Baseline Assessment of Guánica Bay, Puerto Rico in Support of Watershed Restoration

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ABOUT THIS DOCUMENT

This document presents a comprehensive overview of results from a recent baseline assessment of Guánica Bay, Puerto Rico and its surrounding coral reef ecosystem. The report details: (1) a biogeographic assessment of the coral reef ecosystem outside the bay; (2) contaminant (e.g., PAHs, PCBs, pesticides, heavy metals) magnitudes and distributions in surface sediments (inside the bay, outside the bay and in the watershed streams) and in coral tissues (mustard hill coral, *Porites astreoides*); and (3) spatial and temporal patterns in sedimentation rates and surface water nutrient concentrations.

The efforts discussed here were led by the National Centers for Coastal Ocean Science (NCCOS), with significant participation from partners, both internal to NOAA (e.g., Coral Reef Conservation Program) and external to NOAA (e.g., University of Puerto Rico). NCCOS has been proactive in collaborating with other NOAA line offices as well as federal, state and nongovernmental organization partners to maximize cost-sharing efforts and reach its goals. Their efforts and extramural funding has made it possible to complete areas that would have otherwise been unobtainable through federal funding alone.

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EXECUTIVE SUMMARY

David Whitall[1]

Guánica Bay is a major estuary on the southwest coast of Puerto Rico. Significant coral reef ecosystems are present outside the bay. These valuable habitats may be impacted by transport of sediments, nutrients and contaminants from the watershed, through the bay and into the offshore waters. The National Oceanic and Atmospheric Administration's (NOAA) National Centers for Coastal Ocean Science (NCCOS), in consultation with local and regional experts, conducted an interdisciplinary assessment of coral reef ecosystems, contaminants, sedimentation rates and nutrient distribution patterns in and around Guánica Bay. This work was conducted using many of the same protocols as ongoing monitoring work underway elsewhere in the U.S. Caribbean and has enabled comparisons among coral reef ecosystems between this study and other locations in the region.

Guánica Bay, a major estuary in southwest Puerto Rico, is home to many species of coral, fish, seagrass and invertebrates. Photo: NCCOS.

This characterization of Guánica marine ecosystems establishes benchmark conditions that can be used for comparative documentation of future change, including possible negative outcomes due to future land use change, or improvement in environmental conditions arising from management actions.

This report is organized into six chapters that represent a suite of interrelated studies. Chapter 1 provides a short introduction to the study area. Chapter 2 is focused on biogeographic assessments and benthic mapping of the study area, including new surveys of fish, marine debris and reef communities on hardbottom habitats in the study area. Chapter 3 quantifies the distribution and magnitude of a suite of contaminants (e.g., heavy metals, PAHs, PCBs, pesticides) in both surface sediments and coral tissues. Chapter 4 presents results of sedimentation measurements in and outside of the bay. Chapter 5 examines the distribution of nutrients in in the bay, offshore from the bay and in the watershed. Chapter 6 is a brief summary discussion that highlights key findings of the entire suite of studies.

The main findings of each chapter are as follows:

Chapter 1: Introduction

- Historical agricultural land use (sugar cane farming) has given way to a mosaic of land use types. Current land use consists of agriculture, including coffee farming in the mountains, small urban areas and forest.
- Human alteration of the hydrology of the watershed via irrigation and drainage projects has resulted in a complex water cycle within the watershed.
- Offshore marine ecosystem features include both emergent and submergent reefs, seagrass beds, mangrove islands and fringing mangroves.
- The general long shore current direction is east to west.

Chapter 2: Fish Communities and Associated Benthic Habitats in the Guánica Bay Region of Southwest Puerto Rico

- Coral cover was generally low across the study region.
- Gorgonian, sea grass and coral cover were higher in the La Parguera region, to the west of the study area.
- The study region in Guánica Bay had lower coral cover than other studies in the region.

[1] NOAA/NOS/NCCOS

Chapter 3: Contaminants in Surficial Sediments and Coral Tissues of Guánica Bay

- Surface sediment samples demonstrated unusually high concentrations of total chlordane, total PCBs, nickel and chromium.
- A variety of other contaminants (total DDT, total PAHs, arsenic, copper, mercury and zinc) were also at levels which may indicate sediment toxicity.
- Elevated concentrations of pollutants were generally limited to inside the bay, although several contaminants (chlordane, PCBs, chromium) were elevated outside the bay as well.
- With the exception of chromium, all of these contaminants were detected in coral tissues mustard hill coral, (*Porites asteroides*), although it is unclear at what level these contaminants affect coral health.

Chapter 4: Terrigenous Sedimentation Patterns at Reefs Adjacent to the Guánica Bay Watershed

- Sediment trap accumulation rates vary both spatially and temporally.
- The composition of the deposited material remains fairly constant, independent of sedimentation rate. This indicates that re-suspension of sediments is likely very important in this system.
- No east-west pattern in sedimentation rates was observed.

Chapter 5: Spatial and Temporal Variability in Surface Water Nutrients in and around Guánica Bay, Puerto Rico

- Observed nutrient concentrations suggest a strong watershed source of nutrients with dilution as nutrients move into the bay and offshore.
- Phosphorus rarely exceeded previously published threshold values for coral impact; however, approximately 10% of the observations had DIN values which exceeded this proposed threshold.
- In general, there was no obvious nutrient spike related to the wastewater treatment plant (WWTP) in the bay. On one occasion the site closest to the WWTP had an anomalously high nitrogen value, suggesting that something unusual may have occurred around that time, but that was an isolated incident.

Chapter 6: Conclusions

- This suite of environmental data (biological and stressors) represents an important baseline against which future change can be measured. Change might occur due to human induced degradation (e.g., due to coastal development) or due to improvements associated with management actions.
- Further monitoring and assessments are needed to detect changes in the ecosystem over a variety of time scales ranging from relatively short-term responses in sediment loading, to potentially decadal-long recovery processes for reef systems.

Chapter 1: Introduction and Background

David Whitall[1], Laurie Bauer[1,2] and Lia Brune[1]

BACKGROUND

Coral reefs are among the largest and most biologically diverse ecosystems on the planet. Reefs provide a variety of goods and services to people, such as seafood, shoreline protection, recreation and tourism, new medicines, and aesthetics (Burke and Maidens, 2004; Dixon, 1998; Moberg and Folke, 1999). Costanza et al. (1997) found that globally these ecosystem services are valued at $US 375 billion per year.

Yet, over half of the world's coral reefs are threatened by human activity (Bryant et al., 1998). Increased runoff of sediment, nutrients and pollutants has been correlated to the degradation of coral reefs (Fabricius, 2005). In the Caribbean, it is estimated that one third of coral reefs are threatened by land-based sources of pollution, much of which is exacerbated by coastal development (Burke and Maidens, 2004; Moberg and Folke, 1999). Although pollution is a known cause of the decline of coral reefs, details of the relationship between contaminants and corals are not well understood. There are currently no established thresholds for individual pollution stressors indicating concentration limits above which adverse biological effects or the potential for effects occur in coral tissues.

This study presents a baseline assessment of the biological resources of coral reefs off the coast of Guánica Bay, Puerto Rico, as well as the magnitude and spatial distribution of pollutant stressors (sediments, nutrients, contaminants) to this system. This information will provide ecosystem managers a reference point against which to evaluate the success of upland watershed restoration efforts and best management practices (e.g., those planned by the NOAA Restoration Center, USDA Natural Resource Conservation Service and non-governmental organizations.) This baseline assessment supports the Guánica Bay Watershed Management Plan which was created in response to the U.S. Coral Reef Task Force Local Action Strategy for Puerto Rico. The plan addresses land-based sources of pollution and proposes methods to reduce the loss of coral reefs through implementing effective watershed management planning (CWP, 2008).

STUDY AREA

Marine Ecosystem

The study area encompasses Guanica Bay itself and the coral reef ecosystems located offshore. The bay is approximately 3 km long and 2.5 km wide, at its widest point. The mouth of the bay is relatively narrow (less than 1 km), which influences the exchange with the open ocean. There are no living reefs within the bay. The coral reef ecosystem of this region is a complex spatial mosaic of habitat types including extensive coral reefs, submerged aquatic vegetation and mangroves, which in turn support diverse fish communities (Pittman et al., 2010). The shelf within the marine study area is three to four kilometers wide and average water depth is 12 meters. Both emergent and submergent reefs separate the inner shelf from the outer shelf. Sea grass beds and both fringing mangroves and mangrove islands are scattered throughout the study area (Bauer et al., 2012; Kendall et al., 2001). The predominant long shore flow of the area is east to west (CFMC, 1998). A sediment plume originating from Guánica Bay and flowing to the west is commonly observed on satellite imagery.

Watershed Hydrology

Guánica Bay is located on the coast of southwestern Puerto Rico in the Guánica municipality (Figure 1.1). The bay is fed directly by one river, the Rio Loco, which runs year round, but exhibits high flows during the rainy season. In the 1950s, the Southwest Water Project increased the drainage area of the watershed through a series of reservoirs, canals and hydroelectric plants. Consequently, the Guánica watershed actually encompasses five basins and associated reservoirs: Lago Yahuecas, Lago Guayo, Lago Prieto, Lago Luccheti and Lago Loco, as well as the Lajas Irrigation Canal within the Lajas Valley Agricultural Reserve (CWP, 2008). This water diversion system persists today, moving water from the upper watershed to the Lajas Valley to the west. This system not only provides agricultural irrigation water, it also provides drinking water to the municipalities of Sabana Grande, Cabo Rojo, San Germán and Lajas. Drainage canals also move water, primarily originating from overland runoff, to the east where the drainage canal re-joins the Rio Loco just upstream of its entry point into the bay. The Guánica Lagoon, previously situated to the north of the bay, was historically the largest freshwater lagoon in Puerto Rico and may have served as a natural filter and sediment sink prior to the discharge of the Rio Loco into the bay. Following alterations by the Southwest Water Project, a large portion of the Lajas Valley (over 20,000 hectares; Sotomayor and Pérez, 2011) including the lagoon and its surrounding wetlands system, was drained for agricultural use.

1. NOAA/NOS/NCCOS
2. CSS-Dynamac, Fairfax, VA

Figure 1.1. Location of Guanica Bay, Puerto Rico.

Watershed Land Use

Land use in the Guánica Bay/Rio Loco watershed has changed over time (CWP, 2008). Historically, sugar cane, coffee, tobacco and sustenance crops were grown in Lajas Valley. Agricultural production has declined since the 1950s due to a shift in government policy and a move towards small industry. This led to the abandonment of farmlands, most of which became pastureland or urban areas (Cramer and Hobbs, 2007). This conversion of land use from forested to agricultural, and then to urban development has contributed significantly to the increased nutrient and sediment loads, the latter of which can lead to decreases in water clarity, along the shelf of southwest Puerto Rico (Larsen and Webb, 2009). A movement from shade-grown to sun-grown coffee cultivation in Puerto Rico began around the same time period in efforts to increase yields. The sun-grown cultivation method requires more chemical inputs than shade-grown coffee and causes higher runoff due to the lack of canopy cover (Perfecto et al.,1996). Returning to shade-grown coffee cultivation is one of many watershed management practices proposed as part of the Guánica Bay Watershed Management Plan. Other methods for reducing sediment load (e.g., hydroseeding, stream bank stabilization) are also being considered. Increasing urban and industrial developments around Guánica Bay, as well as further upland, have led to increased pollutant and nutrient loadings entering the bay via storm water runoff and wastewater effluent (Warne et al., 2005).

Current land use in the watershed includes 48% forested, 43% agriculture and 9% urban (CWP, 2008). The Guánica State Forest (Bosque Estatal de Guánica) and UNESCO Biosphere Reserve extends to the west and east of Guánica Bay. The northern area of the watershed is generally mountainous and is characterized by a mix of forested and agricultural lands, including coffee plantations, which may be especially important in terms of sediment loss. Closer to the coast, the Lajas Valley Agricultural Reserve extends north of Guánica Bay to the southwest corner of the island. There are two sewage-treatment plants in the tidal areas of the Guánica watershed (CWP, 2008). The largest of these is located on the eastern shore of the bay and discharges its treated effluent into the bay. This facility has tertiary treatment capabilities. Guánica Bay is bordered by the municipality of Guánica, which encompasses the coastal towns of Guánica and Ensenada. The municipality has a population of approximately 22,000 and the regions surrounding Guánica Bay have historically been home to various industries including sugar processing, textiles and fertilizer mixing (U.S. Census Bureau, 2003).

Increased sediment and nutrient discharge into the Rio Loco and ultimately to Guánica Bay's near shore reefs is attributed to land use changes (Warne et al., 2005). The conversion of upland forested lands to agriculture and man-made canals has altered the natural hydrology of the watershed causing upland soil erosion, in-stream channel erosion, loss of lagoons and the downstream transport of sediment (CWP, 2008). A study by Larsen and Santiago-Roman (2001) showed that large quantities of sediment from upland hill-slopes were deposited in lower hill-slopes and stream valleys all across Puerto Rico during the agriculture peak of the mid twentieth century. Larsen and Santiago-Roman (2001) estimated that this stored sediment would be available for dispersal for a century or more, and has the potential to reach the offshore coral reef systems via riverine transport and export from the bay.

PREVIOUS STUDIES

The land use practices and watershed changes outlined above have resulted in large amounts of sediment being distributed in the Rio Loco river valley (CWP, 2008). Storm events and seasonal flooding also transport large amounts of sediment to the coastal waters. Previous work has indicated that contaminant levels are higher in Guánica Bay compared to the surrounding area (Pait et al., 2008; Pait et al., 2009). In particular, concentrations of PCBs (polychlorinated biphenyls) and DDT (dichlorodiphenyltrichloroethane) in sediment samples from within the bay were above established guideline levels that are frequently associated with toxic effects to the biota (Pait, 2008). In addition, among coral tissue mustard hill coral, (*Porites astreoides*) samples collected from southwest Puerto Rico, levels of these same contaminant classes were highest in samples taken closest to Guánica Bay, and there was evidence of a downstream gradient moving away from the bay (Pait et al. et al., 2009).

MANAGEMENT ACTIONS

The threats of upstream watershed practices to coral reefs and the nearshore marine environment have been gaining recognition. Guánica Bay and the adjacent marine waters have been identified as a “management priority area” by the Commonwealth of Puerto Rico and NOAA’s Coral Reef Conservation Program (Commonwealth of Puerto Rico and NOAA CRCP, 2010) and as a U.S. Coral Reef Task Force Priority Watershed (USCRTF, 2011). In a recent Guánica Bay watershed management plan, several critical issues were outlined in regards to land-based sources of pollution (CWP, 2008). These include: upland erosion from coffee agriculture, filling of reservoirs with sediment, in-stream channel erosion, loss of historical Guánica lagoon, legacy contaminants and sewage treatment (CWP, 2008). The plan recommended several management actions that could be taken to reduce impacts of land-based sources of pollution, which form the basis of Guánica watershed restoration efforts. The Guánica watershed restoration project is a multi-disciplinary, multi-faceted effort with numerous federal and territorial partners, including the NOAA National Centers for Coastal Ocean Science (NCCOS) COAST and Biogeography Branches, Coral Reef Conservation Program, Restoration Center, USDA’s Natural Resources Conservation Service and multiple non-governmental organizations. Current and proposed restoration projects in the watershed include the restoration of the historic Guánica Lagoon and adjacent coastal wetlands, conversion of sun-grown to shade-grown coffee, sewage treatment upgrades and stabilization of the Rio Loco river channel. A primary objective of these restoration activities is to reduce nutrient and sediment loading into the marine environment, with the ultimate goal of improving coral reef health.

STUDY OBJECTIVES

In order to measure the effectiveness of restoration efforts, it is critical that baseline data be collected with which potential future changes can be quantified. The objective of the ecological characterization of the Guánica Bay region is to conduct a baseline assessment with which the success of these efforts can be evaluated.

The objectives of this assessment are to:

1. Characterize fish and benthic communities of the Guánica Bay region, and compare these metrics with the neighboring La Parguera study area, which has been monitored by NOAA since 2001 (Pittman et al., 2010).
2. Measure the spatial and temporal differences in sedimentation (flux and sediment characteristics) to the reefs, and identify potential drivers (e.g., precipitation, wave height).
3. Quantify how organic and inorganic contaminants in surface sediments and coral tissues mustard hill coral vary spatially across the study area, and how the magnitudes of those concentrations compare to previously published sediment quality guidelines and regional/national historic values.
4. Determine the spatiotemporal variability in surface water nutrient concentrations in the watershed, bay and reef systems, including the relationship with land use and precipitation.

These data will serve as a baseline with which to compare potential changes following restoration of the Guánica Bay watershed. In future years, follow up assessments will be necessary to track these changes, and determine the efficacy of management efforts.

REFERENCES

Bauer, L.J., K. Edwards, K.K.W. Roberson, M.S. Kendall, S. Tormay, and T.A. Battista. 2012. Shallow-water benthic habitats of Southwest Puerto Rico. NOAA Technical Memorandum NOAA NOS NCCOS 155. Silver Spring, MD.

Bryant, D., L. Burke,J. McManus, M. Spalding. 1998. Reefs at Risk: A Map-based Indicator of Threats to the World's Coral Reefs. World Resources Institute, Washington, DC..

Burke, L. and J. Maidens. Reefs at risk in the Caribbean. 2004. World Resources Institute. Washington, D.C.

Caribbean Fishery Management Council (CFMC). 1998. Essential fish habitat (EFH) generic amendment to the fisheries management plans (FMPs) of the U.S. Caribbean including a draft environmental Assessment: Volume 1. San Juan, PR.

Costanza, R. R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. V. O'Neill, J. Paruelo, R. G. Raskin, P. Sutton, and M. van den Belt. 1997. The value of the world's ecosystem services and natural capital. Nature. 387: 253-260.

Cramer, V. and Hobbs, R. 2007. Old fields: Dynamics and Restoration of abandoned farmland. Island Press, Washington, D.C.

Center for Watershed Protection (CWP). 2008. Guanica Bay Watershed Management Plan. NOAA Coral Reef Program and DRNA, Puerto Rico. http://www.cwp.org/online-watershed-library/cat_view/64-manuals-and-plans/81-watershed-management-plans-and-guidance

Commonwealth of Puerto Rico and NOAA Coral Reef Conservation Program. 2010. Puerto Rico's Coral Reef Management Priorities. NOAA. Silver Spring, MD..

Dixon, J. A. 1998. Economic values of coral reef: what are the issues?" in Hatziolos, M.E., et al. (eds.) Coral reefs: challenges and opportunities for sustainable management. World Bank, Washington, pp. 157–162.

Fabricius, K.E.. 2005. Effects of terrestrial runoff on the ecology of corals and coral reefs: review and synthesis. Mar. Poll. Bull. 50: 125–146.

Kendall, M.S., C.R. Kruer, K.R. Buja, J.D. Christensen, M. Finkbeiner, M.E. Monaco. 2011. Methods used to map benthic habitats of Puerto Rico and the US Virgin Islands. National Oceanic and Atmospheric Administration, National Ocean Service, Biogeography Team Technical Report, Silver Spring, MD.

Larsen, M.C. and A. Santiago-Román. 2001. Geomorphic Processes and Riverine Habitat Mass wasting and sediment storage in a small montane watershed: an extreme case of anthropogenic disturbance in the humid tropics. American Geophysical Union Monograph Series.

Larsen, M.C. and R.M. Webb. 2009. Potential Effects of Runoff, Fluvial Sediment, and Nutrient Discharges on the Coral Reefs of Puerto Rico. Journal of Coastal Research. 25:189-208.

Moberg, F. and C. Folke. Ecological goods and services of coral reef ecosystems. Ecological Economics 1999. 29: 215-233.

Pait, A.S., D.R. Whitall, C.F.G. Jeffrey, C. Caldow, A.L. Mason, G.G. Lauenstein, and J.D. Christensen. 2008. Chemical contamination in southwest Puerto Rico: an assessment of organic contaminants in near shore sediments. Mar. Poll. Bull. 56:580-606.

Pait, A.S.,C.F.G Jeffrey, C. Caldow, D.R. Whitall, S.I. Hartwell, A.L. Mason and J.D. Christensen. 2009. Chemical Contaminants in the Coral Porites astreoides from Southwest Puerto Rico. NOS NCCOS 91. NOAA/NOS/Center for Coastal Monitoring and Assessment. Silver Spring, MD.

Perfecto, I., R.A. Rice, R. Greenberg, R. and M.E. Van der Voort. 1996. Shade Coffee: A Disappearing Refuge for Biodiversity. BioScience. 46: 598-608.

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.

Sotomayor-Ramírez, D. and L. Pérez-Alegría. 2011. An assessment of agricultural activities in the eastern portion of the Lajas Valley Agricultural Reserve, within Lajas and Guánica municipalities. University of Puerto Rico, Mayagüez, Agricultural Experiment Station, Unpublished Manuscript. 16 pp, with appendices.

U.S. Census Bureau. (2003). U.S. Census Bureau http://www.census.gov/popest/archives/2000s/vintage_2002/PRM-EST2002-01.html

U.S. Coral Reef Task Force. 2011. U.S. Coral Reef Task Force Resolution 25.1. http://www.coralreef.gov/meeting25/pdf/resolution.pdf

Warne, A.G., R.M.T. Webb, M.C. Larsen. Water, sediment, and nutrient discharge characteristics in rivers of PR and their potential influence on coral reefs. USGS Sci. Inv. Report 2005-5206. Reston, VA. 2005.

Chapter 2: Fish Communities and Associated Benthic Habitats in the Guánica Bay Region of Southwest Puerto Rico

Laurie Bauer[1,2,3], Kimberly Edwards[1,2] and Chris Caldow[1]

BACKGROUND

The goal of this component of the baseline assessment is to provide a spatially-explicit characterization of fish and benthic communities in the marine ecosystem adjacent to Guánica Bay. The approach to characterizing the marine resources of the region included two complimentary components. Recently, an updated fine-scale benthic habitat map for southwest Puerto Rico, including Guánica Bay, was completed (Bauer et al., 2012). This marks the second such effort NOAA has conducted to map shallow water marine benthic habitats of this region. Components of the new mapping product that mark an improvement over NOAA's previous digital maps (Kendall et al., 2001) include an expanded habitat classification scheme, smaller minimum mapping unit and more recent aerial imagery. In addition, within the extent used for this mapping effort, a larger total area was mapped than in the previous mapping effort. For example, some areas that were mapped as unknown in the previous effort were able to be delineated in the new map due to better remote sensing imagery.

Southwest Puerto Rico is represented by a diverse array of benthic habitats. The spatial distribution of two of the mapped attributes, major geomorphological structure type and dominant biological cover, are displayed in Figures 2.1 and 2.2. Unconsolidated sediments (sand and mud) are most common in the nearshore environment, including Guánica Bay and the La Parguera lagoon (Figure 2.1). Portions of the protected shoreline and offshore cayes are lined with mangroves. Seagrass beds of various patchiness levels are also common in the nearshore areas (Figure 2.2). An extensive reef complex, a mix of both low rugosity (e.g., pavement) and high rugosity (aggregate reef, spur and groove) hardbottom, ranges the entire length of the bank-shelf in southwest Puerto Rico. Algae, which includes both macroalgae and turf algae, was most often the dominant biological cover on coral reef and hardbottom habitats. While live coral was rarely mapped as the major cover, there were exceptions. High density finger coral (*Porites porites)* fields were occastionally present in back reef/reef flat environments, including south of Cayos de Caña Gorda.

The new benthic habitat map serves as a current snapshot of the extent of marine benthic habitats in the region and will support the management and conservation of the watershed and coastal marine waters of Guánica and greater southwest Puerto Rico. Future mapping of southwest Puerto Rico can be used to monitor changes in the benthic habitats in the La Parguera/Guánica region following restoration efforts in the Guánica Bay watershed. One of the major goals of these restoration efforts is to reduce input of sediments, nutrients and contaminants into the marine environment. Reduction in these terruginous inputs could have several potential effects on the marine habitats. For example, improved water clarity would likely benefit growth of seagrass and corals.

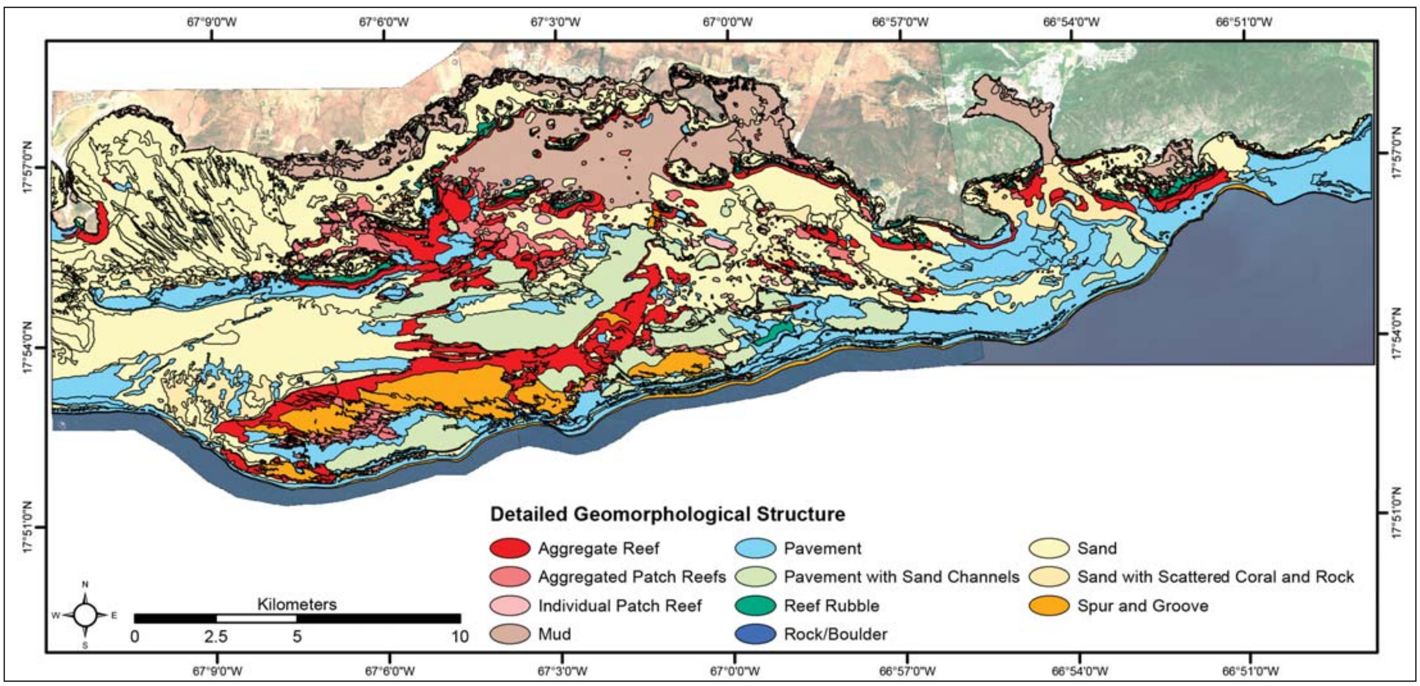

Figure 2.1. Detailed geomorphological structure in the Guánica/La Parguera study area (Bauer et al., 2012).

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2. CSS-Dynamac, Fairfax, VA

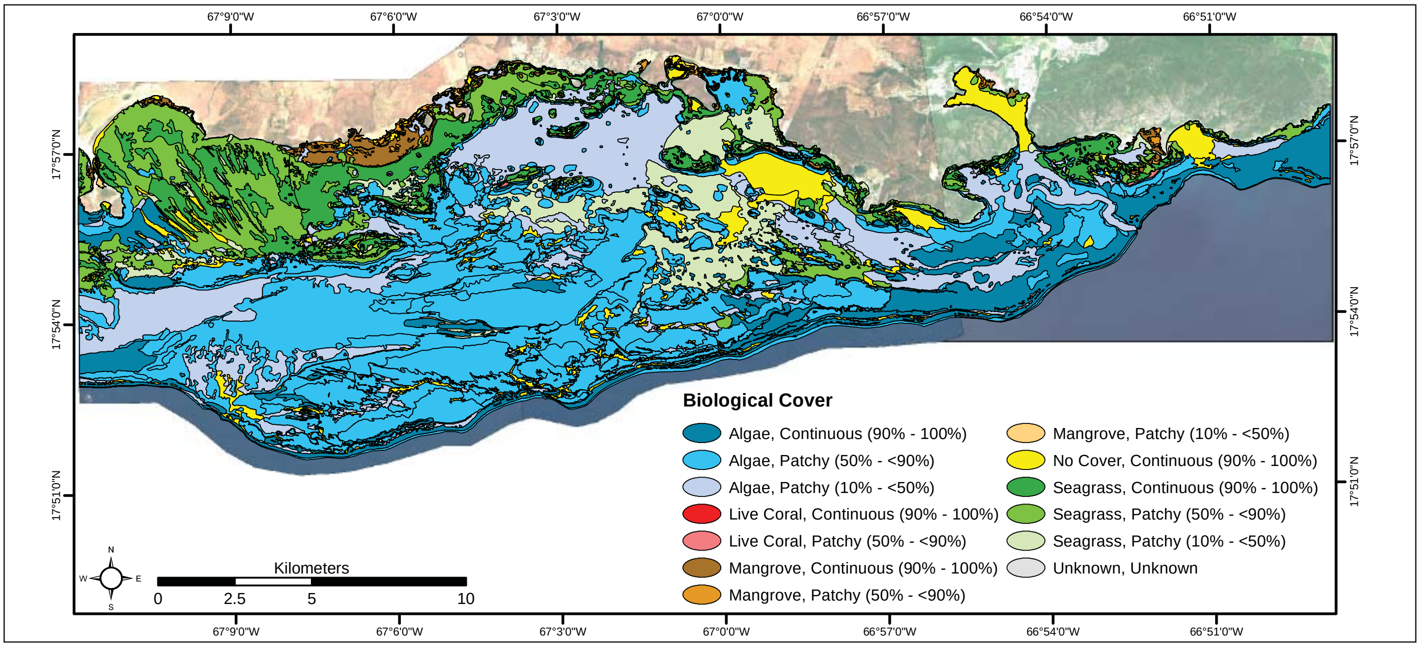

Figure 2.2. Major biological cover in the Guánica/La Parguera study area (Bauer et al., 2012).

More details on the methods and classification scheme used to create the benthic habitat map can be found in Bauer et al. (2012). Here, we expand on the recent mapping efforts by presenting baseline data on fish and benthic communities. It is anticipated that additional monitoring will be conducted following the implementation of the restoration activities to assess potential changes in benthic cover, population estimates and size spectra of fish over time.

Since 2001, the National Centers for Coastal Ocean Science (NCCOS) Biogeography Branch has worked to characterize, monitor and assess the status of the marine environment in neighboring La Parguera, Puerto Rico, located west (downstream) of Guánica Bay (Pittman et al., 2010). The area is designated as a natural reserve by the Puerto Rico Department of Natural and Environmental Resources. While it is important to characterize habitats and fish communities in close proximity to Guánica Bay, it is important to note that the La Parguera region is, with respect to circulation patterns and pollutant transport, an extension of this same system. The prevailing surface currents are driven by the trade winds and flow westward, meaning that effluent from Guánica Bay flows downstream towards La Parguera. Hence, the La Parguera region likely serves as a sink for land-based sources of pollution discharged from Guánica Bay. Due to this oceanographic connectivity, the La Parguera region was also included in this assessment and potential differences between the two regions were explored.

The objectives of this assessment are to:

1. Characterize fish and benthic communities of the Guánica Bay region. This data will serve as a baseline with which to compare potential changes following restoration of the Guánica Bay watershed.
2. Compare fish and benthic community metrics in the Guánica Bay study area with the neighboring La Parguera study area.

METHODS

Site Selection

Using NOAA's established Coral Reef Ecosystem Monitoring protocols (e.g., Pittman et al., 2008 and 2010), field surveys were conducted in August 2010 to characterize the fish communities and associated habitats in the Guánica Bay marine ecosystem. Sites were selected using a random-stratified survey design with habitat type (coral reef and hardbottom, unconsolidated sediments, mangrove) and geographic region (west, central, east of Guánica) as the main strata. The geographic strata were previously designated by the NCCOS COAST Branch for nutrient and contaminant sampling (Chapters 3 and 4). The eastern most strata is located upstream of the Guánica Bay outlet, while the central and western strata are located downstream of the bay (Figure 2.3). Although sample size in the initial baseline assessment was not large enough to make robust statistical comparisons between the three regions, they are noted here for consistency with COAST's complimentary sampling efforts and will be used for qualitative descriptions of patterns across the study region. As more surveys are conducted over time, the increased sample size will enable more comprehensive statistical comparisons between the three geographic strata.

Guánica Bay itself was not included in the survey area due to poor visibility, rendering it unsuitable for visual survey methods. To effectively survey fish using the underwater visual methods, it is necessary that the divers have a minimum of 2 m visibility. During previous dives within the bay, it was determined that visibility was consistently below this threshold.

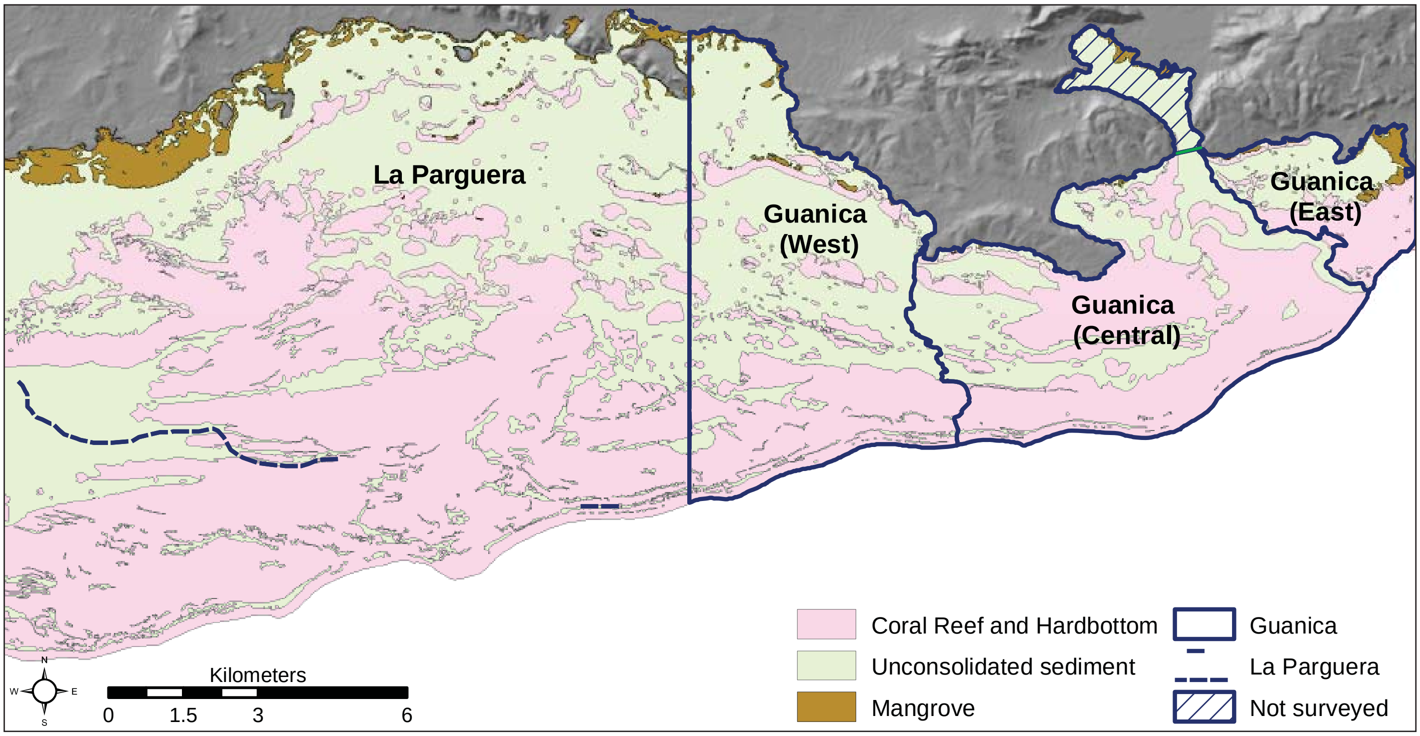

Figure 2.3. Guánica and La Parguera study areas and benthic habitat strata used to stratify surveys of benthic habitat composition, fish communities and marine debris.

The number of sites selected within each strata was determined based on logistics and results from statistical analyses of variance (Menza et al., 2006). As the new benthic habitat map (Bauer et al., 2012) was not completed at the time, the previous NOAA benthic habitat map (Kendall et al., 2001) was used as the basis for site stratification. The "coral reef and hardbottom" strata comprised bedrock, pavement, rubble and coral reef, while the "unconsolidated sediments" stratum comprised submerged aquatic vegetation, as well as uncolonized sand and mud. The "mangrove" stratum comprised the seaward edge of mangrove habitat able to be surveyed with these visual underwater survey methods.

This designated Guánica study area outlined above overlaps with the La Parguera study region that has been monitored since 2001 (Pittman et al., 2010). Data from the La Parguera region considered for this report includes data collected in both August of 2010 and in the previous two summer missions (August 2008-2009). Data from winter missions were not considered as several metrics exhibit seasonal variation (Pittman et al., 2010). For the purposes of this report, the "overlap" area will be considered part of the Guánica Bay study region. The La Parguera study area will refer only to the area west of the Guánica study area boundary (Figure 2.3). Data collected prior to 2008 was summarized in Pittman et al. (2010).

Field methods

The surveys of benthic habitats, fish communities, marine debris and macroinvertebrates were conducted within a 25x4 m transect (100 m^2), along a random heading. Two divers performed the survey at each site (Figure 2.4a-c). One diver was responsible for visual counts and size estimation of fish species. The second diver quantified benthic habitat composition, macroinvertebrates and marine debris. These established monitoring protocols have been used monitor La Parguera and other locations within NOAA's Caribbean Coral Reef Ecosystem Monitoring Project (Pittman et al., 2008; Pittman et al., 2010). The standardized protocols allow for comparisons to be made between different areas. In addition, the protocols include measurement of numerous variables that would be of interest to monitor over time to evaluate changes following the reduction in the input of land-based sources of pollution to the marine environment.

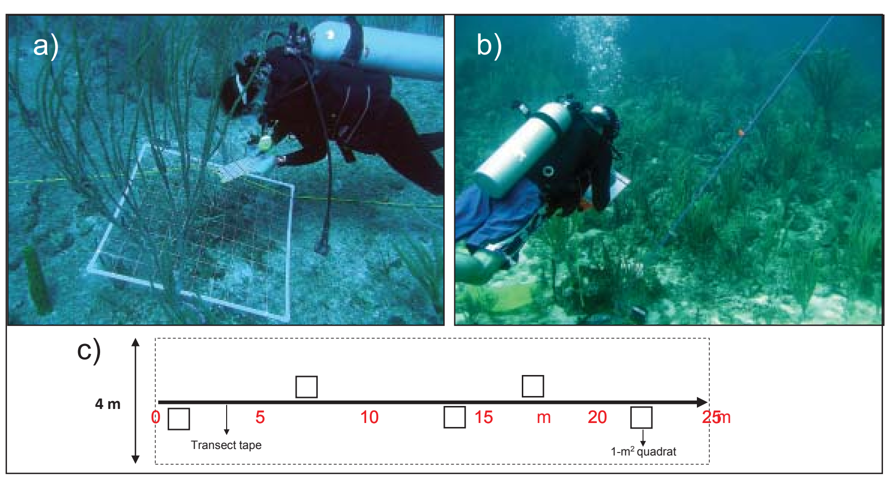

Figure 2.4a-c. Divers collecting data on a) habitat and b) fish composition, and c) schematic representation of the placement of the 1 m^2 quadrat along a 25 m transect tape during fish and benthic community surveys.

Benthic Habitat Composition

The habitat diver first assigned an overall bottom type (i.e., hardbottom, unconsolidated sediments or mangrove) to each transect based on *in situ* observation. Data on the percent cover of abiotic and biotic composition at each survey site were recorded within five 1 m^2 quadrats placed randomly along the 25x4 m transect so that one quadrat falls within every 5 m interval along the transect. The quadrat was placed at each randomly chosen meter mark and systematically alternated from side to side along the transect tape (Figure 2.4c). Several variables were measured to characterize benthic composition and structure (Table 2.1). The quadrat was divided into 100 smaller 10x10 cm squares with string (1 small square = 1% cover) to help the diver with estimation of percent cover. Percent cover was determined by looking at the quadrat from above and visually estimating percent cover in a two-dimensional plane. The information recorded included:

Table 2.1. Abiotic and biotic variables measured in five quadrats along fish transects.

Benthic Variables	Measurements		
	Cover (%)	Height (cm)	Abundance (#)
Abiotic			
Hardbottom	X	X	
Sand	X		
Rubble	X		
Fine sediment/silt	X		
Rugosity			
Water depth			
Biotic			
Corals (by species)	X		
Macroalgae	X	X	
Seagrass (by species)	X	X	
Gorgonians			
Sea rods, whips and plumes	X	X	X
Sea fans	X	X	X
Encrusting form	X		
Sponges			
Barrel, tubes, rope, vase	X	X	X
Encrusting form	X		
Other benthic macrofauna			
Anemonies and hydroids	X		X
Tunicates and zoanthids	X		
Mangroves			
Prop roots			X
Prop roots colonized by algae			X
Prop roots colonized by sponges			X
Prop roots colonized by other biota			X

<u>Abiotic cover</u> - the percent cover (to the nearest 1%) of four abiotic substrate categories (hardbottom, sand, rubble, fine sediments/silt) was estimated within each 1 m^2 quadrat. The maximum height of the hardbottom was also measured.

<u>Biotic cover</u> - the percent cover (to the nearest 0.1%) of algae, seagrass, live corals, sponges, gorgonians and other biota was estimated within each 1 m^2 quadrat. Taxa were identified to the following levels: stony coral-species, seagrass-species, algae-morphological group, sponge-morphological group and gorgonians-morphological group (Table 2.1). For stony and fire corals, the percentage of bleached coral and diseased/dead coral was estimated to the nearest 0.1%. In addition, the presence of elkhorn coral (*Acropora palmata)* or staghorn coral (*A. cervicornis)* either within the transect area (100 m^2) or the vicinity of the sample site was also noted by the divers.

<u>Maximum canopy height</u> - the maximum height of sponges, gorgonians and soft algal groups was recorded to the nearest 1 cm in each quadrat.

<u>Number of individuals</u> - the number of individual upright sponges, gorgonians, non-encrusting anemones and non-encrusting hydroids was recorded in each quadrat.

<u>Rugosity</u> – for hardbottom sites, rugosity was measured by placing a 6 m chain at two randomly selected positions, ensuring no overlap, along the 25 m belt transect. The chain was positioned along the centerline of the transect such that it followed the substrate's relief. The straight-line horizontal distance covered by the chain was measured. An index of rugosity (R) was calculated as the ratio of contoured surface distance (d) to linear distance (L = 6m) using $R=1-d/L$.

Mangrove Habitat Data

At mangrove sites, the survey was conducted along the edge of the mangrove canopy. The transect was laid out as close to the prop roots and as far into the mangroves as possible, up to 2 m, and then out to the edge of the mangrove overhang such that the total area surveyed was still 100 m^2. In this case, some of the survey sites fell on seagrass habitat. This is allowed as the mangrove habitat is defined as a transition zone habitat. Habitat metrics above were collected along with additional mangrove data, including: number of prop roots, number of prop roots colonized by algae, number of prop roots colonized by sponges and number of prop roots colonized by other biota (tunicates, anemones, zooanthids, etc.).

Macroinvertebrate Counts
The habitat diver counted the abundance of spiny lobsters (*Panulirus argus*), long-spined urchins (*Diadema antillarum*) and the abundance/maturity of queen conchs (*Strombus gigas*) within the 25x4 m transect at each site. The maturity of each conch was determined by the presence (mature) or absence (immature) of a flared lip.

Fish Census
Fish surveys were conducted along the 25x4 m transect (100 m^2) using a fixed survey duration (15 minutes) regardless of habitat type or complexity. The number of individuals per species was recorded in 5 cm size class increments up to 35 cm using visual estimation of fork length. If the individual could not be identified to species, they were identified to the extent possible (i.e., genus or family). Individuals greater than 35 cm were recorded as an estimate of the actual fork length to the nearest centimeter.

Marine Debris
The number and type of marine debris within the 100 m^2 transect were recorded. Marine debris size and the area of habitat it affected were estimated. Any flora or fauna that were colonizing the debris item were noted.

DATA ANALYSIS

Benthic Habitat
While many benthic variables were measured during the surveys, data analyses for this report focused primarily on describing differences among major habitat types and broad-scale spatial patterns in the percent cover of the sessile biotic components as described in Table 2.1. Quadrat measurements along each transect were averaged and cumulative coral species richness was calculated for each site. Average site values were used to calculate means and standard errors of measured variables for each habitat type (coral reef and hardbottom, unconsolidated sediment, mangrove) and study area (La Parguera, Guánica) combination. In addition, data were plotted in ArcGIS (v9.3, ESRI) to examine broad spatial patterns in the benthic cover variables. For the above analyses and summary statistics, all 2008-2010 data from within the La Parguera study area were used to provide a detailed site characterization.

Select habitat metrics were tested to identify potential differences between the Guánica and La Parguera study areas. For these tests, a random subset of sites within the La Parguera study area was selected so that the sample size for each bottom type was equal to that of the Guánica study area. Prior to this analysis, we tested the assumption that benthic cover metrics did not differ significantly over the three years of the La Parguera dataset. As the majority of metrics did not vary significantly over this time period, we determined that using a subset of data for statistical tests would not introduce significant bias into the analysis. As the assumption of normality was not met for most variables, non-parametric Wilcoxon-Mann-Whitney tests were used to test differences between study areas (SAS v9.1.3.) Tests were conducted separately for individual strata, as hardbottom, mangrove and unconsolidated sediments generally have distinct benthic communities.

Further analysis was used to explore possible connections between coral variables and the distance from the mouth of Guánica Bay. The in-water distance between each survey location and the bay mouth was calculated using the cost distance raster tool in ArcGIS. Because assumptions of normality could not be met a Spearman's nonparametric correlation test was used to test for associations between distance and percent live coral cover, coral species richness and percent gorgonian cover.

Fish Assemblages
A summary table of all species observed in this characterization across the entire sampling domain was created (see Appendix A, Table A.1). Habitat-wide estimates for community metrics (total density, total biomasss, species richness, Shannon diversity, and density and biomass of trophic groups) were computed for the Guánica and La Parguera study areas employing methods described by Cochran (1977)[1]. Percent occurrence, mean density and biomass (per 100 m^2) and corresponding standard errors (SE) were calculated for each species. Trophic groups include piscivores, herbivores, invertivores and zooplanktivores and were defined for each species based on diet information from Randall (1967). It is important to note that these groups are not mutually exclusive because many fish species can be classified into two or more of these groups based on diet. In those circumstances the trophic group was assigned based on the dominant diet component. Biomass was calculated using published length-weight relationships based on the formula:

$$W = aLb,$$

where L is length in centimeters and W is weight in grams. The midpoint of each size class was used for L values, and the actual length was used for fish >35 cm. For fish in the 0-5 cm size class, 3 cm was used as the mid-point because we do not typically observe fish <1 cm. Values for the a and b coefficients were obtained from FishBase (Froese and Pauly, 2008). Biomass for species with no published length-weight relationships was calculated using terms for the closest congener with most similar morphology.

[1]Small baitfish (i.e., Families Atherinidae, Clupeidae) were omitted from the analysis when calculating overall mean density and biomass and in testing differences between total density and biomass as occasional large schools of small baitfish can highly skew density and biomass estimates in mangrove habitat.

Species diversity was calculated using the Shannon Index (H'), a measure that incorporates both richness and evenness:

$$H' = \Sigma i p_i (\log e p_i)$$

where p_i is the relative abundance of each species.

Data were plotted in ArcGIS to examine broad spatial patterns in the fish metrics. In addition, select families and species of commercial and/or ecological interest were selected for further examination. For each species/family, a summary of the species distribution, size frequency, and mean density and biomass by habitat type and study area was calculated. Age class (juveniles/sub-adults and adult) was identified based on mean length at maturity as identified by FishBase (Froese and Pauly, 2008), García-Cagide et al. (1994) and Ault et al. (2008). Where length at maturity was unknown, one-third of maximum size was used as a proxy as in Pittman et al. (2008, 2010). For the above analyses and summary statistics, all 2008-2010 data from within the La Parguera study area were used to provide a detailed site characterization.

Community metrics and the select families/species were compared between the Guánica and La Parguera study area by habitat type to characterize any potential differences between the two regions[1]. As in the benthic habitat analysis, we ran a preliminary analysis to test whether fish species metrics varied significantly by years within the La Parguera dataset. For the majority of species, no significant differences among years were detected. Consequently, the previously selected random subset of sites within the La Parguera study area was used so that the sample size for each bottom type was equal to that of the Guánica study area. As the assumption of normality was not met for most variables, non-parametric Wilcoxon-Mann-Whitney tests were used to test differences between study areas (SAS v9.1.3).

Differences and similarities in species composition were further examined using multivariate statistical techniques (Primer v.6; Clarke and Warwick, 2001). Data were square-root transformed prior to analysis, and one outlier site was removed due to extremely low fish abundance (n=1 fish). Data were arranged in a species abundance by site data matrix, which was used to construct a triangular matrix of the percentage similarity in community composition between all pairs of sites using the Bray-Curtis Coefficient. The coefficient is a measure of how similar samples are to each other, ranging from 0% (complete dissimilarity) to 100% (complete similarity). Next, non-metric multidimensional scaling (nMDS) was used to place samples in a two-dimensional configuration such that the rank order of the distances between the samples agreed with the rank-order of the similarities from the Bray-Curtis matrix. Sites were coded by bottom type and geographic study area (La Parguera, Guánica) for examination of visual patterns of similarities between sites. These factors were also used to test for significant differences in similarity using Analysis of Similarities (ANOSIM), a multivariate, non-parametric version of ANOVA.

RESULTS AND DISCUSSION

Benthic Habitat

Abiotic Composition

Within the Guánica study area, a total of 61 sites were surveyed: 30 on hardbottom, 22 on unconsolidated sediments and nine in mangrove (Figure 2.5). Three sites identified as hardbottom based on habitat maps (Kendall et al., 2001) were designated as unconsolidated sediment by the survey diver and were subsequently grouped with the unconsolidated sediment surveys in the data analysis.

A total of 188 sites were surveyed during the summer seasons of 2008-2010 in the La Parguera study area, and were comprised of the following: 97 hardbottom, 67 unconsolidated sediments and 24 mangrove (Figure 2.5). Nine sites originally stratified as hardbottom were designated as unconsolidated sediment by the diver, and were subsequently grouped with the unconsolidated sediment surveys in the data analysis. Conversely, one site stratified as an unconsolidated sediment was designated as hardbottom by the diver and was subsequently grouped with the hardbottom surveys in the analysis.

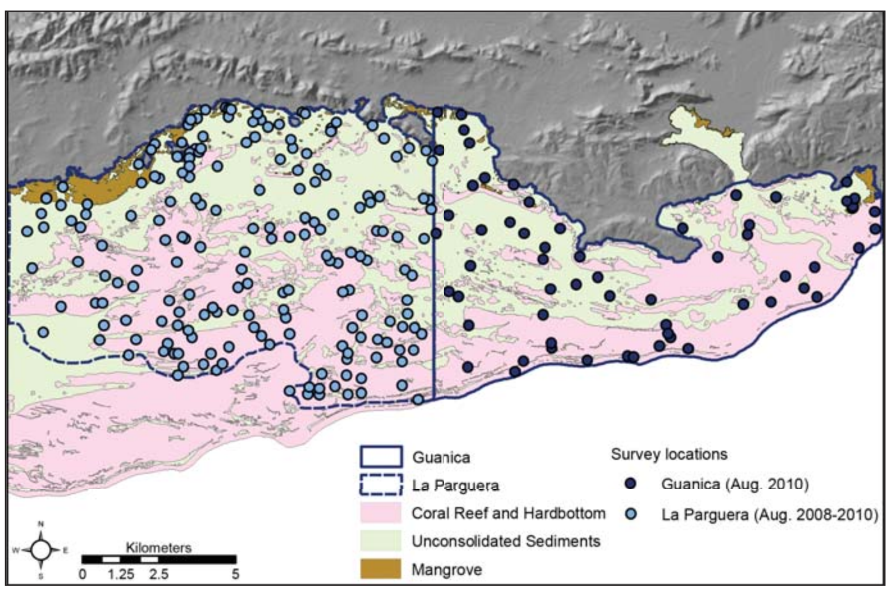

Figure 2.5. Site locations of surveys of benthic habitat composition, fish communities and marine debris. Data include the August 2010 survey in the Guánica study area (N=61) and August 2008-2010 surveys in the La Parguera study area (N=188).

Abiotic composition varied across habitat types but were similar between the two study areas of similar habitat types. (Figure 2.6a and b). Surveys conducted on hardbottom habitat in both the Guánica and La Parguera study areas were predominantly composed of hard substrate, or hardbottom, with small amounts of rubble and sand present, and were devoid of fine sediments. The majority of surveys conducted on unconsolidated sediment habitat were dominated by sand and fine sediment. Hardbottom or rubble substrates were recorded during a few unconsolidated sediment surveys in the Guánica (n=3 of 22) and and La Parguera study areas (n=6 of 67). Most mangrove sites were characterized by sand and fine sediments, with occasional small amounts of rubble.

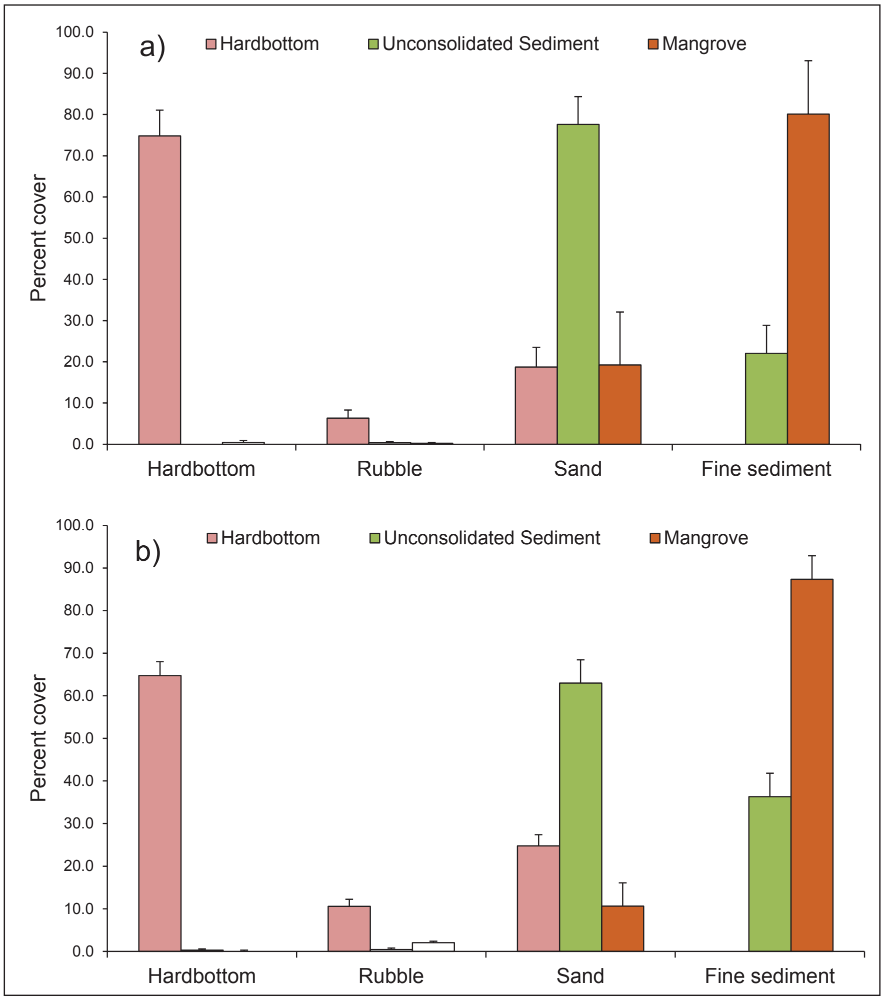

Figure 2.6. Mean (±SE) percent cover of abiotic substrate across hardbottom, unconsolidated sediment, and mangrove surveys in the a) Guánica and b) La Parguera study areas.

Biotic Composition Overview
Here broad information on the habitat groups in both study areas are examined. In subsequent sections, variability of species composition, cover, and diversity within and among habitat groups are covered in more detail. Among hardbottom sites (Figure 2.7a), algae accounted for the highest mean percent cover in the Guánica study area (mean ± SE: 65.3 ± 4.2%), followed by soft corals (3.6 ± 1.0%), sponges (3.4 ± 0.5%) and hard coral (2.2 ± 0.4%). Bare, uncolonized substrate averaged 25.0 ± 4.8%. Algae also accounted for the highest mean percent cover in the La Parguera study area (61.1 ± 2.2%), followed by soft corals (6.0 ± 0.8%), hard corals (4.0 ± 0.4%) and low amounts of sponges (2.0 ± 0.2%). Similar to the Guánica study area, bare, uncolonized substrate averaged 26.0± 2.4%.

Unconsolidated sediment sites (Figure 2.7b) were characterized by an assemblage of submerged aquatic vegetation and low levels of benthic fauna. Seagrass accounted for the highest mean percent cover among the unconsolidated sediment sites in the Guánica study area (8.2 ± 3.2%), followed by algae (5.2 ± 1.2%). All remaining benthic groups surveyed were present in low amounts (<1%). Bare substrate averaged 86.4 (±3.4%). Within the La Parguera study area unconsolidated sediment surveys, seagrass (17.4 ± 2.5%) and algae (12.4 ± 2.1%) also dominated the benthos, with the remaining categories in low amounts as seen in the Guánica study area. Bare substrate averaged 69.4 (± 3.0%).

Algae was the dominant benthic cover on substrates in mangrove habitat within the Guánica study area (Figure 2.7c; 57.4 ± 10.9%) with variable amounts of seagrass present (3.6 ± 2.0%). Bare substrate averaged 38.5 (± 10.9%). Similarly, in mangrove surveys within the La Parguera study area, algae was the dominant cover type (63.0 ± 5.9%), followed by low amounts of seagrass (0.8 ± 0.5%). Bare substrate averaged 35.8 (± 5.7%).

Hard Coral Composition
Percent scleractinian coral in the Guánica study area ranged from 0-10.76% (Figure 2.8). The site with the highest coral cover was located nearshore in 5 meters of water, while sites with moderate cover (3.1-6%) were located almost exclusively in the deeper waters of the outer shelf. The coral community in the Guánica study area was comprised of 21 species, with species richness varying from 0-10 species/site (Figure 2.9). Located on the outer reef edge, the site with the highest coral species richness (10 species/site) had low-moderate coral cover. Within hardbottom surveys, massive starlet coral (*Siderastrea siderea*) represented the species of highest coral cover (0.5 ± 0.1%) followed by mustard hill coral 0.4 ± 0.1%, Figure 2.10a and Figure 2.11). All other species observed averaged <0.3% cover (Figure 2.10a). Bleached coral was present in small amounts (≤0.1%) in one-fifth (six out of 30) of the surveys in hardbottom habitat. Reef rugosity measured on hardbottom sites ranged from 0-0.42, averaging 0.05 (± 0.01). Sites with the highest values were generally located on offshore reefs near the shelf edge and on sloping fore reefs seaward of the lagoon (Figure 2.12).

Within the La Parguera study area, percent scleractinian coral ranged from 0-18.5% (Figure 2.8). Coral cover generally increased with distance from shore. The site with the highest coral cover (18.5%) was located on the outer shelf in 22 meters of water. Over the three years considered in this analysis (2008-2010), 37 species were recorded within the La Parguera study area, with species richness ranging from 0-12 species/site (Figure 2.9). The boulder star coral (*Montastrea annularis)* complex was the dominant scleractinian species in the La Parguera study area (1.4 ± 0.2%), while all remaining species observed averaged <0.5% cover (Figure 2.10b). Little incidence of bleached coral was reported (mean ± SE = 0.01 ± 0.0%). Reef rugosity averaged 0.20 (± 0.02) and ranged from 0-0.47, with higher values located on mid-outer shelf reefs (Figure 2.12).

On hardbottom habitats, percent cover of scleractinian coral and coral species richness were significantly greater in the La Parguera study area compared to the Guánica study area (Table 2.2). Further, percent live coral cover was significantly correlated with distance from the mouth of Guánica Bay (Spearman's Rho=0.19, p=0.036). Coral species richness also showed a positive correlation with increasing distance from the bay, but the statistical significance for this metric was marginal (Speaman's Rho=0.16, p=0.068). Reef rugosity did not vary significantly between the two study areas.

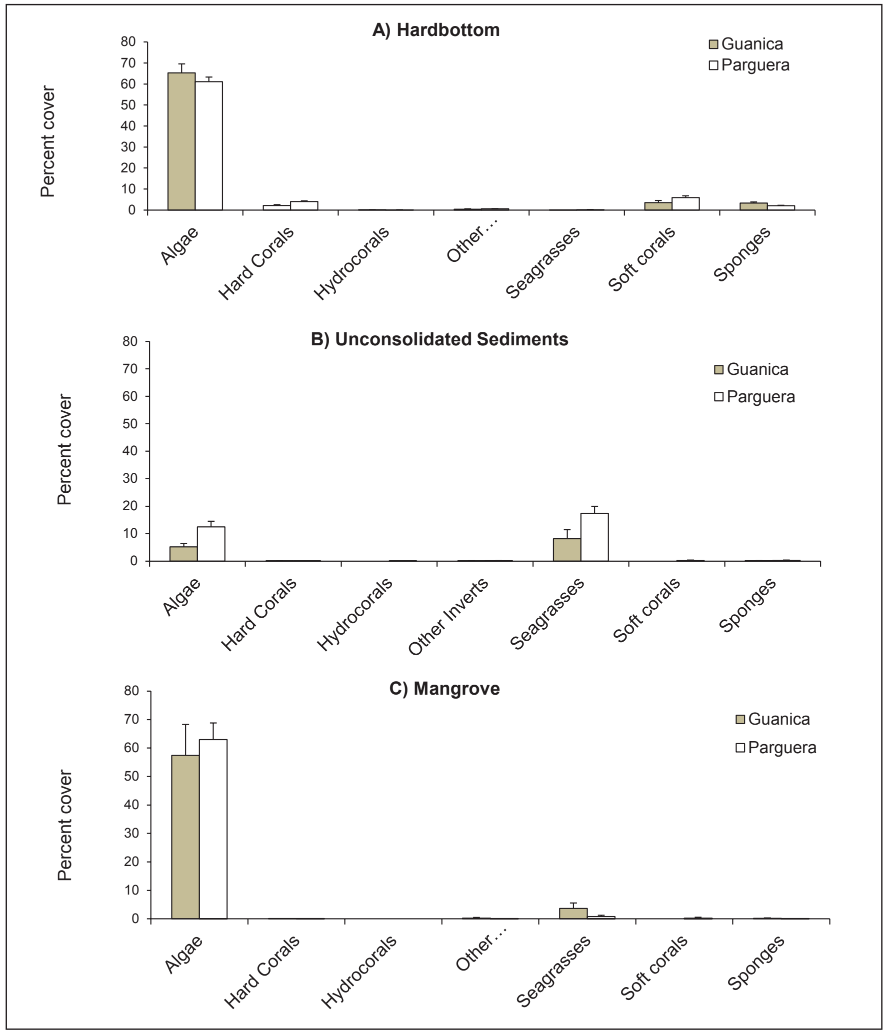

Figure 2.7. Mean (±SE) percent cover of major cover groups in a) hardbottom habitat, b) unconsolidated sediment habitat and c) mangrove habitats in Guánica and La Parguera study areas.

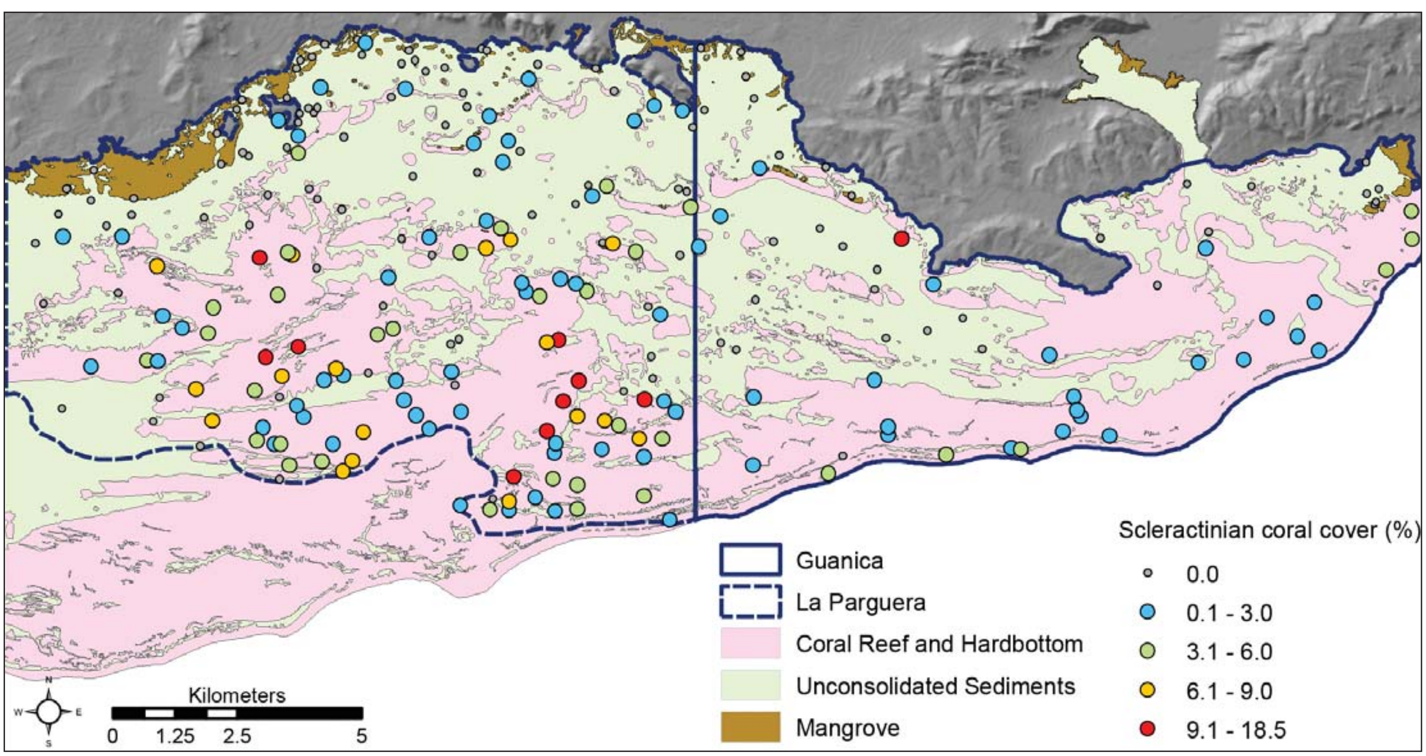

Figure 2.8. Percent cover of scleractinian corals.

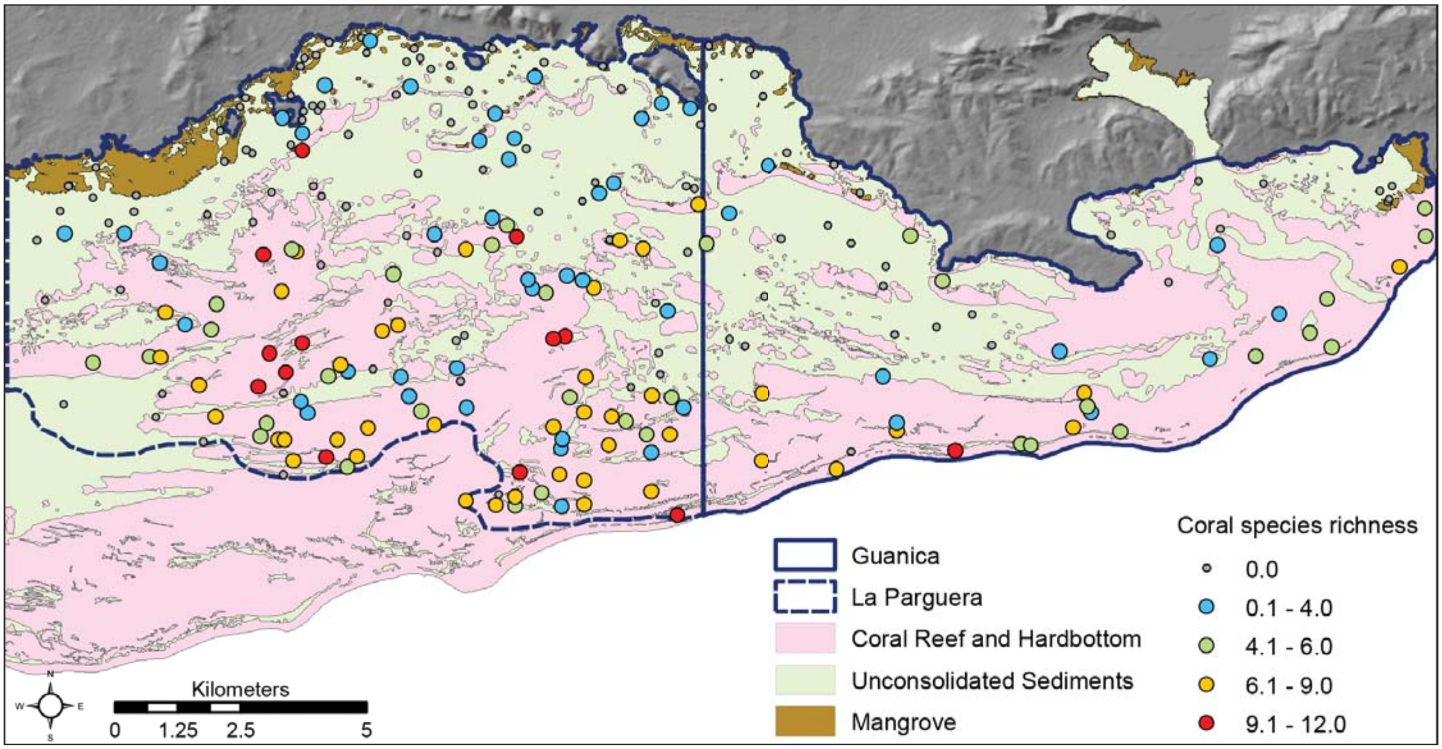

Figure 2.9. Coral species richness.

Staghorn coral and elkhorn coral are each listed as threatened coral species under the Endangered Species Act (USFWS, 1973). Mean cover of both species was estimated at <1% within both study areas. In the Guánica study area, elkhorn coral was present in the quadrat survey at one site and was not reported from any additional sites, while staghorn coral was present in two quadrat surveys and noted within the vicinity at two additional sites. In the La Parguera study area, only staghorn coral was observed. This species was recorded within quadrats at four surveyed sites and was observed anecdotally in the surrounding area of an additional two survey sites. It is important to note that this study's methods were not designed specifically to detect *Acropora* species.

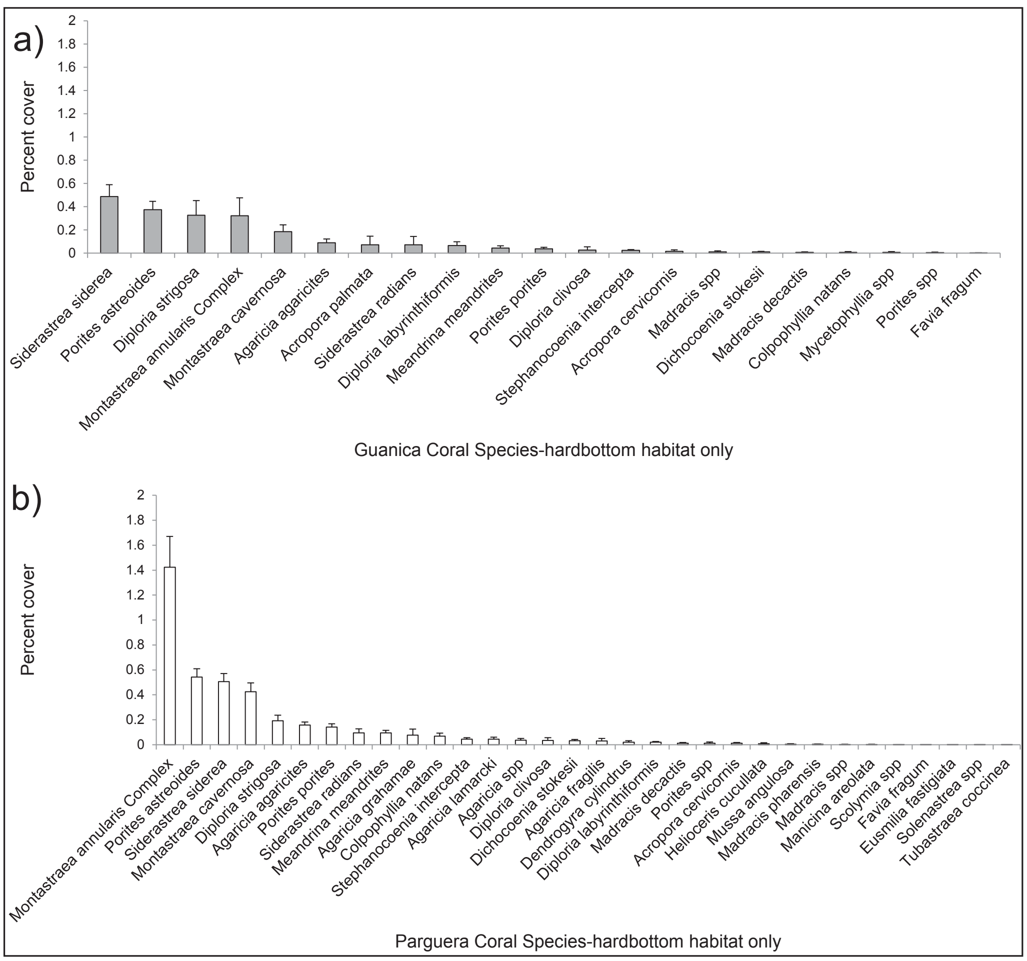

Figure 2.10. Mean (±SE) percent cover of scleractinian coral species in the a) Guánica study area and b) La Parguera study area.

Hydrocorals (*Millepora sp.*) and other invertebrates (see Table 2.1) were present in low amounts in both study areas (<0.3%). Zoanthid cover was primarily limited to hardbottom sites in both study areas with the highest mean cover observed on the mid-shelf reefs of the La Parguera study area and on a nearshore forereef site near Playa Santa within the Guánica study area (Figure 2.13). Zooanthid cover averaged 0.4 (± 0.02)% on hardbottom within the Guánica study area and 0.5 (± 0.01)% in the La Parguera study area.

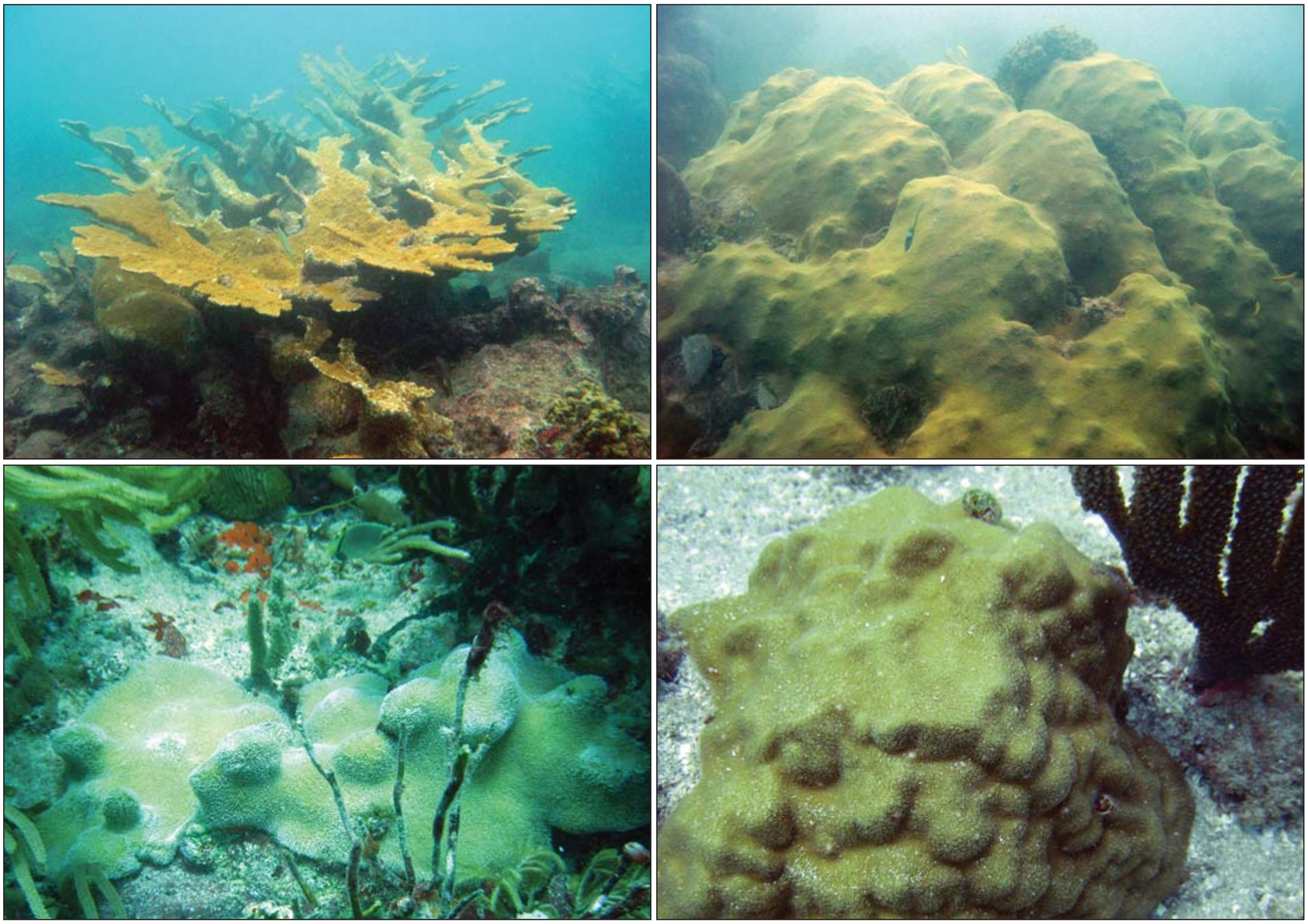

Figure 2.11. Coral species observed in the surveys in southwest Puerto Rico. From left to right: elkhorn coral, boulder star coral complex, pillar coral and mustard hill coral.

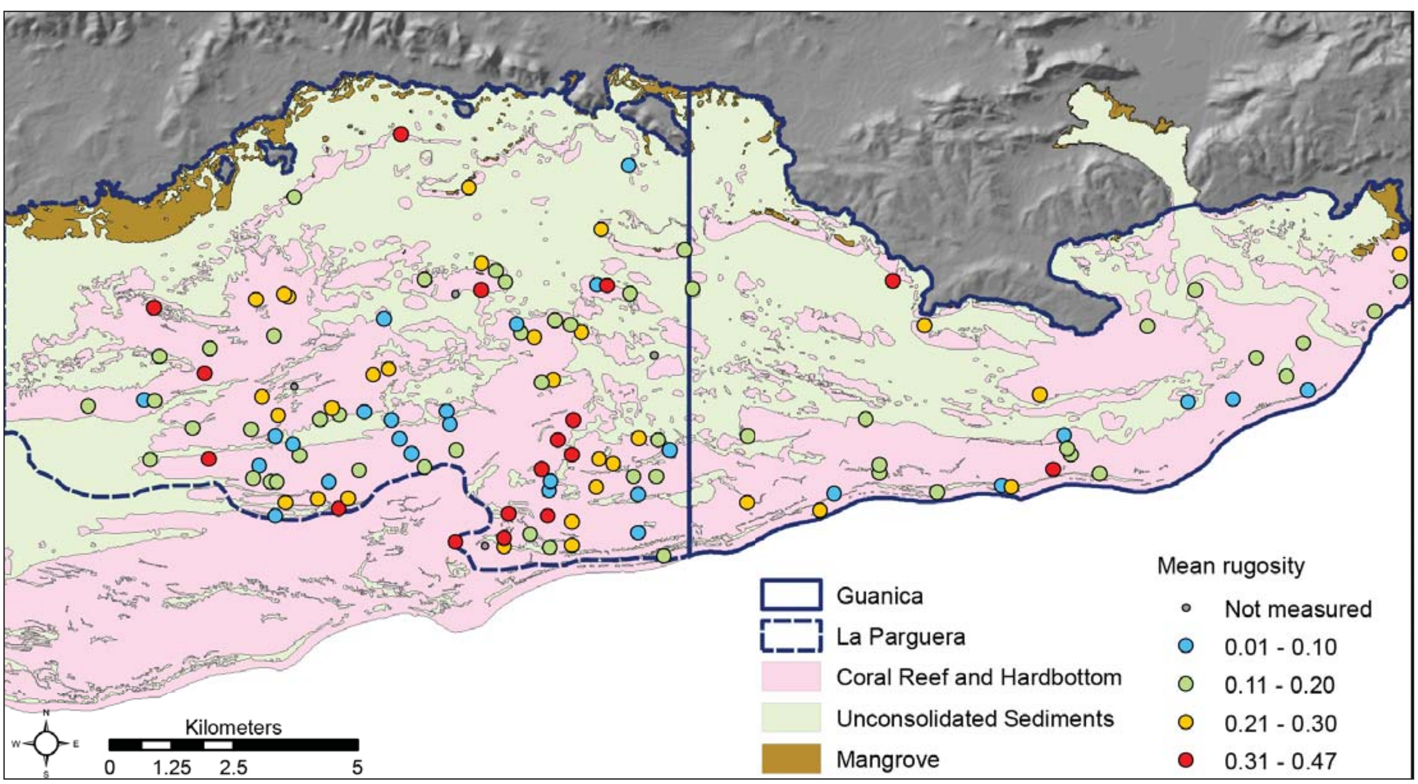

Figure 2.12. Mean rugosity (hardbottom sites only).

Table 2.2. Results of nonparametric analysis (Wilcoxon-Mann Whitney) for benthic habitat metrics between the Guánica and La Parguera study area. As described in the methods, a random subset of the La Parguera (2008-2010) dataset was used. An asterisk (*) Indicates a significant difference at the α=0.05 level.

Hardbottom		
Metric	Z	p-value
% Live coral	2.37	0.018*
Coral species richness	1.99	0.046*
% Gorgonians	2.52	0.011*
% Sponge	-2.59	0.010*
% Macroalgae	-0.82	0.412
% Turf algae	-2.17	0.030*
% Crustose/coralline algae (CCA)	-0.90	0.370
% Filamentous algae/Cyanobacteria	3.82	<0.001*
% Zooanthids	-0.51	0.611
Rugosity	0.68	0.499
Unconsolidated Sediment		
Metric	Z	p-value
% Sponge	2.45	0.014*
% Seagrass	2.09	0.037*
% Macroalgae	1.44	0.149
% Filamentous algae/Cyanobacteria	-1.14	0.253
Mangrove		
Metric	Z	p-value
% Seagrass	-0.90	0.366
% Macroalgae	0.27	0.791
% Filamentous algae/Cyanobacteria	0.18	0.860

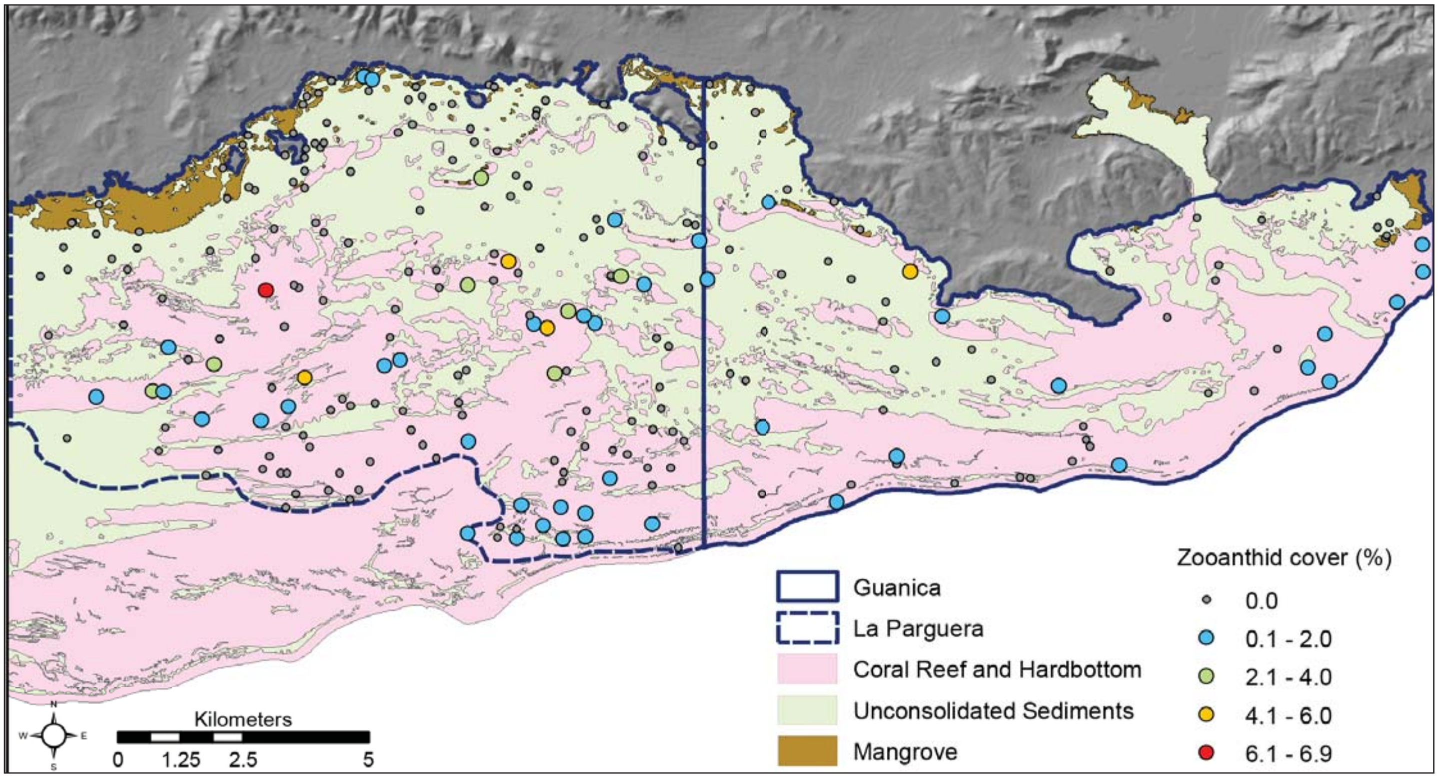

Figure 2.13. Percent cover of zooanthids.

Gorgonian Composition

Gorgonian cover ranged from 0-24.6% in the Guánica study area (Figure 2.14). Sea plumes/rods/whips were the dominant gorgonian morphological cover across hardbottom surveys (2.6 ± 0.6%), followed by sea fans (0.6 ± 0.3%), and encrusting gorgonians (0.3 ± 0.2%). Highest cover was observed both at a nearshore location on fore reef, as well as offshore near the shelf edge.

Gorgonian cover ranged from 0-57.8% in the La Parguera study area (Figure 2.14). Sites on aggregate reef adjacent to the cays within the mid shelf region of the La Parguera study area exhibited the highest gorgonian cover. Sea plumes/rods/whips were the dominant gorgonian morphological cover on hardbottom sites (4.3 ± 0.7%), followed by encrusting gorgonians (1.0 ± 0.2%) and sea fan (0.7 ± 0.2%).

