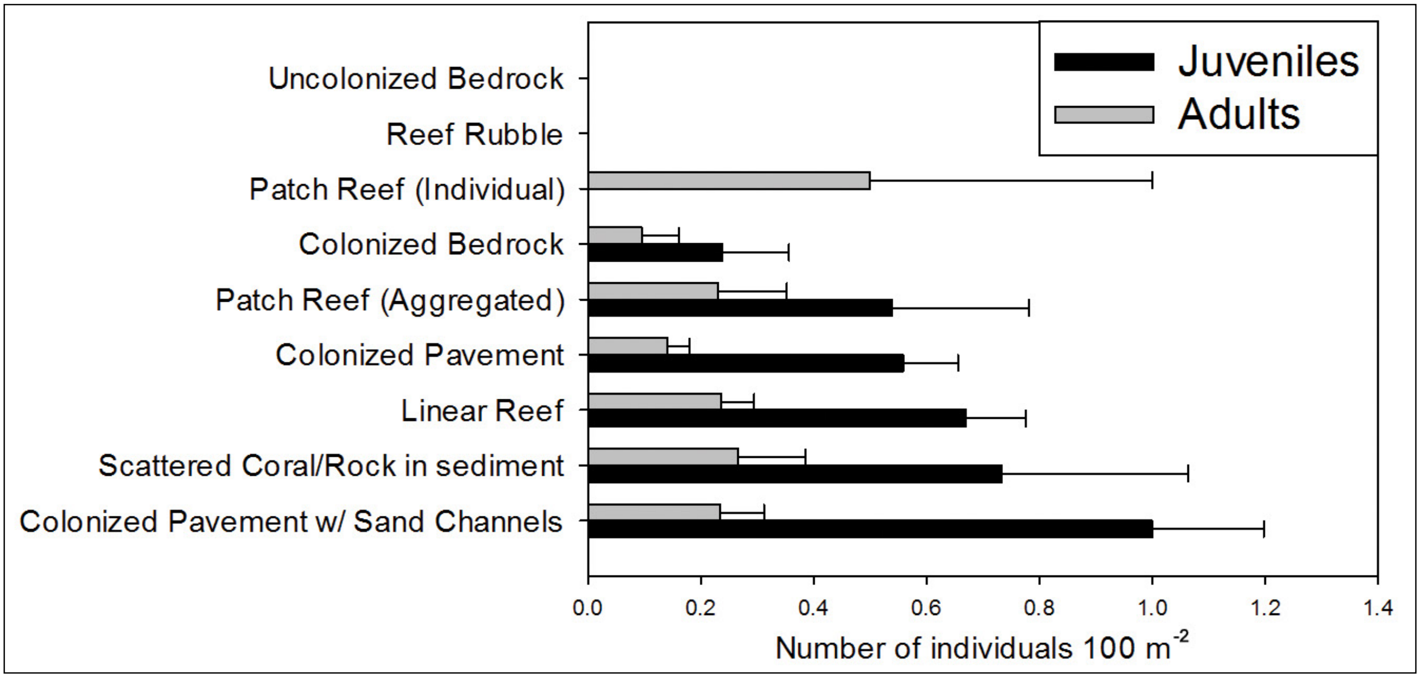


*Figure 4.23. Mean (+SE) density for juvenile/subadult and adult by observer habitat type for Red Hind (*E. guttatus*).*

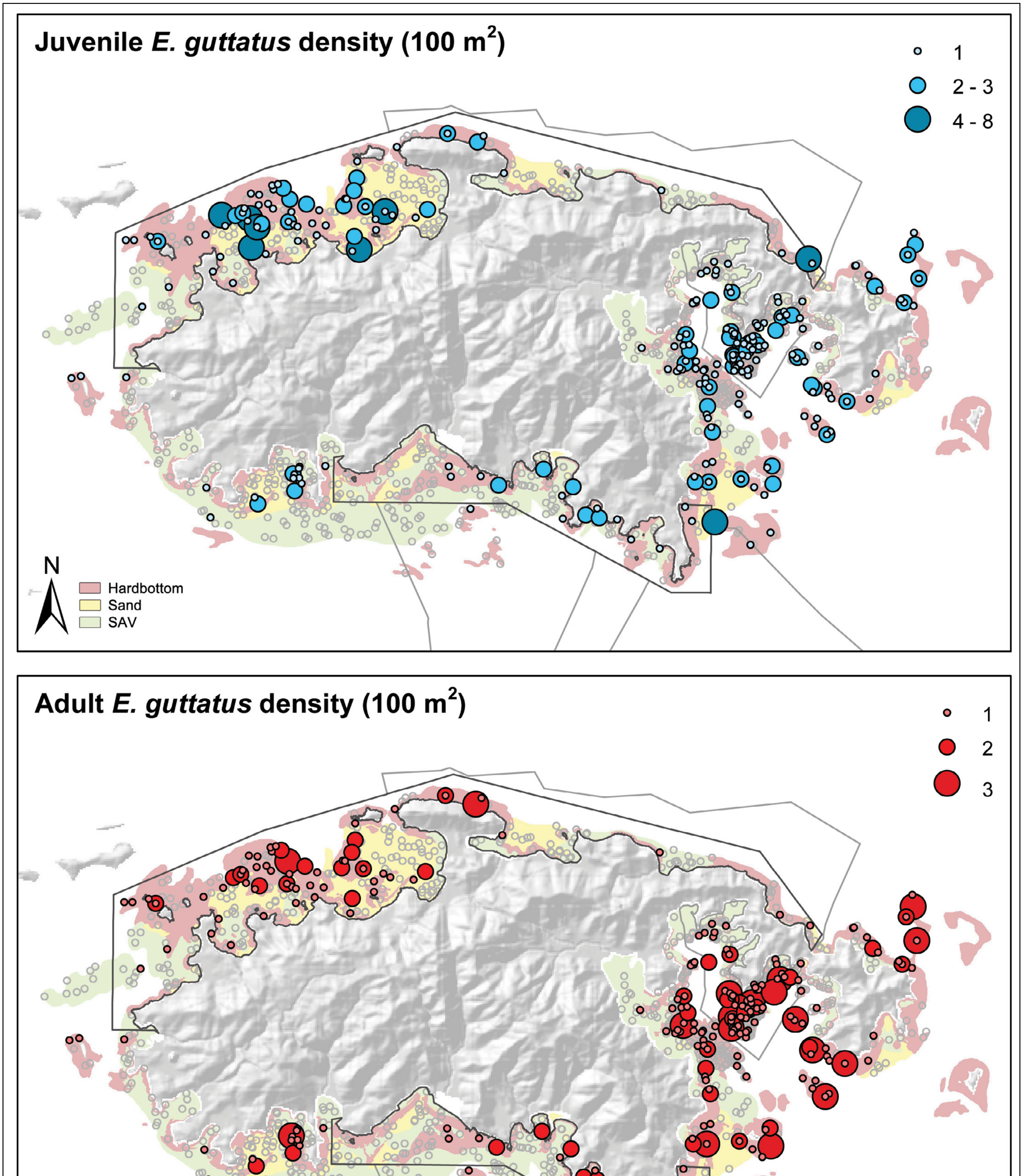


*Figure 4.24. Spatial distributions of juvenile and adult Red Hind (*E. guttatus*) around St. John. Source: K. Stamoulis (University of Hawaii).*

Schoolmaster Snapper (*Lutjanus apodus*)

Schoolmaster Snapper (*Lutjanus apodus*) are known to reach over 65 cm total length (TL; Cervigón,1993) sometimes forming large resting aggregations during the day and feed on fishes, shrimps, crabs, worms, gastropods and cephalopods (Figure 4.25). They are an important species in the commercial trap fishery in the Virgin Islands, but based on SEAMAP-C fisheries-independent trapping data, their rank abundance around St. John declined from 11th overall in 1992-93 to 32nd in 1999-2000 (Whiteman, 2005).

*Figure 4.25. Schoolmaster Snapper (*Lutjanus apodus*) in St. John, USVI. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

Although Schoolmaster Snappers were most common on hardbottom, approximately 19% of all individuals were observed in SAV with nearly all of these individuals occurring outside VIIS (Figure 4.26). Biomass was 3.5 times higher on hardbottom inside VIIS compared with outside, although these differences were not significant.

Overall, 32% of all Schoolmaster Snapper observed were adults (>25 cm TL) but nearly all the individuals outside VIIS were juveniles (93%) while almost half (48%) inside VIIS were adults (Figure 4.27). Schoolmaster Snapper were significantly larger ($t=3.5$, $p<0.001$) inside VIIS ($X=25.9 \pm 12.8$) compared with outside ($X=18.7 \pm 8.8$). When juveniles are excluded (<20 cm) these size differences are not significant ($t=1.7$, $p=0.09$) although the average size inside VIIS ($X=28.8 \pm 11.6$) is 19.5% greater than outside ($X=24.1 \pm 5.9$). Individuals greater than 40 cm (n=6) were only found inside VIIS.

Juveniles were most commonly observed in colonized pavement (40%), colonized bedrock (26%), and linear reef (18%), while adults were encountered primarily in scattered coral/rock in sediment (61%) with lower abundances in colonized bedrock (17%) and colonized pavement (16%; Figure 4.28).

Juveniles were most common around Coral Bay, with a hotspots also observed at the mouth of Great Cruz Bay and scattered locations along the south shore of St. John (Figure 4.29). Adult schoolmasters were also most common in the Coral Bay region with a few isolated locations along the north shore also having a high abundance of adults.

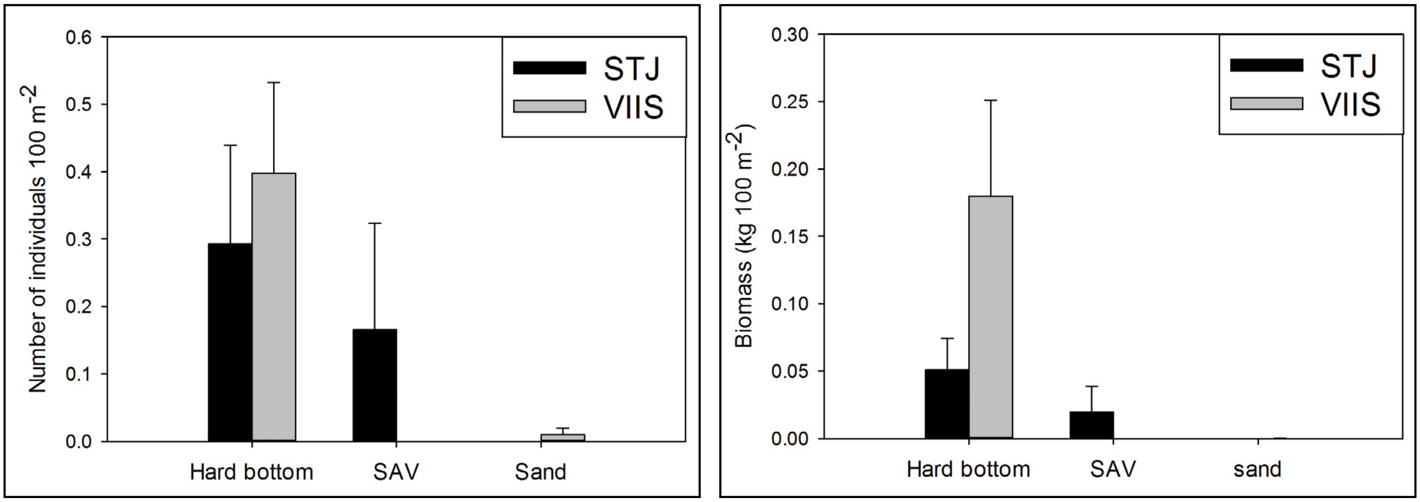


*Figure 4.26. Numerical density and biomass (+SE) of Schoolmaster Snapper (*L. apodus*) by major habitat type inside VIIS and adjacent areas around St. John (STJ).*

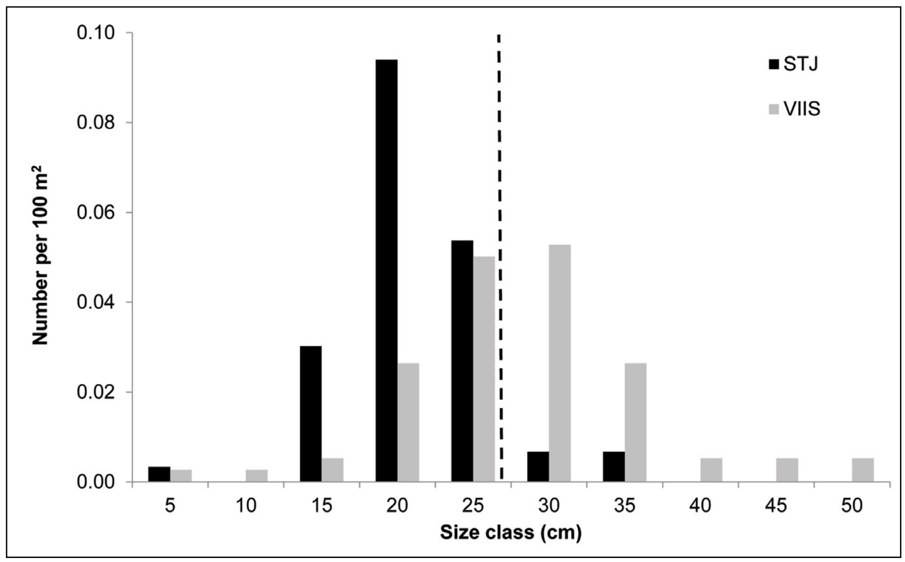


*Figure 4.27. Size frequency distribution for Schoolmaster Snapper (*L. apodus*) inside and outside VIIS. Dashed line represents size class at 1st sexual maturity.*

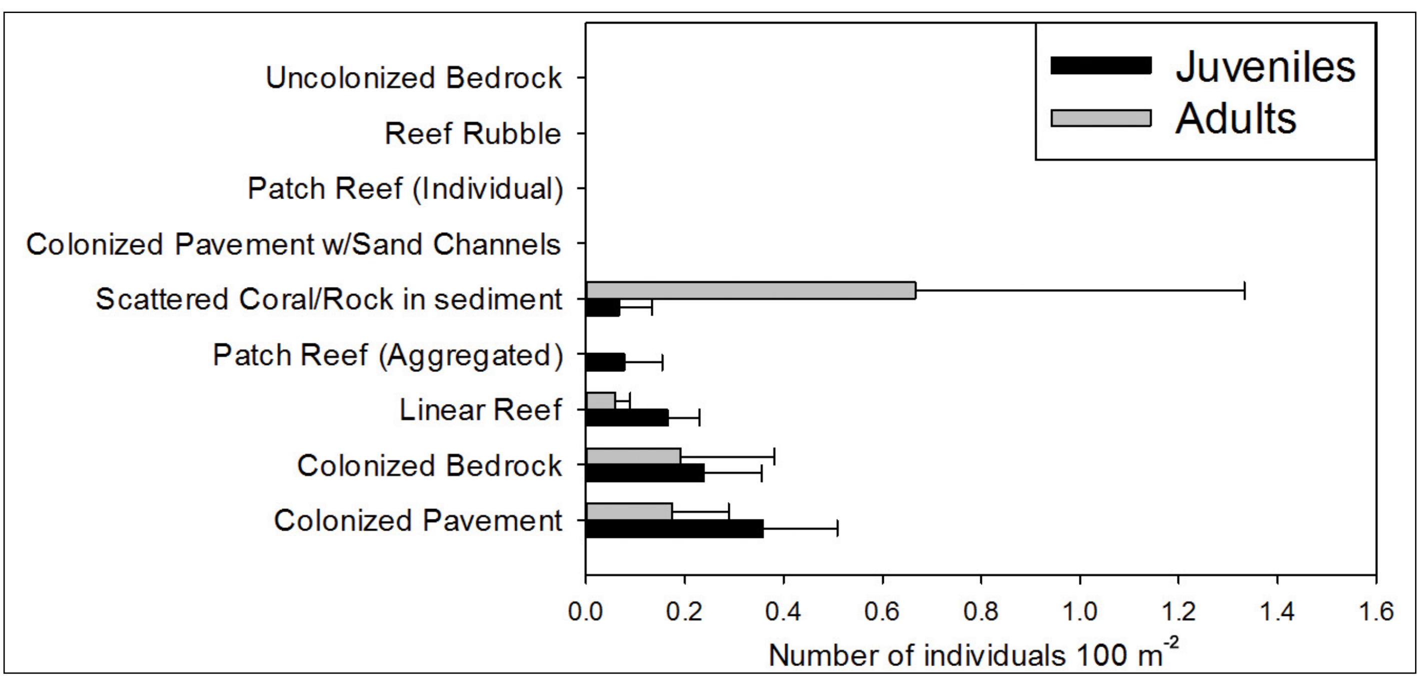


*Figure 4.28. Mean (+SE) density for juvenile/subadult and adult by observer habitat type for Schoolmaster Snapper (*L. apodus*).*

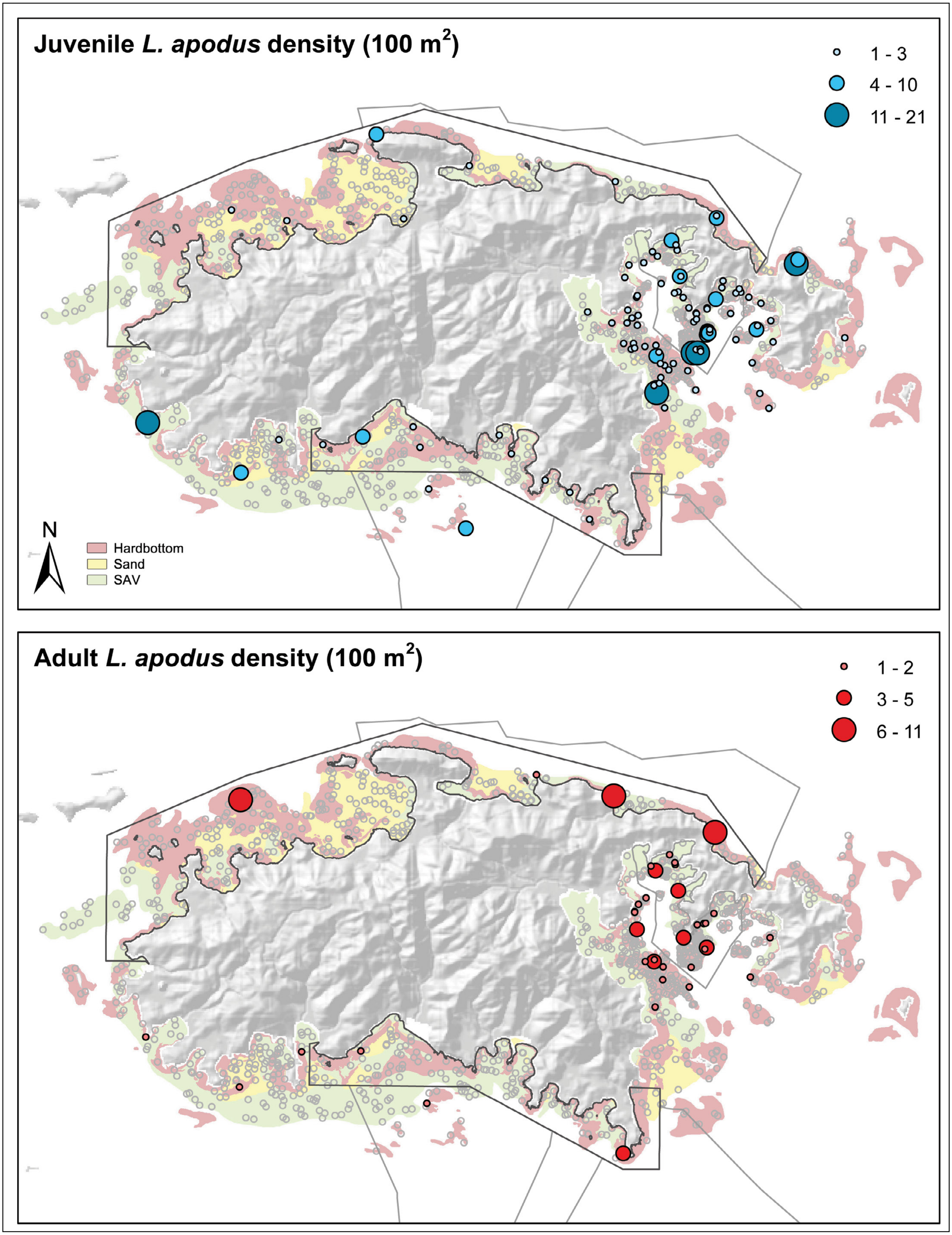


*Figure 4.29. Spatial distributions of juvenile and adult Schoolmaster Snapper (*L. apodus*) around St. John. Source: K. Stamoulis (University of Hawaii).*

Gray Snapper (*Lutjanus griseus*)

Gray Snappers (*Lutjanus griseus*) often form large aggregations and feed mainly at night on small fishes, shrimps, crabs, gastropods, cephalopods and some planktonic items (Figure 4.30). They can reach 89 cm TL and obtain sexual maturity at 26 cm FL in Cuba (García-Cagide et al. 1994) but the largest individuals observed around St. John never exceeded 35 cm. They are an important food fish in many parts of the Atlantic and Caribbean but are not well represented in the catch around St. John.

*Figure 4.30. Gray Snapper (*Lutjanus griseus*) in St. Croix, USVI. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

Only 33 Gray Snapper were quantified on transects around St. John with 82% (n=27) occurring within VIIS. Of these individuals, 80% were observed on hardbottom and the remainder within SAV (Figure 4.31). Similarly, 85% of the biomass of Gray Snapper was observed on hardbottom with 15% in SAV (Figure 4.31). Owing to the small number of individuals observed, it was not possible to construct length frequency distributions. Juveniles were most abundant on linear reef (72%), followed by colonized pavement (29%; Figure 4.32). Adults were found almost exclusively on aggregated patch reefs (92%).

Only one juvenile was observed outside VIIS (20%) while nearly all (96%) of the 27 individuals observed inside VIIS were sub-reproductive size. All of the adults and nearly all of the juveniles were found within Coral Bay (Figure 4.33).

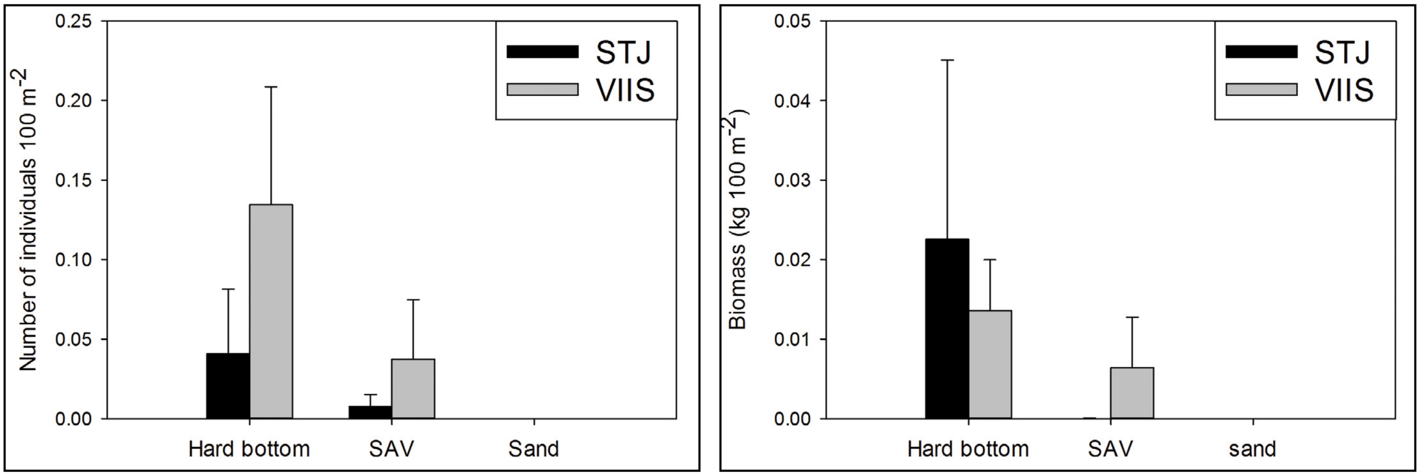


*Figure 4.31. Numerical density and biomass (+SE) of Gray Snapper (*L. griseus*) by major habitat type inside VIIS and adjacent areas around St. John (STJ).*

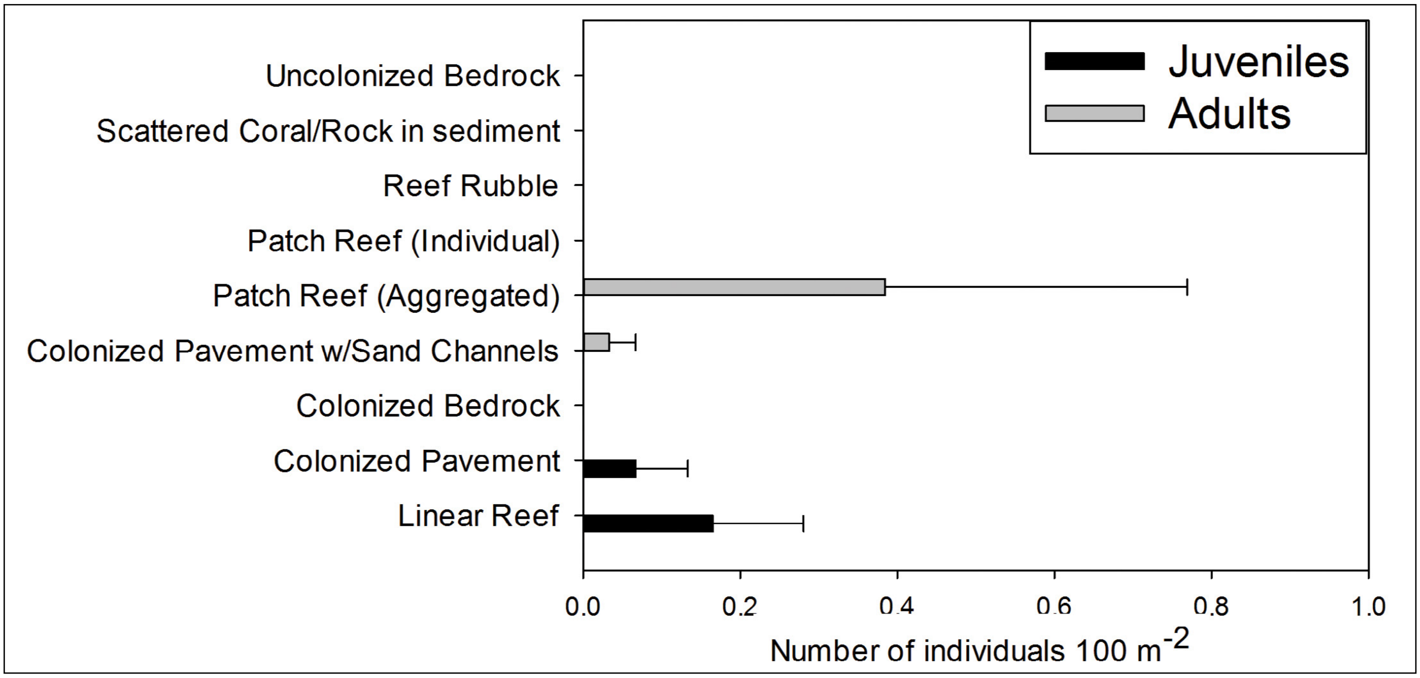


*Figure 4.32. Mean (+SE) density for juvenile/subadult and adult by observer habitat type for Gray Snapper (*L. griseus*).*

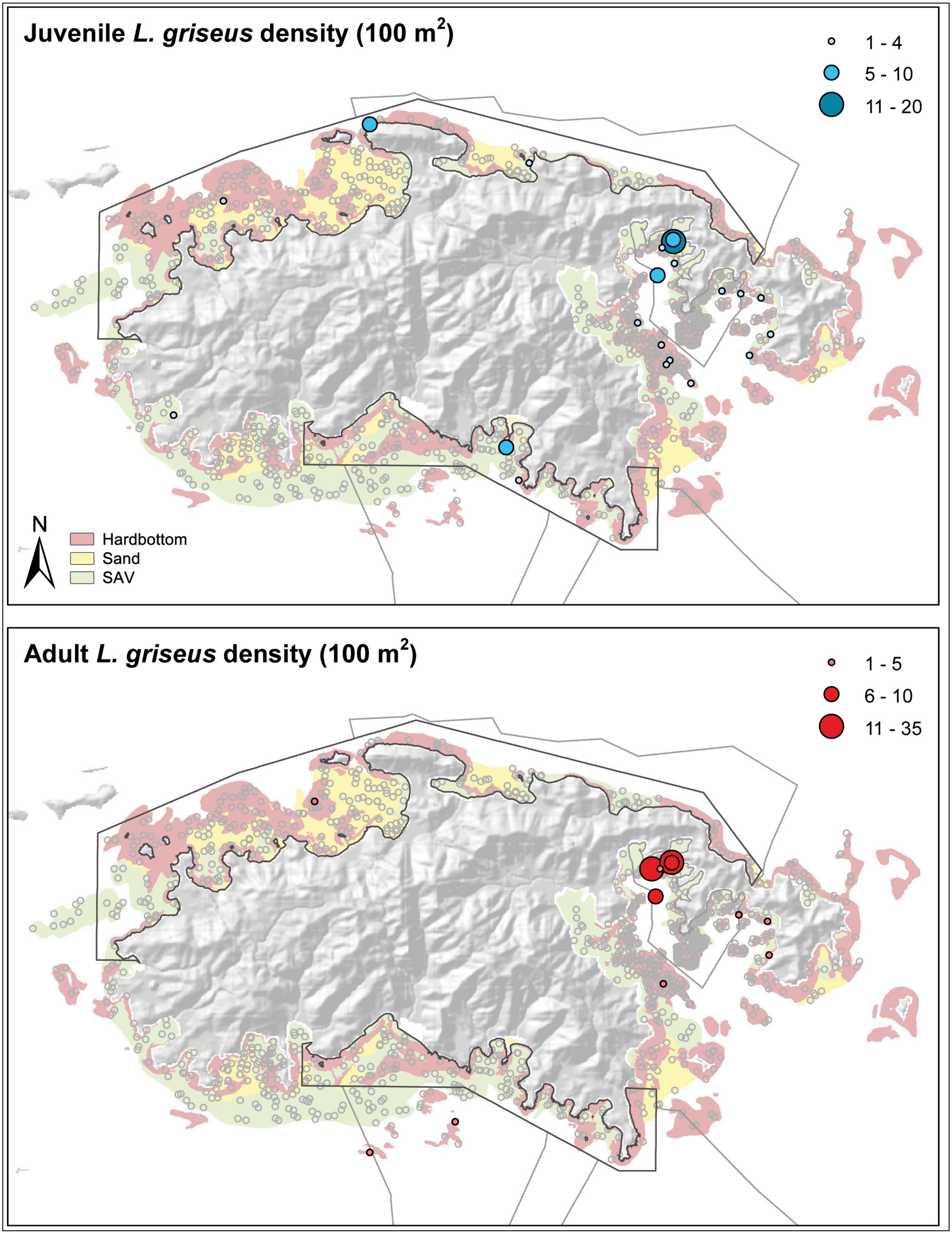


*Figure 4.33. Spatial distributions of juvenile and adult Gray Snapper (*L. griseus*) around St. John. Source: K. Stamoulis (University of Hawaii).*

Lane Snapper (*Lutjanus synagris*)

Lane Snappers (*Lutjanus synagris*) are known to reach 60 cm TL and often form large aggregations, especially during the breeding season (IGFA, 2001; Figure 4.34). They feed at night on small fishes, bottom-living crabs, shrimps, worms, gastropods and cephalopods. They were the 2nd most common snapper observed in traps around St. John, accounting for 31% of the total (Garrison et al. 2004). Acoustically tagged Lane Snapper (both adults and sub-adults) around St. John showed strong site fidelity to daytime resting areas and regular departures at sunset from reef habitats to nocturnal foraging areas in the soft sediment and seagrass beds before returning at sunrise to the reef (Monaco et al., 2009)

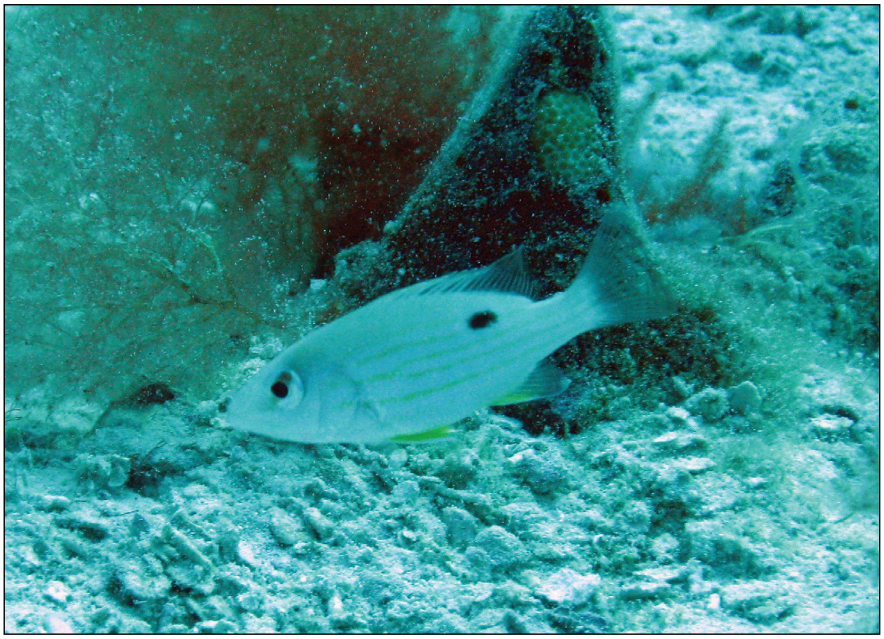


*Figure 4.34. Juvenile Lane Snapper (*Lutjanus synagris*) in St. John, USVI. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

In contrast to most species, Lane Snappers were most common numerically in SAV (41%) and sand (40%), while only 19% of all individuals were found in hardbottom habitats (Figure 4.35). For all three habitat types, numerical density of Lane Snappers was significantly higher inside versus outside VIIS (Kruskal-Wallis=2.42, p=0.012). Although most of the individuals were found in SAV and sand, the majority of the biomass of Lane Snappers (83%) was observed in hardbottom habitat (Figure 4.35).