Gorgonians were rarely observed on soft bottom or mangrove habitats so further investigation was conducted only with hardbottom sites. Results of the Wilcoxon-Mann-Whitney test indicate that total gorgonian cover was significantly greater on hardbottom in the La Parguera study area (Table 2.2). Further, gorgonian cover was positively correlated with distance from the mouth of Guánica Bay (Spearman's Rho=0.21, p=0.017).

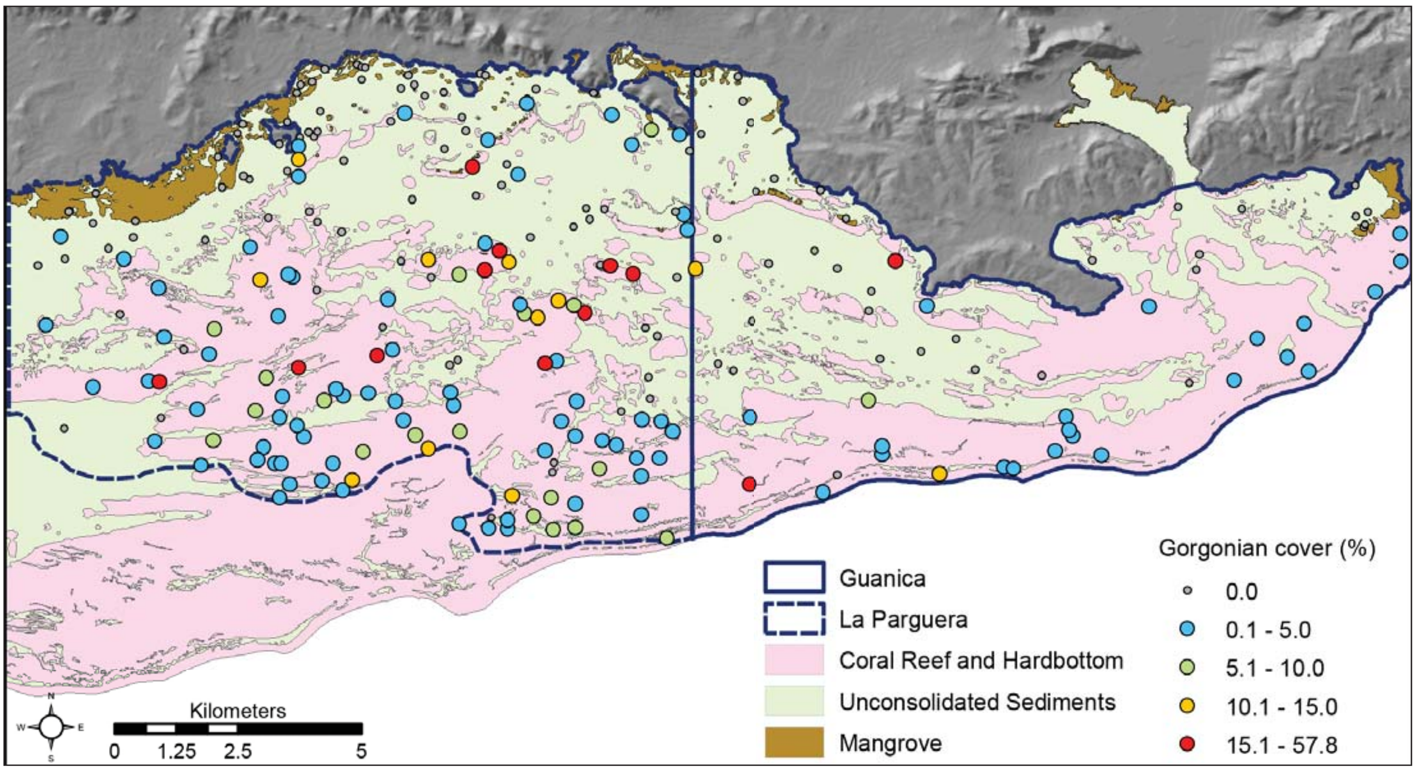

Figure 2.14. Percent cover of gorgonians.

Sponge Composition

Sponge cover ranged from 0-13.3% in the Guánica study area, with the highest values on offshore hardbottom sites (Figure 2.15). Barrel/tube/vase sponge morphology accounted for the majority of sponge cover on hardbottom sites (2.3 ± 0.4%), followed by encrusting sponge (1.0 ± 0.2%). On unconsolidated sediment sites, each sponge morphology was present in very low amounts (<1%).

Sponge cover ranged from 0-11.6% in the La Parguera study area and did not exhibit distinctive spatial patterns (Figure 2.15). Sites with the highest sponge cover were situated in both inshore and offshore locations. Barrel/tube/vase sponge morphology dominated on hardbottom (1.1 ± 0.01%), followed by encrusting sponges (0.9 ± 0.1%).

On hardbottom, percent cover of sponges was significantly greater in surveys within the Guánica study area compared to La Parguera (Table 2.2). On unconsolidated sediments, the pattern was reversed, with higher cover within the La Parguera study area.

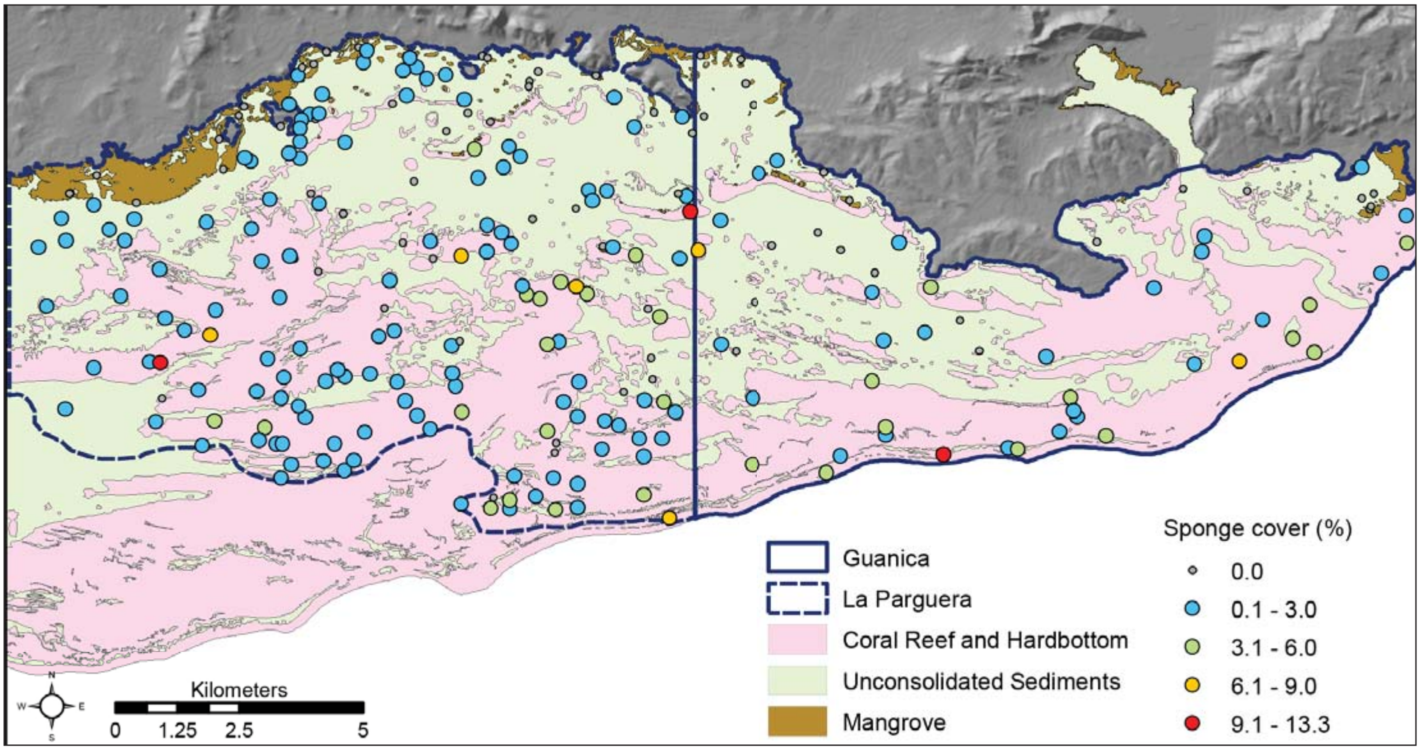

Figure 2.15. Percent cover of sponges.

Algae and Seagrass Composition

Algae was present in virtually all survey transects in the Guánica study area in various morphological forms. Macroalgal cover ranged from 0-78.0% and was present across all habitats (Figure 2.16). The site with the highest macroalgal cover in the Guánica study area was a mangrove site east of Bahia Montalvo. Mean macroalgal cover was higher, but more variable, in mangrove (21.8 ± 10.6%) than hardbottom habitats (18.7 ± 3.1%), and overall low across unconsolidated sediment surveys (3.3 ± 0.8%). Turf algae was a ubiquitous component of the reef community, ranging from 0-89.2% and averaging 43.3 (± 4.2)% across hardbottom habitat (Figure 2.17). Crustose coralline algae (CCA) was typically present in small amounts (ranging from 0-10.8%) and was primarily observed at hardbottom sites further offshore in the outer shelf region (Figure 2.18). Filamentous algae/cyanobacteria generally exhibited highest cover in mangroves (Figure 2.19), averaging 35.3 (±11.7)% across mangrove surveys but only 1.4 (±0.3)% and 1.4 (±0.9)% across hardbottom and unconsolidated sediment surveys, respectively.

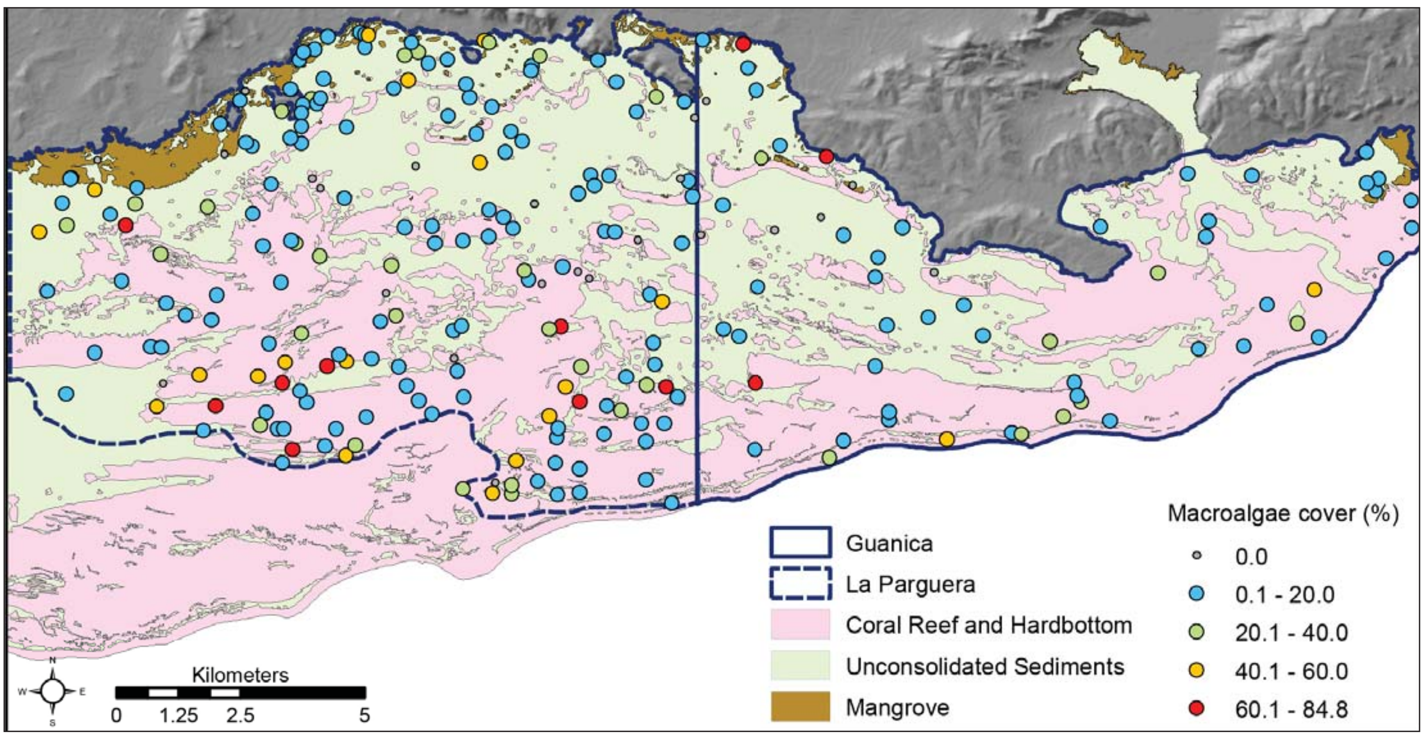

Figure 2.16. Percent cover of macroalgae.

Algal cover patterns were similar in the La Parguera study area. Macroalgae was distributed across all areas, with percent cover ranging from 0-84.8% (Figure 2.16). The site with the highest macroalgal cover was a mangrove site west of the La Parguera waterfront. Macroalgal cover averaged 20.1 (± 2.2)% across hardbottom surveys, with more moderate amounts in mangrove and unconsolidated sediment habitats (10.5 ± 3.0% and 9.4 ± 1.9%, respectively). Turf algae ranged from 0-80.8% and averaged 31.7 (± 2.5%) across hardbottom surveys (Figure 2.17). CCA was primarily restricted to hardbottom habitat (Figure 2.18), where it ranged from 0-9.4% and averaged 1.5 (±0.2)%. Cover values in the higher end of the range were generally located on the mid-outer shelf. Filamentous algae/cyanobacteria ranged from 0-99.4% and were most common in mangrove surveys, but there were occasionally moderate amounts on hardbottom (Figure 2.19). Mean cover was 51.1 (±7.0)% in mangrove, 7.8 (±1.3)% on hardbottom and 0.9 (±0.4)% on unconsolidated sediments.

There were two notable differences in algal cover between the two study areas. On hardbottom habitat, turf algae was significantly higher in the Guánica study area while filamentous/cyanobacteria was significantly higher in the La Parguera study area (Table 2.2).

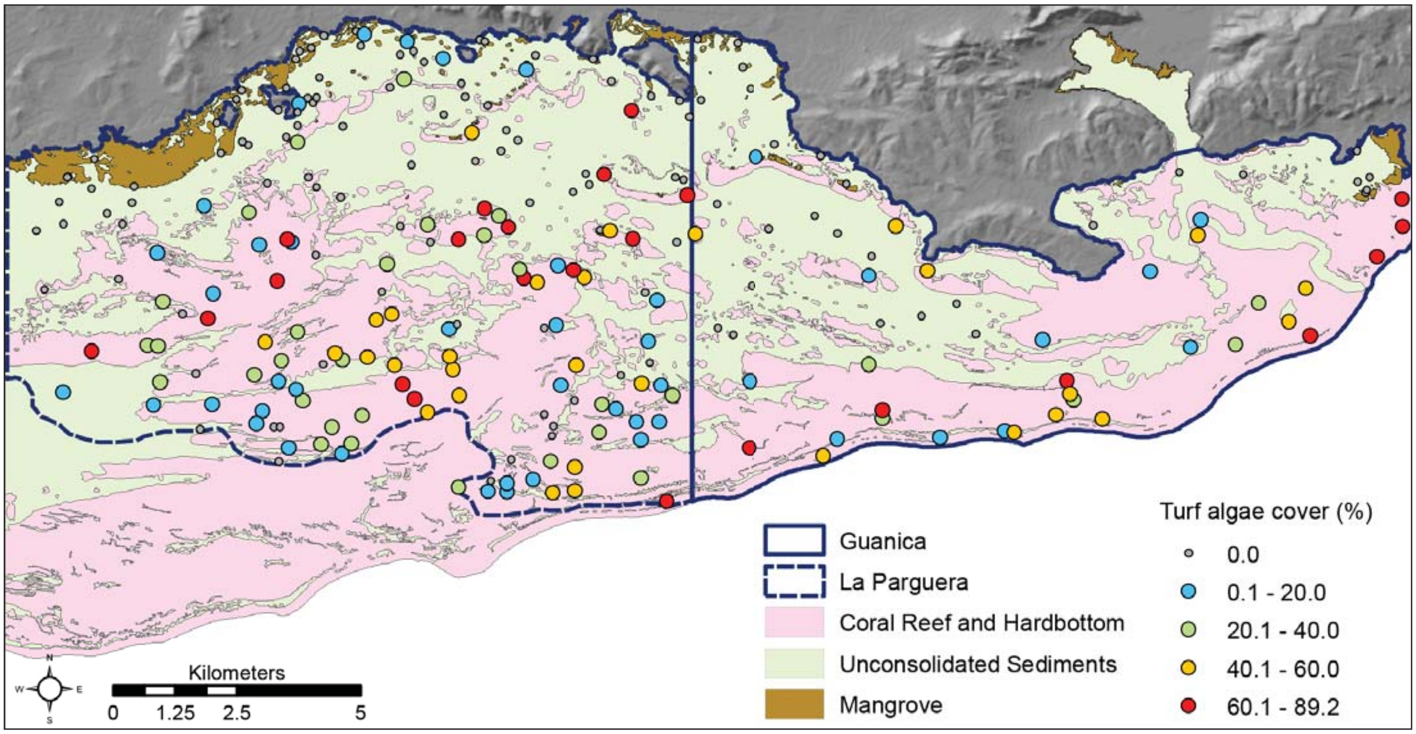

Figure 2.17. Percent cover of turf algae.

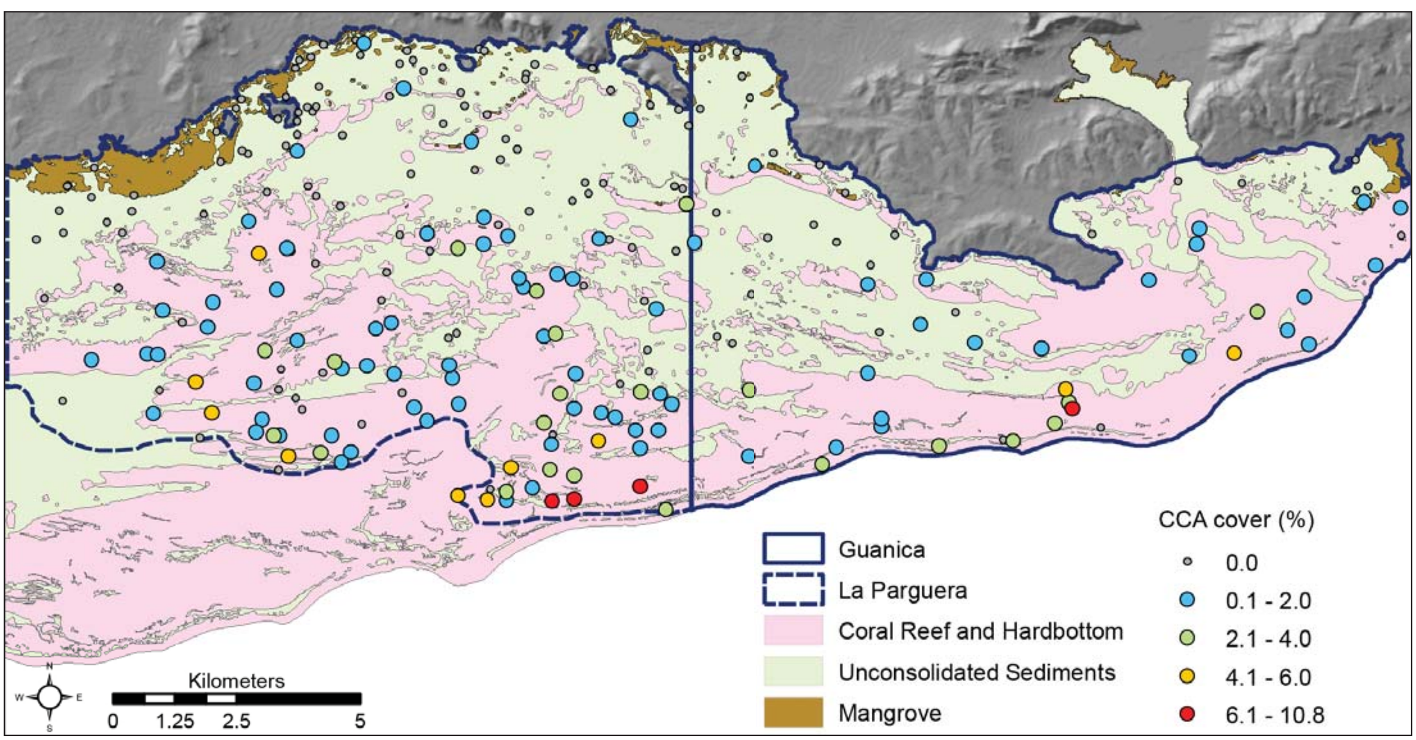

Figure 2.18. Percent cover of crustose coralline algae (CCA).

Seagrass cover in the Guánica study area ranged from 0-57.6% and was highest in the inshore waters on unconsolidated sediment sites, particularly east of Guánica Bay and north of Cayos de Caña Gorda (Figure 2.20). Turtle grass *(Thalassia testudinum)* was the dominant species of seagrass on unconsolidated sediment sites in the Guánica study area (5.7 ± 3.1%), followed by paddle grass *(Halophila decipiens)* (1.8 ± 0.7%), shoal grass *(Haldoule wrightii)* (1.8 ± 0.1%) and manatee grass *(Syringodium filiforme)* (0.5 ± 0.3%; Figure 2.21 and Figure 2.22). Seagrass amounts were lower in mangrove habitat, where total seagrass cover averaged 3.6 (± 2.0)% and was present primarily as turtle grass (Figure 2.7c).

In the La Parguera study area, seagrass cover ranged from 0-71.8% and was also highest in nearshore and lagoonal habitats Figure 2.20). Turtle grass was the dominant species of seagrass on unconsolidated sediment sites, averaging 12.6 (± 2.5%; Figure 2.21), followed by *H. decipens* (4.8 ± 1.3%). *Turtle grass* was the primary species in shallow, nearshore waters, while *H. decipens* exhibited higher cover in the deeper waters seaward of the mangrove cayes. Both shoal grass and manatee grass were less common and generally present in small amounts (<1% cover).

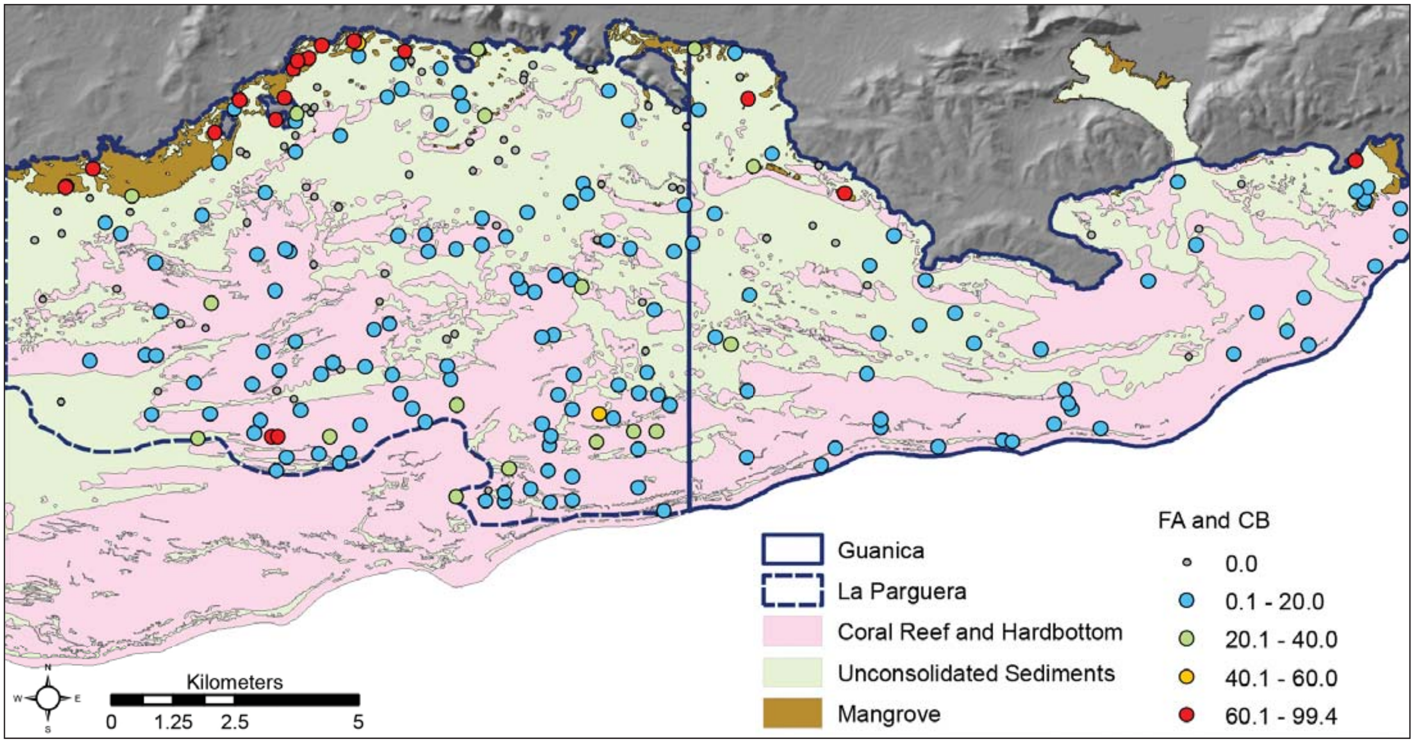

Figure 2.19. Percent cover of filamentous algae (FA) and cyanobacteria (CB).

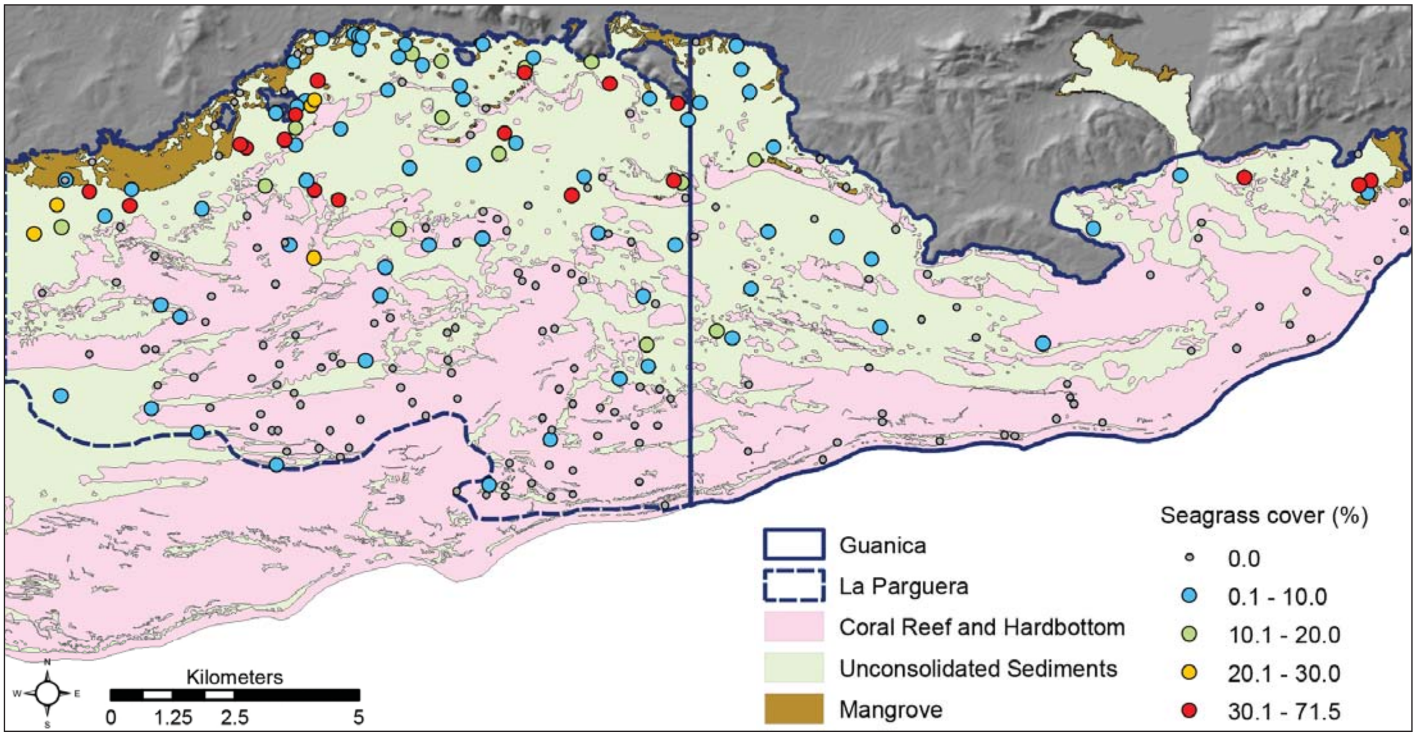

Figure 2.20. Percent cover of seagrass.

There was a significantly greater amount of seagrass on the unconsolidated sediment sites of the La Parguera study area compared to the Guánica study area (Table 2.2). Some of the differences may be attributed to the inherent geomorphological differences between the two study areas. The La Parguera study area contains a much larger protected lagoonal area, spanning almost the entire width of the study area. In the Guánica study area, protected lagoons are located at the western end of the study area (contiguous form the La Parguera study area) and to the east of Guánica Bay, where the highest seagrass cover values were observed in this study. The area in between these two lagoonal features is less protected and much more exposed to waves and currents. Immediately south and west of Guánica Bay, there is an expanse of sand/mud that has patchy seagrass or has been mapped as having no appreciable cover (Bauer et al., 2012). Whether this pattern has been influenced by exposure to sediment runoff from the bay, dredging of the shipping channel or other anthropogenic influences is unknown.

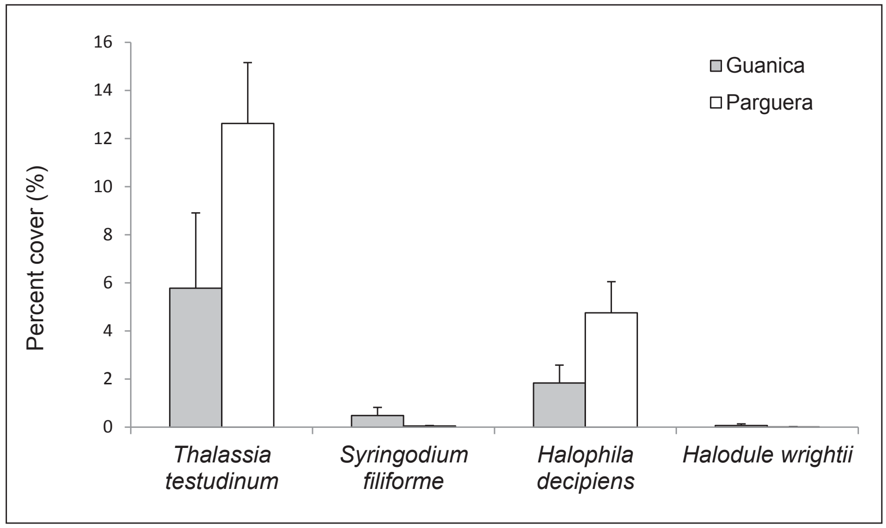

Figure 2.21. Mean percent cover of each seagrass species in unconsolidated sediments sites in the Guánica and La Parguera study areas.

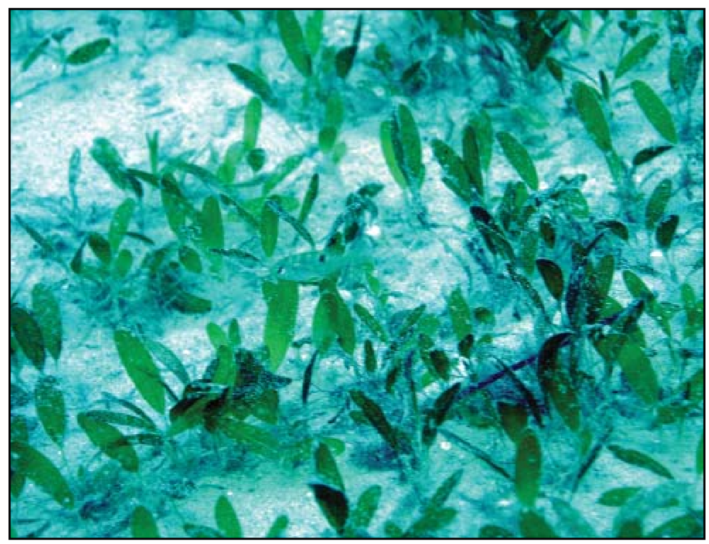

Figure 2.22. Seagrass species documented in Guánica and La Parguera surveys.

Macroinvertebrates

A total of 24 Queen conch (*Eustrombus gigas*) were reported between the two study areas (Figure 2.23), including 18 individuals from the Guánica study area, three of which were mature, and six individuals from the La Parguera study area, two of which were mature. A total of seven lobsters (*Panularis argus*) were reported between the two study areas, only one of which was observed within the Guánica study area. This individual was observed in a survey transect close to the shelf edge (Figure 2.24). Of the remaining six lobsters seen in the La Parguera study area, five were observed at a single mangrove site. All long-spined urchins (*Diadema antillarium*) were observed within the La Parguera study area; no long-spined urchins were observed in the Guánica study area surveys (Figure 2.25).

Marine Debris

A total of 94 marine debris items were found among the field surveys of the Guánica and La Parguera study areas (Table 2.3). The absolute number of debris items within a transect area (100 m^2) ranged from 0-11. Debris was present at a total of 29 sample sites, five in the Guánica study area (8% of sites) and 24 in the La Parguera study area (13% of sites). Derelict fishing gear accounted for 11% of the reported debris, half of which was attributed to derelict traps or trap material. The majority of the debris items were non-fishing related debris (88%), including cans, bottles and tires. Other non-fishing related items included items such as: plastic bags, a plastic tricycle and a visor. The majority of the marine debris items were found along the shoreline and in the mangrove cays of La Parguera (Figure 2.26). The few debris items seen in Guánica were confined to the shoreline and nearshore cays. Only a select few items were found on the outer shelf of La Parguera, and included a relic fish pot, derelict trap material and two glass bottles (Table 2.3).

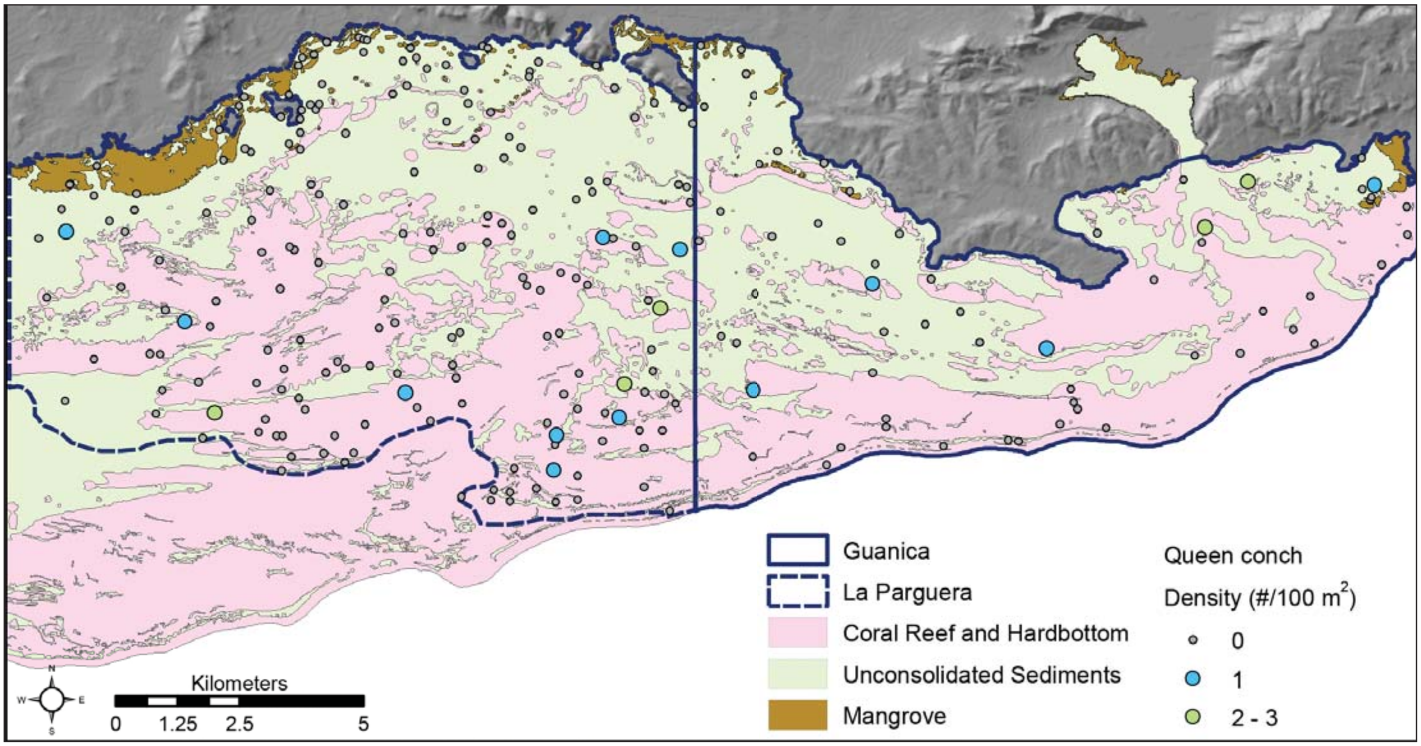

*Figure 2.23. Spatial distribution of Queen conch (*Eustrombus gigas*).*

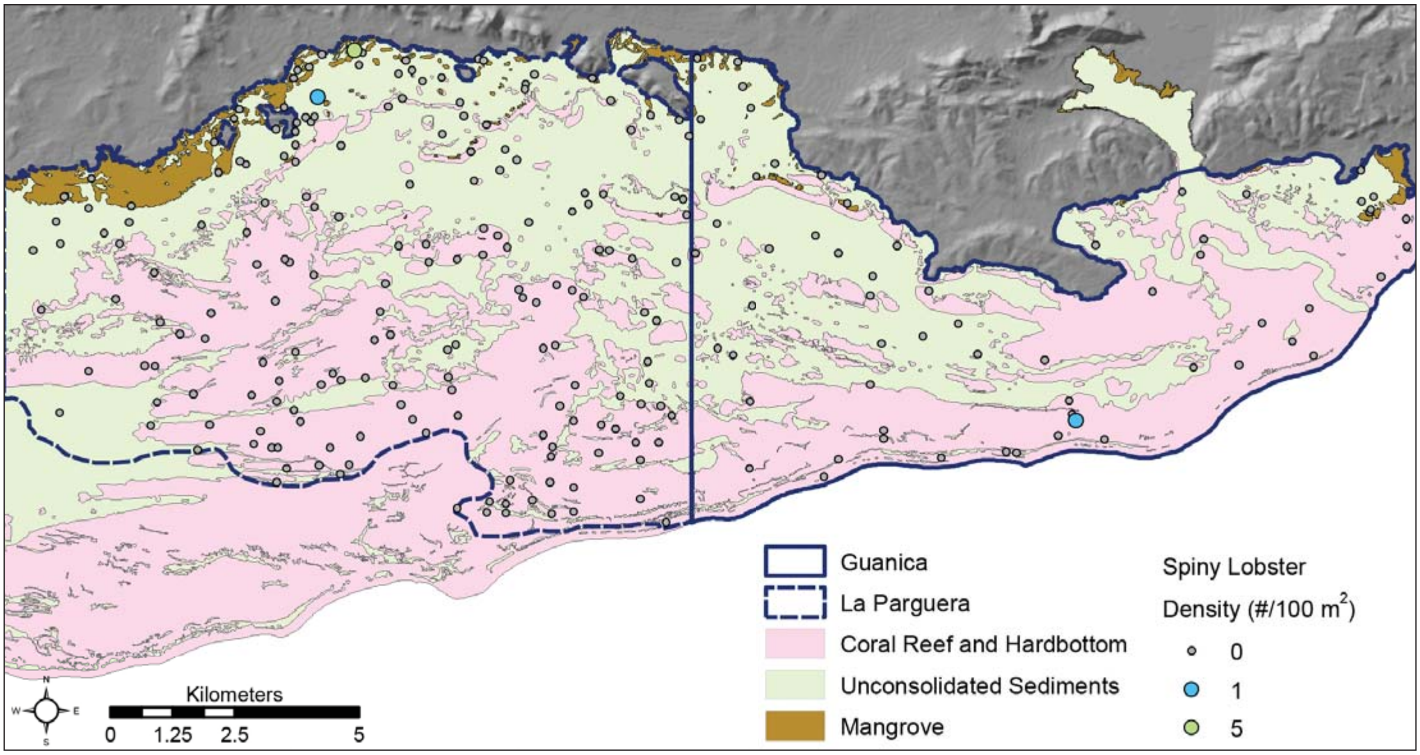

*Figure 2.24. Spatial distribution of spiny lobster (*Panularis argus*).*

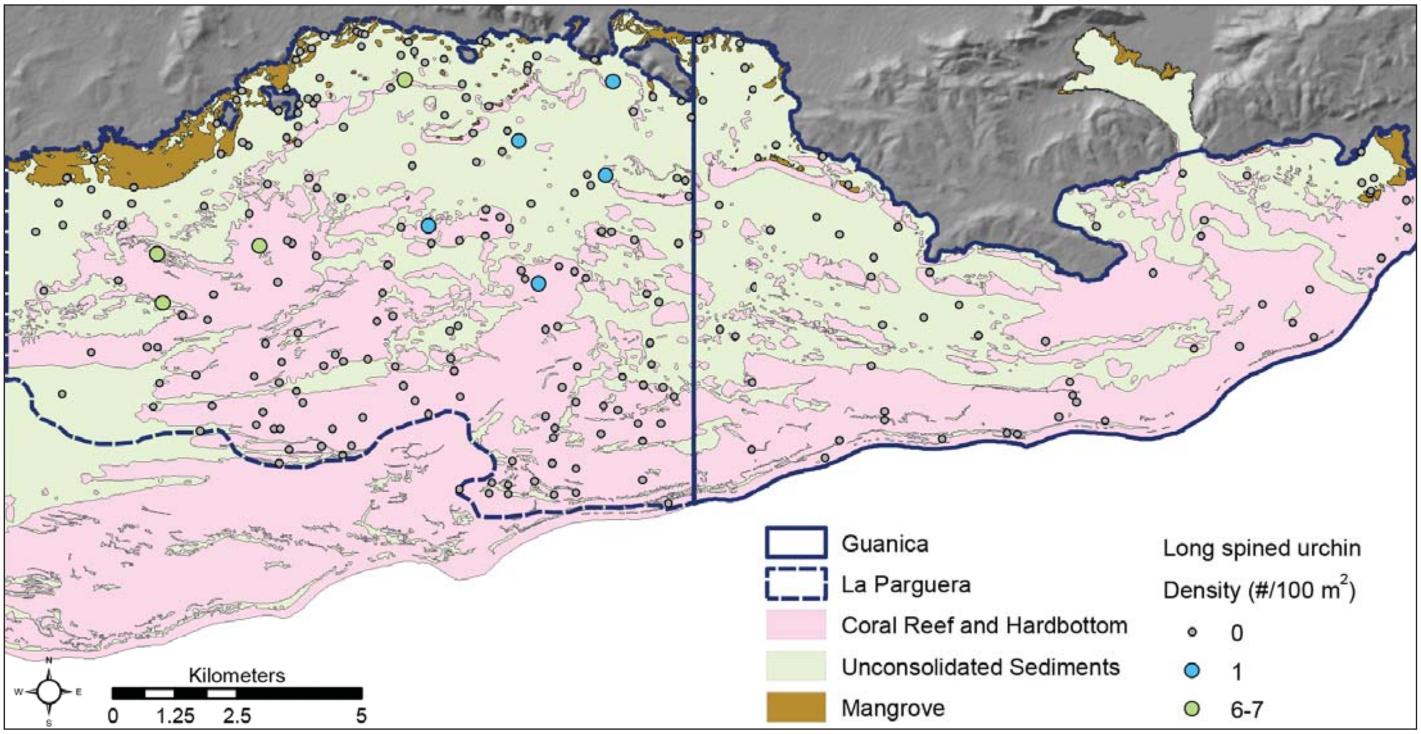

*Figure 2.25. Spatial distribution of long spined urchin (*Diadema antillarium*).*

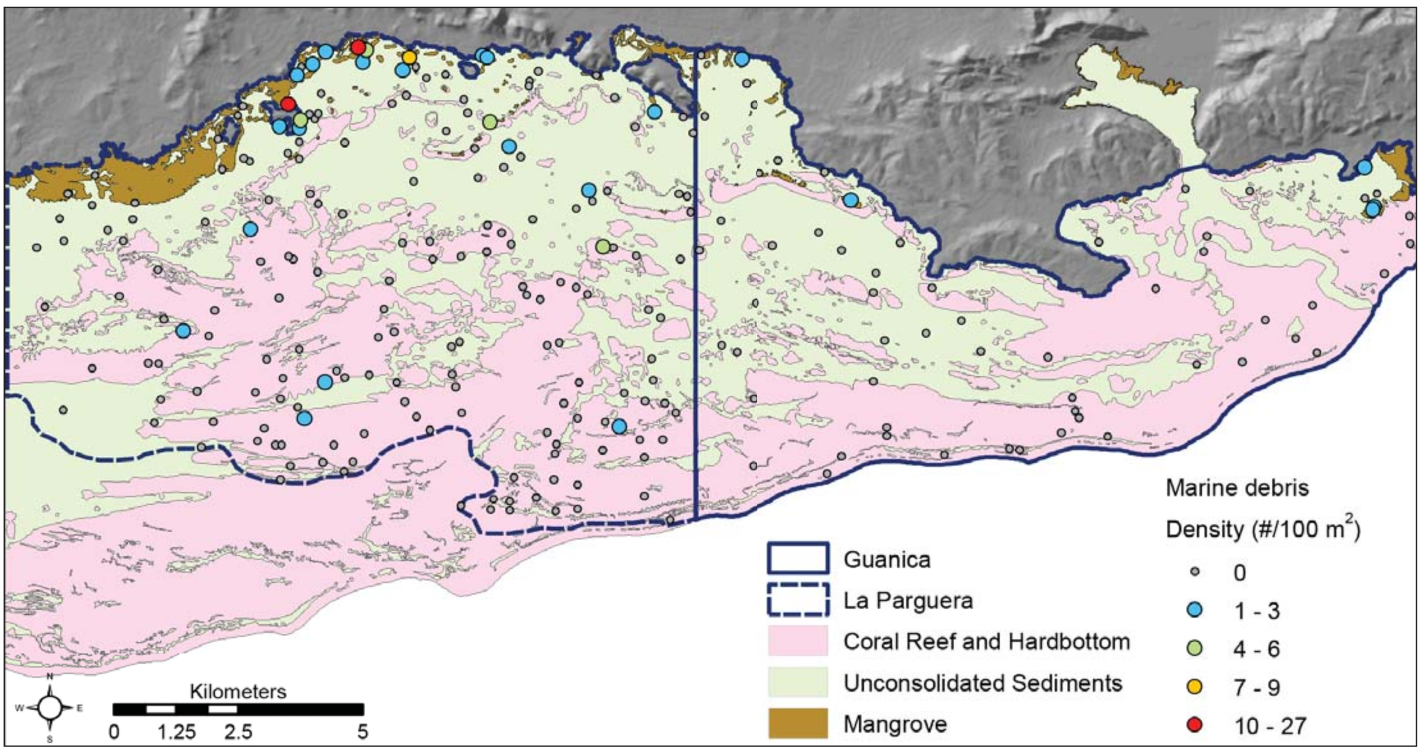

Figure 2.26. Spatial distribution of marine debris.

Table 2.3. Frequency of debris types pooled across all Guánica and La Parguera survey sites.

Debris type	Total number	% of total debris
Fishing line/reel	3	3.2
Derelict trap/materials	6	6.4
Net	2	2.1
Total gear pieces	**11**	**11.7**
Cans	28	29.8
Bottles	26	27.7
Rope	2	2.1
Tires	2	2.1
Other	25	26.6
Total non-gear	**83**	**88.3**
Total debris	**94**	**100**

Fish Assemblages

The fish community observed in the 2010 Guánica study area consisted of 36 taxonomic families and 119 species, while the La Parguera study area was represented by 43 families and 165 species over the 2008-2010 time period. Fish species richness ranged from 1 to 38 species per site (100 m^2) and was highest on hardbottom sites, especially close to the shelf edge, followed by mangrove (Figure 2.27a and Figure 2.28). Unconsolidated sediments were typified by lower species richness. Shannon diversity, which is a product of richness and evenness, followed similar trends (Figure 2.27b and Figure 2.29); however, diversity was significantly greater on hardbottom within the La Parguera study area compared to hardbottom within the Guánica study area (Table 2.4).

Mangrove sites typically exhibited the highest total fish density; however this was largely due to the presence of schooling round herring (*Jenkensia* spp.) and other small unidentified clupeiids, which can be orders of magnitude greater (100s–1,000s) than the next most abundant fish. Following removal of these small baitfish, mean fish density in mangrove was slightly lower compared with hardbottom, although typically more variable (Figure 2.27c). In the Guánica study area, the hardbottom survey sites situated on the southern portion of the shelf generally exhibited the highest density (Figure 2.30). Lowest density was typically observed on unconsolidated sediments, particularly at unvegetated sites. Similar patterns

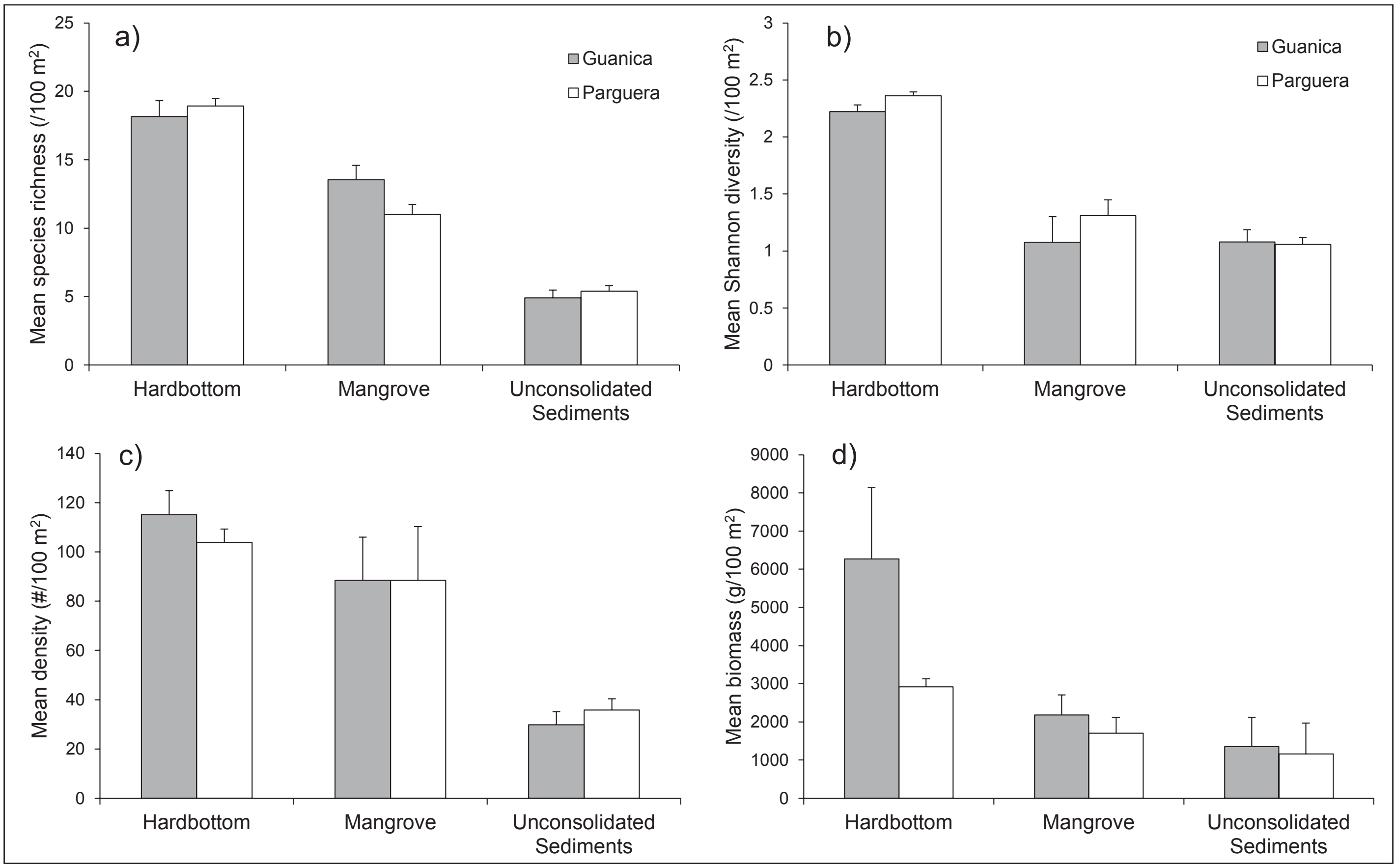

Figure 2.27. Mean (±SE) fish species a) richness, b) Shannon diversity, c) density and d) biomass by habitat type and study area.

were evident in La Parguera study area, with a concentration of high species richness sites near the outer reef edge; however some areas of moderate to high species richness were also found inshore of the reef edge on hardbottom habitats.

Highest levels of biomass generally occurred on hardbottom, particularly offshore near the shelf edge (Figure 2.27d and Figure 2.31). Within the Guánica study area, three out of the five sites with the greatest biomass were located in the far-eastern strata. Unconsolidated sediments were characterized by lowest mean biomass. Higher biomass was recorded at a few unconsolidated sediment sites, often when pelagic species such as great barracuda (*Sphyraena barracuda*), southern stingray (*Dasyatis americana*) or blue runner (*Caranx cryos*) passed through a transect. No significant differences were detected in density and biomass metrics between the two study areas (Table 2.4) for all three habitat types/strata.

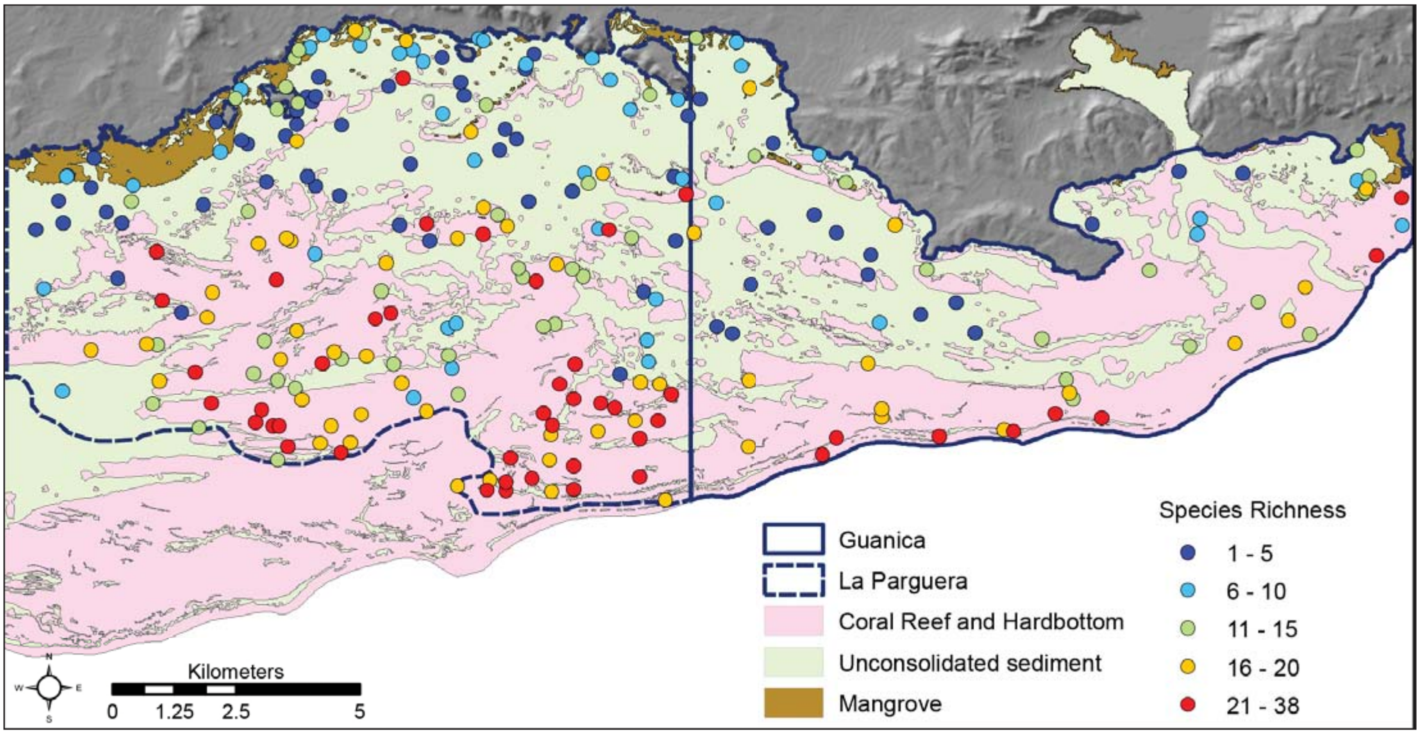

Figure 2.28. Fish species richness.

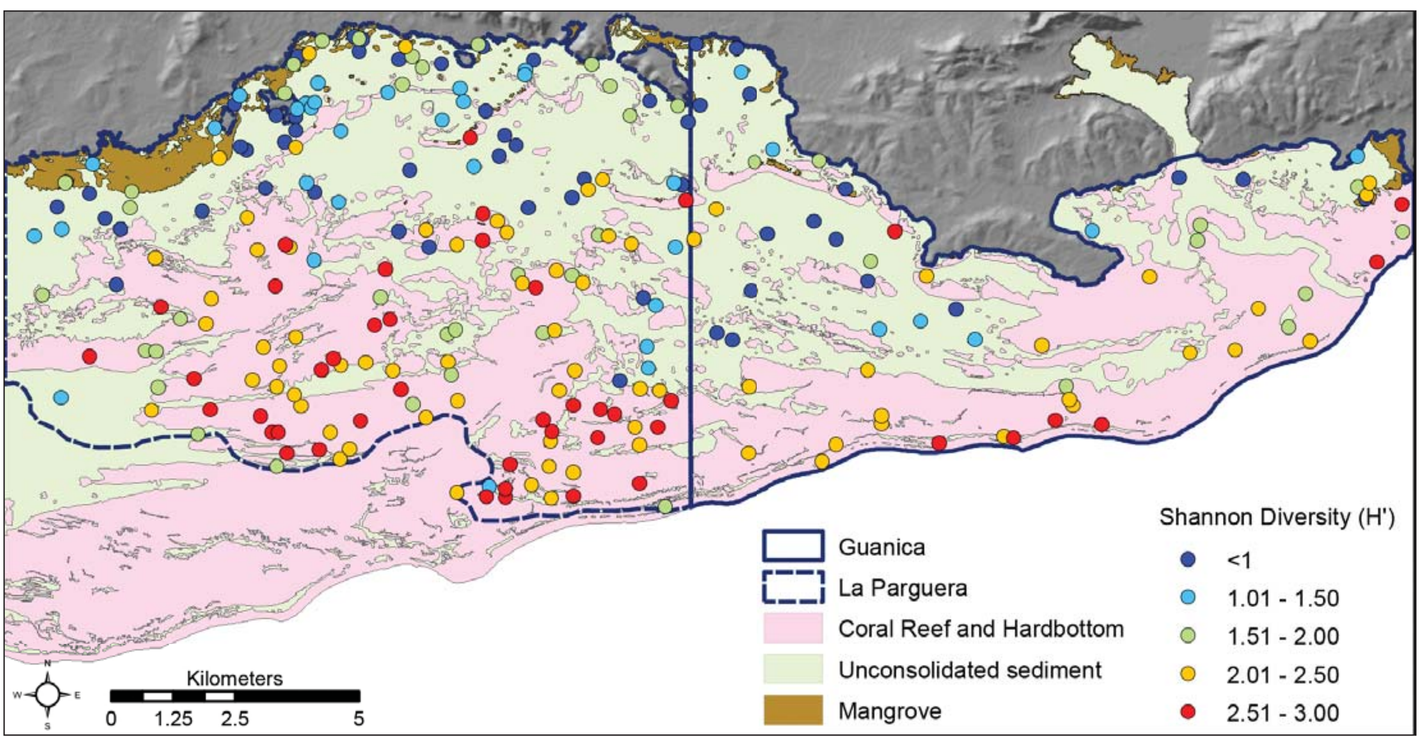

Figure 2.29. Fish species diversity.

The nMDS and ANOSIM analyses further indicate that fish assemblages in southwest Puerto Rico differ by bottom type, but are similar between the two study areas. There was a clear separation of fish communities between hardbottom, unconsolidated sediment and mangrove surveys (Figure 2.32). Mangrove and hardbottom sites tended to be highly clustered, indicating a high degree of similarity in species composition among sites within each respective habitat type. In contrast, unconsolidated sediment sites tended to be more dispersed, indicating more dissimilarity among sites within this group. While sites within this bottom type were generally characterized by low overall abundance, they often varied in their species composition. Within bottom types, there was not a distinct separation of sites between the La Parguera and Guánica study areas. The results of the ANOSIM test also indicate that there was a statistically significant difference in community composition among the three bottom types ($R=0.664$, $p<0.001$), but the effect of study area was not significant ($R=0.111$, $p<0.50$).

Table 2.4. Results of nonparametric analysis (Wilcoxon-Mann Whitney) for fish community metrics between the Guánica and La Parguera study areas. As described in the methods, a random subset of the La Parguera (2008-2010) dataset was used. Bold text with an asterisk (*) indicates a significant difference at the α=0.05 level.

Metric	Hardbottom		Mangrove		Unconsolidated Sediments	
	Z	p-value	Z	p-value	Z	p-value
Total density	1.14	0.255	0.88	0.377	-0.90	0.366
Total biomass	1.06	0.290	0.62	0.536	0.27	0.787
Species richness	-1.08	0.279	1.25	0.210	-1.34	0.180
Shannon diversity	-2.11	**0.035***	0.00	1.000	-1.04	0.296
Herbivore density	0.33	0.739	0.40	0.689	-2.31	**0.021***
Invertivore density	2.27	**0.023***	-0.31	0.757	-0.68	0.495
Piscivore density	1.11	0.269	1.46	0.143	-0.82	0.413
Planktivore density	-1.96	**0.050***	1.18	0.237	-0.36	0.716
Herbivore biomass	-0.42	0.673	-0.97	0.331	-1.21	0.225
Invertivore biomass	1.81	0.070	-1.15	0.251	0.11	0.916
Piscivore biomass	1.77	0.077	0.88	0.377	-0.86	0.391
Planktivore biomass	-1.44	0.150	0.91	0.363	-0.35	0.726
Grouper (Serranidae) density	1.00	0.317	0.89	0.374	-1.40	0.162
Grouper (Serranidae) biomass	0.82	0.410	0.89	0.374	-1.40	0.162
Cephalopholis fulva density	2.43	**0.015***	0.00	1.000	0.00	1.000
Cephalopholis fulva biomass	2.27	**0.023***	0.00	1.000	0.00	1.000
Cephalopholis cruentata density	0.12	0.907	0.89	0.374	-1.40	0.162
Cephalopholis cruentata biomass	0.37	0.711	0.89	0.374	-1.40	0.162
Snappers (Lutjanidae) density	-1.35	0.177	-1.02	0.308	0.54	0.590
Snappers (Lutjanidae) biomass	-0.46	0.644	0.09	0.930	0.00	1.000
Lutjanus apodus density	0.42	0.671	0.36	0.722	0.00	1.000
Lutjanus apodus biomass	0.33	0.740	1.06	0.289	0.00	1.000
Lutjanus synagris density	0.57	0.570	0.89	0.374	1.89	0.059
Lutjanus synagris biomass	0.61	0.544	0.89	0.374	1.68	0.094
Lutjanus griseus density	0.00	1.000	-1.44	0.151	0.00	1.000
Lutjanus griseus biomass	0.00	1.000	-0.18	0.859	0.00	1.000
Ocyurus chrysurus density	-1.99	**0.046***	-1.77	0.077	-1.00	0.315
Ocyurus chrysurus biomass	-1.18	0.239	-1.77	0.077	-0.97	0.331
Grunts (Haemulidae) density	0.42	0.673	-0.88	0.377	-0.50	0.619
Grunts (Haemulidae) biomass	0.78	0.436	-2.03	**0.042***	-0.58	0.560
Haemulon flavolineatum desnity	0.88	0.378	0.98	0.328	0.00	1.000
Haemulon flavolineatum biomass	0.72	0.470	0.00	1.000	0.00	1.000
Haemulon sciurus density	1.31	0.189	0.00	1.000	0.00	1.000
Haemulon sciurus biomass	1.26	0.208	-1.95	0.052	0.00	1.000
Haemulon plumierii density	0.59	0.557	0.61	0.540	-1.40	0.162

Table 2.4 (continued). Results of nonparametric analysis (Wilcoxon-Mann Whitney) for fish community metrics between the Guánica and La Parguera study areas. As described in the methods, a random subset of the La Parguera (2008-2010) dataset was used. Bold text with an asterisk (*) indicates a significant difference at the α=0.05 level.

Metric	Hardbottom		Mangrove		Unconsolidated Sediments	
	Z	p-value	Z	p-value	Z	p-value
Haemulon plumierii biomass	0.62	0.537	0.61	0.540	-1.40	0.162
Haemulon aurolineatum density	0.48	0.633	-0.89	0.374	-0.95	0.340
Haemulon aurolineatum biomass	0.45	0.655	-0.89	0.374	-0.95	0.340
Surgeonfish (Acanthuridae) density	1.05	0.295	1.10	0.273	-0.02	0.981
Surgeonfish (Acanthuridae) biomass	-0.53	0.594	1.21	0.225	0.02	0.981
Acanthurus bahianus density	0.39	0.699	-0.89	0.374	0.56	0.572
Acanthurus bahianus biomass	-1.40	0.162	-0.89	0.374	0.62	0.537
Acanthurus coeruleus density	1.04	0.300	1.37	0.169	-0.95	0.340
Acanthurus coeruleus biomass	1.22	0.223	1.37	0.169	-0.95	0.340
Parrotfish (Scaridae) density	-4.20	**<0.001***	1.32	0.187	-1.74	0.081
Parrotfish (Scaridae) biomass	-2.22	**0.027***	0.64	0.524	-1.28	0.200
Scarus iseri density	-3.15	**0.002***	1.61	0.108	-0.99	0.323
Scarus iseri biomass	-3.12	**0.002***	1.71	0.088	-0.94	0.347
Sparisoma aurofrenatum density	-2.59	**0.010***	0.00	1.000	0.00	1.000
Sparisoma aurofrenatum biomass	-2.14	**0.033***	0.00	1.000	-0.07	0.944
Sparisoma viride density	-1.21	0.225	0.61	0.540	0.00	1.000
Sparisoma viride biomass	-0.92	0.360	0.54	0.587	0.00	1.000
Wrasses (Labridae) density	2.80	**0.005***	1.77	0.077	1.30	0.195
Wrasses (Labridae) biomass	2.74	**0.006***	1.77	0.077	2.05	**0.040***
Goatfish (Mullidae) density	-1.61	0.107	0.00	1.000	-2.05	**0.041***
Goatfish (Mullidae) biomass	-1.43	0.154	0.00	1.000	-2.05	**0.041***
Damselfish (Pomacentridae) density	3.15	**0.002***	0.62	0.534	-0.92	0.355
Damselfish (Pomacentridae) biomass	0.33	0.739	0.53	0.596	-0.92	0.355

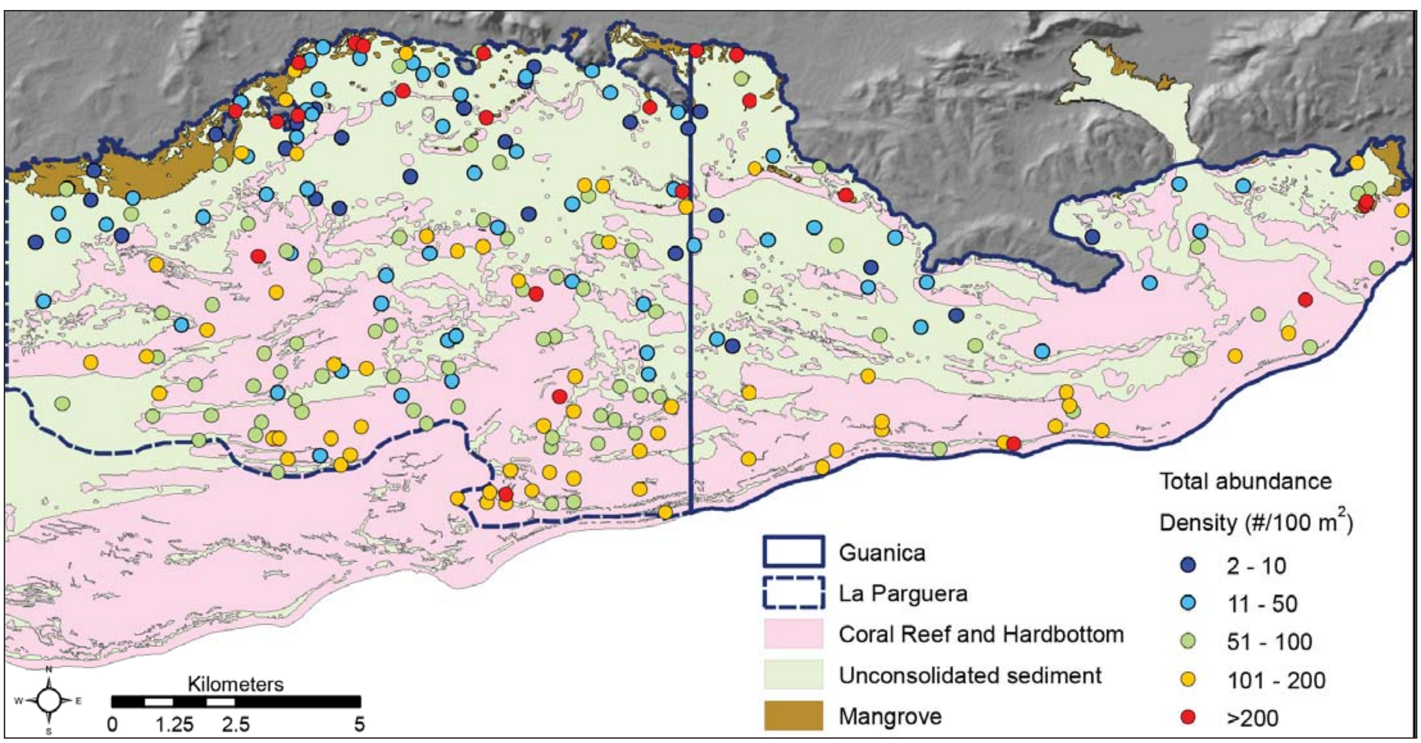

Figure 2.30. Total fish density.

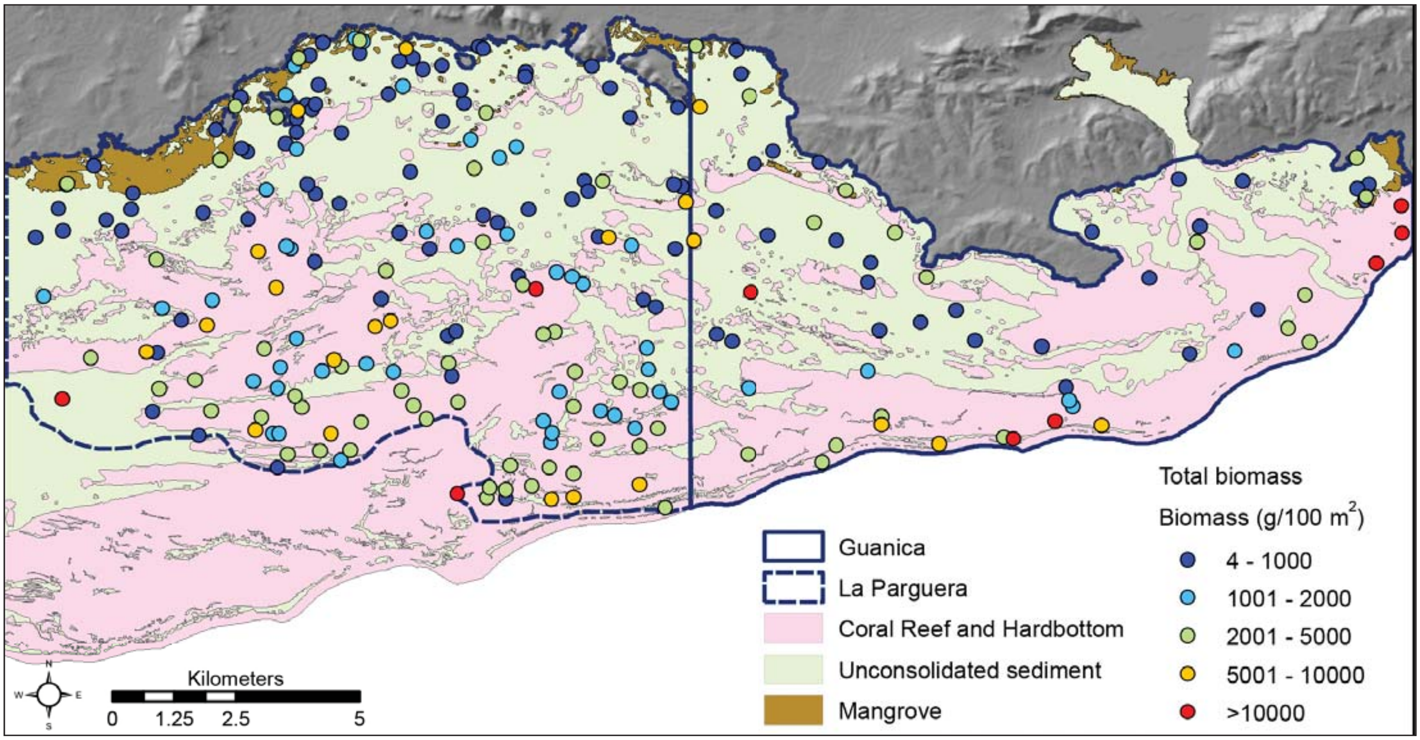

Figure 2.31. Total fish biomass.

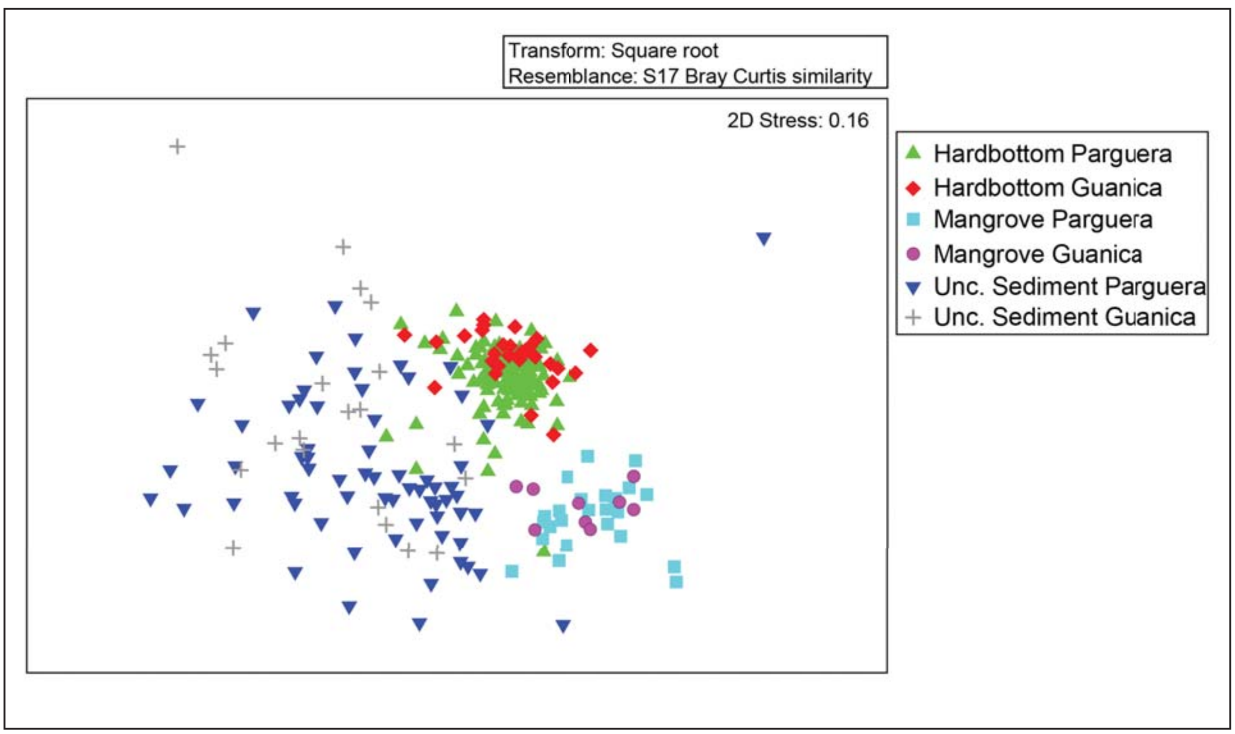

Figure 2.32. Non-metric multidimensional (nMDS) scaling ordination based on between site similarity composition using fish abundance data. Sites are color-coded by habitat type and study area.

Trophic Groups, Families and Species

Biomass and abundance were distributed unevenly throughout trophic groups (Figures 2.33-2.36). On hardbottom and unconsolidated sediments, biomass and density were greatest for herbivores and invertivores (see Appendix A, Tables A.1 and A.2) for species list by trophic guild; Figure 2.33 and Figure 2.34). Mean herbivore abundance and particularly biomass were markedly higher on hardbottom than the other bottom types (Figure 2.33). In particular, the highest values of herbivore biomass were observed on hardbottom along the outer shelf. Herbivore density on unconsolidated sediments varied significantly between the two study areas (Table 2.4) with higher values in the La Parguera study area. Mean herbivore density in mangrove habitat was over three times greater in the La Parguera study area compared to the Guánica study area, but there was no significant difference between the study areas due to high variability between sites (Table 2.4).

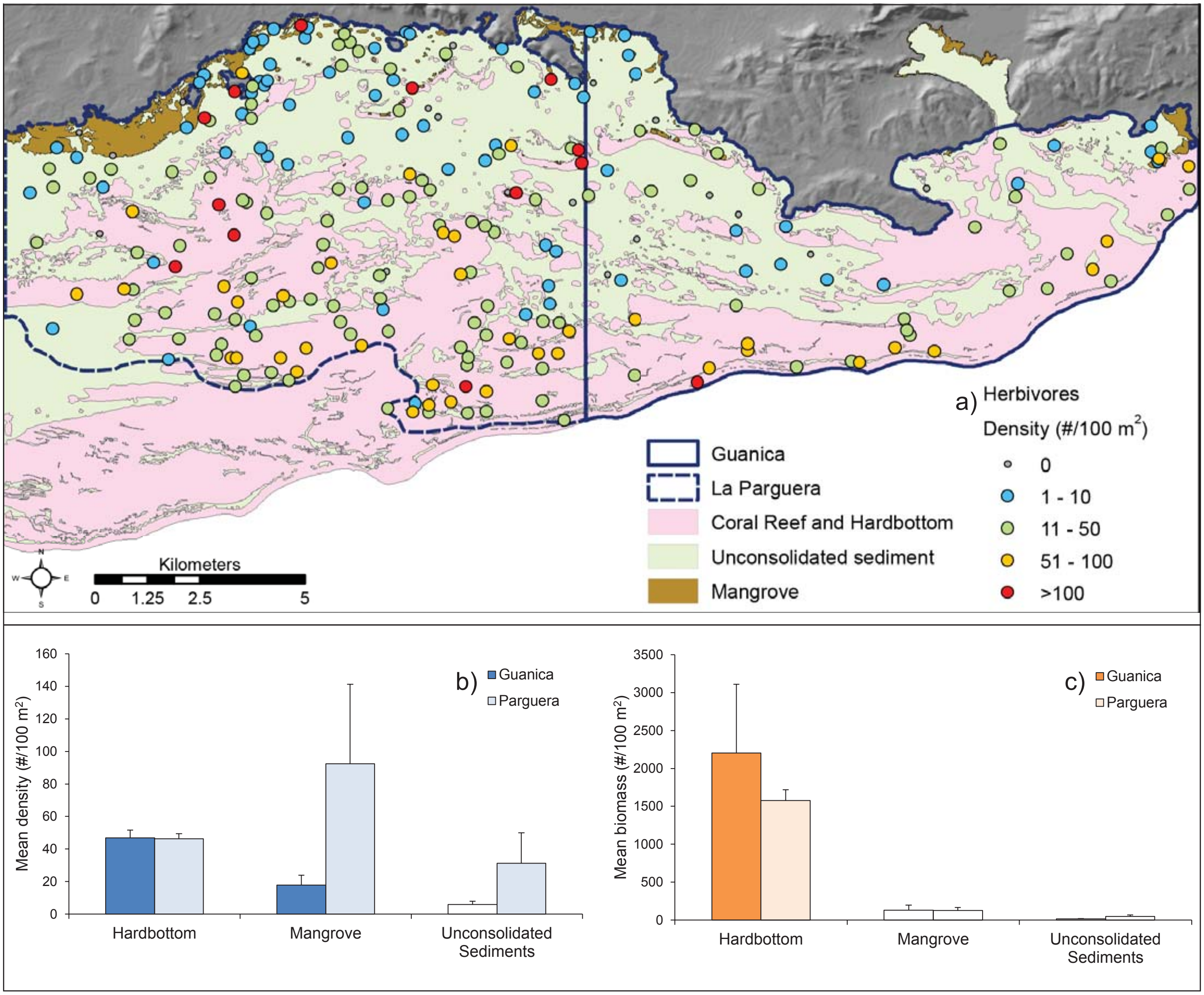

Figure 2.33. a) Spatial distribution, b) mean (±SE) density by habitat and c) mean (±SE) biomass by habitat of herbivores.

Invertivores (I; see Appendix A, Tables A.1, A.2 for complete list) were distributed throughout the study area, with the highest levels observed on hardbottom and in mangrove habitat (Figure 2.34). Results of the non-parametric Wilcoxon tests indicate that density of invertivores were significantly greater in hardbottom surveys within the Guánica study area compared to the La Parguera study area (Table 2.4). This was partly attributed to several sites on the outer shelf in the Guánica study area that were characterized by high densities of a number of invertivore species.

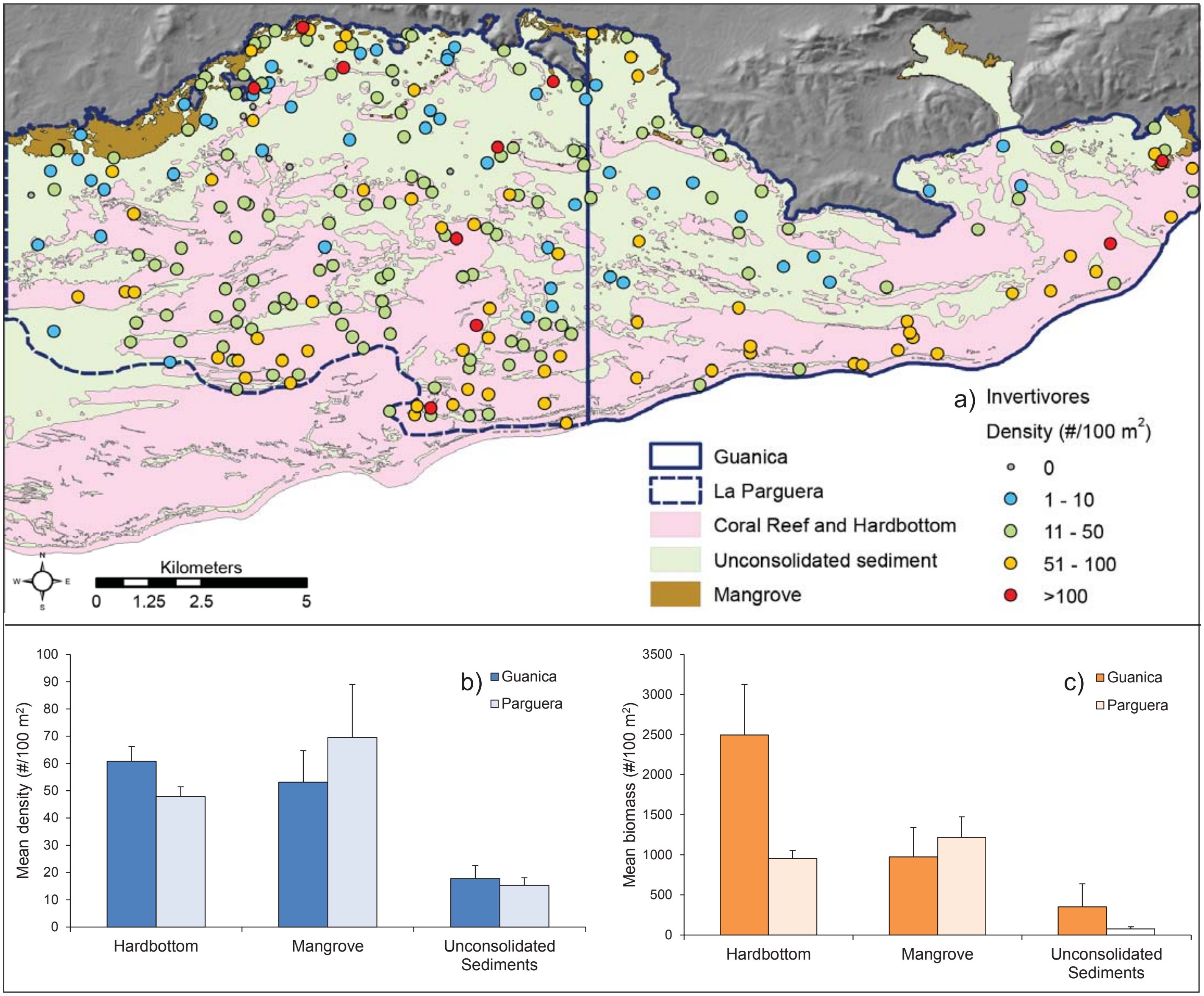

Figure 2.34. a) Spatial distribution, b) mean (±SE) density by habitat and c) mean (±SE) biomass by habitat of invertivores.

Piscivores (P; see Appendix A, Tables A.1, A.2 for complete list) constituted a small percentage of total fish abundance, but made up a greater proportion of the biomass (Figure 2.35). Piscivores were most abundant in mangroves, but individuals were generally juveniles. A few unconsolidated sediment sites exhibited high biomass due to the presence of species such as Great barracuda and Carangidae (jack) species. In mangrove habitats, planktivores exhibited highest mean density and variability due to occasional large schools of small clupeids (Figure 2.36).

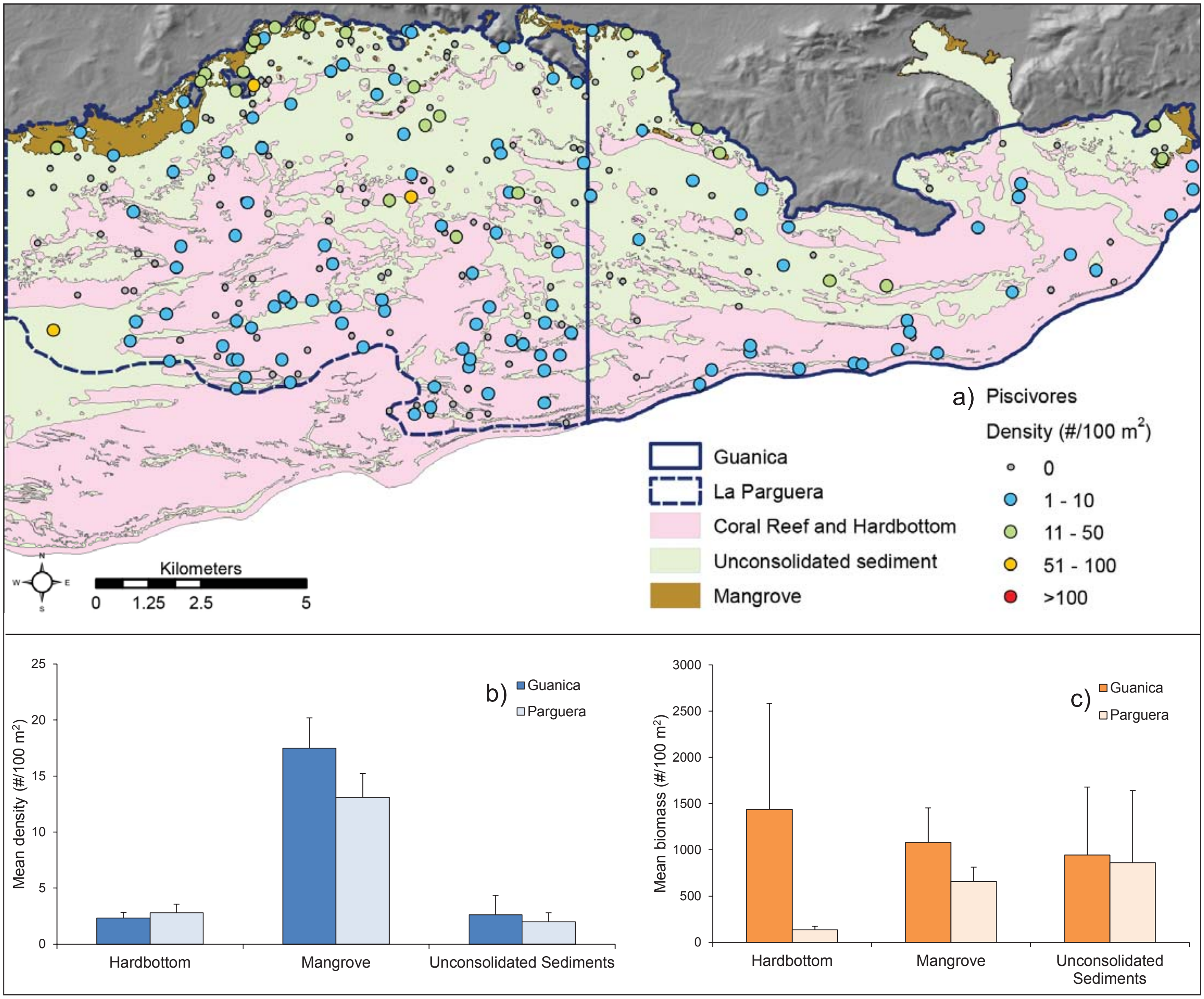

Figure 2.35. a) Spatial distribution, b) mean (±SE) density by habitat and c) mean (±SE) biomass by habitat of piscivores.

For each bottom type and study area, rankings of families with the highest mean abundance and biomass are presented in Figures 2.37 and 2.38. Wrasse (Labridae), damselfish (Pomacentridae) and parrotfish (Scaridae) families were most numerically abundant on hardbottom habitats in both the Guánica and La Parguera study areas; although their rankings differed between study areas (Figure 2.37a-b). Top-ranked families differed for biomass between study areas. Parrotfish ranked highest in terms of biomass in the La Parguera study area, followed by surgeonfishes (Acanthuridae; Figure 2.38b). The three families that accounted for the highest proportional biomass in the Guánica study area were triggerfishes (Balistidae), jacks (Carangidae) and grunts (Haemulidae; Figure 2.38a). All three families exhibited high between site variability.

Within mangrove habitats of both study areas, small schooling fishes in the Clupeidae family were patchily abundant and accounted for the highest mean density, particularly in the Guánica study area (Figure 2.37c-d). Snappers (Family Lutjanidae) accounted for the highest mean biomass in both study areas, followed by grunts (Family Haemulidae; Figure 2.38c-d). In addition, barracudas (Family Sphyraenidae) were commonly observed in mangrove and mean biomass ranked fourth and third among family groups in the Guánica and La Parguera study areas, respectively.

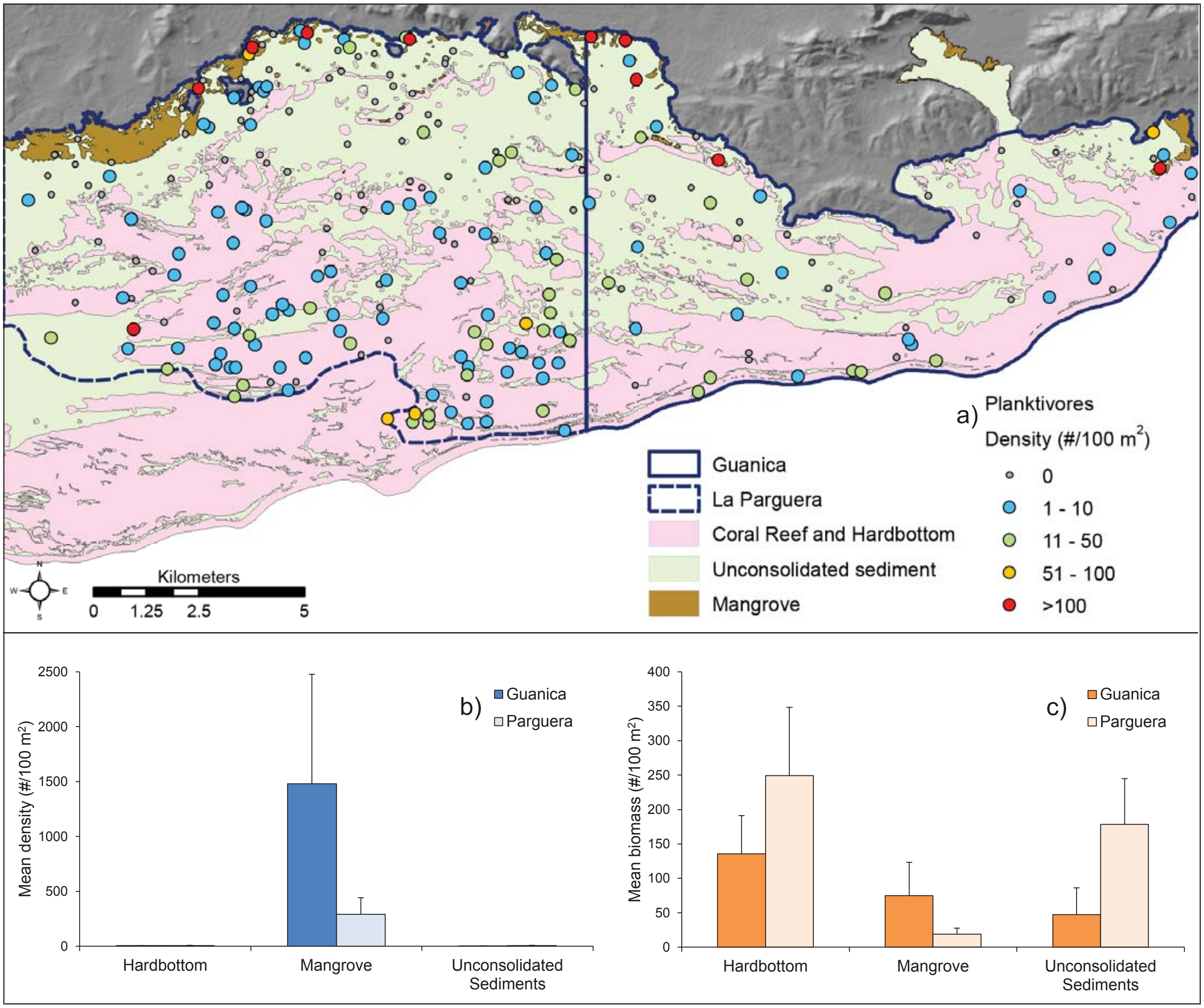

Figure 2.36. a) Spatial distribution, b) mean (±SE) density by habitat and c) mean (±SE) biomass by habitat of planktivores.

Mean density and biomass of family groups were typically variable on unconsolidated sediments. Grunts accounted for the highest mean density in the Guánica study area, while silversides (Atherinidae) dominated in the La Parguera study area with a high standardd error (Figure 2.37e-f). Larger-bodied families Sphyraenidae, Dasyatidae (stingrays) and Carangidae, though infrequently observed, accounted for the highest mean biomass in the Guánica study area, while stingrays were not observed on surveys within the La Parguera study area (Figure 2.38e-f).

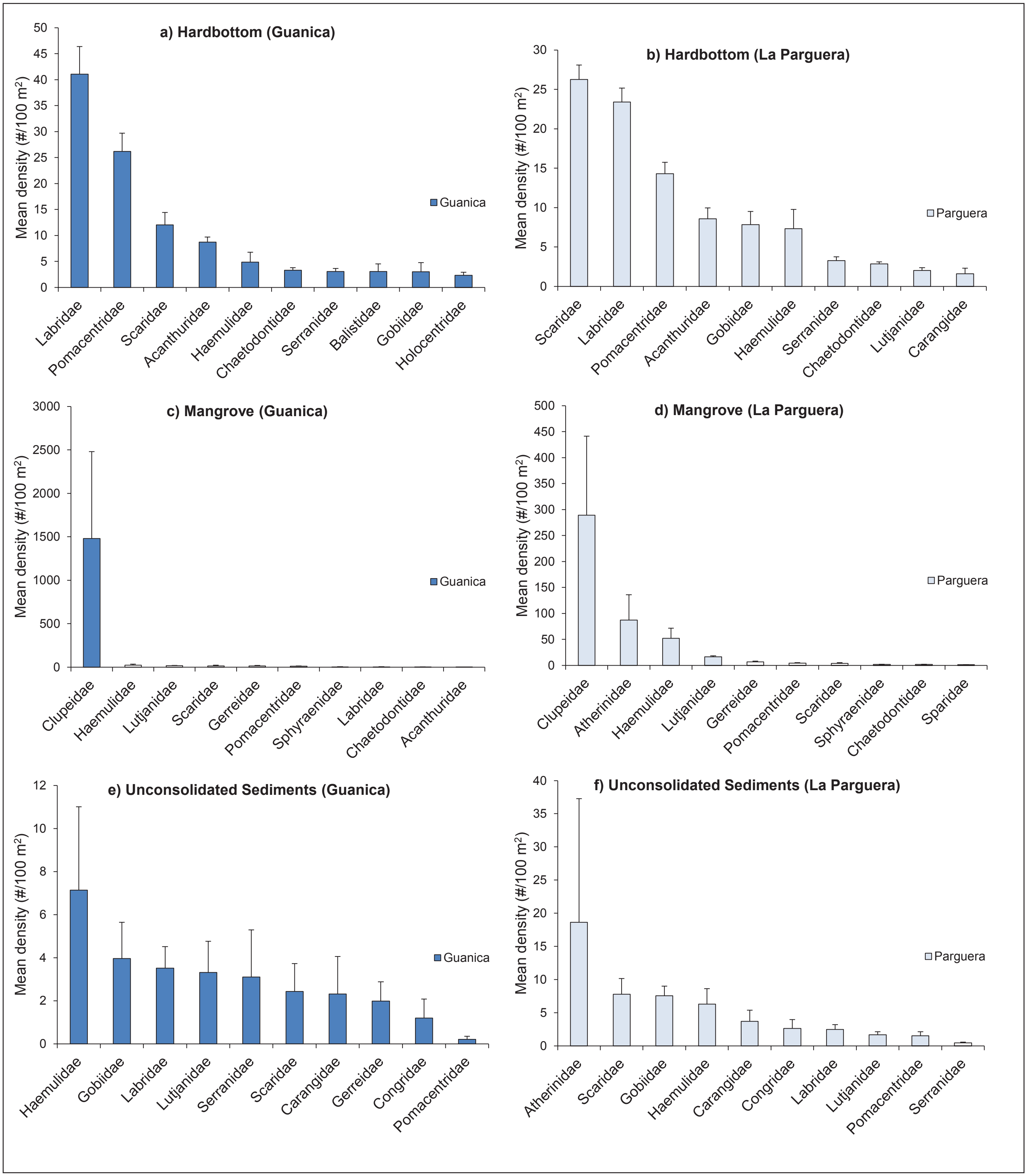

Figure 2.37. Mean (±SE) density of the major (top ten) fish families by habitat and study area.

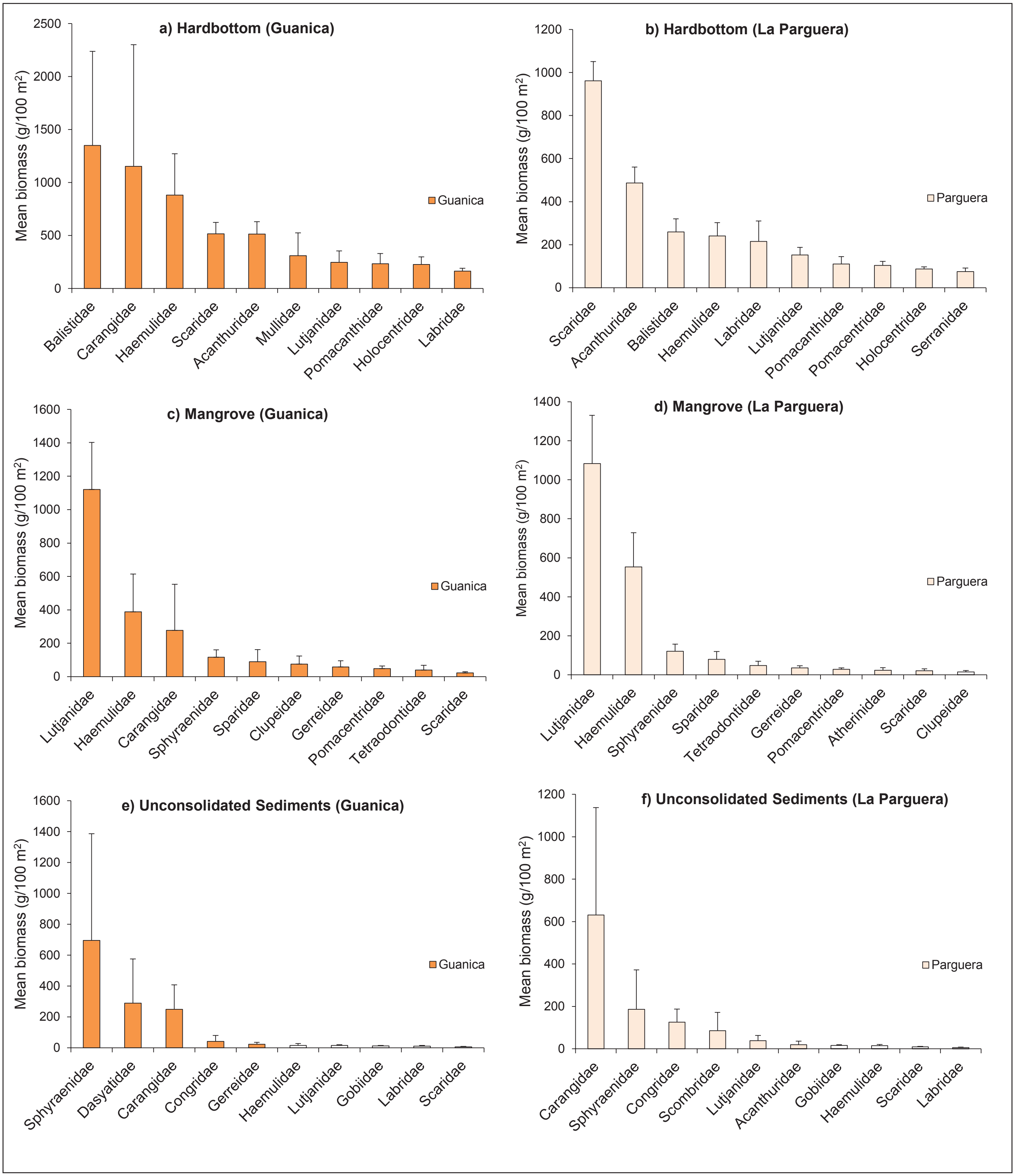

Figure 2.38. Mean (±SE) biomass of the major (top ten) fish families by habitat and study area.

<u>Groupers (Serranidae)</u>

Groupers (*Cephalopholis* and *Epinephelus* spp.) were infrequent, primarily small in size and were almost exclusively found on hardbottom (Figure 2.39). In both study areas, most grouper individuals belonged to three species: coney (*C. fulva*), graysby (*C. cruentata*) and red hind (*E. guttatus*). In addition, one Nassau grouper (*E. striatus*) was documented in the La Parguera study area, while three rock hind (*E. adscensionis*) were observed in the Guánica study area. Across the region, the vast majority of individuals were observed on hardbottom, particularly near the shelf edge (Figure 2.39). While more grouper were observed in Guánica than La Parguera, grouper density and biomass were not significantly different between the two study areas (Table 2.4). Species of the genus *Myctoperca* were conspicuously absent from both study areas.

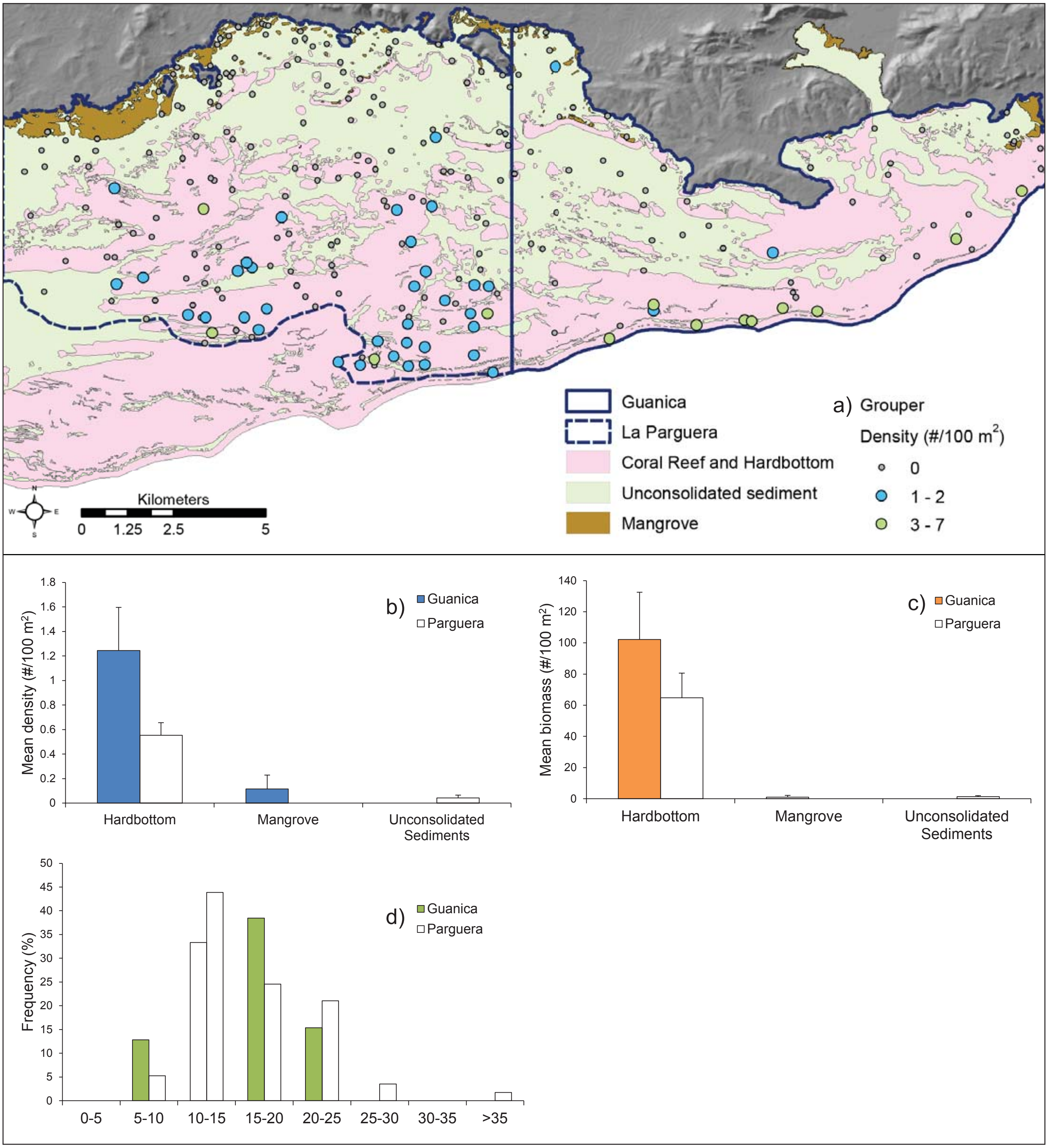

Figure 2.39. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of groupers (Family Serranidae).

Coney was the most abundant grouper species observed in the study and was found exclusively on hardbottom. This species exhibited a strong spatial pattern; the majority of individuals were located offshore close to the shelf edge (Figure 2.40). The majority of individuals were small adults. The species was sighted more frequently in the Guánica study area and results of a non-parametric Wilcoxon-Mann Whitney test indicate abundance and biomass were significantly greater in the Guánica study area (Table 2.4). This may be at least partly due to the species preference for shelf edge environments which was under-sampled in the La Parguera study area because the 2001 map did not extend to the shelf edge.

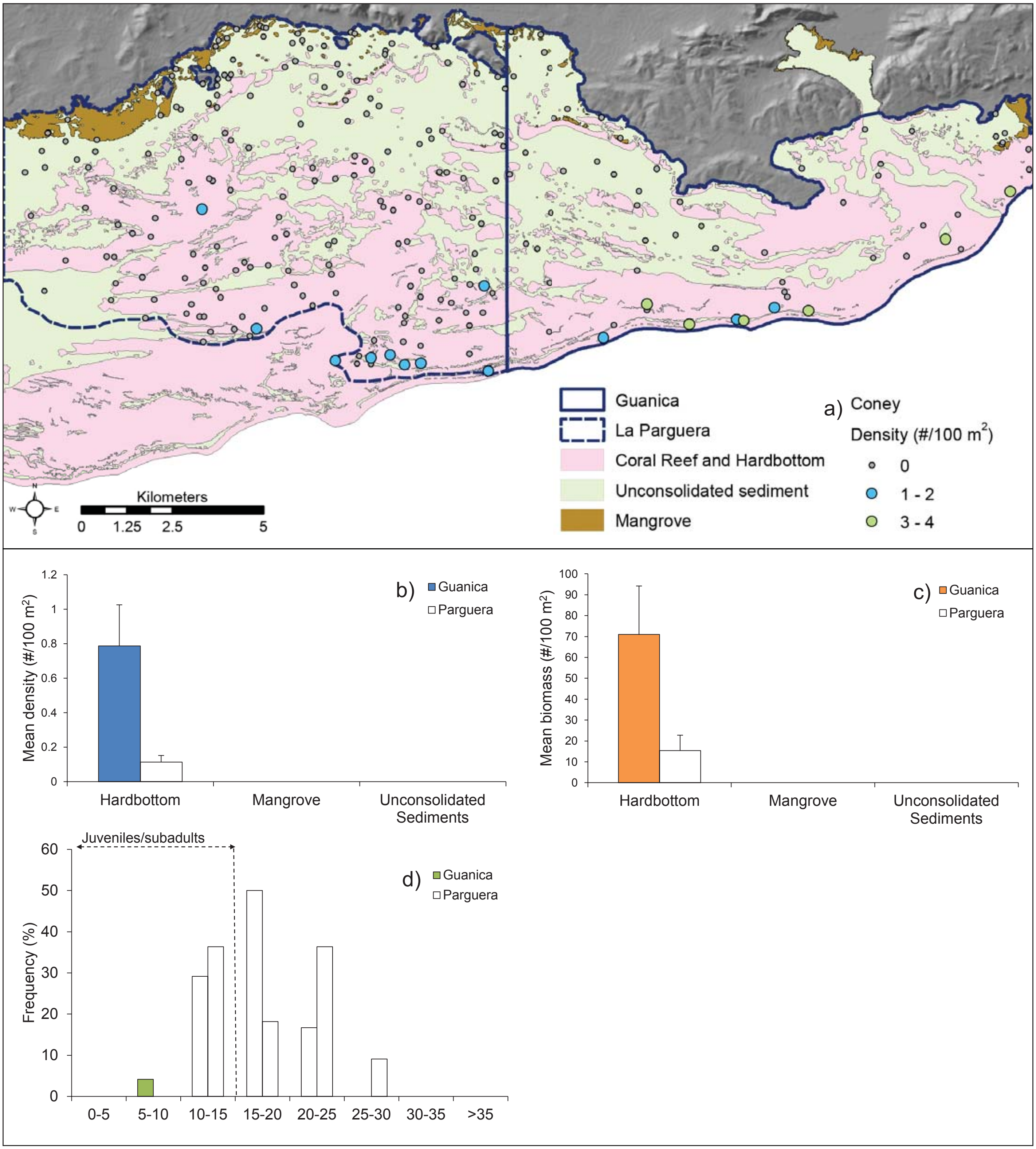

*Figure 2.40. a) Spatial distribution, b) mean (±SE) density by habitat c) mean (±SE) biomass by habitat and d) size (cm) frequency of coney (*Cephalopholis fulva*).*

Graysby exhibited a similar spatial pattern to coney except that the species was occasionally observed in nearshore mangrove and unconsolidated sediment habitats in addition to hardbottom farther out on the shelf. Biomass and density did not differ significantly between study areas in the three surveyed habitats (Figure 2.41b-c and Table 2.4). The species was absent from survey sites in the eastern portion of the Guánica study area (Figure 2.41). The majority of individuals were subadults and small adults (Figure 2.41d). Red hind was infrequently observed, sighted in 4% and 5% of survey transects in the La Parguera and Guánica study areas, respectively. The species was found exclusively on hardbottom.

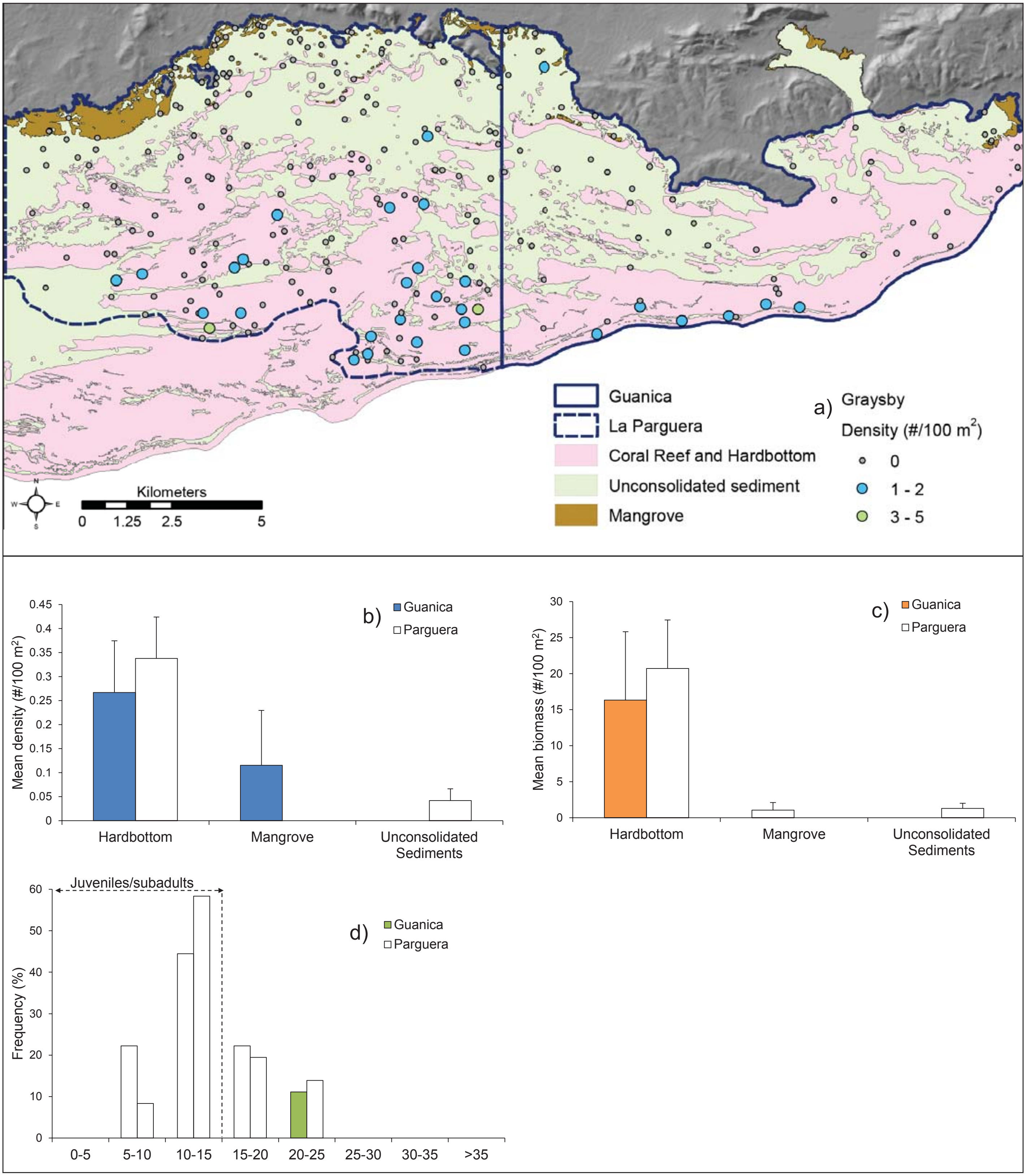

*Figure 2.41. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of graysby (*Cephalopholis cruentata*).*

<u>Snappers (Lutjanidae)</u>

Snappers were detected across the shelf in all investigated habitats but were most abundant in mangroves (Figure 2.42a-b). However, as described below, distribution varied by species and lifestage. The lowest biomass was observed over unconsolidated sediments (Figure 2.42c). Within the two study areas, eight Lutjanid species were documented, with schoolmaster (*Lutjanus apodus*), yellowtail (*Ocyurus chrysurus*), lane (*L. synagris*) and gray snappers (*L. griseus*) accounting for the majority of observations (98%). The remaining species were infrequently sighted (Appendix A, Tables A.1, A.2). Lutjanidae size frequency was skewed toward smaller size classes (<15 cm; Figure 2.42d).

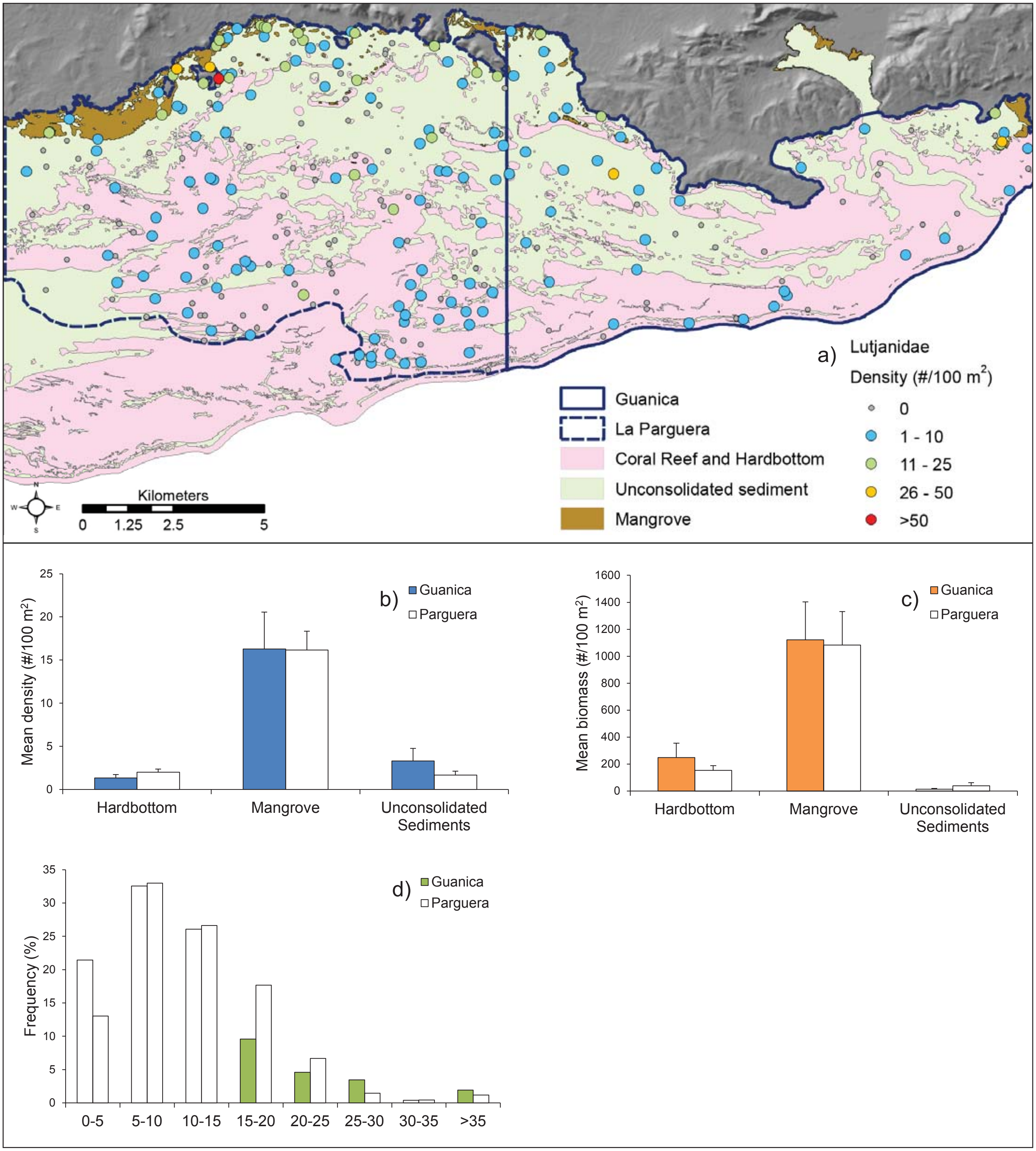

Figure 2.42. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of snappers (Family Lutjanidae).

Schoolmaster were observed at 21% and 19% of survey transects within the Guánica and La Parguera study areas, respectively. They were most abundant in nearshore mangrove fringes and cays (Figure 2.43a-b). A few individuals were observed on offshore hardbottom habitat, while none were observed on unconsolidated sediments. In both study areas, the majority of schoolmaster (>90%) were juveniles/subadults (Figure 2.43d). With the exception of one hardbottom site, all adult-sized individuals, about 7% of the total, were located on mangrove habitat.

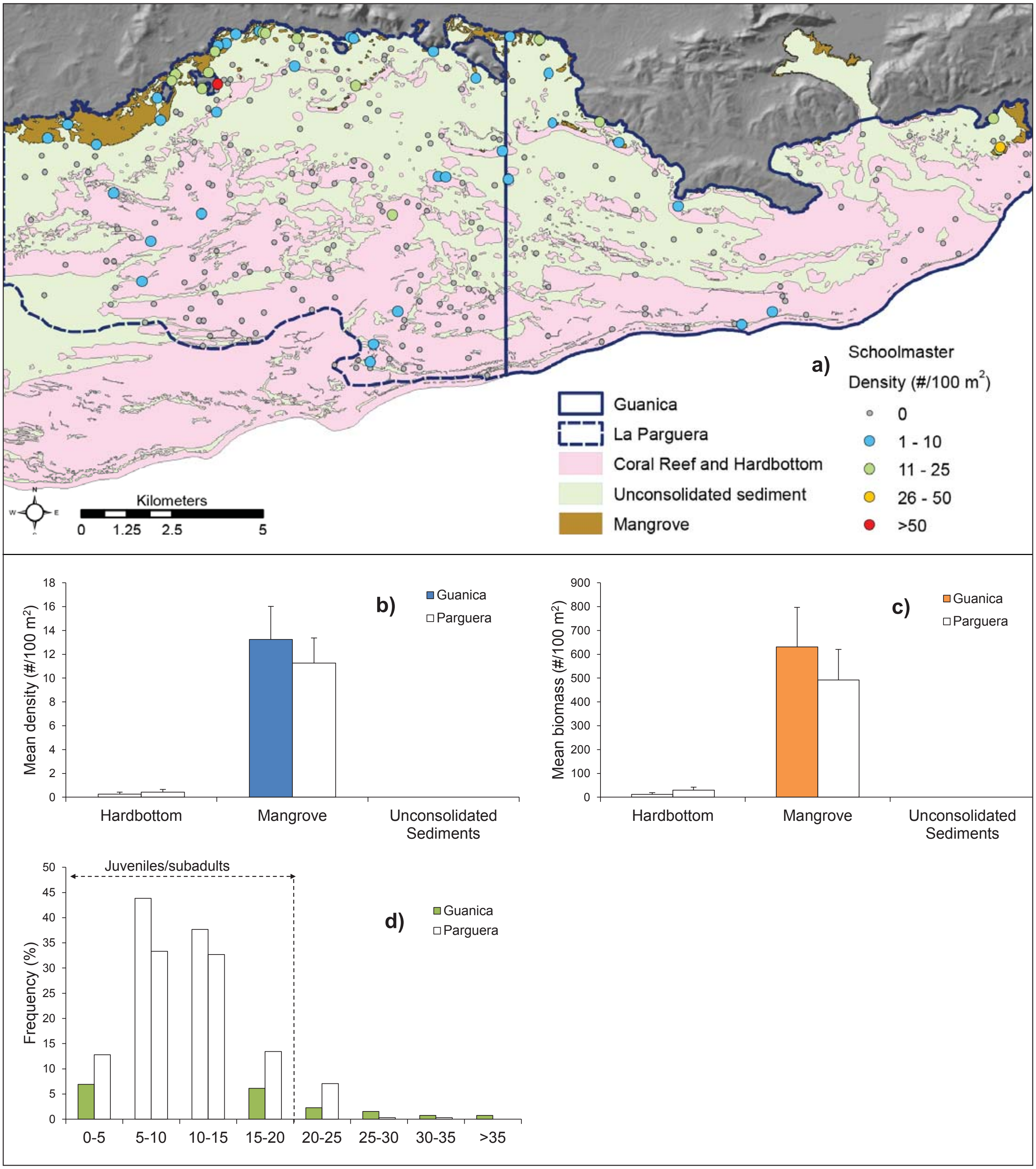

*Figure 2.43. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of schoolmaster (*Lutjanus apodus*).*

Lane snapper were observed in [illegible] of survey transects in the Guánica study area and [illegible] of sites within the La Parguera study area. The majority of sightings occurred on unconsolidated sediment habitat (Figure 2.44a). The species was generally observed in small numbers, but on one survey transect in the Guánica study area [illegible] individuals were observed. There were no significant differences in density or biomass between the two study areas (Table [illegible]). In both study areas, the vast majority of individuals were juveniles in the two smallest size classes ([illegible] and [illegible] cm). Only one adult was recorded in Guánica and no adults in La Parguera (Figure 2.44d).

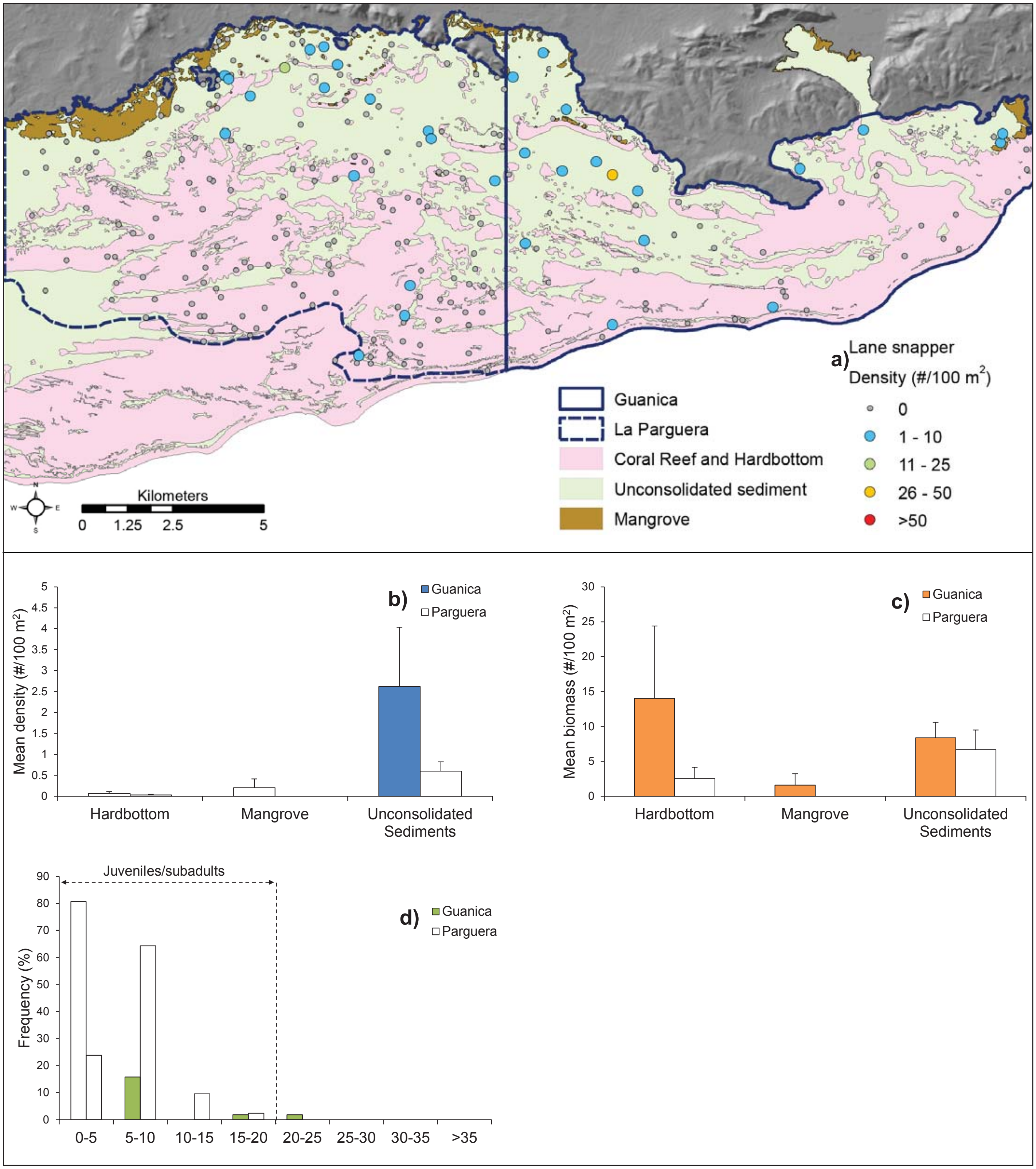

Figure 2.44. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of lane snapper (Lutjanus synagris).

Gray snapper was observed with similar frequency in both study areas (Guánica: 11% of surveys, La Parguera: 10% of surveys) and were exclusively associated with mangrove fringes and cays in both study area (Figure 2.45a,b). The three sites with the highest biomass were located in the fringing mangroves in the western portion of the La Parguera study area, similar to what was observed by Pittman et al. (2010). Observed individuals were comprised primarily of juveniles/sub-adults, but several adults were observed in both study areas (Figure 2.45d).

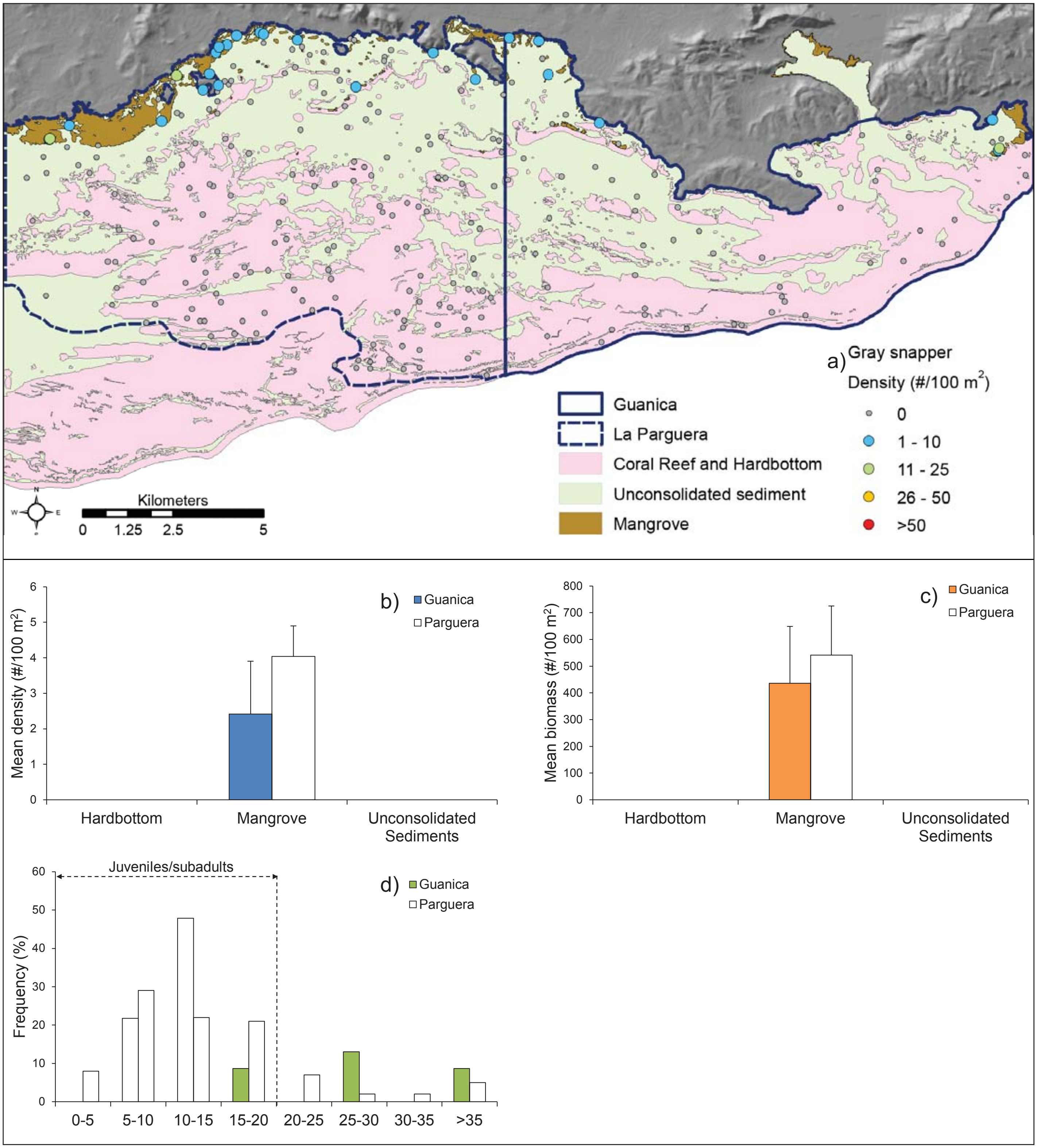

*Figure 2.45. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of gray snapper (*Lutjanus griseus*).*

Mean density of yellowtail snapper was highest on hardbottom, followed by unconsolidated sediments. Mean biomass was also highest on hardbottom. The species was absent from mangroves in Guánica (Figure 2.46a,b). The sites with the highest abundance and biomass were located in the La Parguera study area. Mean density was higher in the La Parguera study area for all three habitats. Results in the nonparametric test indicated a significant difference in density between the two study areas on hardbottom habitat, but not the other two habitats (Table 2.4). This may be due to the high variability exhibited in both study areas. The majority of individuals were juveniles/subadults, particularly those associated with unconsolidated sediments (Figure 2.46d). Thirteen percent of yellowtail snapper present on hardbottom were small-sized adults between 20 and 30 cm.

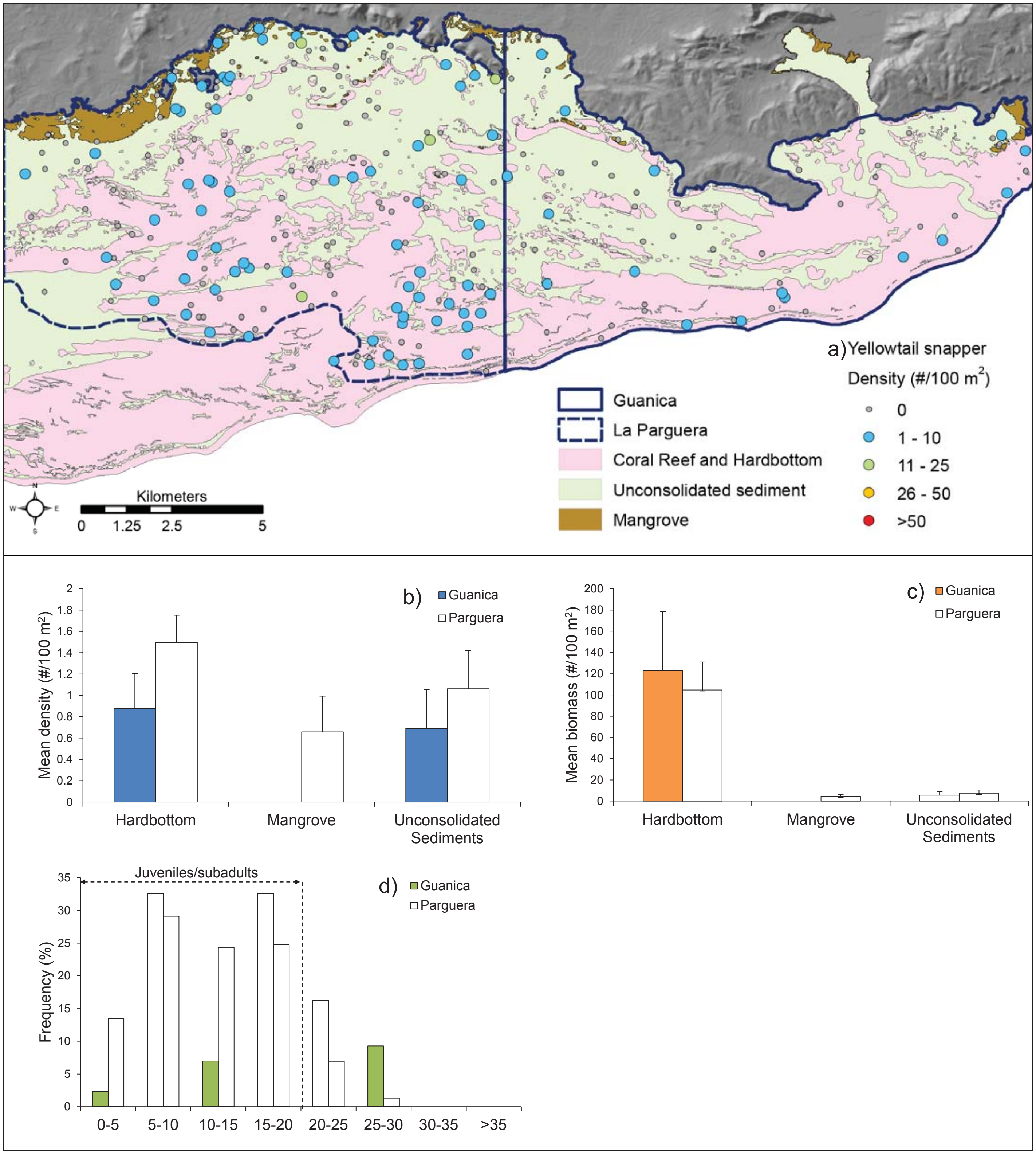

*Figure 2.46. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of yellowtail snapper (*Ocyurus chrysurus*).*

Grunts (Haemulidae)

Fishes in the grunt family were present in all habitat types. Mangrove habitat had the highest mean density in both study areas, while hardbottom and unconsolidated sediment habitats exhibited similar density levels (Figure 2.47b and Table 2.4). In the Guánica study area, biomass was highly variable on hardbottom habitat due to higher than average concentrations of grunt biomass at four survey sites near the shelf edge. The family was represented by 10 species (Appendix A,Tables A.1, A.2). French grunt (*Haemulon flavolineatum*), bluestriped grunt (*H. sciurus*), white grunt (*H. plumierri*) and tomtates (*H. aurolineatum*) were most frequently sighted and had the highest mean abundance and biomass of the Haemulid species. Porkfish (*Anisotremus virginicus*) and the remaining Haemulid species were observed less frequently (Appendix A, Tables A.1, A.2). Approximately one-third of observed grunts in the Guánica study area were small juveniles that could not be identified to the species level, compared to 56% of those in the La Parguera study area. These juveniles were present across all nearshore habitats but were particularly associated with mangroves and unconsolidated sediments (e.g., seagrass beds).

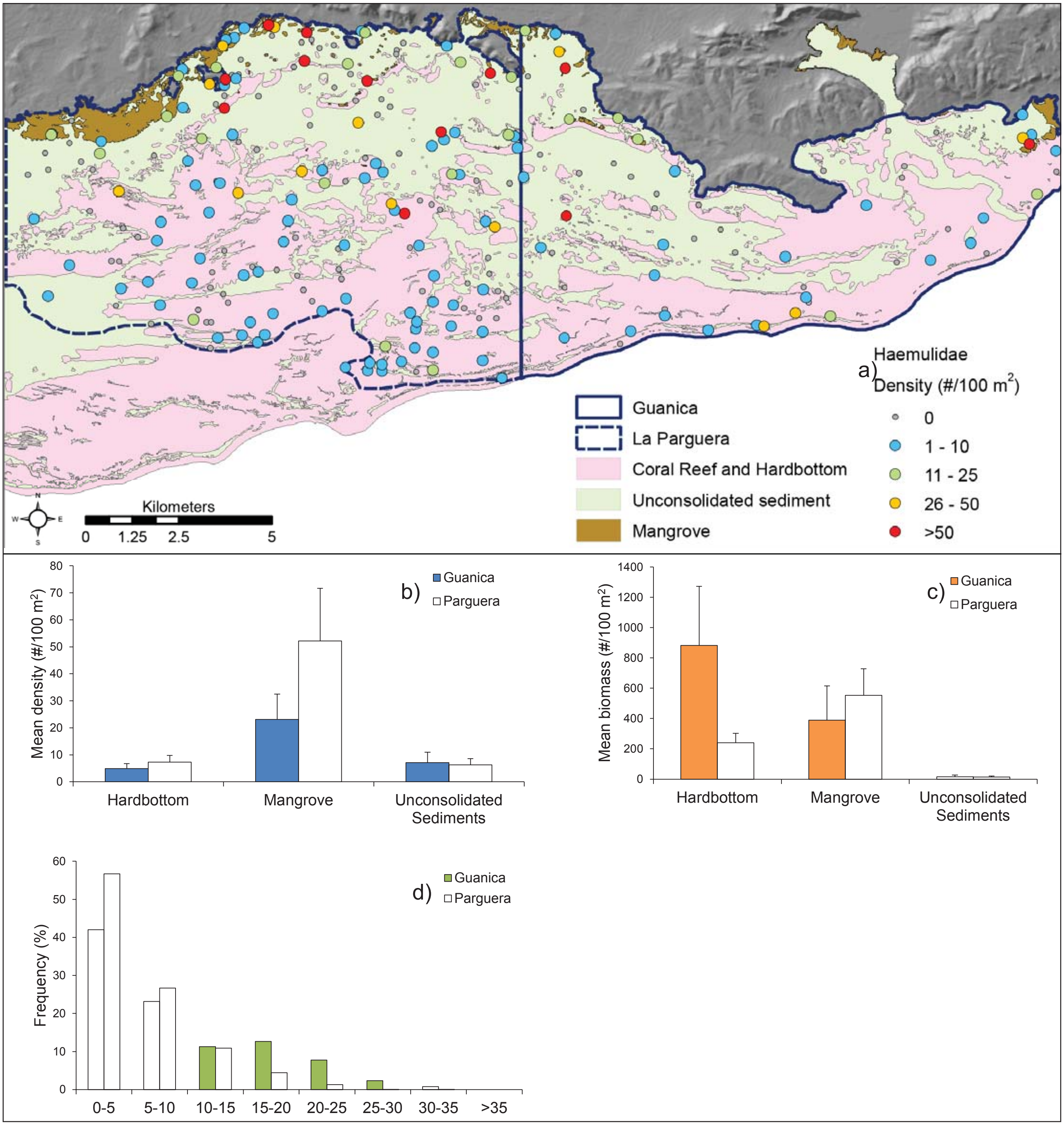

Figure 2.47. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of grunts (Family Haemulidae).

French grunt was present in 39% and 29% of sites within the Guánica and La Parguera study areas, respectively. The species was most commonly observed on hardbottom and mangrove habitats and were nearly absent in unconsolidated sediments across the study area (Figure 2.48). While mean density was highest in mangroves, they were highly variability and the majority of individuals were small juveniles, particularly within the Guánica study area. In contrast, a larger range of sizes were present on hardbottom habitat, although sub-adults and small adults were most common. The site with the highest abundance (355 juveniles) was located in mangrove habitat within the La Parguera study area, while the site with the highest biomass was located on hardbottom near the shelf edge in the Guánica study area.

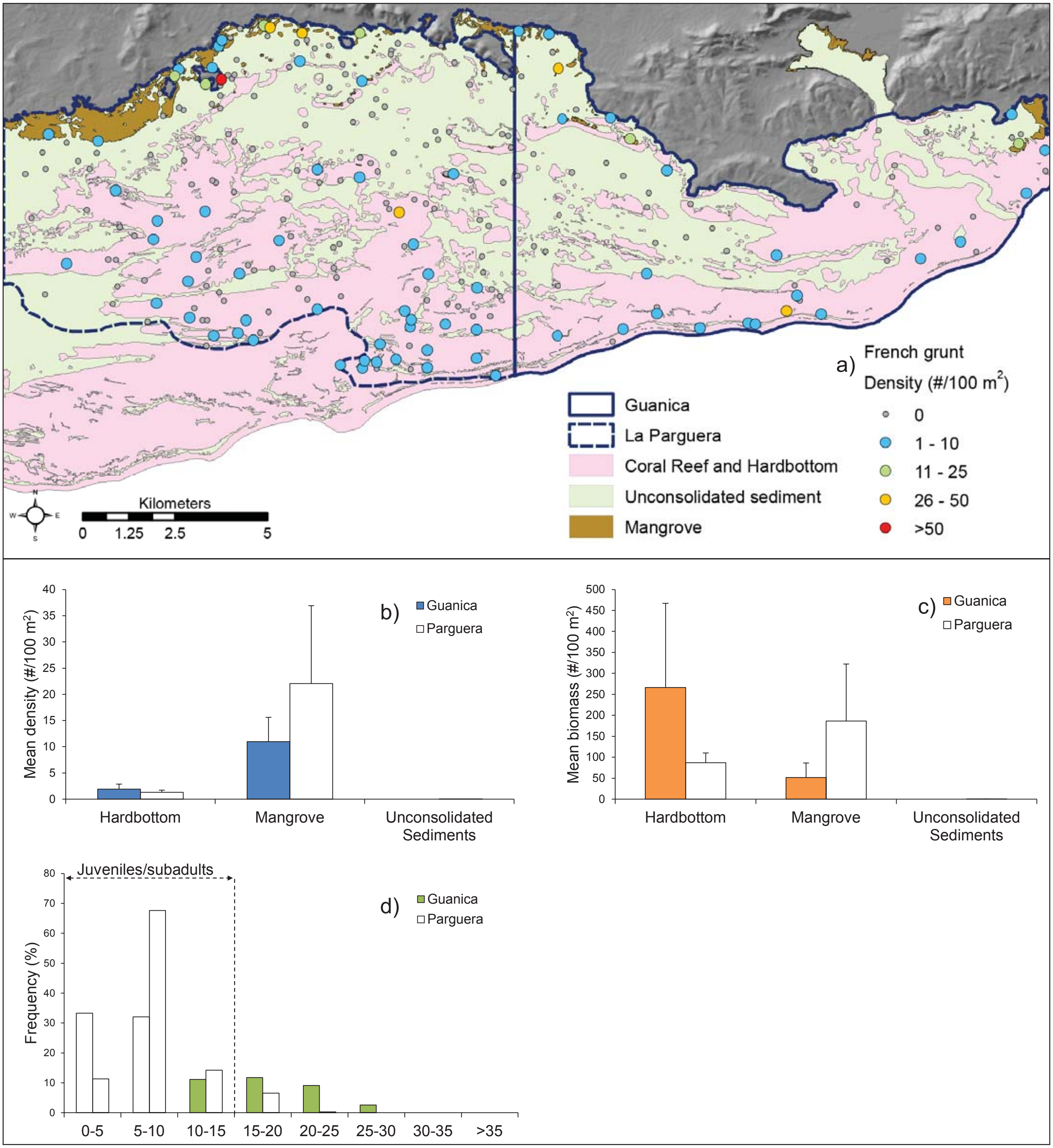

*Figure 2.48. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of French grunt (*Haemulon flavolineatum*).*

Bluestriped grunt was sighted in 20% and 18% of surveys within the Guánica and La Parguera study areas, respectively. The species was most abundant in mangrove habitat; similarly biomass was also highest in mangrove habitat within the La Parguera study area (Figure 2.49a-c). Within the Guánica study area, mean biomass was similar on hardbottom and mangrove and exhibited high spatial variability. While all size classes were present in mangroves, juveniles/sub-adults were most common. Hardbottom was typically inhabited by the larger size classes.

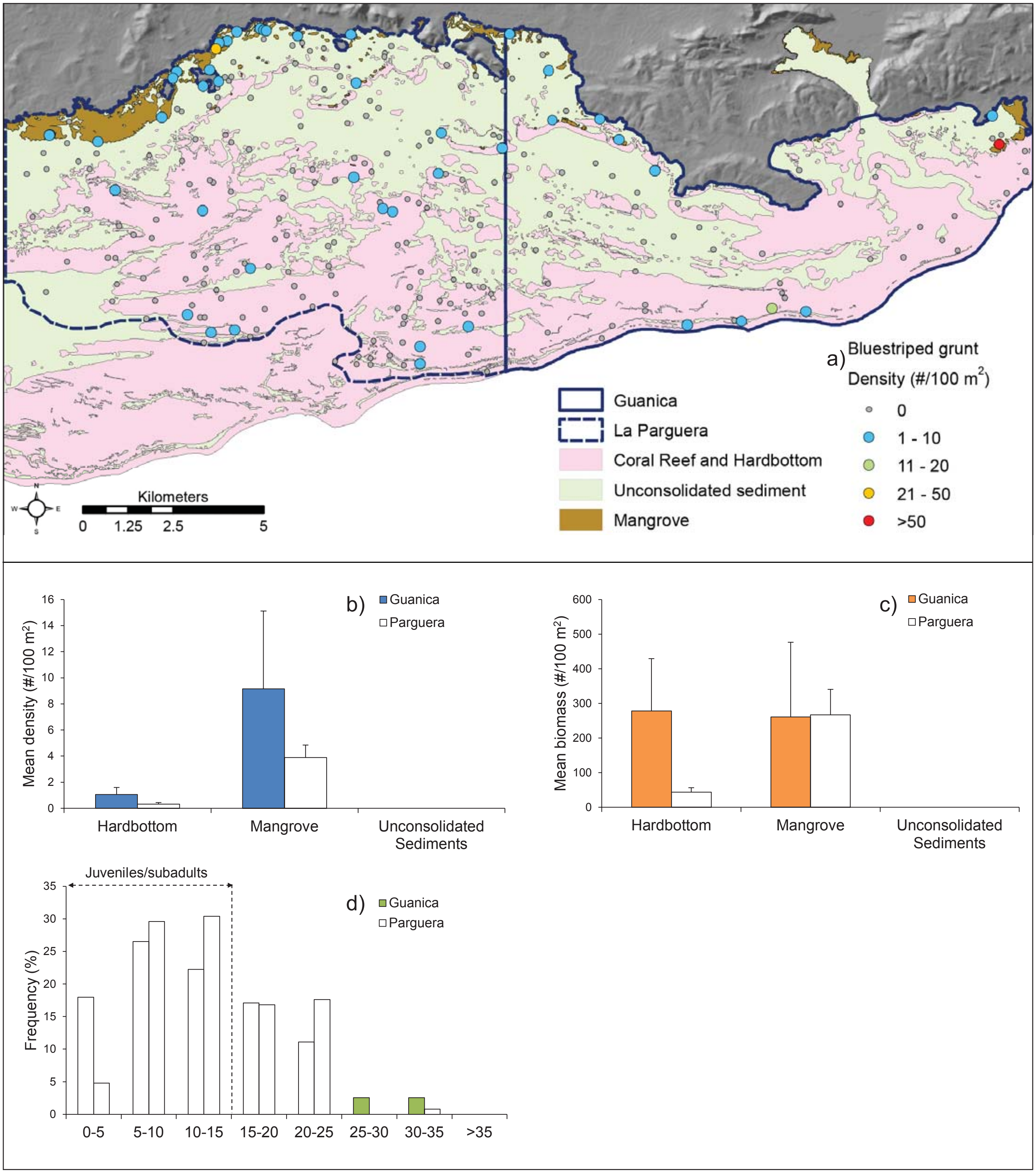

*Figure 2.49. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of bluestriped grunt (*Haemulon sciurus*).*

White grunt was present across all bottom types in the La Parguera study area and only on hardbottom and mangrove habitats within the Guánica study area (Figure 2.50). Small juveniles were most common in mangrove habitat, but in lower densities than the previously described grunt species. In the La Parguera study area, the majority of individuals were juveniles/subadults. In contrast, a larger proportion of observed fish, about 60% of the total, were adult-sized in the Guánica study area.

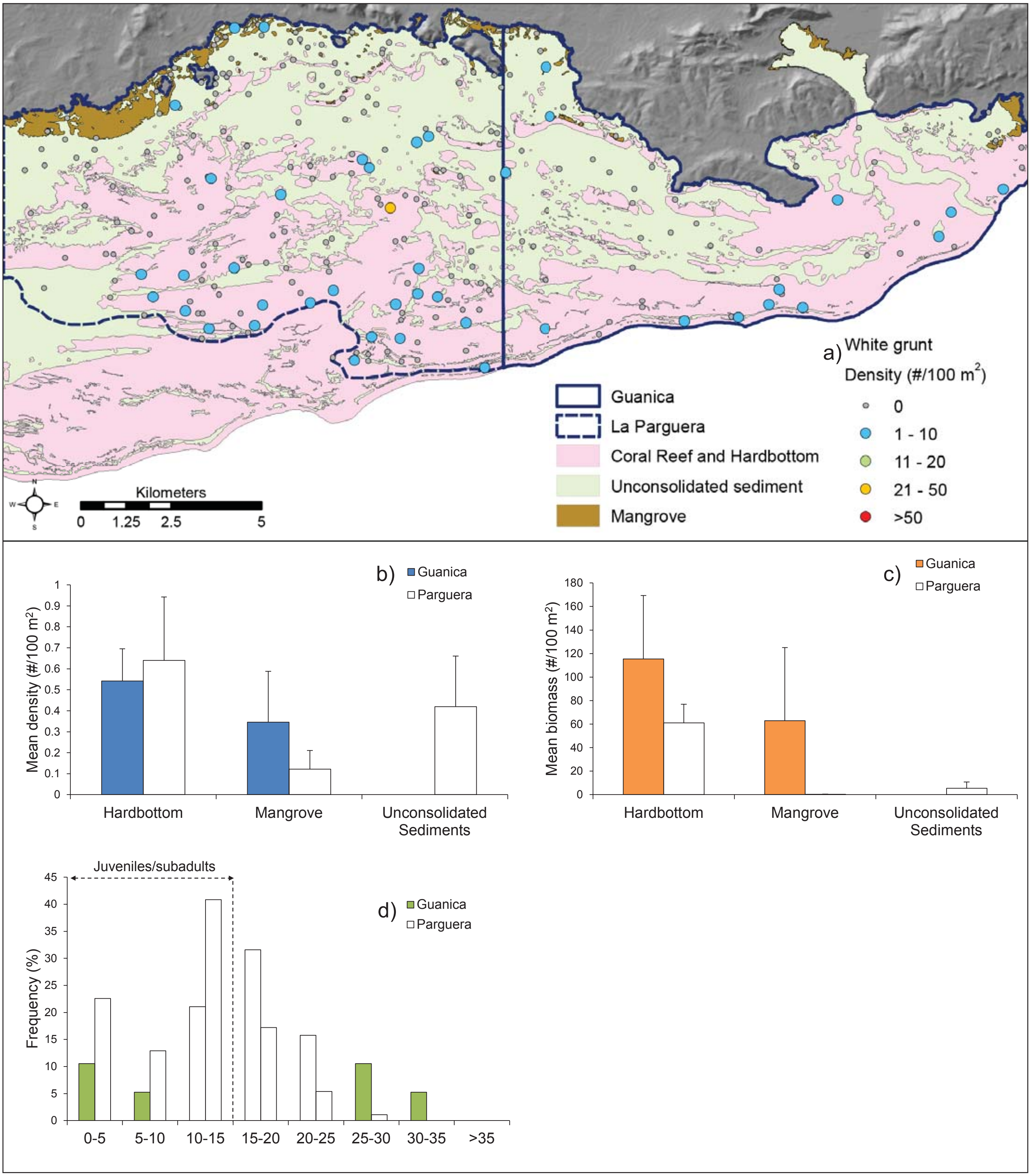

*Figure 2.50. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of white grunt (*Haemulon plumierii*).*

In the Guánica study area, tomate was only documented at three hardbottom sites near the shelf edge, whereas in the La Parguera study area the species was observed at multiple sites spanning the shelf and bottom types, including nearshore and lagoon areas. The species was generally observed at low densities and biomass, with the exception of two offshore hardbottom sites where 17 and 49 individuals were observed (Figure 2.51a-c). There was no significant difference in biomass or density between study areas for the three surveyed habitats (Table 2.4). The majority of observed individuals were sub- adults/small adults, with few small juveniles or large adults (Figure 2.51d).

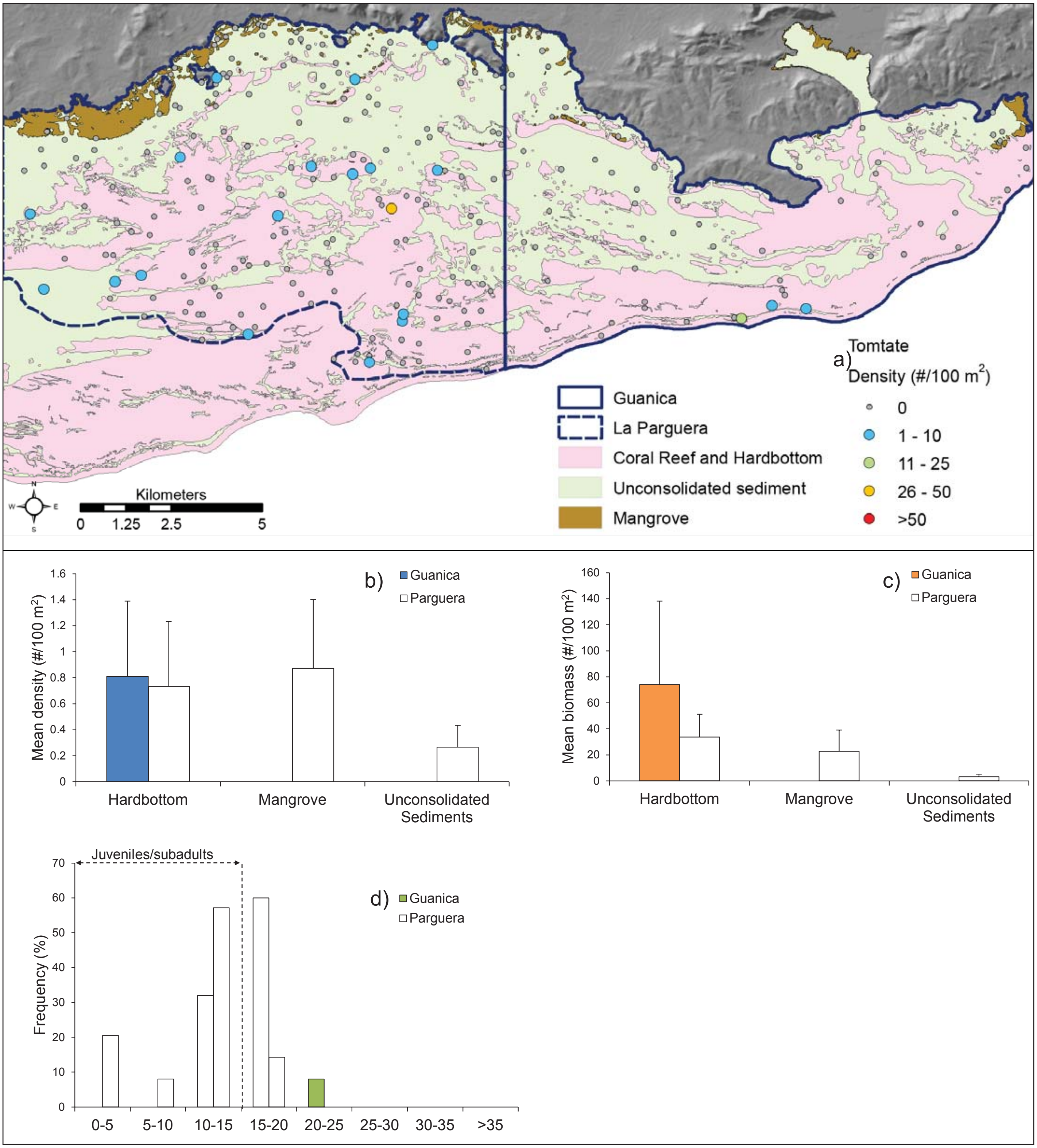

*Figure 2.51. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of tomtate (*Haemulon aurolineatum*).*

Surgeonfish (Acanthuridae)

Surgeonfish (acanthurids) were present in all three habitat types but most abundant in hardbottom habitats (Figure 2.52). In the Guánica study area in particular, fish in the family were more prevalent on the outer shelf than at hardbottom sites in the shallow lagoonal areas. Although sites with the highest abundance were observed in the La Parguera study area, mean density and biomass were not significantly different between the two regions (Table 2.4). Ocean surgeonfish (*Acanthurus bahianus*) and blue tang (*A. coeruleus*) were the most frequently observed species in both study areas, while doctorfish (*A. chirirgus*) were less common (Appendix A, Tables A.1, A.2).

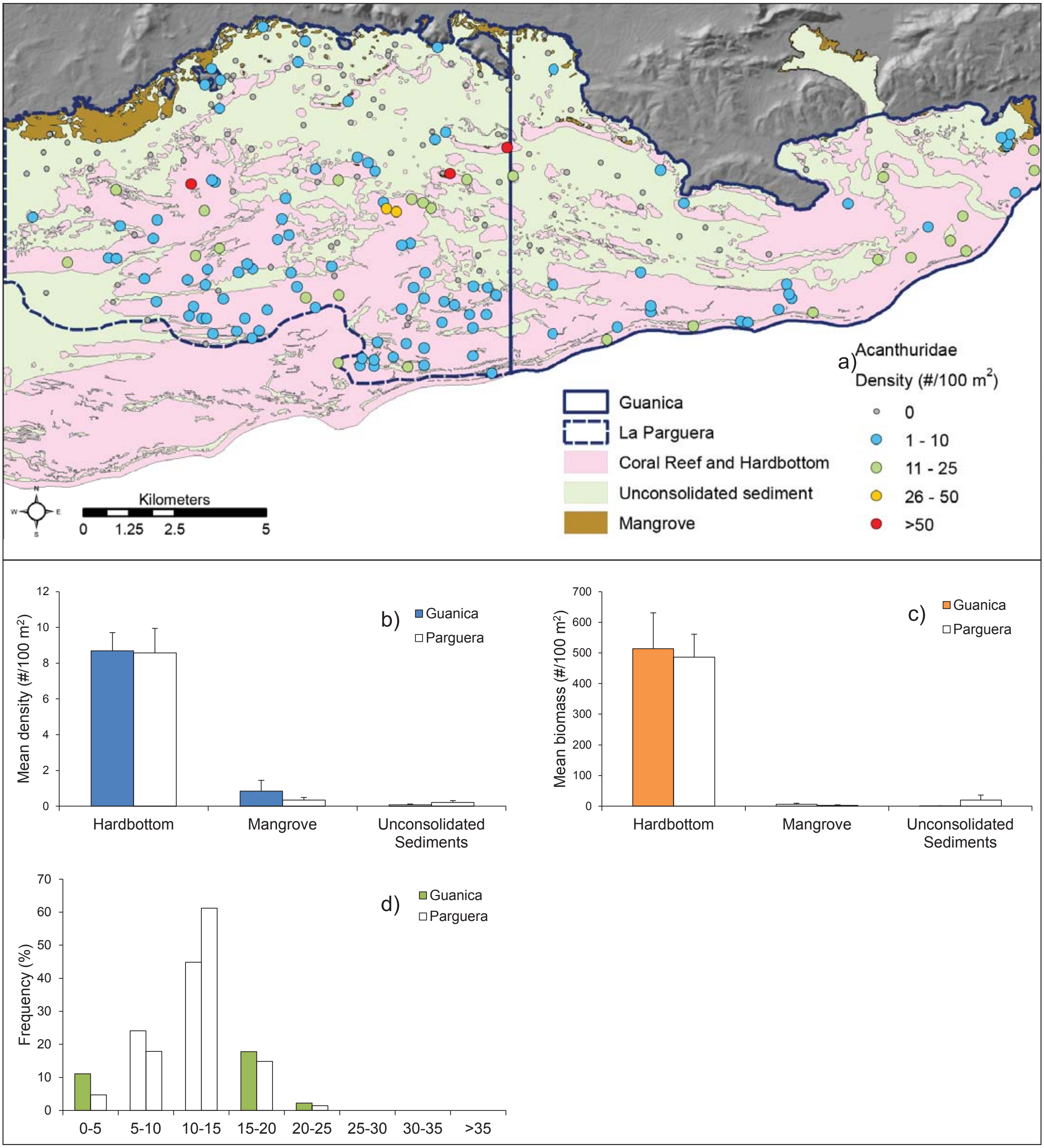

Figure 2.52. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of surgeonfishes (Family Acanthuridae).

Ocean surgeonfish were documented in 39% and 43% of surveys within the Guánica and La Parguera study areas, respectively. The species was observed across the study area, but species density and biomass were highest on hard-bottom habitat outside lagoonal areas (Figure 2.53a-c). The majority of individuals in both study areas were juveniles/sub-adults (Figure 2.53d).

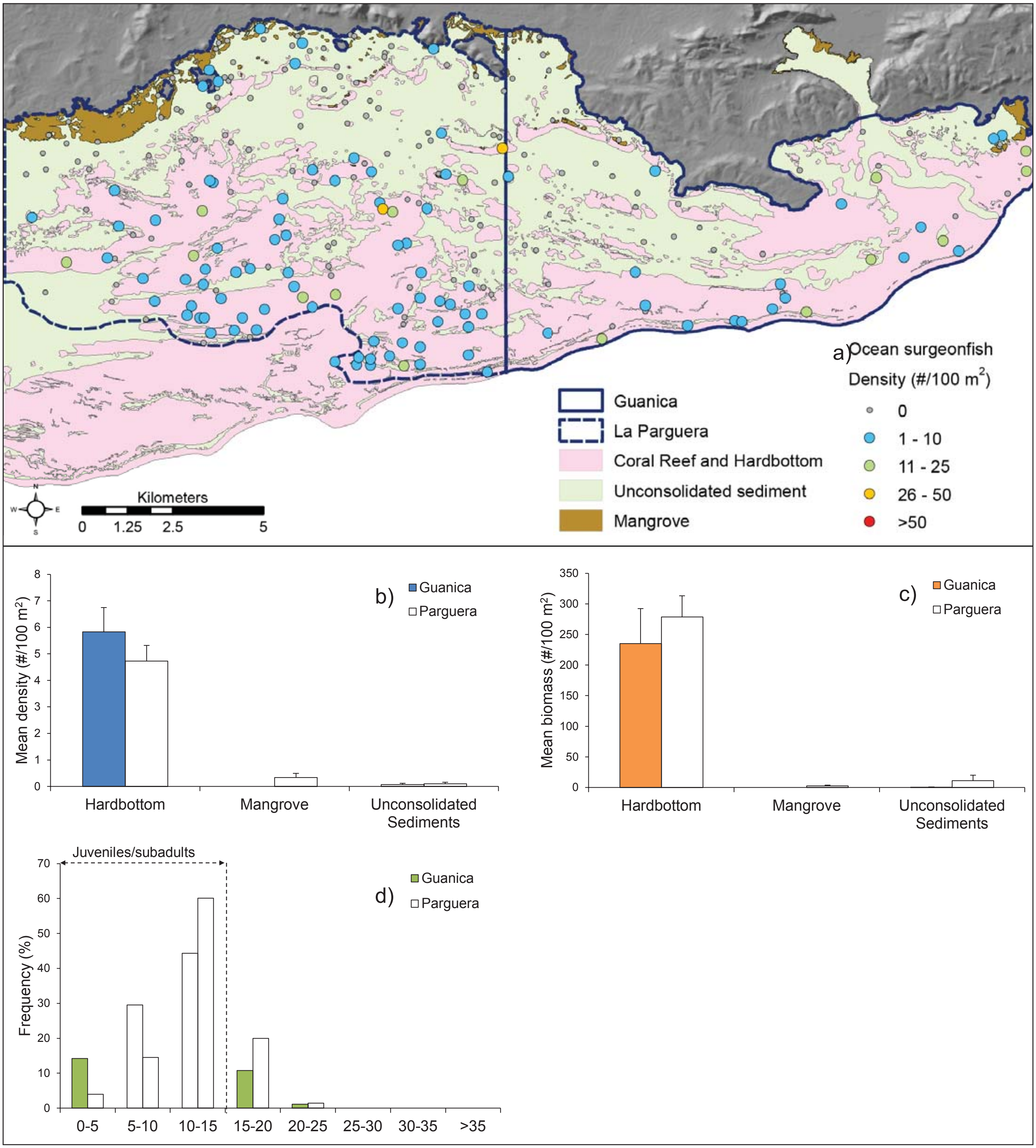

*Figure 2.53. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of ocean surgeonfish (*Acanthurus bahianus*).*

Blue tang occurred in 30% and 27% of surveys within the Guánica and La Parguera stuyd areas, respectively, and exhibited a similar distribution to ocean surgeonfish (Figure 2.54). The species was absent from mangrove habitat in the La Parguera study area and from unconsolidated sediment surveys in the Guánica study area. The majority of individuals were small adults (10-20 cm in total length), with fewer numbers of small juveniles or large adults (Figure 2.54d).

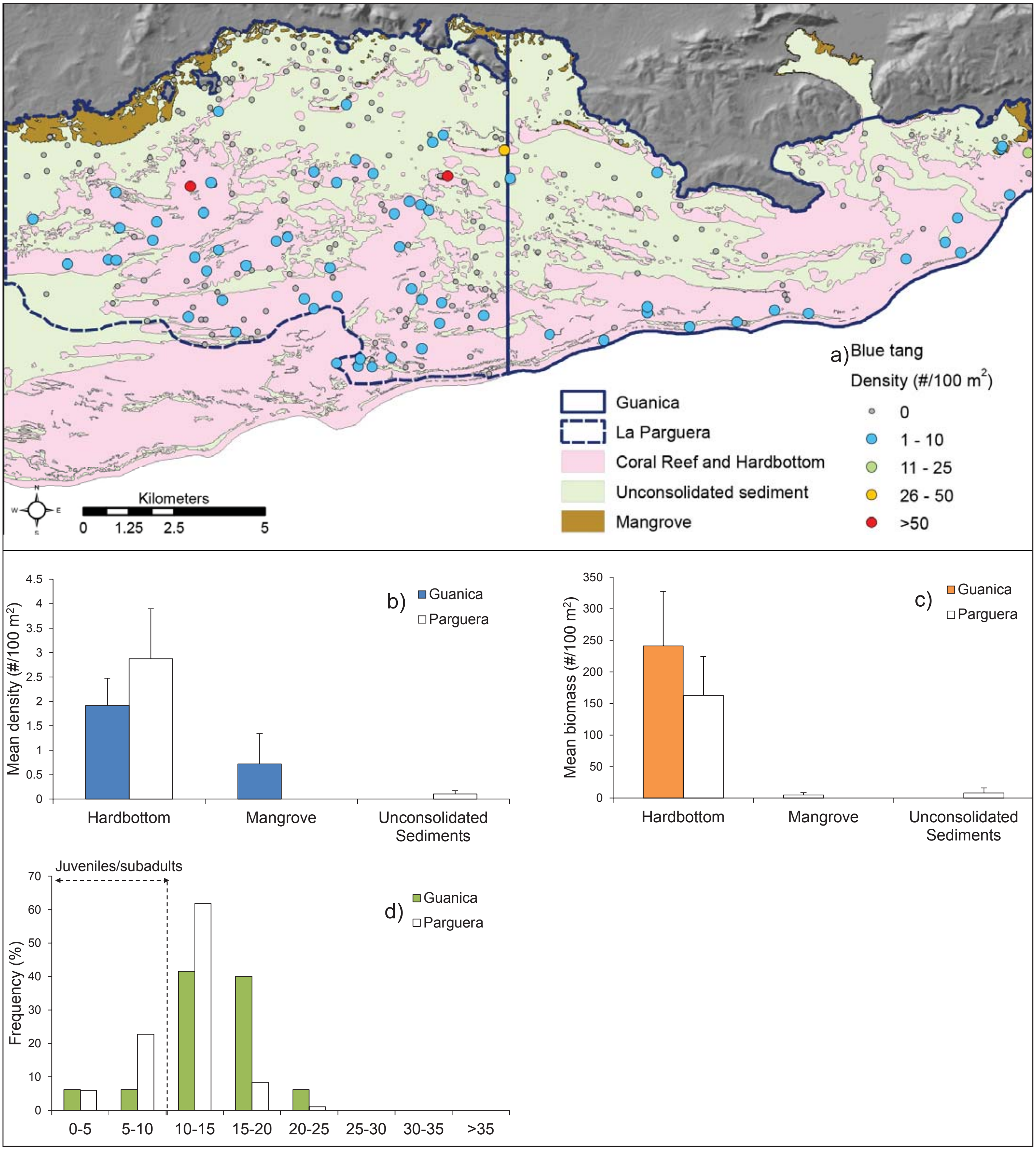

*Figure 2.54. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of blue tang (*Acanthurus coeruleus*).*

<u>Parrotfish (Scaridae)</u>

Parrotfishes (Family Scaridae) were a common component of the southwest Puerto Rico fish community. The family was represented by 11 species, nine of which were present in both study areas and two of which were only recorded in the La Parguera study area (Appendix A, Tables A.1, A.2). In both areas, the species with the highest site frequency, density and biomass were striped parrotfish (*Scarus iseri*), redband parrotfish (*Sparisoma aurofrenatum*), princess parrotfish (*Sc. taeniopterus*) and stoplight parrotfish (*Sp. viride*). Larger-bodied species, such as midnight parrotfish (*Sc. coeruleus*) and rainbow parrotfish (*Sc. guacamaia*), were absent from both study areas. Highest levels of density and biomass were generally observed on hardbottom habitat, with variable density in mangrove habitats (Figure 2.55a-c). Generally small juveniles were prevalent in mangroves (Figure 2.55d), hence biomass in this habitat was relatively low. On hardbottom, Scaridae biomass and density were significantly greater in La Parguera than the Guánica study area (Table 2.4). In the La Parguera study area, 42% of the hardbottom sites had a parrotfish density >25 individuals/100 m^2, compared to only 7% of sites within the Guánica study area. In La Parguera, these "high" density sites were distributed across the shelf, while in Guánica they were restricted to the shelf edge (Figure 2.55a).