Out of a total of 181 Lane Snapper observed around St. John, only 12 individuals (7%) were adults (>20 cm) and of this total, 11 were found inside VIIS (Figure 4.36). Average sizes of individuals were indistinguishable (t=0.03, p=0.97) between inside (X=7.3 ± 11.2) and outside VIIS (X=7.4 ± 7.5). Juveniles were most common in colonized bedrock (28%), followed by aggregated patch reefs (22%), colonized pavement (21%), and linear reef (19%; Figure 4.37). Adults were most abundant in linear reefs habitat (56%) and colonized bedrock (28%). Juveniles were most commonly observed around Coral Bay and along the north shore from Hawksnest to Brown Bay (Figure 4.38). Adults were also most commonly observed in Coral Bay with high abundance in scattered locations around the island.

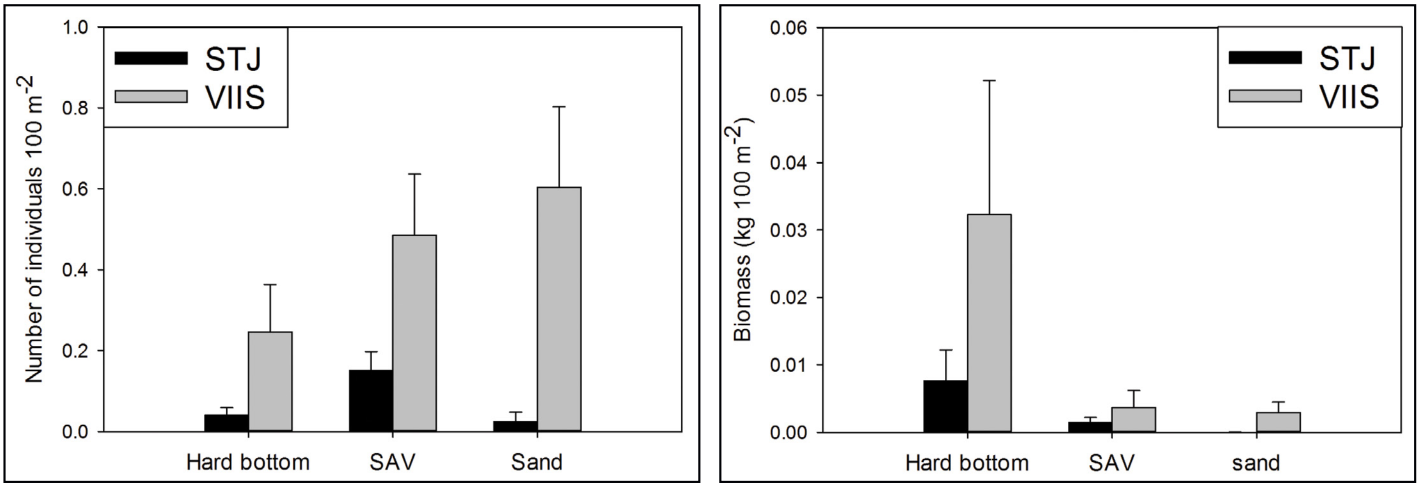


*Figure 4.35. Numerical density and biomass (± SE) of Lane Snapper (*L. synagris*) by major habitat type inside VIIS and adjacent areas around St. John (STJ).*

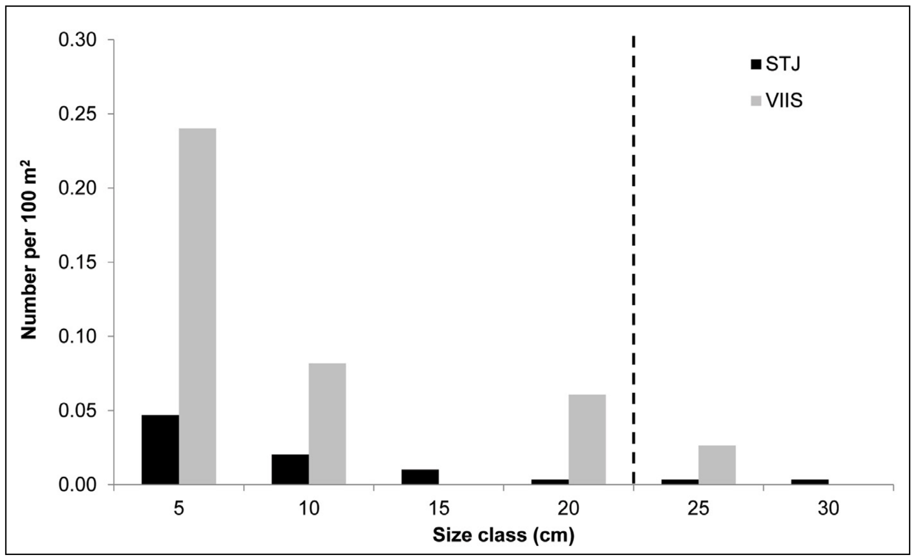


*Figure 4.36. Size frequency distribution for Lane Snapper (*L. synagris*) inside and outside VIIS. Dashed line represents size class at 1st sexual maturity.*

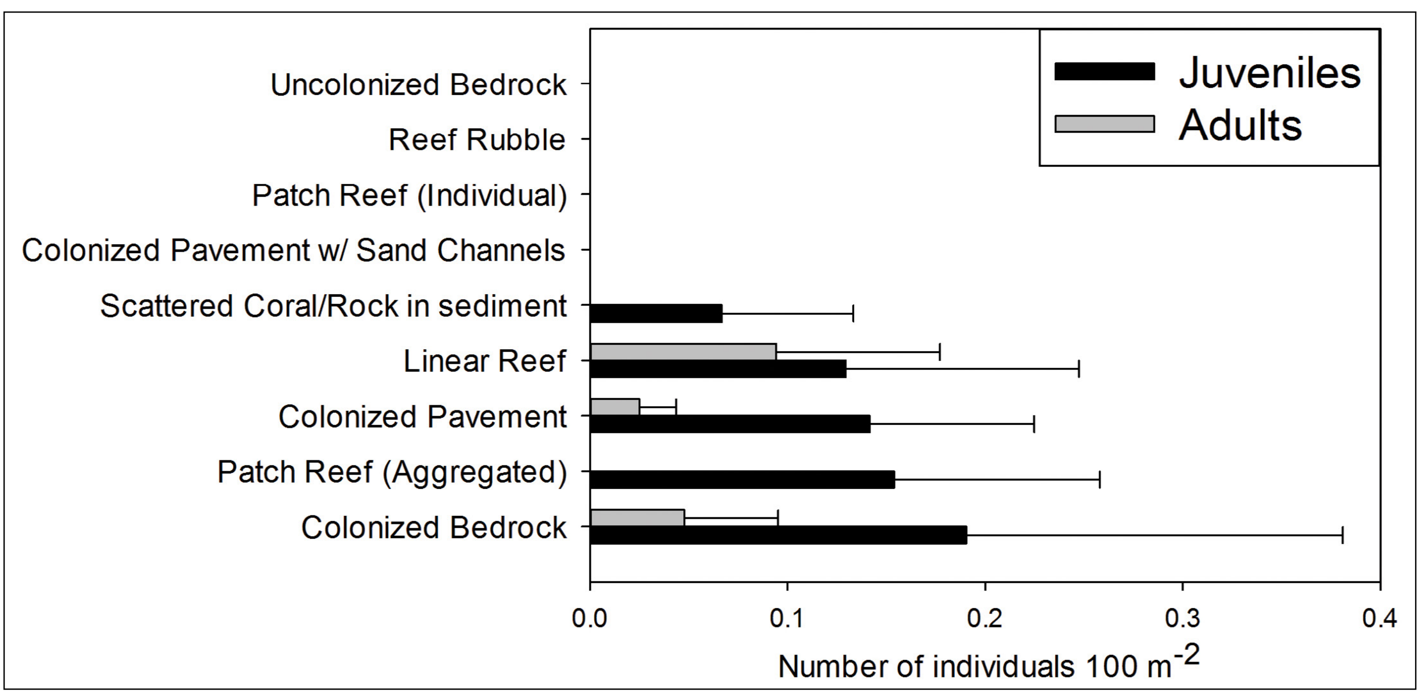


*Figure 4.37. Mean (+ SE) density for juvenile/subadult and adult by observer habitat type for Lane Snapper (*L. synagris*)..*

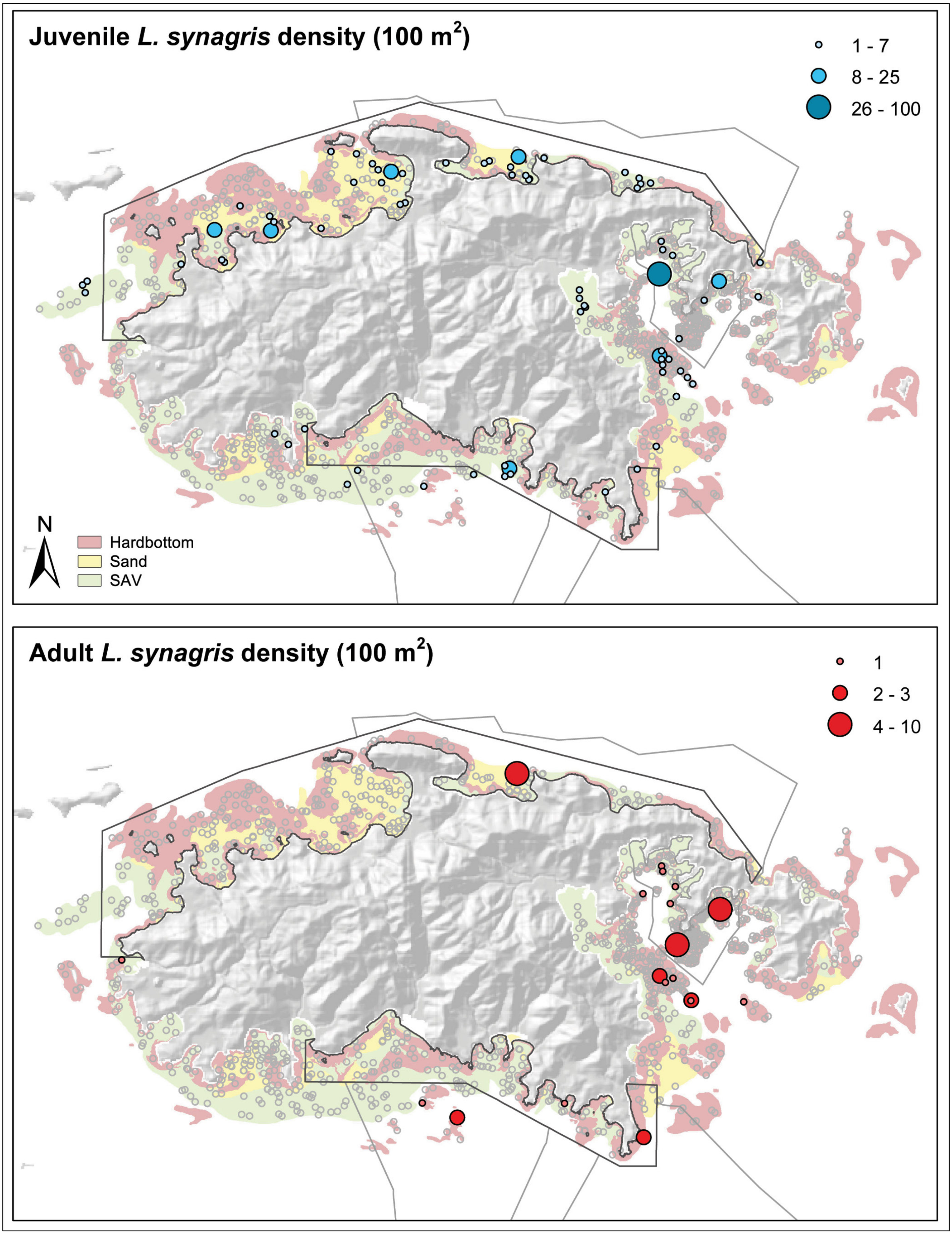


*Figure 4.38. Spatial distributions of juvenile and adult Lane Snapper (*L. synagris*) around St. John. Source: K. Stamoulis (University of Hawaii).*

Yellowtail Snapper (*Ocyurus chrysurus*)

Yellowtail Snapper (*Ocyurus chrysurus*) can grow to over to over 80 cm TL (Cervigón, 1993) and feed mainly at night on a combination of plankton and benthic animals including fishes, crustaceans, worms, gastropods and cephalopods (Figure 4.39). Juveniles feed primarily on plankton. They are an important fisheries resource throughout the region and were the 4th most abundant species captured around St. John in the SEAMAP-C fisheries-independent sampling in 1999-2000. Yellowtail Snapper are the most common snapper observed in traps around St. John, accounting for 59% of the total (Garrison et al., 2004) .

*Figure 4.39. Yellowtail Snapper (*Ocyurus chrysurus*) in St. John, USVI. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

Numerical densities of Yellowtail Snapper were uniformly distributed among major habitat types with 40% of all individuals observed in SAV, 39% in hardbottom, and 21% in sand (Figure 4.40). However, 94% of the total biomass of Yellowtail Snappers around St. John was observed on hardbottom, highlighting the small size of individuals observed in softbottom habitats (Figure 4.40). Only 21% of all individuals were adults and this trend was consistent between management strata (Figure 4.41). Yellowtail Snapper were significantly larger (t=2.5, p=0.01) outside VIIS (X=13.1 ± 14.4) compared with inside VIIS (X=10.1 ± 20.6). This is due to the large number of small juveniles inside VIIS. When only adult Yellowtail are considered (>20 cm), these differences are reversed with significantly (t=2.2, p=0.03) larger sized individuals inside VIIS (X=27.2 ± 10.4) compared to outside VIIS (X=24.6 ± 6.7).

Juveniles were most commonly encountered in scattered coral/rock in sediment (35%), followed by linear reef (17%), aggregated patch reefs (16%), and colonized pavement (15%; Figure 4.42). Adults were most abundant on individual patch reefs (38%), linear reefs (29%), and scattered coral/ rock in sediment (13%). Juveniles were widely scattered around St. John but adults were restricted to offshore features (e.g., Eagle Shoal, Johnson's Reef, White Cliffs, Waterlemon Cay; Figure 4.43).

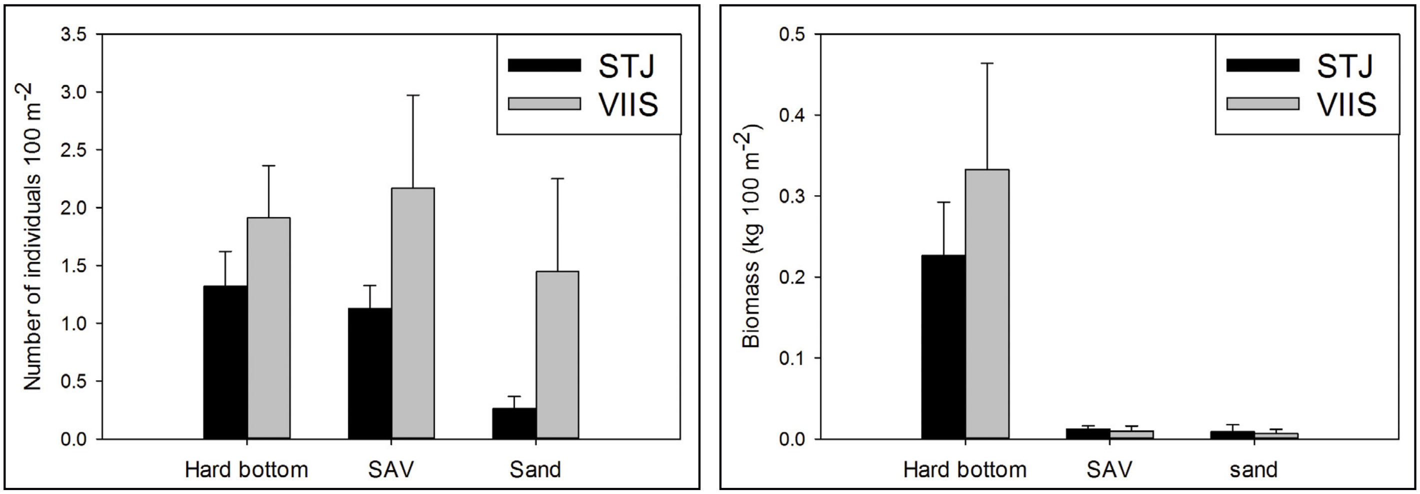


*Figure 4.40. Numerical density and biomass (+SE) of Yellowtail Snapper (*O. chrysurus*) by major habitat type inside VIIS and adjacent areas around St. John (STJ).*

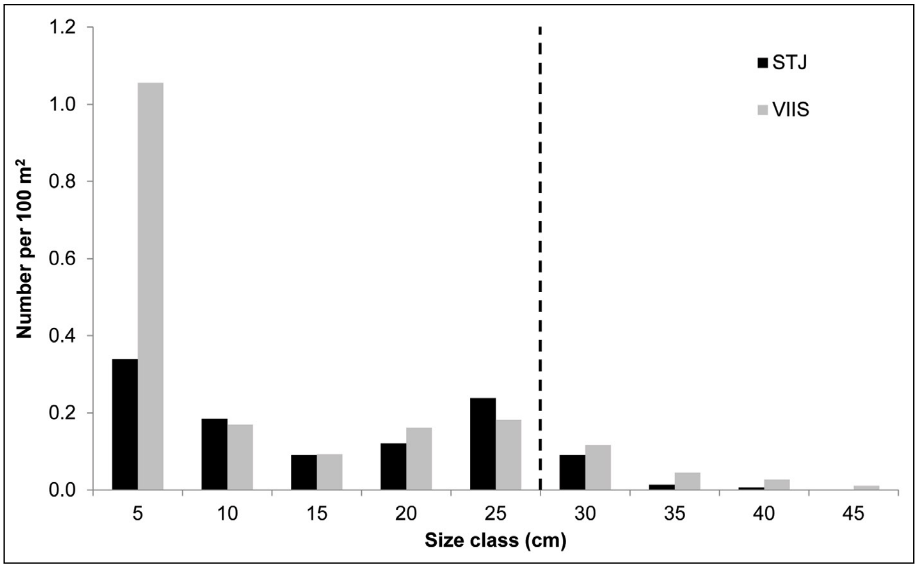


*Figure 4.41. Size frequency distribution for Yellowtail Snapper (*O. chrysurus*) inside and outside VIIS. Dashed line represents size class at 1st sexual maturity.*

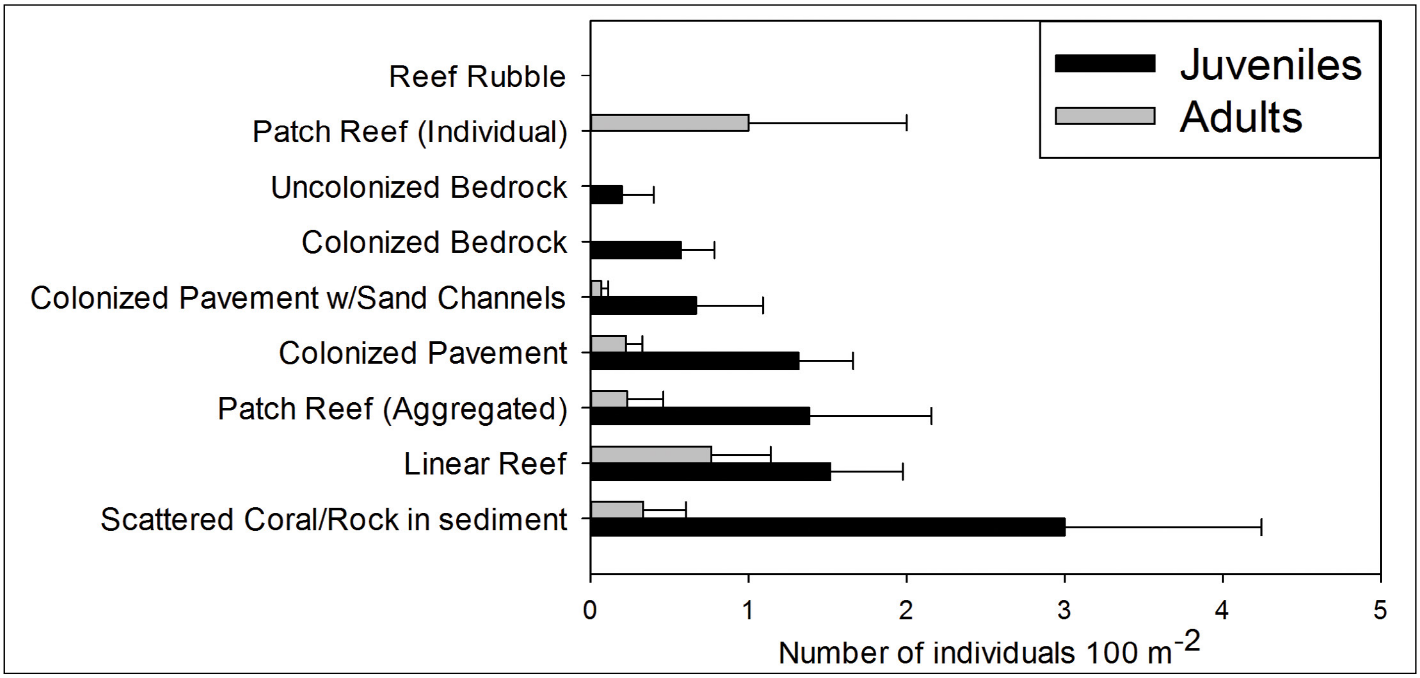


*Figure 4.42. Mean (+SE) density for juvenile/subadult and adult by observer habitat type for Yellowtail Snapper (*O. chrysurus*).*

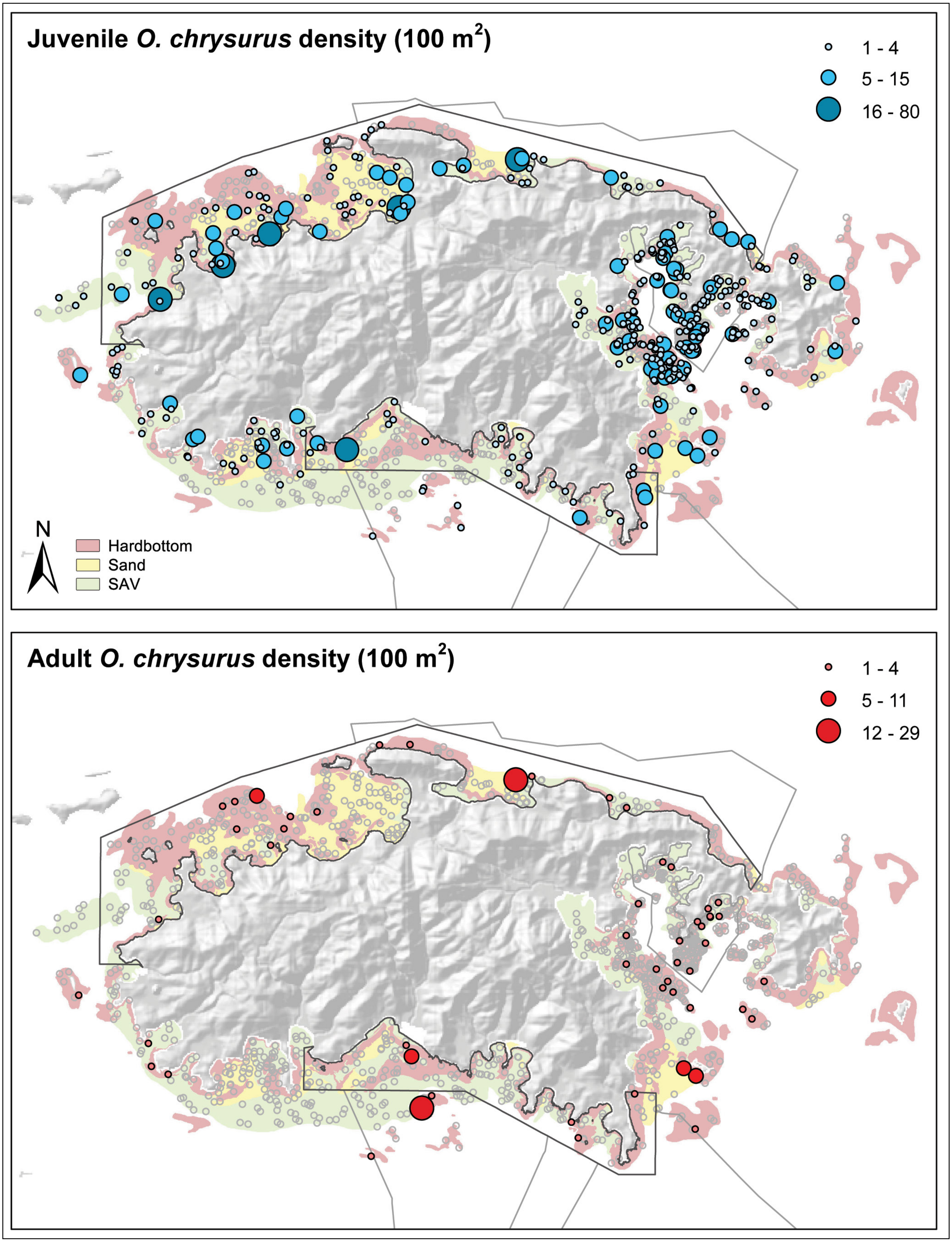


*Figure 4.43. Spatial distributions of juvenile and adult Yellowtail Snapper (*O. chrysurus*) around St. John. Source: K. Stamoulis (University of Hawaii).*

French Grunt (*Haemulon flavolineatum*)

French Grunts (*Haemulon flavolineatum*) have been well studied in the Virgin Islands, with particular emphasis on fine-scale and diel movement patterns (Ogden and Ehrlich; 1977; Helfman et al., 1982; Shulman and Ogden, 1987; Figure 4.44). Adults often occur in large schools on hardbottom habitat while juveniles are abundant in nearshore seagrass beds. French Grunts feed mainly on small crustaceans. Juvenile French Grunt school during the day at coral reef and other hardbottom habitats then disperse around dusk to feed solitarily throughout the night (Helfman et al., 1982). Kendall et al. (2003) found a positive correlation between the presence of juveniles on hardbottom sites and areas of soft bottom within 100 m.

*Figure 4.44. French Grunts (*Haemulon flavolineatum*) in St. John, USVI. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

More than 95% of the individuals and 92% of the biomass of French Grunts were observed on hardbottom (Figure 4.45). The number of individuals on hardbottom was 2.4 times higher inside VIIS compared with outside and biomass was 1.6 times higher inside versus outside VIIS. Overall 76% of all French Grunts observed were below size at 1st sexual maturity (Figure 4.46). The average size of French Grunt was significantly smaller ($t=3.6$, $p<0.001$) inside VIIS ($X=11.1 \pm 10.8$) compared to outside ($X=14.0 \pm 5.2$). However, this was due to the large number of juveniles found inside VIIS. If only adult French Grunts are examined, there is no significant difference ($t=0.1$, $p=0.9$) in average size between inside ($X=17.6 \pm 1.4$) and outside ($X=17.6 \pm 1.2$).

Juveniles were most frequently encountered on uncolonized bedrock (30%), followed by colonized pavement (26%), and linear reef (18%; Figure 4.47). Adult French Grunts were most abundant on linear reef (24%), aggregated patch reefs (21%), and colonized pavement (19%). Juvenile hotspots were encountered near Trunk Bay, the Durloe's and in Coral Bay. Adults were frequently sighted in Coral Bay, but also in many other locations around St. John.(Figure 4.48).

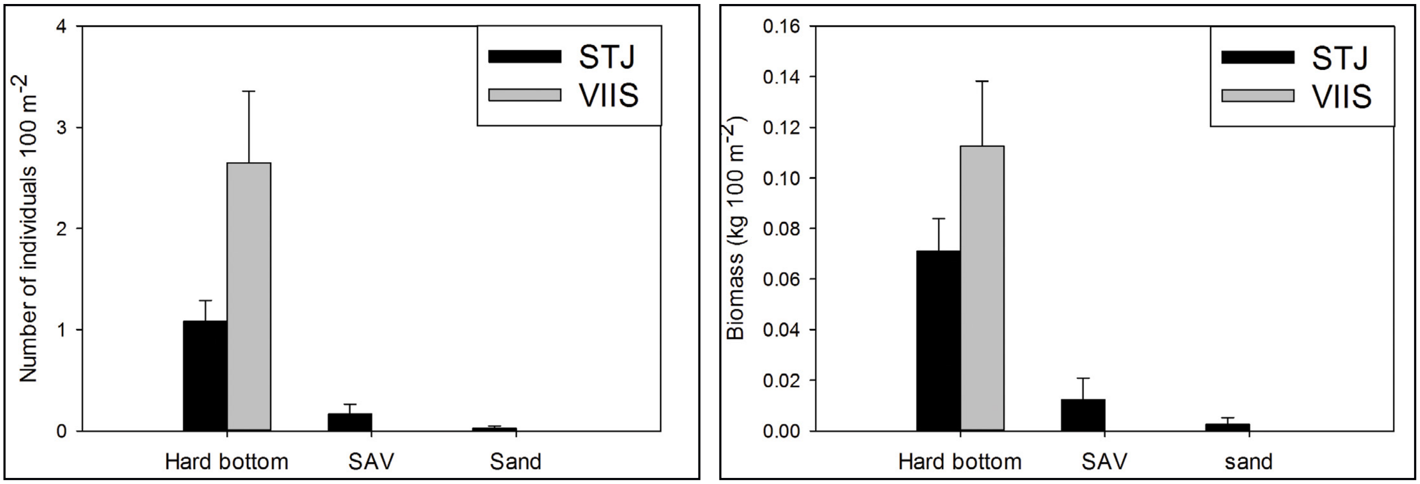


*Figure 4.45. Numerical density and biomass (+SE) of French Grunt (*H. flavolineatum*) by major habitat type inside VIIS and adjacent areas around St. John (STJ).*

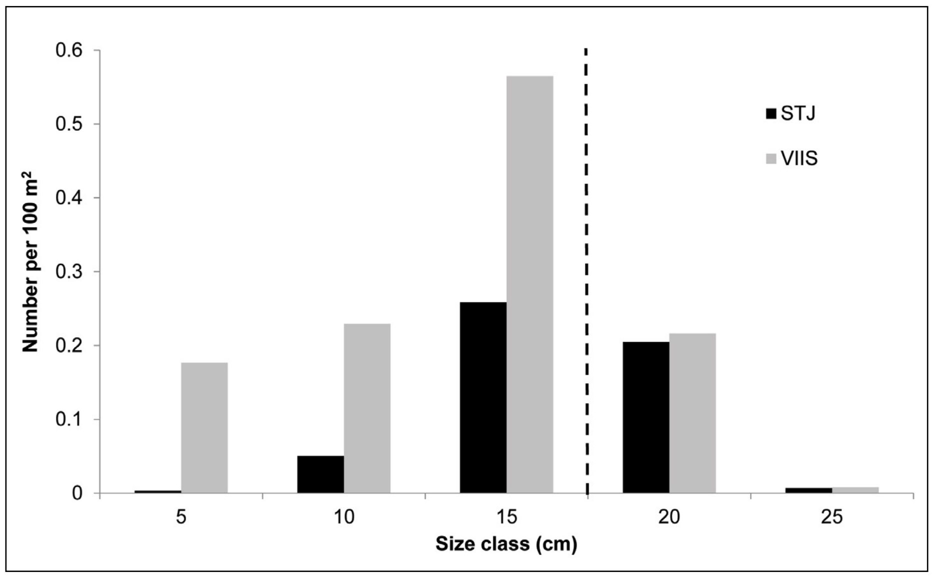


*Figure 4.46. Size frequency distribution for French Grunt (*H. flavolineatum*) inside and outside VIIS. Dashed line represents size class at 1st sexual maturity.*

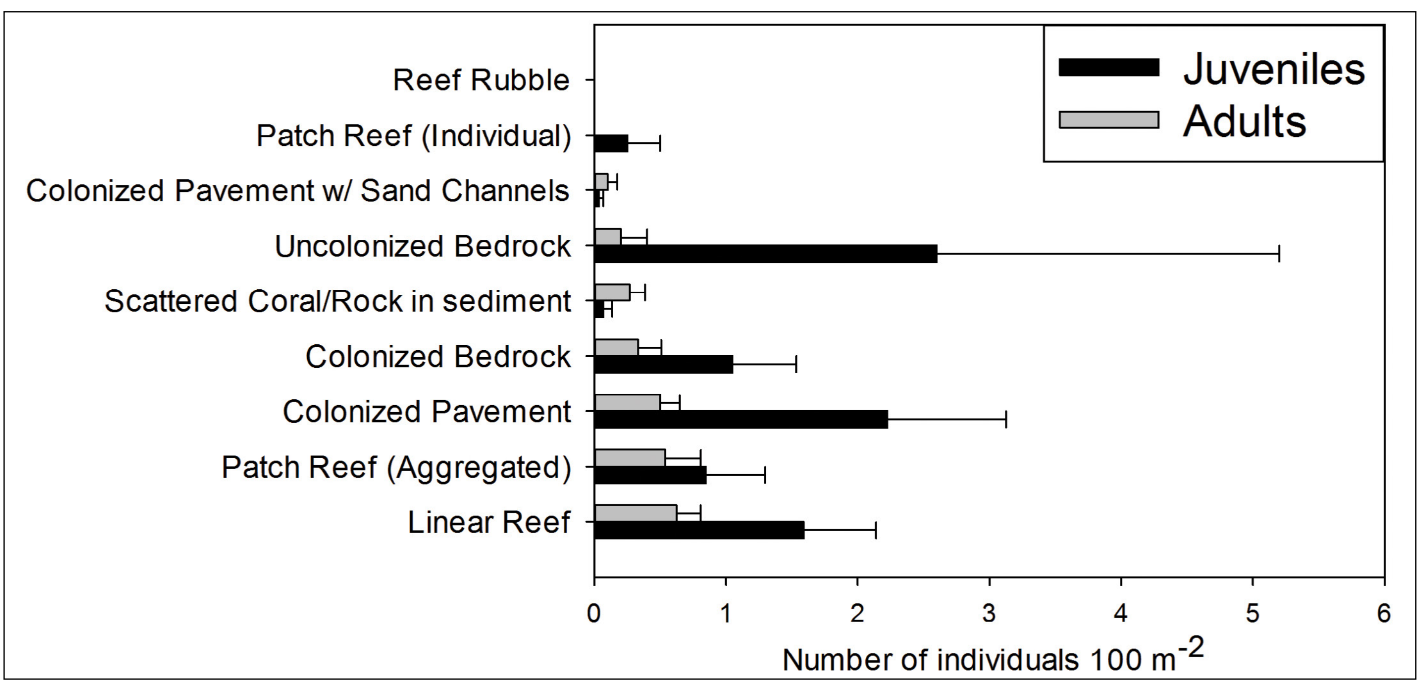


*Figure 4.47. Mean (+SE) density for juvenile/subadult and adult by observer habitat type for French Grunt (*H. flavolineatum*).*

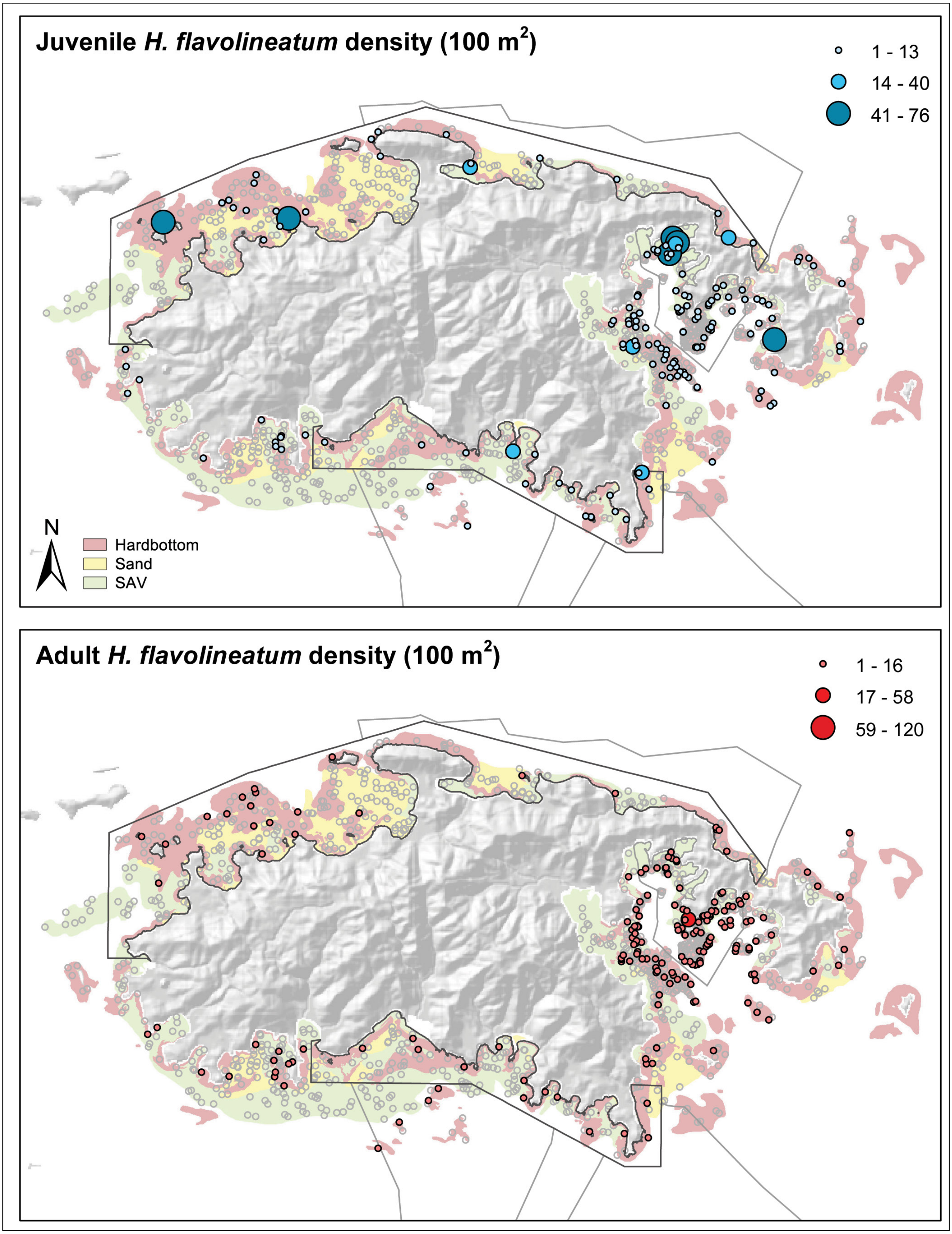


*Figure 4.48. Spatial distributions of juvenile and adult French Grunt (*H. flavolineatum*) around St. John. Source: K. Stamoulis (University of Hawaii).*

Bluestriped Grunt (*Haemulon sciurus*)

Bluestriped Grunt (*Haemulon sciurus*) reach a maximum size of 46 cm TL and are found in small groups over coral and rocky reefs and drop-offs (Figure 4.49). They feed on crustaceans, bivalves, and occasionally on small fishes. Large Bluestriped Grunts showed high site fidelity to nocturnal foraging sites in seagrass beds around St. John (Beets et al., 2003). Acoustically tagged Bluestriped Grunts (both adults and sub-adults) demonstrated little movement in their diurnal shelter sites in the boulder-coral zone, with most individuals making nocturnal migrations into the adjacent seagrass beds. Tracking studies by Hitt et al. (2011) on St. John and St. Thomas revealed high variability in individual movements with daytime activity spaces ranging from 284 to 12,486 m^2 and nocturnal activity spaces ranging from 608 to 25,267 m^2.