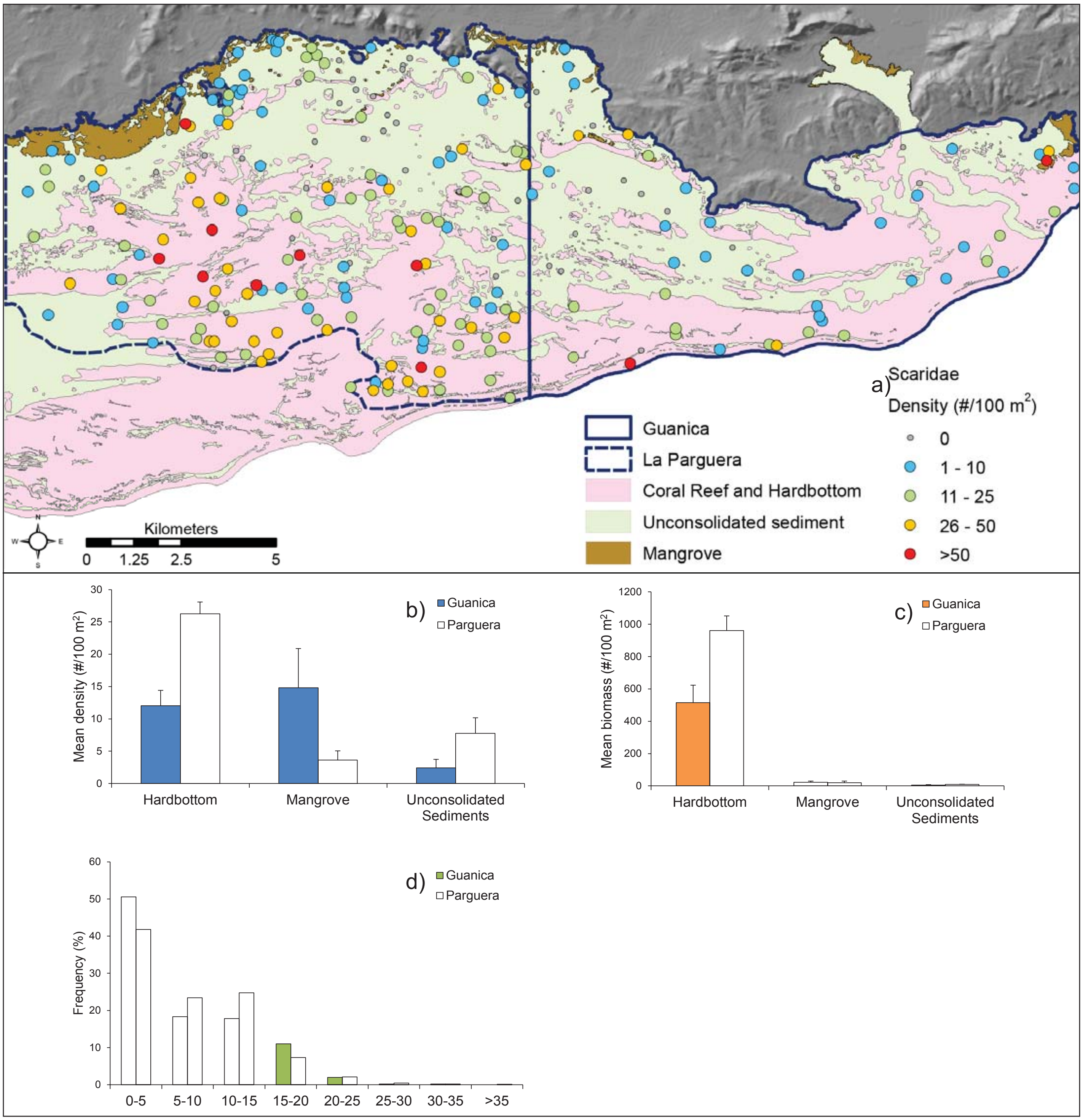

Figure 2.55. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of parrotfish (Family Scaridae).

Striped parrotfish were present in 26% and 49% of surveys within the Guánica and La Parguera study areas, respectively. On hardbottom habitats, the species exhibited significantly higher density and biomass in the La Parguera study area (Table 2.4). There was an opposite trend in mangrove habitat, where the species exhibited higher density and biomass in the Guánica study area, however these differences were not statistically significant (Table 2.4 and Figure 2.56b,c). This species was largely limited to offshore hardbottom and nearshore mangroves in the Guánica study area, while in La Parguera it was distributed across the mid-outer shelf, primarily in hardbottom habitats (Figure 2.56a). In Guánica, the majority of individuals (>80%) were small juveniles (<5 cm), most of which were observed in mangrove habitat (Figure 2.56d). In contrast, the size distribution in the La Parguera study area included greater proportions of larger juveniles and small adults.

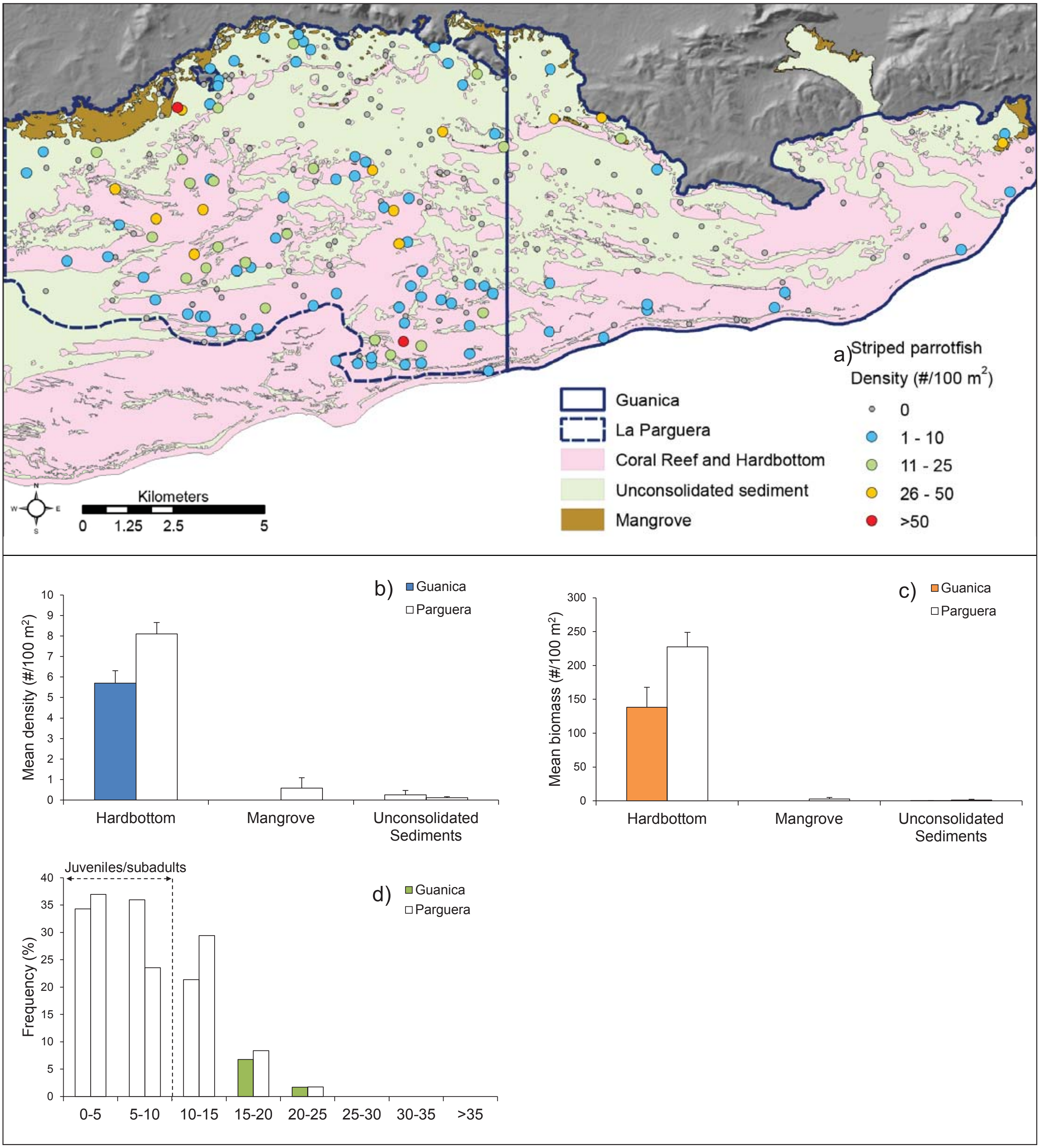

*Figure 2.56. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of striped parrotfish (*Scarus iseri*).*

Redband parrotfish were sighted in about 50% of survey transects in both study areas. However, similar to the previous species, density and biomass on hardbottom were significantly greater in the La Parguera study area compared to the Guánica study area (Table 2.4). Redband parrotfish were uncommon on unconsolidated sediment and mangrove habitats (Figure 2.57a,b). In both study areas, the size distribution was skewed towards the smaller size classes (0-10 cm total length) while 30% and 40% of individuals were small adults in the Guánica and La Parguera study areas, respectively (Figure 2.57d).

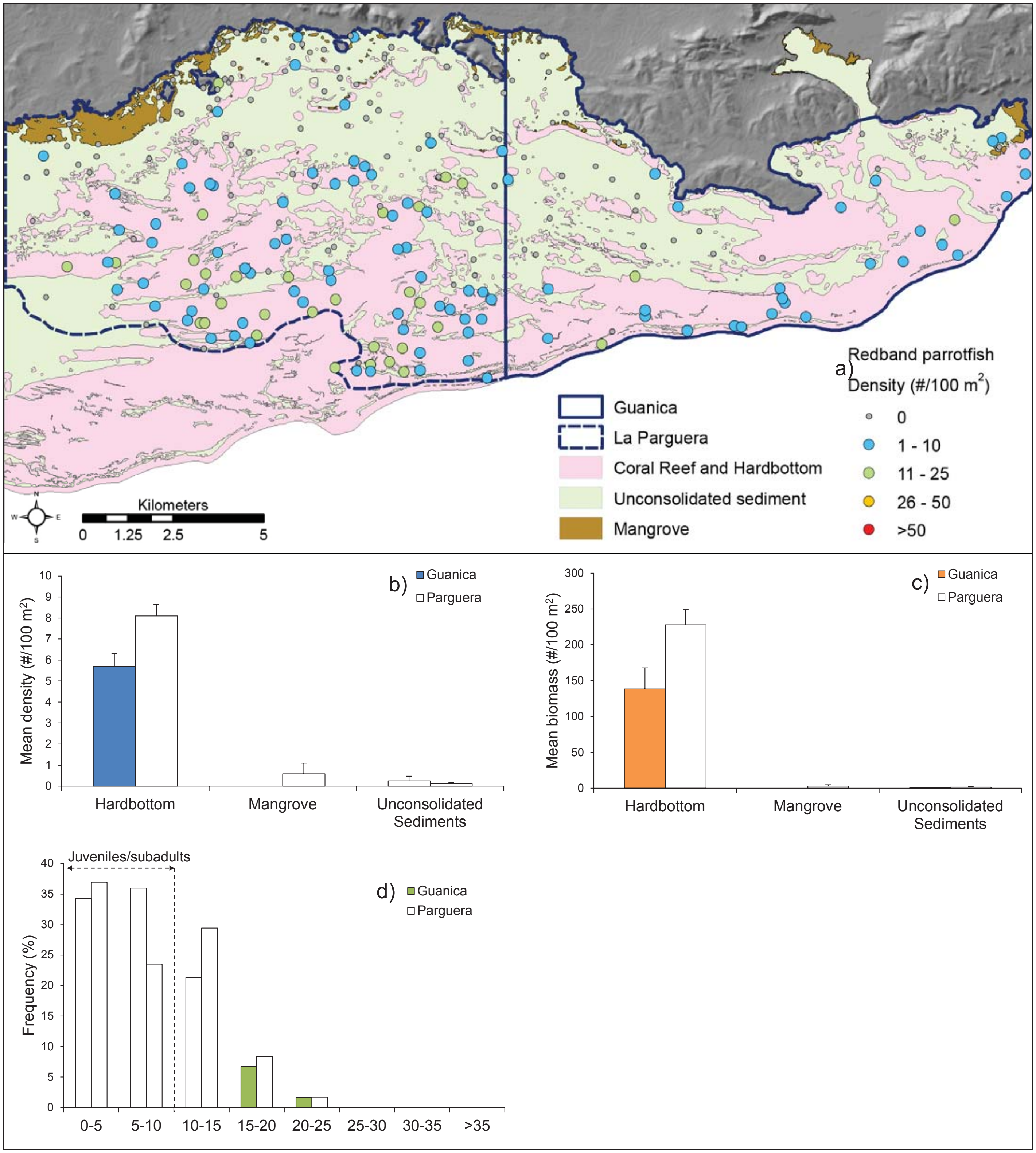

*Figure 2.57. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of redband parrotfish (*Sparisoma aurofrenatum*).*

Stoplight parrotfish were observed in all three habitats but exhibited highest abundance and biomass on hardbottom habitat (Figure 2.58a-c). Species density was greater on hardbottom within the La Parguera study area compared to Guánica, but the nonparametric tests indicated no statistical difference in density or biomass between the two study areas (Table 2.4). The size distribution was more heterogeneous than the previous two parrotfish species, with a mix of small juveniles-small adults (Figure 2.58d). In addition, three large adults (40 cm total length) were observed in the La Parguera study area.

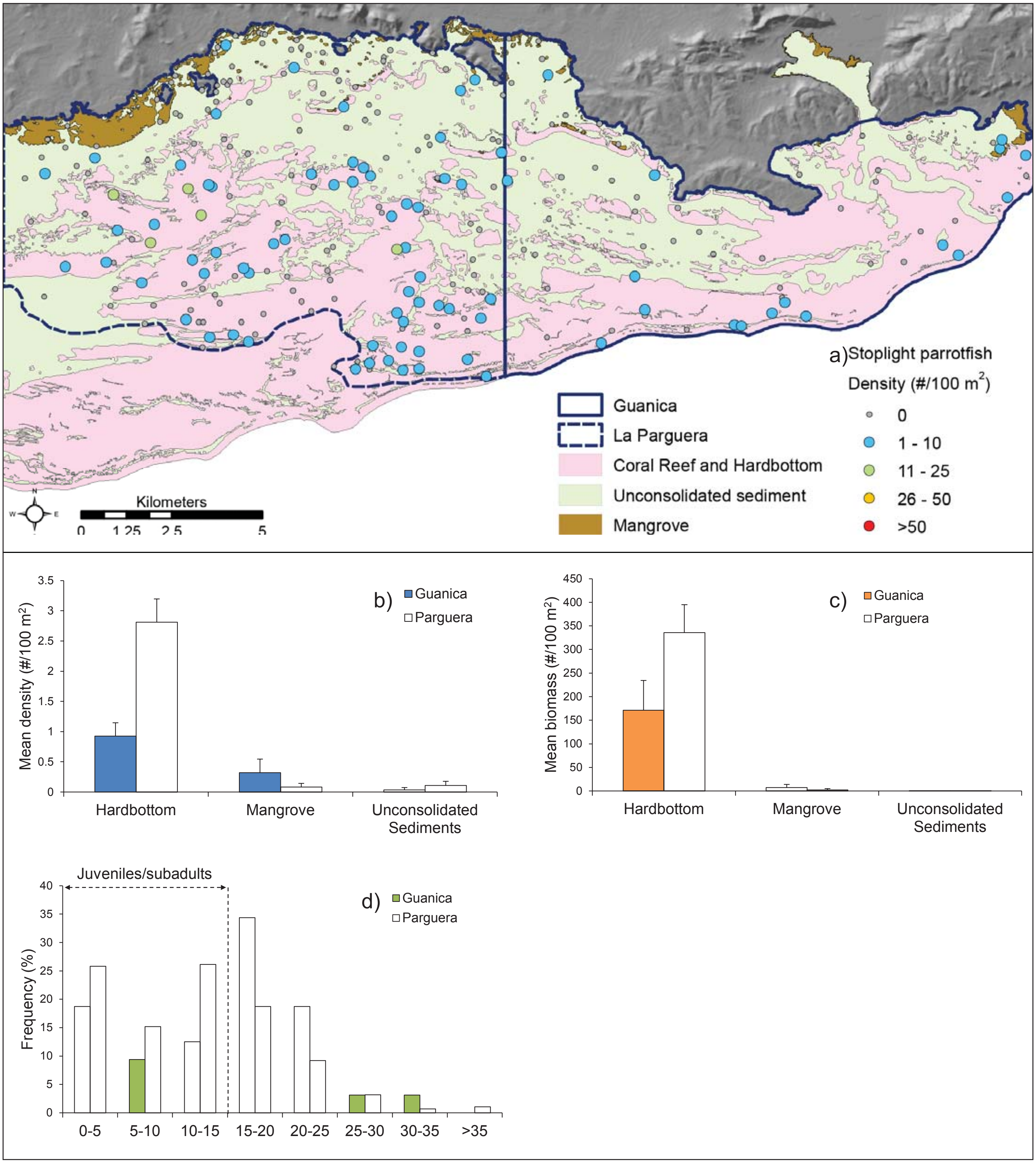

*Figure 2.58. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of stoplight parrotfish (*Sparisoma viride*).*

<u>Wrasses (Labridae)</u>

Wrasses were commonly sighted, occurring in 77% and 64% of surveys in the Guánica and La Parguera study areas, respectively, and nearly all hardbottom surveys. Although observed throughout the study area, sites with the highest densities tended to be located on the mid-outer shelf (Figure 2.59a). On hardbottom, there was a statistical difference in abundance and biomass between the two study areas (Table 2.4) with the Guánica study area exhibiting higher levels of abundance and biomass (Figure 2.59b,c). The family was represented by 13 species, 10 of which were observed in both study areas (Appendix A, Tables A.1, A.2). Two additional species were sighted only in the La Parguera study area and one was unique to the Guánica study area. Bluehead (*Thalassoma bifasciatum*) was the most abundant species in both study areas. Hogfish (*Lachnolaimus maximus*), an economically important species in Puerto Rico, were present in only one survey transect in the Guánica study area and a handful of survey transects in the La Parguera study area. The size distribution skewed towards the smaller size classes (Figure 2.59d) because small-bodied wrasse species were most abundant.

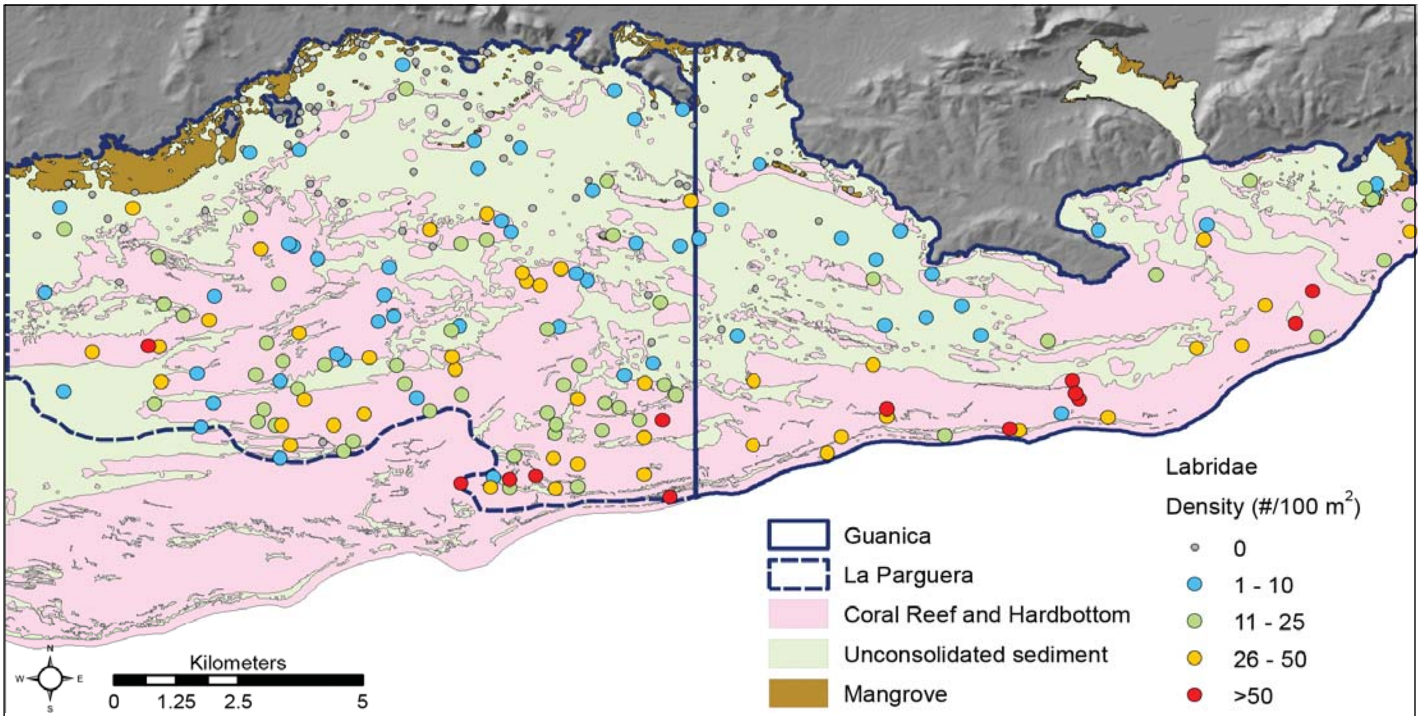

Figure 2.59. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of wrasses (Family Labridae).

Goatfishes (Mullidae)

Fishes of the Family Mullidae were present in [illegible] and [illegible] of survey transects within the Guánica and La Parguera study areas, respectively. The family was represented by two species: spotted goatfish (*Pseudupeneus maculatus*) and yellow goatfish (*Mulloidichthys martinicus*). In the Guánica study area, the family was only observed in hardbottom and generally at low numbers, with the exception of two offshore hardbottom surveys where larger groups of yellow goatfish were observed (Figure 2.60a). In the La Parguera study area, most occurrences of mullids also occurred in hardbottom in the mid/outer shelf. The size distribution varied slightly between the two study areas; the majority of individuals in the Guánica study area were within the [illegible] cm size class, while the majority of individuals observed in the La Parguera study area were [illegible] cm in length (Figure 2.60d).

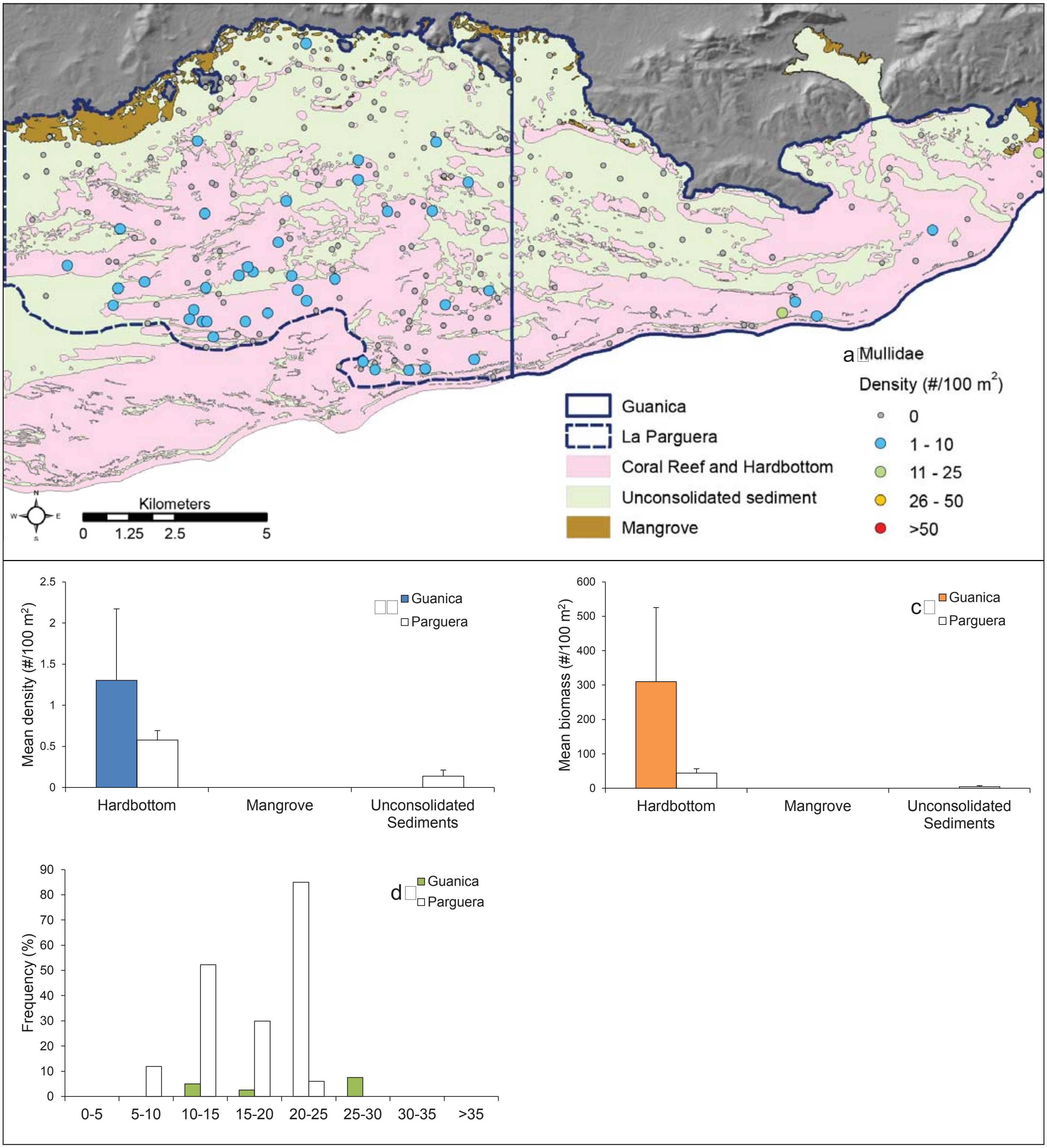

Figure 2.60. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of goatfish (Family Mullidae).

Damselfishes (Pomacentridae)

Damselfishes were present across all bottom types, occurring in 61% and 76% of surveys within the Guánica and La Parguera study areas, respectively. Highest densities and biomass were observed on hardbottom, with intermediate levels in mangrove habitat (Figure 2.61a-c). The family was represented by 11 species, with nine common to both study areas and two observed only in the La Parguera study area (Appendix A ,Tables A.1, A.2). Frequently observed species included the bicolor damselfish (*Stegastes partitus*) and beaugregory (*S. leucostictus*). On hardbottom, damselfishes were significantly more abundant within the Guánica study area (Table 2.4). This was likely due to large numbers of bicolor damselfish at several locations on the outer shelf in the Guánica study area. However, biomass did not vary significantly between the two study regions.

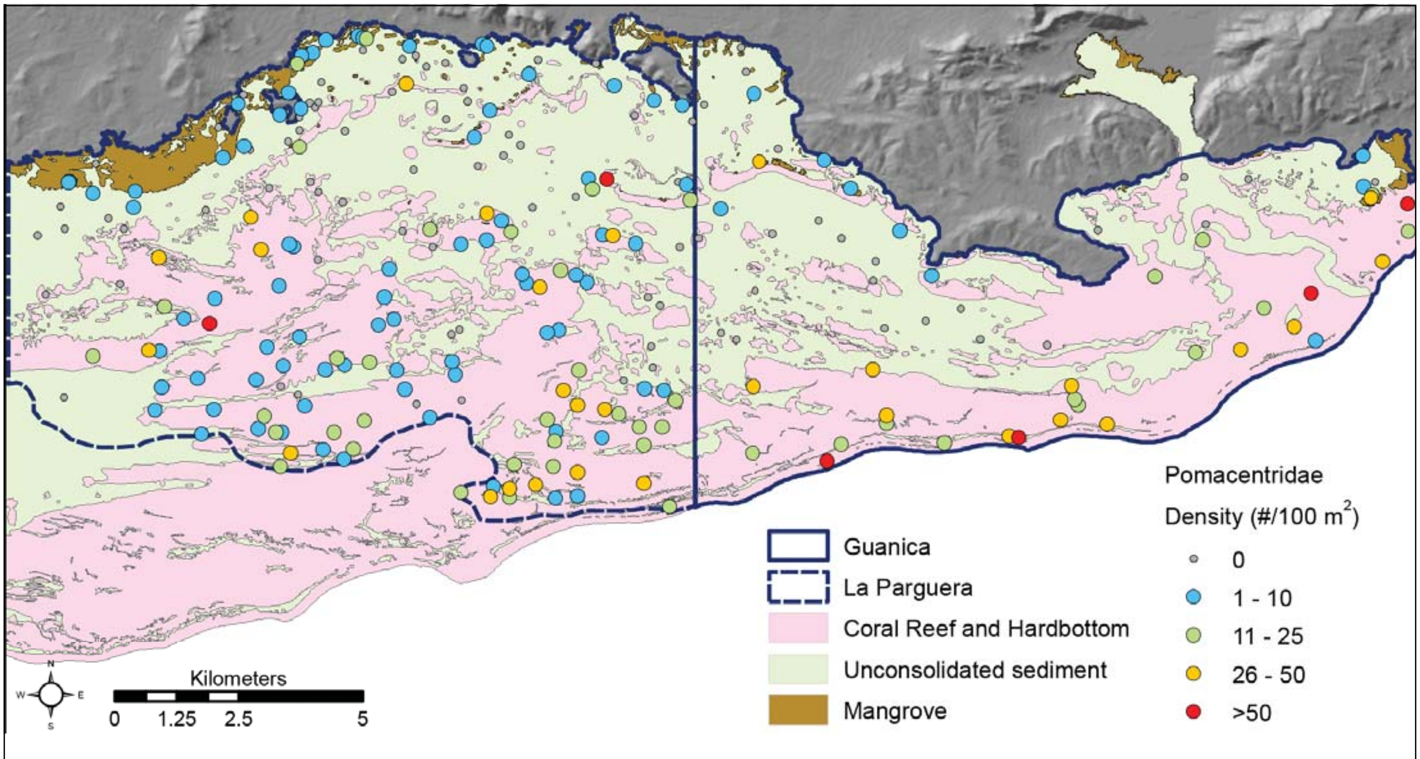

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Figure 2.61. a) Spatial distribution, b) mean (±SE) density by habitat, c) mean (±SE) biomass by habitat and d) size (cm) frequency of damselfishes (Family Pomacentridae).

CONCLUSIONS

The results of this study provide a baseline characterization of the fish and benthic communities of Guánica Bay to assess the efficacy of the watershed restoration efforts. Comparisons were made in the metrics between the Guánica and La Parguera study areas to investigate potential differences in the adjacent regions. For the purposes of this study, the region of southwest Puerto Rico was treated as two study areas; however the Guánica and La Parguera study areas are effectively part of the same continuous system. The prevailing surface currents are driven by the trade winds and flow westward, meaning that effluent from Guánica Bay flows downstream towards the La Parguera region. In addition, the new benthic habitat map of southwest Puerto Rico includes a region south of the current La Parguera study area that was previously unmapped (Bauer et al., 2012). This area contains 28 km^2 of coral reef and hardbottom extending out to the shelf edge, about one-third of which is high rugosity spur and groove structure. It is recommended that the entire system, including this newly mapped area, continue to be monitored in future sampling efforts.

While most metrics observed in this study were similar between the two regions, there is some evidence that the Guánica study area is more degraded than La Parguera. While coral cover was generally low across the entire study region, it was significantly higher in the La Parguera study area. Gorgonian and seagrass cover were similarly higher in the La Parguera study area. Differences in benthic habitat composition may be a factor influencing these differences. However, previous work also indicates that levels of some contaminants exhibit a downstream gradient with higher levels near the mouth of Guánica Bay (see Chapter 3 in this document; Pait et al., 2009). Linkages between chemical contaminants and coral health are poorly understood (Pait et al., 2009) and understanding these cause and effect relationships are critical to effective management and restoration practices.

Recent local studies highlight changes in the Guánica marine benthic community over time. Garcia-Sais et al. (2001) conducted a baseline assessment of the marine fauna at three locations southeast of Guánica Bay, moving west to east: Cayo Coral, Cayo de Cana Gorda and Punta Ventana. The community was dominated by gorgonians and soft corals, with high hard coral cover, averaging ≥25% at all three locations (Garcia-Sais et al., 2001). While these reefs were selected for a specific study purpose and may not be representative of the region as a whole, they recorded higher coral cover than was observed at any site within our Guánica study area (maximum 18.5%). Since the initial baseline survey conducted by Garcia et al. (2001), long-term monitoring has been conducted at their western most site, Cayo Coral. Percent hard coral cover decreased from a mean of 25.3% in 1999 to 8.9% in 2008, for an overall reduction of ~65% (Garcia-Sais et al., 2008b). In a subsequent survey, live coral cover had slightly increased to 10.0% (Garcia-Sais et al., 2009). Boulder star coral in particular showed marketed decline over this time period.

As mentioned previously, while this is the first time that the Caribbean Coral Reef Ecosystem Monitoring Program study has extended as far east as Guánica Bay, the La Parguera region has been monitored one to two times/year since 2001. Pittman et al. (2010) conducted an analysis of fish and benthic data from La Parguera spanning from 2001-2007. This study noted a decrease in live scleractinian coral cover in comparison to the long-term average in Pittman et al. (2010). Coral cover averaged 5.3 ± 0.3% over the 2001-2007 time period, and exhibited a decreasing trend over time. By 2007, mean coral estimates were <5% cover (Pittman et al., 2010). Our results indicate that coral cover has continued to decline within the La Parguera region (4.0 ± 0.4% in the current study's La Parguera study area from 2008-2010) and is even lower in the Guánica study area (2.2 ± 0.4%).

According to Morelock et al. (2001), the Guánica Bay region historically has lower coral cover compared to La Parguera. Morelock et al. (2001) used photo transects to monitor coral cover along the southwest reefs of Puerto Rico, including Guánica, over a 20-year period starting in the late 1970s. They reported mean maximum coral cover at <10% on the Guánica shelf reefs and ~20% on the shelf edge reefs, with lowest cover present at Cayo Cana Gorda near the mouth of Guánica Bay. The data also showed declining trends in coral cover within the region, particularly from 1988-1999 due to increased sediment and nutrient loading from the bay. Warne et al. (2005) estimated the sediment discharge to the bay at 1,000-4,300 m tons/km^2 for the south coast of Puerto Rico, making the Guánica Bay/Rio Loco watershed one of the highest sources of sediment discharge around the island and the site of degraded reefs.

A recent meta-analysis indicated that live coral cover in the Caribbean has declined by 80% over the last three decades (Gardner et al., 2003) which has subsequently led to a "flattening," or decline in rugosity, of reefs over time (Alvarez-Filip et al., 2009). Elkhorn and staghorn coral, which were formerly dominant reef builders in shallow and intermediate forereef communities, respectively, have been devastated by white-band disease (Aronson and Precht, 2001). In Puerto Rico, recovery from the initial 1980s disease outbreak has been limited and Acroporid populations have declined significantly at locations island-wide where they were formerly abundant (Weil et al., 2002). Similarly, Acroporid species were sighted infrequently in this study.

Boulder star coral, another major reef-building coral, has also experienced widespread declines (Edmonds and Elahi, 2007), while the relative abundance of smaller, "weedy" species such as mustard hill coral has increased (Green et al. 2008). While the boulder star coral complex was the number one and number four ranked species in terms of percent cover in the Guánica and La Parguera study areas, respectively, mean cover of this species was low, especially in the Guánica study area. Mustard hill coral ranked second in terms of percent cover in both study areas. Puerto Rican reefs also experienced additional declines in coral cover following a 2005 bleaching event (García-Sais et al., 2008a).

Comparisons across studies are confounded by sampling strategies and make it difficult to identify true environmental changes. Using data collected by the NCCOS Biogeography Branch, including the aforementioned Pittman et al. (2010), allows for more accurate regional comparisons on benthic cover over time due to the standardized survey methodology. Since 2001, the NCCOS Biogeography Branch has regularly monitored habitat and fish communities using the same survey methodology in other U.S. Caribbean locations, including the Buck Island Reef National Monument in St. Croix, U.S. Virgin Islands (Pittman et al., 2008), Vieques, Puerto Rico (Bauer and Kendall, 2010) and the Jobos Bay region (Bauer et al., 2011). Comparatively, reef and hardbottom benthic communities in the Guánica Bay have lower mean coral cover than other regions of Puerto Rico and the U.S. Virgin Islands (2.2%). Similar to the Pittman et al. (2010) assessment in La Parguera, hard coral cover in St. Croix averaged 5.6 ± 0.5% over 2001-2006 (Pittman et al., 2008). In Vieques, coral cover from a 2007 survey averaged 3.4 ± 0.5% (Bauer and Kendall, 2010), while in the Jobos Bay 2009 survey, coral cover averaged 6.5 (±1.2)% (Bauer et al., 2011). At all of these locations, turf algae and macroalgae were the dominant benthic cover types on reef and hardbottom.

Both live coral cover and gorgonian cover were significantly higher in the La Parguera study area compared to Guánica, and both metrics were positively correlated with distance from the mouth of Guánica Bay. While correlation alone does not imply causation, more work is warranted to investigate whether there is a relationship between watershed variables and benthic cover metrics. In Palau, coral metrics (coral cover, richness and density) were positively related to distance from embayments and negatively related to increasing suspended solids (Golbuu et al., 2011).

Table 2.5. Area summary of detailed geomorphological structure within the Guánica and La Parguera study areas from the new southwest Puerto Rico benthic habitat map (Bauer et al., 2012).

Detailed Geomorphological Structure	Guánica Study Area		La Parguera Study Area	
	Area (km²)	Percent of Area	Area (km²)	Percent of Area
Aggregate Reef	5	6.7	13	12.4
Aggregated Patch Reefs	2	2.6	7	6.1
Individual Patch Reef	1	1.1	1	1.0
Mud	6	7.8	24	22.2
Pavement	23	30.7	7	6.5
Pavement with Sand Channels	5	6.0	16	14.7
Reef Rubble	2	2.5	2	2.1
Rock/Boulder	0.1	0.1	0.01	0.0
Sand	25	33.1	32	29.5
Sand with Scattered Coral and Rock	6	7.9	4	3.7
Spur and Groove	1	1.5	2	1.8
Total	**76**	**100**	**108**	**100**

Fish communities were similar between the two study areas, with a few exceptions. On hardbottom habitats, parrotfish were significantly more abundant in the La Parguera study area, while wrasses and damselfish were significantly more abundant in the Guánica study area. Parrotfish abundance and biomass were particularly low in areas south and east of Guánica Bay. The reasons behind this pattern are unknown but could be influenced by a number of factors, including species competition, differences in habitat type, and local fishing pressure. In the Guánica study area, pavement is the dominant hardbottom type, comprising 30.7% of the total area and three-fifths of the total hardbottom (Table 2.5). Higher complexity aggregate reef, in contrast, comprised only 13% of the total hardbottom within the Guánica study area. In contrast, within the La Parguera study area, high rugosity reef types (aggregate reef, spur and groove, patch reefs) altogether comprised ~48% of the total hardbottom. Density of total Scaridae was positively correlated with fine-scale rugosity measurements from this study (Spearman's Rho = 0.47, $p<0.0001$). Previous studies have also shown positive relationships between topographic complexity and parrotfish metrics, including overall parrotfish biomass (Pittman et al., 2009), *Scarus iserti* abundance (Mumby and Wabnitz, 2002) and *Sc. taeniopterus* distribution (Pittman and Brown, 2011). However, despite the differences in composition of detailed hardbottom structure, mean fine-scale rugosity was similar on hardbottom sites between the two study areas. Further data collection may elucidate whether the patterns observed here are persistent over time or were an anomaly.

Pittman et al. (2010) provides a summary and analysis of the fish community data in La Parguera from 2001-2007. Due to the similarity in the two study areas, interannual trends observed in the La Parguera study area may be indicative of patterns in the Guánica region. While overall grouper biomass was significantly lower in 2007 compared to 2001, other metrics, including species richness, total biomass, total grouper abundance, graysby abundance and biomass, and parrotfish abundance and biomass increased over the same time period (Pittman et al., 2010). The increase in small-bodied groupers such as graysby were thought to be influenced by the decline of larger bodied grouper species (Pittman et al., 2010), as in Stallings (2008). Similar to Pittman et al. (2010), in this study the dominant Serranid species included graysby and coney, and a few red hind and rock hind.

However, as Pittman et al. (2010) note, it is likely that the most significant ecosystem changes occurred prior to the start of the monitoring in 2001. The U.S. Caribbean has been subject to region-wide stresses that have affected the wider Caribbean in the last few decades, including a widespread die-off of long-spined urchin in the 1980s, mass Acropora species mortality due to white band disease, coral bleaching, overfishing, and tropical cyclones. Caribbean region-wide declines in fish abundance have been documented (Paddack et al., 2009). Hence, while one of our objectives is to conduct a "baseline" assessment of the Guánica Bay region, it is important to note that this is relative to current conditions. The lack of historical quantitative data leads to a "shifting baseline" away from "pristine" conditions over time (Pauly, 1995; Jackson et al., 2001).

This project represents a unique opportunity to assess the effect of restoration activities on a coral reef ecosystem. Similar efforts that also seek to reduce degradation of reef environments are underway in Hawaii (Conservation Reserve Enhancement Program, http://www.fsa.usda.gov/) and Micronesia (Richmond et al., 2007). The proposed restoration activities, particularly the restoration of the Guánica lagoon, have a common goal of reducing anthropogenic inputs to the marine environment. Hence, it is likely that the most immediate benefits of restoration will be reduced sediment and nutrient loads in the upper reaches of the watershed closest to the sediment source, followed by reduced loadings into the bay and surrounding environment.

While there is little in the literature that demonstrates the effects of restoration efforts on a reef ecosystem, it is possible that the reduction in stressor inputs will have a positive impact on coral reef health. For example, one potential effect is that reduced nutrient inputs could lead to a decline in cover of algal groups. Both macroalgae and cyanobacteria have been shown to inhibit coral recruitment (Kuffner et al., 2006), so reduction of these groups could, in theory, improve coral recruitment over time. Sediment runoff from the terrestrial environment, which is often accompanied by nutrients and a variety of land-based contaminants, has been found to have several negative impacts on hard corals, including direct smothering and light reduction due to turbidity. This can lead to decreased photosynthesis, growth, reproduction and recruitment rates, although the effects vary among coral species, sediment types, and environmental conditions (Rogers, 1990; Fabricius, 2005). A coral's ability to compensate for elevated turbidity is related to water depth such that in turbid environments, coral growth is restricted to shallower water (Yentsch et al., 2002). Although more research has been conducted regarding hard corals' sensitivity to sedimentation, octocorals are also sensitive to sediments (Riegl and Branch, 1995). While the effects of sedimentation on coral reef fishes are not well known, recent experimental work indicated that, as suspended sediment concentrations increase, larval damselfish are less able to discriminate between preferred settlement sites (Wenger et al., 2011). Hence, decreased sediment and associated nutrients/contaminants may reduce stress on fish and benthic biota.

However, it is also possible that improvement to biological metrics such as live coral cover and fish abundance may ultimately require a reduction in multiple stressors, many of which are present at not only local but also regional and global scales (e.g., climate change, ocean acidification, overfishing). While land-based sources of pollution are widely attributed to be a contributing factor to coral reef decline (e.g., Warne et al., 2005; Waddell and Clarke, 2008), identifying the relative contribution of individual anthropogenic activities to reef degradation is challenging (Downs et al., 2005; Fabricius, 2005; Wolanski et al., 2009). Even prior to widespread coral dieoff in the 1980s, the reefs in Guánica were reported to be in poorer condition than nearby La Parguera (Goenaga and Cintron, 1979; Morelock et al., 2001). Whether reduced sediment inputs alone are enough to foster improvements to coral health in the Guánica region are unknown.

REFERENCES

Alvarez-Filip, L., N.K. Dulvy, J.A. Gill, I.M. Côte, and A.R. Watkinson. 2009. Flattening of Caribbean coral reefs: region-wide declines in architectural complexity. Proceedings of the Royal Society B: Biological Sciences 276: 3019-3025.

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

Ault, J.S., S.G. Smith, J. Luo, M.E. Monaco, and R.S. Appeldoorn. 2008. Length-based assessment of sustainability benchmarks for several coral reef fishes in Puerto Rico. Environmental Conservation 35(3): 221-231.

Bauer, L.J., K. Edwards, K.K.W. Roberson, M.S. Kendall, S. Tormey, and T. Battista. 2012. Shallow-water benthic habitats of Southwest Puerto Rico. NOAA Technical Memorandum NOAA NOS NCCOS 155. Silver Spring, MD.

Bauer, L.J., C.F.G. Jeffrey, and K. Roberson. 2011. Baseline Characterization of Fish Communities, Associated Benthic Habitats and Marine Debris of Jobos Bay In: D.R. Whitall et all. (Eds.). A Baseline Assessment of the Ecological Resources of Jobos Bay, Puerto Rico. NOAA Technical Memorandum NOS NCCOS 133. Silver Spring, MD. 188 pp.

Bauer, L.J. and M.S. Kendall. 2010. Characterization of Reef and Hardbottom Habitats, Associated Reef Communities, and Marine Debris in Vieques. pp. 47-87. In: L.J. Bauer and M.S. Kendall (eds.). An Ecological Characterization of the Marine Resources of Vieques, Puerto Rico Part II: Field Studies of Habitats, Nutrients, Contaminants, Fish, and Benthic Communities. NOAA Technical Memorandum NOS NCCOS 110. Silver Spring, MD. 174 pp.

Clarke, K.R. and R.M. Warwick. 2001. Change in marine communities: an approach to statistical analysis and interpretation, 2nd Edition. PRIMER-E. Plymouth Marine Laboratory, Plymouth. 144 pp.

Cochran, W.G. 1977. Sampling Techniques. John Wiley and Sons, Inc., New York. 413 pp.

Downs, C.A., C.M. Woodley, R.H. Richmond, L.L. Lanning, and R. Owen. 2005. Shifting the paradigm of coral-reef 'health' assessment. Marine Pollution Bulletin 51: 486-494.

Edmonds, P.J. and R. Elahi. 2007. The demographics of a 15-year decline in cover of the Caribbean reef coral Montastraea annularis. Ecological Monographs 77: 3-18.

Fabricius, K.E. 2005. Effects of terrestrial runoff on the ecology of corals and coral reefs: review and synthesis. Marine Pollution Bulletin 50: 125-146.

Froese, R. and D. Pauly (eds.). 2008. FishBase. World Wide Web electronic publication. (Online) http://www.fishbase.org version (06/2008), (Accessed 10 June 2011).

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. Instituto de Oceanología Academia de Ciencias de Cuba and Centro de Investigaciones de Quintana Roo (CIQRO) México. 525 pp.

García-Sais, J.R., R. Castro, J. Sabater Clavell, R. Esteves, S. Williams, and M. Carlo. 2009. Monitoring of coral reef communities at Isla Desecheo, Isla de Mona, Rincón, Guánica, Ponce, Caja de Muerto, and Mayaguez. 2008-09. Final Report submitted to the Department of Natural and Environmental Resources (DNER), U. S. Coral Reef National Monitoring Program, NOAA. 212 pp.

García-Sais, J., R. Appeldoorn, T. Battista, L. Bauer, A. Bruckner, C. Caldow, L. Carrubba, J. Corredor, E. Diaz, C. Lilyestrom, G. García-Moliner, E. Hernández-Delgado, C. Menza, J. Morrell, A. Pait, J. Sabater, E. Weil, E.Williams and S. Williams. 2008a. 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.

García-Sais, J.R., R. Castro, J. Sabater Clavell, M. Carlo, R. Esteves and S. Williams. 2008b. Monitoring of coral reef communities at Isla Desecheo, Isla de Mona, Rincón, Guánica, Ponce, Caja de Muerto, and Mayaguez. 2007-08. Final Report submitted to the Department of Natural and Environmental Resources (DNER), U. S. Coral Reef National Monitoring Program, NOAA. Silver Spring, MD. 212 p

García-Sais, J., R., Castro, R. and J. Sabater-Clavell. 2001. Coral reef communities from natural reserves in Puerto Rico: a quantitative baseline assessment for prospective monitoring programs. Vol. 1: Cordillera de Fajardo, Isla Caja de Muertos, Bosque Seco de Guánica, Bahía de Mayaguez. Final Report submitted to the Department of Natural and Environmental Resources (DNER), San Juan, PR. 232 p.

Gardner, T., I.M. Cote, J.A. Gil, A. Grant, and A. Watkinson. 2003. Long-term region-wide declines in Caribbean corals. Science 301: 958-960.

Golbuu, Y., R. van Woesik, R.H. Richmond, P. Harrison, and K.E. Fabricius. 2011. River discharge reduces reef coral diversity in Palau. Marine Pollution Bulletin 62: 824-831.

Goenaga, C., and G. Cintrón. 1979. Inventory of the Puerto Rican Coral Reefs: Puerto Rico Department of Natural Resources (prepared for the Office of Coastal Zone Management, National Oceanic and Atmospheric Administration). 190 p.

Green, D.H., P.J. Edmunds, and R.C. Carpenter. 2008. Increasing relative abundance of Porites astreoides on Caribbean reefs mediated by an overall decline in coral cover. Marine Ecology Progress Series 359: 1-10.

Jackson, J.B.C., M.X. Kirby, W.H. Berger, K.A. Bjorndal, L.W. Botsford, B.J. Bourque, R.H. Bradbury, R. Cooke, Erlandson, J (Erlandson, J); Estes, JA (Estes, JA); Hughes, TP (Hughes, TP); Kidwell, S (Kidwell, S); Lange, CB (Lange, CB); Lenihan, HS (Lenihan, HS); Pandolfi, JM (Pandolfi, JM); Peterson, CH (Peterson, CH); Steneck, RS (Steneck, RS); Tegner, MJ (Tegner, MJ); Warner, RR (Warner, RR) 2001. 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 CCMA 152. Silver Spring, MD. 46 pp.

Kuffner, I.B., L.J. Walters, M.A. Becerro, V.J. Paul, R. Ritson-Williams, and K.S. Beach. 2006. Inhibition of coral recruitment by macroalgae and cyanobacteria. Marine Ecology Progress Series 323: 107-117.

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.

Morelock, J., W.R. Ramírez, A.W. Bruckner, and M. Carlo 2001. Status of coral reefs, southwest Puerto Rico. Caribbean Journal of Science, Special publication No. 4.

Mumby, P.J. and C.C.C. Wabnitz. 2002. Spatial patterns of aggression, territory size, and harem size in five sympatric Caribbean parrotfish species. Environmental Biology of Fishes 63: 265-279.

Paddack, M.J., J.D. Reynolds, C. Aguilar, R.S. Appeldoorn, J. Beets, E.W. Burkett, P.M. Chittaro, K. Clarke, R. Esteves, A.C. Fonseca, G.E. Forrester, A.M. Friedlander, J. García-Sais, G. González-Sansón, L.K.B. Jordan, D.B. McClellan, M.W. Miller, P.P. Molloy, P.J. Mumby, I. Nagelkerken, M. Nemeth, R. Navas-Camacho, J. Pitt, N.V.C. Polunin, M.C. Reyes-Nivia, D.R. Robertson, A. Rodríguez-Ramírez, E.Salas, S.R. Smith, R.E. Spieler, M.A. Steele, I.D. Williams, C.L. Wormald, A.R. Watkinson, and I.M. Côté. 2009. Recent region-wide declines in Caribbean reef fish abundance. Current Biology 19: 590-595.

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. Caribbean Journal of Science 45: 191-203.

Pauly, D. 1995. Anecdotes and the shifting baseline syndrome of fisheries. Trends in Ecology and Evolution 10: 430.

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.

Pittman, S.J., B.M. Costa, and T.A. Battista. 2009. Using Lidar bathymetry and boosted regression trees to predict the diversity and abundance of fish and corals. Journal of Coastal Research 10053: 27-38.

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. and K.A. Brown. 2011. Multi-scale approach for predicting fish species distributions across coral reef seascapes. PLoS ONE 6(5): e20583.doi:10.1371/journal.pone.0020583

Randall, J.E. 1967. Food habits of reef fishes of the West Indies. Studies in Tropical Oceanography (Miami) 5: 665-847.

Richmond, R.H., T. Rongo, Y. Golbuu, S. Victor, N. Idechong, G. Davis, W. Kostka, L. Neth, M. Hamnett, and E. Wolanski. 2007. Watersheds and coral reefs: Conservation science, policy, and implementation. BioScience 57(7): 598-607.

Riegl, B. and G.M. Branch. 1995. Effects of sediment budgets of 4 scleractinian (Boune 1900) and 5 Alcyonacean (Lamouroux 1816) corals. Journal of Experimental Marine Biology and Ecology 186: 259-275.

Rogers, C.S. 1990. Responses of coral reefs and reef organisms to sedimentation. Marine Ecology Progress Series 62: 185–202.

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

US Fish and Wildlife Service (1973) Endangered Species Act of 1973. As Amended by 108th Congress. Washington, DC: Government Printing Office.

Waddell, J.E. and A.M. Clarke (eds.) 2008. 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.

Warne, A.G., R.M.T. Webb and M.C. Larsen. 2005. Water, Sediment, and Nutrient Discharge Characteristics of Rivers in Puerto Rico, and their Potential Influence on Coral Reefs: U.S. Geological Survey Scientific Investigations Report 2005-5206. 58 pp.

Weil, E., E.A. Hernández-Delgado, A.W. Bruckner, A.L. Ortiz, M. Nemeth, and H. Ruiz. 2002. Distribution and status of Acroporid coral (Scleractinia) populations in Puerto Rico. pp. 71-98. In: A.W. Bruckner (ed.). Proceedings of the Caribbean Acropora Workshop: Potential application of the U.S. Endangered Species Act as a conservation strategy. NOAA Technical Memorandum NMFS-OPR-24. Silver Spring, MD. 199 pp.

Wenger, A.S., J.L. Johansen, and G.P. Jones. 2011. Suspended sediment impairs habitat choice and chemosensory discrimination in to coral reef fishes. Coral Reefs 30: 879-887.

Wolanski, E., J.A. Martinez, and R.H. Richmond. 2009. Quantifying the impact of watershed urbanization on a coral reef: Maunalua Bay, Hawaii. Estuarine, Coastal and Shelf Science 84: 259-268.

Yentsch, C.S., C.M. Yentsch, J.J. Cullen, B. Lapointe, D.A. Phinney, and S.W. Yentsch. 2002. Sunlight and water transparency: cornerstones in coral research. Journal of Experimental Marine Biology and Ecology 268: 171-183.

Chapter 3: Contaminants in Surficial Sediments and Coral Tissues of Guánica Bay

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CONTAMINANT BACKGROUND

Chemical Contaminants Analyzed

Since 1986, NOAA's National Status and Trends (NS&T) Program has monitored and assessed the nation's estuarine and coastal waters for chemical contaminants in a variety of matrices (e.g., bivalve tissues, sediments). Characterization of contaminants in coral reefs by NS&T began in 2005. The suite of chemical contaminants routinely analyzed by NS&T and analyzed in the coral tissue and sediment samples for this project is shown in Table 3.1. The analytes include 58 polycyclic aromatic hydrocarbons (PAHs), 31 organochlorine pesticides, 38 polychlorinated biphenyls (PCBs), four butyltins, and 16 trace and major elements. All samples were analyzed using NS&T standard protocols. The analytical protocols have been published previously (Kimbrough et al., 2006; Kimbrough and Lauenstein, 2006). The nature, sources and environmental significance of each of the contaminant classes analyzed for this project are discussed below.

Table 3.1 List of analytes.

PAHs - Low Molecular Weight	PAHs - High Molecular Weight	PCBs	Organochlorine Pesticides
Naphthalene*	Fluoranthene*	PCB8/5	Aldrin
1-Methylnaphthalene*	Pyrene*	PCB18	Dieldrin
2-Methylnaphthalene*	C1-Fluoranthenes/Pyrenes	PCB28	Endrin
2,6-Dimethylnaphthalene*	C2-Fluoranthenes/Pyrenes	PCB29	Heptachlor
1,6,7-Trimethylnaphthalene*	C3-Fluoranthenes/Pyrenes	PCB31	Heptachlor-Epoxide
C1-Naphthalenes	Naphthobenzothiophene	PCB44	Oxychlordane
C2-Naphthalenes	C1-Naphthobenzothiophenes	PCB45	Alpha-Chlordane
C3-Naphthalenes	C2-Naphthobenzothiophenes	PCB49	Gamma-Chlordane
C4-Naphthalenes	C3-Naphthobenzothiophenes	PCB52	Trans-Nonachlor
Benzothiophene	Benz(a)anthracene*	PCB56/60	Cis-Nonachlor
C1-Benzothiophenes	Chrysene*	PCB66	Alpha-HCH
C2-Benzothiophenes	C1-Chrysenes	PCB70	Beta-HCH
C3-Benzothiophenes	C2-Chrysenes	PCB74/61	Delta-HCH
Biphenyl*	C3-Chrysenes	PCB87/115	Gamma-HCH
Acenaphthylene*	C4-Chrysenes	PCB95	2,4'-DDT
Acenaphthene*	Benzo(b)fluoranthene*	PCB99	4,4'-DDT
Dibenzofuran	Benzo(k)fluoranthene*	PCB101/90	2,4'-DDD
Fluorene*	Benzo(e)pyrene*	PCB105	4,4'-DDD
C1-Fluorenes	Benzo(a)pyrene*	PCB110/77	2,4'-DDE
C2-Fluorenes	Perylene*	PCB118	4,4'-DDE
C3-Fluorenes	Indeno(1,2,3-c,d)pyrene*	PCB128	DDMU
Anthracene*	Dibenzo(a,h)anthracene*	PCB138/160	1,2,3,4-Tetrachlorobenzene
Phenanthrene*	C1-Dibenzo(a,h)anthracenes	PCB146	1,2,4,5-Tetrachlorobenzene
1-Methylphenanthrene*	C2-Dibenzo(a,h)anthracenes	PCB149/123	Hexachlorobenzene
C1-Phenanthrene/Anthracenes	C3-Dibenzo(a,h)anthracenes	PCB151	Pentachloroanisole
C2-Phenanthrene/Anthracenes	Benzo(g,h,i)perylene*	PCB153/132	Pentachlorobenzene
C3-Phenanthrene/Anthracenes		PCB156/171/202	Endosulfan II

1. NOAA/NOS/NCCOS
2. Earth Resources Technology Incorporated, Contractor to NOAA, National Marine Fisheries Service, Restoration Center
3. CSS-Dynamac, Fairfax, VA

Table 3.1 (continued). List of analytes.

PAHs - Low Molecular Weight	PAHs - High Molecular Weight	PCBs	Organochlorine Pesticides
C4-Phenanthrene/Anthracenes	Trace Elements	PCB158	Endosulfan I
Dibenzothiophene	Aluminum	PCB170/190	Endosulfan Sulfate
C1-Dibenzothiophenes	Antimony	PCB174	Mirex
C2-Dibenzothiophenes	Arsenic	PCB180	Chlorpyrifos
C3-Dibenzothiophenes	Cadmium	PCB183	
	Chromium	PCB187	Butyltins
	Copper	PCB194	Monobutyltin
	Iron	PCB195/208	Dibutyltin
	Lead	PCB199	Tributyltin
	Manganese	PCB201/157/173	Tetrabutyltin
	Mercury	PCB206	
	Nickel	PCB209	
	Selenium		
	Silver		
	Tin		
	Zinc		

*Compounds used in the calculation of total PAHs
PAHs = polycyclic aromatic hydrocarbons; PCBs = polychlorinated biphenyls

Polycyclic aromatic hydrocarbons (PAHs)

PAHs are associated with the use and combustion of fossil fuels and other organic materials (e.g., wood). Natural sources of PAHs include forest fires and volcanoes. The PAHs analyzed in this study are two to six ring aromatic compounds.

Environmental Effects of PAHs

Although an extensive amount of research has been done on the accumulation and effects of PAHs on aquatic organisms, relatively few studies have been conducted to address the effects of PAHs on corals. Hydrophobic in nature, PAHs readily accumulate in marine organisms through direct exposure (e.g., body surface, gills) or through the food chain (Neff,1985). PAH exposure has been associated with oxidative stress, immune system and endocrine system problems and developmental abnormalities in marine organisms (Hylland, 2006).

Furthermore, a number of individual PAHs including benzo[a]pyrene, benz[a]anthracene, chrysene, benzo[b]fluoranthene, benzo[k]fluoranthene, dibenzo[a,h]anthracene and indeno[1,2,3-c,d]pyrene have been identified as likely carcinogens (USDHHS, 1995). The carcinogenic potential of PAHs in marine organisms is associated with their metabolic breakdown, which generates reactive epoxides that can bind to cellular components such as DNA (Hylland, 2006; Neff, 1985). In addition to accumulating in the coral tissues themselves, PAHs can also affect the zooxanthellae, the symbiotic photosynthetic dinoflagellate algae found within coral tissues. Bioaccumulation appears to be related to the lipid content of both the coral and the algae (Kennedy et al., 1992). Accumulation of PAHs by corals is not an impact by itself; however, the accumulation of a chemical contaminant in an organism increases the likelihood of adverse effects. Solbakken et al. (1984) demonstrated that both phenanthrene and naphthalene were accumulated by the brain coral *(Diploria strigosa)* and green cactus coral *(Madracis decatis)*, and that the lower molecular weight naphthalene was eliminated at a higher rate than phenanthrene (Solbakken et al., 1984). Fluoranthene and pyrene can be toxic to adult corals, particularly in the presence of increased ultraviolet radiation (i.e., phototoxicity; Peachey and Crosby, 1996; Guzman-Martinez et al., 2007).

Polychlorinated biphenyls (PCBs)

PCBs are a class of synthetic organic compounds that have been used in a wide range of applications ranging from electrical transformers and capacitors, to hydraulic and heat transfer fluids, to pesticides and paints. Although no longer manufactured in the U.S., environmental contamination by PCBs is still a potential problem in many environmental systems due to PCBs' environmental persistence and tendency to bioaccumulate. In some cases, use of equipment containing PCBs (e.g., railroad locomotive transformers) is still permitted (CFR, 1998). The structure of PCBs includes a biphenyl ring structure (two benzene rings with a carbon to carbon bond) and chlorine atoms, the latter of which varies in both number and location on the rings. There are 209 PCB congeners (structures) possible.

Environmental Effects of PCBs

Exposure to PCBs has been linked to reduced growth, reproductive impairment and vertebral abnormalities in fish (EPA,1997). Solbakken et al. (1984) quantified the bioconcentration of radiolabeled hexaPCB (2,4,5,2’,4’,5’-hexachlorobiphenyl) in coral. The PCB was rapidly accumulated in brain coral and green cactus coral. However, depuration proceeded at a slow rate; after 275 days nearly 33% of the original radioactivity from the PCB remained in the coral, suggesting that PCBs are quite persistent in coral tissues.

Organochlorine Pesticides

For this study, a total of 31 organochlorine pesticides were analyzed in the sediment and coral samples (Table 3. 1). From the 1950s to the early 1970s, a series of chlorine containing hydrocarbon insecticides were used to control mosquitoes and agricultural pests. One of the best known of these pesticides used during this time period was dichlorodiphenyltrichloroethane, more commonly known as DDT. Before DDT was banned in 1972, an estimated 1.35 billion pounds of DDT were applied in the U.S., the majority of which was applied to cotton crops (EPA, 2009). Organochlorine pesticides, including DDT, are of great environmental concern due to their environmental persistence, potential to bioaccumulate, and toxicity to non-target organisms. These concerns led to their ban in the U.S., but because of their persistence and heavy use in the past, residues of many organochlorine pesticides can be found in the environment, including biota. Other pesticides quantified in this study include: Hexachlorocyclohexane (HCH), chlordane, chlorobenzenes, aldrin, dieldrin, endrin, chlorpyrifos, mirex and endosulfans.

Environmental Effects of Organochlorine Pesticides

Organochlorine pesticides primarily act on biota as neurotoxins. Like PCBs, DDT has also been shown to be endocrine disruptors. DDT and its metabolite DDE have been specifically linked to eggshell thinning in birds, particularly raptors. A number of organochlorine pesticides are also toxic to aquatic life including crayfish, shrimp and some species of fish.

Butyltins

Butyltins (mono-, di-,tri- and tetrabutyltins) have a range of uses from biocides to catalysts to glass coatings. In the 1950s, tributyltin, or TBT, was first shown to have properties as an effective biocide (Bennett, 1996). Beginning in the late 1960s, TBT was incorporated into a very effective antifoulant paint system, quickly becoming one of the most effective paints ever used on boat hulls (Birchenough et al., 2002). TBT was utilized in a polymer boat paint system that released the biocide at a constant, slow rate, which effectively controlled hull fouling organisms such as barnacles, mussels, weeds and algae (Bennett, 1996). In the aquatic environment, TBT experiences both photodegradation and microbial metabolism (Bennett, 1996). The breakdown process involves sequential debutylization resulting in dibutyltin, monobutyltin and finally inorganic tin (Batley, 1996). The half-life of TBT (i.e., the amount of time needed to convert half of the TBT to dibutyltin) in natural waters has been experimentally determined to be on the order of days; further degradation to monobutyltin takes approximately a month (Batley, 1996). Experiments with aerobic sediments have shown that the half-life of TBT is similar to that measured in the water column. In anoxic sediments, however, the half-life of TBT is considerably longer, on the order of two to four years (Batley, 1996).

Environmental Effects of TBT

TBT in the aquatic environment has been associated with endocrine disruption, specifically an imposex condition in marine gastropod mollusks. Beginning in 1989 in the U.S., the use of TBT as an antifouling agent was banned on vessels smaller than 25 m in length (Gibbs and Bryan, 1996). Negri et al. (2002) investigated the effects of TBT in sediments from a shipwreck, on the coral *Acropora microphthalma* from the Great Barrier Reef in Australia. Sediments originally contained approximately 160 µg/g TBT. Even when diluted to 5% of the original TBT concentration, successful settlement of coral larvae in the laboratory was inhibited.

MAJOR AND TRACE ELEMENTS

A total of 16 trace and major elements were measured in sediments, and 14 in coral tissues for this project (Table 3.1). Most of these elements are metals, however, antimony, arsenic and silicon are metalloids; selenium is a nonmetal. All occur naturally to some extent in the environment. Aluminum, iron and silicon are major crustal elements (i.e. components of the earth’s crust). Some trace and major elements in the appropriate concentrations are biologically essential. As their name implies, trace elements such as chromium, cadmium, lead and nickel occur at lower concentrations in crustal material than aluminum, iron and silicon; however, mining and manufacturing processes along with the use and disposal of products containing trace elements can result in elevated concentrations in the environment.

Environmental Effects of Trace Elements

A number of trace elements are toxic at low concentrations. Cadmium, used in metal plating, solders and batteries, has been shown to impair development and reproduction in several invertebrate species, and impedes the ability to osmoregulate in herring larvae (USDHHS, 1999; Eisler, 1985). Mercury is volatile and can enter the atmosphere through processes including mining, manufacturing, combustion of coal and volcanic eruptions, then returning to earth as atmospheric deposition. Effects of mercury on copepods include reduced growth and reproductive rates (Eisler, 1987). Chromium can enter the environment through oil and coal combustion, and various industrial waste streams (textiles and leather production, electroplating). Chromium has been shown to reduce survival and fecundity in the cladoceran *Daphnia magna*, and reduced growth in fingerling chinook salmon (*Oncorhynchus tshawytscha*; Eisler 1986). Copper has a number of uses such as in antifouling paints, wood preservatives, heat exchangers in power plants, electrical wires, coinage and agriculture. Although a biologically essential element, elevated levels of copper can impact aquatic organisms, including adverse effects on reproduction and development in mysid shrimp (Eisler, 1998). In corals, Reichelt-Brushett and Harrison (2005) found that a copper concentration of 20 µg/L significantly reduced fertilization success in lesser star coral *(Goniastrea aspera)*. At copper concentrations at or above 75 µg/L, fertilization success was reduced to 1% or less. Fertilization success was also significantly reduced in the stagehorn coral (*Acropora longicyathus)* at 24 µg/L, a similar concentration level to when effects were observed in lesser star coral.

Additional Data

Sediment samples were also analyzed for total organic carbon (TOC) and grain size. These two pieces of ancillary data are important for assessing the potential for accumulation of contaminants in sediments. In general, for freshwater, estuarine and coastal waters, a positive correlation exists between sediment TOC and chemical contaminants, particularly organic contaminants (Shine and Wallace, 2000; Hassett et al., 1980). Sediment grain size is also an important characteristic that can influence contaminant concentrations. Organic contaminants, as well as a number of metals bind to the smaller silt and clay grain size fractions of sediments, due to the larger surface areas of these fractions. The charge characteristics of clays (small size fraction) lend themselves to preferential attachment of trace and major elements.

The bacteria *Clostridium perfringens*, often used as an indicator of sewage, was analyzed in the sediments collected. This anaerobic, gram positive staining rod-shaped bacteria frequently occurs in the intestines of humans as well as in domestic and wild animals.

Coral tissues were analyzed for lipid content. Tissues with higher lipid content have the potential to sequester more lipophilic compounds, such as organic contaminants.

METHODS

Sampling Design

In order to assess the overall contaminant condition of the ecosystem, and to be able to make geographically explicit conclusions about how pollutants vary spatially, a stratified random sampling design was utilized. Using this approach, all areas had an equal chance of being selected as a sampling site.

Sediment strata were constructed from existing benthic habitat maps and included all non-hard bottom sediments. Six geographic strata were initially articulated for sediment sampling: three within the bay (North Bay, Central Bay, South Bay) and three immediately offshore from the bay (west offshore, central offshore, east offshore). In each of these six strata, five sites were randomly selected (Figure 3.1). If a site could not be sampled (e.g., if the site was inaccessible) a pre-selected randomly determined alternate site from within that strata was sampled. Additionally, three targeted sediment sites were selected in the watershed. These sites included: a site as close to the mouth of the Rio Loco as could be safely sampled, a site on the main stem of the Rio Loco and a site on the agricultural canal that drains the Lajas Valley to the west of the Rio Loco (Figure 3.1). These sites could not be randomly selected due to site access considerations; the exact location of these sites was determined by where stream access was logistically possible. Thirty-three sediment sites were sampled in June 2009. In order to better understand potential watershed sources

Figure 3.1. Sediment sampling sites from June, 2009. Sites were selected in six geographic strata (yellow polygons) in a stratified random design with the exception of the three watershed sites which were targeted.

of contaminants, additional stream samples were collected in February 2013 (Appendix B, Figure B.14).

Coral sampling strata had the same geographic boundaries as the three offshore strata as the sediment sampling, but for the hardbottom areas. Live coral is not present in sufficient quantities within the bay in order to sample the bay. Within each stratum, five sites were randomly selected from hard bottom habitat areas using existing habitat maps (Figure 3.2). If a site could not be sampled (e.g., due to a lack of coral, or the site being inaccessible) a pre-selected randomly determined alternate site from within that strata was sampled. Mustard hill coral *(Porites astreoides)* was chosen for this study as it is a common species of coral in the Caribbean (Humann and DeLoach, 2002), and is easy to find on hard bottom portions of the study area. Colonies of mustard hill coral are generally massive but are often found as encrusting forms, particularly in shallow, surging waters (Veron, 2000). Previous studies (Pait et al., 2009; Bauer and Kendall, 2010; Whitall et al., 2011) also quantified contaminants in *P. astreoides* in other systems in Puerto Rico, which allows for comparison across the region. Coral tissue was collected from 15 sites in June 2009.

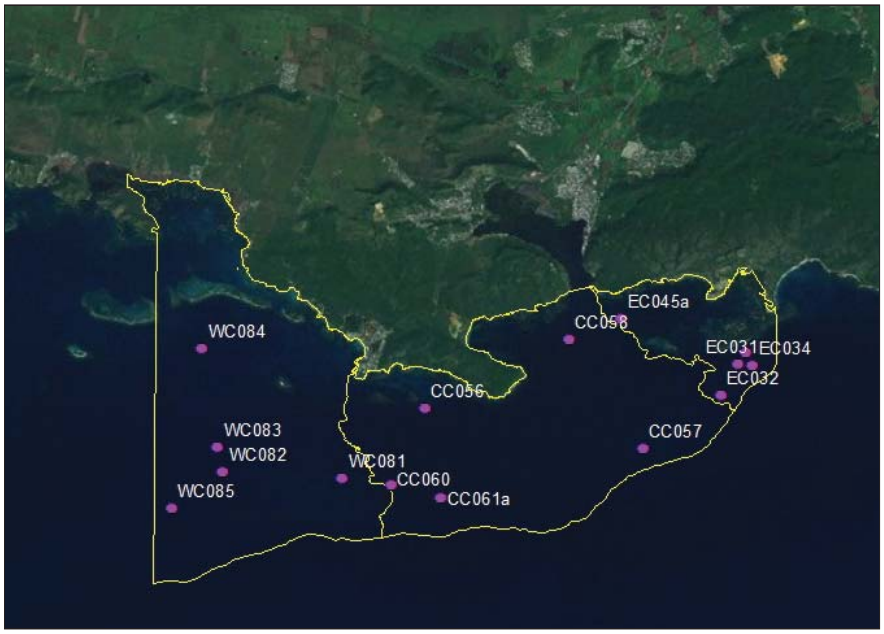

Figure 3.2. Coral (mustard hill coral, Porites astreoides*) sampling sites from June, 2009. Sites were selected in three geographic strata (yellow polygons) in a stratified random design.*

Materials and Methods

Sediment samples were collected using standard NOAA NS&T Program protocols (Lauenstein and Cantillo, 1998). Briefly, a Ponar sediment grab was deployed to collect the sediment samples. Rocks, large coral or shell fragments or bits of seagrass were removed. If an individual grab did not result in 200-300 g of sediment, a second grab was collected and composited with material from the first grab. If enough sediment had not been collected after three deployments of the grab, the site was abandoned and the boat moved on to an alternate, randomly preselected site.

To avoid contamination of samples by equipment and cross contamination between samples, visible sediment was removed from the grab with a brush, then all sampling equipment was rinsed with acetone followed by site water just prior to use. Field personnel handling the samples wore disposable nitrile gloves. The top 3 cm of sediment were collected from the sediment grab using a Kynar-coated sediment scoop. Sediments were placed into a certified clean (IChem®) 250 ml labeled jar, capped and then placed on ice in a cooler. Sediments for grain size analysis were placed in a Whirl-Pak® bag, sealed and placed on ice in a cooler. After returning from the field each day, sediment samples were frozen (-15°C) and the Whirl-Pak® bags used for grain size analysis were refrigerated (4°C), to avoid altering the grain size structure of the sediment that could occur during freezing. Targeted watershed sites were sampled in the same fashion, with the exception that samples were taken by hand (i.e., sediment was scooped into jars) rather than using a dredge.

The coral samples were taken by NOAA SCUBA divers using a hammer and titanium punch. Titanium was used as it was not a target trace element for this project. Prior to each use, the punch was rinsed with acetone and site water to minimize cross-contamination between sites.

Divers collecting the coral samples also wore disposable nitrile gloves. The diver hammered the titanium punch into the coral head which produced a coral core with a diameter of approximately 1.5 cm and a similar core length. Approximately 20 cores were taken at each site and placed underwater in an IChem® certified clean 250 ml jar and then capped. The jar was brought to the surface, drained of water and placed on ice. At the end of each day, the samples were placed in a freezer (-15°C). At the end of the mission, samples were shipped overnight to the laboratory for analysis.

Field researchers measured a suite of water parameters (dissolved oxygen, temperature, salinity and conductivity) at each coral site using a YSI® salinity/conductivity/temperature meter. The instrument probe was submerged to a depth of approximately 0.5 m. Strong offshore currents prevented the measurement of bottom water parameters. Unfortunately, due to equipment failure during the sampling mission, YSI data are not available for the sediment sites.

PAHs were analyzed in the laboratory using gas chromatography/mass spectrometry in the selected ion monitoring mode (Kimbrough et al., 2006). Selected chlorinated organics (PCBs and pesticides) were analyzed using gas chromatography/electron capture detection (Kimbrough et al., 2006). Butyltins were analyzed using gas chromatography/flame photometric detection (Kimbrough et al., 2006).

Silver, cadmium, copper, lead, antimony and tin were analyzed using inductively coupled plasma - mass spectrometry. Aluminum, arsenic, chromium, iron, manganese, nickel, silicon and zinc were analyzed using inductively coupled plasma - optical emission spectrometry. Mercury was analyzed using cold vapor - atomic absorption spectrometry. Selenium was analyzed using atomic fluorescence spectrometry (Kimbrough and Lauenstein et al., 2006). For each element, total elemental concentration (i.e. sum of all oxidation states) was measured.

TOC was quantified via high temperature combustion and subsequent quantification of the CO_2 produced (McDonald et al., 2006). Grain size analysis was carried out using a series of sieving and settling techniques (McDonald et al., 2006). To assess the presence of viable *C. perfringens*, sediment extracts were plated on growth medium and the number of colonies that developed were counted.

STATISTICAL ANALYSIS

All contaminant data were analyzed using JMP® statistical software. The data were first tested for normality using the Shapiro-Wilk test. The data were not normally distributed. A non-parametric multiple comparisons test (Dunn Method for Joint Ranking, α=0.05) was used to evaluate differences between strata. The three watershed sites were targeted rather than randomly selected; these values were included in the summary statistics for the entire study area, but were not included in the analysis of strata. Spearman Rank correlations (α=0.05) were examined to evaluate the relationships between variables.

Providing Context for Results

In addition to comparing contamination results between strata, there are several other ways to evaluate the relative level of contamination of Guánica Bay and the surrounding reef ecosystems. First, and most simply, these findings can be compared to the contaminant concentrations in other studies in Puerto Rico. Second, when the values for Guánica Bay were found to be high compared to the rest of the region, the findings can be placed in a national context by comparing the results to a national dataset, such as NOAA's NS&T Program, which includes sediment chemistry data from over 300 coastal sites throughout the U.S. Finally, the degree of sediment contamination in Guánica Bay can be assessed using NOAA's numerical sediment quality guidelines (SQG) known as ERL (effects range-low) and ERM (effects range-median) developed by Long and Morgan (1990) and Long et al. (1995). The SQG value was not defined for all analytes; existing values are presented in Table 3. 2. These guidelines are statistically derived levels of contamination above which toxic effects would be expected to be observed in benthic organisms with at least a 50% frequency (ERM), and below which effects were rarely (<10 %) expected (ERL). Unfortunately, threshold values are not available for coral body burdens; coral tissue concentrations were compared to other studies in the region.

Table 3. 2. Sediment Quality Guidelines (Long and Morgan, 1990).

Contaminant	Effects Range-Low	Effects Range-Median
Total PAHs (ng/g)	4,022	44,792
Total PCBs (ng/g)	22.7	180
Total DDT (ng/g)	1.58	46.1
Ag (µg/g)	1	3.7
As (µg/g)	8.2	70
Cd (µg/g)	1.2	9.6
Cr (µg/g)	81	370
Cu (µg/g)	34	270
Hg (µg/g)	0.15	0.71
Ni (µg/g)	20.9	51.6
Pb (µg/g)	46.7	NA
Zn (µg/g)	150	410

CONTAMINANT RESULTS AND DISCUSSION

Organics

PAHs in Sediments

Concentrations of total PAHs (sum of 58 PAHs measured in this study) in sediments ranged from 0.6 ng/g to 4,664 ng/g (Figure 3.3), with a mean of 614.8 ng/g (Table 3.3). The PAH concentrations measured in this study are within the range of PAH values measured in sediments in other locations in Puerto Rico (Table 3.4). When comparing measured concentrations to published sediment quality guidelines, total PAHs exceeded the ERL at three sites in the bay. No sites exceeded the ERM. The three sites exceeding the ERL are in the Central Bay and North Bay strata. There are a number of active boat docks near these sites which may be a source of PAHs. Concentrations of total PAHs in the North Bay and Central Bay are significantly greater than concentrations in the central offshore and eastern offshore strata (Dunn Method, α=0.05). The ratios of phenanthrene-to-anthracene (P/A) and fluoranthene-to-pyrene (F/P) have been used as a screening tool to assess the relative contributions of pyrogenic (combustion-related) versus petrogenic (uncombusted) sources of PAHs (Budzinski et al., 1997). Higher levels of uncombusted PAHs would be more indicative of the presence of spilled fuels such as gasoline, or oil. P/A ratios less than 10 are more indicative of pyrogenic sources; F/P ratios greater than one are also thought to be associated with pyrogenic sources. F/P ratios are close to or greater than one at the majority of sites (20 of 33) suggesting that, in the study area as a whole, uncombusted sources are more important than combusted sources of PAHs. There was a well documented oil spill in the area in August of 2007 (Coast Guard News, 2007), which may be a source of some of these PAHs and in conjunction with PAHs from small boat traffic, could explain these patterns.

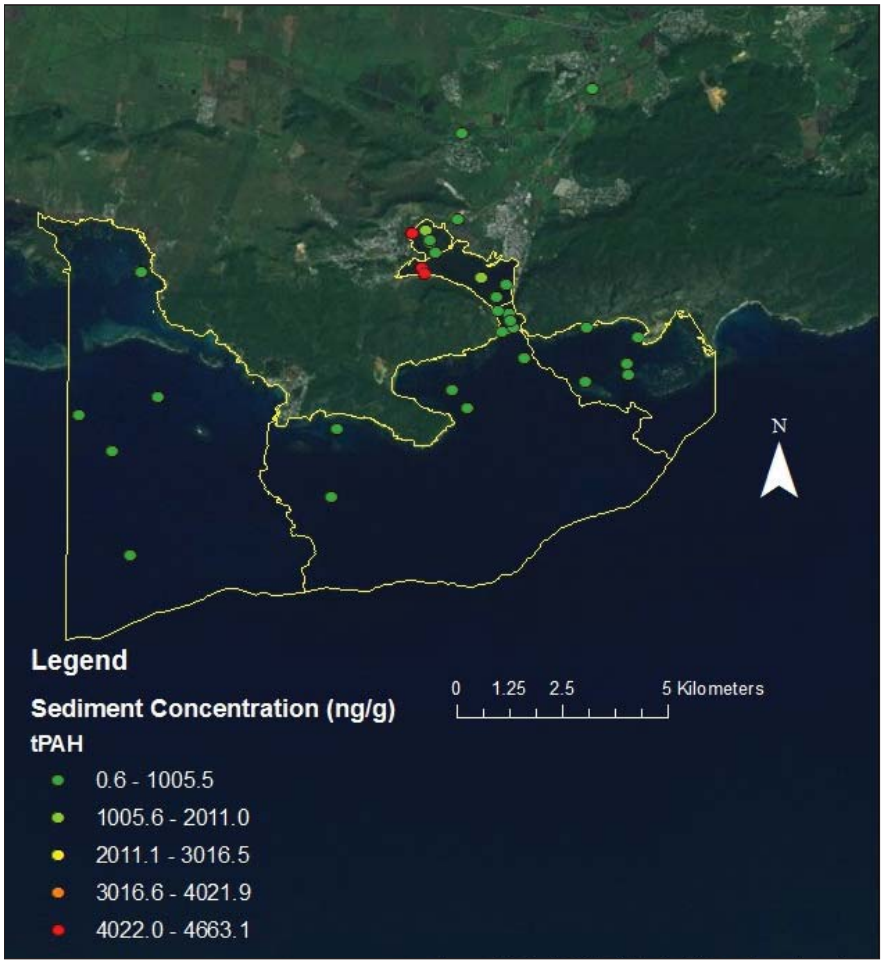

Figure 3.3. Concentrations of total PAHs in sediments. Red indicates an exceedance of the ERL.

Table 3. 3. Summary statistics for organic contaminants quantified in sediments in Guánica Bay. Units are ng/g (butyltins are expressed as ng Sn/g).

Analyte	Mean	Median	Min	Max	StDev
Total PAHs	614.829	74.000	0.637	4663.143	1272.227
Total DDTs	10.132	0.423	0.000	69.246	18.714
Total PCBs	336.068	24.345	0.113	3059.902	661.813
Total HCH	0.056	0.042	0.000	0.253	0.057
Total Chlordane	0.491	0.047	0.000	7.393	1.335
Monobutyltin	0.160	0.000	0.000	0.930	0.232
Dibutyltin	0.135	0.143	0.000	0.790	0.163
Tributyltin	0.079	0.000	0.000	1.190	0.261

Table 3.4. Comparison of Guánica sediment findings (mean and maxima) with three other study sites in Puerto Rico: southwest Puerto Rico (SW PR; Pait et al., 2007), Jobos Bay (Whitall et al., 2011) and Vieques (Pait et al., 2010). Italics denote instances where Guánica was higher than other studies. All raw data from these studies is also available on the NOAA NS&T data server (http://egisws02.nos.noaa.gov/nsandt/). Unless otherwise noted, units of organic pollutants are ng/g; units of trace elements are ug/g.

	Mean				Max			
	Guánica	**SW PR**	**Jobos**	**Vieques**	***Guánica***	**SW PR**	**Jobos**	**Vieques**
Total DDT	*10.13*	2.10	0.54	23.61	*69.25*	46.93	3.28	1,273.66
Total PAHs	*614.8*	80.6	1,061.7	52.3	*4,663.1*	911.2	14,249.9	370.3
Total PCBs	336.07	75.31	2.09	0.45	3,059.90	1,970.28	19.24	4.85
Total HCH	*0.056*	0.043	0.047	0.070	*0.253*	0.685	0.698	1.807
Total Chlordane	*0.49*	0.15	0.21	0.04	*7.39*	2.54	2.21	0.67
TBT (ng Sn/g)	*0.16*	0.01	0.56	0.04	*0.93*	0.29	10.91	1.23
DBT (ng Sn/g)	*0.14*	0.16	0.45	0.26	*0.79*	0.55	3.78	1.27
MBT (ng Sn/g)	*0.08*	0.39	0.66	0.29	*1.19*	1.54	6.60	2.67
Ag	*0.019*	0.002	0.118	0.103	*0.174*	0.081	0.219	0.575
Al	*23,880*	5,983	39,138	35,532	*62,800*	66,600	73,700	118,000
As	*4.77*	1.73	12.59	4.37	*12.80*	10.30	28.10	15.40
Cd	*0.017*	0.009	0.008	0.133	*0.108*	0.223	0.174	1.920
Cr	*286.46*	31.17	18.20	22.45	*1,930.00*	440.00	29.80	178.00
Cu	*29.18*	5.21	33.83	25.91	*102.00*	80.60	73.70	103.00
Fe	*27431*	5,210	26,565	17611	*94,000*	58,900	50,500	50700
Hg	*0.039*	0.004	0.043	0.019	*0.186*	0.095	0.144	0.112
Mn	*469.63*	109.40	510.56	300.91	*2,020.00*	774.00	1,130.00	967.00
Ni	*228.71*	26.56	10.96	7.80	*709.00*	442.00	31.00	38.30
Pb	*6.08*	1.93	7.15	5.42	*31.90*	13.70	16.70	17.60
Sb	0.14	0.09	0.22	0.23	0.43	0.33	0.59	1.19
Se	*0.15*	0.17	0.33	0.26	*0.51*	0.43	1.56	1.20
Si	84,726	18,885	127,849	118,243	220,000	187,000	235,000	451,000
Sn	*0.83*	0.21	1.13	0.66	*3.93*	2.93	2.74	17.10
Zn	*46.71*	7.99	54.21	34.36	*153.00*	90.70	117.00	130.00

PAHs in Coral Tissue

Concentrations of total PAHs in coral tissues ranged from 2.4 ng/g to 8.7 ng/g (Figure 3.4) with a mean of 4.35 ng/g (Table 3.5), which is orders of magnitude lower than values observed in the sediments. The PAH concentrations measured in this study are within the range of PAH values measured in corals in other locations in Puerto Rico (Table 3.6), although values in this study are lower than what was observed farther to the west in La Parguera (Pait et al., 2008). No statistically significant differences existed between the strata (Dunn Method, α=0.05). The coral sampling site with the highest recorded concentration of total PAHs was located near the mouth of Guánica Bay at a concentration of 8.7 ng/g. Elevated concentrations around southwest Puerto Rico may be due to the high intensity recreational boating and storm water from urban areas near the coast. More research is needed to understand the uptake and accumulation of PAHs in corals in order to accurately determine sources and impacts of PAH contaminants.