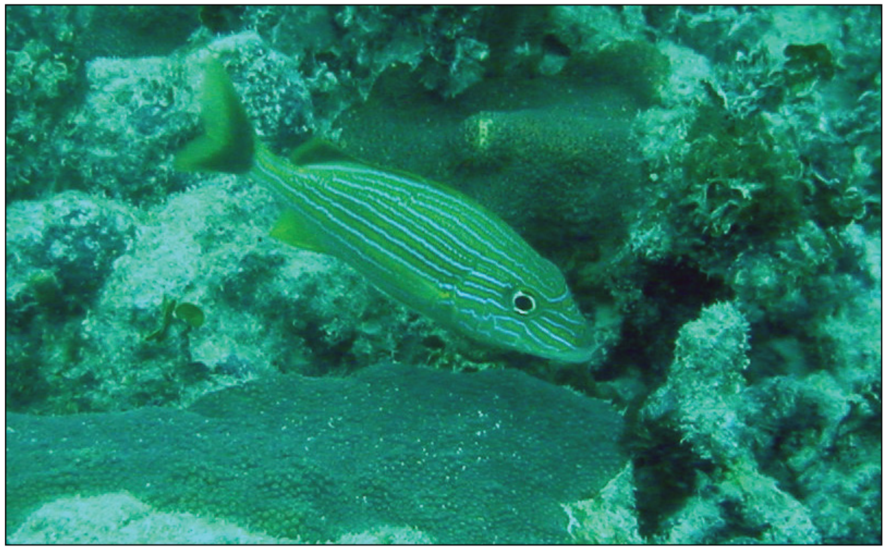


*Figure 4.49. Bluestriped Grunt (*Haemulon sciurus*) in La Parguera, Puerto Rico. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

Nearly all Bluestriped Grunts (94% by number, 99% by weight) were observed on hardbottom (Figure 4.50). Numerical abundance and biomass were both substantially greater inside compared to outside VIIS (3.8 times, 4.4 times, respectively). Over 93% of all Bluestriped Grunts observed around St. John were of reproductive size with no juveniles observed inside VIIS (Figure 4.51). The average size of Bluestriped Grunts was significantly greater (t=2.3, p=0.02) inside VIIS (X=23.5 ± 6.0) compared with outside (X=17.6 + 11.3). This difference becomes indistinguishable (t=0.6, p=0.53) when juveniles <15 cm are excluded (inside VIIS, X=23.5 ± 6.0; outside VIIS, X=22.1 ± 6.2).

All juvenile Bluestriped Grunts were found within colonized pavement while adults were also most commonly found in this habitat type (33%; Figure 4.52). Lesser quantities of adult Bluestriped Grunts were found in colonized bedrock (29%), linear reef (24%), and aggregated patch reefs (9%). Coral Bay supported high occurrence and density of juvenile and adult Bluestriped Grunts (Figure 4.53).

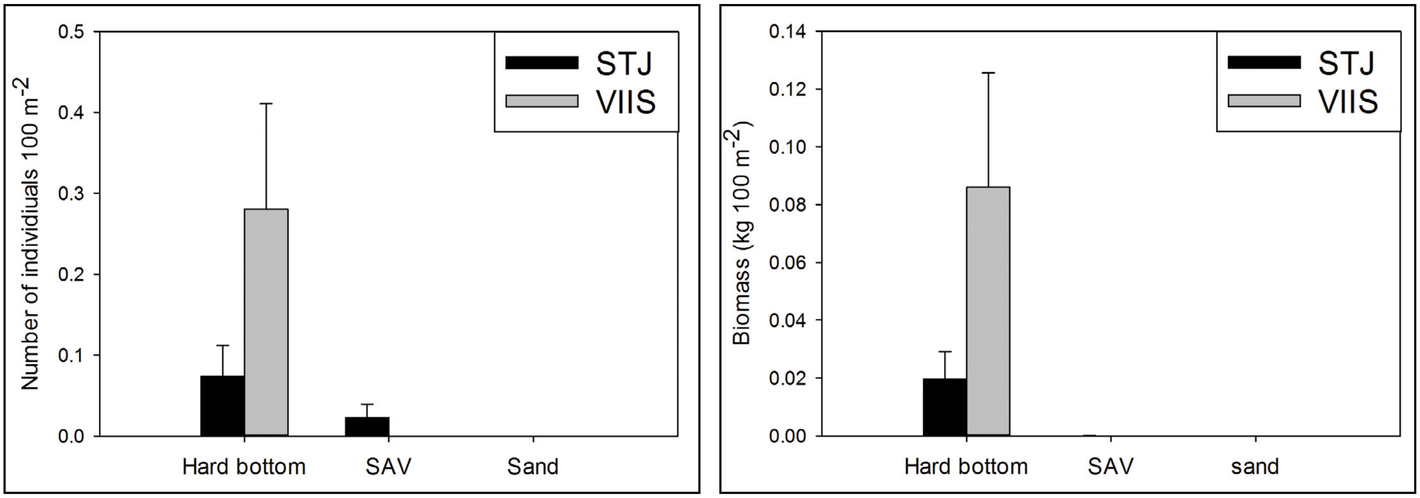


*Figure 4.50. Numerical density and biomass (+SE) of Bluestriped Grunt (*H. sciurus*) by major habitat type inside VIIS and adjacent areas around St. John (STJ).*

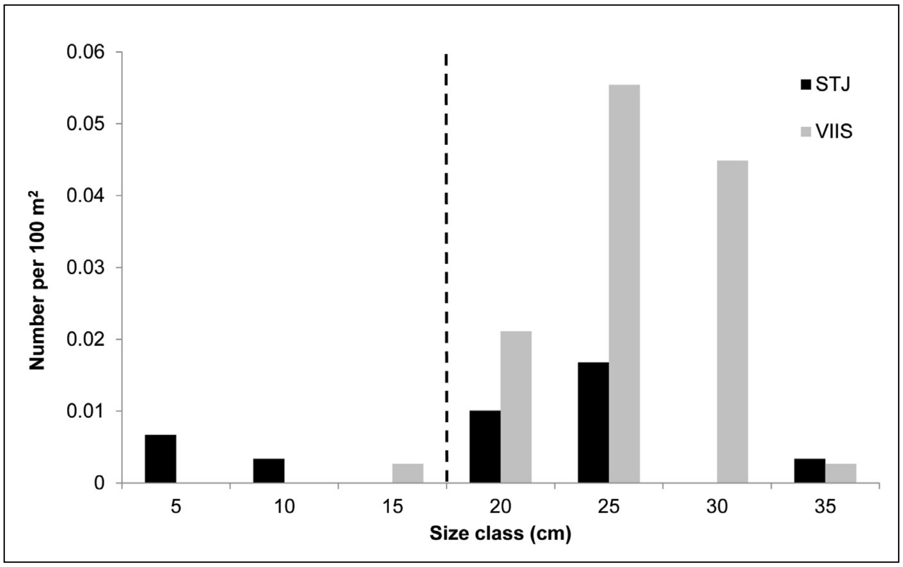


*Figure 4.51. Size frequency distribution for Bluestriped Grunt (*H. sciurus*) inside and outside VIIS. Dashed line represents size class at 1st sexual maturity.*

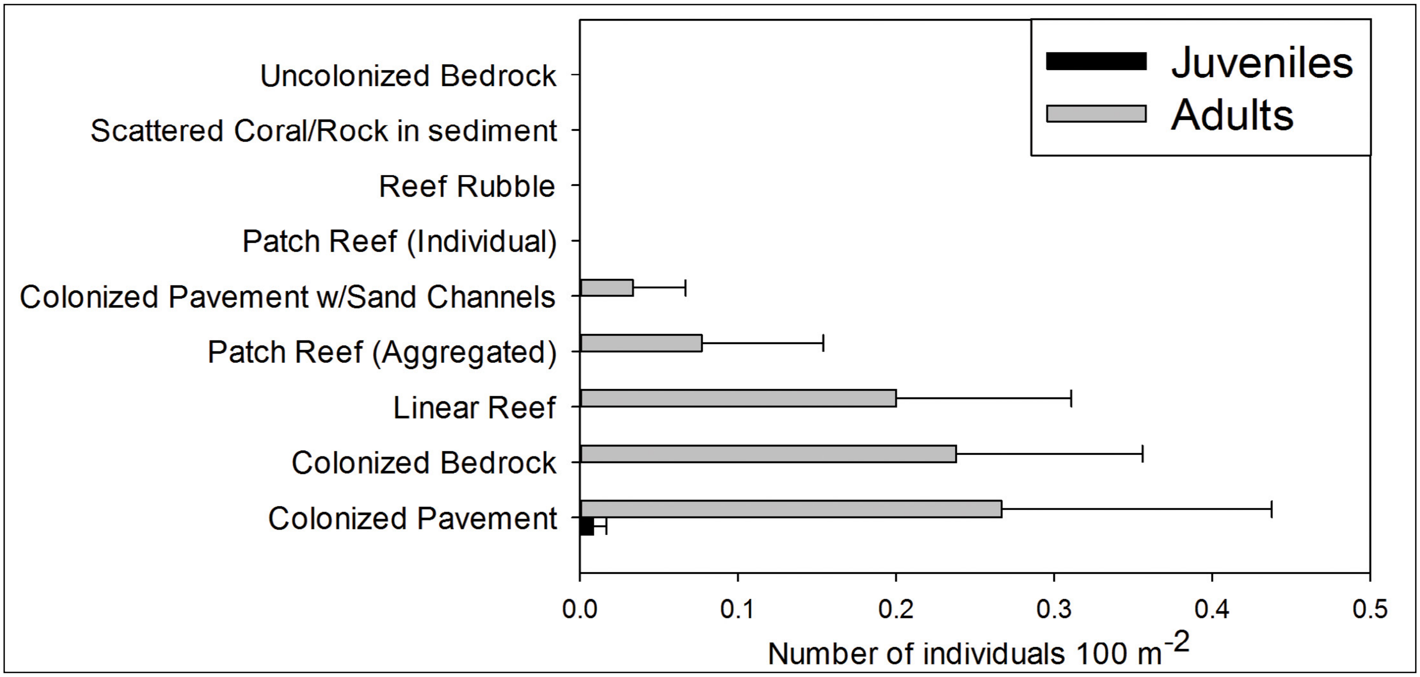


*Figure 4.52. Mean (+SE) density for juvenile/subadult and adult by observer habitat type for Bluestriped Grunt (*H. sciurus*).*

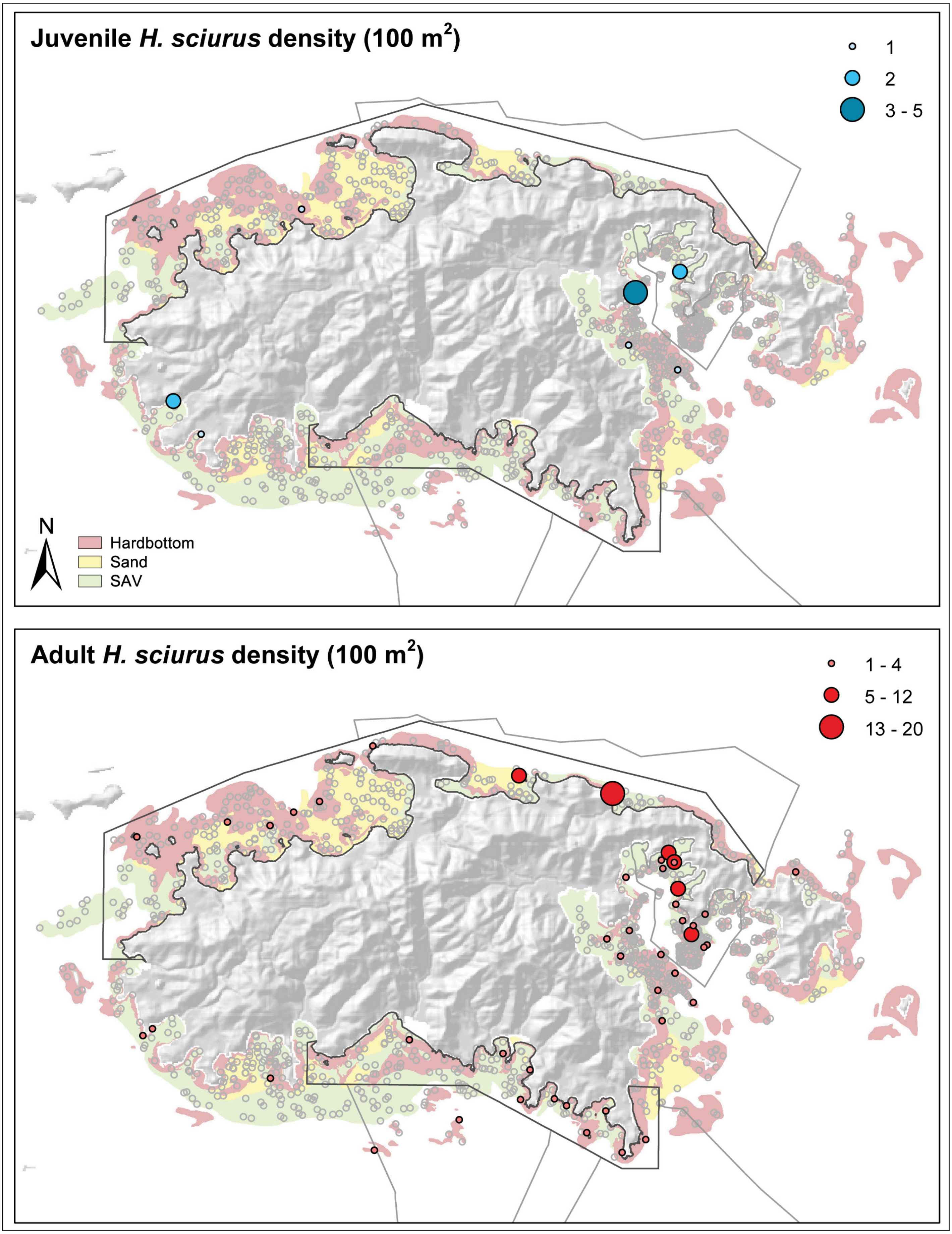


*Figure 4.53. Spatial distributions of juvenile and adult Bluestriped Grunt (*H. sciurus*) around St. John. Source: K. Stamoulis (University of Hawaii).*

Ocean Surgeonfish (*Acanthurus bahianus*)

Ocean Surgeonfishes inhabit shallow reef areas where they can be observed in mixed schools with other surgeonfishes (Figure 4.54). They feed primarily on algae and obtain a maximum size of 38 cm TL. Based on trap surveys conducted by Garrison et al. (2004), Ocean Surgeonfishes accounted for 3.5% of the total abundance and 17% of the surgeonfish abundance in traps around St. John.

*Figure 4.54. Photo of Ocean Surgeonfish (*Acanthurus bahianus*) schooling with Blue Tangs (*A. coeruleus*) in St. John, USVI. Credit: NOAA/NOS/NCCOS/CCMA/ Biogeography Branch.*

Ocean Surgeonfishes were most abundant on hardbottom (82% by number, 91% by weight), followed by SAV (10% by number, 5% by weight), and sand (8% by number, 4% by weight; Figure 4.55). Numerical density and biomass were higher outside VIIS compared with inside (26% and 35%, respectively) while values in SAV and sand were nearly equivalent. Juveniles comprised 92% of all the Ocean Surgeonfishes observed around St. John during the survey period (Figure 4.56). Most Ocean Surgeonfishes observed around St. John were small and average sizes inside (X=9.0 ± 0.3) and outside (X=9.2 ± 0.7) were nearly identical (t=0.37, p=0.71).

Juvenile and adult Ocean Surgeonfish were observed across a wide variety of habitat types (Figure 4.57). Juveniles were most common on colonized bedrock (24%), followed by reef rubble (22%), uncolonized bedrock (10%), and colonized pavement with sand channels (9%; Figure 4.57). Adults were most common in scattered coral/rock in sediment (18%), followed by colonized bedrock (16%), aggregated patch reefs (15%), colonized pavement (15%) and linear reef (14%). Both juveniles and adults were widely distributed around the island of St. John with high occurrence and high density of adults and juveniles in Coral Bay (Figure 4.58).

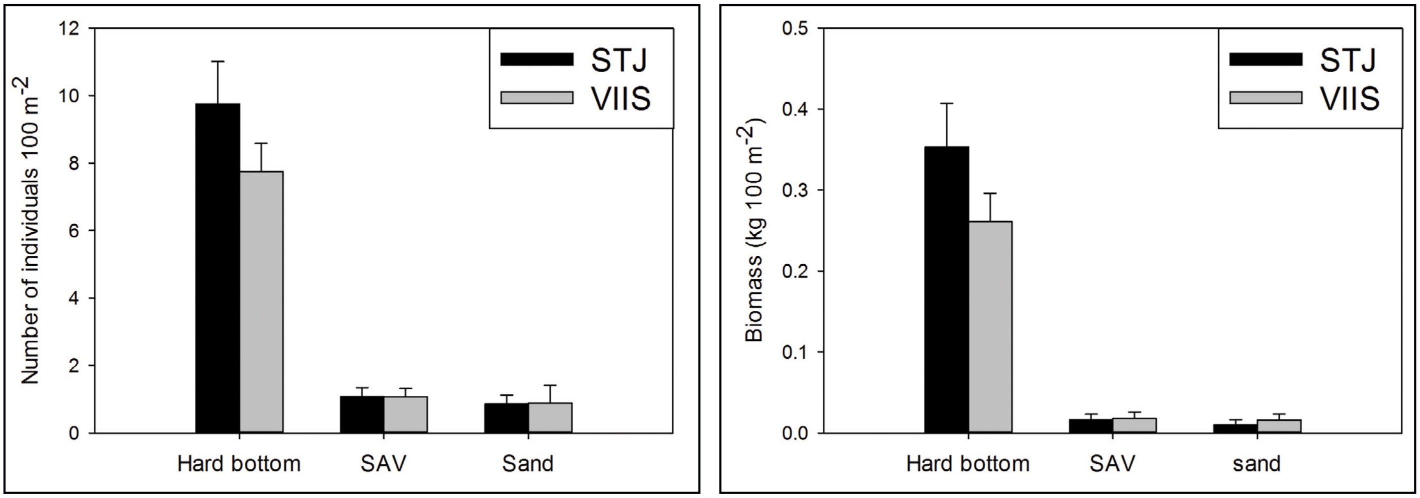


*Figure 4.55. Numerical density and biomass (+SE) of Ocean Surgeonfish (*A. bahianus*) by major habitat type inside VIIS and adjacent areas around St. John (STJ).*

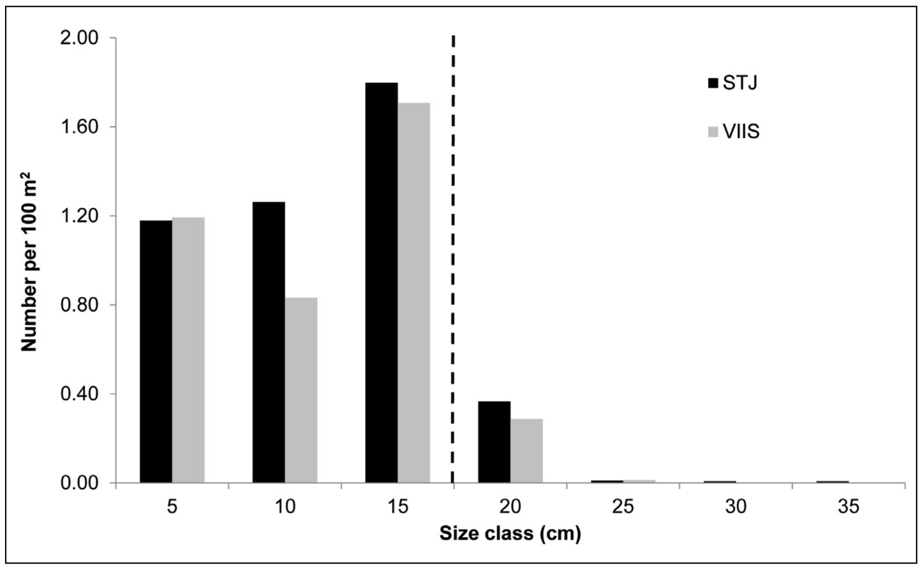


*Figure 4.56. Size frequency distribution for Ocean Surgeonfish (*A. bahianus*) inside and outside VIIS. Dashed line represents size class at 1st sexual maturity.*

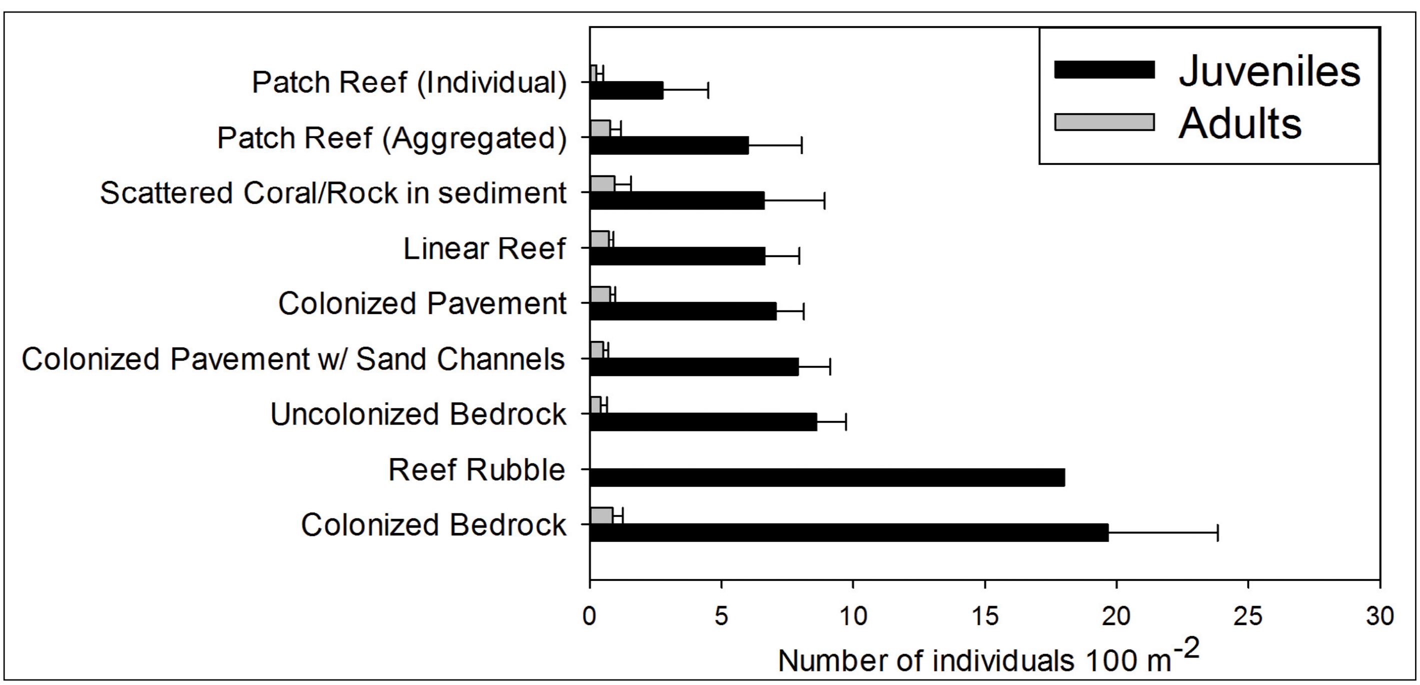


*Figure 4.57. Mean (+SE) density for juvenile/subadult and adult by observer habitat type for Ocean Surgeonfish (*A. bahianus*).*

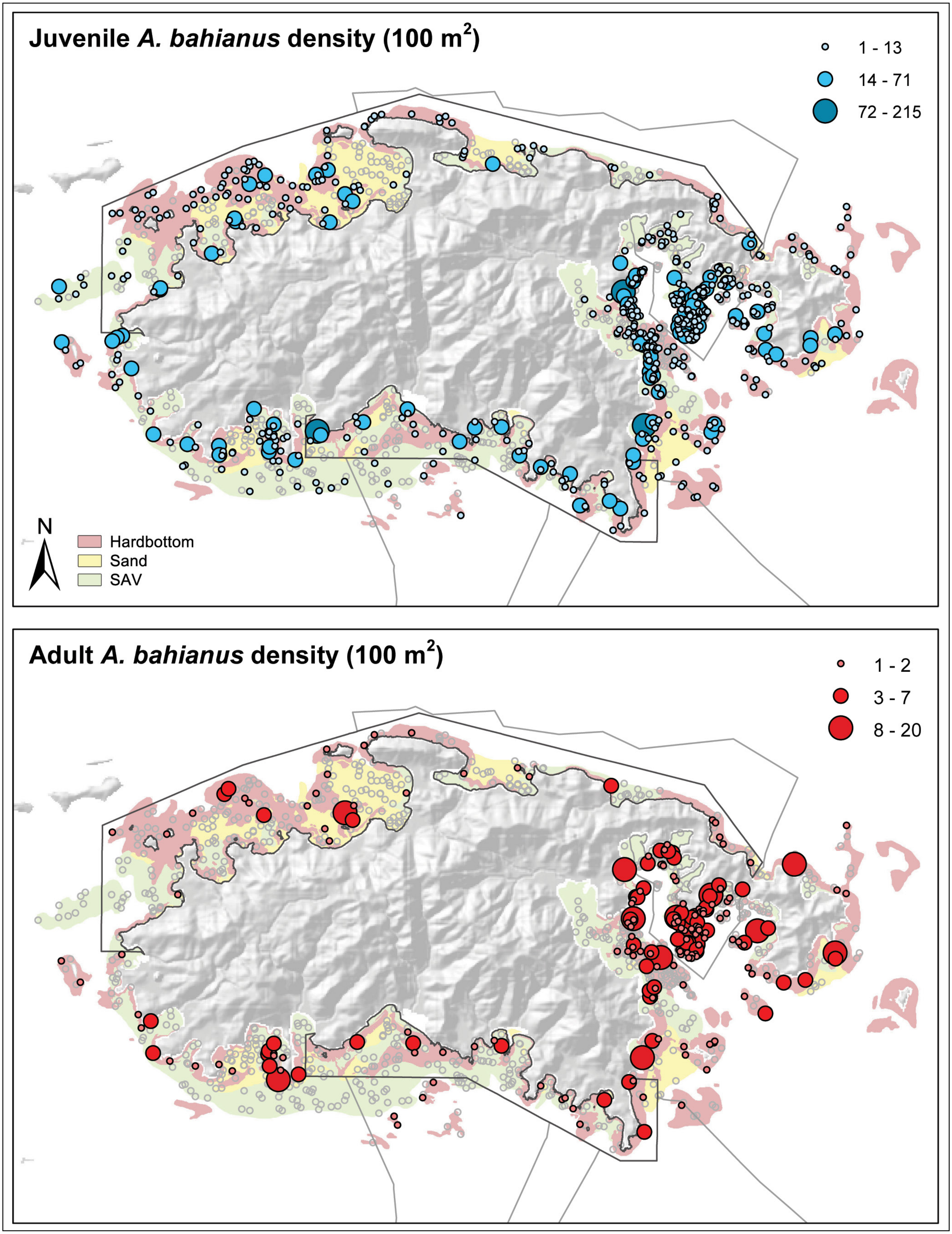


*Figure 4.58. Spatial distributions of juvenile and adult Ocean Surgeonfish (*A. bahianus*) around St. John. Source: K. Stamoulis (University of Hawaii).*

Blue Tang (*Acanthurus coeruleus*)

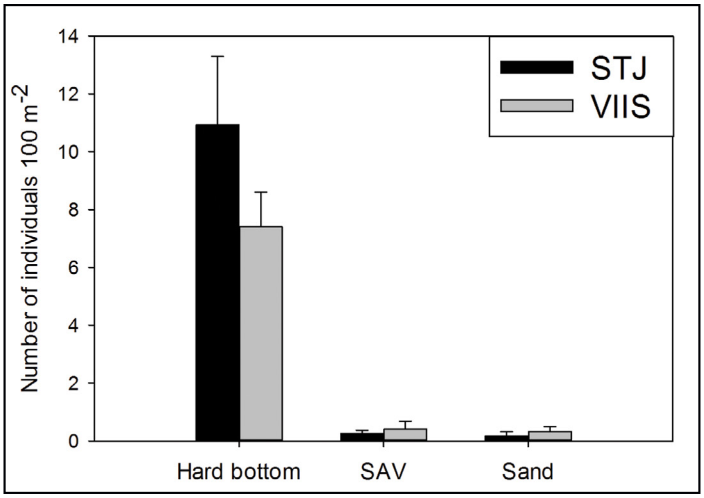
Blue Tangs inhabit shallow reef areas where they can be observed in mixed schools with other surgeonfishes (Figure 4.59). They feed primarily on algae and obtain a maximum size of 39 cm TL. Based on trap surveys conducted by Garrison et al. (2004), Blue Tang accounted for 15% of the total abundance and 75% of the surgeonfish abundance in traps around St. John.

*Figure 4.59. Blue Tang (*A. coeruleus*) in St. John, USVI. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

Slightly more than 94% of the numerical abundance and 93% of the weight of Blue Tang around St. John was observed on hardbottom (Figure 4.60). Numerical abundance and biomass were higher outside compared to inside VIIS (47% higher number, 66% by weight). Values for SAV and sand were nearly equal for both.

Overall, 89% of all Blue Tangs observed around St. John during the survey period were sub-reproductive in size (Figure 4.61). Average size of Blue Tang was not significantly different (t=1.05, p=0.30) inside (X=10.3 ± 8.9) compared with outside (X=10.6 ± 9.8). The same was true when only adults (>10 cm) were considered (t=1.2, p=0.23; inside: X=13.2 ± 4.5; outside: X=13.6 ± 6.0).

Colonized bedrock was the dominant habitat for juvenile Blue Tang (47%), followed by individual patch reefs (11%), colonized pavement (9%), and linear reef (8%; Figure 4.62). Adults were also most abundant on colonized bedrock (24%), followed by linear reef (18%), individual patch reefs (15%), and colonized pavement (14%). Juveniles and adults were widely distributed around the island with some higher concentrations of adults observed in Coral Bay and around the eastern portion of St. John (Figure 4.63).



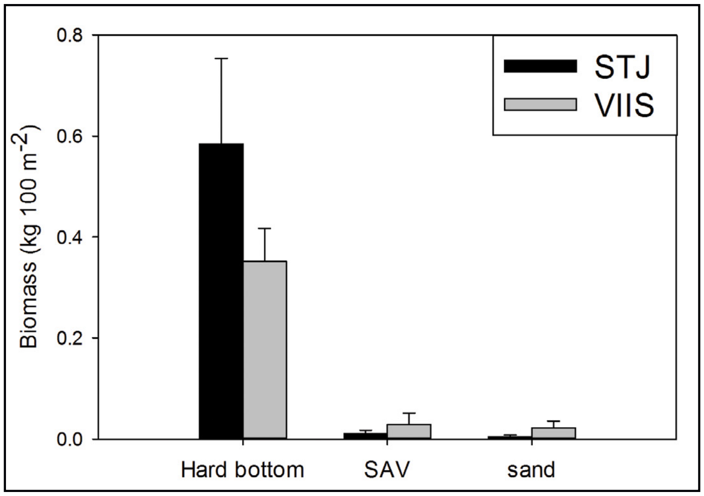


Figure 4.60. Numerical density and biomass (+SE) of Blue Tang (A. coeruleus) by major habitat type inside VIIS and adjacent areas around St. John (STJ).

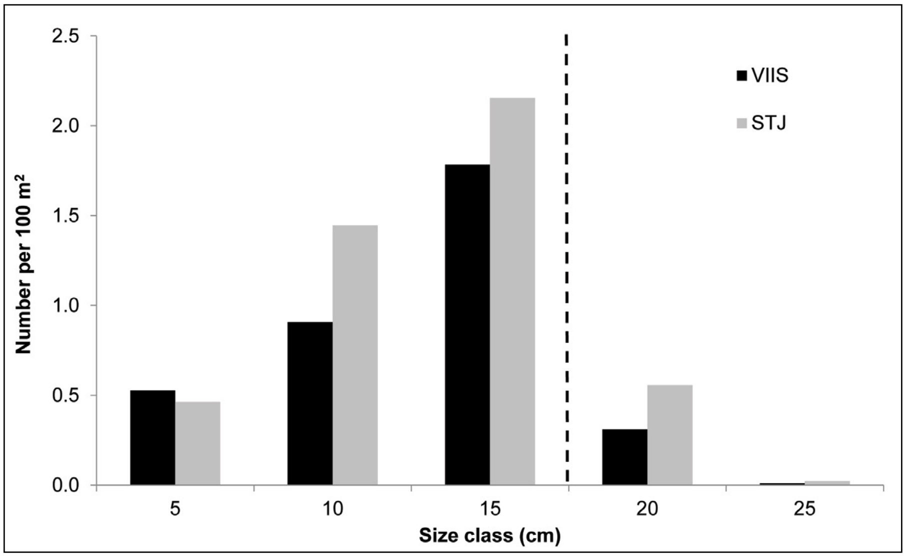


Figure 4.61. Size frequency distribution for Blue Tang (A. coeruleus) inside and outside VIIS. Dashed line represents size class at 1st sexual maturity.

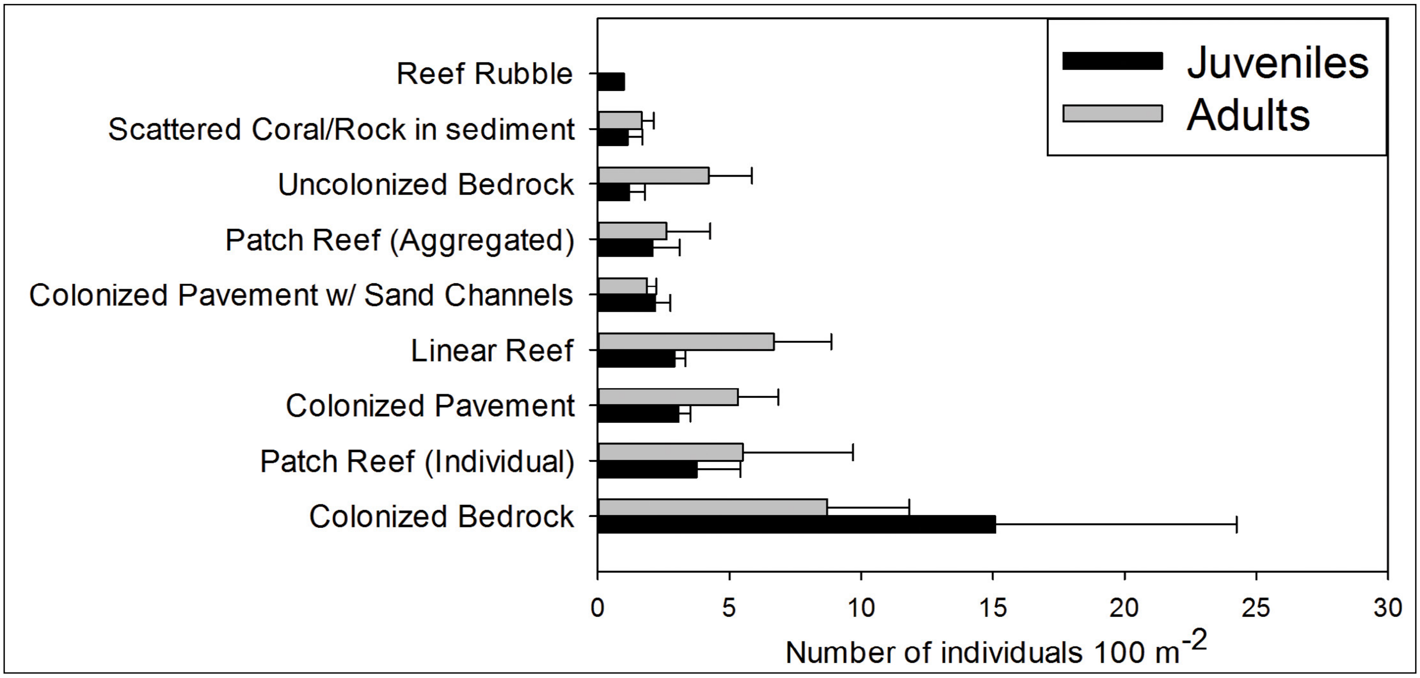


Figure 4.62. Mean (+SE) density for juvenile/subadult and adult by observer habitat type for Blue Tang (A. coeruleus).

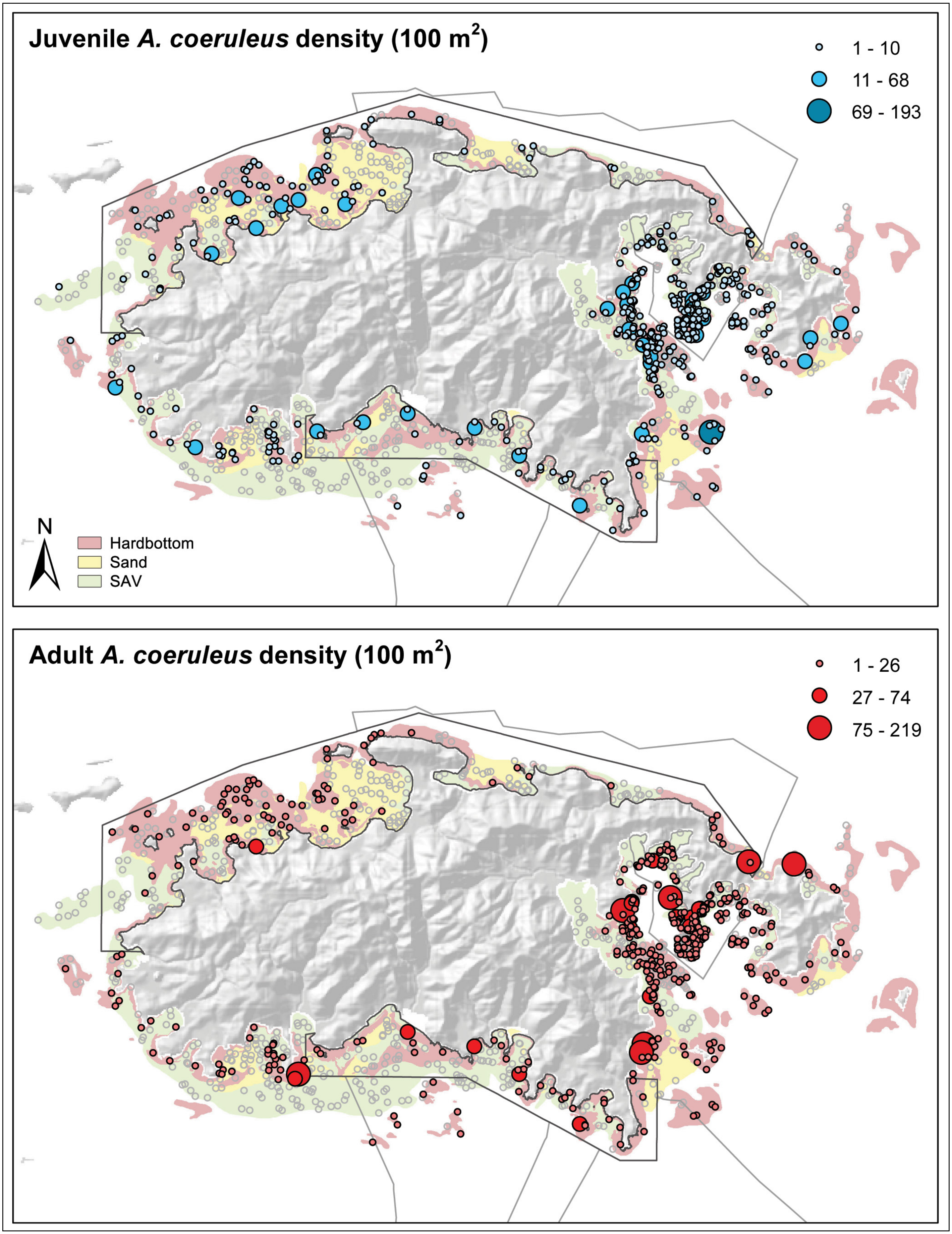


*Figure 4.63. Spatial distributions of juvenile and adult Blue Tang (*A. coeruleus*) around St. John. Source: K. Stamoulis (University of Hawaii).*

Striped Parrotfish (*Scarus iseri*)

Striped Parrotfish are a schooling species that feeds on algae and other plant material (Figure 4.64). They are protogynous hermaphrodites with the terminal males spawning individually with females, while sexually mature males in the striped phase spawn in aggregations.

*Figure 4.64. Photos of juvenile/initial phase Striped Parrotfish (*Scarus iseri*) in St. John, USVI (left) and a terminal phase Striped Parrotfish in La Parguera, Puerto Rico (right). Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

The maximum size is recorded as 35 cm TL. Based on trap surveys conducted by Garrison et al. (2004), Striped Parrotfish accounted for 19% of all parrotfish abundance in traps around St. John.

The majority of the Striped Parrotfish observed were found on hardbottom (86% by number, 90% by weight; Figure 4.65). Among all habitat combined, both numerical abundance and biomass were higher inside compared with outside VIIS (24%, 18%, respectively). Numbers of hardbottom only were 28% higher inside VIIS compared with outside and biomass was 6% higher inside.

Juveniles accounted for more than 94% of all Striped Parrotfish observed around St. John during the survey period (Figure 4.66). When all size classes are considered, the average size of Striped Parrotfish did not differ (t=1.3, p=0.19) inside (X=6.6 ± 12.8) compared with outside (X=7.0 ± 112.0) VIIS. The same trend applied when only adults (>15 cm) were considered (t=1.6, p=0.1; inside: X = 18.8 ± 3.9; outside: X=18.1 ± 2.9).

Juveniles were most common on aggregated patch reefs (23%), followed by linear reef (21%), and individual patch reefs (17%; Figure 4.67). Adults were most common on colonized pavement with sand channels (25%), linear reef (17%), colonized pavement (14%), and individual patch reefs (13%). Both juveniles and adults were widely distributed with higher concentrations observed in Coral Bay and along the northwest portion of the island (Figure 4.68).

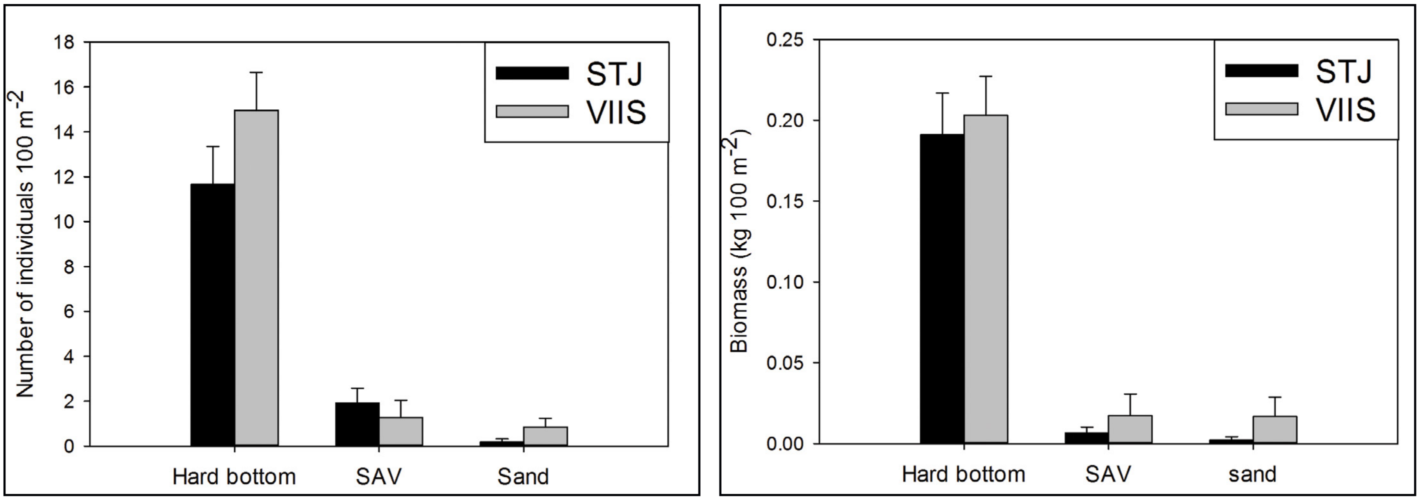


*Figure 4.65. Numerical density and biomass (+SE) of Striped Parrotfish (*S. iseri*) by major habitat type inside VIIS and adjacent areas around St. John (STJ).*

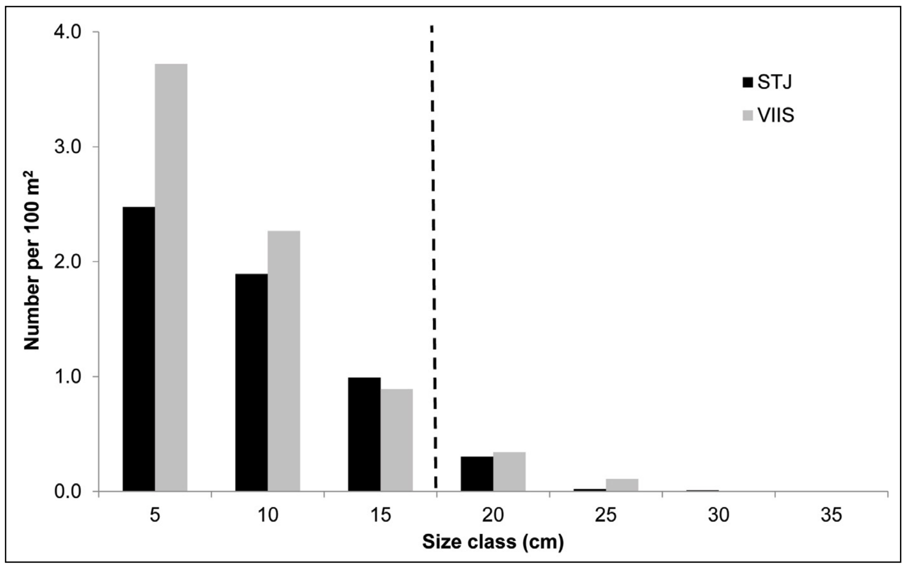


*Figure 4.66. Size frequency distribution for Striped Parrotfish (*S. iseri*) inside and outside VIIS. Dashed line represents size class at 1st sexual maturity.*

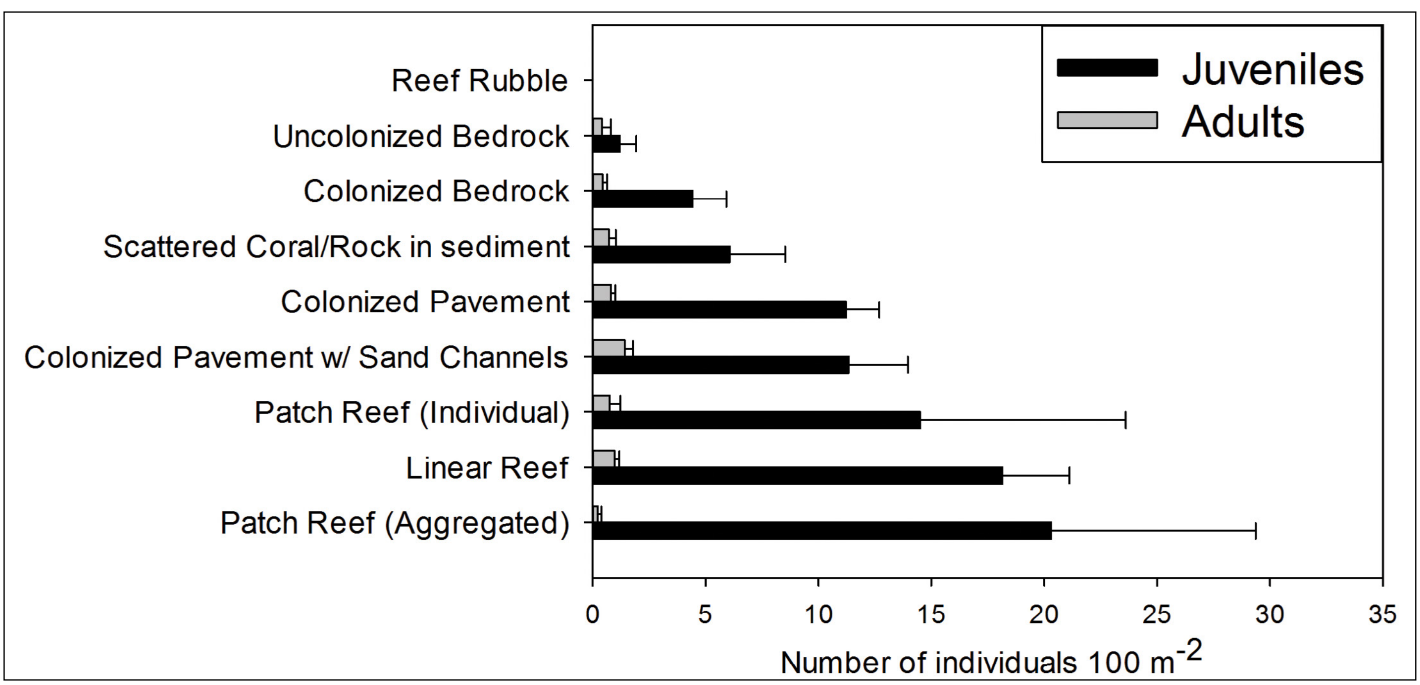


*Figure 4.67. Mean (+SE) density for juvenile/subadult and adult by observer habitat type for Striped Parrotfish (*S. iseri*).*

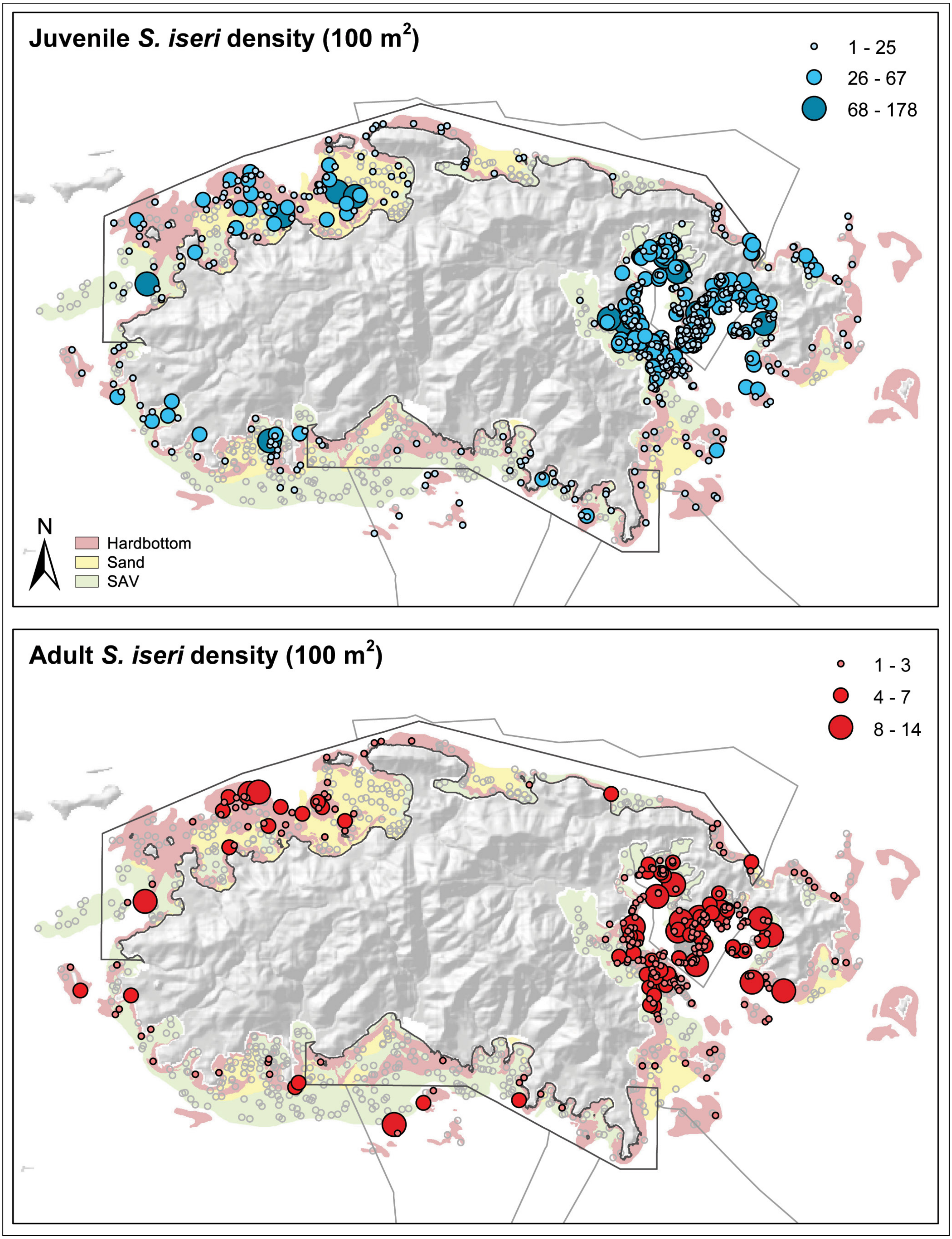


*Figure 4.68. Spatial distributions of juvenile and adult Striped Parrotfish (*S. iseri*) around St. John. Source: K. Stamoulis (University of Hawaii).*

Redband Parrotfish (*Sparisoma aurofrenatum*)

Redband Parrotfish inhabit coral reefs and other hardbottom habitats and feed primarily on algae and other plant material (Figure 4.69). Juveniles are often seen in large schools while adults may occur solitary on in small schools. They are protogynous hermaphrodites and reach a maximum size of 28 cm TL.

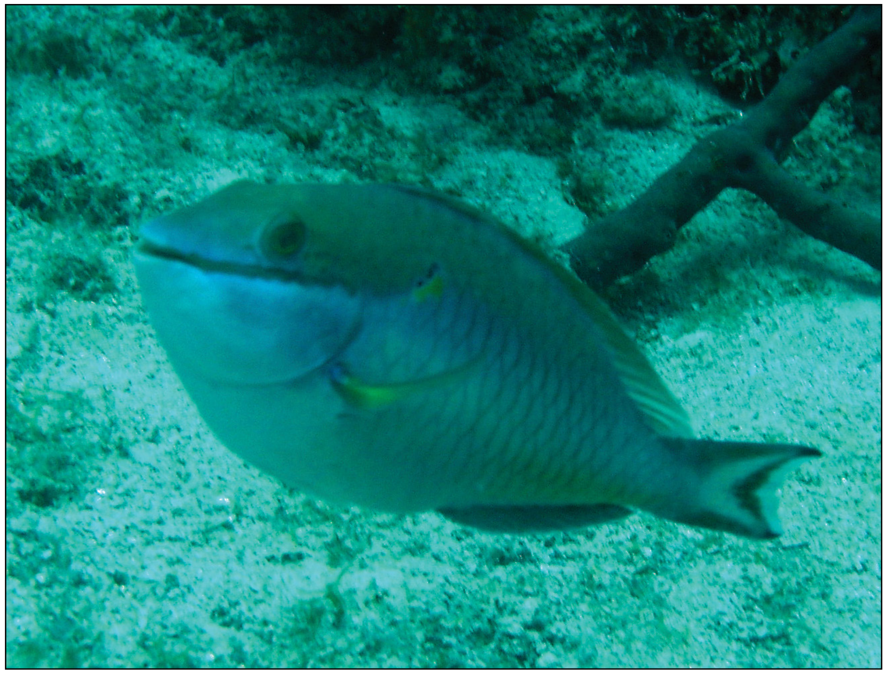


*Figure 4.69. Photo of Redband Parrotfish (*Sparisoma aurofrenatum*) in St. John, USVI. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

The vast majority of the individuals (87%) and biomass (89%) of Redband Parrotfish was observed on hardbottom (Figure 4.70). Overall biomass was 6% higher inside VIIS compared with outside, while numerical abundance outside was 32% than inside VIIS; however, these results were not statistically significant. Although numerical abundance was similar between SAV and sand (7% and 6%, respectively), biomass was greater on sand (9%) compared with SAV (2%).

The majority (82%) of all Redband Parrotfish observed in the study area were juveniles (Figure 4.71). The size distribution of Redband Parrotfish was strongly skewed with the majority of individuals in the smallest size classes. When all size classes were considered, the average size of individuals inside VIIS (X=9.7 ± 10.4) was significantly ($t=3.96$, $p<0.001$) larger than the average size of individuals outside VIIS (X=8.17 ± 10.9) although the absolute difference was small (1.6 cm). When only adult fish (>15 cm) are considered, the average size is still larger inside VIIS (X=19.5 ± 4.1) versus outside VIIS (X=18.8 ± 3.6) but the results are marginal significant ($t=1.9$, $p=0.5$).

Juveniles were most commonly observed in colonized pavement with sand channels (22%), linear reef (21%), and colonized bedrock (16%; Figure 4.72). Adults were most common on individual patch reefs (23%), colonized pavement with sand channels (18%), and linear reef (15%). Both juveniles and adults were distributed across the entire island with high occurrence and density in Coral Bay, Fish Bay, Rendevouz Bay and the reefs in the Johnson Reef area of NW St. John (Figure 4.73).

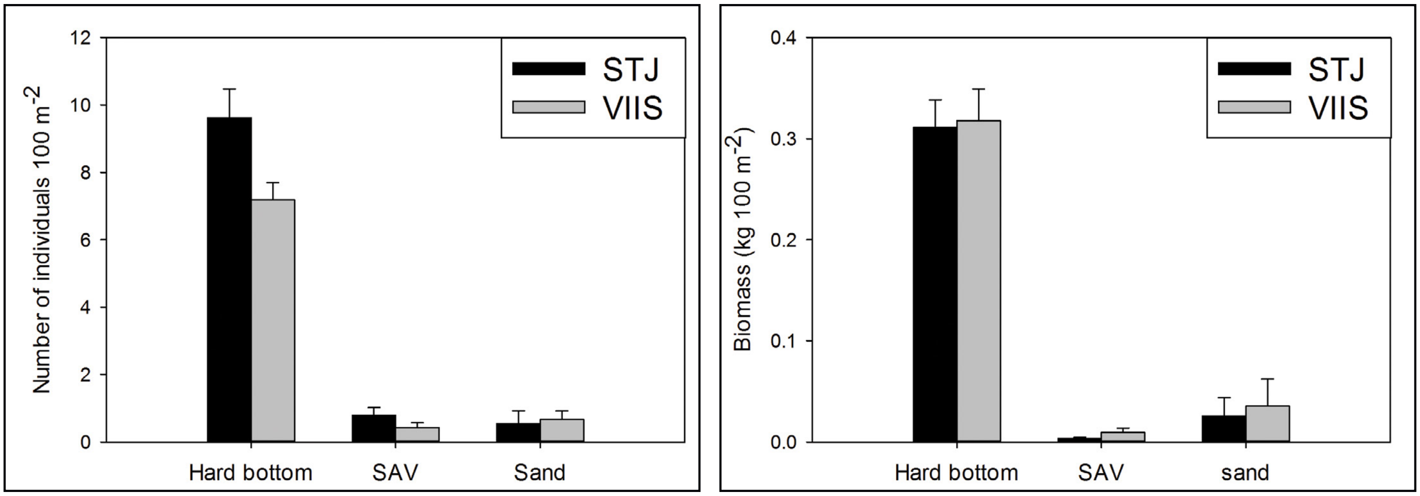


*Figure 4.70. Numerical density and biomass (± SE) of Redband Parrotfish (*S. aurofrenatum*) by major habitat type inside VIIS and adjacent areas around St. John (STJ).*

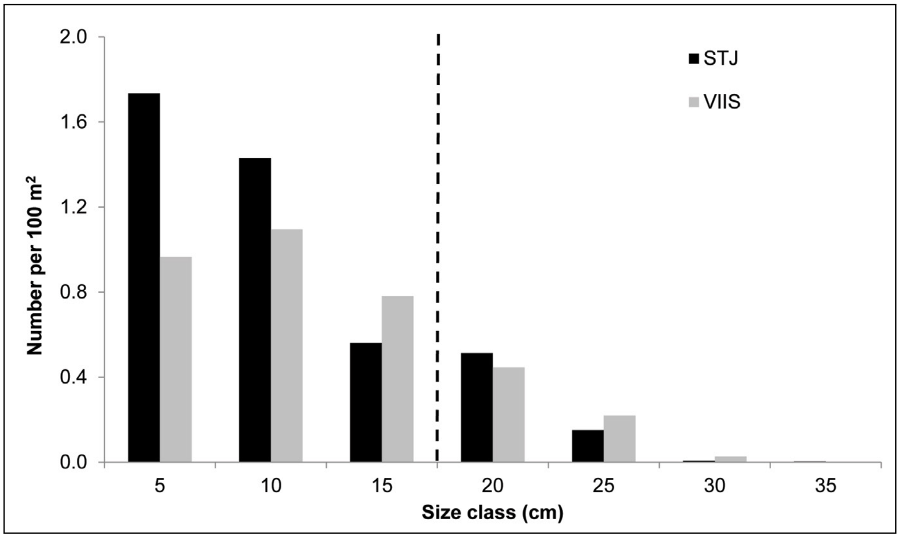


*Figure 4.71. Size frequency distribution for Redband Parrotfish (*S. aurofrenatum*) inside and outside VIIS. Dashed line represents size class at 1st sexual maturity.*

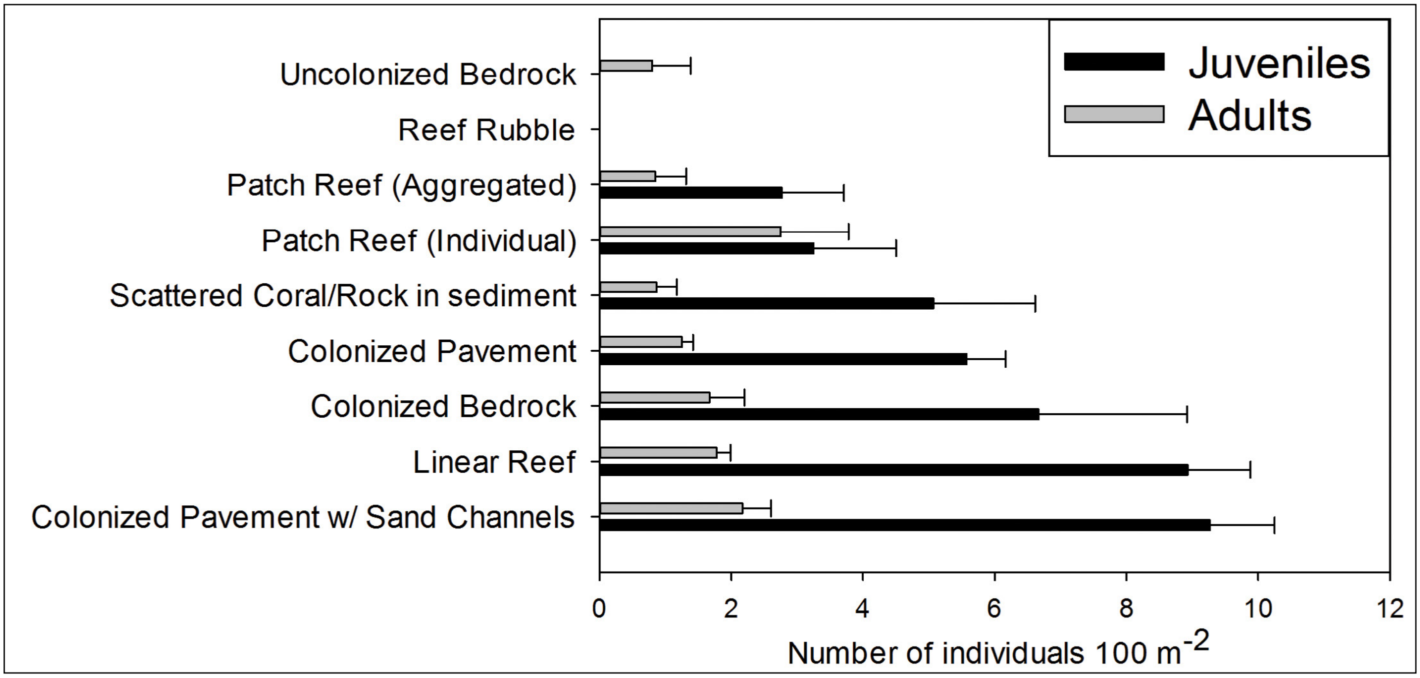


*Figure 4.72. Mean (+ SE) density for juvenile/subadult and adult by observer habitat type for Redband Parrotfish (*S. aurofrenatum*).*

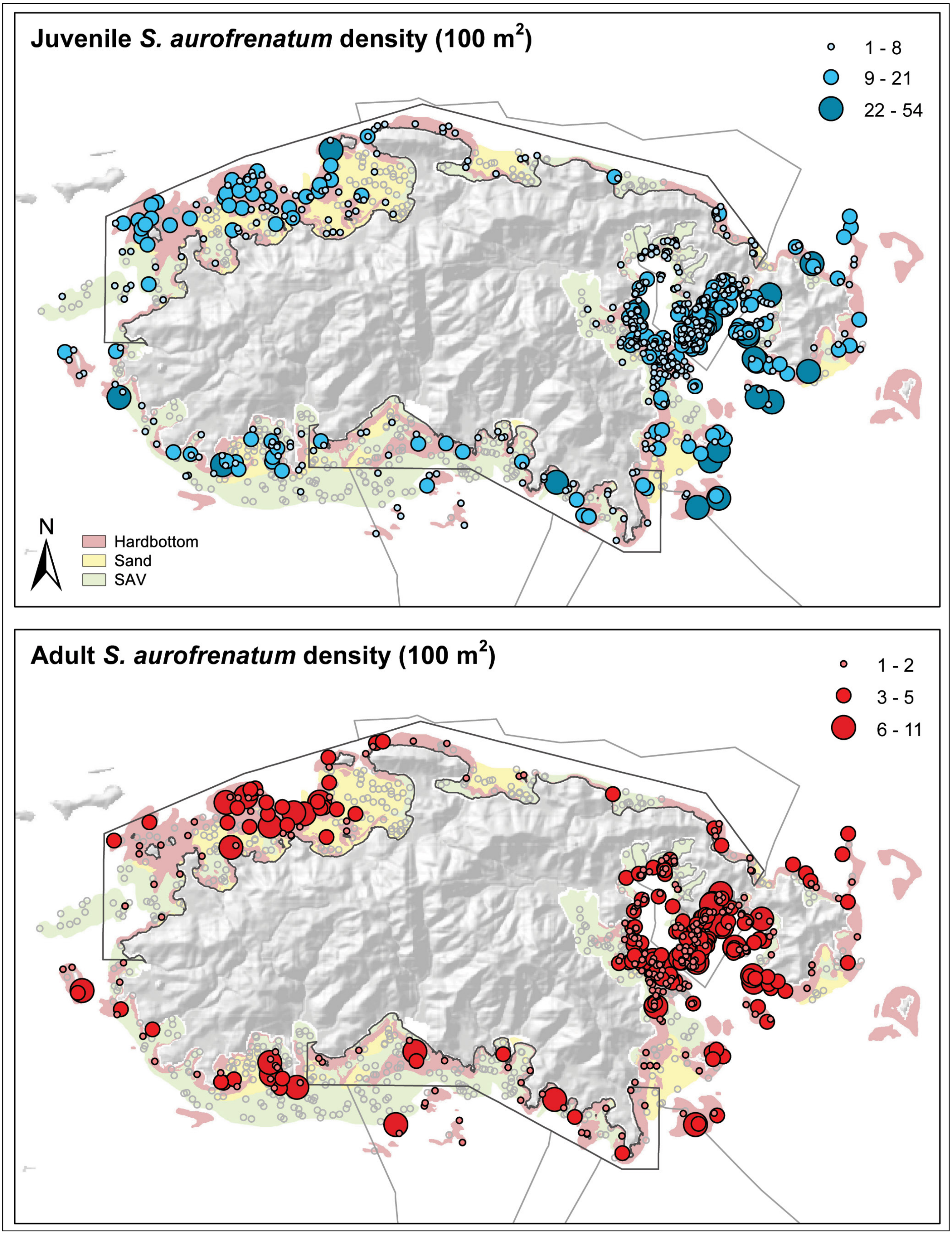


*Figure 4.73. Spatial distributions of juvenile and adult Redband Parrotfish (*S. aurofrenatum*) around St. John. Source: K. Stamoulis (University of Hawaii).*

Queen Triggerfish (*Balistes vetula*)

Queen Triggerfish are found over rocky or coral areas (Figure 4.74). They may form schools, and are sometimes solitary over sand and grassy areas. Queen Triggerfish feed mainly on benthic invertebrates (primarily sea urchins). They were the most abundant species caught in fish traps around St. John in 1999-2000 based on fisheries-independent SEAMAP-C data.