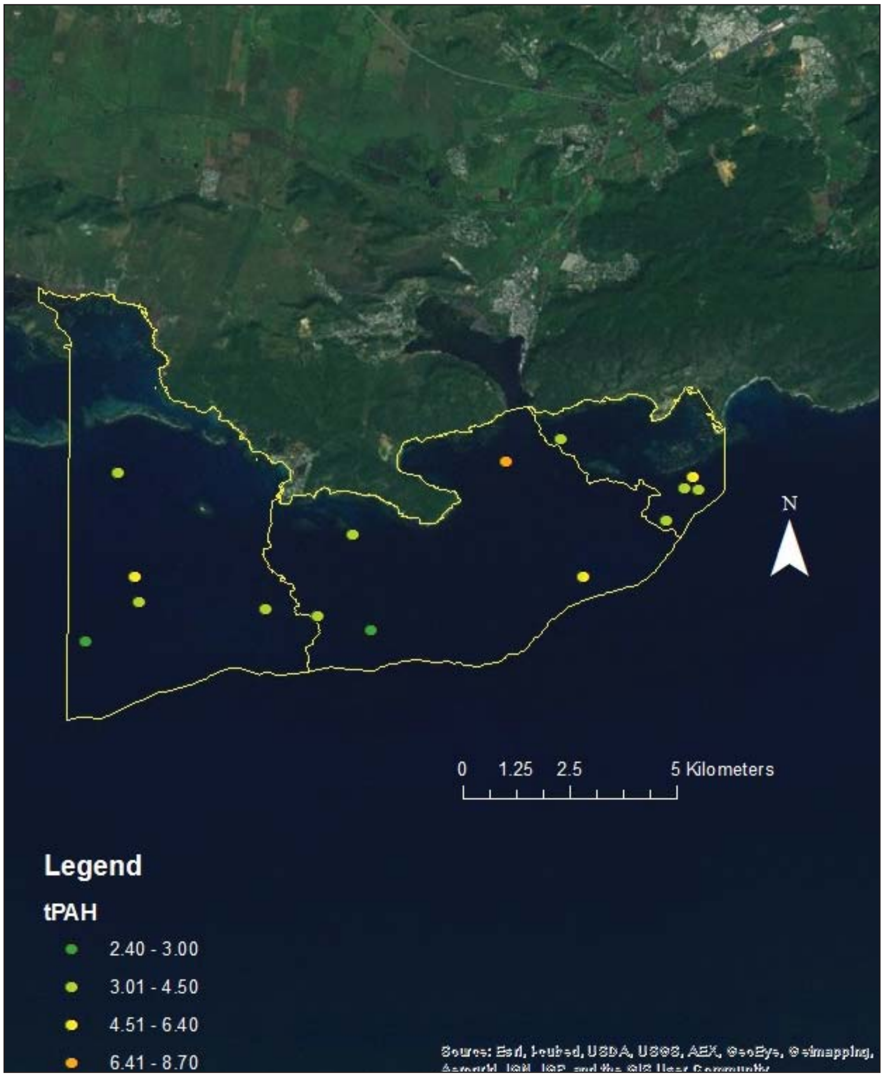

*Figure 3.4. Concentrations of total PAHs in coral (*Porites astreoides*).*

Table 3.5. Summary statistics for organic contaminants quantified in corals in Guánica Bay. Units are ng/g (butyltins are expressed as ng Sn/g).

Analyte	Mean	Median	Min	Max	StDev
Total PAHs	4.353	3.8	2.4	8.7	1.587
Total DDT	0.055	0	0	0.28	0.099
Total PCB	0.464	0.1299	0	3.52	0.9
Total HCH	0.094	0	0	3.52	0.9
Total Chlordane	0.038	0	0	0.52	0.133
Monobutyltin	0	0	0	0	0
Dibutyltin	0	0	0	0	0
Tributyltin	0	0	0	0	0

Table 3.6. Comparison of Guánica coral tissue findings (mean and maximum) with three other study sites in Puerto Rico: southwest Puerto Rico (SW PR; Pait et al., 2007), Jobos Bay (Whitall et al., 2011) and Vieques (Pait et al., 2010). Italics denote instances where Guánica was higher than other studies. All raw data from these studies is also available on the NOAA NS&T data server (http://egisws02.nos.noaa.gov/nsandt/). Units of organic pollutants are ng/g; units of trace elements are µg/g.

	Mean				Maximum			
	Guánica	SW PR	Jobos Bay	Vieques	*Guánica*	SW PR	Jobos Bay	Vieques
Total DDT	*0.06*	0.08	0.04	0.13	*0.28*	0.62	0.60	2.26
Total PAHs	*4.353*	41.91	4.58	15.00	*8.700*	154.00	6.40	21.50
Total PCBs	0.464	1.016	0.120	0.229	3.518	5.390	0.623	1.711
Total HCH	*0.09*	0	0	0	*1.28*	0	0	0.05
Total Chlordane	*0.04*	0	0	0.12	*0.52*	0	0	0.81
Ag	*0.004*	0	0	0.013	*0.055*	0.094	0	0.033
Al	*68.51*	37.80	177.69	30.75	*140.00*	82.20	333.00	103.00
As	*1.42*	0	1.68	0.24	*2.37*	0	2.44	3.42
Cd	*0.27*	0	0.25	0.19	*0.33*	0	0.31	0.29
Cr	*0*	0	0	0.18	*0*	0	0	1.09
Cu	*22.47*	2.06	50.30	0.76	*76.90*	3.54	97.20	6.51
Fe	*100.73*	90.80	212.13	51.20	*480.00*	353.00	480.00	526.00
Hg	*0*	0	0	<0.001	*0*	0	0	0
Mn	*10.48*	3.01	13.33	2.66	*21.20*	5.36	24.60	8.24
Ni	*4.66*	1.32	3.13	0.90	*11.60*	2.70	6.84	8.09
Pb	*0.13*	0	1.43	0.07	*0.32*	0.07	12.50	0.17
Se	*0.19*	0.05	0.18	0.10	*0.25*	0.37	0.26	0.34
Sn	*0.07*	0.02	0.01	0.25	*0.27*	0.14	0.10	0.40
Zn	*9.42*	6.09	8.59	3.43	*25.20*	18.30	16.90	15.20

PCBs in Sediments

Concentrations of total PCBs (sum of 38 PCB congeners analyzed) in sediments ranged from 0.113 ng/g to 3059 ng/g (Figure 3.5) with a mean of 336 ng/g (Table 3. 3). This is markedly higher than sediment concentrations measured in other studies in Puerto Rico (Table 3.4). Because NOAA's NS&T Program has been monitoring contaminants, including PCBs, in the environment since 1986, the high concentrations observed in this study can be put in national and historical context. A number of individual PCB congener concentrations measured in the Guánica Bay area (including two sites in the bay measured as part of Pait et al., 2008) were in the top 10% of all measured sediment concentrations (nationwide, since 1986). Additionally, for some additional congeners the concentrations measured in this study were among the top ten highest individual values ever measured by NS&T (Table 3.7). PCB146, PCB194 and PCB199 were especially high, with sites from Guánica Bay representing seven of the highest 10 values ever measured for those congeners. It is clear that sediment PCB concentrations in Guánica Bay are very high, not only for the region, for the nation as well. PCBs are one of the few analytes where there is strong evidence of transport from the bay to the offshore area, with higher offshore concentrations to the west, as would be expected with the predominant longshore

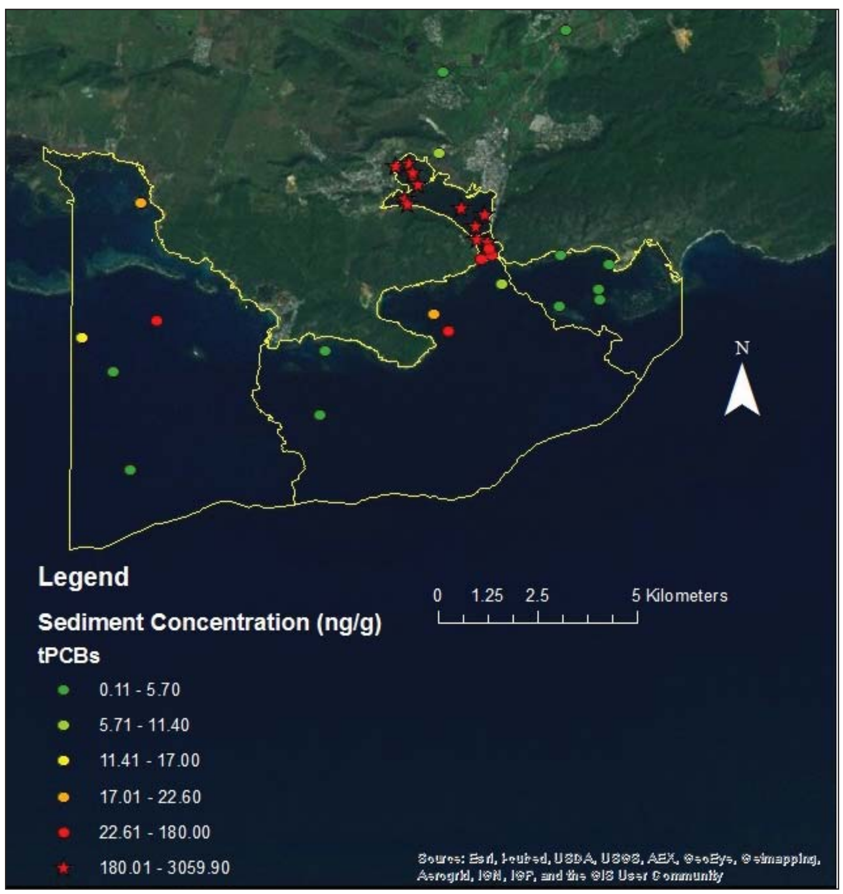

Figure 3.5. Concentrations of total PCBs in sediments. Red indicates an exceedance of the ERL. Red stars indicate exceedance of ERM.

current direction. Statistically, total PCB concentrations in the Central Bay are greater than central offshore and eastern offshore, and the North Bay concentrations are greater than the eastern offshore stratum (Dunn Method, α=0.05).

Table 3.7. Comparison of individual PCB congeners from Guánica Bay with national historic sediment values from NOAA's NS&T program. All NS&T sediment data for each congener were ranked and the frequency of Guánica data points occurring the top 10% and top 10 individual values is presented here. Also shown are the highest ever reported value for each congener. All raw data is available on the NOAA NS&T data server (http://egisws02.nos.noaa.gov/nsandt/).

Analyte	# in top 10	# in top 10%	Highest Value Nationwide	Highest Value Location
PCB146	7	23	26.42	Guánica Bay
PCB199	7	13	20.95	Guánica Bay
PCB194	7	19	35.27	Guánica Bay
PCB183	5	22	27.61	Guánica Bay
PCB151	5	19	39.38	Guánica Bay
PCB156_171_202	4	17	22.93	Guánica Bay
PCB158	4	17	21.8	San Diego Bay
PCB174	3	15	38.02	Guánica Bay
PCB99	3	13	135.325	Lake Michigan
PCB29	2	14	27.82	Lake Erie
PCB49	2	22	795.859	Lake Michigan
PCB149_123	2	14	120	San Diego Bay
PCB31	1	22	1464.887	Lake Michigan
PCB87_115	1	14	82.02	Lake Michigan
PCB74_61	1	13	499.914	Lake Michigan
PCB95	1	11	145	San Diego Bay
PCB187	1	11	186.3	Tampa Bay
PCB180	1	12	350	Puget Sound
PCB52	1	8	996	Tampa Bay
PCB153_132_168	1	6	570.9	Tampa Bay
PCB70	0	14	585.736	Lake Michigan
PCB56_60	0	12	474.747	Lake Michigan
PCB45	0	8	37.589	Lake Michigan
PCB8_5	0	15	596.22	Lake Michigan
PCB201_173_157	0	14	13.4	San Diego Bay
PCB195_208	0	14	260	Puget Sound
PCB110_77	0	8	325.543	Lake Michigan
PCB28	0	13	1118.705	Lake Michigan
PCB18	0	13	588.921	Lake Michigan
PCB170_190	0	12	314.63	Delaware Bay
PCB138_160	0	10	706.4	Tampa Bay
PCB101_90	0	6	406	Boston Harbor
PCB118	0	8	382	Boston Harbor
PCB66	0	7	427.551	Lake Michigan
PCB206	0	7	612.639	St. Simon Sound (GA)
PCB128	0	6	238.5	Tampa Bay
PCB44	0	3	394.022	Lake Michigan
PCB105	0	2	696.9	Tampa Bay

PCBs in Coral Tissues

Concentrations of total PCBs in coral tissues ranged from below limits of detection to 3.52 ng/g (Figure 3.6) with a mean of 0.464 ng/g (Table 3. 5), which is orders of magnitude lower than the concentrations measured in sediments. The measured mean coral concentration of total PCBs in sampled coral tissues is consistent with concentrations found in other reefs of Puerto Rico, despite the fact that PCBs in sediments are very high in Guánica compared to other systems in the region. Both previous studies (Pait et al., 2008) and coral and sediment data from this study suggest a possible downstream concentration gradient of PCBs moving from Guánica Bay westward, but this does not appear to be resulting in unusually high PCB uptake by corals. There were no statistically significant differences between sampling strata for PCBs in coral tissues (Dunn Method, α=0.05).

There is no obvious source of PCBs in the watershed. Because PCBs have historically been used for a wide variety of applications (e.g. electrical applications, hydraulic fluids, pesticides), it may be difficult to identify the original source of these high levels of pollution.

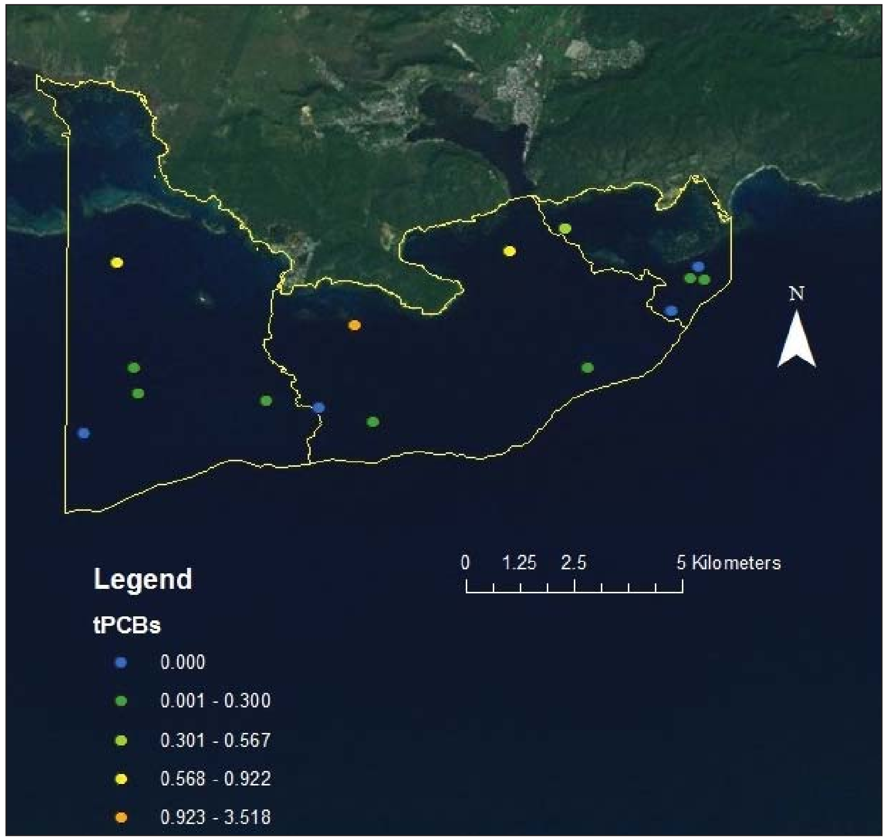

*Figure 3.6. Concentrations of total PCBs in coral (*Porites astreoides*).*

DDT in Sediment

Concentrations of total DDT (sum of parent compound and its degradation products, DDD and DDE) in sediments ranged from below detection limits to 69.25 ng/g (Figure 3.7), with a mean of 10.13 ng/g (Table 3.3). These observed concentrations are somewhat higher than observed in other locations on the south coast of Puerto Rico, but are lower than the highest values measured in the region (Vieques, Puerto Rico; see Table 3.4). When comparing measured concentration to published sediment quality guidelines, total DDT exceeded the ERL at 15 sites, and exceeded the ERM at two sites. Exceedances were limited to strata within the bay, but every site within the bay exceeded the ERL. Statistically, total DDT concentrations in the Central Bay are greater than central offshore and eastern offshore, and the North Bay concentrations are greater than the eastern offshore stratum (Dunn Method, α=0.05). While not included in the statistical analysis, concentrations at the watershed sites were relatively low when compared with the bay. This is relatively unexpected as the watershed should have historically been a source of DDT (see additional discussion of watershed sites below).

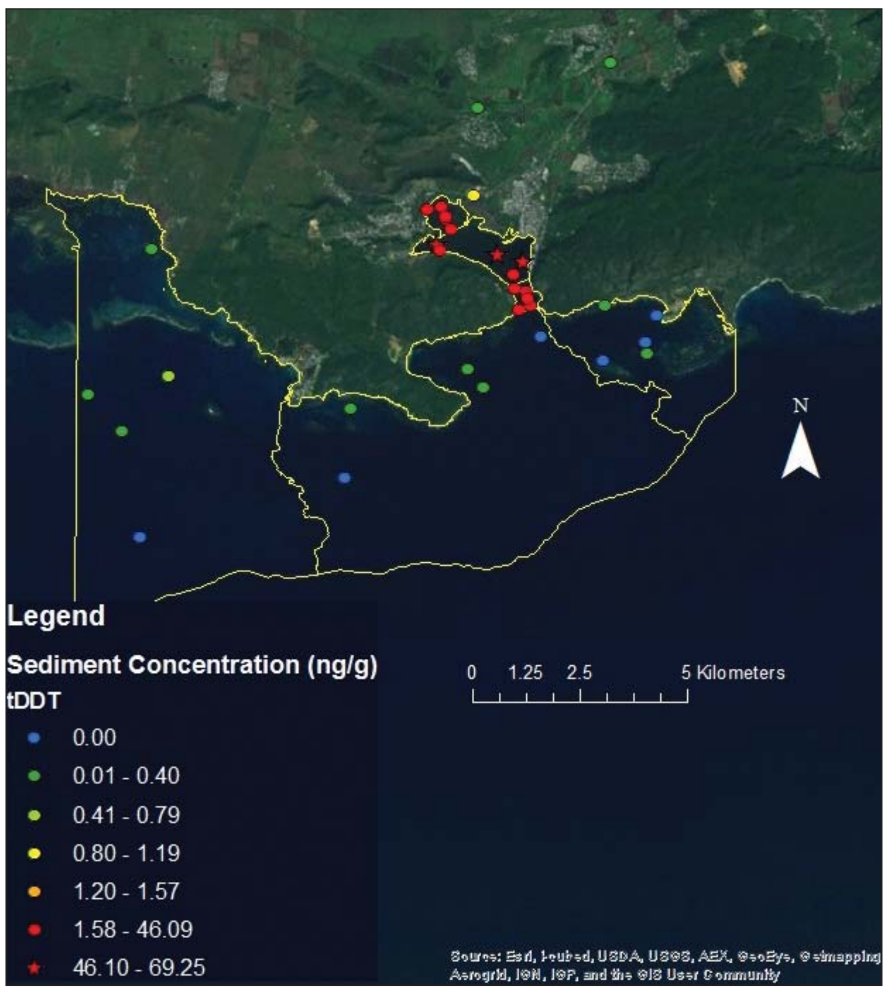

Figure 3.7. Concentrations of total DDT in sediments. Red indicates an exceedance of the ERL. Red stars indicate exceedance of ERM.

Because the measurement of total DDT is made up of both the parent isomers and degradation products, the ratio of parent compounds to degradation products can provide some insight into the relative age or "freshness" of the DDT present. Total DDT concentrations containing higher ratios of the parent compound are more likely to be recently introduced into the environment. The sites with the highest total DDT concentrations were made up primarily of degradation products. Parent material was more prevalent at two sites in the watershed (WS107, WS108) and two sites offshore (CO078a and WO101a), which had greater than two-third parent material. The sites with high parent material had relatively low concentrations so it is possible that this represents areas of slow DDT degradation, rather than a "new" source of DDT, such as a leaking container or illegal use.

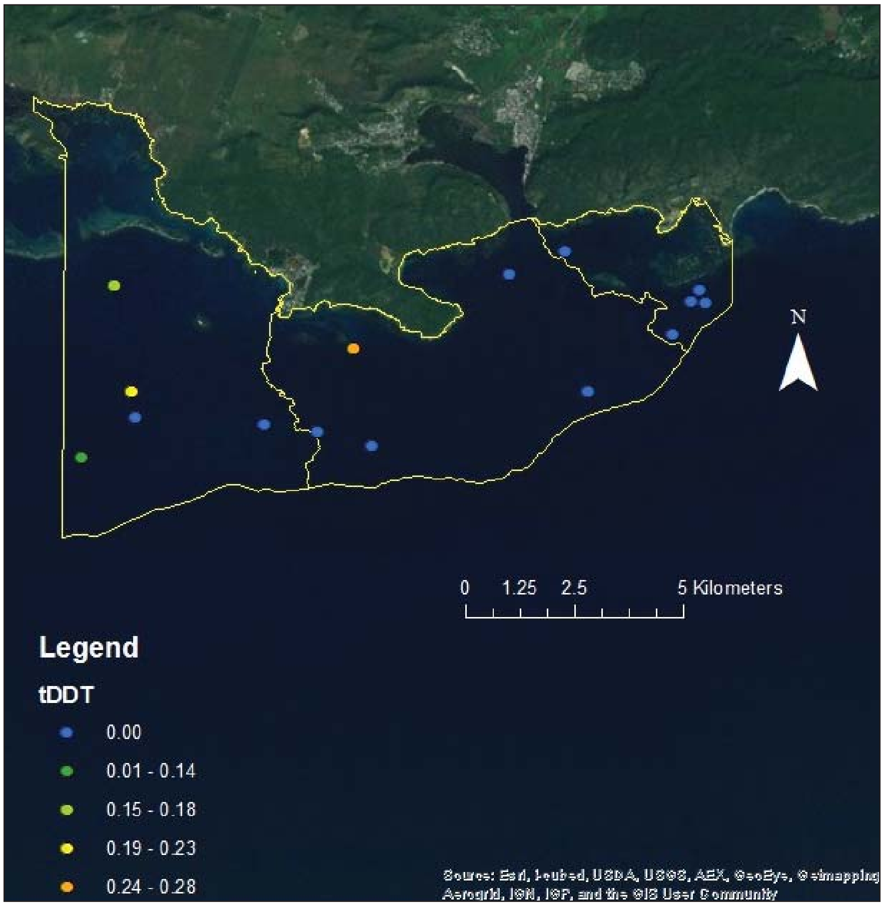

*Figure 3.8. Concentrations of total DDT in coral (*Porites astreoides*).*

DDT in Coral Tissue

Concentrations of total DDT in coral tissues ranged from below detection limits to 0.28 ng/g (Figure 3.8), with a mean of 0.06 ng/g (Table 3.5) which is two orders of magnitude lower than concentrations observed in the sediments. These observed concentrations are similar to concentrations observed in coral tissues in other locations on the south coast of Puerto Rico, but are lower than the highest values measured in the region (Vieques, Puerto Rico; see Table 3.6). No statistically significant differences in concentrations were observed between the three strata (Dunn Method, α=0.05).

A negative correlation was found between DDT concentration in coral tissues and longitude, further suggesting that DDT concentrations increase from east to west (Spearman Rho = -0.6096). Futher study is needed to link near shore oceanographic current data with contaminant data to understand how these pollutants are moving in the near coastal zone.

Despite the fact that DDT was banned in the United States in 1972, it is not surprising that it continues to be found in the environment. The agricultural use of banned organochlorines have been found throughout the Caribbean region until the late 1990s (Alegria et al., 2000). Due to the long time period required for sediment to flush out of the Guánica Bay/ Rio Loco watershed and the general environmental persistance of the breakdown products, DDT used in agriculture will continue to persist in the environment (Larsen and Santiago-Roman, 2001). Furthermore, the transport and atmospheric deposition of African dust in the Caribbean has been noted as a potential source of organochlorine pesticides such as DDT (Nipper et al., 2006).

HCH in Sediments
Sediment concentrations of total HCH ranged from below limits of detection to 0.25 ng/g (Figure 3.9), with a mean of 0.06 ng/g (Table 3.3). This is similar to what has been observed in other studies in Puerto Rico (Table 3.4). No sediment quality guidelines exist for HCH. There were not statistically significant differences between the strata, although some of the higher individual values occurred offshore (Figure 3.9), in contrast to what was observed with most other contaminants.

HCH in Coral Tissues
Total HCH in coral tissues ranged from below limits of detection to 1.28 ng/g (Figure 3.10) with a mean of 0.09 ng/g (Table 3.3). Coral tissue concentrations from study sites to the east (Jobos Bay) and west (La Parguera) did not contain any detectable HCH concentrations (Pait et al., 2009; Whitall et al., 2011). A study conducted in Vieques detected a mean HCH concentration of 0.00346 ng/g in coral tissues (Pait et al., 2010; Table 3.6). There are no statistically significant differences between the strata (Dunn Method, α=0.05). This environmentally persistent insecticide may be present in the system due to historical use in upland agriculture and transported to the reefs via Rio Loco discharge into Guánica Bay.

Chlordane in Sediments
Concentration of total chlordane ranged from below detection levels to 7.39 ng/g (Figure 3.11) with a mean of 0.49 ng/g (Table 3.3). This is higher than previously observed in other regions of Puerto Rico (Table 3.4). When compared with historical, national sediment concentrations from the NS&T program two chemical compounds that make up total chlordane (cis-nonachlor and oxychlordane) had two sites each in the top 10% of all historically measured values, and a third chlordane compound (heptachlor) had one site that was in the top 10 values ever measured nationally. Overall, 17 sites exceeded the ERL, including one site in the North Bay which exceeded the ERM. Most of the sites exceeding SQG were in the bay, but there were three sites offshore (central and western strata) which exceeded the ERL. This was one of only a few contaminants for which high concentrations were observed offshore. There are no statistically significant differences between the strata (Dunn Method, α=0.05).

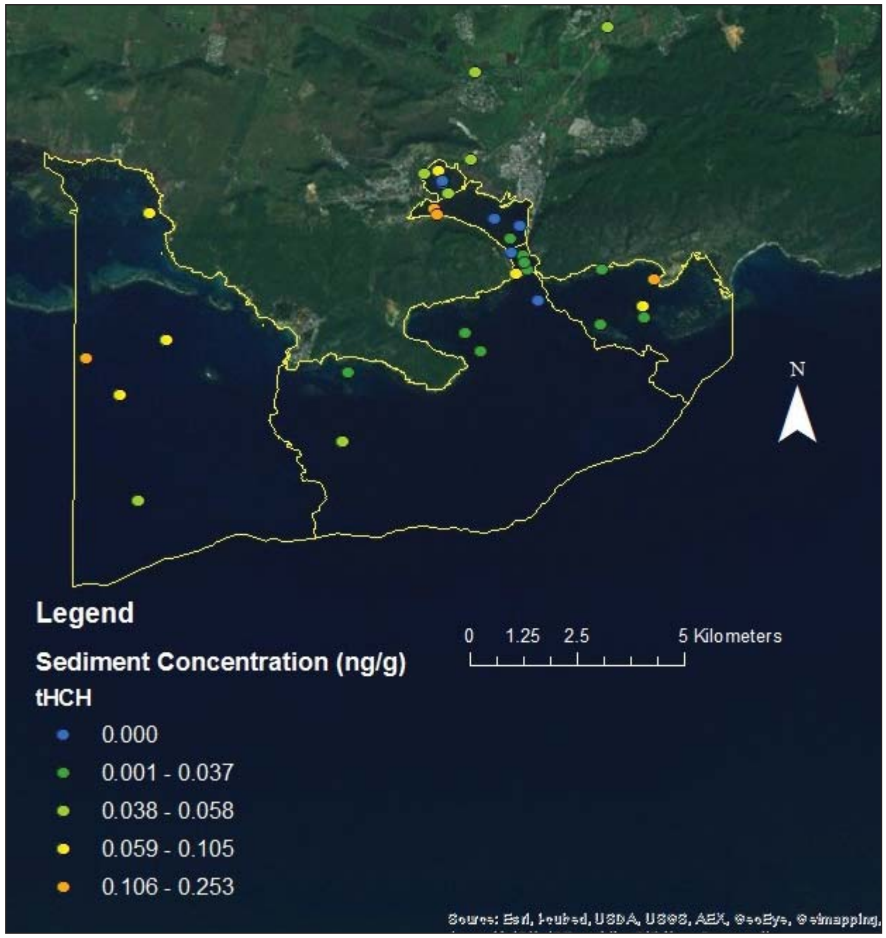

Figure 3.9. Concentrations of total HCH in sediments.

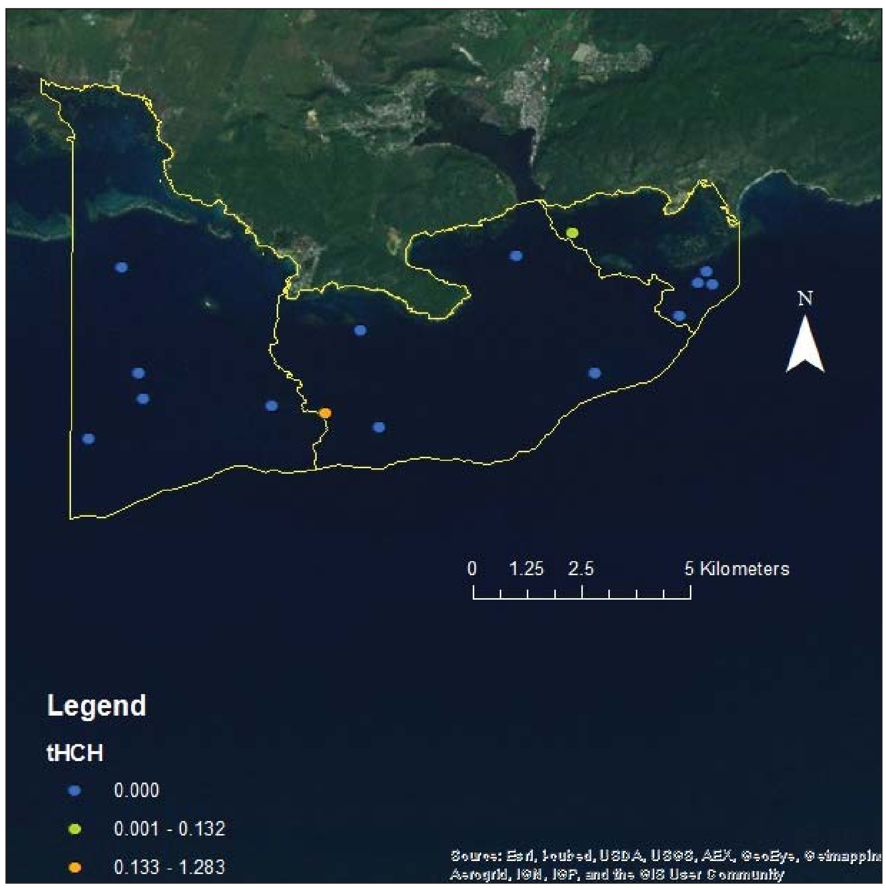

*Figure 3.10. Concentrations of total HCH in coral (*Porites astreoides*).*

Chlordane in Coral Tissues

Chlordane was detected at low levels in coral samples, ranging from below levels of detection to 0.52 ng/g (Figure 3.12) with a mean total concentration of 0.04 ng/g (Table 3.4). Chlordane isomers alpha-chlordane and oxychlordane were both detected in this study. There are no statistically significant differences between the strata (Dunn Method, α=0.05). Previous coral contaminant studies east and west of Guánica Bay did not detect chlordane in coral tissue samples (Table 3.6; Pait et al., 2009). A contaminant study of corals in Vieques, Puerto Rico measured a mean chlordane concentration of 0.12 ng/g (Table 3.6; Pait et al., 2010). Although restricted in 1983 and banned in 1988, chlordane was widely used in the U.S. prior to the ban and is environmentally persistent.

Other Pesticides in Sediments

Other pesticides or pesticide degradation products that were detected in sediments included: aldrin, dieldrin, endrin, tetrachlorobenzene, pentachlorobenzene, hexachlorobenzene, pentachloroanisole, endosulfan II, endosulfan sulfate, chlorpyrifos and mire. Spatial distribution of these contaminants are shown in Figures B.1 to B.11 in Appendix B. There are two isomers of tetrachlorobenzene; only the 1,2,3,4-tetrachlorobenzene isomer was detected in this study. Endosulfan I was not detected at any sites. With a few exceptions (discussed below), these detections were at a few sites, primarily in the bay and watershed, and at relatively low concentrations.

Endosulfan sulfate was only detected at two sites (one in the watershed and one in the North Bay), but these concentrations were both above the NS&T national mean (0.14 ng/g). Chlorpyrifos was only detected at 5 sites, but three of these sites were above the NS&T national mean (0.15 ng/g). Pentachloroanisole, a degradation product of the fungicide pentachlorophenol, was the most widely distributed of these compounds and had numerous observations above the NS&T national mean (0.07 ng/g). Hexachlorobenzene was also detected at a 11 of 33 sites, including three sites (one each in western offshore, central offshore and Central Bay strata) which exceeded the national NS&T average (0.65 n/g). These pesticides include legacy fungicides and insecticides that persist in the environment despite their use being phased out.

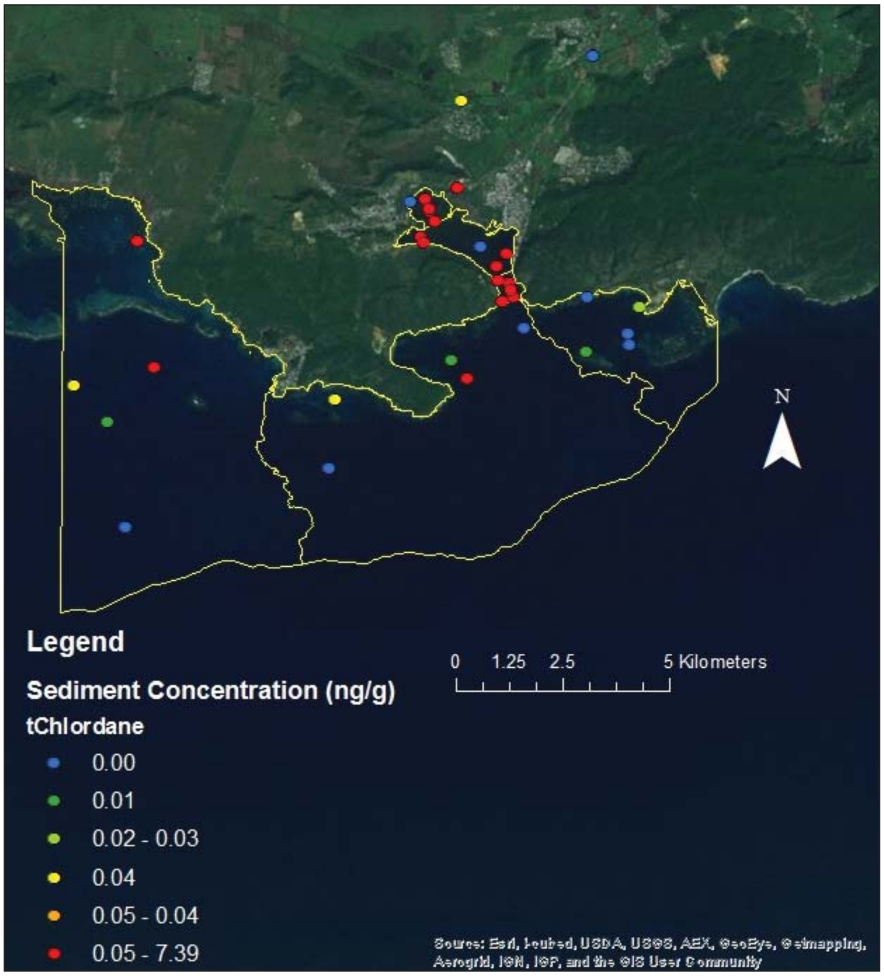

Figure 3.11. Concentrations of total chlordane in sediments. Red indicates an exceedance of the ERL.

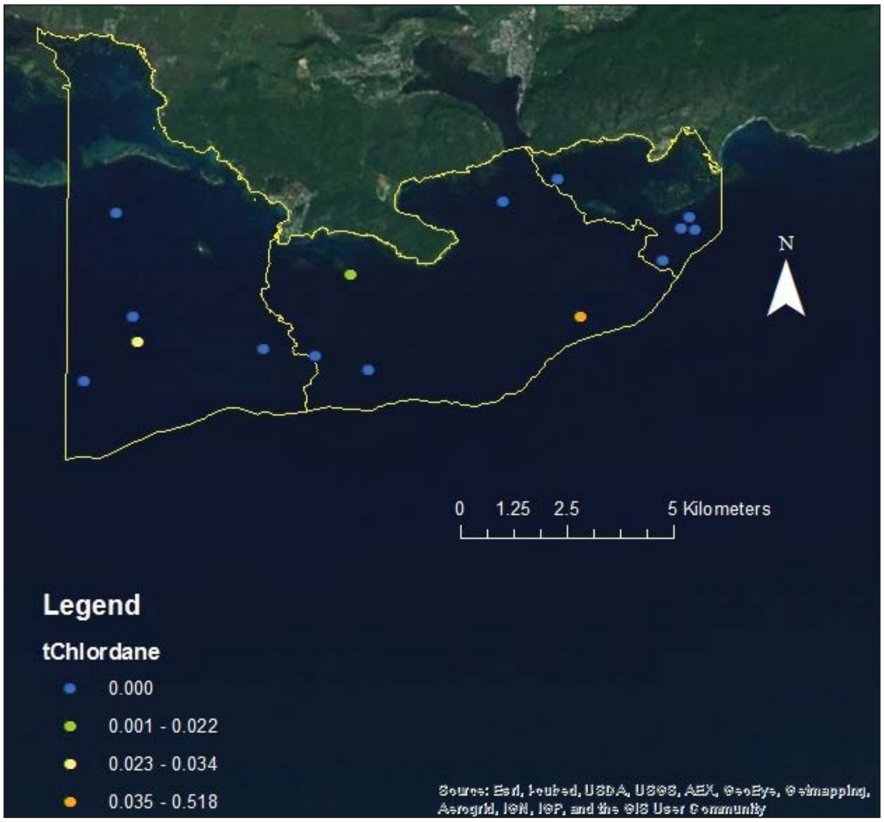

*Figure 3.12. Concentrations of total chlordane in coral (*Porites astreoides*).*

Butyltins in Sediments

Tributyltin (TBT) in sediments ranged from below limits of detection to 0.93 ng/g (Figure 3.13), with a mean of 0.16 ng Sn/g. TBT breakdown products (dibutyltin (DBT) and monobutyltin (MBT)) were also detected in similar concentrations (Table 3.3). There are no statistically significant differences between the strata (Dunn Method, α=0.05) for MBT or DBT. Concentrations of TBT in the Central Bay are greater than the North Bay, central offshore and western offshore strata. These concentrations are within the range of what was observed in southwest Puerto Rico and Vieques, but lower than observed in Jobos Bay (Table 3.4). Tetrabutyltin was not detected in sediments at any sites. Butyltins were not well correlated with elemental tin, suggesting that elemental tin originates from a source other than the breakdown of organotins.

Butyltins in Coral Tissues

No butyltins were detected in coral tissues at any site (data not shown). Butyltins have been previously been detected in coral tissues in other locations in the region (Pait et al., 2007; Pait et al., 2010).

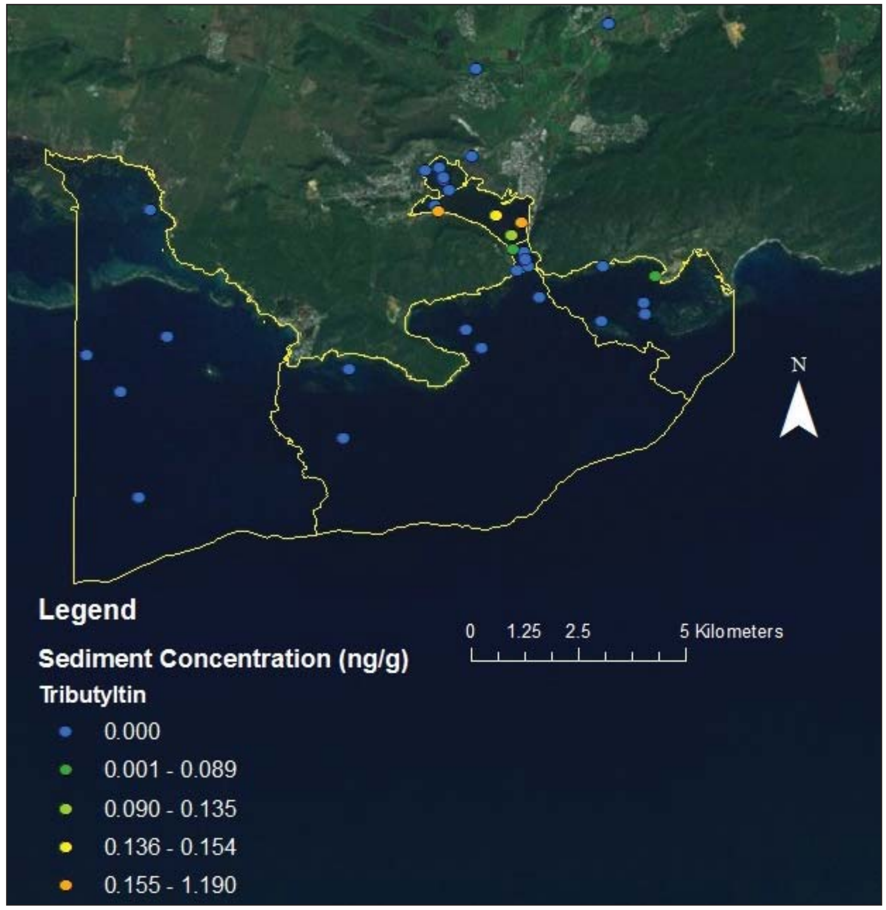

Figure 3.13. Concentrations of TBT in sediments.

Guánica Bay Metals

Sixteen trace and major elements were analyzed in sediments collected from Guánica Bay and the surrounding coastal area. A summary of the means and standard errors for the elements are shown in Table 3.1. The highest mean concentrations of all trace and major elements are those of silicon (73,832 µg/g), iron (22,821 µg/g) and aluminum (21,101 µg/g). Aluminum, iron and silicon are all common elements in the earth's crust, and as such it is not surprising to see higher concentrations of these elements relative to the other 13 trace or major elements. This is comparable with results from Vieques, Puerto Rico (Pait et al., 2010), which found that mean concentrations of the three highest trace or major elements in sediments were aluminum, iron and silicon, respectively.

A discussion of six elements, arsenic, chromium, copper, mercury, nickel and zinc in sediments follows. Brief summaries of the remaining 10 elements are also provided.

Arsenic in Sediments

Concentrations of arsenic in the sediments of Guánica Bay ranged from 1.28 µg/g to 12.80 µg/g (Figure 3.14), with a mean of 4.77 µg/g (Table 3.8), which is within the range of values observed for other systems in Puerto Rico (Table 3.4). There were three sites that exceeded the ERL (8.2 µg/g). No sediments analyzed in this study exceeded the ERM for arsenic.

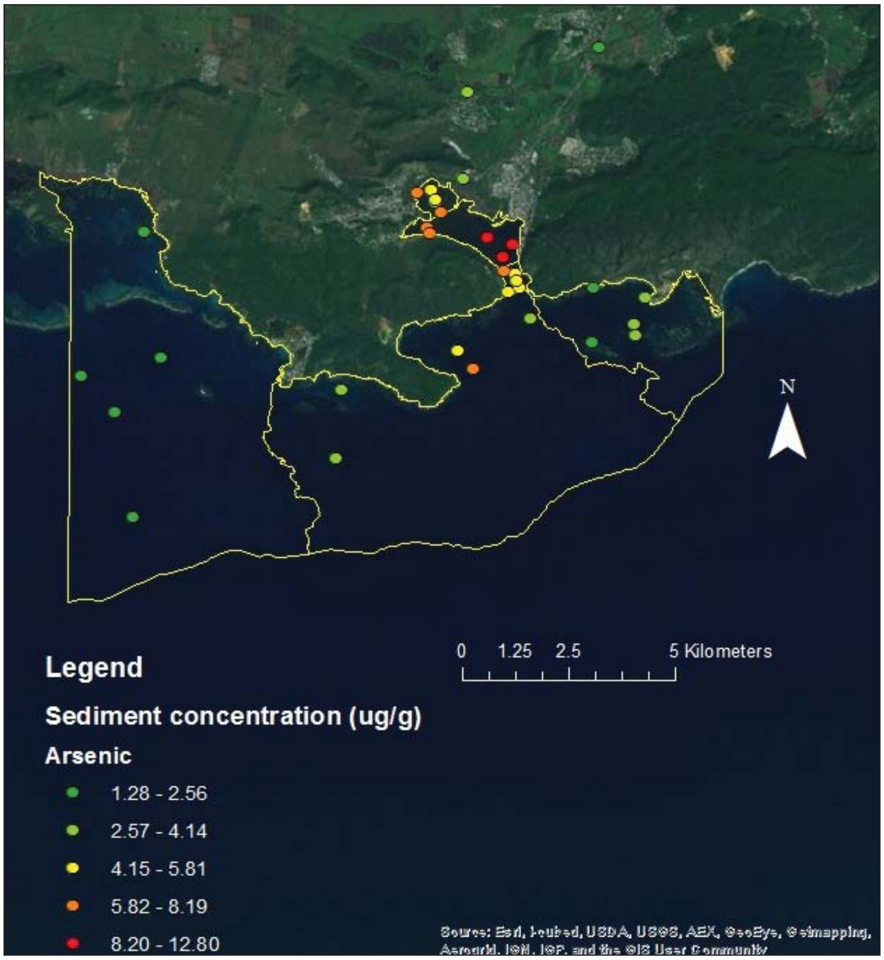

Figure 3.14. Concentrations of arsenic in sediments. Red indicates an exceedance of the ERL.

The highest arsenic sediment concentration of 12.8 µg/g was found in the Central Bay at site CB011. Along with CB011, Central Bay sites CB013 and CB015 (10.4 and 8.35 µg/g respectively) represented the highest concentrations of arsenic in the bay.

Table 3.8. Summary statistics of trace elements in sediments in Guánica Bay.

Element	Mean	Median	Min	Max	St Dev
Ag	0.02	0	0	0.17	0.04
Al	23,880	7,820	613	62,800	24,689
As	4.77	4.14	1.28	12.80	2.73
Cd	0.02	0	0	0.11	0.03
Cr	286.46	71.50	3.48	1,930.00	422.96
Cu	29.18	8.98	0.89	102.00	31.72
Fe	27,431	8,400	705	94,000	29,331
Hg	0.04	0.02	0	0.19	0.05
Mn	470	402	38	2,020	435
Ni	228.71	57.70	4.88	709.00	262.90
Pb	6.08	2.93	0.20	31.90	7.21
Sb	0.14	0.07	0	0.43	0.16
Se	0.15	0.11	0	0.51	0.17
Si	84,726	30,900	2,090	220,000	88,497
Sn	0.83	0	0	3.93	1.12
Zn	46.71	19.70	2.21	153.00	47.26

Statistically, the Central Bay had higher arsenic concentrations than the western and eastern offshore strata (Dunn Method, α=0.05).

Arsenic in Coral Tissues

Arsenic concentrations in coral tissues ranged from 0.91 µg/g to 2.37 µg/g (Figure 3.15), with a mean of 1.42 µg/g (Table 3.9). This is similar to what has been observed in other systems in Puerto Rico (Table 3.6). There were no statistically significant differences between strata for arsenic (Dunn Method, α=0.05). In addition to natural (crustal) sources, arsenic can enter the coastal environment from a variety of sources including treated wood, insecticides and herbicides, all of which may be contributing to the arsenic budget of Guánica Bay.

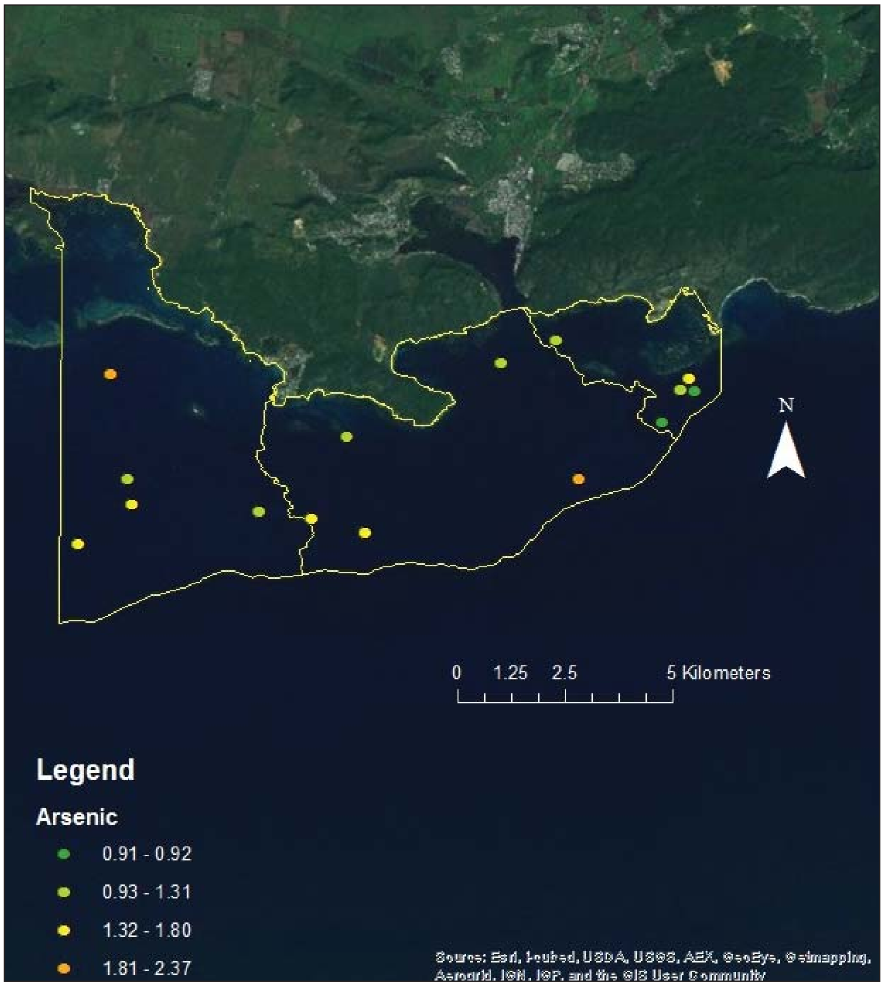

*Figure 3.15. Concentrations of arsenic in coral (*Porites astreoides*).*

Table 3.9. Summary statistics of trace elements in coral tissues in Guánica Bay.

	Mean	Median	Min	Max	St Dev
Ag	0.004	0	0	0.055	0.014
Al	68.51	50.50	24.00	140.00	37.84
As	1.42	1.31	0.91	2.37	0.44
Cd	0.27	0.29	0	0.33	0.08
Cr	0	0	0	0	0
Cu	22.47	16.00	0	76.90	20.18
Fe	100.73	59.30	28.90	480.00	113.95
Hg	0.002	0.002	0.001	0.004	0.001
Mn	10.48	8.66	3.73	21.20	5.48
Ni	4.66	4.79	0.68	11.60	2.83
Pb	0.13	0.12	0.07	0.32	0.06
Se	0.19	0.20	0.13	0.25	0.04
Sn	0.07	0	0	0.27	0.10
Zn	9.42	10.30	1.82	25.20	6.41

Chromium in Sediments

The mean concentration of chromium found in the sediments of Guánica Bay ranged from 3.48 µg/g to 1930 µg/g (Figure 3.16) with a mean of 286.5 µg/g (Table 3.8). The three highest concentrations of chromium for this study were all found at the watershed sites with the highest concentration being 1,930 µg/g found at site WS108. This is an order of magnitude higher than what has been measured in other sites in Puerto Rico (Table 3.4). When comparing chromium data from this study with historical data from NOAA's NS&T Program it is noted that 13 sites from the bay and the watershed were in the top 10% of all measured sediment chromium concentrations (nationwide, since 1986). One sediment chromium value from Guánica Bay (western most watershed site) was among the top 10 highest individual values ever measured by NS&T. It is clear that sediment chromium concentrations in Guánica Bay are very high, not only for the region, for the nation as well. Sixteen of the 33 sites analyzed for this study exceeded the ERL for chromium. Twelve sites met or exceeded the ERM of 370 µg/g with the highest concentration exceeding the ERM by a factor of five.

Statistically, the North Bay had higher chromium concentrations than the western and eastern offshore strata; the Central Bay had higher concentrations than the eastern offshore stratum (Dunn Method, α=0.05).

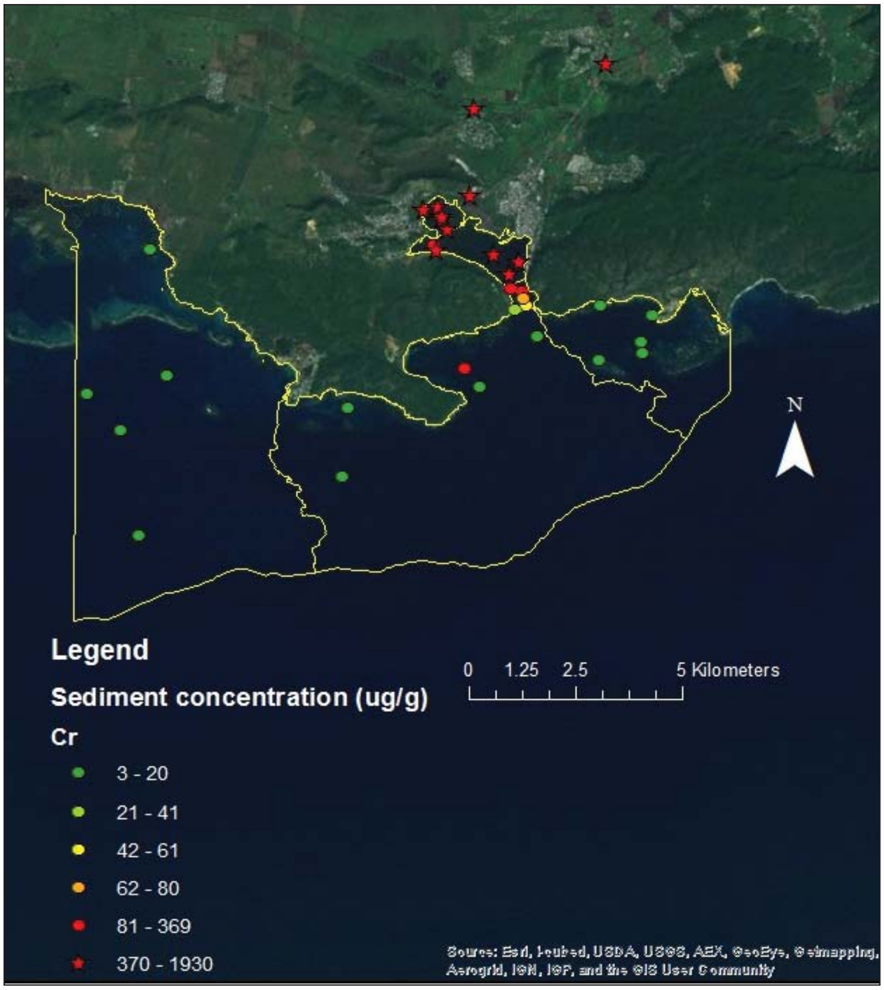

Figure 3.16. Concentrations of chromium in sediments. Red indicates an exceedance of the ERL. Red stars indicate exceedance of ERM.

Chromium in Coral Tissues

Chromium was not detected in any of the coral samples around Guánica Bay (detection limit 0.102 µg/g; data not shown). Coral samples around Jobos Bay and southwest Puerto Rico (Whitall et al., 2011, Pait et al., 2007) also did not detect chromium although a study in Vieques, an island off the eastern coast of Puerto Rico, measured a mean chromium concentration of 0.18 µg/g (Table 3.6; Pait et al., 2009; Pait et al., 2010).

The apparent disconnect between extremely high sediment chromium concentrations and concentrations in coral tissues below limits of detection, in combination with the spatial pattern of chromium in sediments, suggests that chromium pollution is, in general, not reaching the offshore coral reef systems. More research is needed (e.g., water circulation differences between sites) in order to understand this pattern. Chromium in the environment more commonly comes from point sources (discussed below) rather than non-point sources.

Copper in Sediments
Copper concentrations in sediments ranged from 0.89 µg/g to 102 µg/g (Figure 3.17), with a mean concentration of 29.18 µg/g (Table 3.9), which is similar to what has been observed in other studies in Puerto Rico (Table 3.6). The three highest copper concentrations detected in Guánica Bay were all located in the North and Central Bay with the highest concentration being 102 µg/g at CB012.

Thirteen sites in Guánica Bay exceeded the ERL (34 µg/g). No sites exceeded the ERM. All of these exceedances were in the bay or in the watershed.

Statistically, the North Bay had higher copper concentrations than all of the offshore strata; the Central Bay had higher concentrations than the central offshore stratum (Dunn Method, α=0.05).

Copper in Coral Tissues
Copper concentrations in coral tissues ranged from below limits of detection to 76.9 µg/g (Figure 3.18), with a mean of 22.5 µg/g (Table 3.9) which is within the range of values observed in other systems in Puerto Rico (Table 3.6). There were no statistically significant differences between strata for copper concentrations in coral tissues (Dunn Method, α=0.05).

Non-point sources of copper include agricultural pesticides, anti-fouling boat paint and dust from automobile brake pads all of which may contribute to the copper budget of the Guánica system.

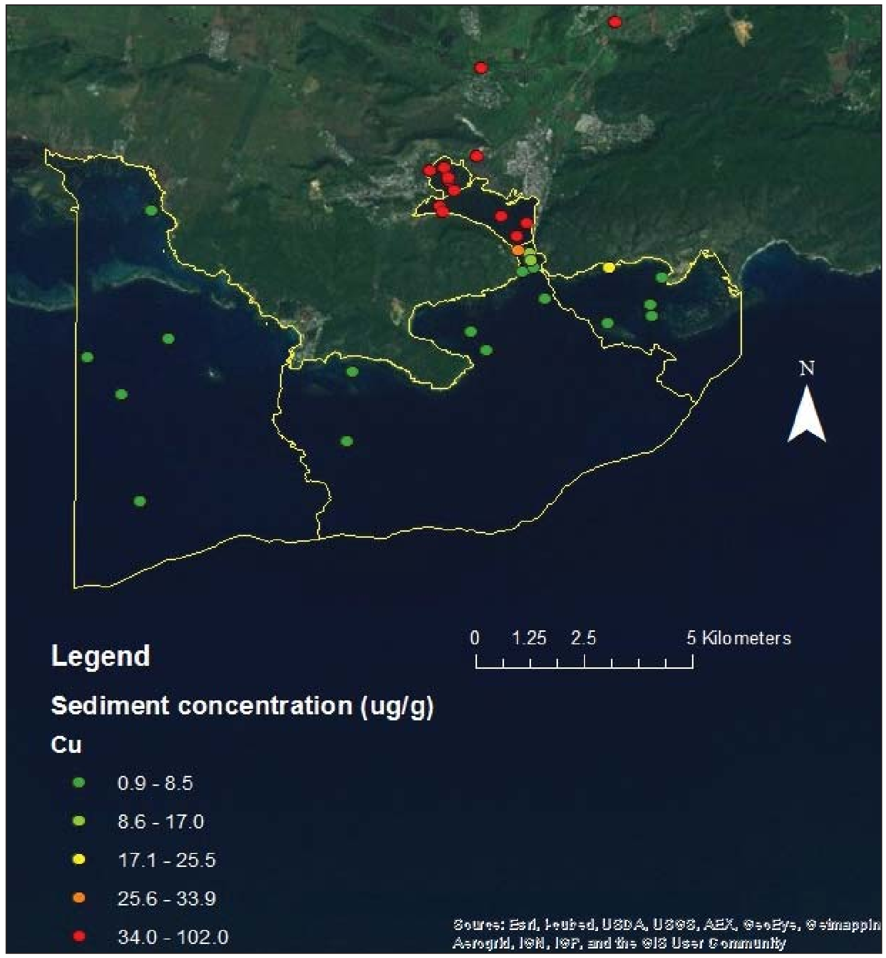

Figure 3.17. Concentrations of copper in sediments. Red indicates an exceedance of the ERL.

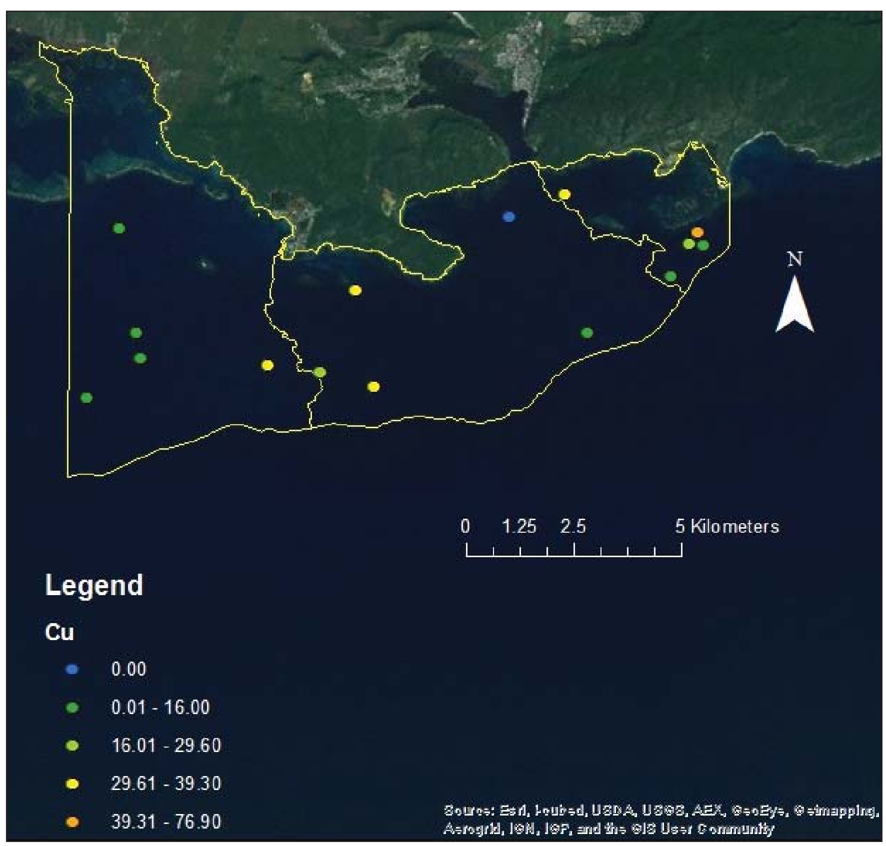

*Figure 3.18. Concentrations of copper in coral (*Porites astreoides*).*

Mercury in Sediments

Detected concentrations of mercury from Guánica Bay sediments ranged from 0.001 µg/g to 0.186 µg/g (Figure 3.19), with a mean of 0.04 µg/g (Table 3.8), which is similar to what has been measured in other studies in Puerto Rico (Table 3.4). The highest concentration of mercury in sediments in Guánica Bay was 0.1860 µg/g at site CB012. The Central Bay accounted for four of the top five concentrations of mercury in the study area.

Two sites in the Central Bay exceeded the ERL (0.15 µg/g). No sites exceeded the ERM. Statistically, the Central Bay had higher mercury concentrations than all of the offshore strata; the North Bay had higher concentrations than the eastern offshore stratum (Dunn Method, α=0.05).

Mercury in Coral Tissues

Mercury concentrations in coral tissues ranged from below 0.001 to 0.004 µg/g (Figure 3.20), with a mean of 0.002 µg/g (Table 3.9), which is similar to what has been measured at other locations in Puerto Rico (Table 3.6). There were not statistically significant differences between strata for mercury concentrations in coral tissue (Dunn Method, α=0.05). Atmospheric deposition may be an important non-point source of Hg in this system.

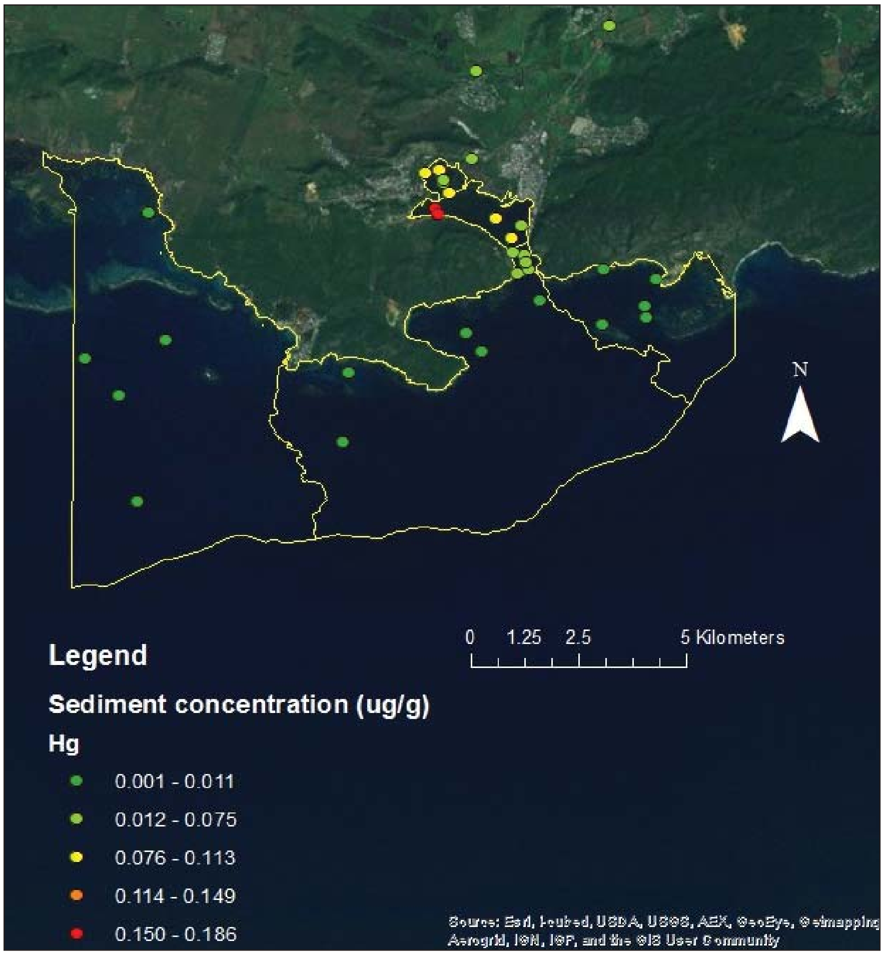

Figure 3.19. Concentrations of mercury in sediments. Red indicates an exceedance of the ERL.

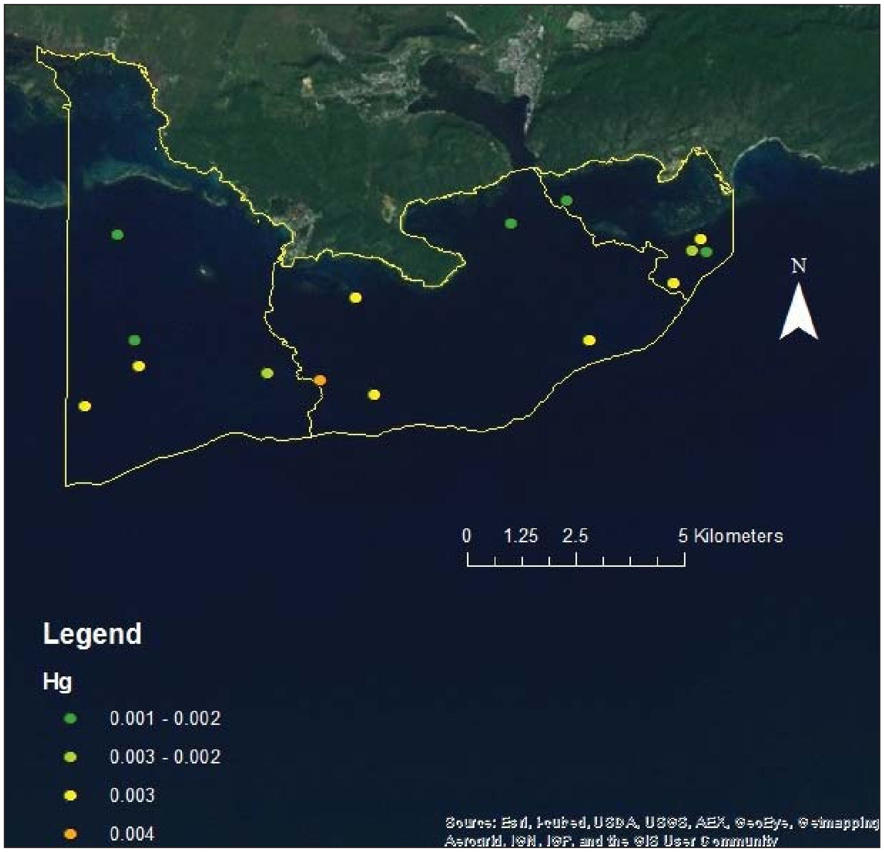

*Figure 3.20. Concentrations of mercury in coral (*Porites astreoides*).*

Nickel in Sediments

Concentrations of nickel detected in the sediments of Guánica Bay ranged from 4.88 to 709 µg/g (Figure 3.21), with a mean of 228.71 µg/g (Table 3.8), which is an order of magnitude higher than nickel sediment values observed elsewhere in the region (Table 3.4). In order to put this in a national and historical context, these values were compared with NOAA NS&T sediment data. Including two sites in Guánica Bay from a previous study (Pait et al., 2007), 19 sites from the bay and the watershed were in the top 10% of all measured sediment nickel concentrations (nationwide, since 1986). The 14 highest individual values ever measured by NS&T were from Guánica Bay. It is clear that sediment nickel concentrations in Guánica Bay are very high, not only for the region, for the nation as well. The two highest concentrations of nickel in this study were detected in the sediments of the watershed at sites WS107 and WS108 (708 and 709 µg/g respectively). The North Bay had the highest nickel concentrations for non-watershed sites in the study area with four of the top five highest detections. Statistically, the North Bay had higher nickel concentrations than all of the offshore strata; the Central Bay had higher concentrations than the central offshore stratum (Dunn Method, α=0.05).

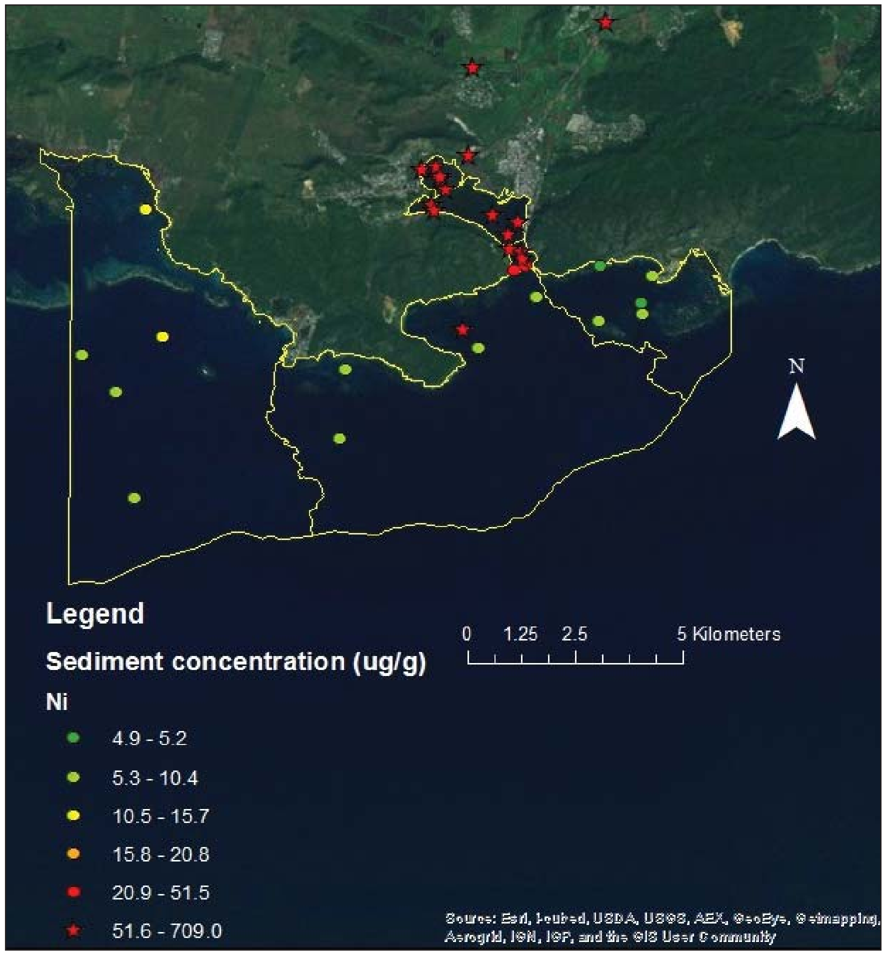

Figure 3.21. Concentrations of nickel in sediments. Red indicates an exceedance of the ERL. Red stars indicate exceedance of ERM.

Seventeen of 33 sites in this study exceeded the ERM (51.6 µg/g) and an additional two sites exceeded the ERL (20.9 µg/g), meaning that almost 60% of the sites samples in this region have the possibility of sediment toxicity due to nickel. These exceedances were primarily in the bay and watershed, although one site in the central offshore stratum exceeded the ERM.

Statistically, the North Bay had higher nickel concentrations than the western and eastern offshore strata; the Central Bay had higher concentrations than the eastern offshore stratum (Dunn Method, α=0.05).

Nickel in Coral Tissues

Concentrations of nickel in coral tissues ranged from 0.68 µg/g to 11.6 µg/g (Figure 3.22), with a mean of 4.66 µg/g (Table 3.9), which is an order of magnitude higher than observed in coral tissues in other studies in Puerto Rico (Table 3.6). There were no statistically significant differences between strata (Dunn Method, α=0.05). Levels of nickel observed in coral tissues, in combination with the observed high sediment nickel concentrations, suggests that the nickel pollution coming from the bay (via the watershed) is reaching the coral reef ecosystem offshore and being incorporated into the tissues above levels that would be asso-

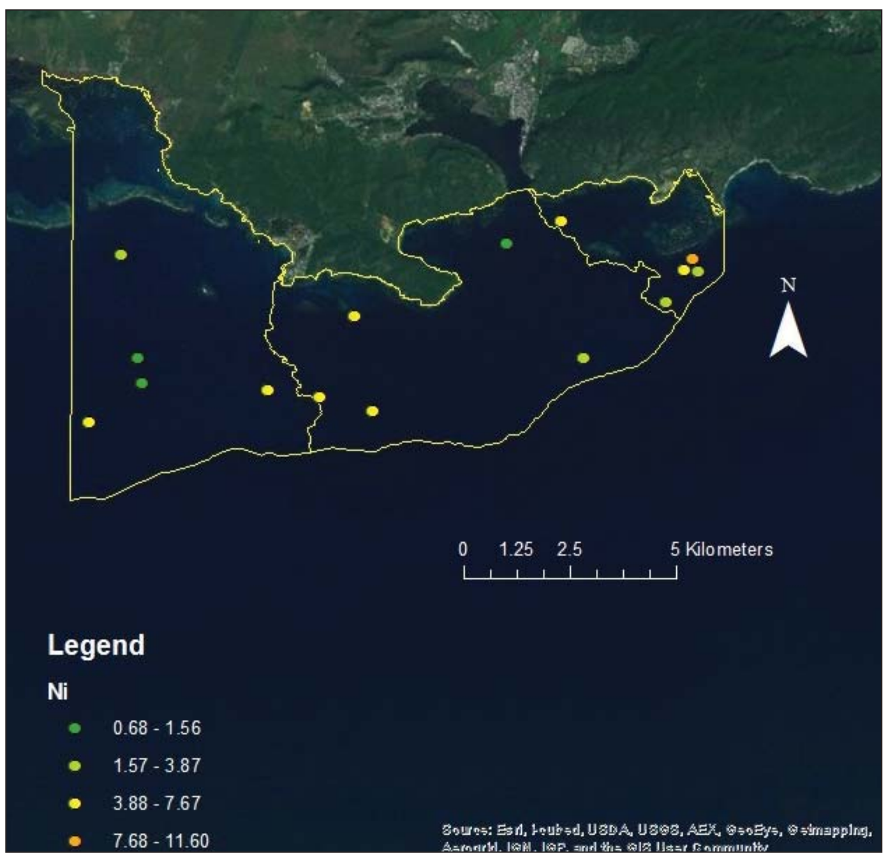

Figure 3.22. Concentrations of nickel in coral (mustard hill coral).

ciated with normal crustal erosion. Potential point sources of nickel are discussed later in the chapter. It is not clear what the likely physiological impact elevated nickel might have on corals; more research is needed to determine potential thresholds above which organismal harm is likely.

Zinc in Sediments

Detected concentrations of zinc in the sediments of Guánica Bay ranged from 2.21 µg/g to 153 µg/g (Figure 3.23), with a mean concentration of 46.71 µg/g (Table 3.8), which is slightly higher than other observed values for the region (Table 3.4). Site WS108 (153 µg/g) exceeded the ERL (150 µg/g). All other sites sampled for zinc in this study were below established guidelines.

Two of the five highest concentrations of zinc were found at the watershed sites WS108 (153 µg/g) and WS106 (108 µg/g). The remaining three were found at CB012 (117 µg/g), NB003 (113 µg/g), and CB015 (104 µg/g). Statistically, the North Bay had higher zinc concentrations than any of the offshore strata; the Central Bay had higher concentrations than the central and western offshore strata (Dunn Method, α=0.05).

Zinc in Coral Tissues

Zinc concentrations in coral tissues ranged from 1.82 µg/g to 25.2 µg/g (Figure 3.24), with a mean of 9.42 µg/g (Table 3.9). This is similar to what has been observed in other sites in the region (Table 3.6). There were no statistically significant differences between strata for zinc (Dunn Method, α=0.05). Automobiles (i.e., tire dust) may be an important non-point source of zinc to this system (Councell et al., 2004).

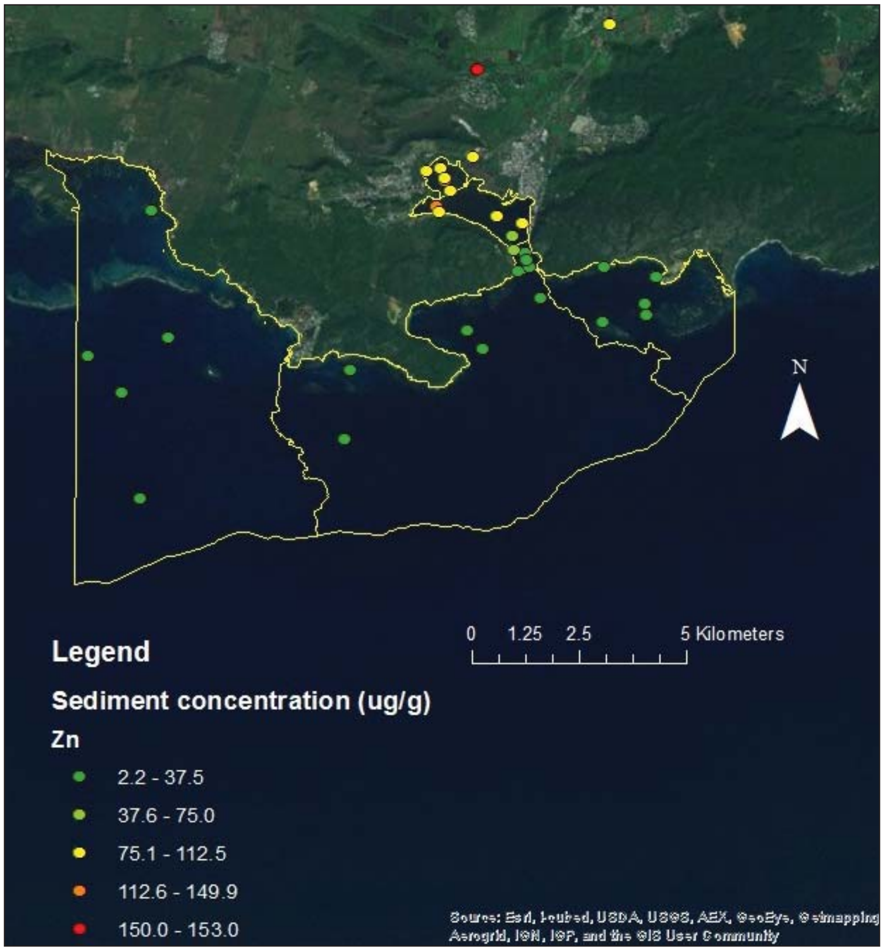

Figure 3.23. Concentrations of zinc in sediments. Red indicates an exceedance of the ERL.

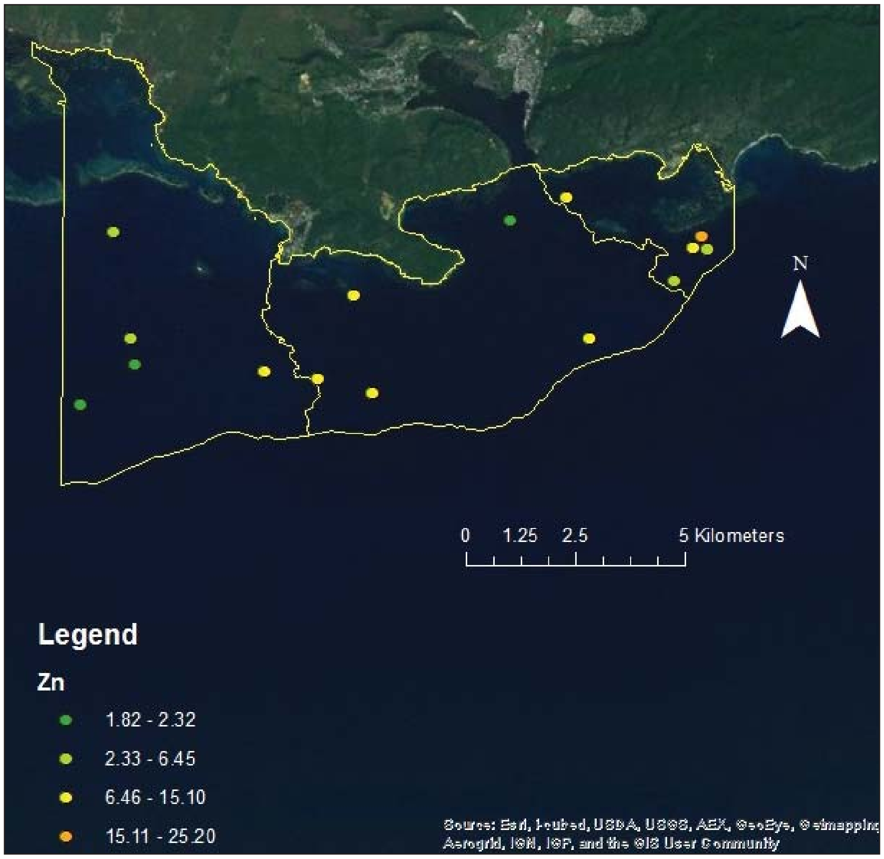

*Figure 3.24. Concentrations of zinc in coral (*Porites astreoides*).*

Other Trace and Major Elements

Ten other trace and major elements were analyzed as part of this study in Guánica Bay. Some of the results are briefly summarized below.

Aluminum

The highest concentration of aluminum detected in this study is from the North Bay site NB003 (62,800 µg/g, Figure 3.25), and the mean was 23,880 µg/g (Table 3.8), which is similar to what has been observed in other sites in the region (Table 3.4). Statistically, the North Bay had higher aluminum concentrations than any of the offshore strata (Dunn Method, α=0.05).

Aluminum in Coral Tissues

Aluminum was detected in coral tissues at values orders of magnitude lower than observed in the sediments (Figure 3.26), with a mean of 68.51 µg/g (Table 3.9) and at similar levels to what has been observed elsewhere in the region (Table 3.6). There are no statistically significant differences between strata for aluminum (Dunn Method, α=0.05).

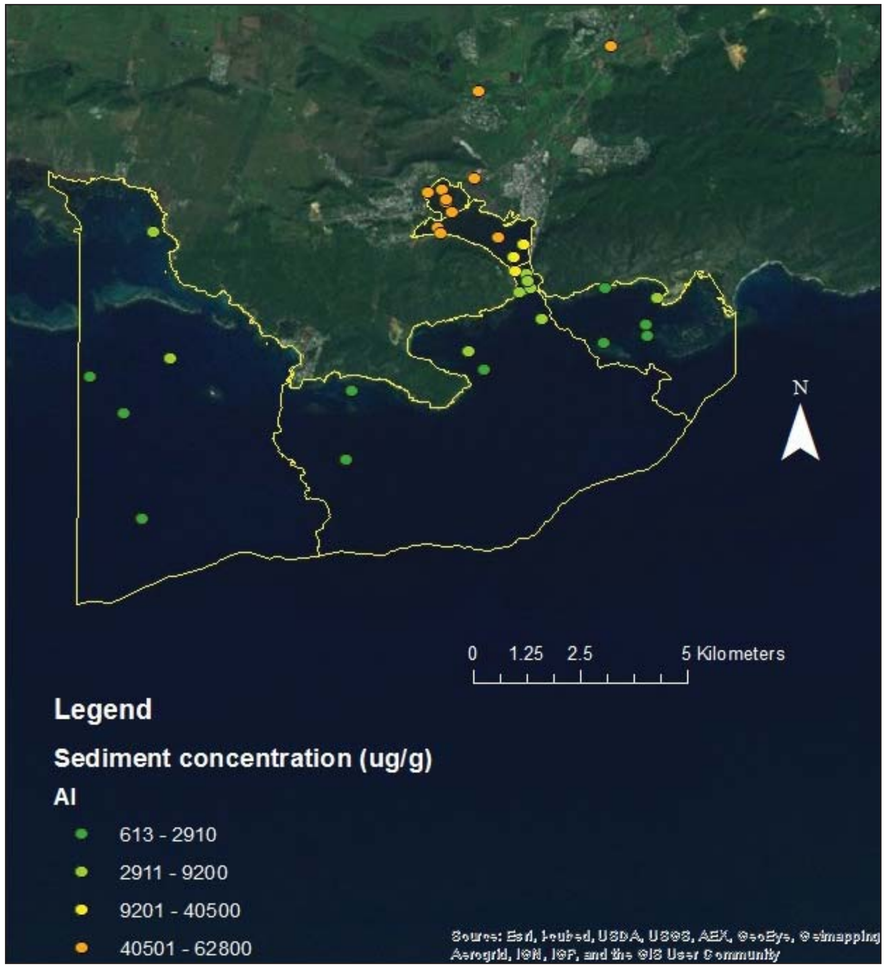

Figure 3.25. Concentrations of aluminum in sediments.