*Figure 4.74. Photo of Queen Triggerfish (*Balistes vetula*) in St. John, USVI. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

Slightly more than 68% of the numerical abundance of Queen Triggerfish was observed on hardbottom with an additional 21% on sand and 11% on SAV (Figure 4.75). However, 91% of the biomass was on hardbottom, highlighting the smaller size of individuals in these other habitat types. Juveniles made up 90% of all individuals observed on softbottom habitats and this likely represents nursery habitat for this species. Adult Queen Triggerfish comprised 54% of individuals observed over the entire study area (Figure 4.76). There was no difference (t=0.11, p=0.91) in the average size of Queen Triggerfish inside (X=23.0 ±11.1) versus outside VIIS (X=23.3 ± 9.0). Average sizes for adults only (>20 cm) were also indistinguishable (t=0.46, p=0.65; inside: X=28.7 ± 6.7; outside: X=27.9 ± 5.3).

On hardbottom habitat, juvenile Queen Triggerfish were most common in colonized pavement with sand channels (36%), followed by linear reef (31%), and aggregated patch reefs (17%; Figure 4.77). Adults were most often found on linear reef (25%), colonized pavement with sand channels (19%), and colonized bedrock (18%). The distribution of Queen Triggerfish was very patchy with juveniles most common along the south shore adjacent to Reef Bay and adults in higher abundance in Coral Bay and along the south shore (Figure 4.78.)

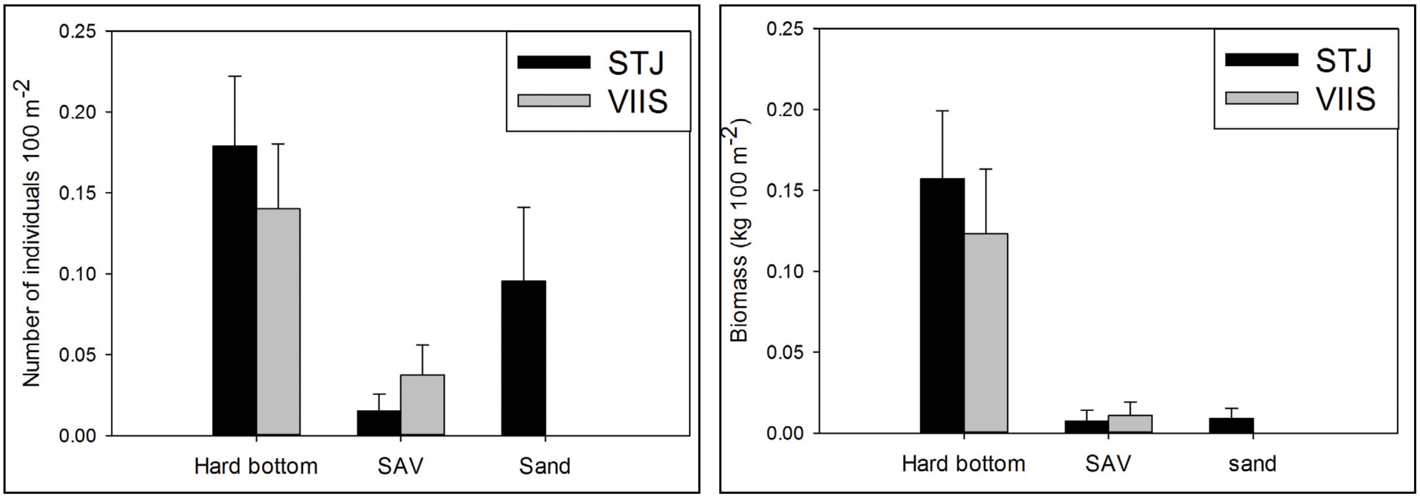


*Figure 4.75. Numerical density and biomass (± SE) of Queen Triggerfish (*B. vetula*) by major habitat type inside VIIS and adjacent areas around St. John (STJ).*

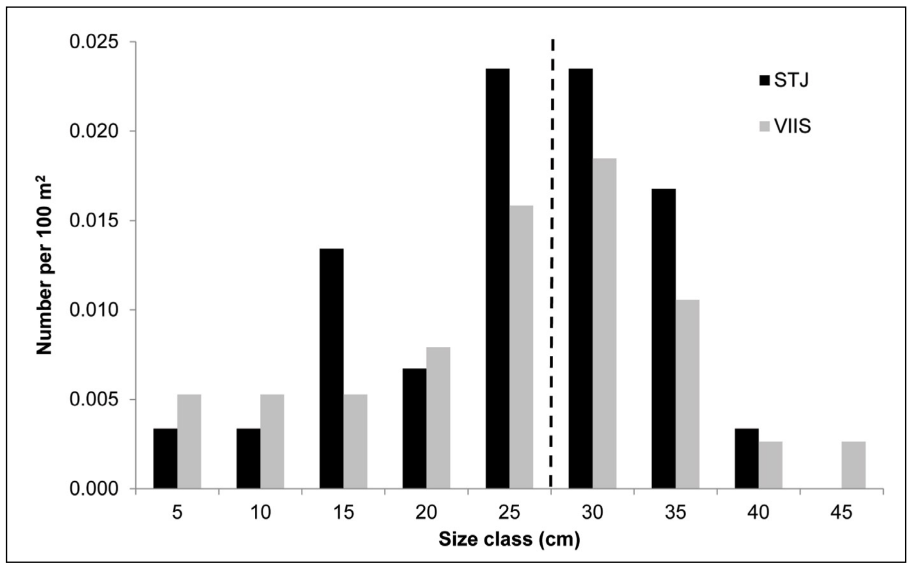


*Figure 4.76. Size frequency distribution for Queen Triggerfish (*B. vetula*) inside and outside VIIS. Dashed line represents size class at 1st sexual maturity.*

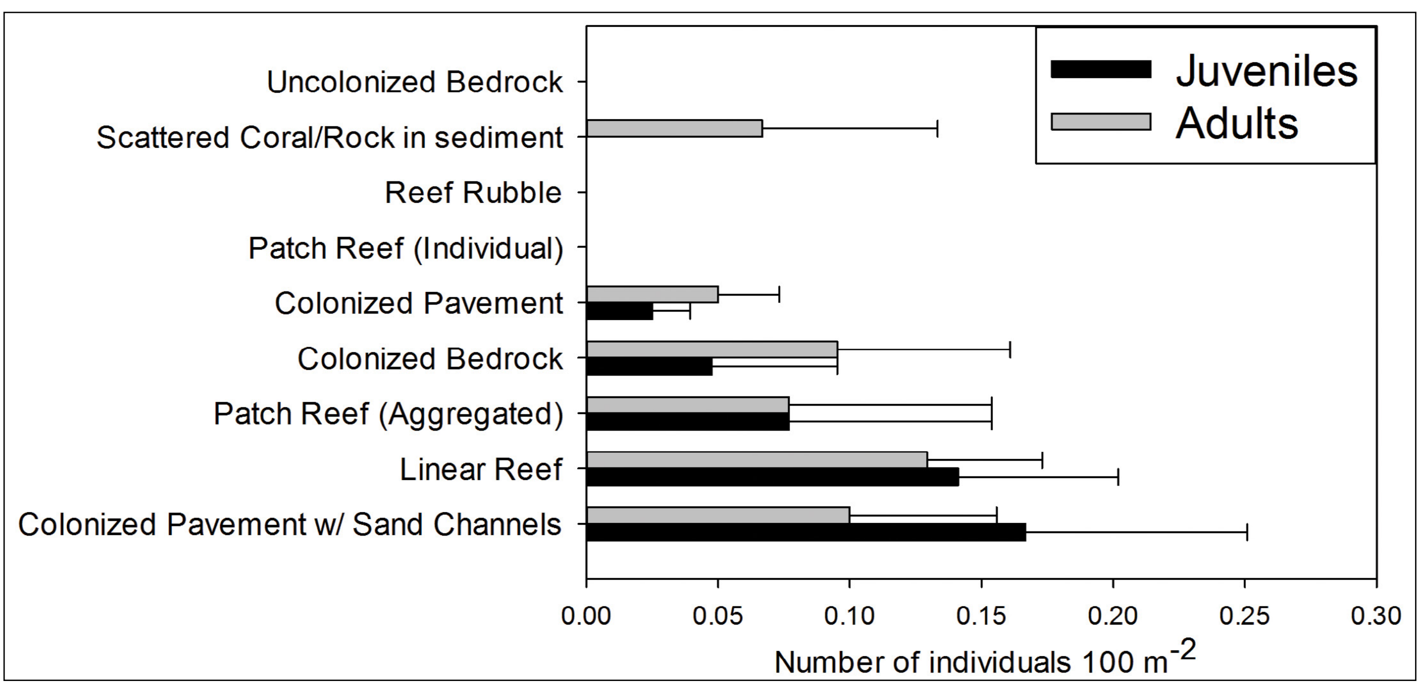


*Figure 4.77. Mean (+ SE) density for juvenile/subadult and adult by observer habitat type for Queen Triggerfish (*B. vetula*)).*

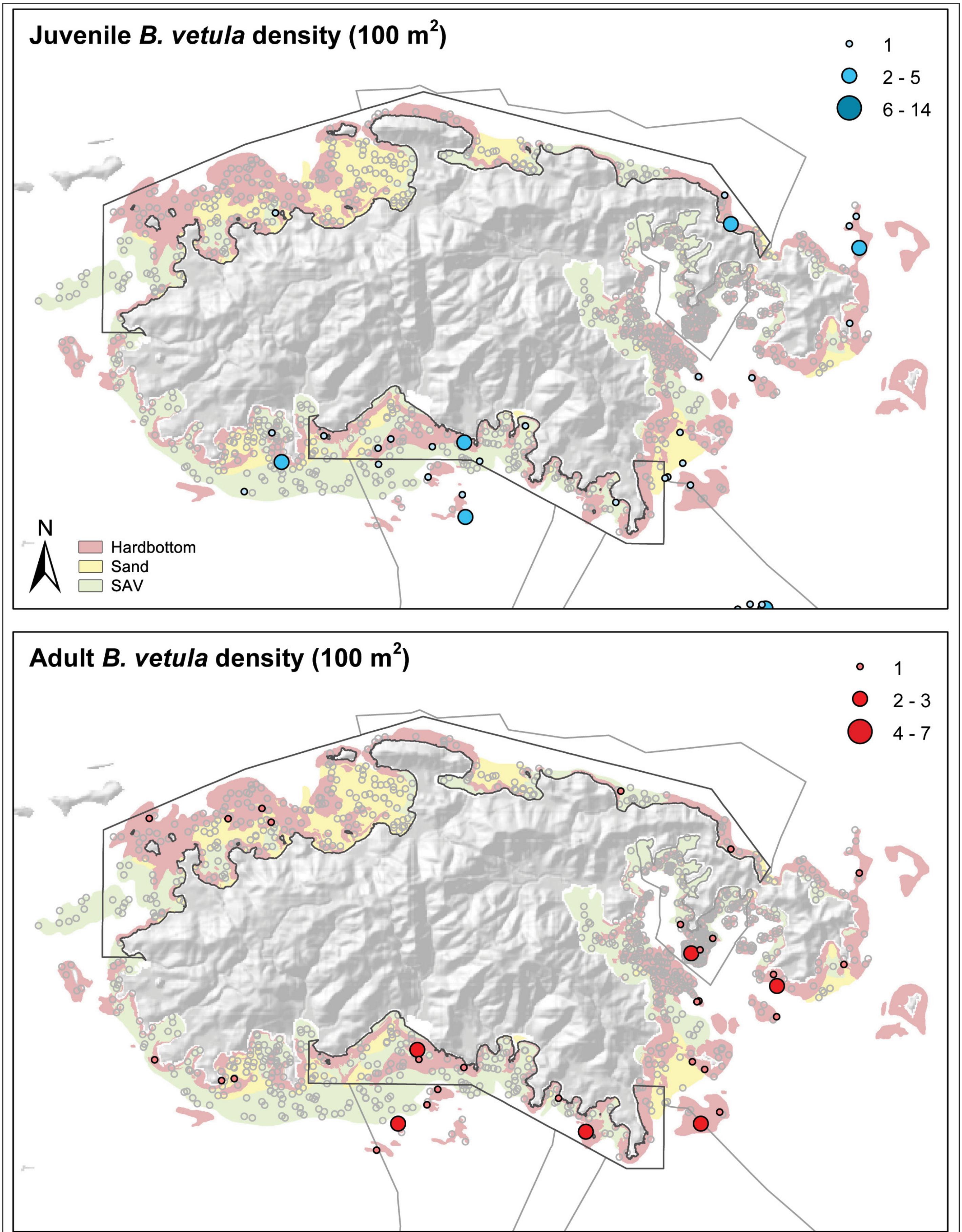


*Figure 4.78. Spatial distributions of juvenile and adult Queen Triggerfish (*B. vetula*) around St. John. Source: K. Stamoulis (University of Hawaii).*

4.3.3. Microhabitat utilization by damselfish

The Threespot Damselfish, *Stegastes planifrons* (Cuvier), is important in mediating interactions among corals, algae, and herbivores on Caribbean coral reefs (Kaufmann 1977). The preferred microhabitat of Threespot Damselfish is thickets of the branching staghorn coral, *Acropora cervicornis*, but within the past few decades, mass mortality of *A. cervicornis* from white-band disease and other factors has rendered this coral a minor ecological component throughout most of its range (Aronson and Precht 2001; Figure 4.79). Survey data from Jamaica (heavily fished), Florida and the Bahamas (moderately fished), the Cayman Islands (lightly to moderately fished), and Belize (lightly fished) indicate that distributional patterns of Threespot Damselfish are positively correlated with live coral cover and topographic complexity (Precht et al. 2010).

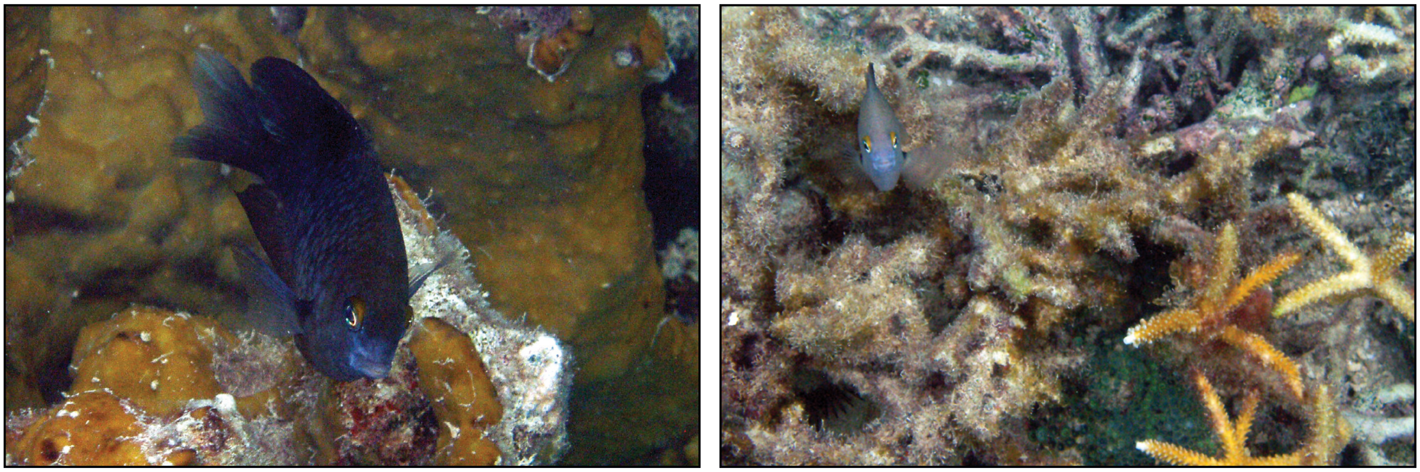


*Figure 4.79. The distinct markings of the Threespot Damselfish (*Stegastes planifrons*; left) and in a* Acropora cervicornis *habitat (right) in La Parguera, Puerto Rico. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

Threespot Damselfish may be a good indicator of areas with relatively high coral cover. To examine this hypothesis further, we plotted the abundance of Threespot Damselfish around St. John with the associated cover of *Montastraea annularis* complex at those same locations. Results show a positive correlation between the cover of *M. annularis* complex and the abundance of Threespot Damselfish (Figure 4.80). The loss of the primary microhabitat of Threespot Damselfish, *A. cervicornis*, has forced a shift in the distribution and recruitment of these damselfish onto remaining high-structured corals, especially the *Montastraea annularis* complex species, affecting coral mortality and algal dynamics throughout the Caribbean (Precht et al., 2010). The loss of structurally complex *Acropora* spp. will likely lead to further degradation of reefs in the Caribbean as Threespot Damselfish are highly territorial and actively kill scleractinian corals by biting the living tissue and cultivating dense algal lawns on the coral skeletons of slow-growing, long-lived, massive corals, like *M. annularis* complex.

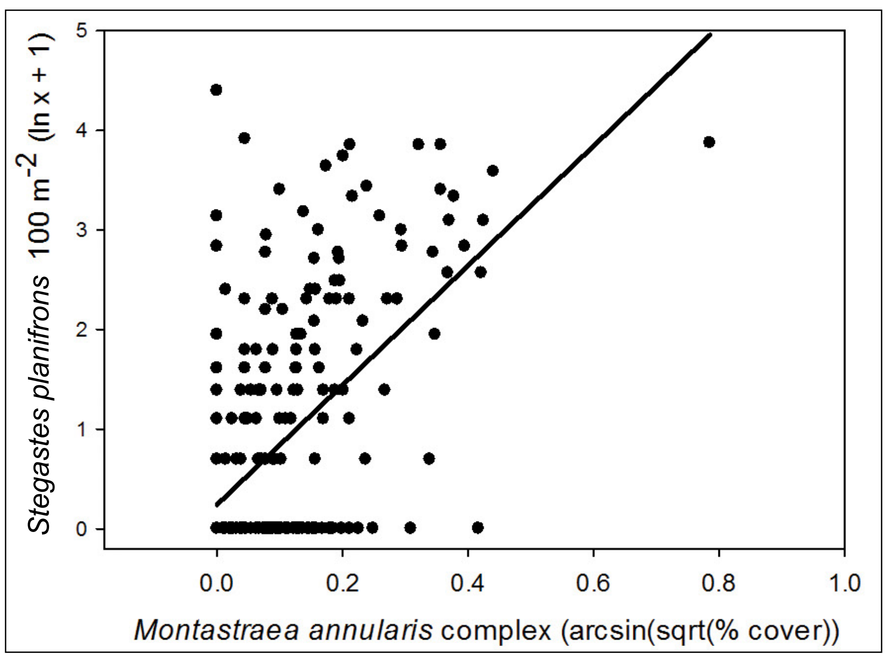


Figure 4.80. Relationship between live cover of Montastraea annularis *complex and the Threespot Damselfish (*S. planifrons*). R^2=0.31, p<0.001, Y=0.28+5.84X.*

4.3.4. Fish metrics of Coral Bay

Overall, fish assemblage characteristics within Coral Bay were similar inside and outside the VICR (Table 4.11.). Fish biomass was 27% higher inside the monument compared with outside and significantly different (p = 0.05). All other metrics were indistinguishable. Fish species richness within Coral Bay was highest along Johnson's Reef and Round Bay (Figure 4.81). Higher biomass was observed near Turner Point while higher diversity was centered around Johnson's Reef, Turner Point and Long Point.

Table 4.11. Fish assemblage characteristics on hard bottom between management strata within Coral Bay. Values are means ± SD.

Assemblage characteristic	Outside Coral Bay Monument	Inside Coral Bay Monument	T	P
Number of species	22.0 (5.9)	21.8 (5.3)	0.28	0.79
Number of individuals/100 m$^{2\backslash}$	1.4 (1.1)	1.5 (1.1)	0.81	0.42
Biomass (kg/100 m^2)	4.4 (3.0)	5.6 (7.2)	2.00	0.05
Diversity	2.3 (0.4)	2.3 (0.9)	0.59	0.55

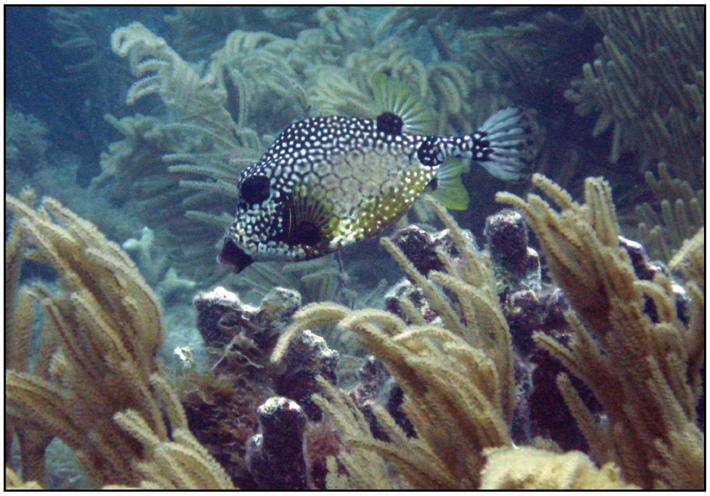


*Figure 4.81. Examples of various species observed and recorded in Coral Bay, St. John, such as Yellowtail Parrotfish (*Sparisoma rubripinne; *left) and Smooth Trunkfish (*Lactophrys triqueter*; right). Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

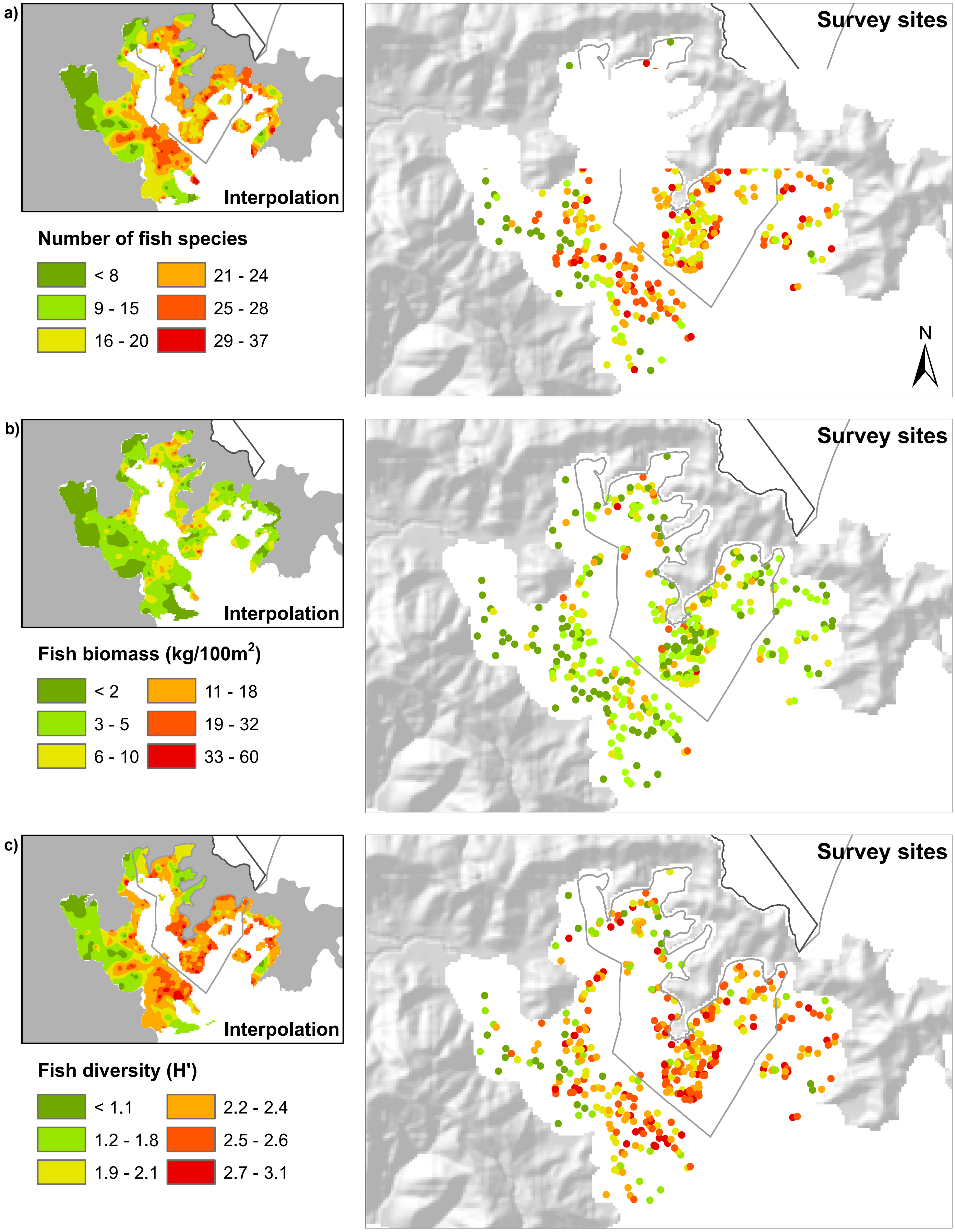


Figure 4.82. Spatial distribution of fish assemblage characteristics within Coral Bay between 2001 and 2009. (a) abundance; (b) biomass; and (c) diversity. Source: K. Stamoulis (University of Hawaii).

4.4. DISCUSSION

This study produced a spatially and temporally comprehensive examination of the fish assemblages around St. John, USVI. One major result of this work was that there are no clear differences in the fish assemblages inside versus outside VIIS, despite 40+ years of "protection" within VIIS. Although a number of species examined showed higher abundance inside VIIS, the overall trends showed no statistical difference. Overall fish biomass in hardbottom habitats was 52 g/m^2, with no difference between management strata. This value lies at the lower end of estimates of reef fish standing stock across the Caribbean (Newman et al., 2006) and likely reflects deterioration of habitat and past and current levels of fishing within the USVI, although direct measures of fishing effort are currently lacking. For many of the commercially important resource species observed around St. John, sub-reproductive juveniles made up a large proportion of the individual species populations. This lack of large individuals of reproductive size is a serious concern for the long-term viability of these populations.

4.4.1. Effects of fishing

Large predatory fishes such as groupers and snappers are uncommon throughout the nearshore USVI and some spawning aggregations have been decimated (Beets and Friedlander, 1999; Beets, 1997; Rogers et al., 2008). In this study, piscivores made up <10% of the total fish biomass and larger groupers were extremely rare (Figure 4.83). Only one Yellowfin Grouper and one Nassau Grouper were found in 677 surveys; both occurred outside VIIS. In a companion study, twelve Nassau grouper were observed during 379 surveys within the VICR; nine of them were seen at Coral Bay and three of same fish were found at the MSR (unpublished data). The two small groupers (Graysby and Coney) made up 75% of the total Epinepheline grouper abundance.

*Figure 4.83. Photo of Nassau Grouper (*Epinephelus striatus*) observed outside the transect in St. John. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

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; Jennings and Polunin, 1997). Studies by Chiappone et al. (2000) in fished and unfished areas in Florida showed that the abundance and biomass of small-bodied groupers (e.g., Coney and Graysby) were higher in areas where fishing had reduced the large-bodied groupers (e.g., *Epinephelus* spp., *Mycteroperca* spp.). 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 Nassau Grouper and have been found to have a more significant impact on the recruitment to patch reefs for a wide range of common fish species. Comparisons of trap data from the early to late 1990s showed major declines in a number of grouper, snapper, and grunt species with increases in small, lower-order trophic level species. The fishing down of the food web by removing higher trophic levels can result in relaxation of top-down control that can lead to a phase transition to ecosystems dominated by lower trophic guilds (Pauly et al., 1998).

Large parrotfishes, such as Blue Parrotfish (*Scarus coeruleus*) and Rainbow Parrotfish (*Scarus guacamaia*), were also extremely rare around St. John with only two individuals of each species observed, and only in 2002 and 2003. Since 1988, only seven individuals in total of these two large and important herbivores have been observed at long-term sampling locations around St. John (Friedlander and Beets, 2008; Friedlander, unpub. data). These species were once common around St. John (Randall 1963) and were likely important in the control of macroalgae due to their large size.

4.4.2. Fish assemblage structure

Overall, most fish assemblage metrics (e.g., richness, numerical abundance, biomass, and diversity) were highest in Coral Bay, along the north shore between Haulover and Newfound Bays, and along the south shore between Lameshur and Salt Pound Bays with the area on the north shore around Johnson's Reef and Trunk and Cinnamon Bays also possessing high fish diversity. High coral cover, along with high habitat complexity and high habitat diversity typified these locations and were positively correlated with most of these fish assemblage metrics. Our study highlights the local significance of Coral Bay for species of the snapper family (Lutjanidae). It is likely that the combination of mangroves, seagrasses and structurally complex coral reefs works synergistically to provide the resources required by snapper (Pittman et al. 2007). These areas have high conservation value and should be managed accordingly.

*Figure 4.84. Wrasses, such as Creole Wrasse (*Clepticus parrae*), are a few species of fish that contribute to most of the numerical abundance (left) and schools of Blue Tangs (*Acanthurus coeruleus*) and Ocean Surgeonfish (*Acanthurus bahianus*) are two of the few species that contribute to much of the biomass (right) in the St. John study region. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

Wrasses, juvenile parrotfishes, and damselfishes made up most of the numerical abundance of fishes around St. John, while larger parrotfishes and surgeonfishes accounted for much of the biomass (Figure 4.84). The absence of groupers, snappers, and grunts among the top species by dominance (% biomass x frequency of occurrence) is a cause for concern since these predators can exert a strong top-down control on the entire coral reef ecosystem and are importance in maintaining ecosystem function (Figure 4.85).

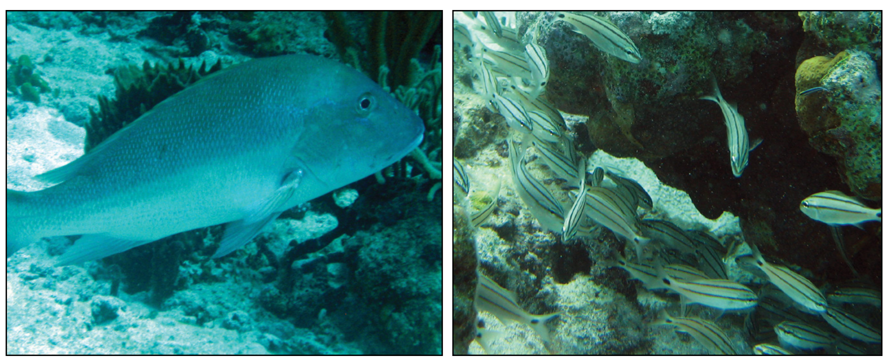


*Figure 4.85. Mutton snapper (*Lutjanus analis*; left) and grunts (in juvenile stage) have reduced in dominance around the St. John study region. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

4.4.3. Spatial distributions and fish-habitat associations

This project also evaluated the utility of the NOAA benthic habitat map classes for predicting differences in fish assemblage structure across multiple habitat classes at different levels of a hierarchical map classification. It was expected that fish assemblages would differ significantly among habitat types due to the structural differences observed during aerial photo interpretation and via underwater observations, which originally resulted in the delineation and classification of distinct map classes. 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 SAV).

Fish assemblages within hardbottom habitats showed high concordance, while sand habitat possessed highly variable assemblages. Of all pairwise habitat comparisons, fish assemblages associated with sand and hardbottom habitats showed the highest dissimilarity (least overlap). At finer thematic resolutions considerable overlap was detected in fish assemblage structure, particularly within hardbottom habitat types, where high similarity occurred between geomorphologically different classes such as patch reefs, linear reefs and colonized bedrock (Figure 4.86). Although differences sometimes occurred, they were not significant and thus we conclude that the assemblage composition did not respond strongly to the different biophysical structures delineated for remote imagery.

*Figure 4.86. Photo of a Barred Hamlet (*Hypoplectrus puella*) observed within a hardbottom habitat in St. John. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

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. Although direct evidence of habitat connectivity cannot be explicitly inferred from our underwater surveys, the current work does demonstrate that many fish species use multiple habitat types. 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. Snappers, grunts, and parrotfishes showed the greatest segregation of adult and juvenile habitat and highlight the importance of linking habitats. Species that have evolved to use all habitat types (seascape generalists) were 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.

Coral Bay appeared to be an important juvenile habitat particularly for several commercially important fisheries species such as Yellowtail Snapper, Schoolmaster Snapper, and several species of parrotfishes (Figure 4.87). Current efforts to reduce sediment loads within the watershed have the potential to improve coastal ecosystem condition in Coral Bay. The importance of Coral Bay as a nursery habitat for many resource species highlights the need to conserve this area and develop appropriate management strategies to improve ecosystem health.

*Figure 4.87. Photo of juvenile Schoolmaster Snapper (*L. apodus*) observed in Coral Bay, St. John. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

The strong evidence for multiple habitat use in tropical marine fishes 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). Lessons from landscape ecology provide a new perspective in marine conservation as habitats within the seascape are not isolated but set within a broader seascape context in which organisms and processes are interacting (Weins, 2009). This seascape approach provides valuable and spatially discrete ecological information that can be integrated in geographic information systems (GIS) and ultimately provide support for the proper configuration of improved marine protected area (MPA) design (Pittman et al., 2007; Wedding et al., 2008).

4.4.4. Temporal trends

Fish assemblage metrics were highly variable without any obvious long-term increasing or decreasing trend over time. Therefore, this study provides no evidence for a positive MPA effect on replenishment of fish populations or diversity during the decade long sampling period (2001-2009). Although a major bleaching event in 2005 had a negative impact on coral cover, the decline in fish species richness was temporary and other assemblage characteristics were unaffected by this event. Total biomass was variable among years but asynchronous between management strata. Declines in fish assemblages due to fishing occurred long before the initiation of this study (Randall, 1963; Beets and Rogers, 2002) and current results reflect temporal variation in a highly altered system.

4.4.5. Summary

Diversity hotspots

- Fish species richness and diversity were high throughout hardbottom habitats in Coral Bay and the areas on the north shore around Johnson's Reef and Trunk and Cinnamon Bays also possessed high fish diversity.

Fish assemblages

- Within VIIS and around St. John, 227 fish taxa from 56 families were recorded. The most frequently observed species was the Slippery Dick which occurred in 55% of all transects, followed by the Ocean Surgeonfish that occurred in 47% of all transects. An additional 25 species were recorded in the offshore waters in and around VICR.
- Species richness on hardbottom habitats averaged 22.6 species per transect with no significant difference between VIIS and outside areas. Numerical abundance averaged 1.7 individuals/m^2 on hardbottom and biomass averaged 55 g/m^2. Neither of these metrics showed any significant difference between VIIS and outside areas. Diversity on hardbottom was significantly higher inside VIIS (2.35) compared with outside (2.24; $p=0.02$)
- Invertivores accounted for 50% of the total fish biomass, followed by herbivores (30%), piscivores (11%), and planktivores (9%). On hardbottom habitats, invertivores accounted for 43% of the total biomass, followed by herbivores (41%), planktivores (9%), and piscivores (7%). Most of the invertivore biomass consisted of small wrasses.
- A positive MPA effect on fish population biomass was not detected. Fish assemblage metrics were highly variable among years with no consistent and observable increase or decrease over time during the study period.
- Coral Bay appeared to be an important juvenile habitat particularly for several commercially important fisheries species such as Yellowtail Snapper, Schoolmaster Snapper, and several species of parrotfishes.
- There was a significant and positive correlation between the Threespot Damselfish and cover of *Montastraea* spp. corals. Threespot Damselfish may therefore be an indicator of coral health.

- Large-bodied groupers were extremely rare during the study. Only one Yellowmouth (*Mycteroperca interstitialis*) and one Nassau (*Epinephelus striatus*) Groupers were observed outside VIIS; nine Nassau occurred in Coral Bay.

4.4.6. Recommendations

A biogeographic process using GIS technology and sampling across the range of habitats present within the seascape has allowed for robust assessment and monitoring of the marine ecosystem within VIIS, VICR and surrounding waters. This information establishes a comprehensive baseline for the entire marine ecosystem surrounding the island of St. John and is useful to the territorial government to help guide future management decisions. The approach taken in this study is similar to those conducted in Buck Island Reef National Monument (BUIS; Pittman et al. 2008), the Reserva Natural de la Parguera, Puerto Rico (Pittman et al., 2010).

Long-term monitoring is necessary to determine the magnitude of the apparent declines for some species and also to track the trajectory of recovery for other species that exhibited an increase in density after several years of decline. This is critical given the inherent natural variability documented during this study and in similar ecosystems around the world. Long-term monitoring efforts may also reveal direction in the change for the many species that were too highly variable from year to year to provide such information over the nine years of data used.

Benthic habitat maps should be periodically updated due to the dynamic nature of coral reef ecosystems. This is particularly important when linking fish seascape structure and when assessing seascape change such as quantifying gain or loss of major habitat types. In addition, acoustic tracking studies may reveal the mechanisms underlying some of the observed temporal changes in fish communities and will determine connectivity between lagoons and coral reefs offshore. Tracking will also provide important information on the time that individual fish spend inside and outside the boundaries of protected areas. Additional mapping, inventory and monitoring efforts are required to explore the deeper water ecosystems around VIIS and VICR boundaries that exist outside NOAA's current benthic habitat maps.

This body of work will greatly contribute to informed decision making in Coastal and Marine Spatial Planning (CMSP) through a spatially-explicit and quantitative description of resource distributions including species of concern and biodiversity patterns. CMSP in the U.S. Caribbean is being implemented through the Regional Ocean Partnerships which recognized that to effectively manage the increasing ocean uses requires "a proactive planning system that will consider multiple uses, reduce conflicts in resource use, and identify usage areas that are both economically efficient for development and ecologically less vulnerable to impacts". Implementation of CMSP in the region will involve making use of best-available data on resource distributions. The data sets utilized in the current report provides the most detailed information available on marine biological distributions around St. John, but also can be used to develop fish-habitat models that are applicable to other areas (Pittman et al., 2007). Fish hotspots and coldspots were mapped by Pittman et al. (2007) and recent studies in neighboring Puerto Rico indicate that similar techniques can also be used to forecast the ecological impacts from future environmental degradation of coral reefs (Pittman et al., 2012). Such data and associated maps will be necessary to avoid damage to sensitive marine habitats and species when citing marine renewable energy installations or when laying subsea cables or dredging, fishing, boating, or coastal development, etc. Fish distribution and body size data can also be used to track status and trends in fish populations and to assess MPA efficacy, particularly for no-take areas, as well as to assess population viability for exploited, endangered and threatened species and to evaluate ecological drivers for species distributions and diversity patterns.

4.5. REFERENCES

Aronson, R.B. and W.F. Precht. 2001. White-band disease and the changing face of Caribbean coral reefs. Hydrobiologia 460:25–38.

Beets, J. 1997. Can coral reef fish assemblages be sustained as fishing intensity increases? pp. 2009-2014. In: H.A. Lessios and I.G. Macintyre (eds.), Proceedings of the 8th International Coral Reef Symposium, Vol. 2. Panama City, Panama. 2118 pp.

Beets, J. and A.M. Friedlander. 1992. Stock analysis and management strategies for red hind, *Epinephelus guttatus*, in the U.S. Virgin Islands. Annu. Proc. Gulf Caribb. Fish Inst. 42:66-79.

Beets, J., A. Friedlander, and W. Tobias. 1994. Stock assessment of the coney, *Epinephelus fulvus*, in the U.S. Virgin Islands. Annu. Proc. Gulf Caribb. Fish. Instit. 43:395-408.

Beets, J. and A. Friedlander. 1999. Evaluation of a conservation strategy: a spawning aggregation closure for red hind, *Epinephelus guttatus*, in the Virgin Islands. Environ. Biol. Fish. 55:91-98.

Beets, J. and C. Rogers. 2000. Changes in fishery resources and reef fish assemblages in a Marine Protected Area in the US Virgin Islands. pp. 449-454. In: M.K. Moosa, S. Soemodihardjo, A. Soegiarto, K. Romimohtarto, A. Nontji, Soekarno, and Suharsono (ed.), Proceedings of the 9th International Coral Reef Symposium, Vol. 1. Bali, Indonesia. 600 pp.

Beets J. and C. Rogers. 2002. Changes in fishery resources and reef fish assemblages in a marine protected area in the U.S. Virgin Islands: the need for a no take marine reserve. pp. 449-454. In: M.K. Moosa, S. Soemodihardjo, A. Soegiarto, K. Romimohtarto, A. Nontji, Soekarno, and Suharsono (ed.), Proceedings of the 9th International Coral Reef Symposium, Vol. 1. Bali, Indonesia. 600 pp.

Beets, J., L. Muehlstein, K. Haught, and H. Schmitges. 2003. Habitat connectivity in coastal environments: patterns and movement of Caribbean coral reef fishes with emphasis on bluestriped grunt, *Haemulon sciurus*. Gulf Caribb. Res. 14:29-42.

Bohnsack J.A., D.L. Sutherland, A. Brown, D.E. Harper, D.B. McClellan. 1986. An Analysis of the Caribbean Biostatistical Database for 1985. NOAA/NMFS/Southeast Fisheries Center, Coastal Resources Division, Contribution No. 86/87–10. 35 pp.

Boulon, R.H., M.E. Monaco, A.M. Friedlander, C. Caldow, J. Christensen, C. Rogers, J. Beets, J. Miller, and S.D. Hile. 2008. An ecological correction to marine reserve boundaries in the U.S. Virgin Islands. pp. 1082-1085. In: B.M Riegl and R.E. Dodge (eds.), Proceedings of the 11th International Coral Reef Symposium, Vol. 2. Ft. Lauderdale, FL. 742 pp.

Caribbean Fisheries Management Council (CFMC). 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. Caribbean Fishery Management Council, San Juan, Puerto Rico. 178 pp. Online: http://www.caribbeanfmc.com/fmp%20REEF%20FISH/RF%20FMP.pdf (Accessed 14 May 2013)

Cervigón, F. 1993. Los peces marinos de Venezuela. Volume 2. Fundación Científica Los Roques, Caracas,Venezuela. 497 pp.

Chiappone M., R. Sluka, and K.S. Sealey. 2000. Groupers (Pisces: Serranidae) in fished and protected areas of the Florida Keys, Bahamas and northern Caribbean. Mar. Ecol. Prog. Ser. 198:261-272.

Clarke, K.R. and R.M. Warwick. 1994. Change in marine communities: an approach to statistical analyses and interpretation. Plymouth Marine Laboratory, Plymouth, UK. PRIMER-E Ltd.

Clarke, K.R. and R.M. Warwick. 1998. A taxonomic distinctness index and its statistical properties. J. Appl. Ecol. 35:523-531.

Clarke, K.R. and R.M. Warwick. 1999. The taxonomic distinctness measure of biodiversity: weighting of step lengths between hierarchical levels. Mar. Ecol. Prog. Ser. 184:21-29.

Collette, B.B. and S.A. Earle (eds.). 1972. Results of the Tektite Program: Ecology of Coral Reef Fishes. Science Bulletin 14 (Los Angeles). Natural History Museum Los Angeles County. Los Angeles, CA. 180 pp.

Faith, D.P., P.R. Minchin, and L. Belbin. 1987. Compositional dissimilarity as a robust measure of ecological distance. Vegetation 68:57-68.

Friedlander, A.M. and J. Beets. 2008. Temporal Trends in Reef Fish Assemblages inside Virgin Islands National Park and around St. John, U.S. Virgin Islands, 1988-2006. NOAA Technical Memorandum NOS NCCOS 70. Silver Spring, MD. 60 pp. Online: http://www.ccma.nos.noaa.gov/publications/VITempTrends.pdf (Accessed 14 May 2013)

García-Cagide, A., R. Claro, and B.V. Koshelev. 1994. Reproducción. pp. 187-262. In R. Claro (ed.) Ecología de los peces marinos de Cuba. Inst. Oceanol. Acad. Cienc. Cuba. and Cen. Invest. Quintana Roo (CIQRO) México. 525 pp.

Garrison, V.H., C.S. Rogers, and J. Beets. 1998. Of reef fishes, overfishing and *in situ* observations of fish traps in St. John, U.S. Virgin Islands. Rev. Biol. Trop. 46 Supl. 5:41-59.

Garrison, V.H., C.S. Rogers, J. Beets, and A.M. Friedlander. 2004. The habitats exploited and the species trapped in a Caribbean island trap fishery. Environ. Biol. Fish. 71:247-260.

Heemstra, P.C. and J.E. Randall 1993. FAO Species Catalogue - Vol. 16. Groupers of the world (Family Serranidae, Subfamily Epinephelinae): An annotated and illustrated catalogue of the grouper, rockcod, hind, coral grouper and lyretail species known to date. FAO Fish. Synop. 125(16). 382 pp.

Helfman, G.S., J.L. Meyer, and W.N. McFarland. 1982. The ontogeny of twilight migration patterns in grunts (Pisces: Haemulidae). Anim. Behav. 30:317–326.

Hitt, S., S.J. Pittman, and K.A. Brown. 2011. Tracking and mapping sun-synchronous migrations and diel space use patterns of *Haemulon sciurus* and *Lutjanus apodus* in the U.S. Virgin Islands. Environmental Biology of Fishes. 92(4):525-538.

Idyll, C.P. and J.E. Randall. 1959. Sport and commercial fishing potential of St. John, Virgin Islands. pp. 1-9. In: Fourth International Gamefish Conference.

International Game Fish Association (IGFA). 2001. Database of IGFA angling records until 2001. IGFA, Fort Lauderdale, USA. Online: http://www.igfa.org/ (Accessed 14 May 2013)

Jennings, S. and N.V.C. Polunin. 1997. Impacts of predator depletion by fishing on the biomass and diversity of non-target reef fish communities. Coral Reefs 16:71-82

Kaufman, L.S. 1977. The threespot damselfish: effects on benthic biota of Caribbean coral reefs. pp. 559-564 In: D.L. Taylor (ed.), Proceedings of the 3rd International Coral Reef Symposium, Vol. 1. Miami, FL. 656 pp.

Kendall, M.S., J.D. Christensen, and Z. Hillis-Starr. 2003. Multi-scale data used to analyze the spatial distribution of French grunts, *Haemulon flavolineatum*, relative to hard and soft bottom in a benthic landscape. Environ. Biol. Fish. 66:19-26.

Legendre, P. and L. Legendre. 1998. Numerical ecology, 2nd ed. Elsevier Science, Amsterdam. 853 pp.

Magurran, A.E. 1988. Ecological diversity and its measurement. Princeton University Press, Princeton. 179 pp.

Menza, C., J. Ault, J. Beets, C. Bohnsack, C. Caldow, J. Christensen, A. Friedlander, C. Jeffrey, M. Kendall, J. Luo, M.E. Monaco, S. Smith, and K. Woody. 2006. A guide to monitoring reef fish in the National Park Service's South Florida/Caribbean Network. NOAA Technical Memorandum NOS NCCOS 39. Silver Spring, MD. 166 pp. Online: http://www.ccma.nos.noaa.gov/publications/Reefmonitoringguide.pdf (Accessed 14 May 2013)

Monaco, M.E., A.M. Friedlander, C. Caldow, J.D. Christensen, J. Beets, J. Miller, C. Rogers, and R. Boulon. 2007. Characterizing reef fish populations and habitats within and outside the U.S. Virgin Islands Coral Reef National Monument: a lesson in MPA design. Fish. Manage. Ecol. 14:33-40.

Monaco, M.E., A.M. Friedlander, S.D. Hile, S.J. Pittman, and R.H. Boulon. 2009. The coupling of St. John, US Virgin Islands marine protected areas based on reef fish habitat affinities and movements across management boundaries. pp. 1029-1032. In: B.M. Riegl and R.E. Dodge (eds.), Proceedings of the 11th International Coral Reef Symposium, Vol. 2. Ft. Lauderdale, FL. 742 pp.

Mora, C., O. Aburto-Oropeza, A. Ayala Bocos, P.M. Ayotte, S. Banks, A.G. Bauman, M. Beger, S. Bessudo, David J. Booth, E. Brokovich, A. Brooks, P. Chabanet, J.E. Cinner, J. Cortés, J.J. Cruz-Motta, A. Cupul Magaña, E.E. DeMartini, G.J. Edgar, D.A. Feary, S.C.A. Ferse, A.M. Friedlander, K.J. Gaston, C.Gough, N.A.J. Graham, A. Green, H. Guzman, M. Hardt, M. Kulbicki, Y. Letourneur, A. López Pérez, M. Loreau, Y. Loya, C. Martinez, I. Mascareñas-Osorio, T. Morove, M.O. Nadon, Y. Nakamura, G. Paredes, N.V.C. Polunin, Morgan S. Pratchett, H. Reyes Bonilla, F. Rivera, E. Sala, S.A. Sandin, G. Soler, R. Stuart-Smith, E. Tessier, D.P. Tittensor, M. Tupper, P. Usseglio, L. Vigliola, L. Wantiez, I. Williams, S.K. Wilson, and F.A. Zapata. 2011. Global human footprint on the linkage between biodiversity and ecosystem functioning in reef fishes. PLoS Biology 9. e1000606.

Nemeth, R. S. 2005. Population characteristics of a recovering US Virgin Islands red hind spawning aggregation following protection. Mar. Prog. Ser. 286:81-97.

Newman, M.J.H., G.A. Paredes, E. Sala, and J.B.C. Jackson. 2006. Structure of Caribbean coral reef communities across a large gradient of fish biomass. Ecol. Letters 9(11):1216-1227.

Ogden, J.C. and P.R. Ehrlich. 1977. The behavior of heterotypic resting schools of juvenile grunts (Pomadasyidae). Mar. Biol. 42:273-280.

Olsen, D. A., and J. A. LaPlace. 1979. A study of a Virgin Islands grouper fishery based on a breeding aggregation. Ann. Proc. Gulf Caribb. Fish. Inst. 31:130-144.

Pauly, D., V. Christensen, J. Dalsgaaard, R. Froese, F. Torres Jr. 1998. Fishing down marine food webs. Science 279:860-863.

Pittman, S.J. and C.A. McAlpine. 2003. Movements of marine fish and decapod crustaceans: Process, theory and application. Adv. Mar. Biol. 44: 205-294.

Pittman, S.J., C. Caldow, S.D. Hile, and M.E. Monaco. 2007. Using seascape types to explain the spatial patterns of fish in the mangroves of SW Puerto Rico. Mar. Ecol. Prog. Ser. 348: 273-284.

Pittman, S.J., S.D. Hile, C.F.G. Jeffrey, C. Caldow, M.S. Kendall, M.E. Monaco, and Z. Hillis-Starr. 2008. Fish assemblages and benthic habitats of Buck Island Reef National Monument (St. Croix, US Virgin Islands) and the surrounding seascape: A characterization of spatial and temporal patterns. NOAA Technical Memorandum 71. Silver Spring, MD. 96 pp. Online: http://ccma.nos.noaa.gov/publications/biogeography/STC_LTR_Final_0309.pdf (Accessed 14 May 2014)

Pittman, S.J., S.D. Hile, C.F.G. Jeffrey, R. Clark, K. Woody, B.D. Herlach, C. Caldow, M.E. Monaco, R. Appeldoorn. 2010. Coral reef ecosystems of Reserva Natural La Parguera (Puerto Rico): Spatial and temporal patterns in fish and benthic communities (2001-2007). NOAA Technical Memorandum NOS NCCOS 107. Silver Spring, MD. 202 pp.

Pittman, S.J., B. Costa, C.F.G. Jeffrey, and C. Caldow. 2012. Importance of seascape complexity for resilient fish habitat and sustainable fisheries. Ann. Proc. Gulf Caribb. Fish. Inst. 63:420-426.

Precht, W.F., R.B. Aronson, R.M. Moody, and L. Kaufman. 2010. Changing patterns of microhabitat utilization by the Threespot Damselfish, *Stegastes planifrons*, on Caribbean reefs. PLoS ONE 5(5): e10835. Online: http://www.plosone.org/article/info%3Adoi%2F10.1371%2Fjournal.pone.0010835 (Accessed 14 May 2013)

Randall, J.E. 1962. Tagging reef fishes in the Virgin Islands. Ann. Proc. Gulf Caribb. Fish. Inst. 14: 201-241.

Randall, J.E. 1963. An analysis of the fish populations of artificial and natural reefs in the Virgin Islands. Caribb. J. Sci. 3:31-47.

Randall, J.E. 1965. Grazing effect on sea grasses by herbivorous reef fishes in the West Indies. Ecology 46:255-260.

Randall, J.E. 1967. Food habits of reef fishes of the West Indies. Stud. Trap. Oceanogr. 5:665-847.

Randall, J.E. 1968. Caribbean Reef Fishes. T.F.H. Publications, Hong Kong. 319 pp.

Rogers C.S., J. Miller, E.M. Muller, P. Edmunds, R.S. Nemeth, J.P. Beets, A.M. Friedlander, T.B. Smith, R. Boulon, C.F.G. Jeffrey, C. Menza, C. Caldow, N. Idrisi, B. Kojis, M.E. Monaco, A. Spitzack, E.H. Gladfelter, J.C. Ogden, Z. Hillis-Starr, I. Lundgren, W.B. Schill, I.B. Kuffner, L.L. Richardson, B.E. Devine, andJ.D. Voss. 2008. Chapter 8 - Ecology of Coral Reefs in the US Virgin Islands. pp. 303-374. In: B. Riegl and R.E. Dodge (eds.) Coral Reefs of the USA. Springer, Berlin. 806 pp.

SAS Institute. 2006. SIGMAPLOT© Software. SAS Institute Inc., Cary, NC, USA.

Shulman, M.J. and J.C. Ogden. 1987. What controls tropical reef fish populations: recruitment or benthic mortality? An example in the Caribbean reef fish *Haemulon flavolineatum*. Mar. Ecol. Prog. Ser. 39: 233-242.

Stallings, C.D. 2008. Indirect effects of an exploited predator on recruitment of coral-reef fishes. Ecology 89: 2090-2095.

Stallings, C.D. 2009. Fishery-independent data reveal negative effect of human population density on Caribbean predatory fish communities. PLoS One 4(5): e5333. Online: http://www.plosone.org/article/info%3Adoi%2F10.1371%2Fjournal.pone.0005333 (Accessed 11 July 2012)

Thompson, R. and J.L. Munro. 1978. Aspects of the biology and ecology of Caribbean reef fishes: Serranidae (hinds and groupers). J. Fish Biol. 12(2):115-146.

Warwick, R.M. and K.R. Clarke. 1995. New ‘biodiversity’ measures reveal a decrease in taxonomic distinctness with increasing stress. Mar. Ecol. Prog. Ser. 129:301-305.

Wedding, L., A. Friedlander, M. McGranaghan, R. Yost, and M.E. Monaco. 2008. Using bathymetric Lidar to define nearshore benthic habitat complexity: implications for management of reef fish assemblages in Hawaii. Remote Sens. Environ. 112:4159-4165.

Whiteman E.A. 2005. SEAMAP-C USVI, St. Croix & St. Thomas/St. John Fisheries Independent Trap and Line Survey, 1992-2002, SUMMARY REPORT: Data analysis and conclusions. UPR Sea Grant College Program/ NMFS Cooperative Southeast Area Monitoring and Assessment Program-Caribbean (SEAMAP-C) University of Puerto Rico Sea Grant College Program, Mayagüez, Puerto Rico.

Whiteman, E.A., C.A. Jennings, and R.S. Nemeth. 2005. Sex structure and potential female fecundity in a red hind (*Epinephelus guttatus*) spawning aggregation: applying ultrasonic imaging. J. Fish. Biol. 66:983-995.

Wiens, J.A. 2009. Landscape ecology as a foundation for sustainable conservation. Landsc. Ecol. 24:1053-1065.

Appendix A: Sampling Methodology

To assist in monitoring coral reef ecosystem resources and to achieve a better understanding of fish habitat relationships in the U.S. Caribbean, NOAA National Centers for Coastal Science (NCCOS) Center for Coastal Monitoring and Assessment's Biogeography Branch (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 Caribbean Coral Reef Ecosystem Monitoring (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. In St. John, two 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 of bedrock, pavement, rubble and coral reefs and the "soft" stratum comprised of sand, seagrasses and macroalgal beds.

This report uses underwater census data collected from 2001 to 2009 and occurred once a year during the summer season in July. This data set, which comprises 677 surveys, is part of a broader monitoring study that has conducted 1358 surveys throughout the St. John study region between 2001 and 2009. There are two complementary components to the biological field methods: (1) fish surveys and (2) benthic habitat composition surveys.

FISH SURVEYS

Fish were surveyed with consistent visual census protocol for 15 minutes along a 25 m long by 4 m wide belt transect (100 m^2; Figure A.1). 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. To decrease the total time spent writing, four letter codes are used that consist of the first two letters of the genus name followed by the first two letters of the species name. In the rare case that two species have the same four-letter code, the first letter of the species name where a difference occurs is used as the last letter of the code. If the fish can only be identified to the family or genus level then this is all that is recorded. If the fish cannot be identified to the family level then no entry is necessary.

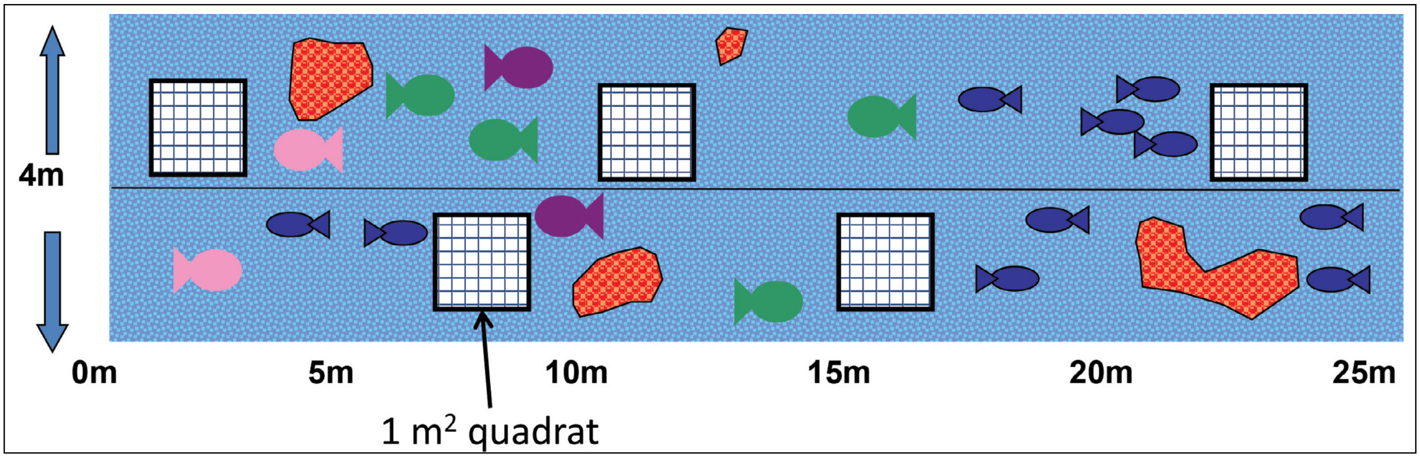


Figure A.1. Schematic of fi sh transect with random habitat quadrat placement.

BENTHIC HABITAT COMPOSITION SURVEYS

The method presented in this report for benthic habitat composition data collection is the full-scale habitat composition census.

Full-scale Habitat Composition Census

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 (Figure A.1). Percent cover is estimated within the quadrat in a two-dimensional plane perpendicular to the observer's line of vision (Figure A.1).

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., 2001), 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 m2 quadrat.

<u>Biotic footprint</u> - 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.

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

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

<u>Hardbottom rugosity</u> - 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.

<u>Proximity of structure</u> - 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.

<u>Marine debris data</u>

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 fl ora or fauna that colonized the debris. Marine debris data collection began in 2007.

<u>Queen conch</u>

The abundance of immature and mature queen conch (*Strombus 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. Conch were included in the survey protocol from August 2004 onward.

<u>Caribbean spiny lobster</u>

Abundance of Caribbean spiny lobsters (*Panulirus argus*) was reported for the period 2005-2007. 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.

Photography

The point count or habitat diver will take at least two photos in different directions at each site to maintain an anecdotal and permanent visual description of the sites that were sampled. Proper care and maintenance is necessary for all camera and camera housings. It is important to maintain the cameras and housings before, after, and in between dives.

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 1-m^2 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.

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.

REFERENCES

Menza, C., J. Ault, J. Beets, C. Bohnsack, C. Caldow, J. Christensen, A. Friedlander, C. Jeffrey, M. Kendall, J. Luo, M.E. Monaco, S. Smith, and K. Woody. 2006. A guide to monitoring reef fish in the National Park Service's South Florida/Caribbean Network. NOAA Technical Memorandum NOS NCCOS 39. Silver Spring, MD. 166 pp. http://ccma.nos.noaa.gov/news/feature/FishMonitoring.html.

Kendall, M.S., C.R. Kruer, K.R. Buja, J.D. Christensen, M. Finkbeiner, R. Warner, and M.E. Monaco. 2002. Methods used to map the benthic habitats of Puerto Rico and the U.S. Virgin Islands. NOAA Technical Memorandum 152. Silver Spring, MD. http://ccma.nos.noaa.gov/products/biogeography/usvi_pr_mapping/manual.pdf

Appendix B: Rapid Habitat Assessment (RHA) Publications

Citations

Boulon, R.H., M.E. Monaco, A.M. Friedlander, C. Caldow, J. Christensen, C. Rogers, J. Beets, J. Miller, and S.D. Hile. 2008. An ecological correction to marine reserve boundaries in the U.S. Virgin Islands. pp. 1082- 1085. In: B.M Riegl and R.E. Dodge (eds.), Proceedings of the 11th International Coral Reef Symposium, Vol. 2. Ft. Lauderdale, FL. 742 pp.

ABSTRACT

Marine protected areas (MPAs) are important tools for management of marine ecosystems. While desired, ecological and biological criteria are not always feasible to consider when establishing protected areas. In 2001, the Virgin Islands Coral Reef National Monument (VICR) in St. John, US Virgin Islands was established by Executive Order. VICR boundaries were based on administrative determination of Territorial Sea boundaries and land ownership at the time of the Territorial Submerged Lands Act of 1974. VICR prohibits almost all fishing and other extractive uses. Surveys of habitat and fishes inside and outside of VICR were conducted in 2002-07. Based on these surveys, areas outside VICR had significantly more hard corals; greater habitat complexity; and greater richness, abundance and biomass of reef fishes than areas within VICR, further supporting results from 2002-2004 (Monaco et al., 2007). The administrative (political) process used to establish VICR did not allow a robust ecological characterization of the area to determine the boundaries of the MPA Efforts are underway to increase amounts of complex reef habitat within VICR by swapping a part of VICR that has little coral reef habitat for a Territorially-owned area within VICR that contains a coral reef with higher coral cover.

Monaco, M.E., A.M. Friedlander, C. Caldow, J.D. Christensen, J. Beets, J. Miller, C. Rogers, and R. Boulon. 2007. Characterizing reef fish populations and habitats within and outside the U.S. Virgin Islands Coral Reef National Monument: a lesson in MPA design. Fish. Manage. Ecol. 14:33-40.

ABSTRACT

Marine protected areas are an important tool for management of marine ecosystems. Despite their utility, ecological design criteria are often not considered or feasible to implement when establishing protected areas. In 2001, the Virgin Islands Coral Reef National Monument (VICRNM) in St John, US Virgin Islands was established by Executive Order. The VICRNM prohibits almost all extractive uses. Surveys of habitat and fishes inside and outside of the VICRNM were conducted in 2002–2004. Areas outside the VICRNM had significantly more hard corals, greater habitat complexity,and greater richness, abundance and biomass of reef fishes than areas within the VICRNM. The administrative process used to delineate the boundaries of the VICRNM did not include a robust ecological characterisation of the area. Because of reduced habitat complexity within the VICRNM, the enhancement of the marine ecosystem may not be fully realised or increases in economically important reef fishes may take longer to detect.