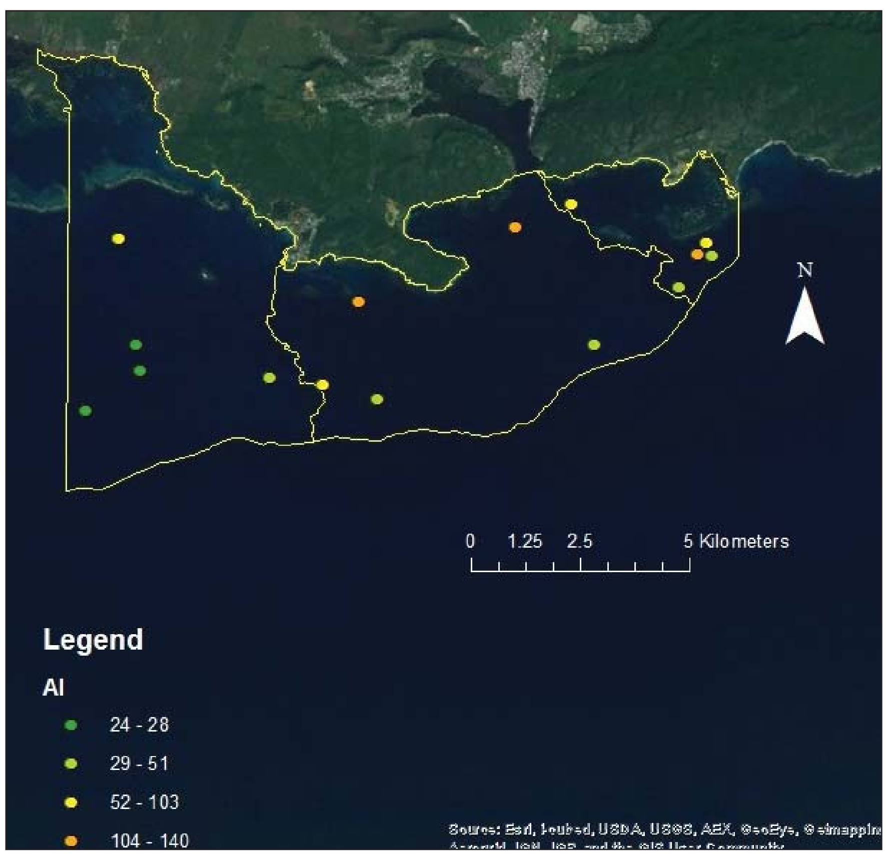

Figure 3.26. Concentrations of aluminum in coral (mustard hill coral).

Antimony in Sediments

Concentration of antimony in Guánica Bay sediments is ranged from below limits of detection to 0.433 µg/g (Figure 3.27), with a mean of 0.143 µg/g (Table 3.8), which is similar to what has been observed at other sites in the region (Table 3.4). The highest concentration of antimony detected is 0.433 µg/g at the North Bay site NB005. Statistically, the North Bay had higher antimony concentrations than any of the offshore strata and the South Bay stratum (Dunn Method, α=0.05).

Antimony in Coral Tissues

Antimony was not measured in coral tissues.

Cadmium in Sediments

Concentrations of cadmium in sediments from Guánica Bay ranged from below detection limits to 0.108 µg/g (Figure 3.28), with a mean of 0.017 µg/g (Table 3.8), which is within the range of values observed within the region (Table 3.4). No Guánica Bay sediment sites exceeded any thresholds or guidelines for cadmium. The highest observed concentration was found at watershed site WS106. The three watershed sites represented three of the top four detected concentrations of cadmium in Guánica Bay. There are no significant differences between strata for cadmium (Dunn Method, α=0.05).

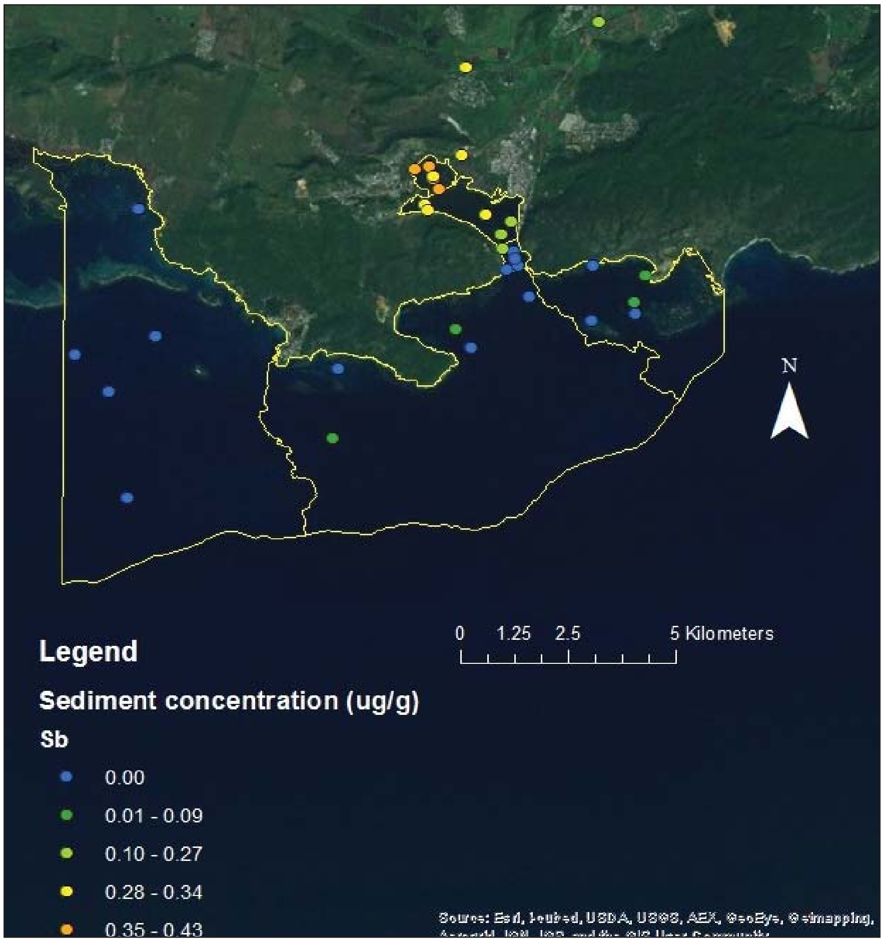

Figure 3.27. Concentrations of antimony in sediments.

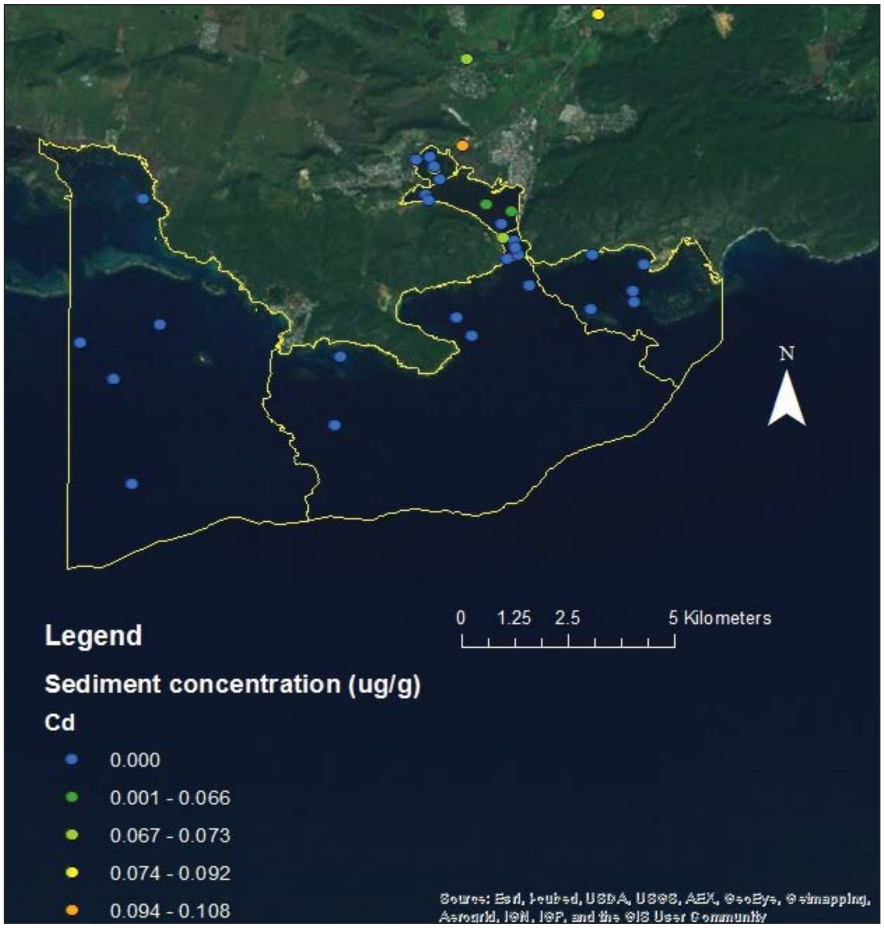

Figure 3.28. Concentrations of cadmium in sediments.

Cadmium in Coral Tissues

Cadmium concentrations in coral tissues ranged from below limits of detection to 0.33 µg/g (Figure 3.29), with a mean of 0.269 µg/g (Table 3.9), which is similar to what has been observed in other studies in the region (Table 3.6). There were no statistically significant differences between strata for cadmium in coral tissues (Dunn Method, α=0.05).

Iron in Sediments

The highest concentration of iron detected in this study is from the watershed site WS108 (62,800 µg/g, Figure 3.30), and the mean was 27,431 µg/g (Table 3.8) which is on the high end of what has been observed in other sites in the region (Table 3. 4), but not unusually high compared to historical NS&T national data. The three watershed sites represented three of the top five detections of iron for this study. Statistically, the North Bay had higher iron concentrations than any of the offshore strata (Dunn Method, α=0.05).

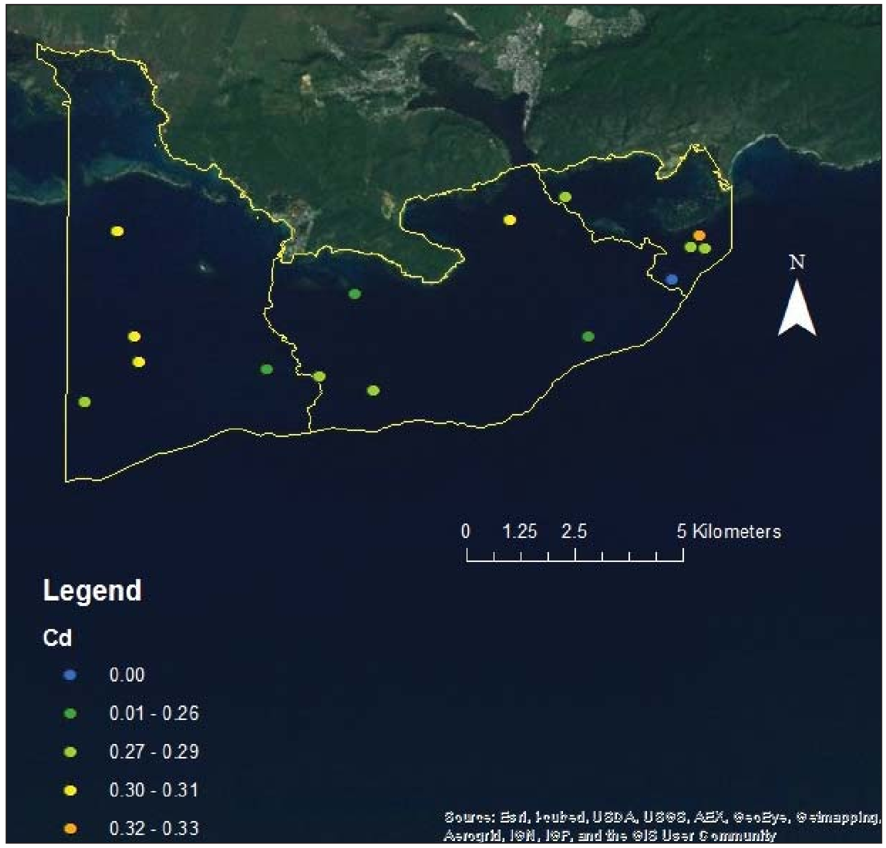

Figure 3.29. Concentrations of cadmium in coral (mustard hill coral).

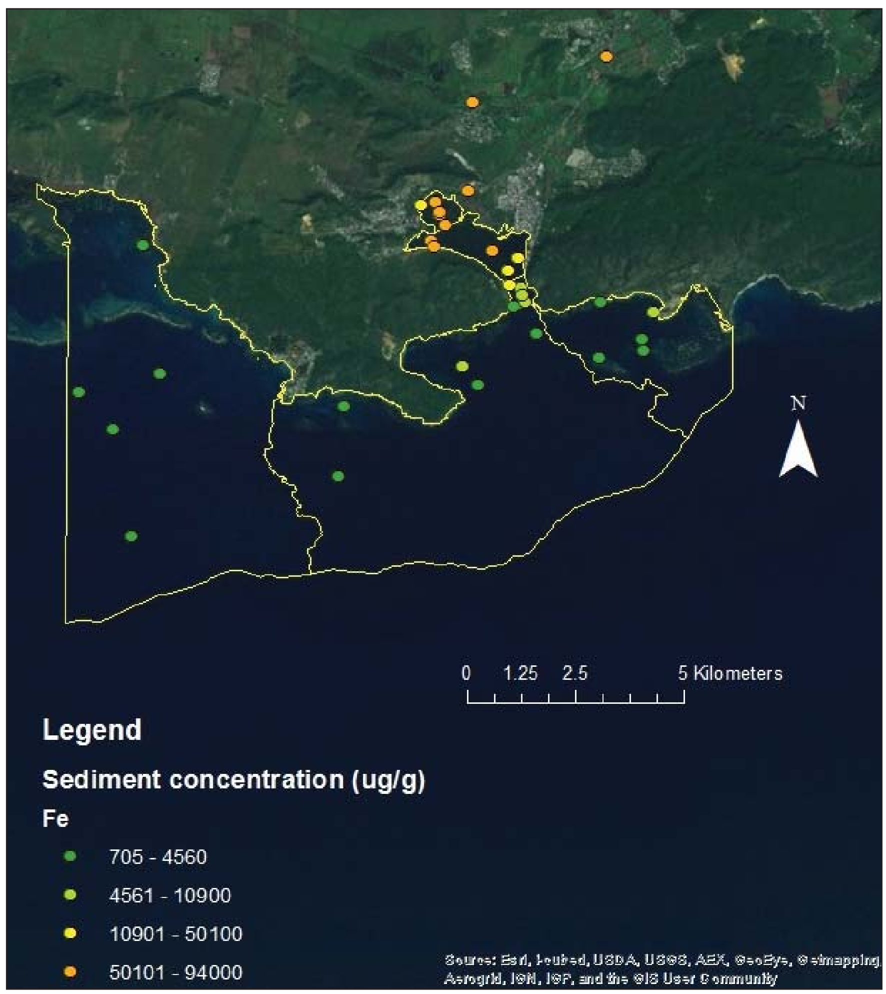

Figure 3.30. Concentrations of iron in sediments.

Iron in Coral Tissues
Iron was detected in coral tissues at values orders of magnitude lower than observed in the sediments (Figure 3.31), with a mean of 100.73 µg/g (Table 3.9) and at similar levels to what has been observed elsewhere in the region (Table 3.6). There are no statistically significant differences between strata for iron (Dunn Method, α=0.05).

Lead in Sediments
The highest concentration of lead detected in the sediments of Guánica Bay is 31.9 µg/g found at the Central Bay site CB012 (Figure 3.32). The mean concentration of lead was 6.08 µg/g (Table 3.8), which is within the range of values reported for the region (Table 3.4). The Central Bay accounted for the top three lead concentrations detected in this study. No sediment sites from Guánica Bay exceeded any thresholds or guidelines for lead. Statistically, the Central Bay had higher lead concentrations than any of the offshore strata, while the North Bay had higher concentrations than the eastern offshore stratum (Dunn Method, α=0.05).

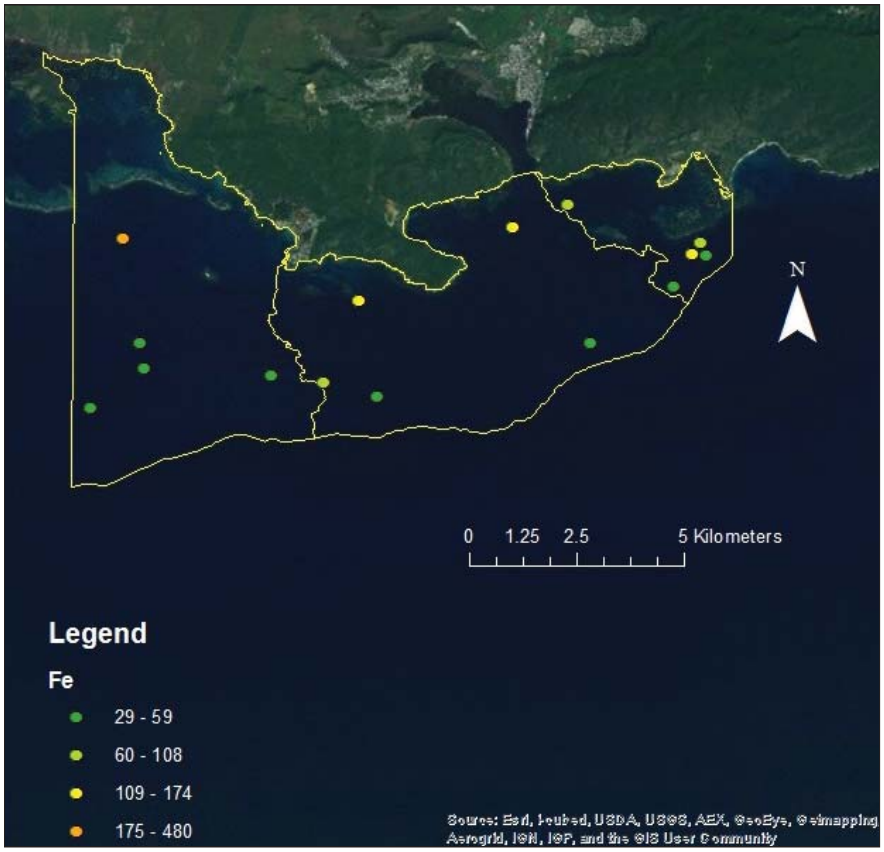

Figure 3.31. Concentrations of iron in coral (mustard hill coral).

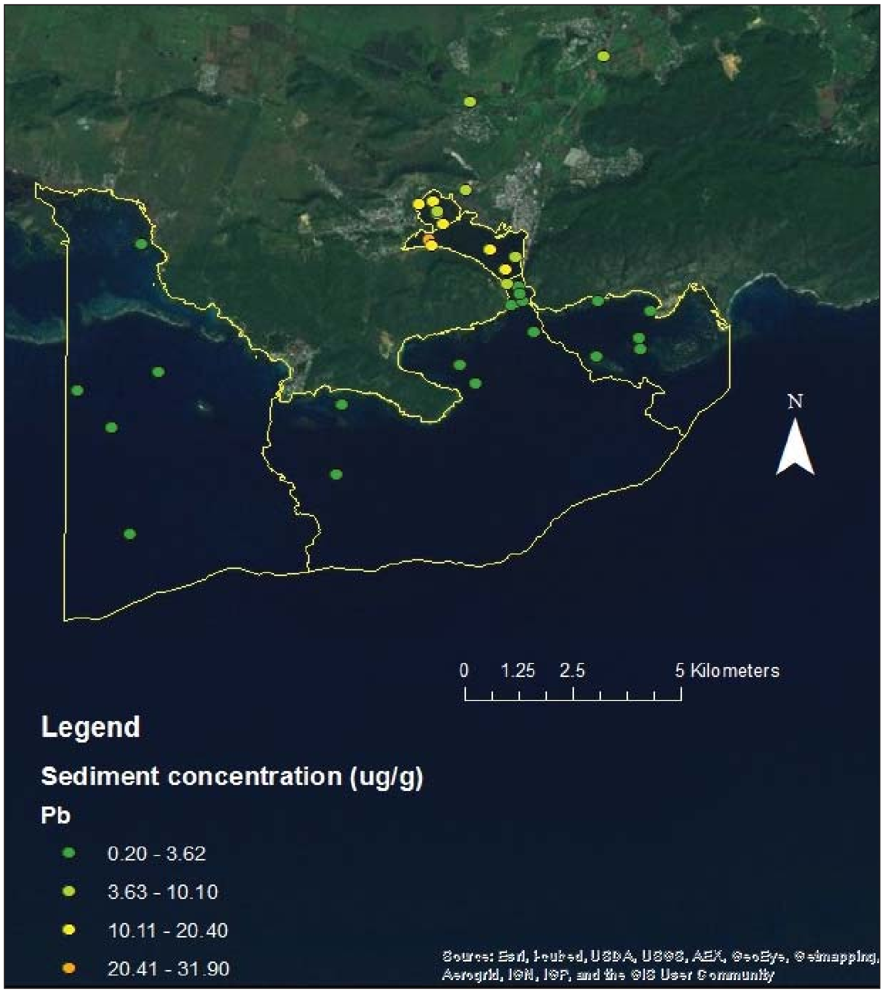

Figure 3.32. Concentrations of lead in sediments.

Lead in Coral Tissues
Lead was detected in coral tissues at values orders of magnitude lower than observed in the sediments (Figure 3.33, Table 3.9), and at similar levels to what has been observed elsewhere in the region (Table 3.6). There are no statistically significant differences between strata for lead (Dunn Method, α=0.05).

Manganese in Sediments
The highest concentration of manganese detected was 2,020 µg/g at the watershed site WS108 (Figure 3.34). The mean concentration of manganese in the sediments of Guánica Bay is 469.63 µg/g (Table 3.8) which is higher than observed at other sites in the region (Table 3.4) and slightly higher than the NS&T national mean (425.82 µg/g). The three watershed sites represented three of the top five detections of manganese for this study. Statistically, the Central Bay had higher manganese concentrations than any of the offshore strata (Dunn Method, α=0.05).

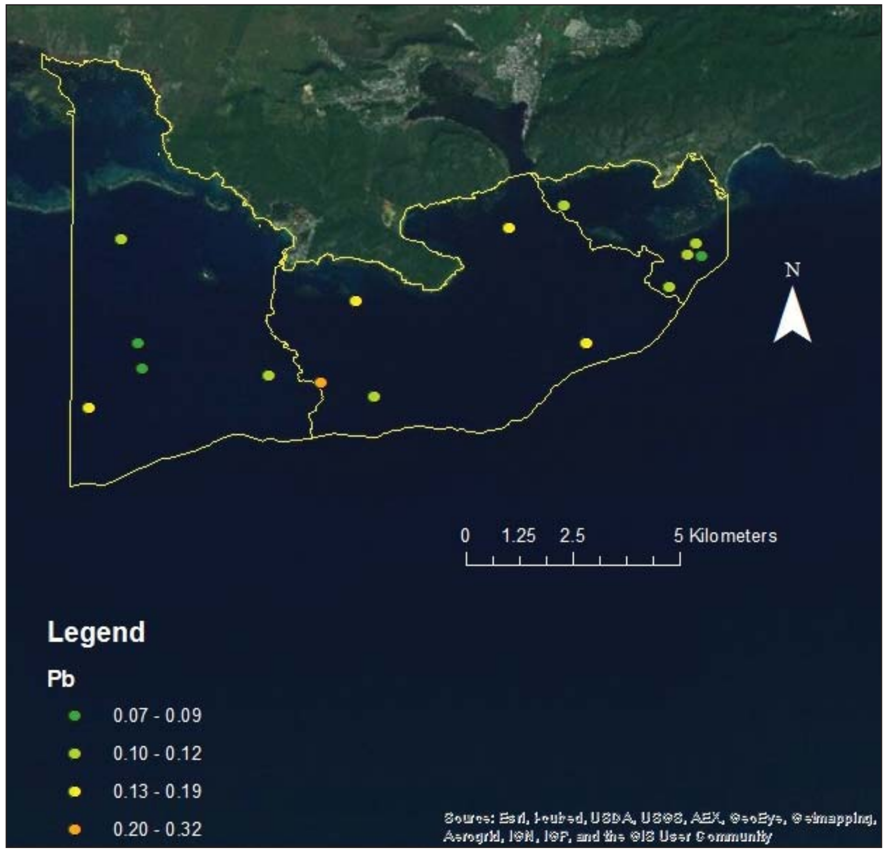

Figure 3.33. Concentrations of lead in coral (mustard hill coral).

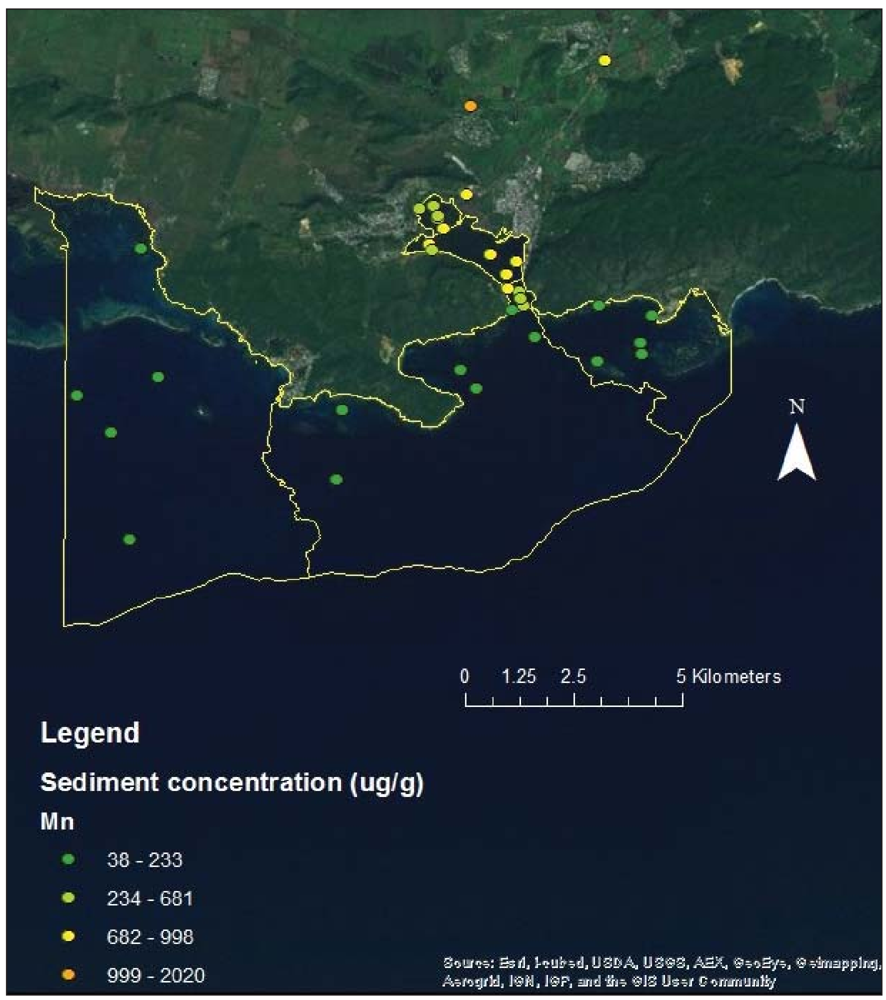

Figure 3.34. Concentrations of manganese in sediments.

Manganese in Coral Tissues
Manganese was detected in coral tissues at values orders of magnitude lower than observed in the sediments (Figure 3.35), with a mean of 10.48 µg/g (Table 3.9) and at similar levels to what has been observed elsewhere in the region (Table 3.6). There are no statistically significant differences between strata for manganese (Dunn Method, α=0.05).

Selenium in Sediments
The highest concentration of selenium detected in sediments from Guánica Bay is 0.512 µg/g at the Central Bay site CB012 (Figure 3.36). The mean concentration of selenium is 0.154 µg/g (Table 3.8), which is similar to what has been observed by other sites in the region (Table 3.4). Selenium concentrations are higher in the north stratum than the eastern offshore stratum (Dunn Method, α=0.05).

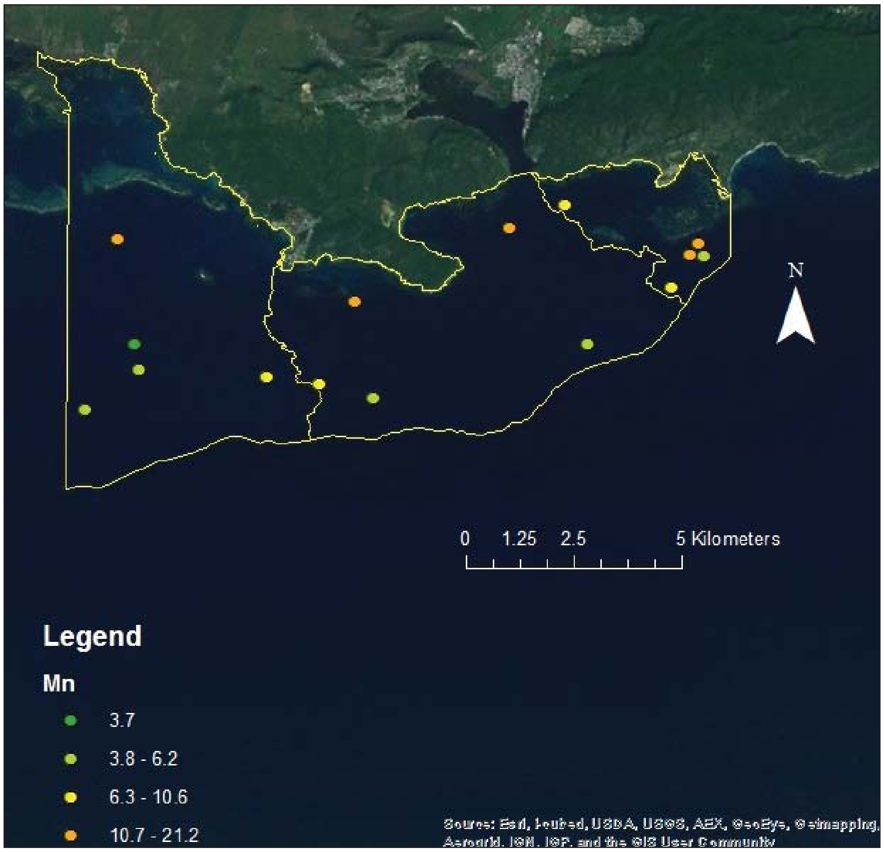

Figure 3.35. Concentrations of manganese in coral (mustard hill coral).

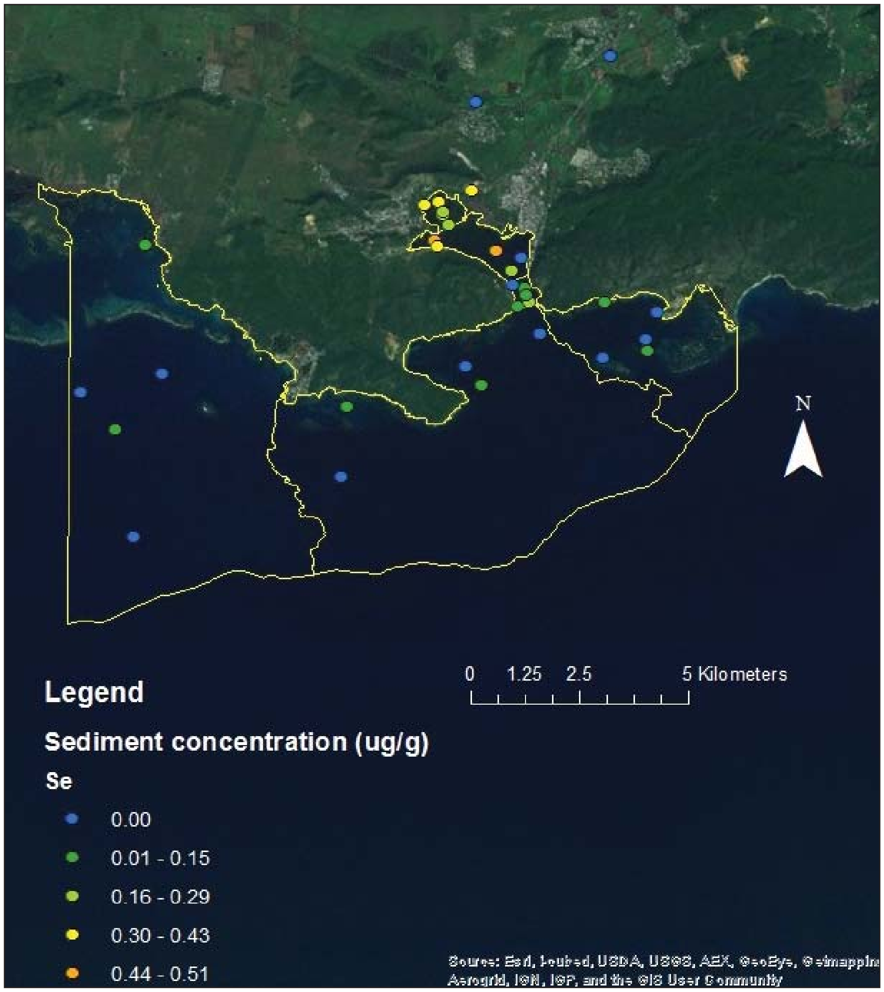

Figure 3.36. Concentrations of selenium in sediments.

Selenium in Coral Tissues
Selenium was detected in coral tissues at values orders of magnitude lower than observed in the sediments (Figure 3.37), with a mean of 0.19 µg/g (Table 3.9), which is similar to levels that have been observed elsewhere in the region (Table 3.6). There are no statistically significant differences between strata for selenium (Dunn Method, α=0.05).

Silicon in Sediments
The highest concentration of silicon detected is 220,000 µg/g at North Bay site NB05 (Figure 3.38). The mean concentration of silicon in the sediments of Guánica Bay is 84,726 µg/g (Table 3.8), which is within the range of values observed elsewhere in the region (Table 3.4). Statistically, the North Bay had higher silicon concentrations than any of the offshore strata and the Central Bay is higher than eastern offshore (Dunn Method, α=0.05).

Silicon in Coral Tissues
Silicon was not measured in coral tissues.

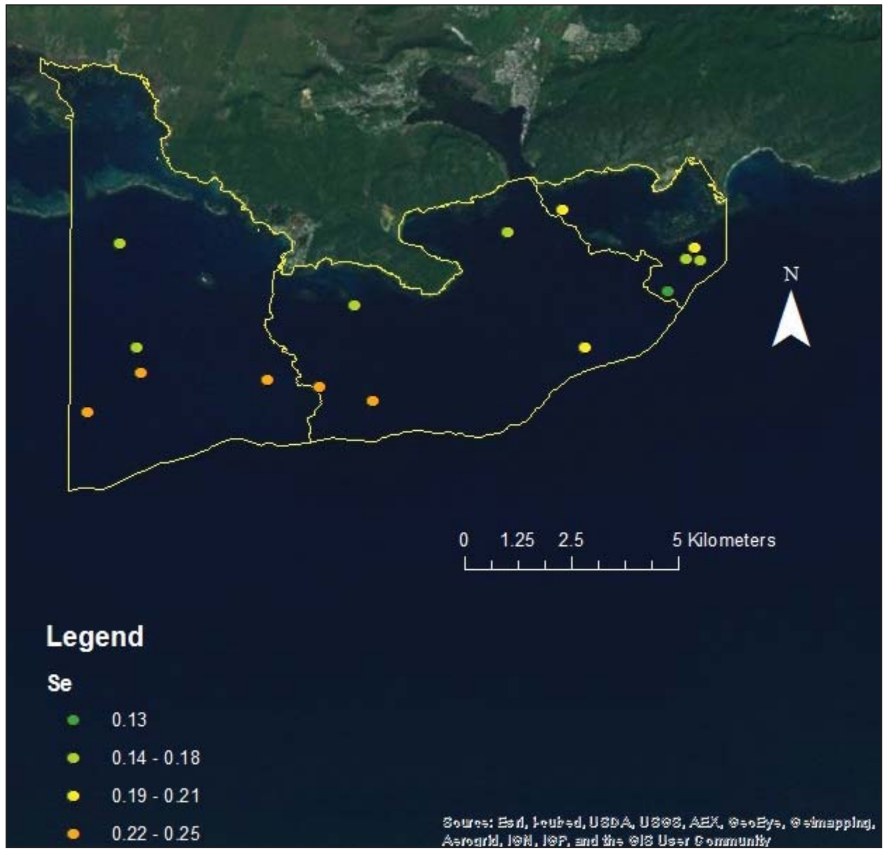

Figure 3.37. Concentrations of selenium in coral (mustard hill coral).

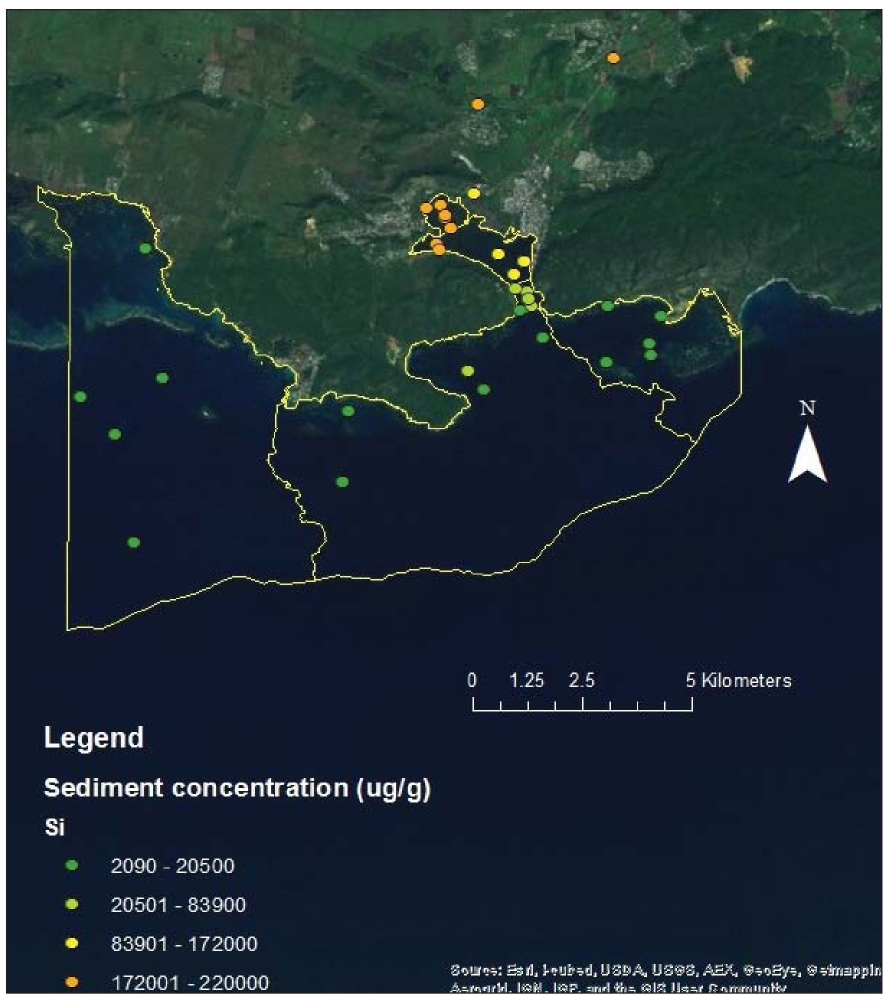

Figure 3.38. Concentrations of silicon in sediments.

Silver in Sediments

The highest concentration of silver detected in the sediments of Guánica Bay is 0.174 µg/g at North Bay site NB01 (Figure 3.39). The mean concentration of silver is 0.019 µg/g (Table 3.8), which is within the range of reported values for the region (Table 3.4). No sediment sites in Guánica Bay exceeded any thresholds or guidelines for silver. There are no significant differences between any strata for silver concentrations in this study (Dunn Method, α=0.05).

Silver in Coral Tissues

Silver was detected in coral tissues at values lower than observed in the sediments (Figure 3.40), with a mean of 0.004 µg/g (Table 3.9) which are similar levels to what has been observed elsewhere in the region (Table 3.6). There are no statistically significant differences between strata for silver (Dunn Method, α=0.05).

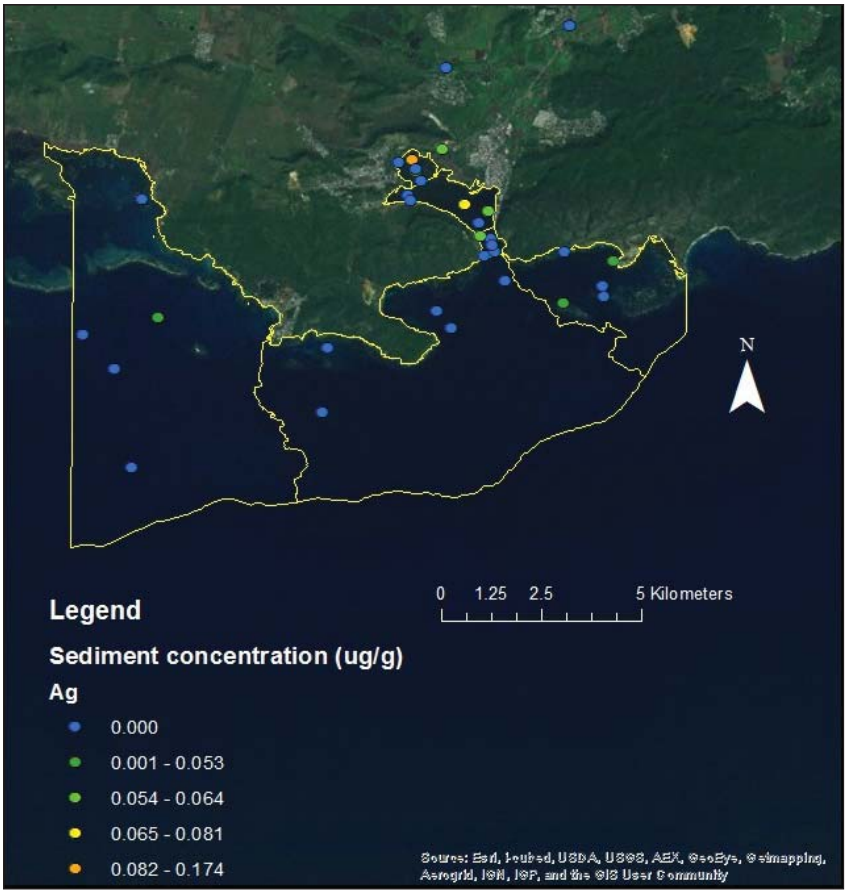

Figure 3.39. Concentrations of silver in sediments.

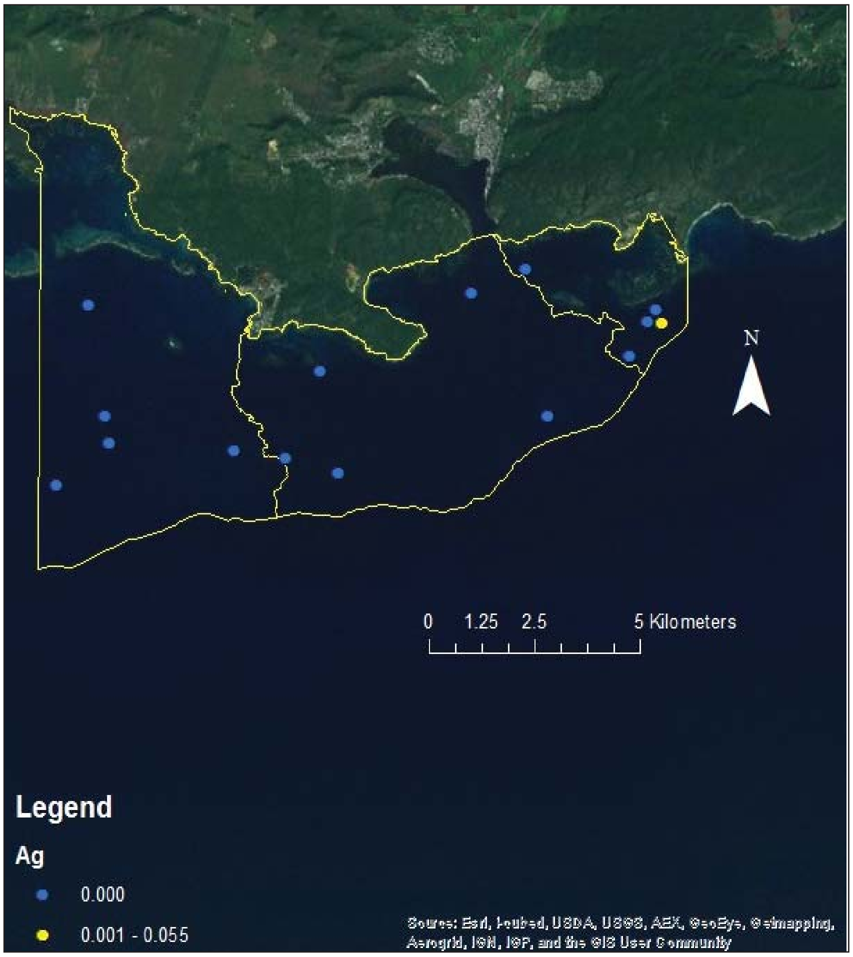

*Figure 3.40. Concentrations of silver in coral (*Porites astreoides*).*

Tin in Sediments

The highest concentration of tin detected in this study is 3.93 µg/g at the watershed site WS108 (Figure 3.41). The mean concentration of tin in Guánica Bay sediments is 0.830 µg/g (Table 3.8), which is similar to what has been observed elsewhere in the region (Table 3.4). Tin concentrations are higher in the North Bay and Central Bay strata when compared with the central and western offshore strata (Dunn Method, α=0.05).

Tin in Coral Tissues

Concentrations of tin in coral tissues were lower than concentrations observed in sediments, with a mean of 0.07 µg/g (Table 3.9), which is similar to what has been reported for other systems in the region (Table 3.6). Tin was detected in coral tissues only in the central stratum (i.e., immediately offshore from the Bay; Figure 3.42). Interestingly, tin was the only contaminant for which there is a statistically significant difference between strata for corals (Dunn Method, α=0.05), but it is likely because of the prevalence of non-detects in the eastern and western strata rather than unusually high tin in the central stratum.

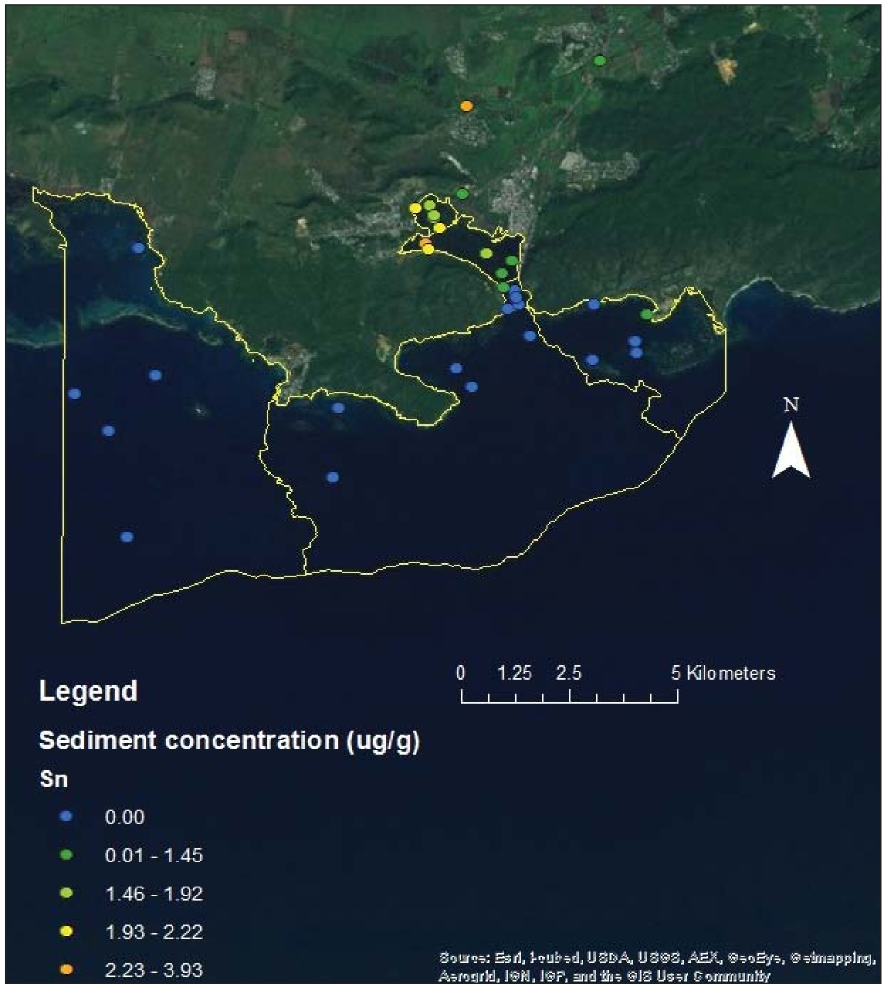

Figure 3.41. Concentrations of tin in sediments.

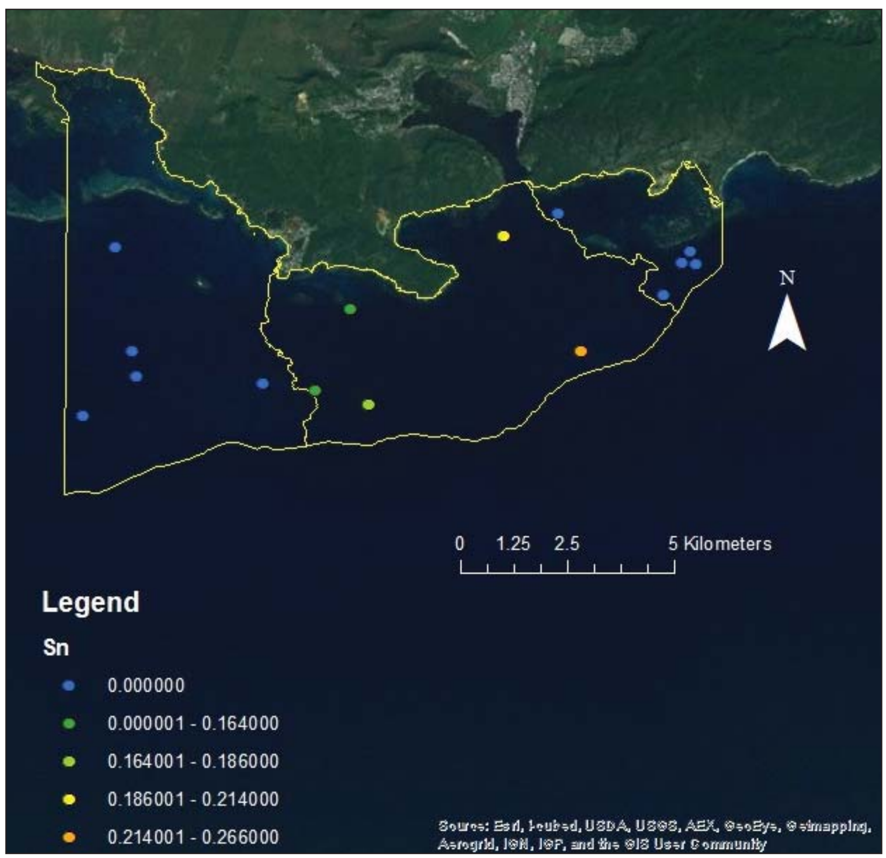

Figure 3.42. Concentrations of tin in coral (mustard hill coral).

Sewage Indicators in Sediments

C. perfrigens was detected in 23 of 33 sediment samples collected, including every site in the bay and watershed, with a maximum of 3311 CFU/g (colony forming units per gram) at a site in the North Bay (Figure 3.43) and a mean of 454 CFU/g. All sites with detections were relatively close to shore. Statistically, *C. perfrigens* was higher in the North Bay and Central Bay than eastern offshore and western offshore strata (Dunn Method, α=0.05). This is a similar range of values that has been reported elsewhere in Puerto Rico (Pait et al., 2007; Pait et al., 2010). Sources of the elevated levels of *C. perfringens* would include wastewater treatment plants (WWTP) or septic systems that discharge to Guánica Bay or the Rio Loco, along with possible animal sources from agriculture in the Lajas Valley.

Although *C. perfringens* is a common cause of foodborne illnesses, no health guidelines exist for concentrations of *C. perfringens* in sediments. A more severe form of the disease is often fatal and results from ingesting large numbers of the active bacteria. This pathogen also has the capability of forming spores which can persist in soils and sediments in areas subject to human and animal fecal pollution.

Figure 3.43. Concentrations of C. perfringens *in sediments.*

Relationship of Contaminants with Grain Size, Total Organic Carbon and Lipid Content

None of the sediment contaminants measured in this study were well correlated (Spearman correlation) with percent total organic carbon (TOC). This is somewhat surprising given the tendency for contaminants, especially organics, to bind to sediments with higher organic content. Conversely, all sediment contaminants, with the exception of cadmium, HCH, monobutyltin and dibutlytin, are significantly correlated with grain size (percent fine, defined as sum of silt and clay fractions). However, correlation coefficients (Spearman ρ) are only above 0.7 for mercury, lead, PAHs, chlordane, PCBs and PAHs, indicating that these analytes are well correlated with grain size. When this subset of analytes is normalized to grain size, the statistical analysis of differences between strata changes; specifically, there are far fewer differences between the strata (mercury, lead and chlordane have no statistically significant differences between strata). While this highlights the importance of grain size in understanding spatial patterns, it is important to consider that from an ecological response perspective, higher concentrations, whether driven by grain size or some other pattern, can be detrimental to organisms in the ecosystem. Coral tissue contaminants were not significantly correlated with tissue lipid content (Spearman, α=0.05).

Surface Water Temperature, Salinity and Dissolved Oxygen

Surface water YSI data (temperature, salinity and dissolved oxygen) for the coral sampling sites are shown in Table 3.11. There is very little variability between sites for these three parameters indicating fairly uniform offshore surface water characteristics. The only difference between strata is for temperature, for which the western strata is significant cooler than the central strata (Dunn Method, α=0.05). The measurements in the western strata were made in the early morning, whereas the measurements in the central strata were made mid-day. This suggests that radiant heating of the surface water explains this observed difference.

Interpretation of Watershed Sites

Caution must be used when interpreting observed concentrations of contaminants in the three watershed sites from 2009 (Figure 3.1) and the supplemental watershed sites from 2013 (see Appendix B, Figure B.12). First, these sites were not randomly selected and therefore cannot be included in the statistical analysis by strata of the sites selected via stratified random design. Second, these sites represent a very different hydrologic environment than the sites in the bay and offshore. It is not clear to what extent deposited sediments are retained at these sites; based on field observations, it seems likely that some of the stream sites are probably scoured during large storm events. The site closest to the mouth of the Rio Loco (i.e., the farthest downstream watershed site) is more likely to be a depositional area which may retain significant sediments and associated contaminants. Additionally, it is tidally influenced, and its contaminant concentration may reflect both watershed inputs, as well as reverse flow from the bay. Presence of contaminants in the stream sites may shed some light on where pollutants are originating from, but these sites, in the absence of other information about pollutant sources, cannot provide definitive information. Results for selected analytes from the supplemental sampling are shown in Appendix B (Figures B.13-B.18). Chlordane is widely distributed in the watershed (Appendix B, Figure B.13). This is likely due to its wide use as an insecticide, including for termites, combined with its environmental persistence. PCBs do not appear to be coming from the watershed (Appendix B, Figure B.14). It is not clear if there is a near bay source of PCBs that is leaching into the bay via the stormwater outflow pipes or via groundwater, or if these stream values represent contamination present in the bay being transported to these sites via tidal actions. Similarly, DDT does not appear to have a strong watershed stream source (Appendix B, Figure B.15), which is surprising given its historical widespread use in agriculture and for mosquito control. Nickel (Appendix B, Figure B.16) and chromium (Appendix B, Figure B.17) are quite elevated across the watershed which suggests some sort of diffuse, or possibly natural source of these metals. Geological characterizations of southwest Puerto Rico have identified stream sediment anomalies for both nickel and chromium (USGS, 1998). This is attributed to these metals natural presence in the characteristic geology of the area (podiform chromite terrane). It is possible that anthropogenically enhanced erosion is exacerbating naturally high metals levels, resulting in the usually high marine sediment metals concentrations for nickel and chromium. Zinc is high near the outflow pipes near the bay (Appendix B, Figure B.18) which may be indicative of an urban automobile source (i.e., car tire dust; Councell et al., 2004).

Table 3.11. YSI data from coral sampling sites (depth=0.5 m).

Site	Strata	DO	Temp	Salinity	Conductivity
CC056	Central	5.92	29.1	34.4	56.6
CC057	Central	6.13	28.7	34.2	52.1
CC058	Central	5.79	28.9	34.4	52.3
CC060	Central	6.13	29	34.2	55.9
CC061a	Central	6.04	29	34.1	55.8
EC031	Eastern	5.89	28.7	34.5	55.2
EC032	Eastern	6.11	28.5	34.3	55.7
EC034	Eastern	5.89	28.5	34.5	55.8
EC041a	Eastern	5.91	28.4	34.3	55.7
EC045a	Eastern	5.51	28.6	34.5	56.1
WC081	Western	6.1	28.8	34.2	52.2
WC082	Western	7.04	28.1	34.4	52.3
WC083	Western	6.04	28.1	34.4	55.5
WC084	Western	5.91	28.2	34.5	55.8
WC085	Western	5.85	27.8	34.5	52.7

Potential Point Sources of Pollutants in the Watershed

Pollutant specific non-point sources are discussed above. A review of U.S. EPA National Pollution Discharge Elimination System permits for the subwatersheds of the study region does not point to any obvious dischargers that could explain the elevated levels of pollution in this study area (USEPA, 2013). All of the permitted dischargers within the watershed are drinking water and wastewater treatment plant facilities, none of which have discharged large amounts of metals or organic pollutants. There is an active landfill in the eastern portion of the study area (Figure 3.1) which could be a source of a variety of pollutants. Historically, there was a chemical plant (Gonzalez Chemical Industries), which manufactured ammonia fertilizer and sulfuric acid (Chem. Eng. News, 1957). While it is possible that this production facility contributed to the pollution legacy of the bay, there is no obvious, documented link between these products and the type of pollution observed in this system.

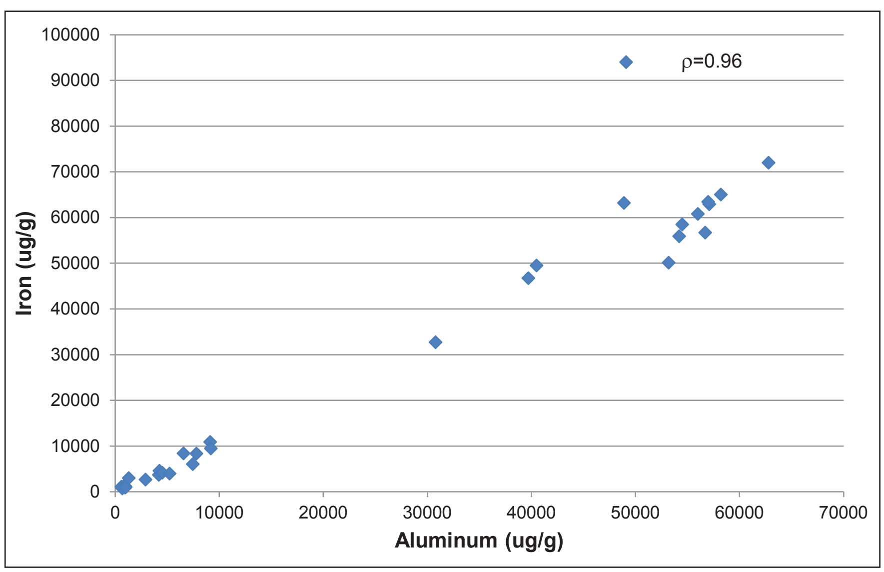

Figure 3.44. Correlation of sediment aluminum concentrations with sediment iron concentrations. ρ value is Spearman rho (α=0.05).

Additionally, because the bay is oriented on a north-south axis, latitude was used as a proxy for relative strength of watershed source/dilution within the bay. In other words, within the watershed and bay, if the watershed was a strong source of an individual pollutant it would be expected that the more northerly sites would be most impacted by watershed and the more southerly sites less so (due to dilution). Latitude was significantly but weakly correlated (Spearman, ρ=0.48) with the sewage indicator *C. perfringens*, suggesting a watershed-based source of sewage, rather than discharges from boats. Latitude was also significantly correlated with concentration (Spearman, α=0.05) for all metals except silver, arsenic, lead, selenium and mercury, suggesting a strong watershed source of these metals. The strongest relationships were for iron (ρ=0.87), silicon (ρ=0.80), nickel (ρ=0.85), chromium (ρ=0.84) and zinc (ρ=0.78). However, because metals are naturally occurring in the earth's crust, it is unclear if these are natural or anthropogenic sources. Another way to potentially examine the nature of the source of these metals is to look at the ratio of each metal to aluminum, the primary element in the earth's crust and generally not considered to be a pollutant. If a metal is well correlated with aluminum, it is more likely to have a natural (erosion) source (see also Apeti et al., 2012). All metals with the exception of silver were significantly correlated with aluminum (Spearman, α=0.05). Elements which make up large portions of the earth's crust (e.g. silicon and iron) are especially well correlated with aluminum (e.g., Figure 3.44). Other metals, such as nickel and chromium, were also well correlated with aluminum, although sites with disproportionately high concentrations (relative to aluminum were observed (Figures 3.45 and 3.46). These "outliers" may represent anthropogenic sources of these metals are prevalent at those sites.

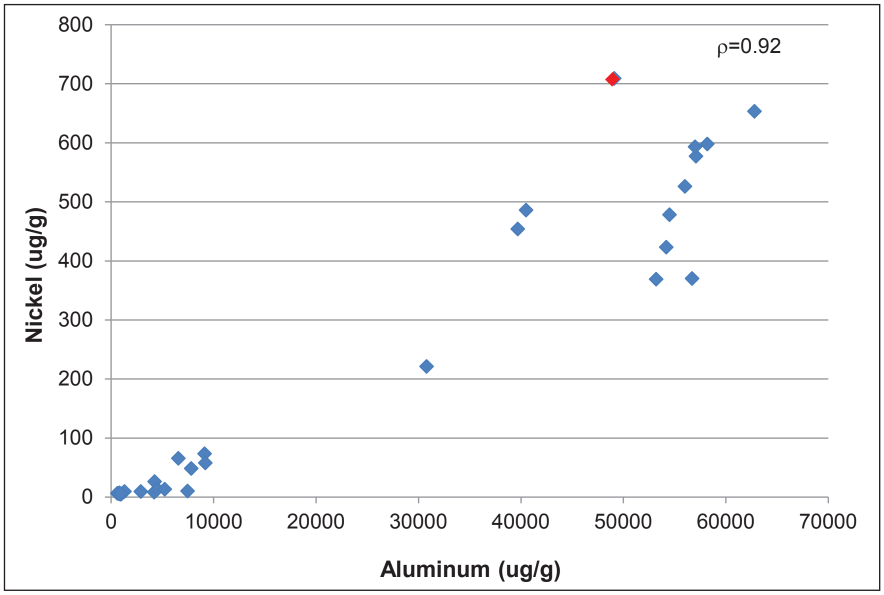

Figure 3.45. Correlation of sediment aluminum concentrations with sediment nickel concentrations. Red diamonds indicate higher than expected (relative to aluminum) concentrations of nickel. ρ value is Spearman rho (α=0.05).

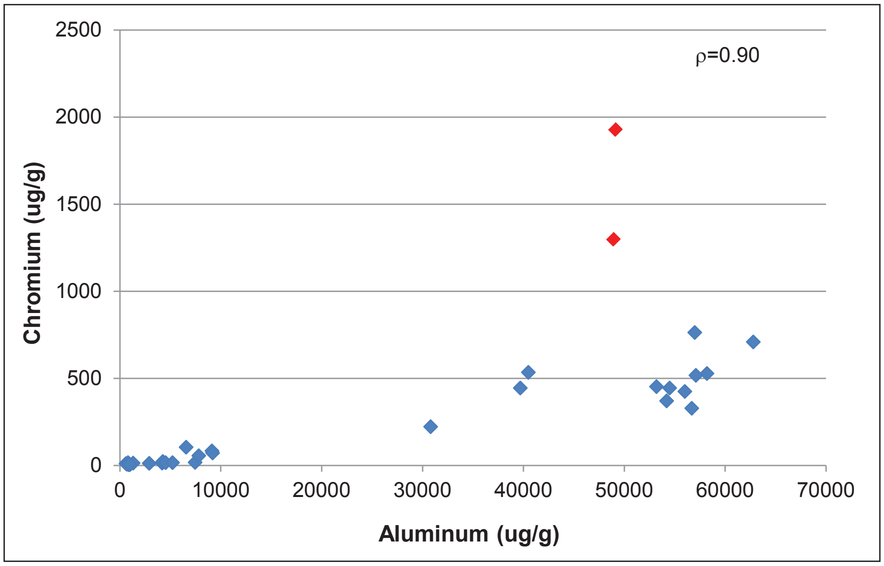

Figure 3.46. Correlation of sediment aluminum concentrations with sediment chromium concentrations. Red diamonds indicate higher than expected (relative to aluminum) concentrations of chromium. ρ value is Spearman rho (α=0.05).

Fate of Contaminants from Guánica Bay

In general, pollutant levels were highest inside the bay itself and lower offshore. In order to test if the easterly flow from Guánica Bay is having a pollutant impact to the west, multivariate nonparametric tests were conducted correlating contaminant concentration in sediments (offshore strata only) and corals to longitude (i.e., long shore position). In general, contaminant distributions were not well correlated with longitude (Spearman α=0.05). A negative correlation was found between DDT in coral tissue concentration and longitude, suggesting that DDT concentrations increase from east to west (Spearman Rho = -0.6096). However, this pattern was not observed in sediment data. It is possible that offshore currents are sufficient to "flush" the offshore sediment system frequently; whereas pollutants can accumulate in the bay because it is not as exposed to ocean currents.

Ecological Significance of Findings

Since trace elements occur naturally in the environment, including some which are micronutrients for many organisms, the mere presence of trace elements in corals tissues may or may not represent a stressor to the coral. However, organic contaminants serve no positive purpose in coral physiology. At best, the organic pollutants might not harm the corals in small concentrations, but at worst these may represent serious stress to coral health and physiology.

While some studies exist linking specific contaminants to deleterious effects in corals (e.g., Solbakken et al., 1984; Peachey and Crosby, 1996; Reichelt-Brushett and Harrison, 2005; Guzman-Martinez et al., 2007), further research is needed to link observed contaminants in coral tissues with sub-lethal responses in the coral. This might be accomplished by ecotoxicological studies in coral mesocosm experimental facilities, or in the field with genetic analysis of stressor genes in combination with field measurements of contamination (e.g., Edge et al., 2005).

In the absence of experimental ecotoxicological studies on the causal relationships between contaminants and coral reef communities, a number of multivariate statistical methods could be used to examine the correlational relationships between contaminants and the biological communities (e.g., fish and corals) they may impact. Among these methods are regression-based techniques to model the response of biological variables (e.g., species presence/absence, abundance, biomass) to contaminants as well as classification and ordination methods (see Legendre and Legendre, 1998 for detailed descriptions of many of these methods).

These multivariate statistical methods generally require spatially and temporally co-located environmental and species sample data. Although co-located data (or even approximately co-located within the range of spatial autocorrelation) was not available from the contaminant and biological datasets described in this chapter and in Chapter 2, respectively, as an exploratory analysis a simple correlation analysis was performed using contaminant sample data and modeled predictions of biological data at the locations of the contaminant samples. Although not statistically robust, this was intended as a demonstration of the type of analysis that could be performed with co-located data. Geostatistical methods were used to generate continuous gridded model predictions at 30m horizontal grid resolution for fish biomass, fish abundance, fish species richness, fish diversity, total coral and coral species richness from the point sample data in the dataset presented in Chapter 2. Geostatistical methods are based on the premise that neighboring samples are more similar than those farther apart and that the spatial structure of the data (spatial autocorrelation) can be estimated, modeled and used to make predictions at locations that have not been sampled (Cressie, 1993). For each of the biological variables, spatial autocorrelation in the point sample data was quantified and modeled using semivariogram analysis. Fitted semivariogram model parameters were used to perform ordinary kriging of the sample data. Values for each of the biological variables were then extracted from the model predictions at the locations of contaminant sample data, and Pearson's product-moment correlation coefficients were calculated for each biological variable-contaminant pair and tested for statistical significance ($\alpha=0.05$). As an example of the potential utility of this type of analysis, the tests showed a relatively strong ($r > 0.6$) negative relationship between total PCBs and all the biological variables except fish biomass and diversity.

It is important to note, however, that a number of statistical assumptions are violated in using values extracted from the modeled biological datasets to perform correlation analysis in this manner and that the datasets generated by geostatistical methods represent statistical model predictions, and as such may vary greatly in their accuracy and may or may not reflect actual conditions at any given location. At the same time, this exploratory analysis suggested some potential relationships between contaminants and coral reef communities that could be further explored through the collection of additional and spatially and temporally co-located data. In addition, semivariogram analysis of the sample datasets can inform future survey design by indicating the range of spatial autocorrelation in the variables of interest.

CONCLUSIONS

Overall, the chemical contamination of Guánica Bay is higher than what has been observed in other systems in Puerto Rico. For some analytes, the bay is polluted relative to national historic sediment values. High levels of pollution are generally limited to inside the bay itself. While there is evidence of transport of pollutants to the offshore coral reef ecosystem, contaminant concentrations in coral tissues in this study were consistent with concentrations seen in other regions of Puerto Rico.

There are a variety of potential sources of pollution to Guánica Bay (e.g., WWTP, landfill, legacy industries, agriculture, boating, impervious surface runoff), but there is no one "smoking gun" which identifies any one source of primary concern. This speaks to the need for an integrated management strategy which addresses multiple sources of land-based pollution.

REFERENCES

Alegria, H.A., T.F. Bidleman, and T.J. Shaw. 2000. Organochlorine pesticides in ambient air of Belize, Central America. Environ. Sci. Technol. 34:1953–1958.

Apeti, D., D. Whitall, A. Pait, Z. Dieppa, A. Zitello, and G. Lauenstein. 2012. Characterization of land-based sources of pollution in Jobos Bay, Puerto Rico: status of heavy metal concentration in bed sediment. Environ Monit Assess 184:811–830

Batley, G. 1996. Distribution and fate of tributyltin. In: S. J. de Mora (ed.) Tributyltin: A Case Study of an Environmental Contaminant. Cambridge University Press. Cambridge, England. 301 pp.

Bauer, L.J. and M.S. Kendall (eds.). 2010. An Ecological Characterization of the Marine Resources of Vieques, Puerto Rico Part II: Focused Field Studies of Habitats, Nutrients, Contaminants, Fish, and Benthic Communities. NOAA Technical Memorandum NOS NCCOS Silver Spring, MD. 240 pp.

Bennett, R.F. 1996. Industrial manufacture and applications of tributyltin compounds. In: S. J. de Mora (Ed.), Tributyltin: A Case Study of an Environmental Contaminant. Cambridge University Press. Cambridge, England. 301 pp.

Birchenough, A.C., N. Barnes, S.M. Evans, H. Hinz, I. Kronke and C. Moss. 2002. A review and assessment of tributyltin contamination in the North Sea, based on surveys of butyltin tissue burdens and imposex/intersex in four species of neogastropods. Marine Pollution Bulletin 44: 534-543.

Bryant, D., L. Burke, J. McManus, and M. Spalding. 1998. Reefs at Risk: A Map-based Indicator of Threats to the World's Coral Reefs. World Resources Institute, Washington, DC.

Budzinski, H., I. Jones, J. Bellocq, C. Pierard, and P. Garrigues. 1997. Evaluation of sediment contamination by polycyclic aromatic hydrocarbons in the Gironde estuary. Mar. Chem. 58: 85-97.

Cantillo, A.Y., G.G. Lauenstein, W.E. Johnson, and T.P. O'Connor. 1999. Status and Trends of contaminant levels in biota and sediments of Tampa Bay. NOAA Regional Report Series 5. 43 pp.

Chem. Eng. News. 1957. "Petrochemicals 'Arrive' in P. R." Chem. Eng. News. Volume 35 (15). 20 pp.

Code of Federal Regulations (CFR). 1998. PCB allowable uses provisions. 40 CFR 761.80, July 1998. Available at: http://www.access.gpo.gov/nara/cfr/cfr-Table 3.-search.html.

Coast Guard News. 2007. Coast Guard Update on the Oil Spill on Puerto Rican Coast. http://coastguardnews.com/coast-guard-update-on-the-oil-spill-on-puerto-rican-coast/2007/09/12/

Councell, T.B., K.U. Duckenfield, E.R. Landa, and E. Callender. 2004. Tire-wear particles as a source of zinc to the environment. Environ. Sci. Technol. 38: 4206-4214.

Cressie, N. 1993. Statistics for spatial data. Wiley, New York.

DNER (Puerto Rico Department of Natural and Environmental Resources). 2002. Field, R. (Editor) , E.N. Laboy, J. Capella, and. P.O. Robles, and C.M. Gonzalez. Jobos Bay Estuarine Profile: A National Estuarine Research Reserve. 107 pp.

Edge, S.E, M.B. Morgan, D.R. Gleason, and T.W Snell. 2005. Development of a coral cDNA array to examine gene expression profiles in Montastraea faveolata exposed to environmental stress. Marine Pollution Bulletin 51: 507–523

Eisler, R. 1985. Cadmium hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish and Wildlife Service Biological Report 85(1.2). 30 pp.

Eisler, R. 1986. Chromium hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish and Wildlife Service Biological Report 85(1.6). 60 pp.

Eisler, R. 1987. Mercury hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish and Wildlife Service Biological Report 85(1.10). 63 pp.

Eisler, R. Copper hazards to fish, wildlife, and invertebrates: a synoptic review (Contaminant Hazard Reviews Report No. 33). US Department of the Interior and US Geological Survey. 1998.

EPA (U.S. Environmental Protection Agency). 1997. Management of polychlorinated biphenyls in the United States. Environmental Protection Agency. Available at: http://www.chem.unep.ch/pops. Office of Pollution Prevention and Toxics, U.S. Environmental Protection Agency. 6 pp.

EPA (U.S. Environmental Protection Agency). 2009. DDT regulatory history: a brief survey (to 1975). Online: http://www.epa.gov/history/topics/ddt/02.htm.

EPA (U.S. Environmental Protection Agency). 2013. Permit Compliance System (PCS) and Integrated Compliance Information System

(ICIS) database. http://www.epa.gov/enviro/facts/pcs-icis/search.html
Gibbs, P.E. and G.W. Bryan. 1996. TBT-induced imposex in neogastropod snails: masculinization to mass extinction. In: S. J. de Mora (ed.), Tributyltin: A Case Study of an Environmental Contaminant. Cambridge University Press. Cambridge, England. 301 pp.

Guzman-Martinez, M.D.C., P. Ramirez-Romero and A.T. Banaszak. 2007. Photoinduced toxicity of the polycyclic aromatic hydrocarbon, fluoranthene, on the coral, Porites divaricata. Journal of Environmental Science and Health, Part A 42(10): 1495-1502.

Hassett, J.J., J.C. Means, W.L. Banwart and S.G. Wood. 1980. Sorption related properties of sediments and energy related pollutants. U.S. Environmental Protection Agency. EPA-600/3-8-041. 133 pp.

Humann, P. and N. DeLoach. 2002. Reef Coral Identifi cation: Florida, Caribbean Bahamas. Second Edition. New World Publications. Jacksonville, Florida. 278pp.

Hylland, K. 2006. Polycyclic aromatic hydrocarbons (PAH) ecotoxicology in marine ecosystems. Journal of Toxicology and Environmental Health, Part A 69: 109-123.

Kennedy, C.J., N.J. Gassman and P.J. Walsh. 1992. The fate of benzo[a]pyrene in the scleractinian corals Favia fragum and Montastrea annularis. Marine Biology 113: 3131-318.

Kimbrough, K.L. and G.G. Lauenstein (eds). 2006. Major and trace element analytical methods of the National Status and Trends Program: 2000-2006. NOAA Technical Memorandum NOS NCCOS 29. Silver Spring, MD. 19 pp.

Kimbrough, K.L., G.G. Lauenstein and W.E. Johnson (eds). 2006. Organic contaminant analytical methods of the National Status and Trends Program: Update 2000-2006. NOAA Technical Memorandum NOS NCCOS 30. Silver Spring, MD. 137 pp.

Larsen, M.C. and A. Santiago-Román. 2001. Geomorphic Processes and Riverine Habitat Mass wasting and sediment storage in a small montane watershed: an extreme case of anthropogenic disturbance in the humid tropics. American Geophysical Union Monograph Series.

Lauenstein, G.G., and A.Y. Cantillo. 1998. Sampling and Analytical Methods of the National Status and Trends Program Mussel Watch Project: 1993-1996 Update. NOAA Tech. Memo. 130, NOS ORCA. Silver Spring, Maryland. 233 pp.

Legendre, P. and L. Legendre. 1998: Numerical ecology. 2nd English edition. Elsevier, Amsterdam

Long, E.R. and L.G. Morgan. 1990. The Potential for Biological Effects of Sediment-Sorbed Contaminants Tested in the National Status and Trends Program. NOAA Tech. Memo NOS OMA 52. NOAA. Seattle, WA. 175 pp.

Long, E.R., D.D. MacDonald, S.L. Smith, and F.D. Calder. 1995. Incidence of adverse biologival effects within ranges of chemical concentrations in marine and estuarine sediments. Environmental Management 19: 81-97.

McDonald, S.J., D.S. Frank, J.A. Ramirez, B. Wang and J.M. Brooks. 2006. Ancillary methods of the National Status and Trends Program: 2000-2006 Update. NOAA Technical Memorandums NOS NCCOS 28. Silver Spring, MD. 17 pp.

Neff, J.M. 1985. Polycyclic aromatic hydrocarbons. In: G.M. Rand and S.R. Petrocelli (eds.), Fundamentals of Aquatic Toxicology. Hemisphere Publishing Corporation. New York. 666 pp.

Neff, J.M., S.A. Stout, and D.G. Gunster. 2005. Ecological risk assessment of polycyclic aromatic hydrocarbons in sediments: identifying sources and ecological hazard. Integrated Environmental Assessment and Management 1: 22-33.

Negri, A.P., L.D. Smith, N.S. Webster and A.J. Heyward. 2002. Understanding ship-grounding impacts on a coral reef: potential effects of anti-foulant paint contamination on coral recruitment. Marine Pollution Bulletin 44: 111-117.

Nipper, M., J.W. Tunnell, C.R. Scott, V.H. Garrison, D. Griffin, and G. Smith. 2006. Effects of Globally Transported African Dust to Caribbean Marine Ecosystems. USGS. Reston, VA. 2006.

Pait, A.S., D.R. Whitall, C.F.G. Jeffrey, C. Caldow, A.L. Mason, J.D. Christensen, M.E. Monaco, and J. Ramirez. 2007. An assessment of chemical contaminants in the marine sediments of southwest Puerto Rico. NOS NCCOS 52. Silver Spring, MD. 116pp.

Pait, A.S., D.R. Whitall, C.F.G. Jeffrey, C. Caldow, A.L. Mason, G.G.Lauenstein, and. J.D. Christensen. 2008. Chemical contamination in southwest Puerto Rico: an assessment of organic contaminants in nearshore sediments. Marine Pollution Bulletin 56: 580-606.

Pait, A.S., C.F.G. Jeffrey, C. Caldow, D.R. Whitall, S.I. Hartwell, A.L. Mason, and J.D. Christensen. 2009. Chemical Contaminants in the Coral Porites astreoides from Southwest Puerto Rico. NOAA Technical Memorandum NOS NCCOS 91. Silver Spring, MD. 40 pp.

Pait, A.S., A.L. Mason, D.R. Whitall, J.D. Christensen, and S.I. Hartwell. 2010. Assessment of chemical contaminants in sediments and corals in Vieques In: Bauer and Kendall (eds.), An Ecological Characterization of the Marine Resources of Vieques, Puerto Rico Part II: Field Studies of Habitats, Nutrients, Contaminants, Fish, and Benthic Communities. NOAA Technical Memorandum NOS NCCOS 110. Silver Spring, MD. 174 pp.

Peachey, R.L. and D.G. Crosby. 1996. Phototoxicity in tropical reef animals. Mar. Environ. Res. 42: 359-362.
Pait, A.S., A.L. Mason, D.R. Whitall, J.D. Christensen, and S.I. Hartwell. 2010. Assessment of Contaminants in Sediments and Corals in Vieques. In: Bauer and Kendall (eds.), An Ecological Characterization of the Marine Resources of Vieques, Puerto Rico Part II: Field Studies of Habitats, Nutrients, Contaminants, Fish, and Benthic Communities. NOAA Technical Memorandum NOS NCCOS 110. Silver Spring, MD. 174 pp.

Reichelt-Brushett, A.J. and P.L. Harrison. 2005. The effect of selected trace elements on the fertilization success of several scleractinian coral species. Coral Reefs 24: 524-534.

Shine, J., and G. Wallace. 2000. Chemical aspects of organic carbon and ecosystem stress in benthic ecosystems. In: Ad Hoc Benthic Indicator Group - results of initial planning meeting. IOC Technical Series No 57, UNESCO. 65pp.

Solbakken, J.E., A.H. Knap, T.D. Sleeter, C.E. Searle and K.H. Palmork. 1984. Investigation into the fate of 14C-labeled xenobiotics (napthalene, phenanthrene, 2,4,5,2',4',5'-hexachlorobiphenyl, octachlorostyrene) in Bermudian corals. Marine Ecology Progress Series 16: 149-154.

Veron, J.E.N. 2000. Corals of the World. Volume 3. Australian Institute of Marine Science. Townsville, Australia..

U.S. Department of Human Health and Services (USDHHS). 1995. Polycyclic aromatic hydrocarbons toxicology profile. Agency for Toxic Substance and Disease Registry (ATSDR). Atlanta, GA. 487 pp.

U.S. Department of Human Health and Services (USDHHS). 1999. Toxicological profile for cadmium. Agency for Toxic Substance and Disease Registry (ATSDR). Atlanta, GA. 439 pp.

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.

US Geological Survey (USGS). 1998. Geology, Geochemistry, Geophysics, Mineral Occurrences and Mineral Resource Assessment for the Commonwealth of Puerto Rico. Open File Report 98-38. http://pubs.usgs.gov/of/1998/of98-038/ Accessed 7/31/2013.

Chapter 4: Terrigenous Sedimentation Patterns at Reefs Adjacent to the Guánica Bay Watershed

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BACKGROUND

Increased terrestrial runoff due to coastal development and watershed land-use changes is one of the primary threats to coral reef ecosystems worldwide (Burke et al., 2011). Elevated sedimentation levels have been linked to several types of reef degradation including fewer coral species, less live cover, reduced recruitment, lower growth rates and calcification, altered species composition and lower rates of reef accretion (ISRS, 2004; Rogers, 1990). Sedimentation can cause burial and smothering of corals and tissue necrosis (Erftemeijer et al., 2012). Increased terrestrial runoff into Guánica Bay and adjacent coastal waters is a primary concern of the Guánica Bay Watershed Management Plan (CWP, 2008). The work presented here is a component of a larger effort to characterize reef communities and the chemical/biological stressors affecting them in order to inform decision-making, as well as serve as a baseline to quantify the effectiveness of implemented watershed restoration efforts. This work specifically seeks to address project needs by determining the composition and accumulation rates of terrigenous materials accumulating in Guánica Bay and on adjacent reefs.

METHODOLOGY

Study Area

The study area encompasses Guánica Bay (approximately 17° 57' N, 66° 54.4' W, mouth of bay) and adjacent nearshore reefs in southwest Puerto Rico (Figure 4.1). The insular shelf here extends ~3 to 5 km offshore before dropping steeply to oceanic depths from the shelf break at a depth of ~20 m. The shelf contains both emergent and submerged linear reefs oriented roughly parallel to shore and rising from depths of ~10-20 m. The reefs are separated by open sandy and hardbottom areas (Kendall et al., 2001). In order to assess the influence of Guánica Bay on the sedimentary dynamics of adjacent reefs, nine reef sites were selected along an ~14 km long, shore-parallel transect roughly centered on the mouth of the bay. In addition, two (non-reef) sites were chosen to characterize sedimentary dynamics of Guánica Bay itself. These are sites S01 at the mouth of the Rio Loco where it empties into Guánica Bay and site S02 at the mouth of Guánica Bay where it opens into adjacent coastal waters. The study area was divided into four strata based on proximity to the bay. These are Bay (sites S01 and S02), East (sites S08, S09 and S10), South (sites S06, S07 and S11) and West (sites S03, S04 and S05).

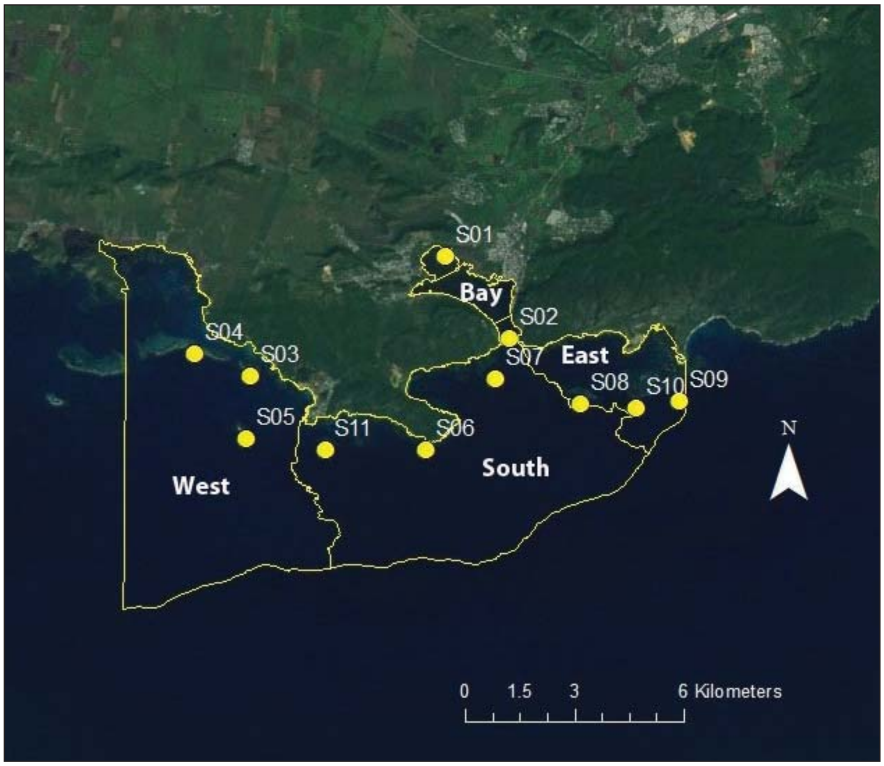

Figure 4.1. Map of study area showing location of sediment traps. Study area is divided into four strata based on proximity to Guánica Bay. These are Bay, East, South and West.

Sediment Traps

Sediment traps were clear plastic tubes 30-cm high with an internal diameter of 6.4 cm (Figure 4.2). Tops of the traps were covered with a 1 cm^2 nylon mesh to prevent entry of small fish and invertebrates. Traps were secured with plastic cable ties to steel rods that were driven into the hard reef surface. Due to the soft mud and sand substrate at the Bay sites (S01 and S02), the steel rods were attached to a plastic crate that was anchored to the seafloor with a cinder block. In all cases, the traps were situated so that the tops of the traps were ~60 cm above the surrounding seafloor. Similar traps and techniques have been used in previous studies (e.g., Bothner et al., 2006; Storlazzi et al., 2009). At reef sites, a pair

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of duplicate sediment traps was placed at a depth of ~10 m. At three of the reef sites (S04, S07 and S10), an additional pair of traps was placed at a depth of ~5 m and are designated as S04S, S07S and S10S, respectively. At the Bay sites, traps were situated at depths of 2 m (S01) and 8 m (S02). Traps were collected and processed monthly from August 2009 through July 2012. In some cases, weather or other logistical issues did not allow for the usual monthly collection and the traps were collected after an approximately two-month sampling interval. This includes December 2009-January 2010, August-September 2011, April-May 2012 and June-July 2012.

Figure 4.2. Simple tube traps used to collect suspended sediments. Sediment traps were clear plastic tubes 30-cm high with an internal diameter of 6.4 cm. Tops of the traps were covered with a 1 cm^2 nylon mesh to prevent entry of small fish and invertebrates. Traps were secured with plastic cable ties to steel rods that were driven into the hard reef surface. Traps were situated so that the tops of the traps were ~60 cm above the surrounding seafloor.