Monaco, M.E., A.M. Friedlander, S.D. Hile, S.J. Pittman, and R.H. Boulon. 2009. The coupling of St. John, US Virgin Islands marine protected areas based on reef fish habitat affinities and movements across management boundaries. pp. 1029-1032. In: B.M. Riegl and R.E. Dodge (eds.), Proceedings of the 11th International Coral Reef Symposium, Vol. 2. Ft. Lauderdale, FL. 742 pp.

ABSTRACT

NOAA's Biogeography Branch, National Park Service (NPS), US Geological Survey, and the University of the Virgin Islands (UVI) are using acoustic telemetry to quantify spatial patterns and habitat affinities of reef fishes. The objective of the study is to define the movements of reef fishes among habitats within and between the Virgin Islands Coral Reef National Monument (VICRNM), the Virgin Islands National Park (VIIS), and Territorial waters. In order to better understand species' habitat utilization patterns among management regimes, we deployed an array of hydroacoustic receivers and acoustically tagged reef fishes. A total of 150 fishes, representing 18 species and 10 families were acoustically tagged along the south shore of St. John. Thirty six receivers were deployed in shallow nearshore bays and across the shelf to depths of approximately 30 m. Example results include the movement of lane snappers and blue striped grunts that demonstrated diel movement from reef habitats during daytime hours to offshore seagrass beds at night. The array comprised of both nearshore and cross shelf location of receivers provides information on fine to broad scale fish movement patterns across habitats and among management units to examine the strength of ecological connectivity between management areas and habitats.

Appendix C: Composition of Macroinvertebrates

BACKGROUND

The stratified random sampling design utilized by this study also provided limited opportunities to conduct surveys and observe the broad-scale spatial distribution of three macroinvertebrates: queen conch (*Strombus gigas*); long-spined sea urchins (*Diadema antillarum*); and the spiny lobster (*Panilurus argus*). Queen conch are ecologically important components of faunal assemblages that occur in Caribbean coastal ecosystems, 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. Several territorial and federal regulations have been implemented to reduce the harvest of conch from U.S. Caribbean territories (CFMC, 1996). Current fishing regulations allow for the harvest of two conch per person per day in the VIIS but no harvest of conch within the VICR (USVI DPNR, 2012).

Like queen conch, spiny lobster is a treasured local delicacy the U.S. Virgin Islands, and local populations are targeted by both commercial and artisanal fishers. A recent analysis of commercial landings data to determine allowable catch limits for lobsters have estimated median landings from St. Thomas and St. John to be 119,902 lb (54,501 kg) per year between 2000 and 2008 (CMFC, 2011). Fishing pressure on spiny lobster is controlled under federal regulations implemented by the CFMC since 1985 and by territorial regulations implemented by Puerto Rico's Department of Natural and Environmental Resources (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). Harvest of up to two lobsters per person per day is allowed in the VIIS but is prohibited in the VICR (USVI DPNR, 2012).

The long-spined sea urchin is 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).

RESULTS

Abundance and distribution of macroinvertebrates

Queen conch (*Strombus gigas*)

A total of 228 queen conch were observed in the study region between 2004 and 2009, of which 56% were juveniles (i.e., no fl ared lip; Figure C.1). Overall conch were most abundant in SAV (X=0.54 ± 0.11), followed by sand (X=0.41 ± 0.13), and hardbottom (X=0.13 ± 0.05). These spatial patterns were similar for both immature as well as mature conch (Figure C.2). Although immature and mature conch densities were higher inside VIIS compared with outside, there were no statistically significant differences between management strata for either age class or overall (all p>0.05).

Higher concentrations of adult conch were observed in seagrass habitats, primarily on the South shore of St. John (Figure C.3). Juvenile conch were scattered around St. John with no apparent spatial pattern (Figure C.3). Total conch abundance in SAV was higher in 2008 and 2009 compared with previous years, particularly inside VIIS (Figure C.4). No clear temporal trends were apparent in hardbottom or sand habitats.

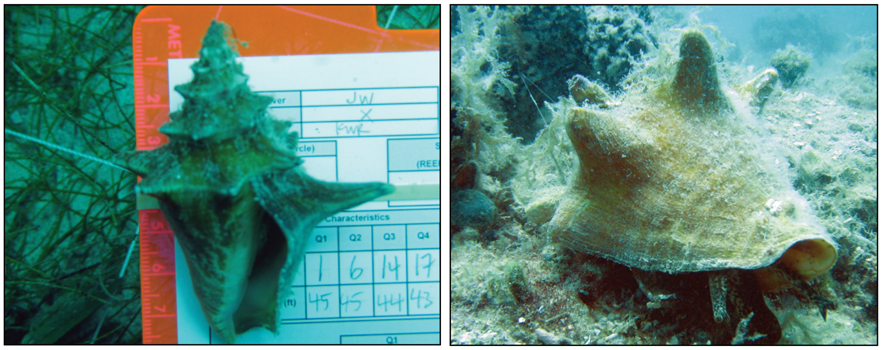


*Figure C.1. Photos of queen conch (*Strombus gigas*): juvenile (left) and mature (right) stages in St. John, USVI. Credit: NOAA/NCCOS/CCMA/Biogeography Branch.*

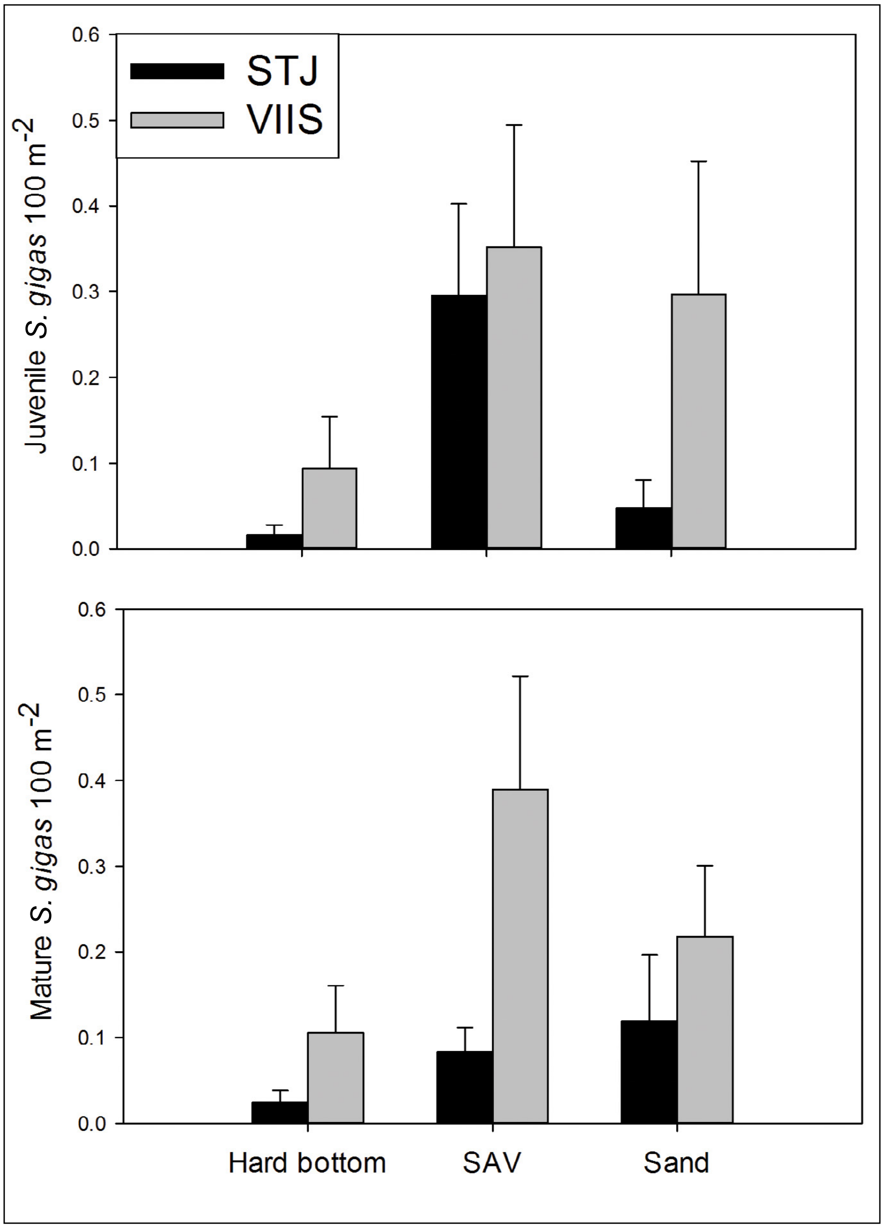


Figure C.2. Numerical density (mean + SE) for all queen conch (S. gigas) observed on transects inside and outside VIIS by dominant habitat types in the study region between 2004 and 2010 for immature (top) and mature (bottom) stages.

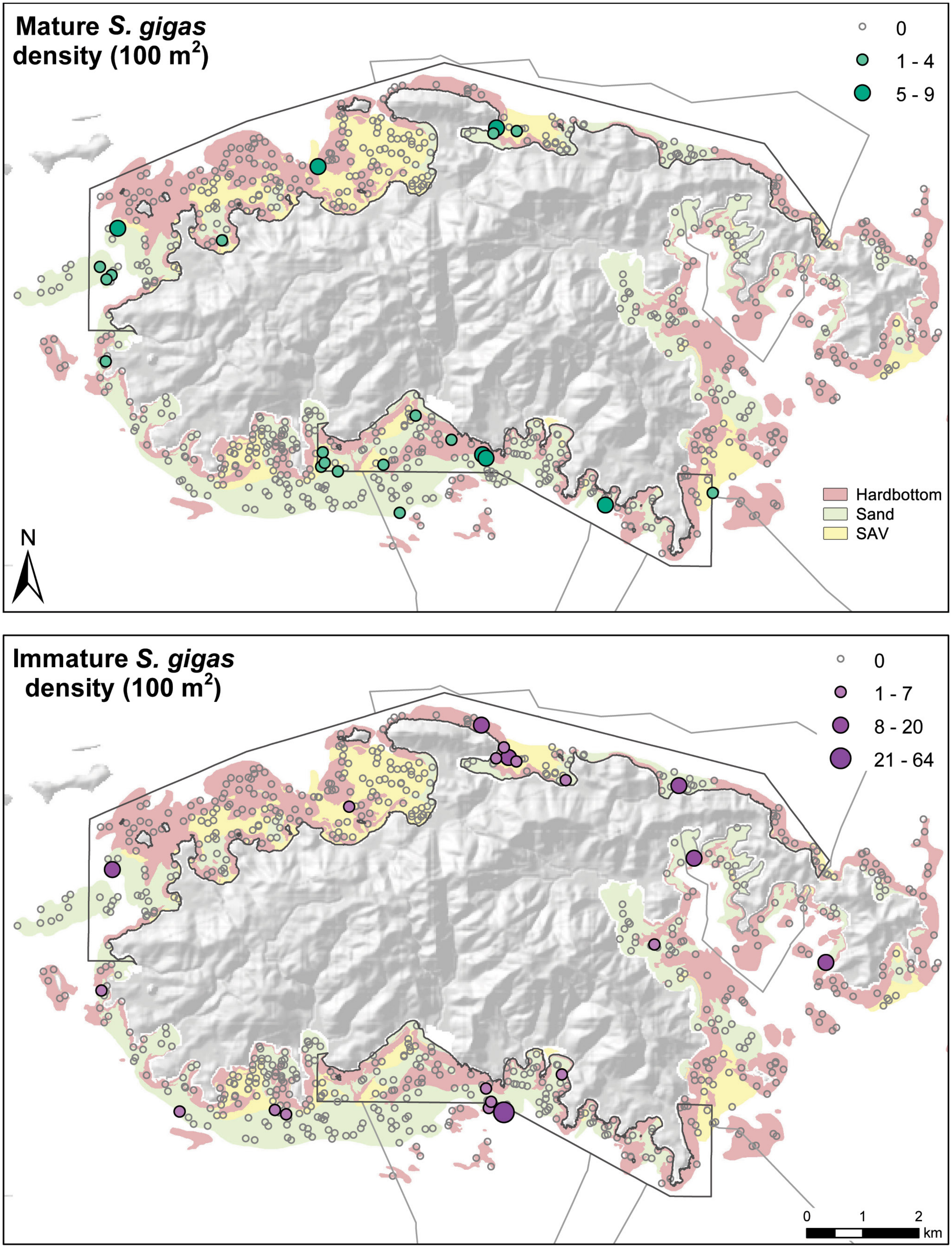


*Figure C.3. Spatial distributions of juvenile (immature; top) and adult (mature; bottom) queen conch (*S. gigas*) density in the study region between 2004 and 2009. Source: K. Stamoulis (University of Hawaii).*

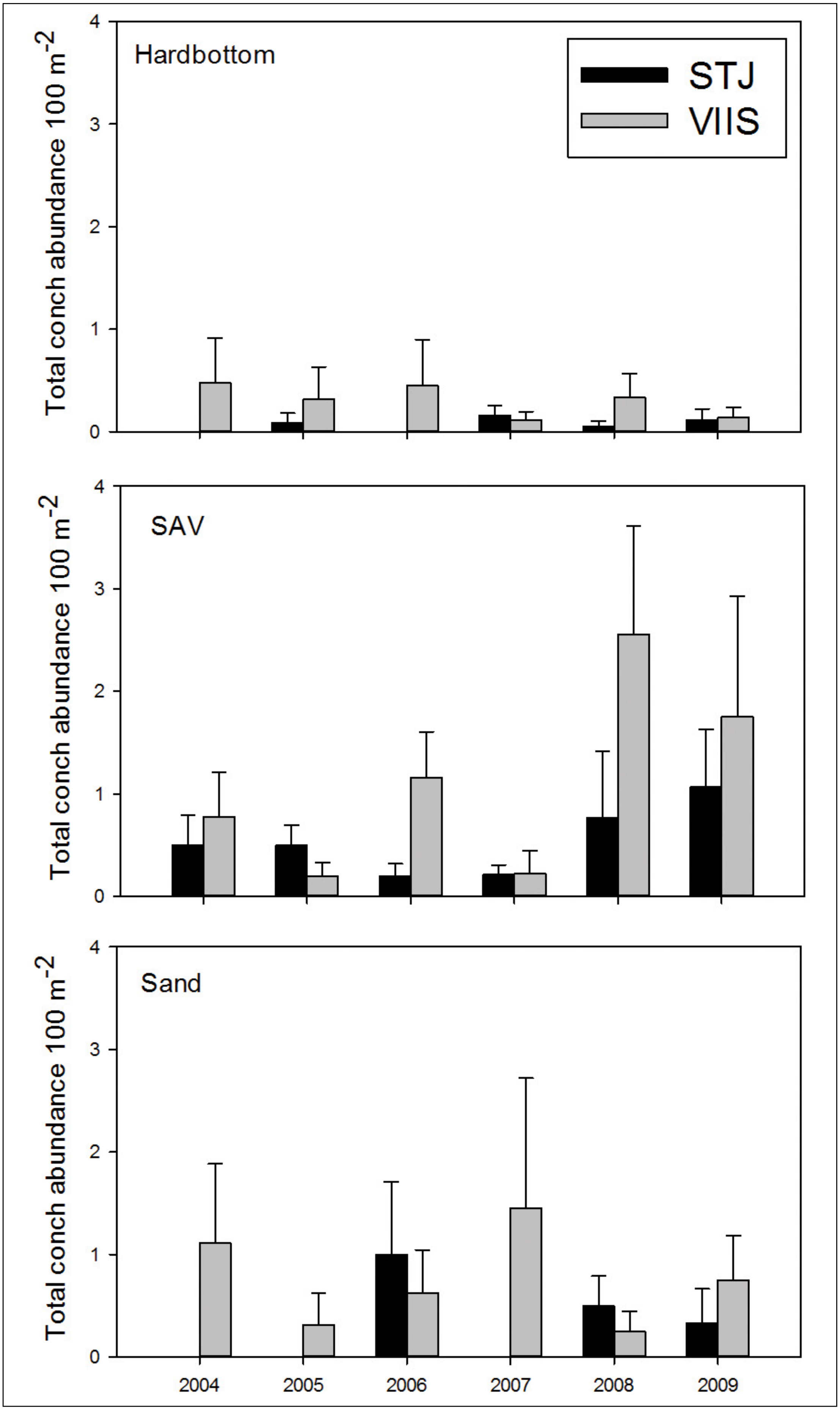


*Figure C.4. Total queen conch (*S. gigas*) density (mean number 100/m² + SE) within major habitat types and between management strata for a) hardbottom, b) SAV, and c) sand.*

Long-spined sea urchin (*Diadema antillarum*)

The long-spined sea urchin was only observed on 3.5% of all surveys (hardbottom = 5.4%, SAV = 2.5%, sand = 1.4%; Figure C.5). The overall average number of individuals observed was 0.9 ± 8.8 but the coefficient of variation was very large (COV=8.9) owing to the large number of surveys without urchins. Numerical density in hardbottom (2.0 ± 0.7), was an order of magnitude higher than density within SAV (0.4 ± 0.3), and two orders of magnitude higher than urchin density within sand habitat (0.02 ± 0.1; Figure C.6). Densities of urchins on hardbottom inside VIIS (X=0.7 ± 4.3) was five times lower than densities observed outside VIIS (X=3.7 ± 19.1). Highest densities of long-spined sea urchins were observed scattered along the south shore of St. John, with lower densities found in the northwest portion of the island (Figure C.7).

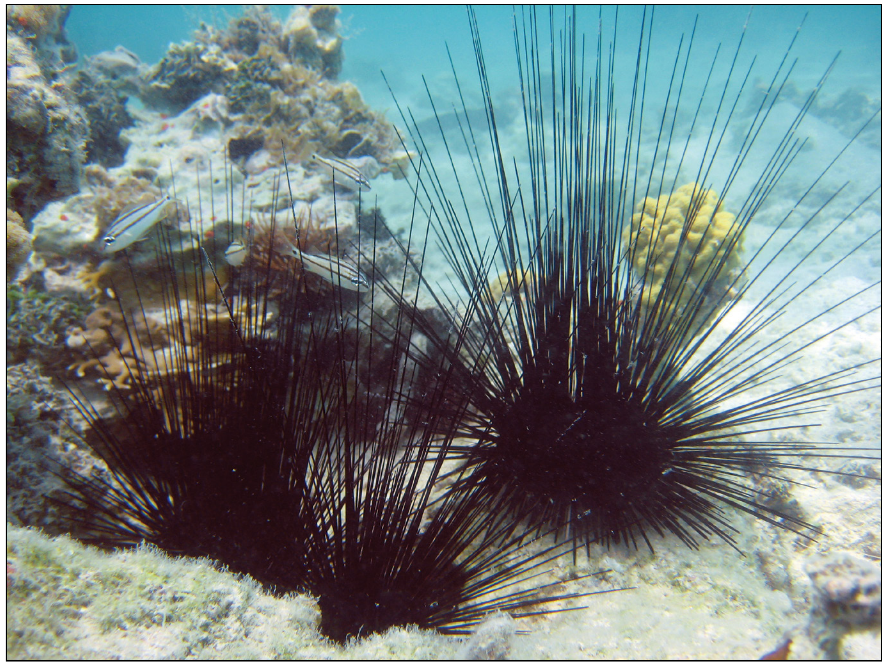


*Figure C.5. Photo of long-spined sea urchin (*Diadema antillarum*) in St. John, USVI. Credit: NOAA/NCCOS/CCMA/Biogeography Branch.*

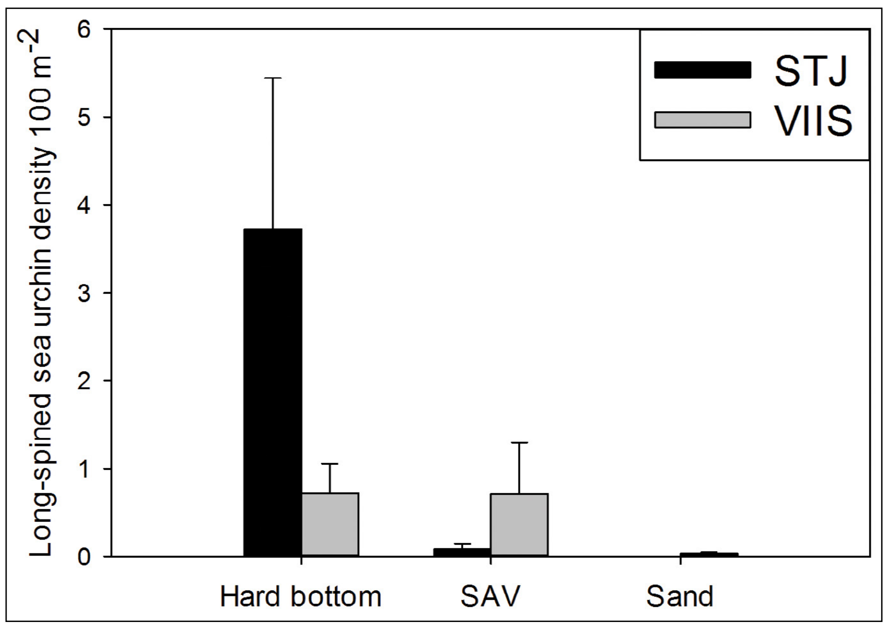


*Figure C.6. Density (mean + SE) for long-spined sea urchin (*D. antillarum*) inside and outside VIIS by dominant habitat types in the study region between 2004 and 2009.*

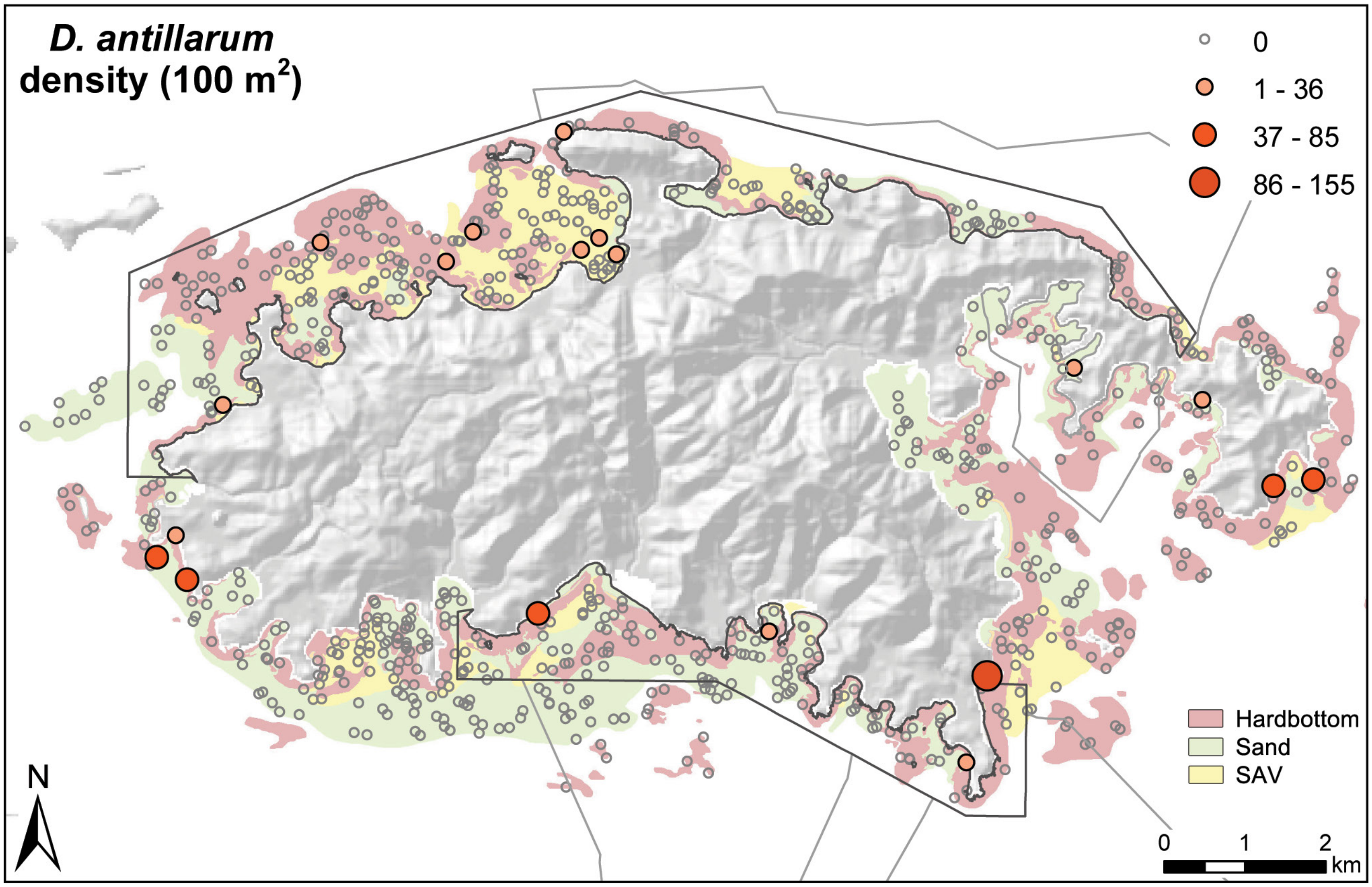


*Figure C.7. Spatial distribution of long-spined sea urchins (*D. antillarum*) in the study region between 2004 and 2009. Source: K. Stamoulis (University of Hawaii).*

<u>Caribbean spiny lobster (*Panulirus argus*)</u>

Only 10 Caribbean spiny lobster were recorded over the study area from 2004 to 2009 (Figure C.8). All individuals were found in hardbottom areas but occurred in <1% of surveys within this habitat (X=0.04 ± 0.03 100/m2). Density outside VIIS was more than seven times higher than density inside VIIS but the small number of total observations and high variance makes comparisons difficult (Figure C.9). Overall lobsters were only observed on the south shore of St. John (Figure C.10).

The techniques to survey lobsters in this study likely underestimate of abundance. Lobsters are cryptic and crevice dwelling animals that are best surveyed using dedicated lobster census techniques and supplemented with night time surveys when some lobsters are more active and therefore more visible. Therefore, these data should be used with caution when making inferences about lobster populations around St. John.

*Figure C.8. Photo of Caribbean spiny lobster (*Panulirus argus*) in St. John, USVI. Credit: NOAA/NOS/NCCOS/CCMA/Biogeography Branch.*

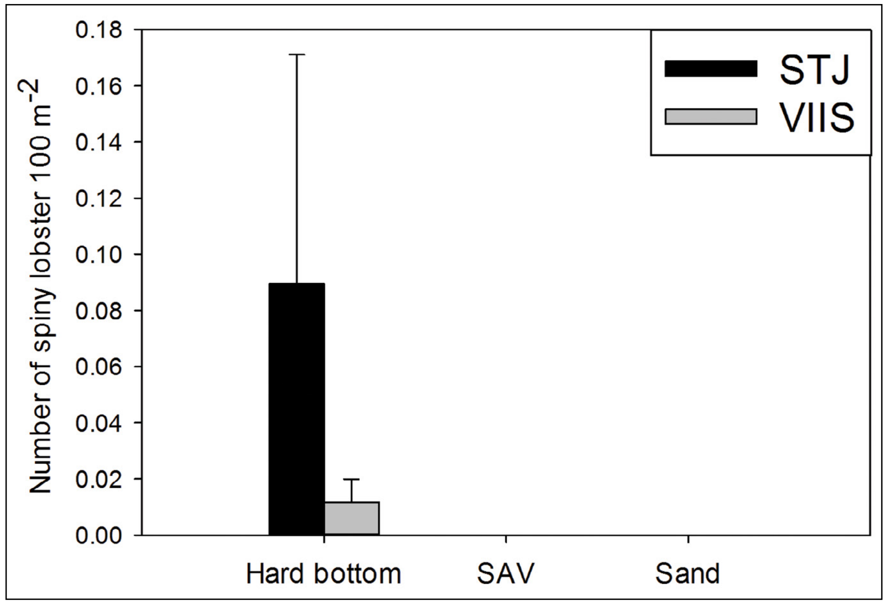


*Figure C.9. Density (X + SE) for Caribbean spiny lobster (*P. argus*) inside and outside VIIS by dominant habitat types in the study region between 2004 and 2009.*

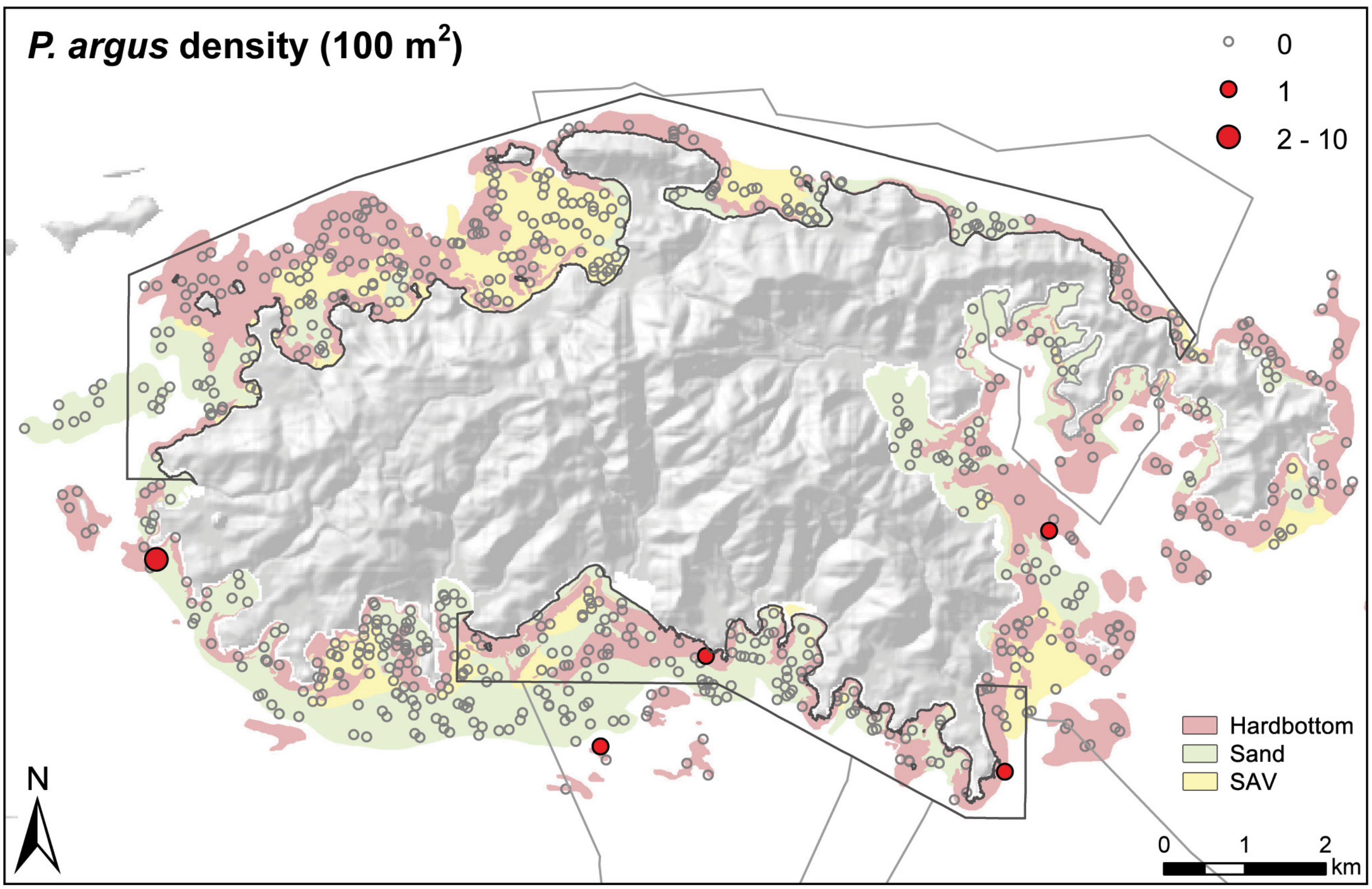


*Figure C.10. Spatial distribution of Caribbean spiny lobster (*P. argus*) in the study region between 2004 and 2009. Source: K. Stamoulis (University of Hawaii).*

REFERENCES

Appeldoorn, R.S. 1994. Queen conch management and research: status, needs, and priorities. pp. 301-320. In: R.S. Appeldoorn and B. Rodriguez (eds.). Queen Conch biology, fisheries, and mariculture. Fundación, Científíca. Los Roques, Caracas Venezuela. 356 pp.

Appeldoorn, R.S. and B. Rodriguez, (eds.). 1994. Queen Conch biology, fisheries, and mariculture. Fundación, Científíca. Los Roques, Caracas Venezuela. 356 pp.

Caribbean Fisheries Management Council (CFMC). 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. Caribbean Fishery Management Council, San Juan, Puerto Rico. 178 pp. Online: http://www.caribbeanfmc.com/fmp%20REEF%20FISH/RF%20FMP.pdf (Accessed 14 May 2013)

Caribbean Fishery Management Council (CFMC). 1996. Fishery Management Plan, Regulatory Impact Review, and Final Environmental Impact Statement for the Queen Conch Resources of Puerto Rico and the U.S. Virgin Islands. Caribbean Fishery Management Council and the National Marine Fisheries Service. 56 pp. Online: http://www.strombusgigas.com/fishery_management_plans.htm (Accessed 14 May 2013)

Caribbean Fisheries Management Council (CFMC). 2011. Comprehensive Annual Catch Limit (ACL) Amendment For The U.S. Caribbean. http://www.caribbeanfmc.com/fmp%20ACLS/final%202011_Caribbean_ ACL_Amendment_FEIS_102511.pdf (Accessed 14 May 2013)

Knowlton, N. 2001. The future of coral reefs. Proceedings of the National Academy of Sciences of the U.S.A. 98(10): 5419-5425.

Lessios, H.A. 1998. Mass mortality of *Diadema antillarum* in the Caribbean: what have we learned? Ann. Rev. Ecol. Syst. 19: 371-393.

Lessios, H., D.R. Robertson, and J.D. Cubit. 1984. Spread of *Diadema* mass mortality through the Caribbean. Science 226(4672): 335-337.

Randall, J.E., R.E. Schroeder, and W.A. Starck II. 1964. Notes on the biology of the echinoid *Diadema antillarum*. Carib. J. Sci. 4: 421-43.

Sammarco, P.W. 1972. Some aspects of the ecology of *Diadema antillarum* (Philippi): food preferences and effect on grazing. Special problems report to the West Indies Lab. of Fairlaigh Dickinson University, St. Croix. 47 pp.

U.S. Virgin Islands Department of Planning and Natural Resources (USVI DPNR). 2012. United States Virgin Islands Commercial & Recreational Fisher's Information Handbook. DPNR, VI Fish and Wildlife and Division of Environmental Enforcement.

Weil, E., F. Losada, and D. Bone. 1984. Spatial variations in density and size of the echinoid *Diadema antillarum* Philippi on some Venezuelan coral reefs. Bijdragen tot de Dierkunde. 54: 73-82.

Wells, S.M., R.M. Pile, and N.M. Collins. 1985. Queen or pink conch. In: The IUCN invertebrate red data book. pp 79-90. IUCN. Gland, Switzerland. 632 pp.

Appendix D: Fish Landings Data Table

Table D.1. 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 (*Lutjanus synagris*)	Lutjanidae	0.03
Yellowtail snapper (*Ocyurus chrysurus*)	Lutjanidae	2.89
Triggerfishes	Balistidae	29.68
Squirrelfishes	Holocentridae	4.84
Mutton snapper (*Lutjanus analis*)	Lutjanidae	0.13
Other snappers	Lutjanidae	1.04
Hogfish	Labridae	1.06
Trunkfish	Ostraciidae	0.08

Table D.2. Total captured biomass (kg) of each species from all gear types south of St. John in 1992-93 and 1999-2000. Species are ordered according to rank position in 1992-93. Data from SEAMAP-C 2005.

		1992-93			1999-2000		
Species	**Common name**	**Biomass**	**%**	**Rank**	**Biomass**	**%**	**Rank**
Epinephelus guttatus	Red Hind	63.20	26.13	1	68.48	20.82	3
Canthidermis sufflamen	Ocean Triggerfish	59.60	24.64	2			
Balistes vetula	Queen Triggerfish	39.33	16.26	3	75.86	23.06	1
Cephalopholis fulva	Coney	23.87	9.87	4	75.23	22.87	2
Ocyurus chrysurus	Yellowtail Snapper	22.89	9.46	5	16.38	4.98	4
Mycteroperca venenosa	Yellowfin Grouper	5.00	2.07	6	1.56	0.48	20
Haemulon plumierii	White Grunt	3.74	1.55	7	0.89	0.27	30
Cephalopholis cruentata	Graysby	2.57	1.06	8	6.55	3.85	9
Holocentrus rufus	Longspine Squirrelfish	2.19	0.91	9	2.89	0.88	12
Calamus species	Porgy species	2.18	0.9	10			
Lutjanus apodus	Schoolmaster Snapper	2.06	0.85	11	0.78	0.24	32
Epinephelus striatus	Nassau Grouper	1.80	0.74	12			
Aluterus monocerus	Unicorn Filefish	1.80	0.74	13			
Xanthichthys ringens	Sargassum Triggerfish	1.23	0.51	14			
Halichoeres radiatus	Puddingwife	1.23	0.51	15	0.17	0.05	42
Holocentrus adscensionis	Squirrelfish	1.17	0.48	16	2.92	0.89	11
Acanthurus chirurgus	Doctorfish	0.99	0.41	17	2.11	0.64	15
Lutjanus jocu	Dog Snapper	0.76	0.31	18			
Malacanthus plumieri	Sand Tilefish	0.71	0.29	19			
Melichthys niger	Black Durgon	0.68	0.28	20	1.64	0.50	19

Caribbean Fisheries Management Council (CFMC). 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. Caribbean Fishery Management Council, San Juan, Puerto Rico. 178 pp. Online: http://www.caribbeanfmc.com/fmp%20REEF%20FISH/RF%20FMP.pdf (Accessed 14 May 2013)

Appendix E: Fish Species Summary Tables

Table E.1. Fish species list and summary data on frequency, abundance and biomass (2001-2009) for the St. John, U.S. Virgin islands study region.

Family		Frequency		Abundance			Biomass (g)		
Species Name	**Common Name**	**Total**	**%**	**Total**	**Mean**	**± SE**	**Total**	**Mean**	**± SE**
Acanthuridae									
Acanthurus bahianus	Ocean Surgeonfish	320	47.27	2907	4.294	0.357	94235.074	139.195	14.380
Acanthurus chirurgus	Doctorfish	92	13.59	405	0.598	0.111	28924.571	42.725	7.160
Acanthurus coeruleus	Blue Tang	252	37.22	2726	4.027	0.555	139008.546	205.330	36.083
Albulidae									
Albula vulpes	Bonefish	1	0.15	2	0.003	0.003	5031.180	7.432	7.432
Apogonidae									
Apogon aurolineatus	Bridle Cardinalfish	1	0.15	1	0.001	0.001	0.459	0.001	0.001
Apogon binotatus	Barred Cardinalfish	5	0.74	7	0.010	0.005	10.429	0.015	0.012
Apogon lachneri	Whitestar Cardinalfish	1	0.15	3	0.004	0.004	1.378	0.002	0.002
Apogon maculatus	Flamefish	3	0.44	5	0.007	0.004	2.296	0.003	0.002
Apogon UNK	CARDINALFISH species	4	0.59	12	0.018	0.011	5.512	0.008	0.005
Apogon pseudomaculatus	Twospot Cardinalfish	2	0.30	3	0.004	0.003	1.522	0.002	0.002
Apogon quadrisquamatus	Sawcheek Cardinalfish	3	0.44	4	0.006	0.004	1.837	0.003	0.002
Apogon townsendi	Belted Cardinalfish	4	0.59	13	0.019	0.011	20.398	0.030	0.023
Astrapogon stellatus	Conchfish	3	0.44	4	0.006	0.004	1.998	0.003	0.002
Atherinidae									
Atherinomorus UNK	SILVERSIDE species	1	0.15	750	1.108	1.108	197.933	0.292	0.292
Aulostomidae									
Aulostomus maculatus	Trumpetfish	47	6.94	57	0.084	0.013	4308.716	6.364	1.455
Balistidae									
Balistes vetula	Queen Triggerfish	45	6.65	56	0.083	0.014	42873.912	63.329	13.013
Melichthys niger	Black Durgon	1	0.15	1	0.001	0.001	151.770	0.224	0.224
Blenniidae									
Ophioblennius macclurei	Redlip Blenny	36	5.32	104	0.154	0.034	355.937	0.526	0.123
Parablennius marmoreus	Seaweed Blenny	5	0.74	6	0.009	0.004	6.347	0.009	0.008
Scartella cristata	Molly Miller	1	0.15	1	0.001	0.001	0.363	0.001	0.001
Bothidae									
Bothus lunatus	Peacock Flounder	10	1.48	13	0.019	0.007	342.561	0.506	0.329
Bothus ocellatus	Eyed Flounder	1	0.15	1	0.001	0.001	6.051	0.009	0.009
Bothus UNK	LEFTEYE FLOUNDER sp	13	1.92	16	0.024	0.007	58.636	0.087	0.049
Callionymidae									
Paradiplogrammus bairdi	Lancer Dragonet	63	9.31	159	0.235	0.041	171.141	0.253	0.071
Carangidae									
Carangoides bartholomaei	Yellow Jack	2	0.30	6	0.009	0.007	2086.203	3.082	2.188
Caranx crysos	Blue Runner	9	1.33	112	0.165	0.121	36195.189	53.464	47.783
Caranx lugubris	Black Jack	1	0.15	1	0.001	0.001	890.275	1.315	1.315
Carangoides ruber	Bar Jack	89	13.15	808	1.194	0.347	18322.409	27.064	6.285
Decapterus UNK	SCAD species	1	0.15	10	0.015	0.015	216.966	0.320	0.320
Decapterus macarellus	Mackerel Scad	7	1.03	861	1.272	0.743	41866.086	61.841	42.679
Trachinotus goodei	Palometa	1	0.15	6	0.009	0.009	7095.375	10.481	10.481
Chaenopsidae									
Acanthemblemaria maria	Secretary Blenny	5	0.74	9	0.013	0.007	1.795	0.003	0.001
Acanthemblemaria spinosa	Spinyhead Blenny	7	1.03	10	0.015	0.006	1.994	0.003	0.001
Chaenopsis UNK	PIKEBLENNY species	10	1.48	10	0.015	0.005	13.232	0.020	0.009
Chaenopsis limbaughi	Yellowface Pikeblenny	14	2.07	18	0.027	0.008	66.456	0.098	0.062
Chaenopsis ocellata	Bluethroat Pikeblenny	8	1.18	13	0.019	0.008	46.377	0.069	0.035
Emblemaria pandionis	Sailfin Blenny	2	0.30	2	0.003	0.002	0.399	0.001	0.000
Chaetodontidae									
Chaetodon capistratus	Foureye Butterflyfish	187	27.62	552	0.815	0.070	13123.307	19.385	2.476
Chaetodon ocellatus	Spotfin Butterflyfish	2	0.30	4	0.006	0.004	351.579	0.519	0.482
Chaetodon sedentarius	Reef Butterflyfish	15	2.22	24	0.035	0.010	605.108	0.894	0.392
Chaetodon striatus	Banded Butterflyfish	36	5.32	67	0.099	0.021	1932.662	2.855	0.716
Cirrhitidae									
Amblycirrhitus pinos	Redspotted Hawkfish	19	2.81	27	0.040	0.010	22.876	0.034	0.012

Family *Species Name*	Common Name	Frequency Total	Frequency %	Abundance Total	Abundance Mean	Abundance ± SE	Biomass (g) Total	Biomass (g) Mean	Biomass (g) ± SE
Clupeidae									
Clupeidae UNK	Clupeidae Family species	0	0.00	0	0.000	0.000	0.000	0.000	0.000
Jenkinsia UNK	HERRING species	0	0.00	0	0.000	0.000	0.000	0.000	0.000
Congridae									
Conger triporiceps	Manytooth Conger	1	0.15	1	0.001	0.001	3.466	0.005	0.005
Heteroconger longissimus	Brown Garden Eel	19	2.81	634	0.936	0.319	29585.679	43.701	17.836
Dactylopteridae									
Dactylopterus volitans	Flying Gurnard	5	0.74	5	0.007	0.003	13.648	0.020	0.013
Dasyatidae									
Dasyatis americana	Southern Stingray	13	1.92	15	0.022	0.006	333537.912	492.670	189.261
Diodontidae									
Chilomycterus antennatus	Bridled Burrfish	1	0.15	1	0.001	0.001	12.782	0.019	0.019
Diodon holocanthus	Balloonfish	9	1.33	15	0.022	0.009	280.252	0.414	0.205
Diodon hystrix	Porcupinefish	2	0.30	2	0.003	0.002	104.567	0.154	0.109
Echendeidae									
Echeneis naucrates	Sharksucker	4	0.59	11	0.016	0.011	4422.977	6.533	3.796
Ephippidae									
Chaetodipterus faber	Atlantic Spadefish	2	0.30	7	0.010	0.008	876.970	1.295	0.944
Gerridae									
Eucinostomus gula	Silver Jenny	2	0.30	15	0.022	0.019	107.564	0.159	0.124
Eucinostomus jonesii	Slender Mojarra	1	0.15	2	0.003	0.003	38.472	0.057	0.057
Eucinostomus melanopterus	Flagfin Mojarra	2	0.30	94	0.139	0.137	438.827	0.648	0.619
Gerres cinereus	Yellowfin Mojarra	13	1.92	30	0.044	0.016	1181.647	1.745	0.657
Ginglymostomatidae									
Ginglymostoma cirratum	Nurse Shark	6	0.89	6	0.009	0.004	132456.575	195.652	125.974
Gobiidae									
Bollmannia boqueronensis	White-eye Goby	5	0.74	42	0.062	0.045	123.226	0.182	0.104
Coryphopterus dicrus	Colon Goby	98	14.48	262	0.387	0.057	171.714	0.254	0.037
Coryphopterus eidolon	Pallid Goby	12	1.77	35	0.052	0.018	22.939	0.034	0.012
Coryphopterus glaucofraenum	Bridled Goby	266	39.29	2101	3.103	0.395	2061.289	3.045	0.441
Coryphopterus lipernes	Peppermint Goby	1	0.15	1	0.001	0.001	0.655	0.001	0.001
Coryphopterus personatus/hyalinus	Masked/glass Goby	0	0.00	0	0.000	0.000	0.000	0.000	0.000
Ctenogobius saepepallens	Dash Goby	61	9.01	934	1.380	0.363	647.054	0.956	0.243
Elacatinus UNK	GOBY species	1	0.15	2	0.003	0.003	0.502	0.001	0.001
Elacatinus chancei	Shortstripe Goby	2	0.30	3	0.004	0.003	0.753	0.001	0.001
Elacatinus dilepis	Orangesided Goby	1	0.15	1	0.001	0.001	0.251	0.000	0.000
Elacatinus evelynae	Sharknose Goby	102	15.07	222	0.328	0.039	55.741	0.082	0.010
Elacatinus louisae	Spotlight Goby	1	0.15	2	0.003	0.003	0.502	0.001	0.001
Gnatholepis thompsoni	Goldspot Goby	98	14.48	368	0.544	0.092	221.208	0.327	0.068
Gobiidae UNK	GOBY species	17	2.51	49	0.072	0.026	32.114	0.047	0.017
Gobiosoma grosvenori	Rockcut Goby	1	0.15	3	0.004	0.004	1.966	0.003	0.003
Microgobius carri	Seminole Goby	12	1.77	31	0.046	0.016	22.210	0.033	0.021
Nes longus	Orangespotted Goby	83	12.26	394	0.582	0.102	1165.053	1.721	0.411
Oxyurichthys stigmalophius	Spotfin Goby	1	0.15	2	0.003	0.003	4.936	0.007	0.007
Priolepis hipoliti	Rusty Goby	1	0.15	1	0.001	0.001	0.376	0.001	0.001
Grammatidae									
Gramma loreto	Fairy Basslet	29	4.28	81	0.120	0.028	34.570	0.051	0.015
Haemulidae									
Anisotremus surinamensis	Black Margate	1	0.15	6	0.009	0.009	1951.675	2.883	2.883
Haemulon album	Margate (White)	2	0.30	15	0.022	0.021	6737.207	9.952	7.268
Haemulon aurolineatum	Tomtate	50	7.39	1055	1.558	0.474	8825.460	13.036	3.889
Haemulon chrysargyreum	Smallmouth Grunt	3	0.44	3	0.004	0.003	162.423	0.240	0.157
Haemulon carbonarium	Caesar Grunt	13	1.92	18	0.027	0.009	2188.969	3.233	1.129
Haemulon UNK	GRUNT species	34	5.02	3302	4.877	2.202	1654.436	2.444	0.899
Haemulon flavolineatum	French Grunt	119	17.58	609	0.900	0.188	29737.007	43.925	7.311
Haemulon macrostomum	Spanish Grunt	6	0.89	13	0.019	0.011	2231.928	3.297	1.905
Haemulon parra	Sailors choice	1	0.15	1	0.001	0.001	226.446	0.334	0.334
Haemulon plumierii	White Grunt	27	3.99	101	0.149	0.044	10410.149	15.377	6.255
Haemulon sciurus	Bluestriped Grunt	26	3.84	60	0.089	0.034	17140.817	25.319	10.211
Haemulon striatum	Striped Grunt	3	0.44	3	0.004	0.003	292.915	0.433	0.268

Family *Species Name*	Common Name	Frequency Total	 %	Abundance Total	 Mean	 ± SE	Biomass (g) Total	 Mean	 ± SE
Holocentridae									
Holocentrus adscensionis	Squirrelfish	38	5.61	65	0.096	0.018	6170.957	9.115	2.443
Holocentrus rufus	Longspine Squirrelfish	138	20.38	334	0.493	0.071	28147.961	41.577	5.480
Myripristis jacobus	Blackbar Soldierfish	25	3.69	101	0.149	0.060	7554.261	11.158	3.427
Neoniphon marianus	Longjaw Squirrelfish	4	0.59	4	0.006	0.003	134.154	0.198	0.099
Sargocentron coruscum	Reef Squirrelfish	1	0.15	1	0.001	0.001	37.760	0.056	0.056
Sargocentron vexillarium	Dusky Squirrelfish	5	0.74	8	0.012	0.006	250.840	0.371	0.203
Inermiidae									
Inermia vittata	Boga	7	1.03	348	0.514	0.377	149.827	0.221	0.133
Kyphosidae									
Kyphosus sectator	Chub (Bermuda/Yellow)	6	0.89	86	0.127	0.075	33254.237	49.120	31.840
Labridae									
Bodianus rufus	Spanish Hogfish	30	4.43	45	0.066	0.016	6502.018	9.604	3.027
Clepticus parrae	Creole Wrasse	14	2.07	406	0.600	0.294	11558.734	17.073	7.908
Halichoeres bivittatus	Slippery dick	375	55.39	5110	7.548	0.615	21967.950	32.449	2.843
Halichoeres garnoti	Yellowhead Wrasse	289	42.69	2861	4.226	0.289	15974.677	23.596	2.148
Halichoeres maculipinna	Clown Wrasse	121	17.87	550	0.812	0.129	2331.524	3.444	0.748
Halichoeres pictus	Rainbow Wrasse	42	6.20	343	0.507	0.121	954.169	1.409	0.419
Halichoeres poeyi	Blackear Wrasse	98	14.48	204	0.301	0.037	886.822	1.310	0.254
Halichoeres radiatus	Puddingwife	70	10.34	136	0.201	0.029	852.963	1.260	0.311
Lachnolaimus maximus	Hogfish	4	0.59	4	0.006	0.003	498.583	0.736	0.506
Thalassoma bifasciatum	Bluehead	268	39.59	7560	11.167	1.000	17933.584	26.490	2.574
Xyrichtys martinicensis	Rosy Razorfish	93	13.74	1312	1.938	0.351	11122.209	16.429	4.236
Xyrichtys novacula	Pearly RazorFish	1	0.15	3	0.004	0.004	40.631	0.060	0.060
Xyrichtys splendens	Green Razorfish	57	8.42	179	0.264	0.053	815.048	1.204	0.279
Labrisomidae									
Labrisomus nuchipinnis	Hairy Blenny	1	0.15	1	0.001	0.001	8.183	0.012	0.012
Malacoctenus aurolineatus	Goldline Blenny	5	0.74	12	0.018	0.010	8.123	0.012	0.007
Malacoctenus boehlkei	Diamond Blenny	26	3.84	37	0.055	0.013	8.891	0.013	0.003
Malacoctenus gilli	Dusky Blenny	1	0.15	2	0.003	0.003	0.481	0.001	0.001
Malacoctenus UNK	SCALY BLENNY species	7	1.03	19	0.028	0.017	6.793	0.010	0.006
Malacoctenus macropus	Rosy Blenny	20	2.95	34	0.050	0.012	23.015	0.034	0.008
Malacoctenus triangulatus	Saddled Blenny	69	10.19	159	0.235	0.041	94.008	0.139	0.036
Malacoctenus versicolor	Barfin Blenny	8	1.18	15	0.022	0.009	7.119	0.011	0.006
Lutjanidae									
Lutjanus analis	Mutton Snapper	14	2.07	21	0.031	0.010	43107.014	63.674	20.492
Lutjanus apodus	Schoolmaster	32	4.73	127	0.188	0.053	39561.747	58.437	19.037
Lutjanus cyanopterus	Cubera Snapper	1	0.15	8	0.012	0.012	24.646	0.036	0.036
Lutjanus griseus	Gray Snapper	8	1.18	33	0.049	0.021	5787.401	8.549	4.512
Lutjanus jocu	Dog Snapper	2	0.30	3	0.004	0.003	1168.465	1.726	1.226
Lutjanus mahogoni	Mahogany Snapper	6	0.89	16	0.024	0.013	1983.009	2.929	1.517
Lutjanus synagris	Lane Snapper	60	8.86	181	0.267	0.050	7334.913	10.834	5.134
Lutjanus UNK	SNAPPER species	19	2.81	57	0.084	0.031	44.357	0.066	0.022
Ocyurus chrysurus	Yellowtail Snapper	221	32.64	1028	1.518	0.219	88318.252	130.455	35.702
Malacanthidae									
Malacanthus plumieri	Sand Tilefish	19	2.81	32	0.047	0.015	2284.442	3.374	1.301
Microdesmidae									
Ptereleotris helenae	Hovering Goby	39	5.76	115	0.170	0.036	171.990	0.254	0.088
Monacanthidae									
Aluterus scriptus	Scrawled Filefish	1	0.15	1	0.001	0.001	45.753	0.068	0.068
Cantherhines pullus	Orangespotted Filefish	15	2.22	23	0.034	0.011	949.753	1.403	0.518
Monacanthus ciliatus	Fringed Filefish	41	6.06	87	0.129	0.030	54.057	0.080	0.019
Monacanthus UNK	FILEFISH species	16	2.36	25	0.037	0.010	40.771	0.060	0.036
Monacanthus tuckeri	Slender Filefish	46	6.79	97	0.143	0.029	158.944	0.235	0.094
Mullidae									
Mulloidichthys martinicus	Yellow Goatfish	25	3.69	158	0.233	0.150	11815.118	17.452	5.652
Pseudupeneus maculatus	Spotted Goatfish	153	22.60	364	0.538	0.064	16990.940	25.097	4.395
Muraenidae									
Echidna catenata	Chain Moray	1	0.15	1	0.001	0.001	411.600	0.608	0.608
Gymnothorax funebris	Green Moray	1	0.15	1	0.001	0.001	0.000	0.000	0.000

Family / Species Name	Common Name	Frequency Total	Frequency %	Abundance Total	Abundance Mean	Abundance ± SE	Biomass (g) Total	Biomass (g) Mean	Biomass (g) ± SE
Gymnothorax miliaris	Goldentail Moray	2	0.30	2	0.003	0.002	23.890	0.035	0.026
Gymnothorax moringa	Spotted Moray	7	1.03	7	0.010	0.004	739.836	1.093	0.680
Gymnothorax vicinus	Purplemouth Moray	1	0.15	1	0.001	0.001	244.404	0.361	0.361
Myliobatidae									
Aetobatus narinari	Spotted Eagle Ray	1	0.15	1	0.001	0.001	18997.188	28.061	28.061
Opistognathidae									
Lonchopisthus micrognathus	Swordtail Jawfish	3	0.44	9	0.013	0.008	16.827	0.025	0.017
Opistognathus aurifrons	Yellowhead Jawfish	77	11.37	311	0.459	0.088	1043.634	1.542	0.428
Opistognathus UNK	JAWFISH species	5	0.74	22	0.032	0.015	5.464	0.008	0.004
Opistognathus macrognathus	Banded Jawfish	3	0.44	3	0.004	0.003	52.529	0.078	0.072
Ostraciidae									
Lactophrys bicaudalis	Spotted Trunkfish	4	0.59	4	0.006	0.003	139.650	0.206	0.123
Lactophrys trigonus	Trunkfish	3	0.44	3	0.004	0.003	1046.379	1.546	1.302
Lactophrys triqueter	Smooth Trunkfish	18	2.66	20	0.030	0.007	1431.497	2.114	0.835
Paralichthyidae									
Syacium UNK	SAND FLOUNDER sp	3	0.44	3	0.004	0.003	2.981	0.004	0.004
Pempheridae									
Pempheris schomburgkii	Glassy Sweeper	3	0.44	147	0.217	0.159	1362.925	2.013	1.389
Pomacanthidae									
Centropyge argi	Cherubfish	5	0.74	18	0.027	0.014	12.243	0.018	0.010
Holacanthus ciliaris	Queen Angelfish	12	1.77	12	0.018	0.005	1869.260	2.761	1.049
Holacanthus tricolor	Rock Beauty	31	4.58	42	0.062	0.012	3064.110	4.526	1.778
Pomacanthus arcuatus	Gray Angelfish	40	5.91	73	0.108	0.022	25129.495	37.119	12.976
Pomacanthus paru	French Angelfish	17	2.51	17	0.025	0.006	4950.369	7.312	2.921
Pomacentridae									
Abudefduf saxatilis	Sergeant Major	48	7.09	522	0.771	0.338	18417.693	27.205	14.553
Abudefduf taurus	Night Sergeant	1	0.15	4	0.006	0.006	179.832	0.266	0.266
Chromis cyanea	Blue Chromis	93	13.74	1865	2.755	0.600	5112.623	7.552	1.403
Chromis multilineata	Brown Chromis	37	5.47	947	1.399	0.786	2675.216	3.952	1.155
Microspathodon chrysurus	Yellowtail Damselfish	65	9.60	199	0.294	0.062	9077.893	13.409	2.429
Stegastes adustus	Dusky Damselfish	59	8.71	326	0.482	0.083	2559.593	3.781	0.730
Stegastes diencaeus	Longfin Damselfish	69	10.19	413	0.610	0.158	3626.223	5.356	1.196
Stegastes leucostictus	Beaugregory	152	22.45	964	1.424	0.182	4368.432	6.453	0.893
Stegastes partitus	Bicolor Damselfish	266	39.29	3276	4.839	0.465	6403.307	9.458	1.234
Stegastes planifrons	Threespot Damselfish	125	18.46	1256	1.855	0.263	12824.066	18.942	3.284
Stegastes variabilis	Cocoa Damselfish	145	21.42	844	1.247	0.144	4576.983	6.761	0.892
Priacanthidae									
Heteropriacanthus cruentatus	Glasseye Snapper	9	1.33	10	0.015	0.005	1720.414	2.541	1.125
Scaridae									
Cryptotomus roseus	Bluelip Parrotfish	158	23.34	651	0.962	0.092	3942.795	5.824	0.844
Scarus UNK	PARROTFISH species	1	0.15	100	0.148	0.148	47.790	0.071	0.071
Scarus coeruleus	Blue Parrotfish	2	0.30	2	0.003	0.002	3588.032	5.300	3.949
Scarus guacamaia	Rainbow Parrotfish	1	0.15	1	0.001	0.001	275.949	0.408	0.408
Scarus iseri	Striped Parrotfish	256	37.81	4474	6.609	0.605	62757.990	92.700	8.884
Scarus taeniopterus	Princess Parrotfish	142	20.97	781	1.154	0.135	24644.115	36.402	5.588
Scarus vetula	Queen Parrotfish	43	6.35	81	0.120	0.023	19063.539	28.159	7.314
Sparisoma UNK	PARROTFISH species	3	0.44	5	0.007	0.004	2.248	0.003	0.002
Sparisoma atomarium	Greenblotch Parrotfish	118	17.43	786	1.161	0.157	960.515	1.419	0.222
Sparisoma aurofrenatum	Redband Parrotfish	298	44.02	2651	3.916	0.260	98702.979	145.795	11.742
Sparisoma chrysopterum	Redtail Parrotfish	30	4.43	51	0.075	0.015	6064.219	8.957	2.297
Sparisoma radians	Bucktooth Parrotfish	79	11.67	328	0.484	0.084	493.771	0.729	0.169
Sparisoma rubripinne	Yellowtail Parrotfish	49	7.24	150	0.222	0.054	22160.730	32.734	6.990
Sparisoma viride	Stoplight Parrotfish	205	30.28	1025	1.514	0.135	108319.808	160.000	18.429
Sciaenidae									
Equetus lanceolatus	Jackknife Fish	2	0.30	3	0.004	0.003	7.625	0.011	0.008
Equetus punctatus	Spotted Drum	10	1.48	10	0.015	0.005	1164.631	1.720	0.734
Odontoscion dentex	Reef Croaker	2	0.30	4	0.006	0.005	120.033	0.177	0.125
Pareques acuminatus	Highhat	3	0.44	3	0.004	0.003	11.321	0.017	0.012
Scombridae									
Scomberomorus regalis	Cero	5	0.74	12	0.018	0.010	20195.937	29.832	17.065

Family *Species Name*	Common Name	Frequency Total	Frequency %	Abundance Total	Abundance Mean	Abundance ± SE	Biomass (g) Total	Biomass (g) Mean	Biomass (g) ± SE
Scorpaenidae									
Scorpaena plumieri	Spotted Scorpionfish	1	0.15	1	0.001	0.001	428.531	0.633	0.633
Serranidae									
Cephalopholis cruentata	Graysby	58	8.57	81	0.120	0.017	11278.998	16.660	3.311
Cephalopholis fulva	Coney	58	8.57	101	0.149	0.023	12665.749	18.709	3.803
Diplectrum formosum	Sand Perch	13	1.92	35	0.052	0.018	154.951	0.229	0.093
Epinephelus adscensionis	Rock Hind	1	0.15	1	0.001	0.001	29.883	0.044	0.044
Epinephelus guttatus	Red Hind	142	20.97	247	0.365	0.034	59985.708	88.605	10.364
Epinephelus striatus	Nassau Grouper	1	0.15	1	0.001	0.001	84.142	0.124	0.124
Hypoplectrus aberrans	Yellowbelly Hamlet	13	1.92	18	0.027	0.008	146.875	0.217	0.089
Hypoplectrus chlorurus	Yellowtail Hamlet	42	6.20	65	0.096	0.018	543.244	0.802	0.169
Hypoplectrus guttavarius	Shy Hamlet	8	1.18	11	0.016	0.006	79.680	0.118	0.058
Hypoplectrus indigo	Indigo Hamlet	5	0.74	13	0.019	0.012	99.496	0.147	0.070
Hypoplectrus nigricans	Black Hamlet	64	9.45	105	0.155	0.022	1175.528	1.736	0.271
Hypoplectrus UNK	HAMLET species	66	9.75	95	0.140	0.019	481.698	0.712	0.146
Hypoplectrus puella	Barred Hamlet	148	21.86	300	0.443	0.040	1881.290	2.779	0.332
Hypoplectrus unicolor	Butter Hamlet	44	6.50	54	0.080	0.014	598.849	0.885	0.184
Liopropoma rubre	Peppermint basslet	2	0.30	2	0.003	0.002	6.166	0.009	0.009
Mycteroperca interstitialis	Yellowmouth Grouper	1	0.15	1	0.001	0.001	572.519	0.846	0.846
Rypticus saponaceus	Greater Soapfish	1	0.15	1	0.001	0.001	29.071	0.043	0.043
Serranus baldwini	Lantern Bass	62	9.16	275	0.406	0.064	452.910	0.669	0.140
Serranus UNK	SEABASS species	9	1.33	16	0.024	0.010	27.540	0.041	0.019
Serranus tabacarius	Tobaccofish	60	8.86	108	0.160	0.024	857.443	1.267	0.249
Serranus tigrinus	Harlequin Bass	103	15.21	221	0.326	0.038	1847.768	2.729	0.501
Serranus tortugarum	Chalk Bass	112	16.54	2741	4.049	0.602	3229.551	4.770	1.210
Sparidae									
Calamus calamus	Saucereye Porgy	29	4.28	44	0.065	0.013	8000.299	11.817	2.966
Calamus UNK	PORGY species	1	0.15	1	0.001	0.001	1.042	0.002	0.002
Calamus penna	Sheepshead Porgy	1	0.15	1	0.001	0.001	736.869	1.088	1.088
Calamus pennatula	Pluma Porgy	12	1.77	13	0.019	0.006	3272.453	4.834	2.050
Diplodus argenteus caudimacula	Silver Porgy	1	0.15	1	0.001	0.001	227.954	0.337	0.337
Sphyraenidae									
Sphyraena barracuda	Great Barracuda	19	2.81	22	0.032	0.009	74855.025	110.569	32.790
Sygnathidae									
Cosmocampus elucens	shortfin Pipefish	9	1.33	10	0.015	0.005	2.739	0.004	0.002
Hippocampus UNK	PIPEFISH species	2	0.30	2	0.003	0.002	0.081	0.000	0.000
Hippocampus reidi	Longsnout Seahorse	1	0.15	1	0.001	0.001	0.633	0.001	0.001
Syngathus dawsoni	NOT RECORDED	4	0.59	4	0.006	0.003	2.248	0.003	0.003
Synodontidae									
Synodus intermedius	Sand Diver	89	13.15	161	0.238	0.052	1794.722	2.651	0.750
Synodus saurus	Bluestriped Lizardfish	3	0.44	4	0.006	0.004	7.557	0.011	0.006
Tetraodontidae									
Canthigaster rostrata	Sharpnose Puffer	227	33.53	430	0.635	0.049	899.259	1.328	0.152
Sphoeroides spengleri	Bandtail Puffer	41	6.06	57	0.084	0.016	218.736	0.323	0.082
Sphoeroides testudineus	Checkered Puffer	8	1.18	14	0.021	0.010	150.267	0.222	0.164

Appendix F: Project Team

Laurie Bauer (NCCOS CCMA)
Jim Beets (UH)
Rafe Boulon (NPS)
Marilyn Brandt (NPS)
Andy Davis (NPS)
Chris Caldow (NCCOS CCMA)
John Christensen (NOAA CRCP)
Bryan Costa (NCCOS CCMA)
Randy Clark (NCCOS CCMA)
Michael Coyne (Seaturtle.org)
Kimberly Edwards (NCCOS CCMA)
Alan Friedlander (USGS)
Kelly Gleason (ONMS PR)
Ricky Grober
Kim Haughty
Jimmy Herlan
Sarah Davidson Hile (NCCOS CCMA)
Susie Holst (NOAA CRCP)
Chris Jeffrey (NCCOS CCMA)
Olaf Jensen
Thomas Kelley (NPS)
Matt Kendall (NCCOS CCMA)
Tom McGrath (NCCOS CCMA)
Charles Menza (NCCOS CCMA)
Jeff Miller (NPS)
Wendy Morrison
Mark Monaco (NCCOS CCMA)
Erinn Muller (NPS/NOAA/FIT)
Simon Pittman (NCCOS CCMA)
Caroline Rogers (USGS)
Ben Ruttenberg (NMFS SEFSC)
Carrie Stengel (NPS)
Jason Vasques (USVI DPNR)
Jenny Waddell (NOAA CRCP)
Kimberly Woody Roberson (NCCOS CCMA)
Rob Waara (NPS)

This report was made possible through the participation of the individuals recognized above. Their efforts with data collection, fi eld support, and report production are highly appreciated.

U.S. Department of Commerce

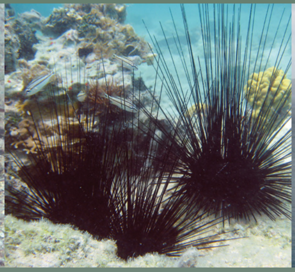
Rebecca Blank, *Acting Secretary*

National Oceanic and Atmospheric Administration

Kathryn Sullivan, *Acting Under Secretary for Oceans and Atmosphere*

National Ocean Service

Holly Bamford, *Assistant Administrator for Ocean Service and Coastal Zone Management*

The mission of the National Centers for Coastal Ocean Science is to provide managers with scientific information and tools needed to balance society's environmental, social and economic goals. For more information, visit: http://www.coastalscience.noaa.gov/.

Coral reef ecosystems of St. John, U.S. Virgin Islands - NOAA Technical Memorandum NOS NCCOS 152