Laboratory Analyses

In the laboratory, material from one trap of each duplicate pair was separated by wet sieving into a coarse/sand fraction (>63 µm) and fine/mud fraction (<63 µm). Samples from all traps (sieved and unseived) were oven dried at 60°C and dry weight was determined. Monthly totals were taken as the average of each pair of traps. Trap accumulation rates were determined as total weight (mg)/area of trap opening (cm^2)/number of days in the field and expressed as mg cm^{-2} d^{-1}. A subset of samples was ground in an agate mortar for compositional analyses. Bulk carbon composition, including total carbon, total inorganic carbon and total organic carbon, was determined by carbon coulometry techniques (Engleman et al., 1985) conducted at the Limnological Research Center/National Lacustrine Core Facility, University of Minnesota – Twin Cities. Coulometric results were converted to percent calcium carbonate, percent organic material and percent other/terrigenous material.

Wave and Rainfall Data

Wave data are from the Caribbean Coastal Ocean Observing System Ponce Buoy located at 17° 51.61´N, 66° 31.42´W, ~42 km southeast of the mouth of Guánica Bay. From this data set, average daily wave heights were determined for each sampling interval over the study period from August 2009 - July 2012. Gaps in wave data are due to technical issues or servicing of the buoy. Rainfall data were collected at the Isla Magueyes Marine Laboratories in La Parguera, Puerto Rico, ~15 km west of the mouth of Guánica Bay. Data were taken from climatological records kept by the Bio-Optical Oceanography Laboratory, Department of Marine Sciences, University of Puerto Rico at Mayagüez. Total rainfall was determined for each sampling interval over the study period from August 2009 - July 2012.

Statistical Analysis

Because the data was not normally distributed (Shapiro-Wilk test), non-parametric techniques (Dunn Method, α=0.05) were used to examine statistical differences between the strata.

RESULTS

Trap accumulation data for the entire sampling period of August 2009 through July 2012 are presented here. Bulk compositional data are presented only for the period of August 2009 through May 2011.

Average Bulk Composition

Average composition of trap sediments in terms of percent calcium carbonate, percent organic material and percent terrigenous material is shown in Figures 4.3 and 4.4. Not surprisingly, the Bay sites consistently have higher percentages of terrigenous material than the reef sites, though there is a

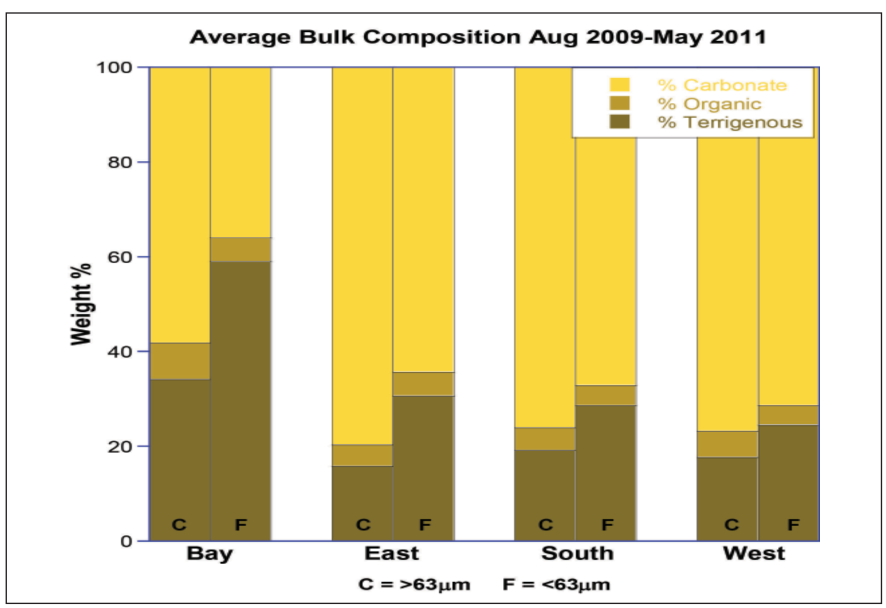

Figure 4.3. Average bulk composition of trap sediments from August 2009 through May 2011 given in weight percent calcium carbonate, organic material and terrigenous material for the coarse (>63 µm) and fine fractions (<63 µm).

considerable difference in composition between the two bay sites (S01 and S02). In all cases, the fine fraction (<63 µm) contains a higher percentage of terrigenous material. Organic contents are low, typically less than 10% with an average of ~5%. Site S01 near the mouth of the Rio Loco in Guánica Bay has the highest average terrigenous content at ~87%, with the remainder roughly equally split between carbonate and organic material. In contrast, Site S02 at the mouth of Guánica Bay has an average terrigenous content of ~30%, with calcium carbonate and organic contents of ~65% and ~5%, respectively. Reef sites have an average percent terrigenous material that ranges from ~21% to 28%, with averages of 24% at East sites (S08, S09 and S10), 26% at South sites (S07, S06 and S11) and 23% at West sites (S03, S04 and S05). The remainder consists primarily of calcium carbonate (aragonite and magnesium calcite), representing *in situ* production, with an average of only ~5% organic material.

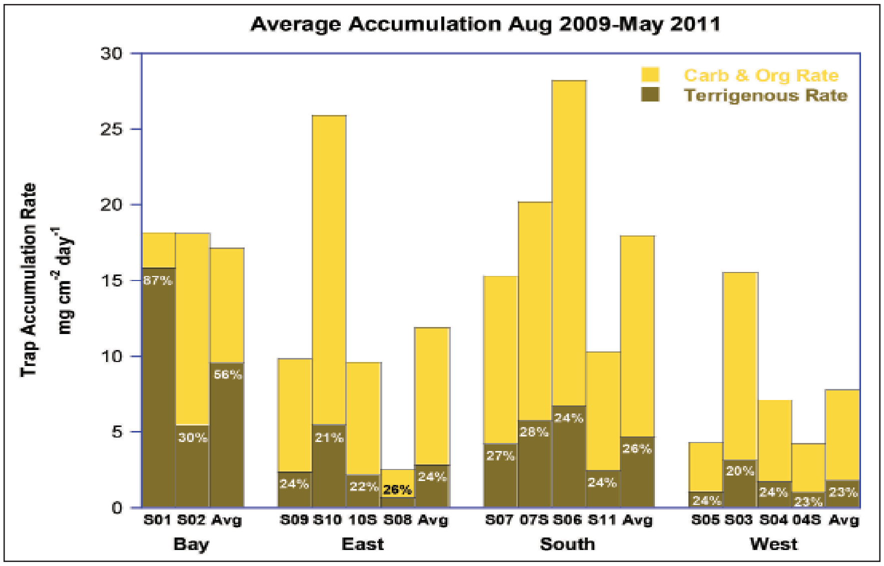

Figure 4.4. Average trap accumulation from August 2009 through May 2011, showing percent terrigenous material.

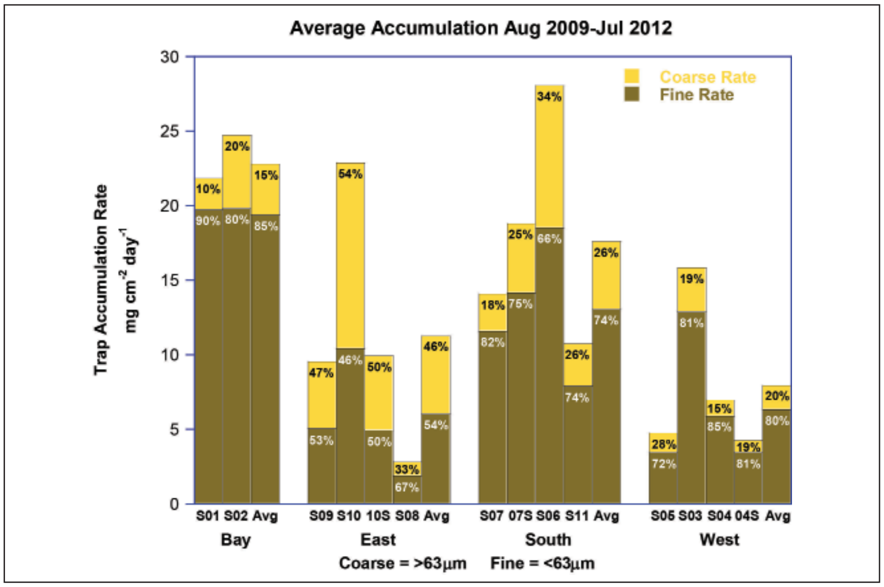

Figure 4.5. Average trap accumulation rate from August 2009 through July 2012, showing percentages of coarse-(>63 µm) and fine-grained (<63 µm) material.

Average Trap Accumulation Rates

Trap accumulation rates are highly variable on both spatial and temporal scales. Average accumulations rates are shown in Figures 4.4 and 4.5. Over the entire sampling period of August 2009 through July 2012 (Figure 4.5), the Bay sites had average trap accumulation rates of 22 mg cm^{-2} d^{-1} (S01) and 25 mg cm^{-2} d^{-1} (S02), with 80-90% of this being fine-grained material (<63 µm). In general, these rates are higher than average accumulation rates at the reef sites over the same period, though two of the reef sites (S06 and S10) had average rates that met or exceeded those measured in the bay. Reef sites had average trap accumulation rates that ranged from 3 to 28 mg cm^{-2} d^{-1}, with averages of 11 mg cm^{-2} d^{-1} at East sites (S08, S09 and S10), 18 mg cm^{-2} d^{-1} at South sites (S07, S06 and S11) and 8 mg cm^{-2} d^{-1} at West sites (S03, S04 and S05). Trap sediments at South and West sites consisted of considerably more fine-grained material averaging 74% and 80%, respectively, while trap sediment at East sites averaged 54% fine-grained material.

Temporal Patterns

Temporal patterns in trap accumulation and composition are shown in Figures 4.6 to 4.12. Figure 4.6 compares trap accumulation over time between the two Bay sites, S01 (Rio Loco mouth) and S02 (Guánica Bay mouth). Although the patterns are similar, there is not a consistent relationship between the two sites, indicating that separate drivers are affecting sedimentary dynamics at these sites. Terrigenous content at the river mouth is very uniform and remains within a few percent of the average value of 87% (Figure 4.7). Terrigenous content at the bay mouth is more variable ranging from 22-41%, but in most cases is very close to the average of 30%. Figure 4.8 and 4.9 provide more detail on sedimentary dynamics at the mouth of Guánica Bay (site S02), the pathway through which sediments from the bay must travel to reach adjacent reefs. In most cases, increases in trap accumulation correspond with decreases in the percentage of fine-grained material (Figure 4.8). This suggests that peaks in trap accumulation are associated with high-energy events during which fine-grained material is not able to settle. In general, trap accumulation and terrigenous accumulation (trap accumulation rate multiplied by percent terrigenous material) vary together (Figure 4.9). In contrast, the percent terrigenous typically decreases during peaks in trap accumulation. Thus, increases in trap accumulation do not necessarily correspond to increases in the relative amount of terrigenous material.

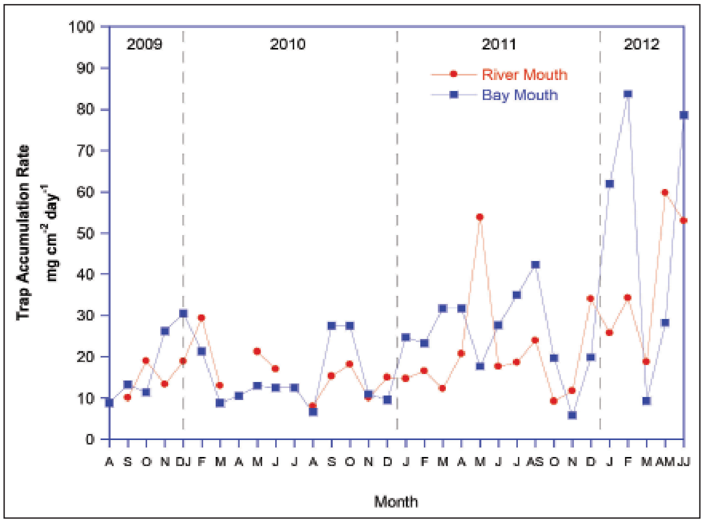

Figure 4.6. Trap accumulation rates by month from August 2009 through July 2012 for the two Bay sites, S01 at the mouth of Rio Loco and S02 at the mouth of Guánica Bay.

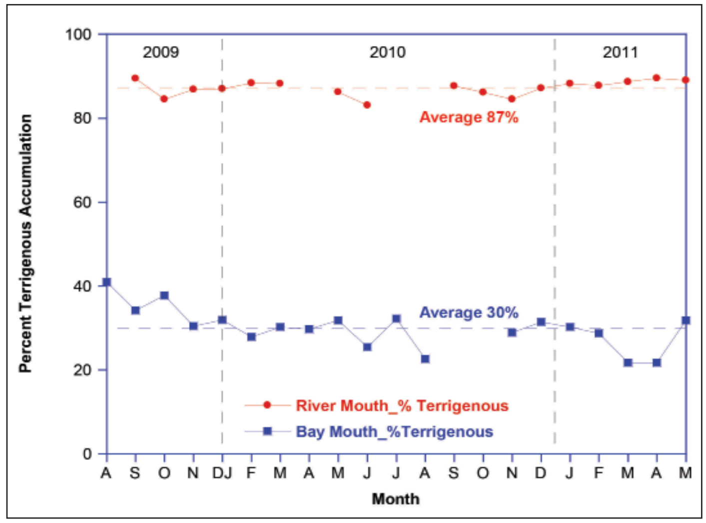

Figure 4.7. Percent terrigenous accumulation by month from August 2009 through May 2011 for the two Bay sites, S01 at the mouth of Rio Loco and S02 at the mouth of Guánica Bay.

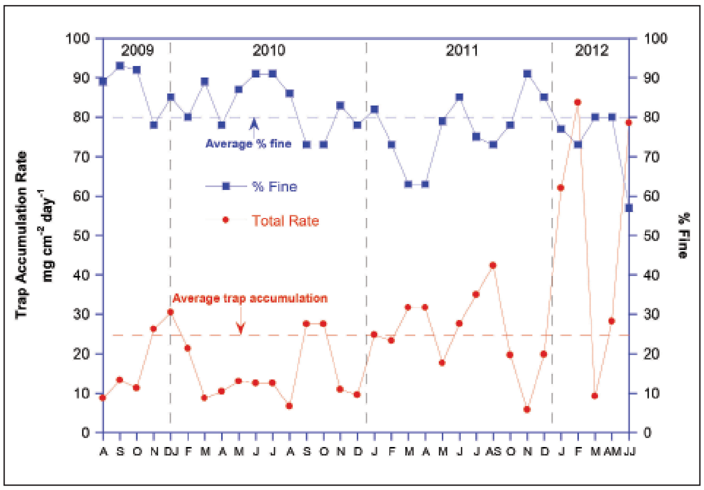

Figure 4.8. Comparison of trap accumulation rate with percent fine-grained material (<63 µm) by month from August 2009 through July 2012 for site S02 at the mouth of Guánica Bay. Note that increases in trap accumulation are typically associated with a decrease in the percentage of fine-grained material.

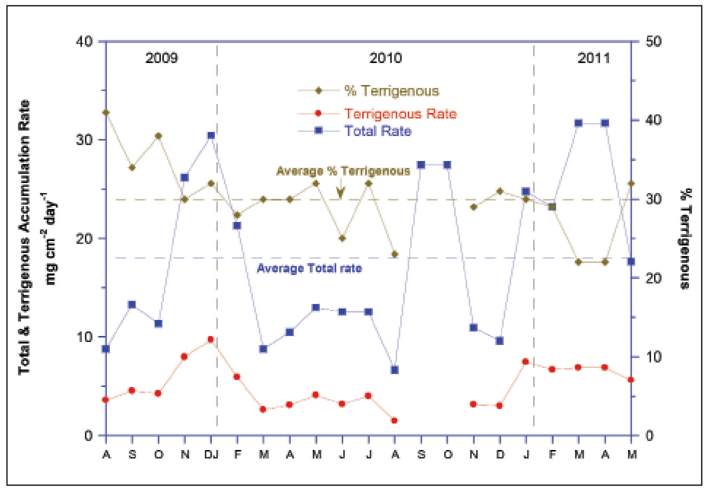

Figure 4.9. Comparison of total trap accumulation rate, terrigenous accumulation rate and percent terrigenous material by month from August 2009 through May 2011 for site S02 at the mouth of Guánica Bay. Note that total and terrigenous accumulation rates vary together, while increases in trap accumulation rate are typically associated with decreases in percent terrigenous material.

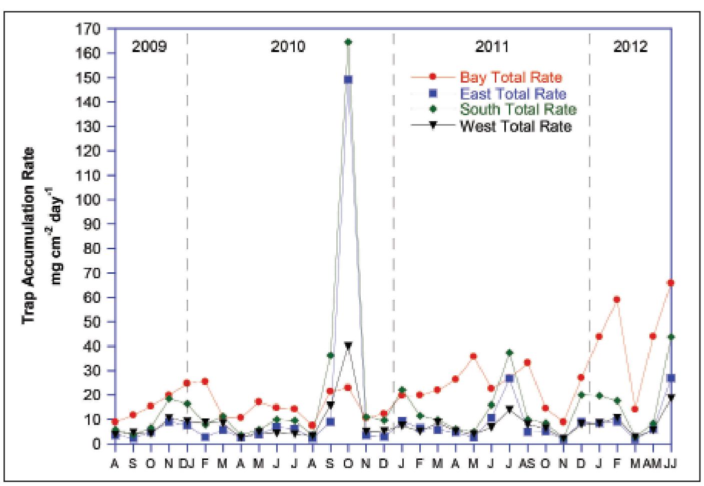

Figure 4.10. Comparison of average trap accumulation rates by month for the four different strata of the study area (Bay, East, South and West). Note that the reef sites (East, South and West) exhibit similar temporal patterns, while the Bay sites behave somewhat differently. The record is punctuated by several high-magnitude events, most notably October 2010, July 2011 and June-July 2012.

Table 4.1. Results of Dunn Method (α=0.05) to determine differences between strata for total sedimentation rate and terrigenous sedimentation rate. Note: p-values in italics denote statistical significance.

Variable	Stratum	Stratum	p-Value
Sedimentation Rate	River Mouth	East	*<0.001*
Sedimentation Rate	River Mouth	Central	*0.004859*
Sedimentation Rate	West	East	1
Sedimentation Rate	River Mouth	Bay Mouth	1
Sedimentation Rate	West	Central	0.303524
Sedimentation Rate	East	Central	0.079343
Sedimentation Rate	Central	Bay Mouth	*0.003877*
Sedimentation Rate	West	Bay Mouth	*<0.001*
Sedimentation Rate	West	River Mouth	*<0.001*
Sedimentation Rate	East	Bay Mouth	*<0.001*
Terrigenous sedimentation rate	River Mouth	East	*<0.001*
Terrigenous sedimentation rate	River Mouth	Central	*<0.001*
Terrigenous sedimentation rate	River Mouth	Bay Mouth	0.772657
Terrigenous sedimentation rate	West	East	1
Terrigenous sedimentation rate	West	Central	0.058889
Terrigenous sedimentation rate	East	Central	*0.024139*
Terrigenous sedimentation rate	Central	Bay Mouth	*0.012323*
Terrigenous sedimentation rate	West	Bay Mouth	*<0.001*
Terrigenous sedimentation rate	East	Bay Mouth	*<0.001*
Terrigenous sedimentation rate	West	River Mouth	*<0.001*

Figure 4.10 compares trap accumulation rates among the four strata of the study area. Table 4.1 shows the statistical differences (Dunn Method, α=0.05) between the strata. Reef sites (East, South and West) exhibit a very similar temporal pattern with concomitant increases and decreases in trap accumulation, indicating a uniform driver of sedimentary dynamics among the reef sites. The Bay stratum displays a slightly different temporal pattern and there is not a consistent temporal relationship between the Bay and the reef sites. East and West sites generally have similar accumulation rates and remain below ~10 mg cm^{-2} d^{-1} much of the time. South sites display a similar temporal pattern in accumulation to the other reef sites, but typically record higher rates. The record is punctuated by several high-magnitude events with accumulation rates that, in some cases, are an order of magnitude higher than the long-term average, most notably October 2010, July 2011 and June-July 2012.

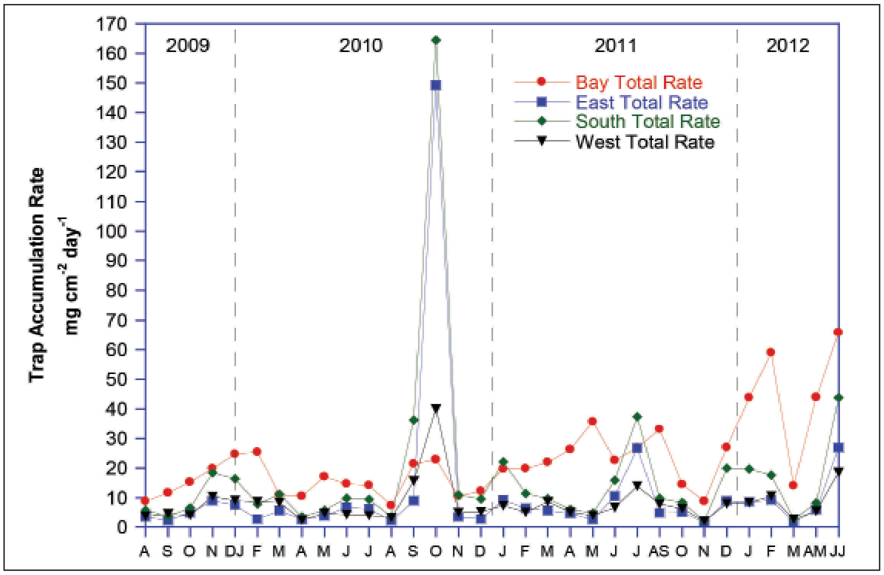

Figure 4.11. Comparison of trap accumulation rate with percent fine-grained material (<63 μm) by month from August 2009 through July 2012 for South sites (S06, S07, S07S and S11). Note that increases in trap accumulation are typically associated with a decrease in the percentage of fine-grained material.

To examine sedimentary dynamics at reef sites in more detail, results from the South sector are shown in Figures 4.11 and 4.12. These are representative of sedimentary dynamics at the other reef sites. At the South sites, over much of the study period, trap accumulation remains close to or below the long-term average of ~18 mg cm^{-2} d^{-1}. The exceptionally high rates recorded during October 2010 greatly affect the long-term average. Removing this one event from the record would reduce the long-term average by about 27% down to ~ 13 mg cm^{-2} d^{-1}. Peaks in trap accumulation (e.g., October 2010, July 2011 and June-July 2012) are associated with decreases in percent fine material (Figure 4.11). As with site S02 at the mouth of Guánica Bay (Figure 4.8), this indicates that these exceptionally high accumulation rates are associated with high-energy events during which fine-grained material is not able to settle. In general, total trap accumulation and terrigenous accumulation vary directly together (Figure 4.12). In contrast, the percent terrigenous has more of an inverse relationship with total trap accumulation. Peaks in trap accumulation are associated with decreases in percent terrigenous (e.g., October 2010), while the highest percent terrigenous values are associated with relatively low (below average) accumulation rates. In addition, while accumulation rates are highly variable, the percent terrigenous material is relatively uniform, remaining within ~3% of the long-term average over most of the record.

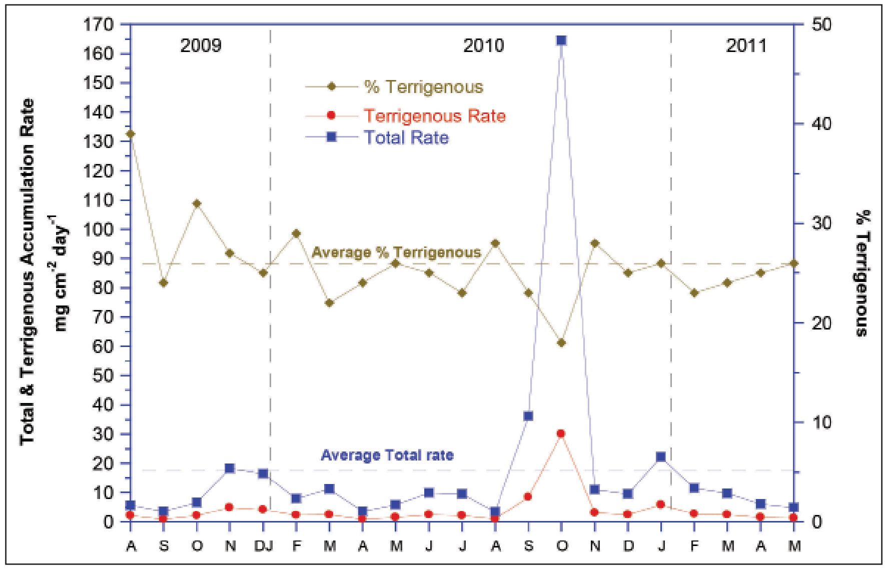

Figure 4.12. Comparison of total trap accumulation rate, terrigenous accumulation rate and percent terrigenous material by month from August 2009 through May 2011 for South sites (S06, S07, S07S and S11). Note that total and terrigenous accumulation rates vary together, while increases in trap accumulation rate are typically associated decreases in percent terrigenous material.

DISCUSSION

Results of a three-year sediment-trap study establish an important baseline of spatial and temporal trends in sediment accumulation rates and composition at nearshore reefs adjacent to the Guánica Bay watershed. In general, Bay sites (S01 and S02) had higher accumulation rates and a higher percentage of terrigenous material than the reef sites (Figures 4.4 and 4.5). While terrigenous content at the mouth of the Rio Loco (S01) was fairly uniform and remained within a few percent of the average value of 87%, terrigenous content at the mouth of Guánica Bay (S02) was more variable and considerably lower, ranging from 22-41%, but in most cases remaining close to the average of 30% (Figure 4.7). This marked decrease in terrigenous content from the mouth of the Rio Loco to the mouth of Guánica Bay is likely a function of two factors. First, it indicates that Guánica Bay serves as an important sink of terrigenous materials delivered to the coast by the Rio Loco. Second, the composition of sediments accumulating at the mouth of the bay is greatly affected by *in situ* production of carbonate sediments on the insular shelf. In addition, because site S02 is situated within the primary conduit through which terrigenous sediments from the bay must pass to reach adjacent reefs, the composition of trap sediments at S02 represents an upper end number of what would be expected at adjacent reefs. That is, reef sites would not be expected to receive sediments with a terrigenous content above ~30%, the average at site S02. Indeed, all reef sites have average terrigenous contents below 30%, ranging from 21% to 28%.

All reef sites exhibit similar temporal patterns of trap accumulation rates (Figure 4.10), indicating that they are responding to the same driver (or drivers) of sedimentary dynamics. The similarity of these long-term records also indicates that the sediment traps were recording actual trends in sedimentary dynamics and not isolated occurrences at an individual site. Among the reef sites, South sites (S06, S07 and S11) generally have higher average trap accumulation rates and higher percent terrigenous material than East and West sites (Figures 4.4 and 4.5). This is consistent with their location just to the west of the mouth of Guánica Bay and mean westward flowing coastal currents along the shelf (Warne et al., 2005). Average trap accumulation rates and percent terrigenous material among the reef sites are comparable to, but slightly higher than accumulation rates and percent terrigenous material measured at inner- and middle-shelf reefs off La Parguera, a few kilometers to the west of Guánica (Hernandez et al., 2009). This is consistent with the reefs off La Parguera being further removed from terrestrial influence with no major river inputs (Hernandez et al., 2009; Warne et al., 2005). However, because different types of sediment traps were used for the La Parguera study, it is difficult to directly compare these results (Storlazzi et al., 2011).

Average chronic sedimentation rates of 10 mg cm^{-2} d^{-1} have been proposed as a threshold for initiating stress responses in coral reefs including fewer coral species, reduced live coral cover, lower growth rates and slower rates of reef accretion (Rogers, 1990). Pastorok and Bilyard (1985) suggested that rates of 10-50 mg cm^{-2} d^{-1} could be considered as moderate to severe. However, other studies indicate that some corals and reefs can tolerate sedimentation rates >400 mg cm^{-2} d^{-1} (Erftemeijer et al., 2012 and references therein). Still, studies by Nemeth and Nowlis (2001) and Smith et al. (2008) in the U.S. Virgin Islands lend support to 10 mg cm^{-2} d^{-1} as a reasonable "working" threshold for the region and to increased sedimentation as a primary driver of reef degradation at nearshore sites. In the current study, all South sites have average trap accumulation rates in excess of 10 mg cm^{-2} d^{-1} and, thus, may be considered at least moderately sediment stressed. In contrast, most East and West sites have average trap accumulation rates below 10 mg cm^{-2} d^{-1}. Only site S10 in the East sector and S03 in the West have average trap accumulation rates above 10 mg cm^{-2} d^{-1}.

Trap accumulation rates are highly variable on both spatial and temporal scales and can range by an order of magnitude from one monthly sampling interval to the next. In contrast, sediment composition remains relatively uniform over time (Figures 4.9, 4.10 and 4.12). The rough inverse relationship between trap accumulation rate and percent fine-grained material (Figures 4.8 and 4.11) suggests that elevated trap accumulation rates are associated with high-energy events, such as wave events that can resuspend large amounts of bottom sediments. Terrigenous accumulation rates vary directly with total trap accumulation. While terrigenous accumulation rates are variable over time, the percentage of terrigenous material remains fairly constant, or deceases slightly during peak accumulation events (Figure 4.13). Even the percentage of terrigenous material in the fine fraction remains relatively constant over time. Thus, peaks in trap accumulation rates do not correspond to increases in the relative amount of terrigenous material in trap sediments, i.e., they do not correspond to influxes of new terrigenous material. Similar temporal patterns in accumulation rates among the sites without corresponding changes in composition of sediments point to resuspension of bottom sediments by wave action as a primary driver of sedimentary dynamics at these reefs. This is further supported by comparison of trap accumulation rates with rainfall and wave height records (Figure 4.14).

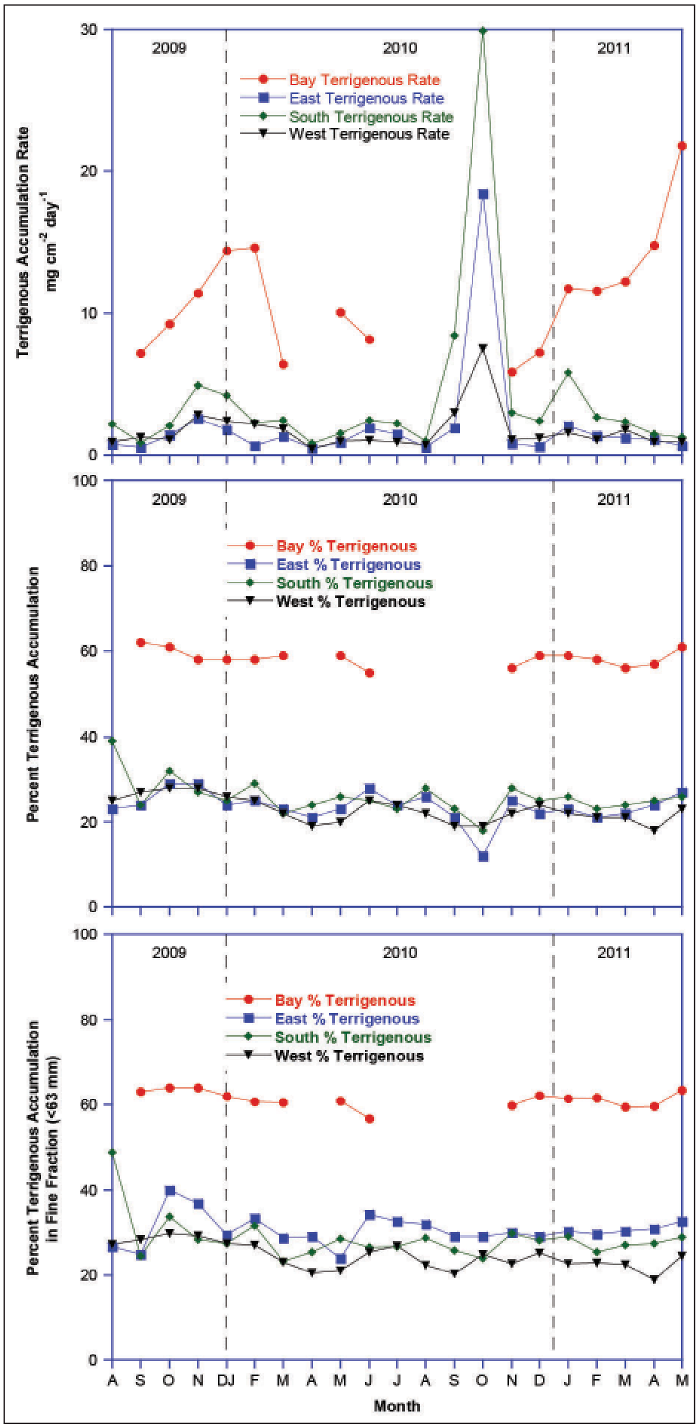

Figure 4.13. Terrigenous accumulation rates by month in the four strata of the study area from August 2009 through May 2011 (top). Percent terrigenous accumulation by month in the four strata of the study area from August 2009 through May 2011 (center). Percent terrigenous accumulation in the fine fraction (<63 µm) by month in the four strata of the study area from August 2009 through May 2011(bottom). Note that while terrigenous accumulation rates are variable, the percentage of terrigenous accumulation remains quite constant both in the whole sample and in the fine fraction.

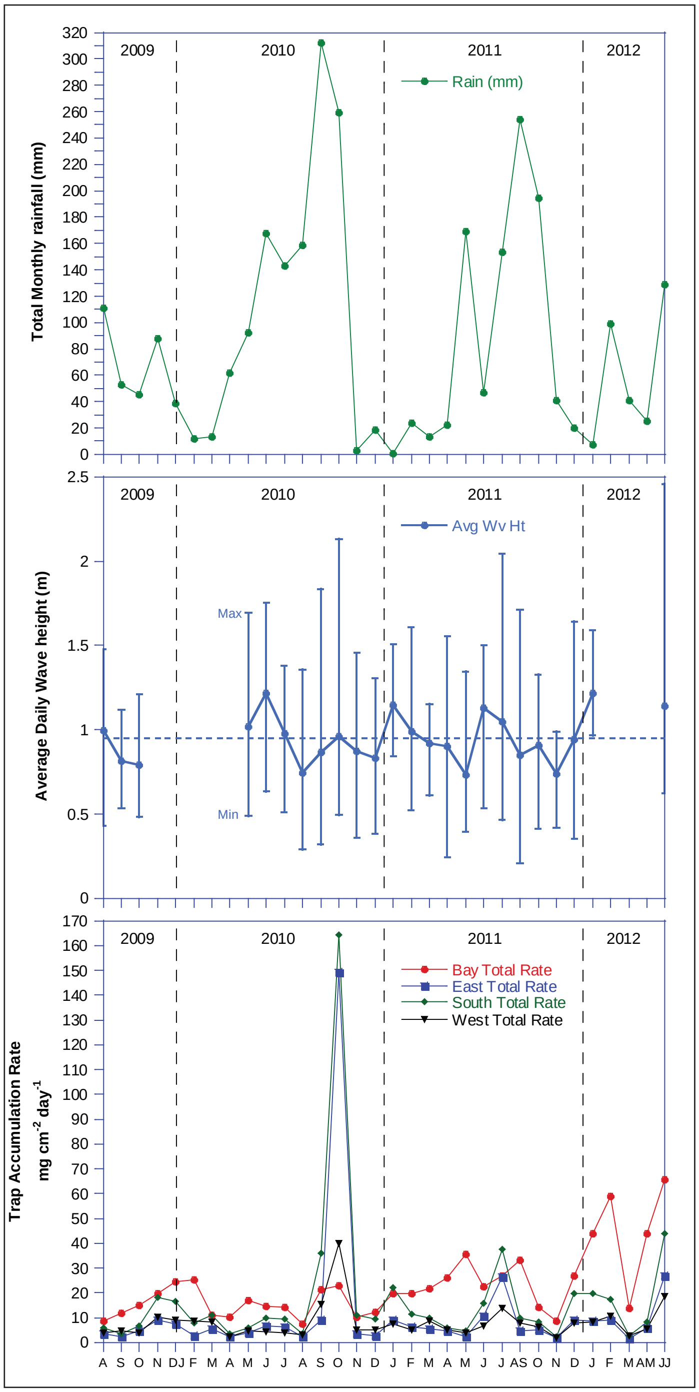

Figure 4.14. Total rainfall during each of the sampling intervals from August 2009 through July 2012 (top). Average daily wave height during each of the sampling intervals from August 2009 through July 2012 (center). Solid line shows average daily wave height during each sampling interval. Vertical bars show range of average daily waves heights during each interval. Average trap accumulation rates by month for the four different strata of the study area (Bay, East, South and West; bottom). Note that high-magnitude events in October 2010, July 2011 and June-July 2012 correlate with the three highest (max) values of average daily wave height, each greater than 2 m.

Three notable peaks in trap accumulation occur in October 2010, July 2011 and June-July 2012. While these are intervals of moderate to high rainfall, they do not stand out in the rainfall record and other intervals with similar or higher levels of rainfall do not correspond to such high peaks in trap accumulation. When compared to the wave height record, the peaks in trap accumulation in October 2010, July 2011 and June-July 2012 correspond to the three highest values of average daily wave height recorded over the three-year study period, each over 2 m.

When assessing the degree to which a reef is potentially influenced by terrigenous sedimentation, it is important to consider in conjunction both total sediment accumulation and the percentage of terrigenous material (Figure 4.15). Reefs negatively impacted by terrigenous sedimentation would be expected to plot towards the upper right in Figure 4.15, i.e., they would be experiencing both high sediment accumulation rates along with a high percentage of terrigenous material in the sediment. Note that site S02 at the mouth of Guánica Bay plots furthest to the right with the highest percentage of terrigenous material as well as a high average accumulation rate, which is consistent with this site's location. All reef sites plot to the left of S02. Sites S07 and S07S (S07S are shallow traps at 5 m water depth, just adjacent to S07) plot closest to S02 consistent with their close proximity to the mouth of Guánica Bay and indicative of terrigenous sediment stress. Site S06 also plots in the upper right, suggestive of terrigenous sediment stress and consistent with its location just down current from the mouth of Guánica Bay. It has the highest average accumulation rate with a moderate percentage of terrigenous material. Together, these factors result in S06 having the highest average terrigenous accumulation rate of the reef sites. Among the other reef sites spatial (i.e., east-west) trends are not readily apparent as plotted in Figure 4.15.

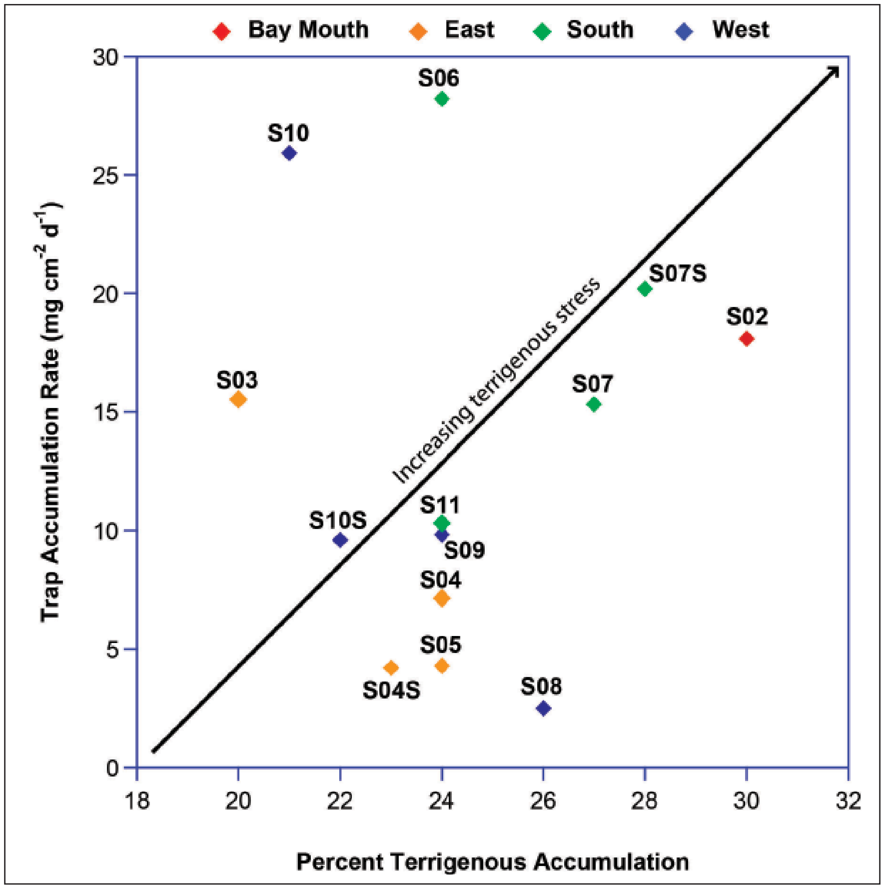

Figure 4.15. Average trap accumulation versus percent terrigenous material at all reef sites and at the mouth of Guánica Bay (S02). The arrow indicate potential for increased stress due to terrigenous sedimentation. Sites that plot in the upper right of the graph are exposed to higher level of terrigenous accumulation.

Guánica Bay is certainly a source of terrigenous materials to the adjacent coastal ocean. Observations by field personnel during the course of the study frequently included visible plumes of turbid water exiting the bay, and this is reinforced by the high terrigenous content of those sites closest to the mouth of the bay, i.e., S02, S07 and S07S. The lack of consistent east-west trends among the other sites in either trap accumulation rates or percent terrigenous material is suggestive of several causative factors. First, the Guánica coast is an open, energetic and well-mixed system. Terrestrial inputs are quickly diluted by the large reservoir of shelf sediments. Terrigenous influx events are difficult to identify with sediment traps, because these events are typically coincident with storms and high waves when traps are primarily collecting resuspended, well-mixed sediments. Second, the assumption of relatively uniform westerly flow along the Guánica shelf may be overly simplistic. Terrigenous materials exiting Guánica Bay may be being transported in multiple directions, which would account for the lack of east-west trends relative to the mouth of the bay. However, this would suggest a general decrease in relative amounts of terrigenous materials with distance from the bay, which is not evident in trap samples. Finally, it is estimated that 1,000 to 4,300 t km^{-2} yr^{-1} of suspended sediment are discharged to coastal waters of south/southwestern Puerto Rico, much of this delivered to the coast east of Guánica (Warne et al., 2005). Prevailing flow along the insular shelf could transport this material westward and constitute an important source of terrigenous sediments reaching the reefs off Guánica.

CONCLUSIONS

The Guánica coastline is an open, energetic and well-mixed system. Sediment trap accumulation rates are highly variable on both spatial and temporal scales. Low-frequency, high-magnitude, resuspension events are extremely important components of the sedimentary dynamics of these reefs. Sediment composition remains fairly uniform on both spatial and temporal scales. Spikes in the percentage terrigenous material (i.e., terrigenous influx events) are not evident. The uniform composition of trap sediments regardless of trap accumulation rate indicates that resuspension of bottom sediments is a primary driver of trap accumulation rates and patterns. Those sites closest to Guánica Bay show higher trap accumulation rates and percentage terrigenous material. However, there are no consistent east-west trends in either trap accumulation rates or percent terrigenous material, with respect to proximity to the bay. While Guánica Bay is a local source of terrigenous materials it is not necessarily the sole or primary source. Other up-current sources to the east are likely equally important.

REFERENCES

Bothner, M.H., Reynolds, R.L., Casso, M.A., Storlazzi, C.D., Field, M.E., 2006. Quantity, composition, and source of sediment collected in sediment traps along the fringing coral reef off Molokai, Hawaii. Marine Pollution Bulletin 52, 1034.

Burke, L., Reytar, K., Spalding, M., Perry, A., 2011. Reefs at risk revisited. World Resources Institute, Washington, DC. 114 pp.

Center for Watershed Protection (CWP). 2008. Guánica Bay Watershed Management Plan. NOAA Coral Reef Program and DRNA, Puerto Rico. http://www.cwp.org/online-watershed-library/cat_view/64-manuals-and-plans/81-watershed-management-plans-and-guidance

Engleman, E.E., L.L. Jackson, and D.R. Norton. 1985. Determination of carbonate carbon in geological materials by coulometric titration. Chemical Geology 53: 125.

Erftemeijer, P.L.A., B. Riegl, B.W. Hoeksema, and P.A. Todd. 2012. Environmental impacts of dredging and other sediment disturbances on corals: A review. Marine Pollution Bulletin 64: 1737-1765.

Hernandez, R., C. Sherman, E. Weil, and P. Yoshioka. 2009. Spatial and temporal patterns in reef sediment accumulation and composition, southwestern insular shelf of Puerto Rico. Caribbean Journal of Science 45: 138-150.

ISRS (International Society for Reef Studies). 2004. The effects of terrestrial runoff of sediments, nutrients and other pollutants on coral reefs. International Society for Reef Studies. 18 pp.

Kendall, M.S., M.E. Monaco, K.R. Buja, J.D. Christensen, C.R. Kruer, M. Finkbeiner, and R.A. Warner. 2001. Methods used to map the benthic habitats of Puerto Rico and the U.S. Virgin Islands. NOAA National Ocean Service, Silver Spring, MD. 44 pp.

Nemeth, R.S. and J.S. Nowlis. 2001. Monitoring the effects of land development on the near-shore reef environment of St. Thomas, USVI. Bulletin of Marine Science 69: 759-775.

Pastorok, R.A. and G.R. Bilyard. 1985. Effects of sewage pollution on coral-reef communities. Marine Ecology Progress Series 21: 175-189.

Rogers, C.S. 1990. Responses of coral reefs and reef organisms to sedimentation. Marine Ecology Progress Series 62: 185-202.

Smith, T.B., R.S. Nemeth, J. Blondeau, J.M. Calnan, E. Kadison, and S. Herzlieb. 2008. Assessing coral reef health across onshore to offshore stress gradients in the US Virgin Islands. Marine Pollution Bulletin 56: 1983.

Storlazzi, C., M. Field, and M. Bothner. 2011. The use (and misuse) of sediment traps in coral reef environments: theory, observations, and suggested protocols. Coral Reefs 30: 23-38.

Storlazzi, C.D., M.E. Field, M.H. Bothner, M.K. Presto, and A.E. Draut. 2009. Sedimentation processes in a coral reef embayment: Hanalei Bay, Kauai. Marine Geology 264: 140.

Warne, A.G., R.M.T. Webb, and M.T. Larsen. 2005. Water, sediment, and nutrient discharge characteristics of rivers in Puerto Rico, and their potential influence on coral reefs, U.S. Geological Survey Scientific Investigations Report 2005-5206. 58 pp.

Chapter 5: Spatial and Temporal Variability in Surface Water Nutrients in and around Guánica Bay, Puerto Rico

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BACKGROUND

Primary productivity in marine systems is most often limited by nitrogen, but phosphorus can be co-limiting under certain circumstances. Estuarine and coastal systems can vary from nitrogen limitation to phosphorus limitation over space and time. Nutrient enrichment can result in algal blooms, changes in algal community composition (including increasing occurrence and extent of harmful algal blooms) and increases in hypoxia/anoxia (Bricker et al., 2007).

In tropical systems, excess nutrient loads can cause increases in macroalgal growth and can have deleterious effects on corals, such as macroalgae outcompeting and overgrowing corals. Furthermore, nitrogen and phosphorus can impact corals directly by lowering fertilization success (Harrison and Ward, 2001), and reducing both photosynthesis and calcification rates (Marubini and Davis, 1996). Useful reviews summarizing the biogeochemistry of nitrogen and phosphorus in coral systems have been previously published (e.g., McCook, 1999; Bell, 1992).

Land-based contributions of nutrients to coastal systems originate from a variety of sources. Phosphorus and reactive nitrogen can enter the environment from chemical fertilizers (agriculture, lawns, golf courses), industrial sources, animal waste and human waste (Galloway et al., 2003). Additionally, nitrogen can be contributed from biological nitrogen fixation and atmospheric nitrogen deposition (originating from fossil fuel combustion and ammonia volatilization from agriculture; Mathews et al., 2002).

Although a comprehensive watershed nutrient budget is beyond the scope of this study, both human waste and agriculture are potentially important sources of new nutrients to Guánica Bay. There are two wastewater treatment plant (WWTP) facilities in the study area. The primary facility, Puerto Rico Aqueduct and Sewer Authority Guánica Sewage Treatment Plant, is located on the eastern shore of the bay. Plans are being considered to add additional households to the WWTP inflow. It employs tertiary treatment, which removes a large portion of the nitrogen from the waste stream. A second, much smaller WWTP sits on the coast to the east of the bay. It treats waste from public facilities at a beach access and likely represents only minor nutrient inputs to the coastal system.

Agriculture in the watershed includes low density pasture land, a variety of crop lands and coffee farms (see further discussion in Chapter 1).

Objectives

The goals of this portion of the baseline assessment of Guánica Bay were to:

1. quantify magnitude and spatiotemporal variability of surface water nutrients in the bay and the offshore coral reef ecosystem;
2. establish a baseline of nutrient conditions against which to measure changes in the future; and
3. link observed concentrations of nutrients to hydrologic forcing factors and possible nutrient sources.

METHODS

Nutrient Sampling

Eighteen sites were randomly selected from six geographic strata (three sites per strata, Figure 5.1). From 2009 to 2012, each of these sites was visited monthly to collect surface water grab samples. Additionally, two targeted sites in the bay, mouth of Rio Loco (N31) and mouth of bay (N32), and three targeted riverine sites in the watershed were sampled: the southernmost site (N31) being close to the outflow point of the river into the bay, the westernmost site (N33) which is on the agricultural drainage canal upstream from where it joins the Rio Loco, and the northernmost site which is on the main stem of the Rio Loco (N34; Figure 5.1). The targeted sites were selected in order to capture nutrient signals at hydrologically important locations, i.e., mouth of river, mouth of bay, the drainage canal and the mainstem of the Rio Loco. Watershed sites were targeted rather than random due to logistical considerations regarding site access.

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Nutrient samples were collected in high density polyethylene bottles from 0.1 m below the surface. At extremely shallow sites (e.g., riverine sites at lower flows), samples were taken at half the distance to the bottom. In this situation, care was taken to exclude sediment from the samples. Bottles were rinsed three times with site water prior to sampling. Nitrile or latex gloves were worn by field personnel to avoid contamination of the samples during handling. Samples were stored on ice, in the dark while in the field and frozen at -20°C upon returning to the lab and not thawed until immediately prior to analysis. Samples were not filtered so that total nutrient levels could be analyzed, rather than only dissolved levels.

Figure 5.1. Map of sampling sites.

Analytical Methods Used for the Analysis of Nutrients in Water

Nutrient laboratory analyses were performed at a contract lab (Geochemical and Environmental Research Group, Texas A&M University). Water samples were analyzed for a standard suite of nutrient analytes: nitrate (NO_3^-), nitrite (NO_2^-), orthophosphate (HPO_4=), ammonium (NH_4^+), urea ($(NH_2)_2CO$), total nitrogen and total phosphorus (Table 5.1).

Nitrate and nitrite analyses were based on the methodology of Armstrong et al. (1967). Orthophosphate was measured using the methodology of Bernhardt and Wilhelms (1967) with the modification of hydrazine as reductant. Silicate determination was accomplished using the methods of Armstrong et al. (1967) using stannous chloride. Ammonium analysis was based on the method of Harwood and Kuhn (1970) using dichloro-isocyanurate as the oxidizer. Urea was measured using diacetyl-monoximine and themicarbozide. The total concentrations of nitrogen and phosphorus were determined after an initial decomposition step. This method involves persulfate oxidation while heating the sample in an autoclave (115°C, 20 minutes; Hansen and Koroleff, 1999). After oxidation of the samples, nutrient determination was conducted on the Technicon II analyzer for nitrate and orthophosphate.

Table 5.1. Details on analytical methods for nutrients. Units are expressed as mg-N/L or mg-P/L.

Analyte	Method Detection Limit (µM)	Method Detection Limit (mg/L)	Standard Range (µM)	Standard Range (mg/L)
NO_3^-	0.177	0.010	3.85 - 30.14	0.23 - 1.86
NO_2^-	0.010	0.0004	0.09 - 0.72	0.006 - 0.033
$HPO_4^=$	0.030	0.002	0.35 - 2.18	0.021 - 0.21
$HSIO_3^-$	0.155	0.014	4.05 - 30.08	0.25 - 2.80
NH_4^+	0.070	0.001	0.42 - 3.44	0.026 - 0.062
Urea	0.205	0.012	0.59 - 4.42	0.036 - 0.265

Statistical Analysis of Data

Data were analyzed using JMP® statistical software. The data were first tested for normality using the Shapiro-Wilk test. Because the data were not normally distributed, non-parametric statistical tests were used; a Dunn's multiple comparison test was used to examine differences between sites, seasons and years. Spearman's rank correlation coefficient test was used to examine relationships between analytes. Because they were not randomly selected, targeted sites were not included in analyses of differences between strata, but were included in correlative analyses and in summary statistics because they help to characterize the overall status of the system.

RESULTS AND DISCUSSION

Summary Statistics

Summary statistics (mean, median, maximum, minimum, standard deviation) for each analyte are presented in Table 5.2. These statistics are inclusive of the five targeted sites. These values are similar to what has been observed in other comparable studies in Puerto Rico (Table 5.3). Watershed sites presented in this study had higher concentrations than data collected for Jobos Bay and southwest Puerto Rico, but those studies did not include any sites in the watershed, so this is not surprising.

Table 5.2. Summary statistics for nutrient analytes across all sites. Units are expressed as mg-N/L or mg-P/L.

	Mean	Median	StDev	Max	Min
NO_3^-+NO_2^-	0.14	0.01	0.32	3.68	0.00
NH_4^+	0.05	0.01	0.23	6.45	0.00
Urea	0.01	0.00	0.02	0.42	0.00
Total N	0.46	0.28	0.56	10.94	0.01
$HPO_4^=$	0.05	0.01	0.11	1.48	0.00
Total P	0.08	0.03	0.14	1.95	0.00

Table 5.3. Comparison with other sites in Puerto Rico. Vieques data from Whitall et al., 2010; Jobos data from Whitall et al., 2011. Units are expressed as mg-N/L or mg-P/L.

	Guánica				Vieques			Jobos		
	Mean	Median	StDev	Max	Mean	StDev	Max	Mean	StDev	Max
NH_4^+	0.05	0.01	0.23	6.45	0.00	0.00	0.01	No data	No data	No data
Urea	0.01	0.00	0.02	0.42	0.01	0.00	0.03	No data	No data	No data
Oxidized N	0.14	0.01	0.32	3.68	0.00	0.00	0.00	0.01	0.01	0.12
Total N	0.46	0.28	0.56	10.94	0.13	0.23	2.77	No data	No data	No data
$HPO_4^=$	0.05	0.01	0.11	1.48	0.00	0.00	0.01	0.01	0.01	0.17
Total P	0.08	0.03	0.14	1.95	0.01	0.00	0.01	No data	No data	No data

Spatial Patterns

Spatial patterns for mean and maximum values for each analyte are shown in Figure 5.2 through 5.13. In general, data show a pattern of dilution, with the highest concentrations for all analytes occurring in the watershed and becoming diluted as they move into the bay, and further diluted offshore (Figures 5.14 through 5.16). This suggests that the watershed, rather than the near bay environment, is the primary source of nutrients to the system. Measurements of flux values of nutrients were beyond the scope of this study, but if there were strong sources of nutrients being discharged directly into the bay, this clear pattern of dilution shown by the concentration data would not be as well defined. One exception to this pattern of observations was for nitrate/nitrite which had a high maximum value observed near the WWTP. This elevated value occurred on one date in 2011, and appears to be an aberration, i.e. this was the only date during the study period that this occurred. It is not known what caused this spike, but it is possible that there was an event at the WWTP that caused a temporary spike in oxidized nitrogen outflow.

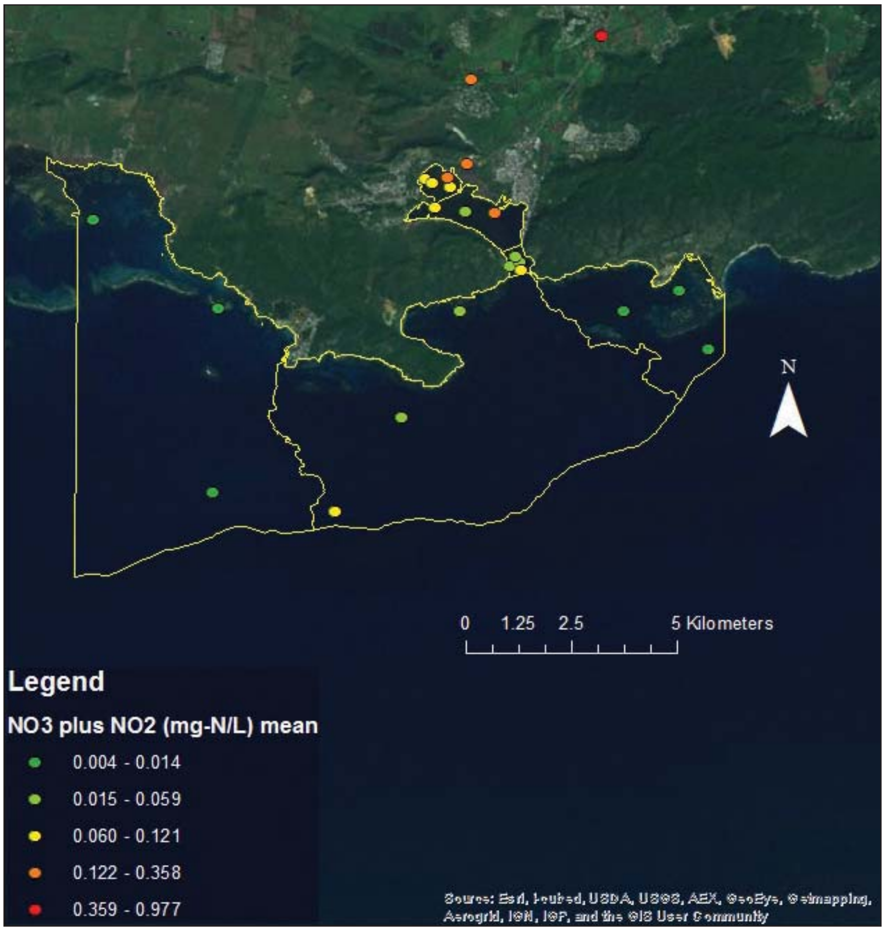

Figure 5.2. Oxidized nitrogen (nitrate plus nitrite as mg-N/L) mean concentrations.

Offshore nutrient patterns do not suggest a strong downstream (east to west) gradient as might be expected, with the prevailing current structure moving along shore from east to west. There were no significant differences among the three offshore strata (Dunn's test, α=0.05).

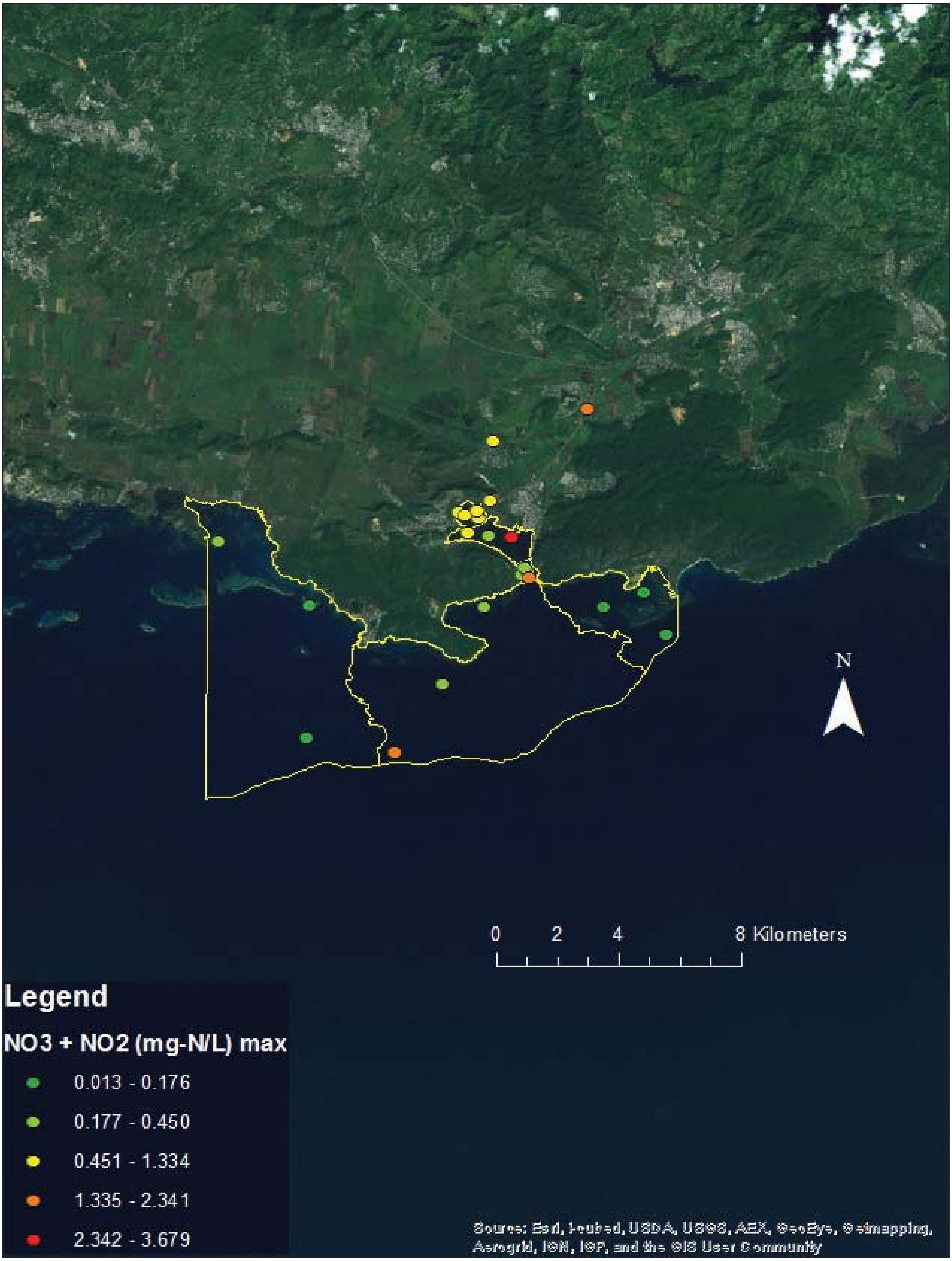

Figure 5.3. Oxidized nitrogen (nitrate plus nitrite as mg-N/L) maximum concentrations.

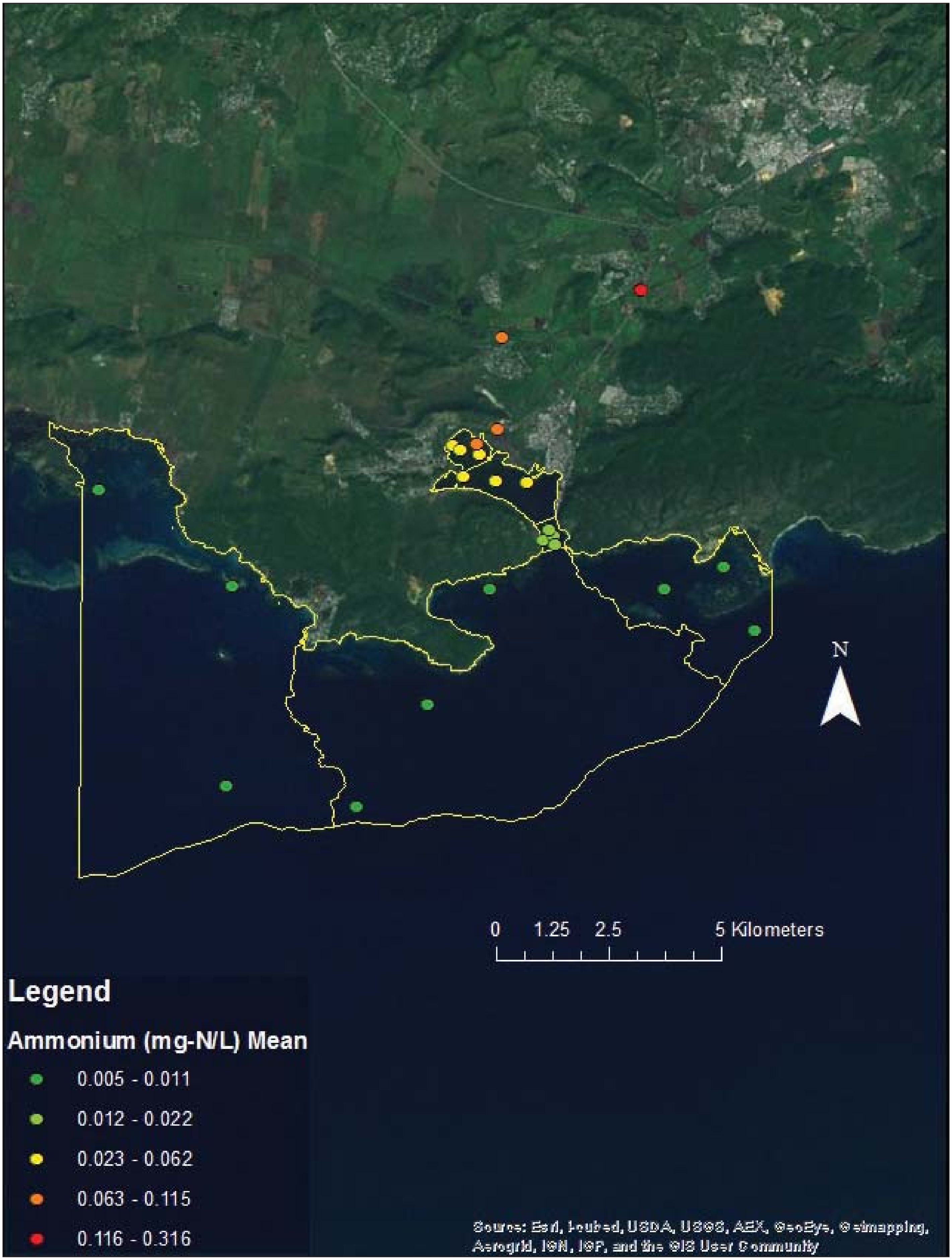

Figure 5.4. Ammonium mean concentrations (as mg-N/L).

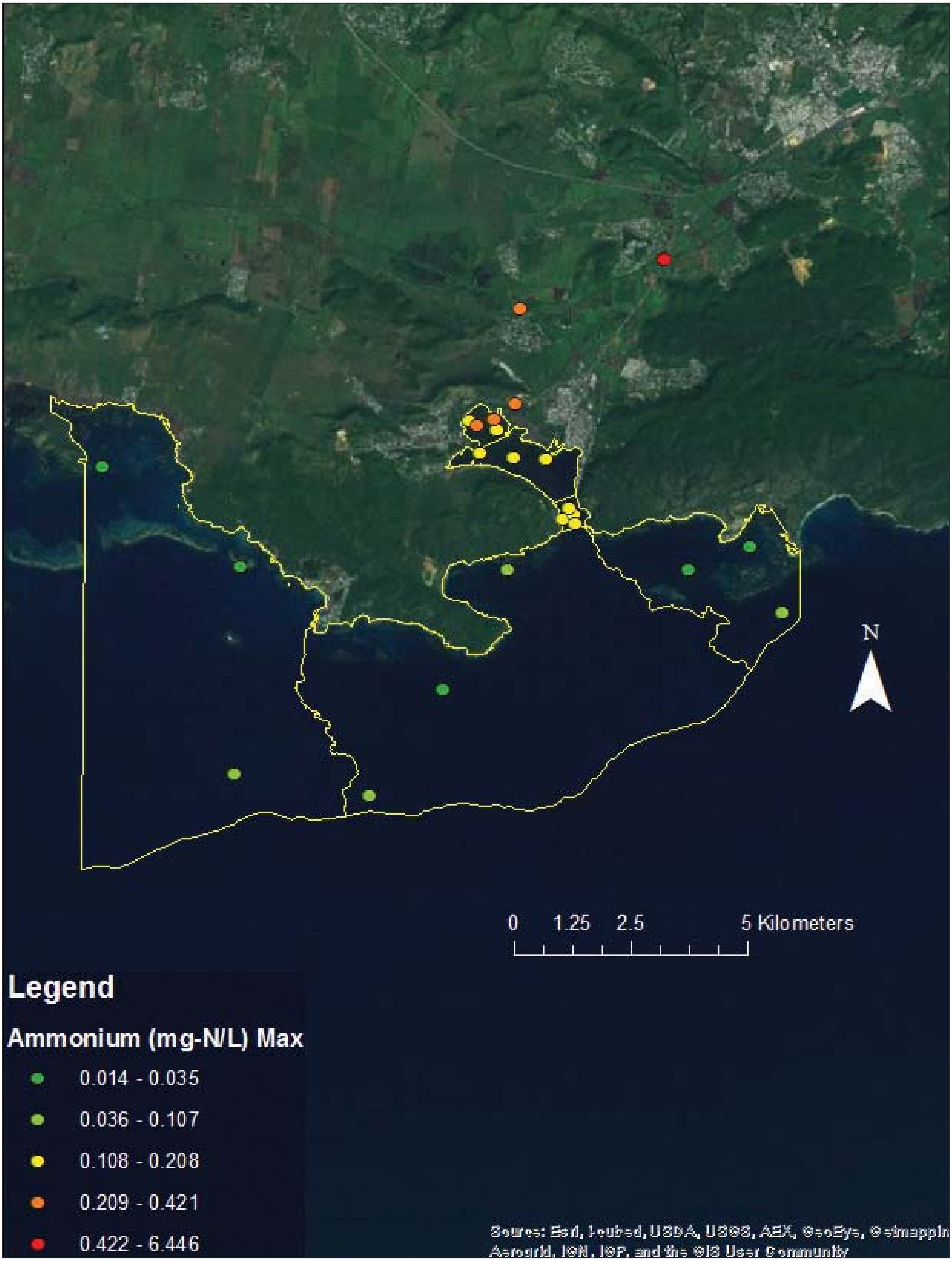

Figure 5.5. Ammonium maximum concentrations (as mg-N/L).

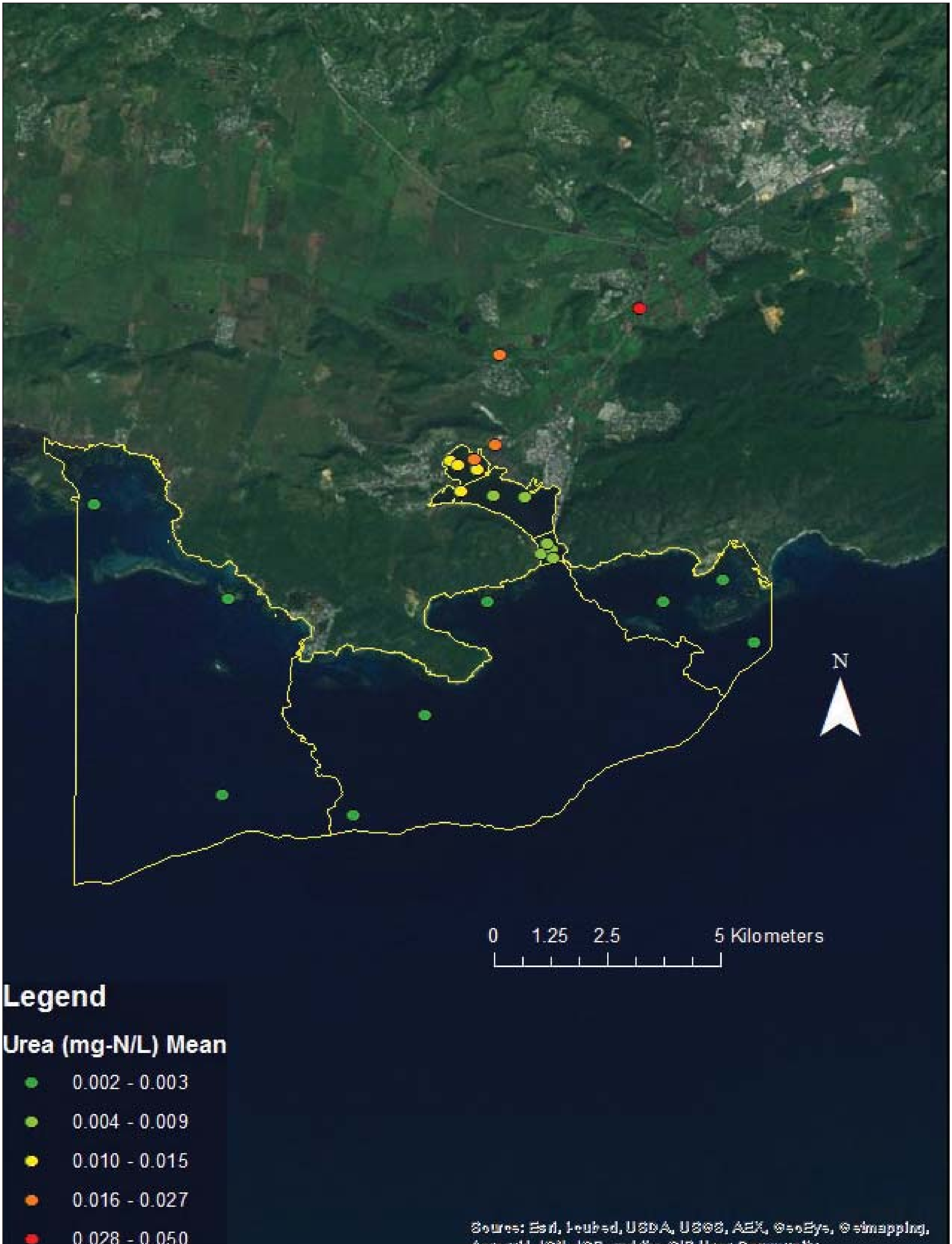

Figure 5.6. Urea mean concentrations (as mg-N/L).

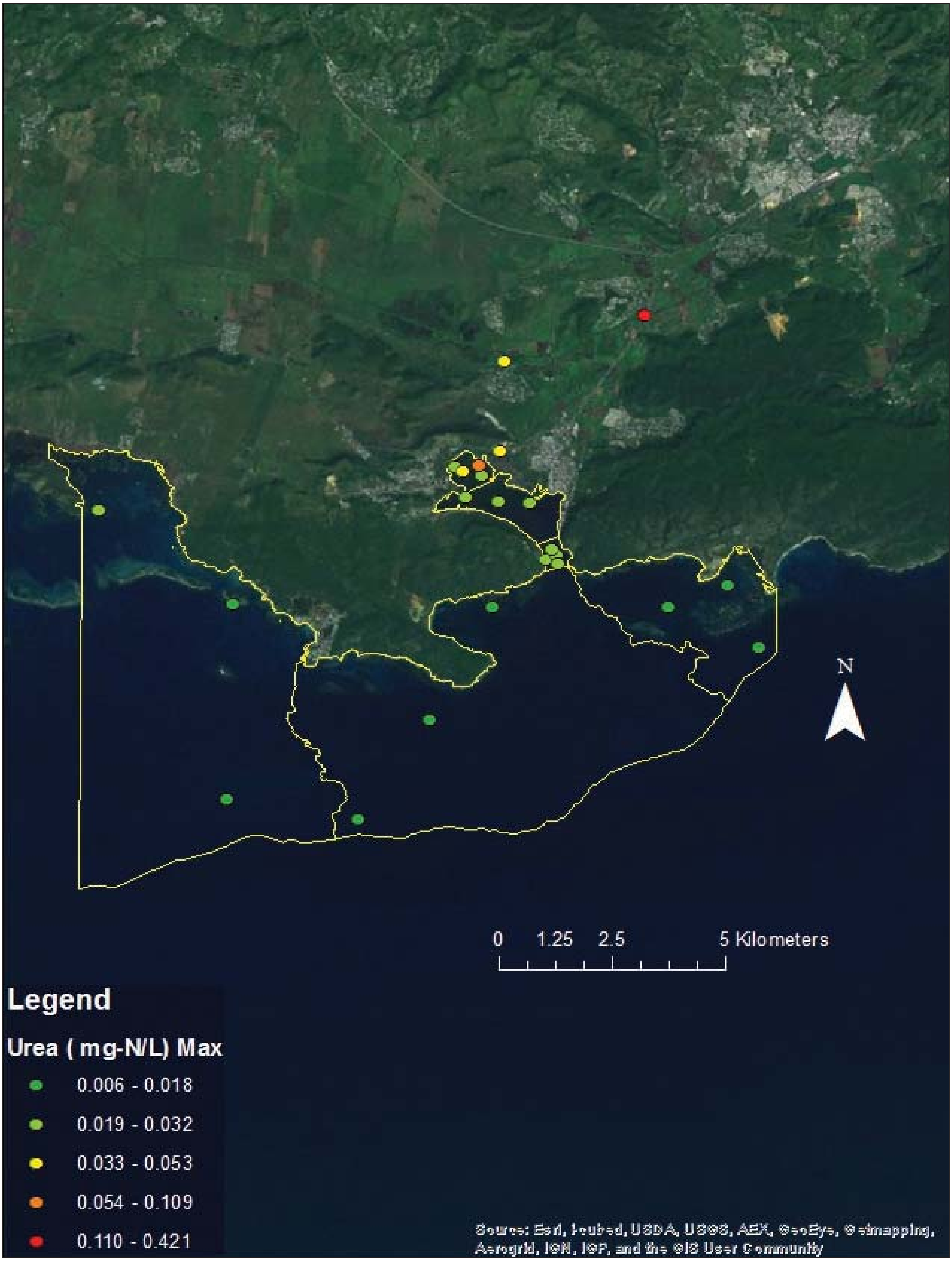

Figure 5.7. Urea maximum concentrations (as mg-N/L).

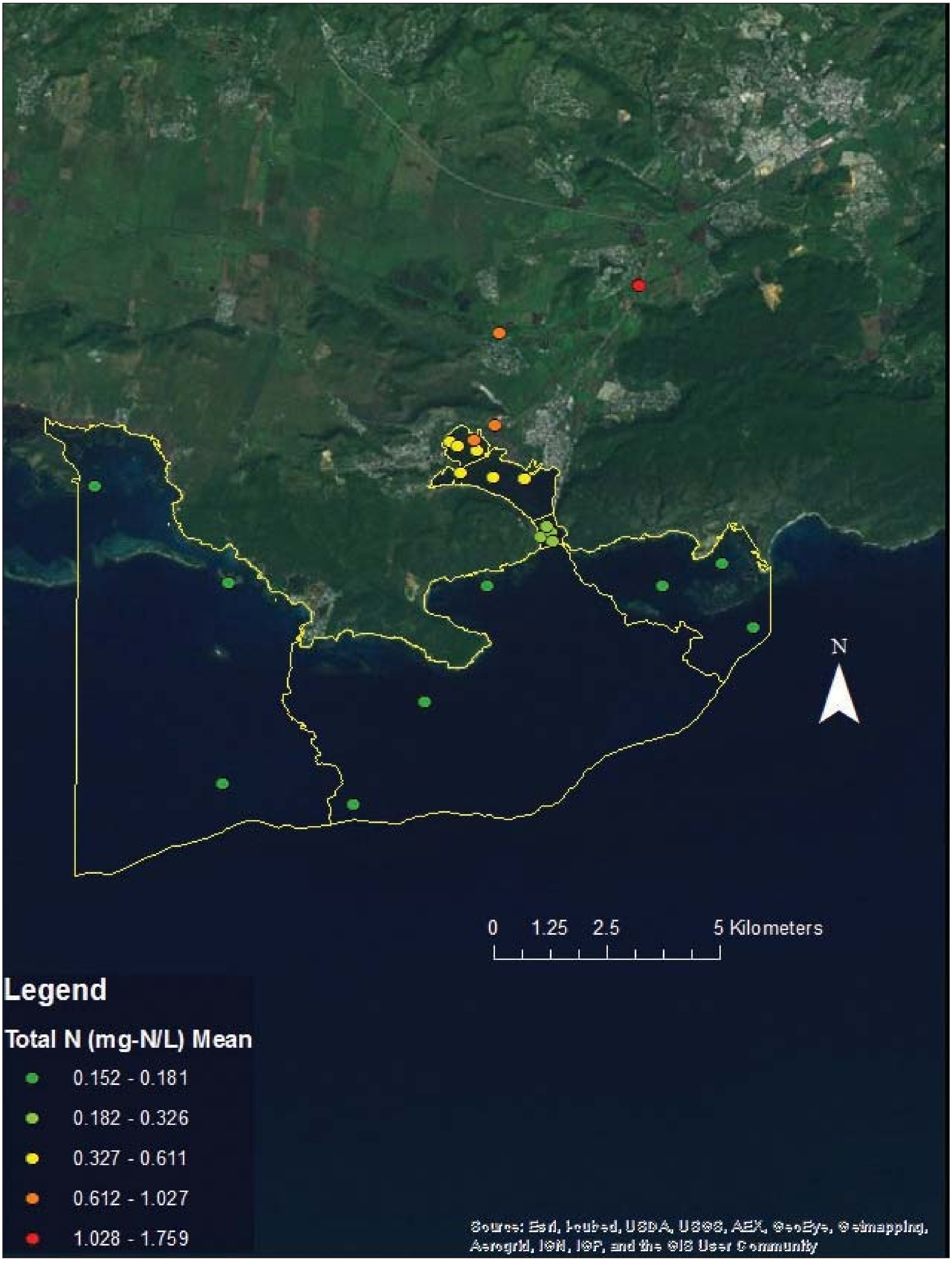

Figure 5.8. Total nitrogen mean concentrations (as mg-N/L).

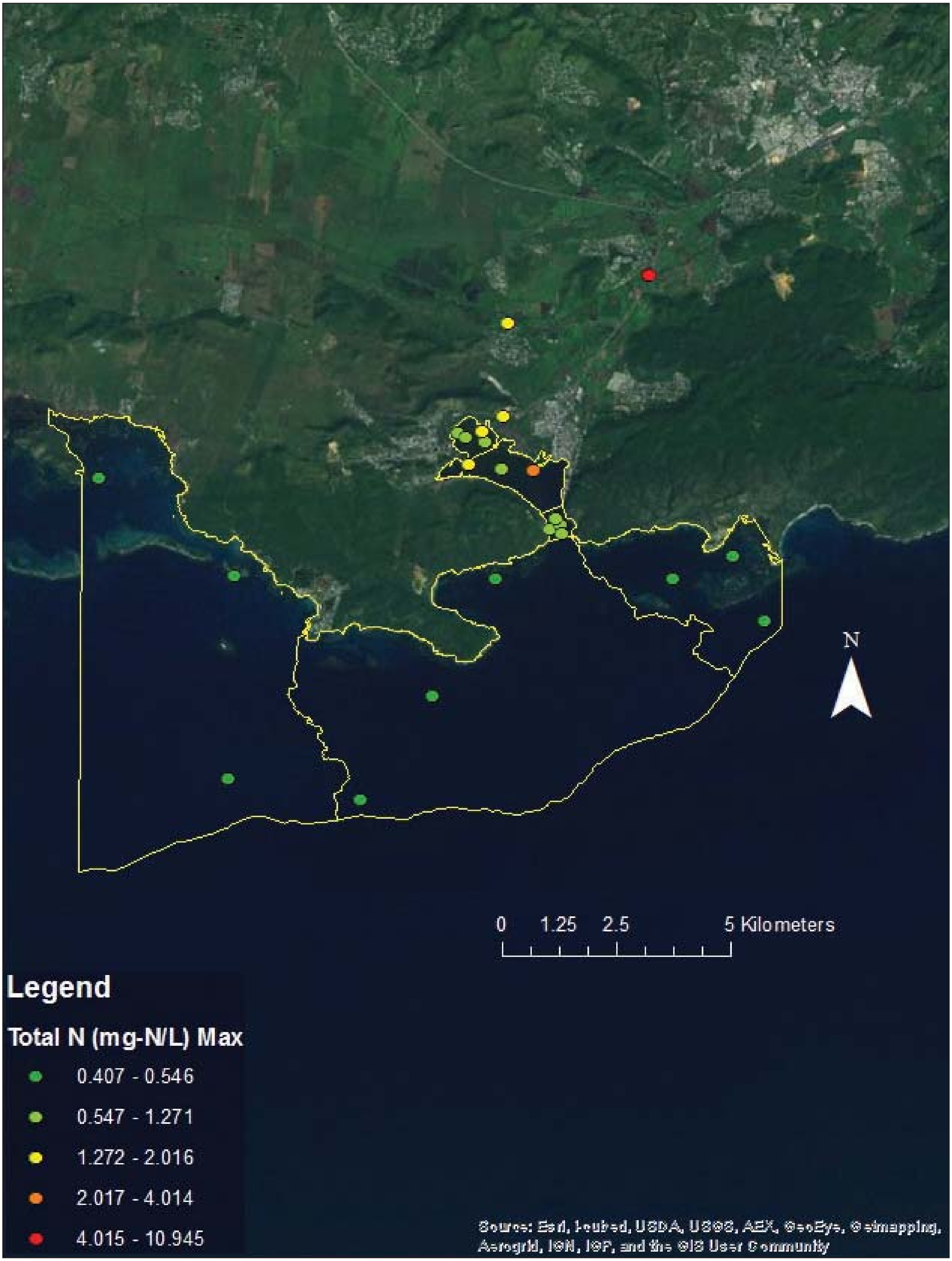

Figure 5.9. Total nitrogen maximum concentrations (as mg-N/L).

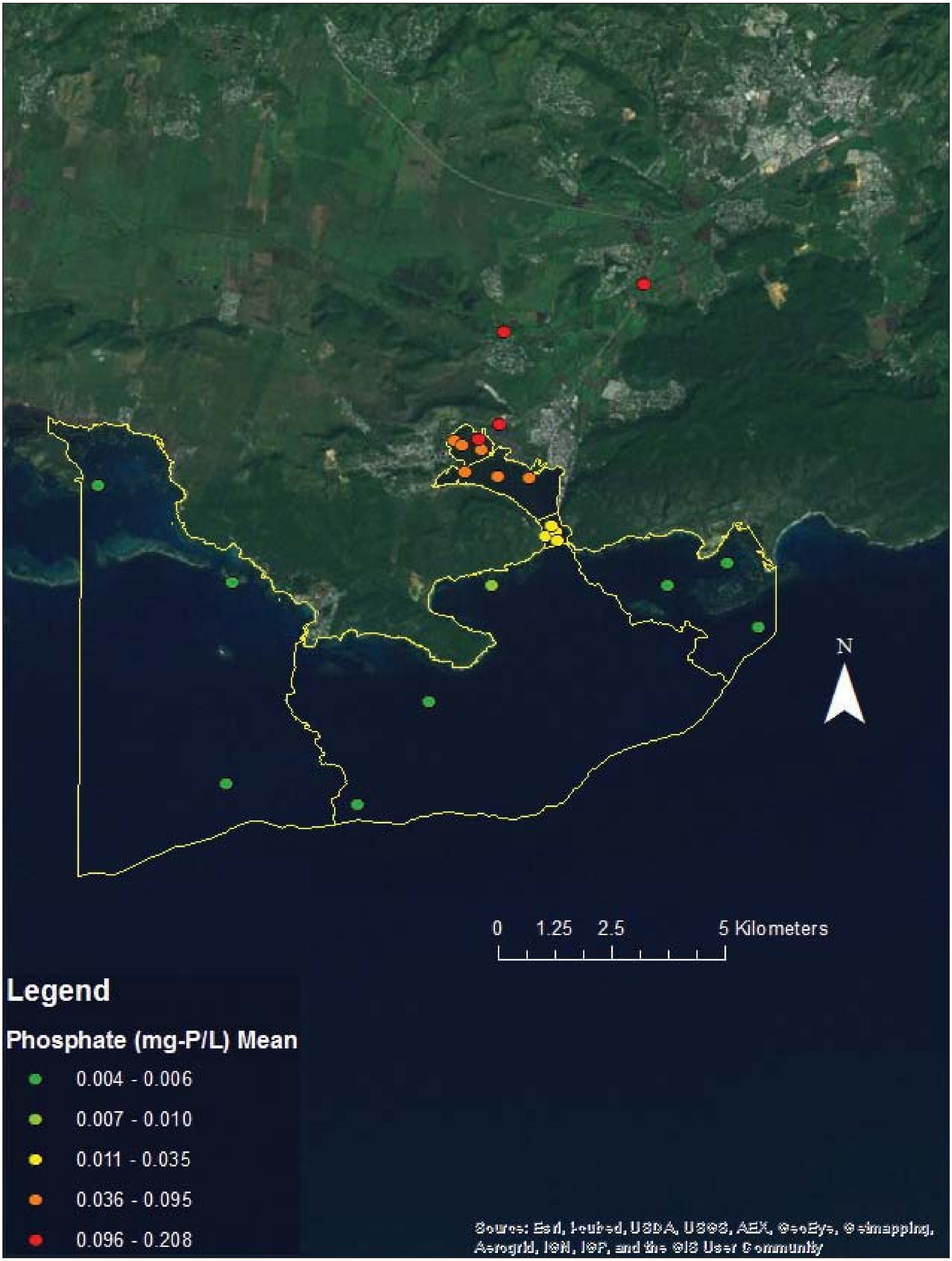

Figure 5.10. Orthophosphate mean concentrations (as mg-P/L).

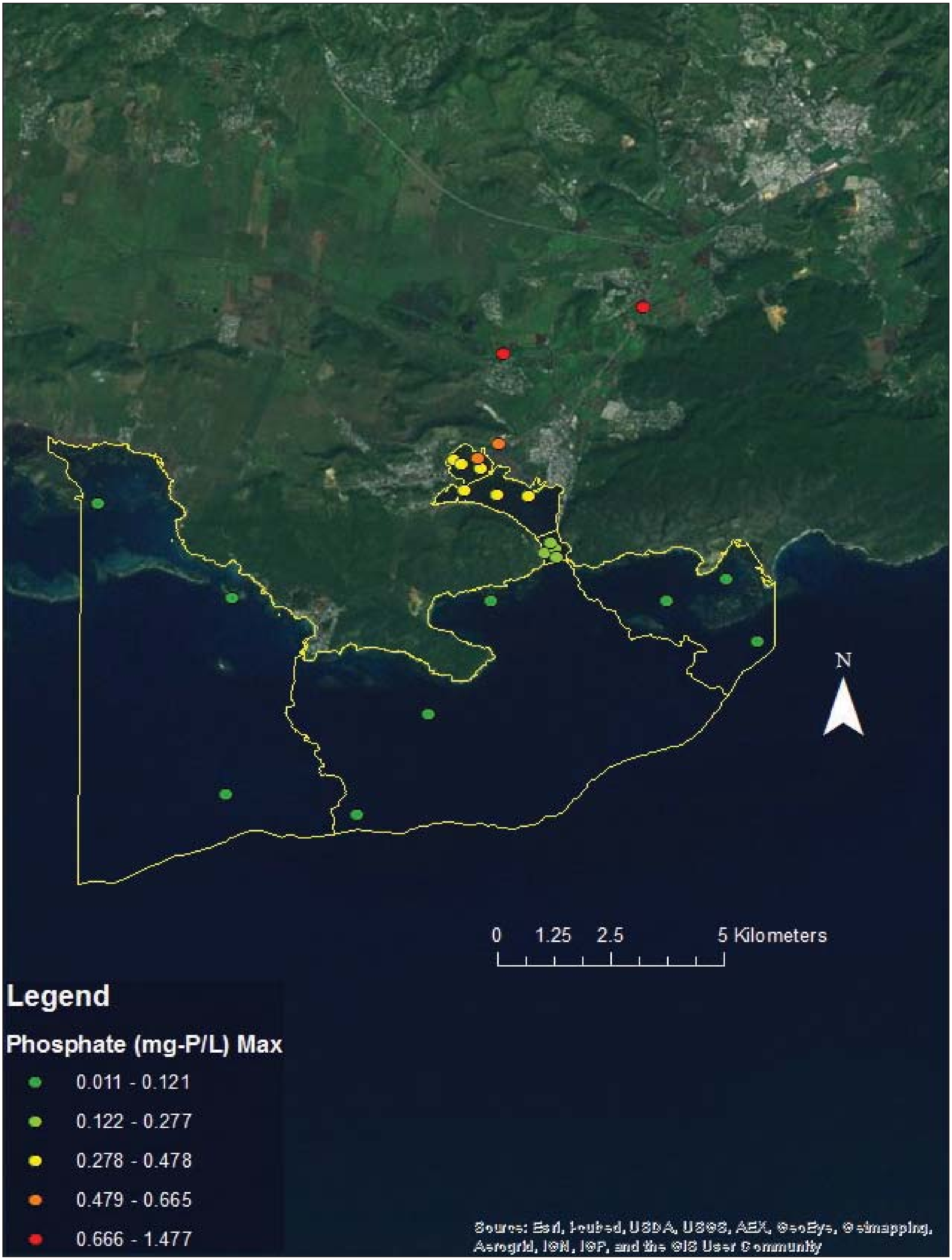

Figure 5.11. Orthophosphate maximum concentrations (as mg-P/L).

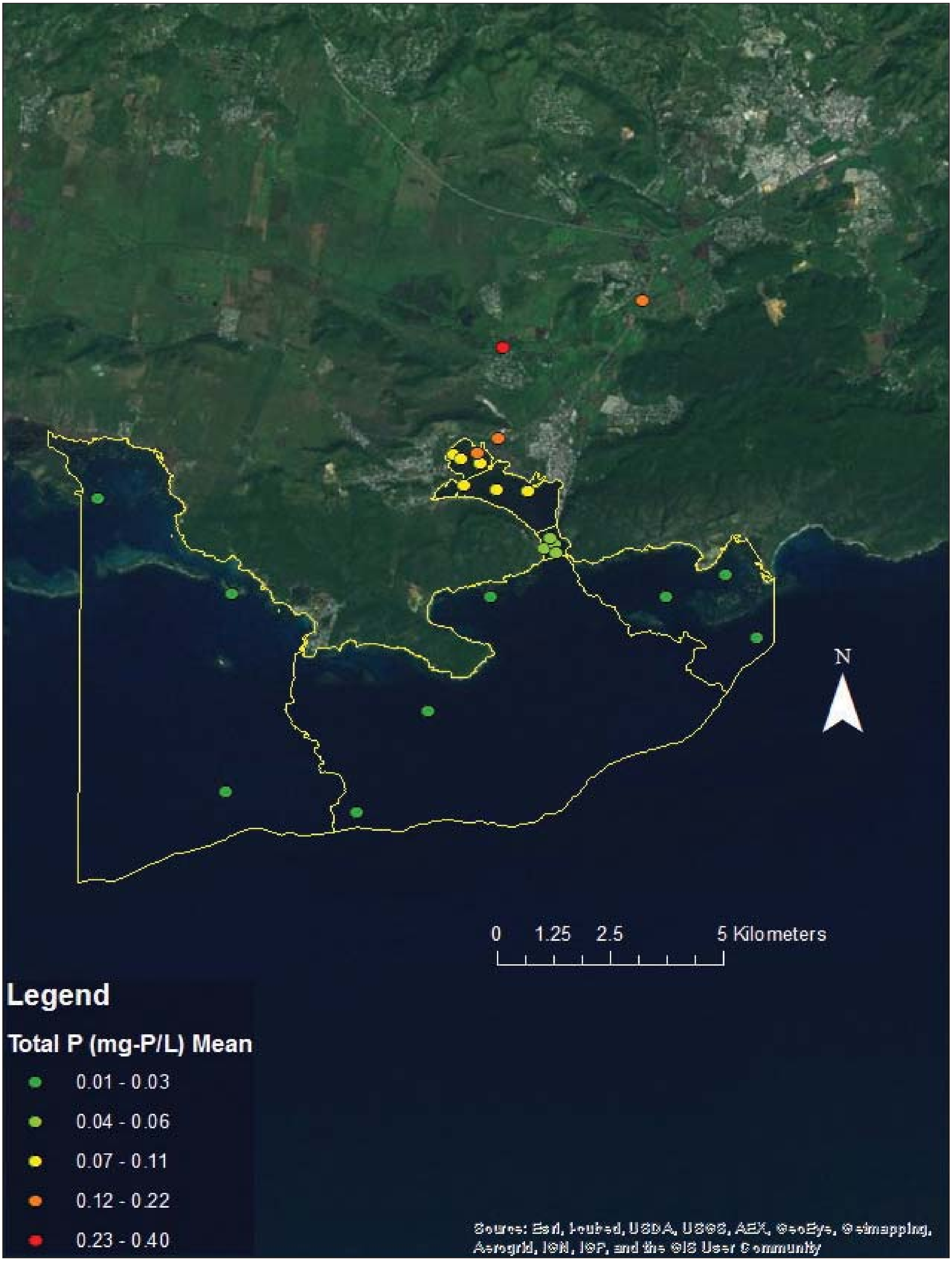

Figure 5.12. Total phosphorus mean concentrations (as mg-P/L).

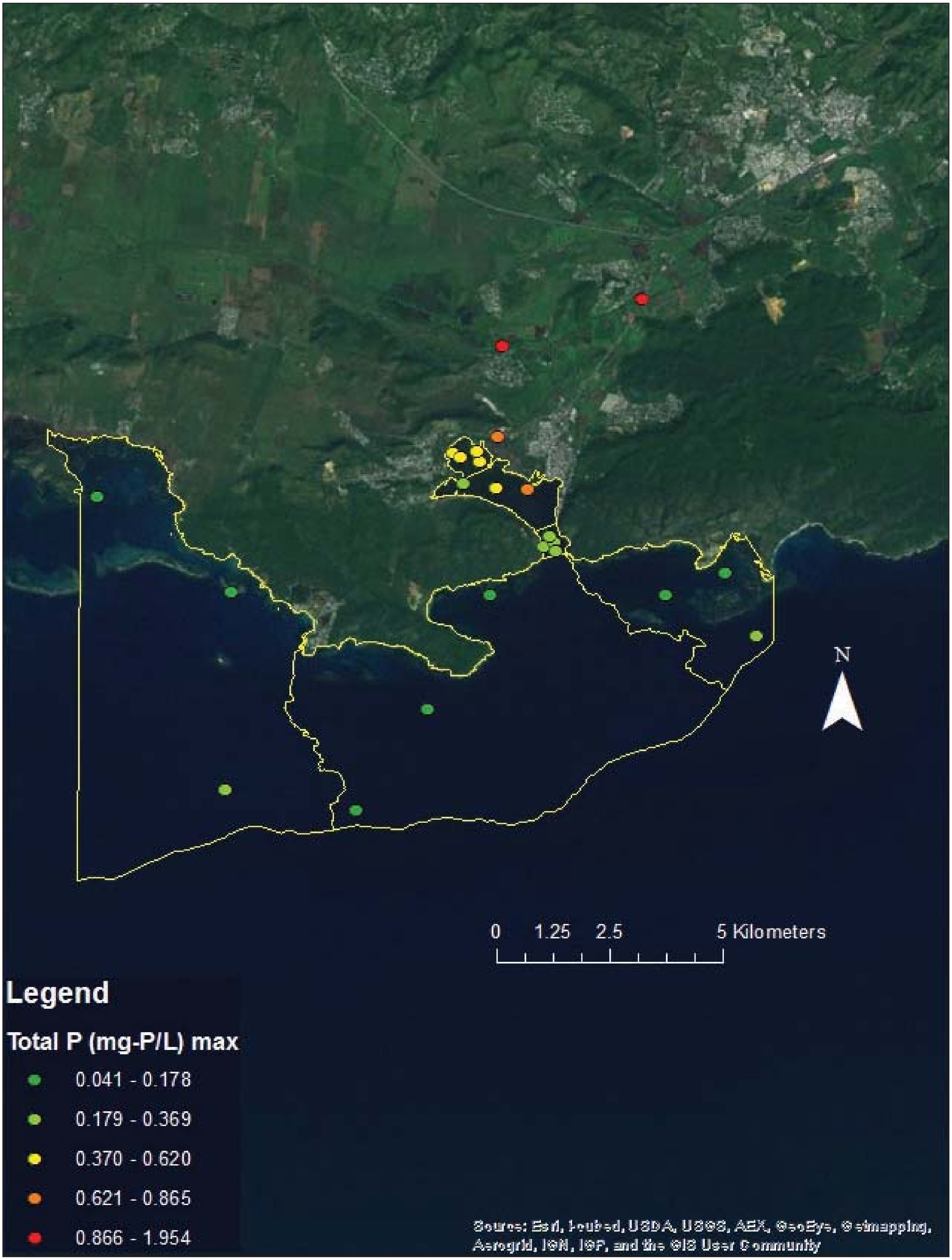

Figure 5.13. Total phosphorus maximum concentrations (as mg-P/L).

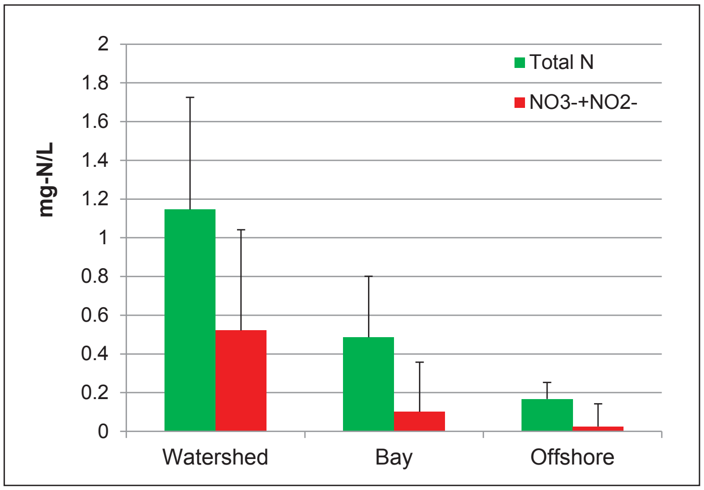

Figure 5.14. Total nitrogen and oxidized nitrogen (nitrate plus nitrite, as mg-N/L) means and standard deviations by geographic area.

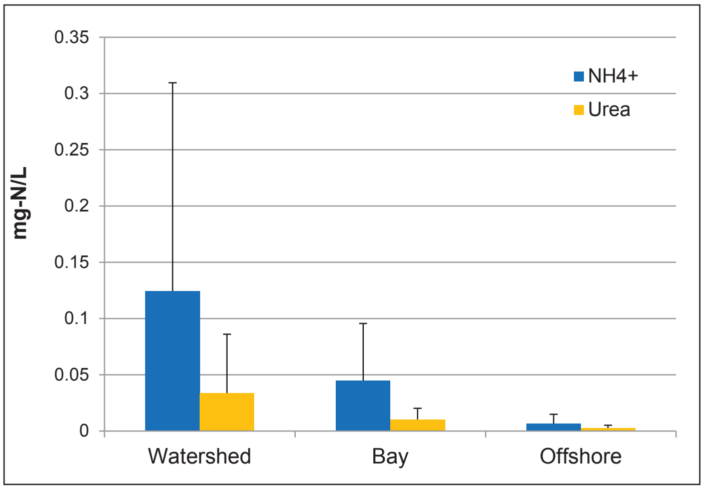

Figure 5.15. Ammonium and urea means and standard deviations (as mg-N/L) by geographic area.

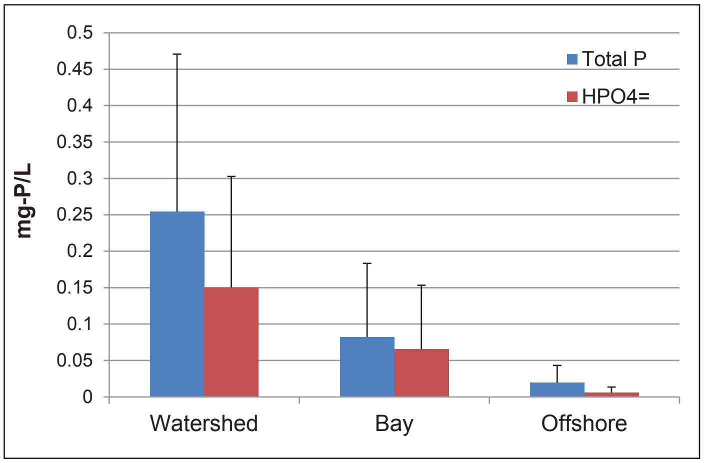

Figure 5.16. Total phosphorus and orthophosphate means and standard deviations (as mg-P/L) by geographic area.

For every analyte except nitrate/nitrite, there were statistically significant differences between all individual bay strata and all individual offshore strata, as well as differences between the North Bay/Central Bay strata and the South Bay. In all cases the northern strata were higher than the southern strata (e.g., Central Bay is greater than South Bay). The majority of these statistical relationships were very strong ($p<0.001$). For nitrate/nitrite differences existed only between the North Bay and the South Bay and offshore strata, with the North Bay being higher. A matrix of statistical comparisons between strata is shown in Table 5.4.

Temporal Patterns

There are statistically significant differences between the sampling years for all analytes. Seasonally, urea, ammonium and orthophosphate were all statistically higher during the rainy season than during the dry season (Wilcoxon test $\alpha=0.05$). This is expected because, while oxidized nitrate percolates quickly to groundwater, urea, ammonia and phosphorus fluxes are driven more by surface water runoff, which is tightly linked to precipitation.

Relationship Between Analytes

Spearman's rank correlation coefficient test showed that, beyond analytes which are autocorrelated (e.g., ammonium and total nitrogen), ammonium, urea and orthophosphorus were significantly ($\alpha=0.05$) and strongly ($\rho>0.70$) correlated with each other. This is not surprising because these nutrients are most strongly associated with runoff, as opposed to oxidized forms of nitrogen that tend to percolate to groundwater.

Relationship with Precipitation

Because riverine flow values were not available, nutrient flux could not be calculated. However, nutrient concentrations may be compared to precipitation, as a proxy for flow. Relating precipitation (and by proxy overland flow and riverine inputs) to stream and marine nutrient data for Guánica is complicated by the large degree to which the hydrology of the system has been altered by humans. In addition to there being a time lag between when rains fall in the upper watershed and when nutrients are delivered to the bay, there are also lags due to the reservoirs in the upper watershed, and the diversion of water from the mainstem of the Rio Loco to the Lajas valley (see further discussion in Chapter 1).

Table 5.4. Results of Dunn Test (α=0.05) comparing nutrient concentrations between strata.

Analyte	Strata	Strata	p Value
HPO4=	Offshore East	Bay North	< 0.001
HPO4=	Offshore West	Bay North	< 0.001
HPO4=	Offshore Central	Bay North	< 0.001
HPO4=	Offshore East	Bay Central	< 0.001
HPO4=	Offshore West	Bay Central	< 0.001
HPO4=	Offshore Central	Bay Central	< 0.001
HPO4=	Offshore East	Bay South	< 0.001
HPO4=	Offshore West	Bay South	< 0.001
HPO4=	Offshore Central	Bay South	< 0.001
HPO4=	Bay South	Bay North	< 0.001
HPO4=	Bay South	Bay Central	0.0028
Total P	Offshore East	Bay North	< 0.001
Total P	Offshore Central	Bay North	< 0.001
Total P	Offshore West	Bay North	< 0.001
Total P	Offshore East	Bay Central	< 0.001
Total P	Offshore Central	Bay Central	< 0.001
Total P	Offshore West	Bay Central	< 0.001
Total P	Offshore East	Bay South	< 0.001
Total P	Offshore Central	Bay South	< 0.001
Total P	Bay South	Bay North	< 0.001
Total P	Offshore West	Bay South	< 0.001
Total P	Bay South	Bay Central	0.0010
NH4+	Offshore West	Bay North	< 0.001
NH4+	Offshore Central	Bay North	< 0.001
NH4+	Offshore East	Bay North	< 0.001
NH4+	Offshore West	Bay Central	< 0.001
NH4+	Offshore Central	Bay Central	< 0.001
NH4+	Offshore East	Bay Central	< 0.001
NH4+	Bay South	Bay North	< 0.001
NH4+	Offshore West	Bay South	< 0.001
NH4+	Offshore Central	Bay South	< 0.001
NH4+	Offshore East	Bay South	< 0.001
NH4+	Bay South	Bay Central	< 0.001
NO3-+NO2-	Offshore West	Bay North	< 0.001
NO3-+NO2-	Offshore East	Bay North	0.0013
NO3-+NO2-	Bay South	Bay North	0.0019
NO3-+NO2-	Offshore Central	Bay North	0.0020
Total N	Offshore Central	Bay North	< 0.001
Total N	Offshore East	Bay North	< 0.001
Total N	Offshore West	Bay North	< 0.001
Total N	Offshore Central	Bay Central	< 0.001
Total N	Offshore East	Bay Central	< 0.001
Total N	Offshore West	Bay Central	< 0.001
Total N	Offshore Central	Bay South	< 0.001
Total N	Bay South	Bay North	< 0.001

Table 5.4 (continued). Results of Dunn Test (α=0.05) comparing nutrient concentrations between strata.

Analyte	Strata	Strata	p Value
Total N	Offshore East	Bay South	< 0.001
Total N	Offshore West	Bay South	< 0.001
Total N	Bay South	Bay Central	< 0.001
Urea	Offshore Central	Bay North	< 0.001
Urea	Offshore East	Bay North	< 0.001
Urea	Offshore West	Bay North	< 0.001
Urea	Offshore Central	Bay Central	< 0.001
Urea	Offshore East	Bay Central	< 0.001
Urea	Offshore West	Bay Central	< 0.001
Urea	Bay South	Bay North	< 0.001
Urea	Offshore Central	Bay South	< 0.001
Urea	Bay South	Bay Central	0.0016
Urea	Offshore East	Bay South	0.0120
Urea	Offshore West	Bay South	0.0139
Urea	Bay North	Bay Central	0.0312

In order to minimize the confounding nature of the altered hydrology, a coastal rain gauge was desirable for comparison with nutrient concentrations. The coastal rain gauge with the most complete data set during the study period was located at the University of Puerto Rico Marine Lab in Lajas (UPR, 2013). A suite of time series graphs are presented in Appendix C. A representative graph for each analyte from each stratum is presented (42 graphs total). In general, the coastal precipitation values track relatively well with nutrient concentrations. While not all large precipitation events result in a nutrient spike, it is rare to see a nutrient spike without a corresponding precipitation event. One notable example of such an exception occurred on 17 March 2011, when an anomalously high nitrate+nitrite value was observed at site N23, which is located near the WWTP (Appendix C, Figure C.25). This concentration was an order of magnitude higher than was what typically measured at that site, and while it is unclear what caused this spike, it is possible that it is related to some sort of incident at the WWTP. This was the only sampling date that demonstrated a high value at this site. Event based sampling (i.e. storm flow sampling) in future studies would further clarify the relationship between precipitation events and nutrient concentrations.

Impact to Coral Reefs

Lapointe (1997) previously proposed nutrient threshold values for coral reefs. This previous work postulated that for reefs in Florida, 0.014 mg/L inorganic nitrogen (DIN) and 0.031 mg/L soluble reactive phosphorus (SRP) were the thresholds above which coral reefs are impacted adversely. In the offshore waters, where coral reefs currently exist, inorganic phosphorus values exceeded this threshold on two sampling dates: 10 July 2009, at sites N12 and N27, and on 9 September 2009 at site N29. This suggests that coral reefs in this system are rarely exposed to SRP values that would be indicative of problems with macroalgae. DIN, on the other hand, exceeded the proposed threshold at the offshore sites for approximately 10% of the data points. There were 88 data points (out of 878) with a DIN value of greater than 0.014 mg/L. Every offshore site had at least one sampling date with a DIN concentration greater than 0.014 mg/L. Sites in the central stratum had more frequent exceedances, with site N27 having the most frequent (18 times) high DIN. A similar number of average monthly exceedances occurred during the wet and the dry seasons. This may be due to a source which does not vary much through time (e.g., groundwater discharge) or may be confounded by other seasonal forcing factors (e.g., physical oceanography). Regardless, these high DIN values suggest that corals may be at risk from high nutrient levels.

Implications for Watershed Management

The observed patterns in nutrient concentrations suggest that lower watershed point sources (i.e., the WWTP) are not the dominant source of nutrients to the system. This is not to say that the WWTP isn't an important source of nutrients, but it needs to be considered along with sources of nitrogen from farther up in the watershed. Agricultural fertilizer use and houses with septic systems are likely other major sources of nutrients. Nitrogen may also be introduced into the system by natural biological nitrogen fixation or by anthropogenically enhanced atmospheric nitrogen deposition. Because nutrients, especially nitrogen, are multi-source pollutants, a comprehensive nutrient management strategy that targets multiple sources of nutrient inputs is required. Additional data, including additional spatial coverage and flow measurements, would shed additional light on nutrient dynamics in this system.

CONCLUSIONS

These data demonstrate that nutrients are reaching the bay via surface runoff and/or groundwater discharge. These pathways of transport may be of concern to nutrient sensitive offshore coral reef ecosystems.

These data make up part of a larger baseline assessment of this system (see Chapters 2, 3 and 4 of this report), which will be used to evaluate the planning for and efficacy of implementation of watershed best management practices. The water quality parameters discussed here will likely respond very quickly to changes in the watershed, making them especially useful for documenting short term changes to the system.

REFERENCES

Armstrong, F. A. J., C. R. Stearns and J. D. H. Strickland. 1967. The measurement of upwelling and subsequent biological processes by means of the Technicon AutoAnalyzer™ and associated equipment. Deep-Sea Res. 14(3): 381-389.

Bell, P.R.F. 1992. Eutrophication and coral reefs: Some examples in the Great Barrier Reef lagoon. Water Research 26: 553-568.

Bernhardt, H. and A. Wilhelms. 1967. The continuous determination of low level iron, soluble phosphate and total phosphate with the Autoanalyzer. Technicon Symposium.

Bricker, S., B. Longstaff, W. Dennison, A. Jones, K. Boicourt, C. Wicks and J. Woerner. 2007. Effects of Nutrient Enrichment in the Nation's Estuaries: A Decade of Change, National Estuarine Eutrophication Assessment Update. NOAA Coastal Ocean Program Decision Analysis Series No. 26. National Centers for Coastal Ocean Science, Silver Spring, MD. 322 pp.

Galloway J.N., J.D. Aber, J.W. Erisman, S.P. Seitzinger, R.H. Howarth, E.B. Cowling and B.J. Cosby. 2003. The nitrogen cascade. BioScience 53: 341-356.

Hansen, H.P. and F. Koroleff. 1999. Determination of nutrients. In:Grasshoff, K., K. Kremling, M. Ehrhardt (eds.). Methods of Seawater Analysis, 3rd ed. Wiley-VCH, Weinheim, ISBN:3-527-29589-5pp. 159– 228.

Harrison, P.L. and S. Ward. 2001. Elevated levels of nitrogen and phosphorus reduce fertilisation success of gametes from scleractinian reef corals. Marine Biology 139: 1057-1068.

Harwood, J. E. and A. L. Kuhn (1970). "A colorimetric method for ammonia in natural waters." Water Research 4: 805 - 811.

Lapointe, B.E. 1997. Nutrient thresholds for bottom-up control of macroalgal blooms on coral reefs in Jamaica and southeast Florida. Limnol. Ocean 42: 1119-1131.

Marubini, F. and P.S. Davies. 1996. Nitrate increases zooxanthellae population density and reduces skeletogenesis in corals. Marine Biology 127: 319-328.

Mathews, L.G, F.R. Homans and K.W. Easter. 2002. Estimating the benefits of phosphorus pollution reductions: An application in the Minnesota River. Journal of the American Water Resources Association 38: 1217-1223.

McCook, L.J. 1999. Macroalgae, nutrients and phase shifts on coral reefs: scientific issues and management consequences for the Great Barrier Reef. Coral Reefs 18: 357-367.

University of Puerto Rico. 2013. Bio-optical Oceangraphy Laboratory Weather Dataset. http://bio-optics.uprm.edu/weather.html Accessed 8/19/2013.

Whitall, D.R., A.L. Mason, A.S. Pait, V Ransibrahmanakul and J.D. Christensen. 2010. Characterization of Spatial and Temporal Nutrient Dynamics. pp. 151-168. In: Bauer and Kendall (eds.), An Ecological Characterization of the Marine Resources of Vieques, Puerto Rico Part II: Field Studies of Habitats, Nutrients, Contaminants, Fish, and Benthic Communities. NOAA Technical Memorandum NOS NCCOS 110. Silver Spring, MD. 174 pp.

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.

Chapter 6: Conclusions

David Whitall[1] and Laurie Bauer[1,2]

This study represents an interdisciplinary data set that enhances our understanding of the ecosystems in and around Guánica Bay, Puerto Rico. These data will serve as a useful baseline against which to measure future change. Change in a system, including assessment of management efficacy, cannot be measured without this type of baseline study. Different environmental parameters have the potential to change at different rates. For example, water quality may change in response to management activities rather quickly (on the order of weeks to months), whereas, metrics of coral reef health (e.g., percent live coral) may respond much more slowly (years to decades). Future monitoring and assessments need to take this into account in order to adequately track change.

While coral cover was generally low across the entire study region, there is some evidence that the Guánica study region is more degraded that the La Parguera region to the west. Percent cover of hard corals, gorgonians and seagrass were higher in the La Parguera study area compared to Guánica. Fish community metrics were generally similar between the two regions, although there were some differences in species and family composition. It is not well known how sediment, nutrients and toxins are affecting coral and fish communities, and understanding these cause and effect relationships are critical to effective management and restoration practices. It is possible that distance from Guánica Bay, which is fairly polluted, may partially explain these differences.

Numerous analytes (total chlordane, total DDT, total PAHs, total PCBs, arsenic, chromium, copper, mercury, nickel and zinc) were measured at levels in sediments that suggest possible toxicity to benthic infauna. Several analytes (PCBs, nickel, chromium, chlordane) were extremely high compared to values measured elsewhere in the U.S. Many of these pollutants were detected in coral tissues, although at lower concentrations than observed in the sediments. Because contaminant threshold values do not exist for coral, it is unclear what effect the observed contaminant levels might have on coral health. Although the highest level of contamination was observed inside the bay (i.e., away from the reefs), the mangrove areas around the bay are important nursery habitats for a variety of fish species. Future studies should consider fish tissue contaminants to assess whether there is an ecological or seafood safety issue related to contaminants in the bay.

Sediment trap accumulation rates are spatiotemporally variable, with accumulated sediment composition (i.e., terrigenous versus carbonate) being relatively uniform. Sediment re-suspension is likely very important in this system and while Guánica Bay is likely an important sediment source, other up current sources may be important as well.

Surface water nutrient values measured in the study area were similar to values observed elsewhere in the coastal waters of Puerto Rico. In general, nutrient concentrations were highest in the watershed, and concentrations decreased with dilution into the bay and offshore. Nutrient concentrations track fairly well with precipitation patterns, with higher phosphorus, ammonium and urea concentrations during the wet season. In offshore waters, phosphorus rarely exceeded previously proposed coral health thresholds; whereas, nitrogen exceeded thresholds 10% of the time.

Although there is no singular stressor or response variable that captures the relationship between pollution and ecosystem health for this system, the preponderance of evidence presented in this report suggests that this system is experiencing anthropogenic stress, which may be resulting in coral decline.

This suite of environmental data (biological and stressors) represent an important baseline against which to measure future change, such as improvements due to watershed restoration, or degradation due to further development in the area. Further monitoring and assessments are needed in order to detect changes in the ecosystem over a variety of time scales ranging from relatively short-term responses in sediment loading, to potentially decadal-long recovery processes for reef systems.

1. NOAA/NOS/NCCOS COAST
2. CSS-Dynamac, Fairfax, VA

Appendix A

Table A.1. Mean species site frequency, density, and biomass for fish species observed within the Guánica study area in the 2010 survey.

Species	Common name	Family	Trophic Group	% of Surveys	Mean Density (SE)	Mean Biomass (SE)
Abudefduf saxatilis	Sergeant major	Pomacentridae	I	11%	0.27 (0.15)	10.37 (6.17)
Acanthurus bahianus	Ocean surgeonfish	Acanthuridae	H	39%	3.08 (0.48)	122.86 (29.85)
Acanthurus chirurgus	Doctorfish	Acanthuridae	H	11%	0.49 (0.24)	19.93 (8.09)
Acanthurus coeruleus	Blue tang	Acanthuridae	H	30%	1 (0.29)	125.76 (45.16)
Anisotremus virginicus	Porkfish	Haemulidae	I	10%	0.28 (0.17)	76.62 (58.44)
Archosargus rhomboidalis	Sea bream	Sparidae	H	7%	0.14 (0.14)	28.29 (28.16)
Aulostomus maculatus	Trumpetfish	Aulostomidae	P	5%	0.05 (0.03)	6.58 (4.59)
Balistes vetula	Queen triggerfish	Balistidae	I	13%	0.24 (0.08)	160.15 (63.17)
Bathygobious soporator	Frillfin goby	Gobiidae	I	2%	0.00 (0.00)	0.00 (0.00)
Belonidae sp.	Needlefish family sp.	Belonidae	P	5%	0.00 (0.00)	0.00 (0.00)
Bodianus rufus	Spanish hogfish	Labridae	I	7%	0.1 (0.05)	11.22 (6.79)
Calamus calamus	Saucereye porgy	Sparidae	I	10%	0.12 (0.05)	17.36 (7.27)
Cantherhines macrocerus	American whitespotted filefish	Monacanthidae	I	2%	0.02 (0.02)	1.85 (1.86)
Cantherhines pullus	Orangespotted filefish	Monacanthidae	I	3%	0.04 (0.02)	1.65 (1.15)
Canthigaster rostrata	Sharpnose puffer	Tetraodontidae	I	20%	0.33 (0.09)	2.44 (0.86)
Carangoides ruber	Bar jack	Carangidae	P	11%	1.04 (0.82)	18.26 (14.06)
Caranx crysos	Blue runner	Carangidae	P	5%	0.29 (0.2)	104.37 (75.97)
Caranx latus	Horse-eye jack	Carangidae	P	2%	0.00 (0.00)	0.44 (0.44)
Caranx sp.	Jack sp.	Carangidae	P	2%	0.1 (0.1)	597.46 (598.23)
Cephalopholis cruentata	Graysby	Serranidae	P	11%	0.14 (0.06)	8.52 (4.96)
Cephalopholis fulva	Coney	Serranidae	I	15%	0.41 (0.12)	37.02 (12.1)
Chaetodon capistratus	Foureye butterflyfish	Chaetodontidae	I	44%	1.14 (0.21)	25.48 (5.23)
Chaetodon ocellatus	Spotfin butterflyfish	Chaetodontidae	I	3%	0.05 (0.04)	4.69 (3.31)
Chaetodon sedentarius	Reef butterflyfish	Chaetodontidae	I	2%	0.02 (0.02)	0.07 (0.07)
Chaetodon striatus	Banded butterflyfish	Chaetodontidae	I	25%	0.55 (0.11)	29.66 (8.9)
Chromis cyanea	Blue chromis	Pomacentridae	PL	8%	1.58 (0.76)	4.16 (2.07)
Chromis multilineata	Brown chromis	Pomacentridae	I	2%	0.07 (0.07)	0.33 (0.33)
Clupeidae sp.	Herring family sp.	Clupeidae	PL	7%	0.24 (0.12)	0.01 (0.01)
Coryphopterus dicrus	Colon goby	Gobiidae	I	3%	0.18 (0.15)	0.23 (0.17)
Coryphopterus glaucofraenum	Bridled goby	Gobiidae	I	18%	0.74 (0.32)	1.65 (0.66)
Cosmocampus elucens	Shortfin pipefish	Syngnathidae	PL	3%	0.04 (0.03)	0.15 (0.11)
Cryptotomus roseus	Bluelip parrotfish	Scaridae	H	8%	0.31 (0.19)	0.77 (0.41)
Ctenogobius saepepallens	Dash goby	Gobiidae	I	2%	0.02 (0.02)	0.06 (0.06)
Dasyatis americana	Southern stingray	Dasyatidae	I	2%	0.02 (0.02)	137.41 (137.11)
Elacatinus evelynae	Sharknose goby	Gobiidae	I	10%	0.22 (0.10)	0.38 (0.14)

Table A.1 (continued). Mean species site frequency, density, and biomass for fish species observed within the Guánica study area in the 2010 survey.

Species	Common name	Family	Trophic Group	% of Surveys	Mean Density (SE)	Mean Biomass (SE)
Epinephelus adscensionis	Rock hind	Serranidae	I	2%	0.05 (0.05)	0.75 (0.75)
Epinephelus guttatus	Red hind	Serranidae	I	5%	0.05 (0.03)	7.02 (4.58)
Equetus punctatus	Spotted drum	Sciaenidae	I	2%	0.02 (0.02)	0.07 (0.07)
Eucinostomus gula	Silver jenny	Gerreidae	I	8%	0.55 (0.34)	9.6 (5.93)
Eucinostomus melanopterus	Flagfin mojarra	Gerreidae	I	11%	0.33 (0.24)	0.93 (0.76)
Gerres cinereus	Yellowfin mojarra	Gerreidae	I	15%	0.10 (0.07)	0.59 (0.37)
Ginglymostoma cirratum	Nurse shark	Ginglymostoma-tidae	I	2%	0.02 (0.02)	0.05 (0.05)
Gnatholepis thompsoni	Goldspot goby	Gobiidae	H	15%	0.83 (0.62)	1.42 (0.53)
Gramma loreto	Fairy basslet	Grammatidae	I	2%	0.12 (0.12)	0.09 (0.10)
Haemulon aurolineatum	Tomtate	Haemulidae	I	5%	0.42 (0.3)	38.6 (33.48)
Haemulon chrysargyreum	Smallmouth grunt	Haemulidae	I	3%	0.04 (0.03)	0.73 (0.67)
Haemulon flavolineatum	French grunt	Haemulidae	I	39%	1.01 (0.52)	139.02 (104.99)
Haemulon parra	Sailors choice	Haemulidae	I	3%	0.00 (0.00)	0.02 (0.02)
Haemulon plumierii	White grunt	Haemulidae	I	21%	0.28 (0.08)	60.32 (28.19)
Haemulon sciurus	Bluestriped grunt	Haemulidae	I	20%	0.56 (0.29)	145.58 (78.78)
Haemulon sp.	Grunt sp.	Haemulidae	I	11%	3.39 (1.85)	7.21 (5.86)
Halichoeres bivittatus	Slippery dick	Labridae	I	44%	3.44 (0.73)	17.8 (4.2)
Halichoeres cyanocephalus	Yellowcheek wrasse	Labridae	P	2%	0.02 (0.02)	0.15 (0.15)
Halichoeres garnoti	Yellowhead wrasse	Labridae	I	41%	3.27 (0.72)	24.92 (6.05)
Halichoeres maculipinna	Clown wrasse	Labridae	I	16%	0.42 (0.14)	2.87 (1.34)
Halichoeres poeyi	Blackear wrasse	Labridae	I	13%	0.64 (0.3)	2.32 (0.96)
Halichoeres radiatus	Puddingwife	Labridae	I	7%	0.07 (0.03)	0.53 (0.26)
Heteroconger longissimus	Brown garden eel	Congridae	PL	3%	0.57 (0.42)	19.28 (18.75)
Holacanthus ciliaris	Queen angelfish	Pomacanthidae	I	7%	0.07 (0.03)	21.05 (11.18)
Holacanthus tricolor	Rock beauty	Pomacanthidae	I	5%	0.05 (0.03)	2.19 (1.47)
Holocentrus adscensionis	Squirrelfish	Holocentridae	I	23%	0.48 (0.13)	43.75 (16.69)
Holocentrus rufus	Longspine squirrelfish	Holocentridae	I	16%	0.41 (0.15)	33.62 (14.35)
Hypoplectrus chlorurus	Yellowtail hamlet	Serranidae	I	2%	0.02 (0.02)	0.32 (0.32)
Jenkinsia sp.	Herring species	Clupeidae	PL	8%	2.09 (1.50)	0.10 (0.07)
Lachnolaimus maximus	Hogfish	Labridae	I	2%	0.02 (0.02)	2.14 (2.14)
Lactophrys triqueter	Smooth trunkfish	Ostraciidae	I	5%	0.05 (0.03)	5.31 (3.37)
Lutjanus apodus	Schoolmaster	Lutjanidae	P	21%	0.16 (0.08)	7.41 (3.42)
Lutjanus cyanopterus	Cubera snapper	Lutjanidae	P	3%	0.02 (0.02)	46.1 (46.09)
Lutjanus griseus	Gray snapper	Lutjanidae	I	11%	0.00 (0.00)	0.69 (0.33)
Lutjanus jocu	Dog snapper	Lutjanidae	P	2%	0.00 (0.00)	0.01 (0.01)
Lutjanus mahogoni	Mahogany snapper	Lutjanidae	P	5%	0.05 (0.03)	5.26 (3.84)

Table A.1 (continued). Mean species site frequency, density, and biomass for fish species observed within the Guánica study area in the 2010 survey.

Species	Common name	Family	Trophic Group	% of Surveys	Mean Density (SE)	Mean Biomass (SE)
Lutjanus synagris	Lane snapper	Lutjanidae	I	26%	1.28 (0.68)	11.3 (5.52)
Malacanthus plumieri	Sand tilefish	Malacanthidae	I	8%	0.1 (0.05)	28.18 (16.55)
Malacoctenus triangulatus	Saddled blenny	Labrisomidae	I	11%	0.12 (0.04)	0.46 (0.17)
Melichthys niger	Black durgon	Balistidae	H	16%	1.34 (0.78)	543.86 (466.47)
Microgobius carri	Seminole goby	Gobiidae	H	2%	0.03 (0.03)	0.06 (0.06)
Microgobius signatus	Microgobius signatus	Gobiidae	H	2%	0.07 (0.07)	0.09 (0.09)
Microspathodon chrysurus	Yellowtail damselfish	Pomacentridae	H	5%	0.2 (0.15)	6 (3.93)
Monacanthus ciliatus	Fringed filefish	Monacanthidae	H	2%	0.02 (0.02)	0.05 (0.06)
Monacanthus tuckeri	Slender filefish	Monacanthidae	PL	2%	0.02 (0.02)	0.43 (0.43)
Mulloidichthys martinicus	Yellow goatfish	Mullidae	I	3%	0.62 (0.46)	158.05 (112.56)
Myripristis jacobus	Blackbar soldierfish	Holocentridae	I	8%	0.31 (0.17)	40.98 (22.4)
Nes longus	Orangespotted goby	Gobiidae	H	7%	1.37 (0.72)	3.77 (1.9)
Ocyurus chrysurus	Yellowtail snapper	Lutjanidae	PL	26%	0.79 (0.24)	66.77 (29.02)
Ophioblennius macclurei	Redlip blenny	Blenniidae	H	5%	0.27 (0.21)	1 (0.72)
Opistognathus aurifrons	Yellowhead jawfish	Opistognathidae	PL	5%	0.3 (0.2)	1.57 (0.9)
Pareques acuminatus	Highhat	Sciaenidae	I	3%	0.03 (0.02)	1.56 (1.41)
Pomacanthus arcuatus	Gray angelfish	Pomacanthidae	I	7%	0.17 (0.08)	98.32 (49.59)
Pomacanthus paru	French angelfish	Pomacanthidae	I	2%	0.02 (0.02)	0.09 (0.09)
Pseudupeneus maculatus	Spotted goatfish	Mullidae	I	5%	0.05 (0.03)	3.61 (2.26)
Ptereleotris helenae	Hovering goby	Microdesmidae	PL	3%	0.16 (0.14)	0.16 (0.12)
Pterois volitans	Red lionfish	Scorpaenidae	P	2%	0.02 (0.02)	2.27 (2.27)
Rypticus saponaceus	Greater soapfish	Serranidae	I	2%	0.02 (0.02)	1.55 (1.55)
Scarus iseri	Striped parrotfish	Scaridae	H	26%	0.92 (0.29)	15.31 (8.92)
Scarus taeniopterus	Princess parrotfish	Scaridae	H	16%	1.98 (1)	84.02 (36.58)
Scarus vetula	Queen parrotfish	Scaridae	H	2%	0.02 (0.02)	3.73 (3.72)
Scorpaena plumieri	Spotted scorpionfish	Scorpaenidae	I	2%	0.02 (0.02)	2.11 (2.11)
Serranus baldwini	Lantern bass	Serranidae	P	15%	0.49 (0.2)	1.71 (0.82)
Serranus tabacarius	Tobaccofish	Serranidae	P	5%	0.06 (0.03)	0.38 (0.22)
Serranus tigrinus	Harlequin bass	Serranidae	I	21%	1.05 (0.68)	2.77 (0.82)
Serranus tortugarum	Chalk bass	Serranidae	PL	8%	0.79 (0.41)	0.77 (0.36)
Sparisoma atomarium	Greenblotch parrotfish	Scaridae	H	7%	0.09 (0.05)	0.52 (0.29)
Sparisoma aurofrenatum	Redband parrotfish	Scaridae	H	49%	3.1 (0.34)	72.32 (15.26)
Sparisoma radians	Bucktooth parrotfish	Scaridae	H	13%	0.45 (0.24)	0.96 (0.58)
Sparisoma rubripinne	Yellowtail parrotfish	Scaridae	H	3%	0.04 (0.03)	3.77 (2.61)
Sparisoma sp.	Parrotfish sp.	Scaridae	H	3%	0.05 (0.05)	0.42 (0.42)
Sparisoma viride	Stoplight parrotfish	Scaridae	H	28%	0.5 (0.12)	89.48 (32.75)
Sphoeroides testudineus	Checkered puffer	Tetraodontidae	I	3%	0.00 (0.00)	0.06 (0.05)
Sphyraena barracuda	Great barracuda	Sphyraenidae	P	20%	0.07 (0.03)	402.29 (333.89)

Table A.1 (continued). Mean species site frequency, density, and biomass for fish species observed within the Guánica study area in the 2010 survey.

Species	Common name	Family	Trophic Group	% of Surveys	Mean Density (SE)	Mean Biomass (SE)
Stegastes diencaeus	Longfin damsel-fish	Pomacentridae	H	7%	0.02 (0.02)	0.08 (0.06)
Stegastes leucostictus	Beaugregory	Pomacentridae	I	18%	0.18 (0.08)	1.09 (0.5)
Stegastes partitus	Bicolor damselfish	Pomacentridae	H	44%	11.02 (1.45)	32.98 (11)
Stegastes planifrons	Threespot dam-selfish	Pomacentridae	I	7%	0.43 (0.41)	3.73 (3.48)
Stegastes variabilis	Cocoa damselfish	Pomacentridae	H	2%	0.02 (0.02)	0.23 (0.23)
Synodus intermedius	Sand diver	Synodontidae	P	2%	0.02 (0.02)	0.13 (0.13)
Thalassoma bifasciatum	Bluehead	Labridae	I	39%	13.6 (2.44)	22.87 (9.93)
Xyrichtys martinicensis	Rosy razorfish	Labridae	I	5%	0.19 (0.13)	0.74 (0.58)
Xyrichtys novacula	Pearly razorFish	Labridae	I	2%	0.07 (0.07)	0.13 (0.13)
Xyrichtys splendens	Green razorfish	Labridae	I	21%	1.27 (0.47)	4.83 (1.84)

Table A.2. Mean species site frequency, density, and biomass for fish species observed within the La Parguera study area in the August 2008-2010 surveys.

Species	Common name	Family	Trophic Group	% of Surveys	Mean Density (SE)	Mean Biomass (SE)
Abudefduf saxatilis	Sergeant major	Pomacentridae	I	7%	0.02 (0.01)	0.37 (0.29)
Abudefduf taurus	Night sergeant	Pomacentridae	H	1%	0.01 (0.01)	0.24 (0.24)
Acanthemblemaria maria	Secretary blenny	Chaenopsidae	PL	1%	0.01 (0.01)	<0.01 (<0.01)
Acanthemblemaria sp.	Tube Blenny sp.	Chaenopsidae	PL	1%	0.01 (0.01)	<0.01 (<0.01)
Acanthostracion quadricornis	Scrawled cowfish	Ostraciidae	I	1%	0.01 (0.01)	0.04 (0.04)
Acanthurus bahianus	Ocean surgeonfish	Acanthuridae	H	43%	2.48 (0.3)	148.7 (18.29)
Acanthurus chirurgus	Doctorfish	Acanthuridae	H	12%	0.5 (0.14)	23.37 (6.02)
Acanthurus coeruleus	Blue tang	Acanthuridae	H	27%	1.53 (0.53)	87.63 (32.03)
Amblycirrhitus pinos	Redspotted hawkfish	Cirrhitidae	I	1%	0.01 (0.01)	0.01 (0.01)
Anisotremus virginicus	Porkfish	Haemulidae	I	6%	0.04 (0.02)	3.22 (1.78)
Archosargus rhomboidalis	Sea bream	Sparidae	H	5%	<0.01 (<0.01)	0.23 (0.12)
Atherinomorus sp.	Silverside sp.	Atherinidae	H	4%	130.91 (121.98)	34.55 (32.19)
Aulostomus maculatus	Trumpetfish	Aulostomidae	P	3%	0.03 (0.01)	1.78 (0.92)
Balistes vetula	Queen triggerfish	Balistidae	I	12%	0.15 (0.03)	110.02 (26.14)
Bodianus rufus	Spanish hogfish	Labridae	I	1%	0.01 (0.01)	1.07 (1.03)
Bollmannia boqueronensis	White-eye goby	Gobiidae	H	3%	0.07 (0.03)	0.34 (0.15)
Bothus lunatus	Platefish	Bothidae	P	1%	0.01 (0.01)	0.07 (0.07)
Calamus calamus	Saucereye porgy	Sparidae	I	4%	0.06 (0.02)	6.59 (2.68)
Calamus penna	Sheepshead porgy	Sparidae	I	1%	0.01 (0.01)	0.54 (0.55)
Calamus pennatula	Pluma porgy	Sparidae	I	3%	0.04 (0.02)	7.69 (3.78)
Calamus sp.	Porgy sp.	Sparidae	I	1%	0.01 (0.01)	0.65 (0.65)
Cantherhines pullus	Orangespotted filefish	Monacanthidae	I	1%	0.01 (0.01)	0.55 (0.55)
Canthigaster rostrata	Sharpnose puffer	Tetraodontidae	I	27%	0.42 (0.06)	0.76 (0.15)
Carangoides ruber	Bar jack	Carangidae	P	10%	1.05 (0.44)	26.85 (15.97)
Caranx crysos	Blue runner	Carangidae	P	3%	0.44 (0.27)	262.9 (243.54)
Cephalopholis cruentata	Graysby	Serranidae	P	13%	0.19 (0.05)	11.26 (3.49)
Cephalopholis fulva	Coney	Serranidae	I	5%	0.06 (0.02)	7.89 (3.8)
Chaenopsis limbaughi	Yellowface pikeblenny	Chaenopsidae	I	1%	0.01 (0.01)	<0.01 (<0.01)
Chaenopsis sp.	Pikeblenny sp.	Chaenopsidae	I	1%	0.01 (0.01)	0.08 (0.06)
Chaetodon capistratus	Foureye butterflyfish	Chaetodontidae	I	49%	1.45 (0.13)	21.19 (2.02)
Chaetodon ocellatus	Spotfin butterflyfish	Chaetodontidae	I	1%	0.01 (0.01)	0.02 (0.02)
Chaetodon striatus	Banded butterflyfish	Chaetodontidae	I	4%	0.07 (0.03)	1.4 (0.56)
Chromis cyanea	Blue chromis	Pomacentridae	PL	12%	0.95 (0.26)	2.17 (0.57)
Chromis multilineata	Brown chromis	Pomacentridae	I	2%	0.1 (0.08)	0.42 (0.34)
Clepticus parrae	Creole wrasse	Labridae	PL	4%	0.7 (0.34)	62.32 (48.12)
Clupeidae sp.	Clupidae family sp.	Clupeidae	PL	3%	0.63 (0.36)	0.03 (0.02)
Coryphopterus dicrus	Colon goby	Gobiidae	I	10%	0.21 (0.06)	0.32 (0.09)
Coryphopterus eidolon	Pallid goby	Gobiidae	H	2%	0.06 (0.04)	0.08 (0.05)

Table A.2 (continued). Mean species site frequency, density, and biomass for fish species observed within the La Parguera study area in the August 2008-2010 surveys.

Species	Common name	Family	Trophic Group	% of Surveys	Mean Density (SE)	Mean Biomass (SE)
Coryphopterus glaucofraenum	Bridled goby	Gobiidae	I	30%	2.16 (0.52)	1.89 (0.39)
Coryphopterus lipernes	Peppermint goby	Gobiidae	I	3%	0.09 (0.05)	0.09 (0.04)
Coryphopterus personatus/hyalinus	Masked/glass goby	Gobiidae	I	10%	2.17 (0.81)	1.62 (0.56)
Cryptotomus roseus	Bluelip parrotfish	Scaridae	H	13%	0.45 (0.13)	1.58 (0.59)
Ctenogobius saepepallens	Dash goby	Gobiidae	I	3%	0.09 (0.04)	0.07 (0.04)
Decapterus macarellus	Mackerel scad	Carangidae	PL	2%	1.04 (0.62)	22.98 (12.11)
Decapterus sp.	Scad sp.	Carangidae	PL	1%	0.07 (0.07)	1.12 (1.12)
Diplectrum bivittatum	Dwarf sand perch	Serranidae	P	1%	0.01 (0.01)	0.02 (0.02)
Echeneis naucrates	Sharksucker	Echeneidae	PL	1%	0.03 (0.02)	1.79 (1.52)
Elacatinus chancei	Shortstripe goby	Gobiidae	I	1%	0.01 (0.01)	<0.01 (<0.01)
Elacatinus dilepis	Orangesided goby	Gobiidae	I	1%	0.02 (0.02)	0.02 (0.02)
Elacatinus evelynae	Sharknose goby	Gobiidae	I	16%	0.33 (0.06)	0.2 (0.05)
Epinephelus guttatus	Red hind	Serranidae	I	4%	0.05 (0.02)	9.48 (3.94)
Epinephelus striatus	Nassau grouper	Serranidae	P	1%	0.01 (0.01)	5.32 (5.32)
Equetus punctatus	Spotted drum	Sciaenidae	I	1%	0.02 (0.02)	0.1 (0.09)
Eucinostomus gula	Silver jenny	Gerreidae	I	2%	0.02 (0.02)	0.78 (0.66)
Eucinostomus melanopterus	Flagfin mojarra	Gerreidae	I	9%	0.19 (0.16)	0.12 (0.08)
Eucinostomus sp.	Mojarra sp.	Gerreidae	I	1%	0.07 (0.07)	0.1 (0.1)
Euthynnus alletteratus	Little tuny	Scombridae	P	1%	0.03 (0.03)	31.28 (31.35)
Gerres cinereus	Yellowfin mojarra	Gerreidae	I	10%	0.05 (0.02)	0.44 (0.24)
Gnatholepis thompsoni	Goldspot goby	Gobiidae	H	14%	0.73 (0.18)	0.3 (0.07)
Gobiidae sp.	Goby sp.	Gobiidae	I	1%	0.02 (0.02)	0.01 (0.01)
Gramma loreto	Fairy basslet	Grammatidae	I	8%	0.57 (0.19)	0.34 (0.1)
Haemulon aurolineatum	Tomtate	Haemulidae	I	10%	0.51 (0.27)	18.9 (9.06)
Haemulon carbonarium	Caesar grunt	Haemulidae	I	2%	0.07 (0.05)	3.75 (2.82)
Haemulon chrysargyreum	Smallmouth grunt	Haemulidae	I	1%	0.01 (0.01)	0.18 (0.17)
Haemulon flavolineatum	French grunt	Haemulidae	I	29%	0.73 (0.22)	45.4 (12.14)
Haemulon macrostomum	Spanish grunt	Haemulidae	I	1%	0.01 (0.01)	0.02 (0.02)
Haemulon parra	Sailors choice	Haemulidae	I	2%	<0.01 (<0.01)	0.07 (0.05)
Haemulon plumierii	White grunt	Haemulidae	I	15%	0.53 (0.19)	34.05 (8.56)
Haemulon sciurus	Bluestriped grunt	Haemulidae	I	18%	0.18 (0.06)	23.41 (6.54)
Haemulon striatum	Striped grunt	Haemulidae	I	1%	0.01 (0.01)	0.25 (0.25)
Haemulon sp.	Grunt sp.	Haemulidae	I	18%	4.88 (1.54)	2.91 (1.03)
Halichoeres bivittatus	Slippery dick	Labridae	I	28%	1.99 (0.36)	10.73 (2.07)
Halichoeres garnoti	Yellowhead wrasse	Labridae	I	40%	2.98 (0.29)	27.15 (3.35)
Halichoeres maculipinna	Clown wrasse	Labridae	I	10%	0.2 (0.06)	0.29 (0.08)
Halichoeres pictus	Rainbow wrasse	Labridae	I	1%	0.01 (0.01)	0.03 (0.03)
Halichoeres poeyi	Blackear wrasse	Labridae	I	16%	0.39 (0.09)	1.87 (0.55)

Table A.2 (continued). Mean species site frequency, density, and biomass for fish species observed within the La Parguera study area in the August 2008-2010 surveys.

Species	Common name	Family	Trophic Group	% of Surveys	Mean Density (SE)	Mean Biomass (SE)
Halichoeres radiatus	Puddingwife	Labridae	I	5%	0.06 (0.02)	1.02 (0.5)
Heteroconger longissimus	Brown garden eel	Congridae	PL	4%	1.46 (0.65)	64.93 (29.62)
Heteropriacanthus cruentatus	Glasseye snapper	Priacanthidae	PL	1%	0.01 (0.01)	0.21 (0.21)
Holacanthus tricolor	Rock beauty	Pomacanthidae	I	7%	0.11 (0.03)	7.56 (2.59)
Holocentrus adscensionis	Squirrelfish	Holocentridae	I	7%	0.1 (0.03)	6.44 (2.52)
Holocentrus rufus	Longspine squirrelfish	Holocentridae	I	31%	0.46 (0.05)	34.47 (4.13)
Hypoplectrus chlorurus	Yellowtail hamlet	Serranidae	I	9%	0.13 (0.04)	1.18 (0.46)
Hypoplectrus guttavarius	Shy hamlet	Serranidae	I	1%	0.01 (0.01)	0.06 (0.04)
Hypoplectrus indigo	Indigo hamlet	Serranidae	I	2%	0.02 (0.01)	0.06 (0.04)
Hypoplectrus nigricans	Black hamlet	Serranidae	I	2%	0.02 (0.01)	0.12 (0.06)
Hypoplectrus puella	Barred hamlet	Serranidae	I	12%	0.17 (0.04)	0.98 (0.25)
Hypoplectrus unicolor	Butter hamlet	Serranidae	I	10%	0.12 (0.03)	0.85 (0.28)
Hypoplectrus sp.	Hamlet sp.	Serranidae	I	6%	0.1 (0.03)	0.22 (0.09)
Inermia vittata	Boga	Inermiidae	PL	1%	0.01 (0.01)	0.15 (0.15)
Jenkinsia sp.	Herring sp.	Clupeidae	PL	2%	0.36 (0.27)	0.02 (0.01)
Lachnolaimus maximus	Hogfish	Labridae	I	4%	0.04 (0.02)	2 (0.86)
Lactophrys bicaudalis	Spotted trunkfish	Ostraciidae	I	2%	0.02 (0.01)	0.43 (0.31)
Lactophrys trigonus	Trunkfish	Ostraciidae	I	1%	0.01 (0.01)	1.36 (0.97)
Lonchopisthus micrognathus	Swordtail jawfish	Opistognathidae	P	3%	0.05 (0.02)	0.31 (0.14)
Lutjanus analis	Mutton snapper	Lutjanidae	I	1%	0.01 (0.01)	11.68 (11.68)
Lutjanus apodus	Schoolmaster	Lutjanidae	P	19%	0.25 (0.12)	16.73 (6.61)
Lutjanus cyanopterus	Cubera snapper	Lutjanidae	P	1%	<0.01 (<0.01)	0.11 (0.11)
Lutjanus griseus	Gray snapper	Lutjanidae	I	10%	0.01 (<0.01)	1.56 (0.53)
Lutjanus jocu	Dog snapper	Lutjanidae	P	1%	0.01 (0.01)	0.47 (0.45)
Lutjanus mahogoni	Mahogany snapper	Lutjanidae	P	1%	0.02 (0.02)	7.54 (7.54)
Lutjanus synagris	Lane snapper	Lutjanidae	I	9%	0.3 (0.11)	4.52 (1.6)
Malacanthus plumieri	Sand tilefish	Malacanthidae	I	3%	0.04 (0.02)	10.01 (4.41)
Malacoctenus boehlkei	Diamond blenny	Labrisomidae	I	2%	0.02 (0.01)	0.02 (0.02)
Malacoctenus macropus	Rosy blenny	Labrisomidae	I	3%	0.04 (0.02)	0.03 (0.01)
Malacoctenus triangulatus	Saddled blenny	Labrisomidae	I	2%	0.02 (0.01)	0.06 (0.03)
Malacoctenus sp.	Scaly Blenny sp.	Labrisomidae	I	1%	0.01 (0.01)	<0.01 (<0.01)
Melichthys niger	Black durgon	Balistidae	H	1%	0.04 (0.04)	24.72 (18.85)
Microgobius carri	Seminole goby	Gobiidae	H	1%	0.01 (0.01)	0.02 (0.02)
Microgobius signatus	Microgobius signatus	Gobiidae	H	1%	0.04 (0.03)	0.01 (0.01)
Microspathodon chrysurus	Yellowtail damselfish	Pomacentridae	H	14%	0.35 (0.11)	20.46 (5.58)
Monacanthus ciliatus	Fringed filefish	Monacanthidae	H	1%	0.01 (0.01)	0.01 (<0.01)
Monacanthus tuckeri	Slender filefish	Monacanthidae	PL	1%	0.01 (0.01)	0.01 (0.01)
Mulloidichthys martinicus	Yellow goatfish	Mullidae	I	3%	0.09 (0.05)	8.69 (5.5)
Myripristis jacobus	Blackbar soldierfish	Holocentridae	I	3%	0.05 (0.02)	4.28 (1.93)
Neoniphon marianus	Longjaw squirrelfish	Holocentridae	I	2%	0.03 (0.01)	0.62 (0.41)

Table A.2 (continued). Mean species site frequency, density, and biomass for fish species observed within the La Parguera study area in the August 2008-2010 surveys.

Species	Common name	Family	Trophic Group	% of Surveys	Mean Density (SE)	Mean Biomass (SE)
Nes longus	Orangespotted goby	Gobiidae	H	14%	1.65 (0.42)	6 (1.76)
Ocyurus chrysurus	Yellowtail snapper	Lutjanidae	PL	38%	1.29 (0.22)	57.45 (13.64)
Ophioblennius macclurei	Redlip blenny	Blenniidae	H	2%	0.02 (0.01)	0.05 (0.04)
Opistognathus aurifrons	Yellowhead jawfish	Opistognathidae	PL	9%	0.32 (0.09)	0.56 (0.19)
Opistognathus macrognathus	Banded jawfish	Opistognathidae	P	1%	0.01 (0.01)	0.04 (0.04)
Oxyurichthys stigmalophius	Spotfin goby	Gobiidae	H	1%	0.01 (0.01)	<0.01 (<0.01)
Paradiplogrammus bairdi	Lancer dragonet	Callionymidae	I	2%	0.05 (0.02)	0.06 (0.03)
Pareques acuminatus	Highhat	Sciaenidae	I	1%	0.01 (0.01)	<0.01 (<0.01)
Pomacanthus arcuatus	Gray angelfish	Pomacanthidae	I	8%	0.09 (0.02)	47.47 (16.21)
Pomacanthus paru	French angelfish	Pomacanthidae	I	1%	0.02 (0.02)	1.84 (1.81)
Prognathodes aculeatus	Longsnout butterflyfish	Chaetodontidae	I	1%	0.01 (0.01)	0.07 (0.07)
Pseudupeneus maculatus	Spotted goatfish	Mullidae	I	18%	0.27 (0.05)	16.03 (4.04)
Ptereleotris helenae	Hovering goby	Microdesmidae	PL	2%	0.05 (0.03)	0.27 (0.21)
Sargocentron vexillarium	Dusky squirrelfish	Holocentridae	I	1%	0.01 (0.01)	0.2 (0.2)
Scarus guacamaia	Rainbow parrotfish	Scaridae	H	1%	<0.01 (<0.01)	0.03 (0.03)
Scarus iseri	Striped parrotfish	Scaridae	H	49%	6.16 (1.11)	93.15 (16.04)
Scarus taeniopterus	Princess parrotfish	Scaridae	H	34%	2.79 (0.38)	94 (14.34)
Scarus vetula	Queen parrotfish	Scaridae	H	2%	0.03 (0.02)	0.58 (0.34)
Scomberomorus regalis	Cero	Scombridae	P	1%	0.02 (0.02)	19.95 (14.13)
Scorpaena plumieri	Spotted scorpionfish	Scorpaenidae	I	1%	0.01 (0.01)	0.59 (0.59)
Serranus baldwini	Lantern bass	Serranidae	P	5%	0.1 (0.03)	0.1 (0.04)
Serranus flaviventris	Twinspot bass	Serranidae	P	1%	0.01 (0.01)	0.03 (0.03)
Serranus tabacarius	Tobaccofish	Serranidae	P	9%	0.15 (0.04)	0.28 (0.08)
Serranus tigrinus	Harlequin bass	Serranidae	I	15%	0.29 (0.05)	1.43 (0.3)
Serranus tortugarum	Chalk bass	Serranidae	PL	6%	0.49 (0.21)	0.35 (0.12)
Sparisoma atomarium	Greenblotch parrotfish	Scaridae	H	17%	0.55 (0.13)	0.63 (0.2)
Sparisoma aurofrenatum	Redband parrotfish	Scaridae	H	50%	4.22 (0.29)	117.71 (11.01)
Sparisoma chrysopterum	Redtail parrotfish	Scaridae	H	1%	0.02 (0.01)	6.44 (4.78)
Sparisoma radians	Bucktooth parrotfish	Scaridae	H	19%	1.44 (0.3)	1.49 (0.43)
Sparisoma rubripinne	Yellowtail parrotfish	Scaridae	H	4%	0.07 (0.04)	10.33 (7.73)
Sparisoma sp.	Parrotfish sp.	Scaridae	H	2%	0.04 (0.03)	0.05 (0.04)
Sparisoma viride	Stoplight parrotfish	Scaridae	H	35%	1.5 (0.2)	172.66 (30.64)
Sphoeroides spengleri	Bandtail puffer	Tetraodontidae	I	3%	0.03 (0.01)	0.15 (0.1)
Sphoeroides testudineus	Checkered puffer	Tetraodontidae	I	5%	0.02 (0.01)	0.2 (0.09)
Sphyraena barracuda	Great barracuda	Sphyraenidae	P	12%	0.03 (0.01)	102.34 (90.65)
Stegastes adustus	Dusky damselfish	Pomacentridae	H	5%	0.84 (0.38)	6.94 (3.49)
Stegastes diencaeus	Longfin damselfish	Pomacentridae	H	7%	0.37 (0.14)	2.33 (0.98)
Stegastes leucostictus	Beaugregory	Pomacentridae	I	30%	1 (0.27)	4.87 (1.03)
Stegastes partitus	Bicolor damselfish	Pomacentridae	H	44%	3.35 (0.35)	6.46 (1.73)
Stegastes planifrons	Threespot damselfish	Pomacentridae	I	8%	0.5 (0.19)	4.88 (1.76)
Stegastes variabilis	Cocoa damselfish	Pomacentridae	H	19%	0.6 (0.13)	5.63 (1.22)

Table A.2 (continued). Mean species site frequency, density, and biomass for fish species observed within the La Parguera study area in the August 2008-2010 surveys.

Species	Common name	Family	Trophic Group	% of Surveys	Mean Density (SE)	Mean Biomass (SE)
Syacium sp.	Sand Flounder sp.	Paralichthyidae	I	1%	0.01 (0.01)	<0.01 (<0.01)
Syngathus dawsoni	NOT RECORDED	Syngnathidae	I	1%	0.01 (0.01)	0.01 (0.01)
Synodus foetens	Inshore lizardfish	Synodontidae	P	1%	0.01 (0.01)	<0.01 (<0.01)
Synodus intermedius	Sand diver	Synodontidae	P	3%	0.03 (0.01)	0.21 (0.12)
Thalassoma bifasciatum	Bluehead	Labridae	I	43%	6.62 (0.81)	6.8 (1.56)
Xyrichtys martinicensis	Rosy razorfish	Labridae	I	3%	0.2 (0.1)	0.36 (0.21)
Xyrichtys novacula	Pearly razorFish	Labridae	I	1%	0.01 (0.01)	0.03 (0.03)
Xyrichtys splendens	Green razorfish	Labridae	I	1%	0.01 (0.01)	0.02 (0.02)

Appendix B

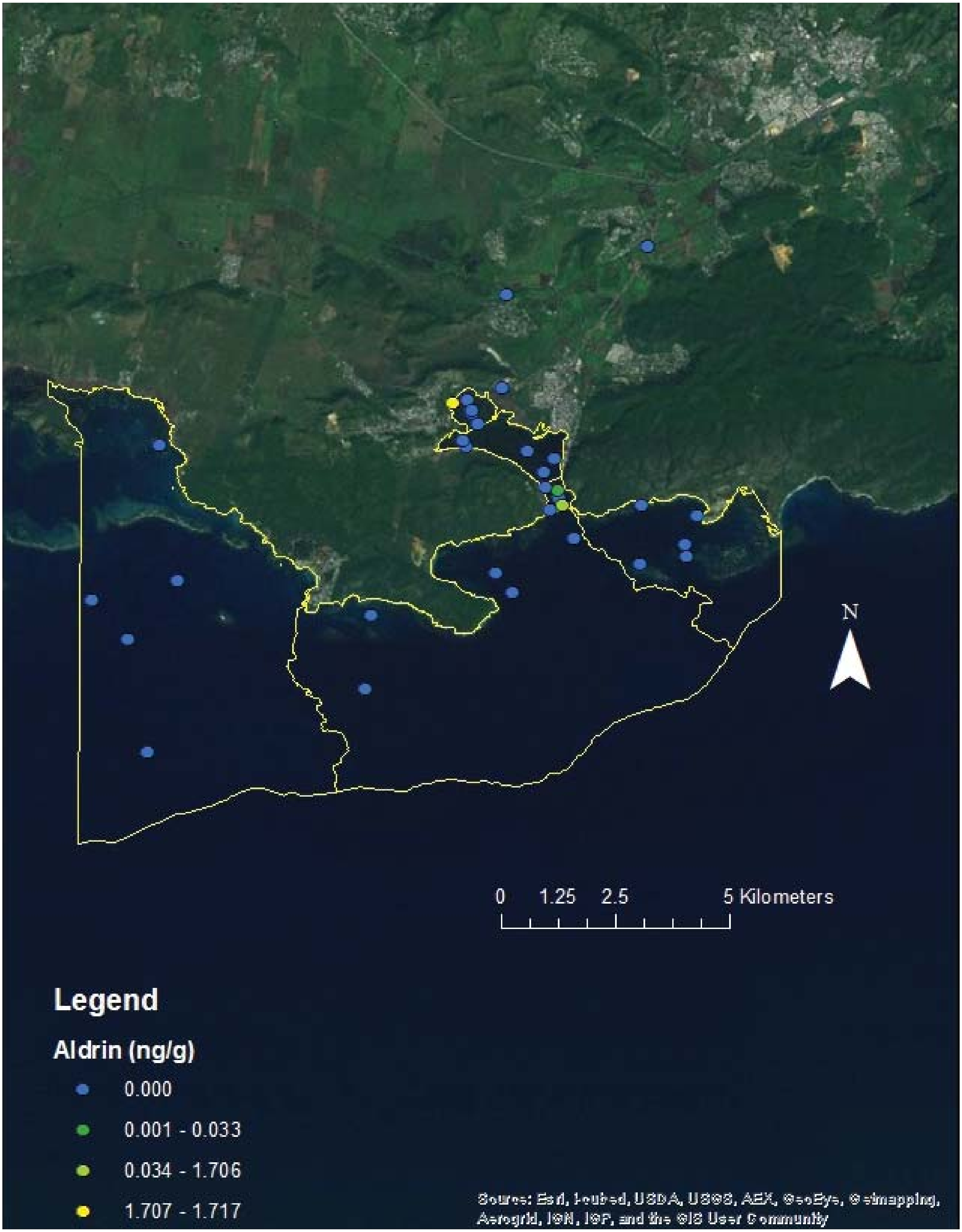

Figure B.1. Concentrations of aldrin (ng/g) in sediments.

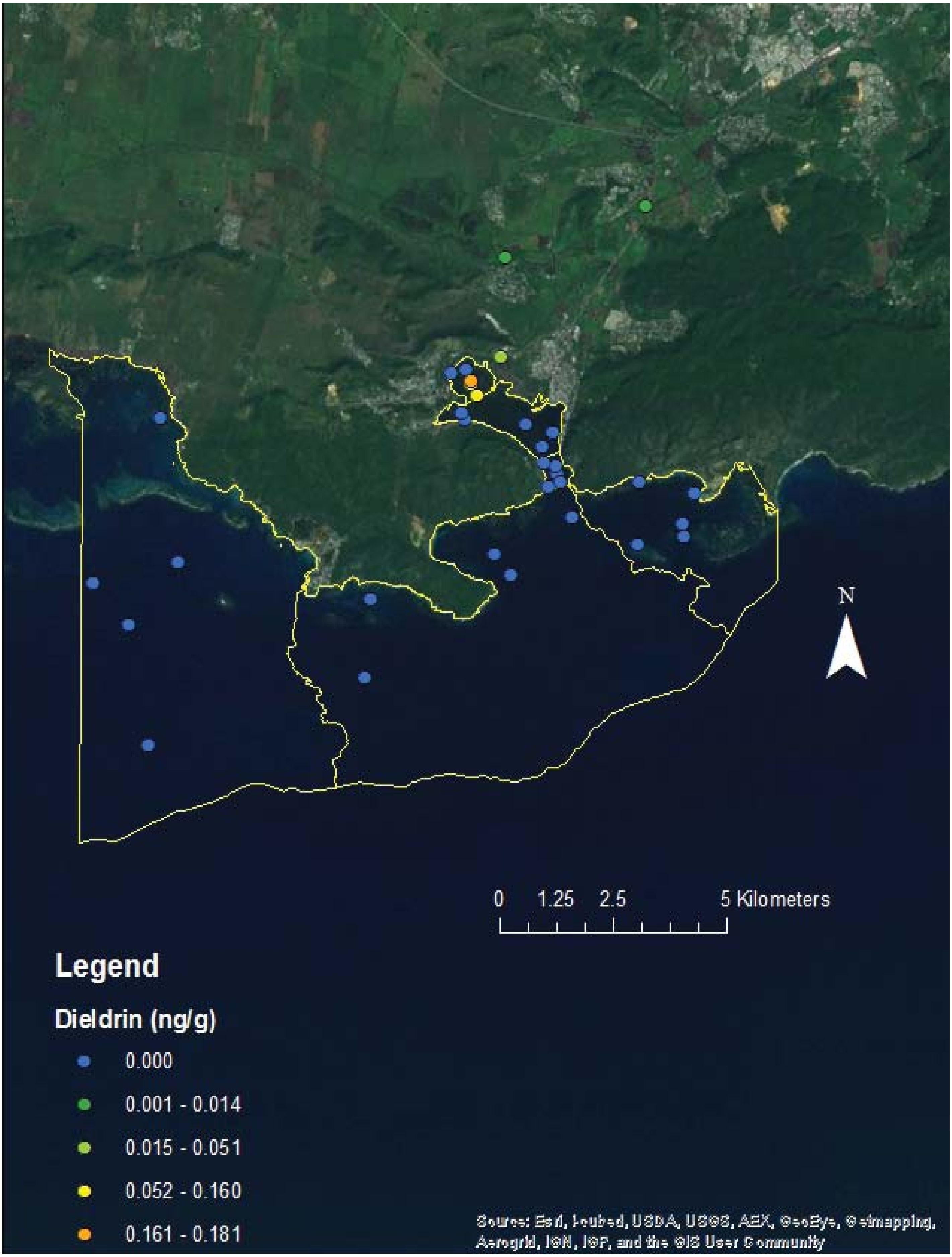

Figure B.2. Concentrations of dieldrin (ng/g) in sediments.

Figure B.3. Concentrations of endrin (ng/g) in sediments.

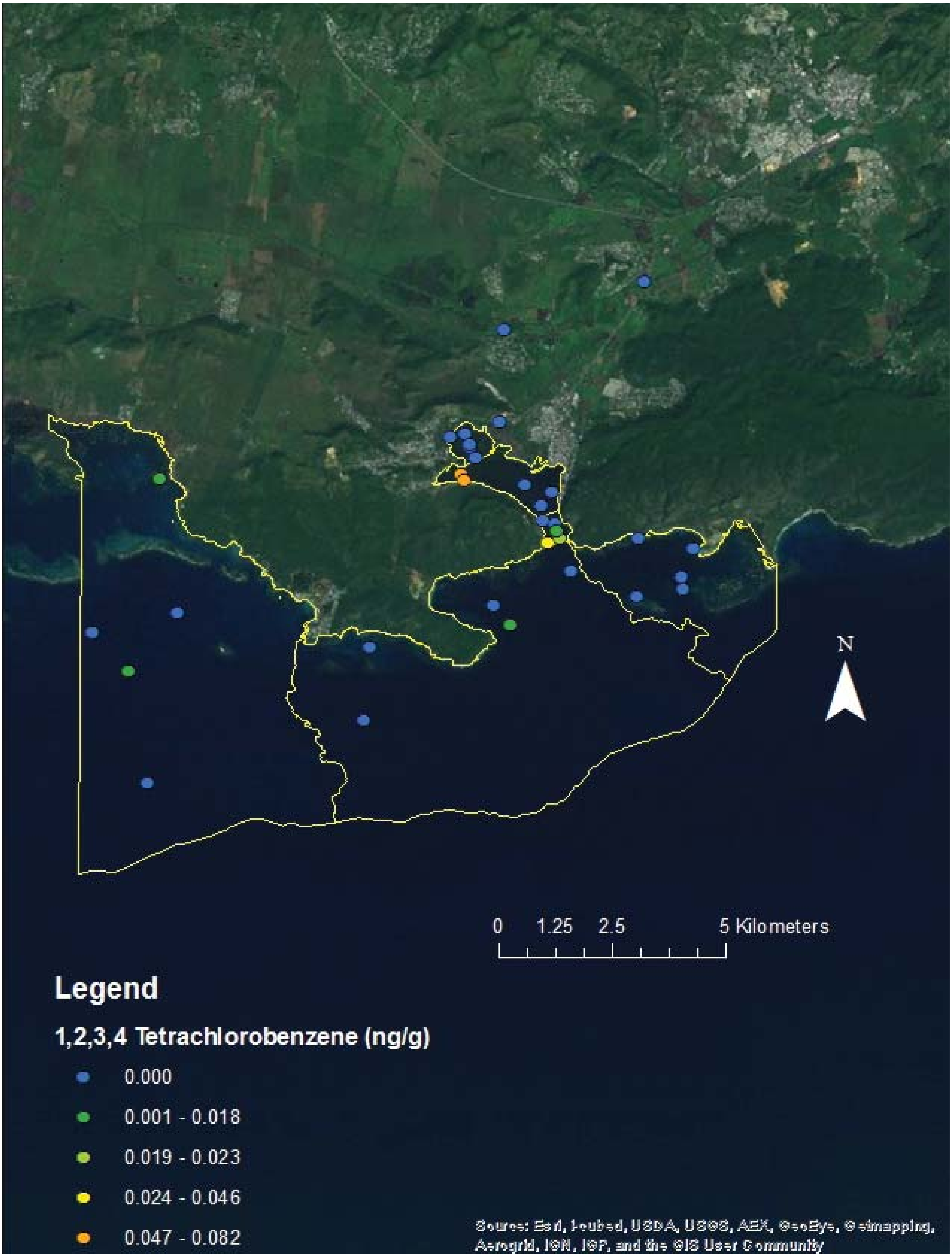

Figure B.4. Concentrations of tetrachlorobenzen (ng/g) in sediments.

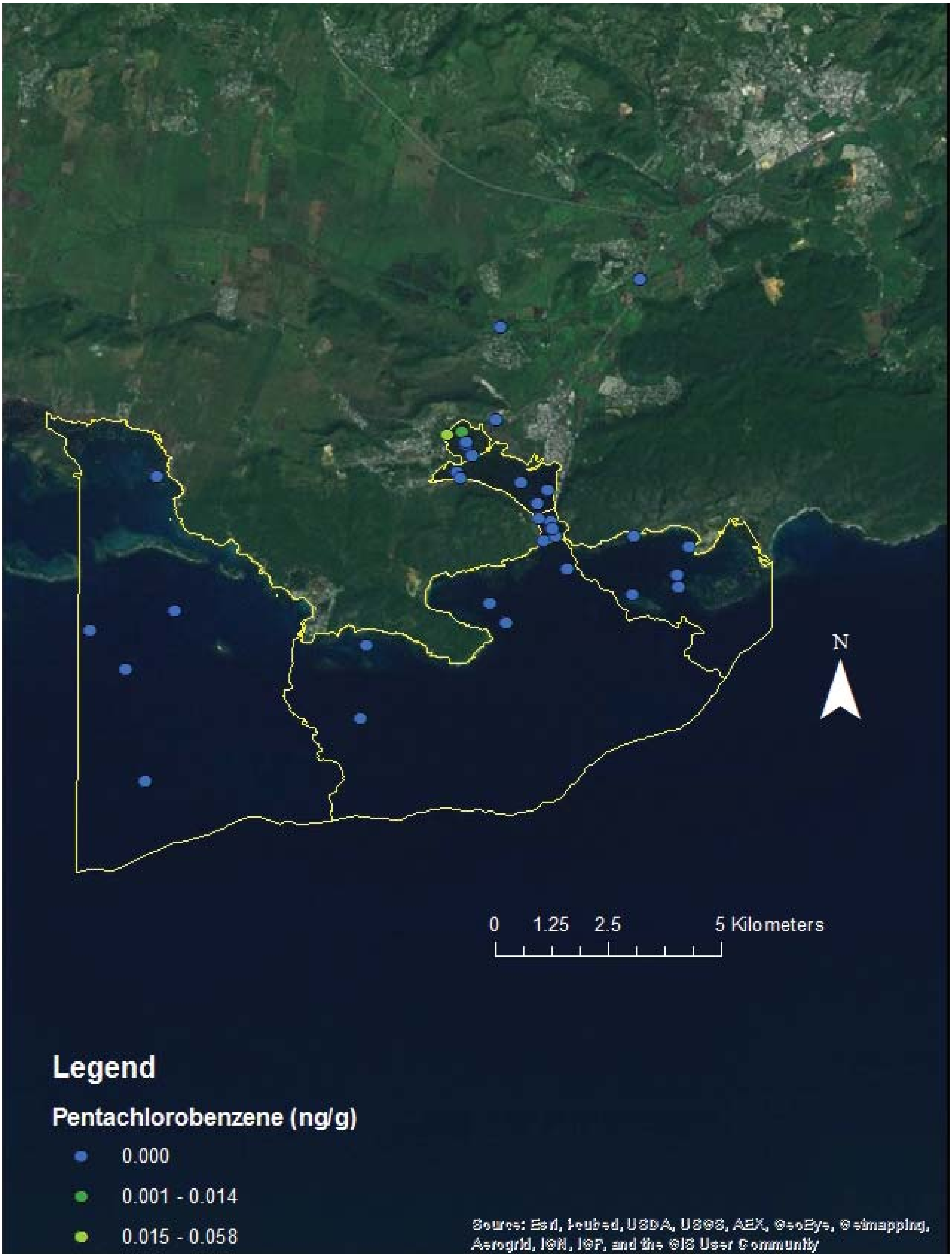

Figure B.5. Concentrations of pentachlorobenzene (ng/g) in sediments.

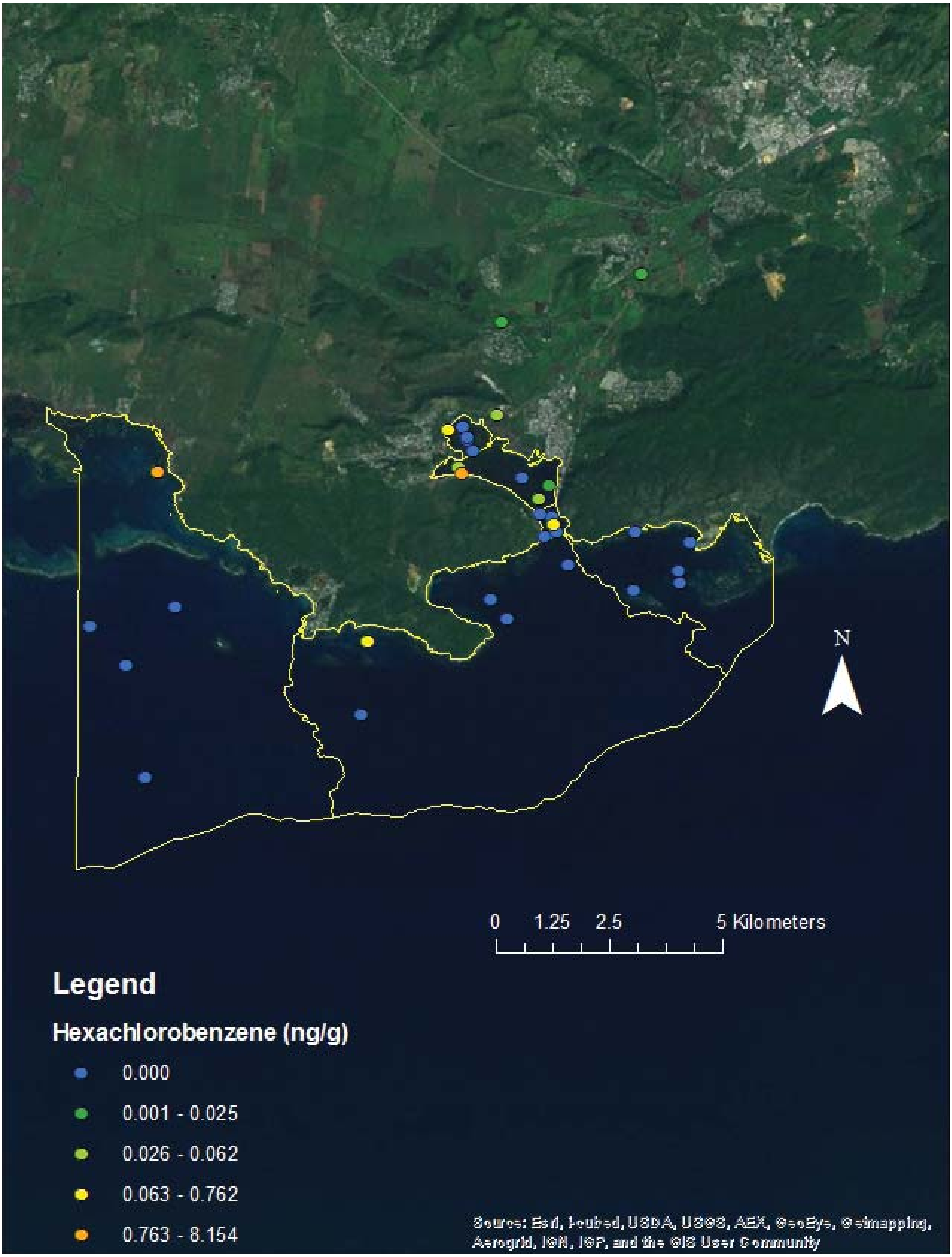

Figure B.6. Concentrations of hexachlorobenzene (ng/g) in sediments.

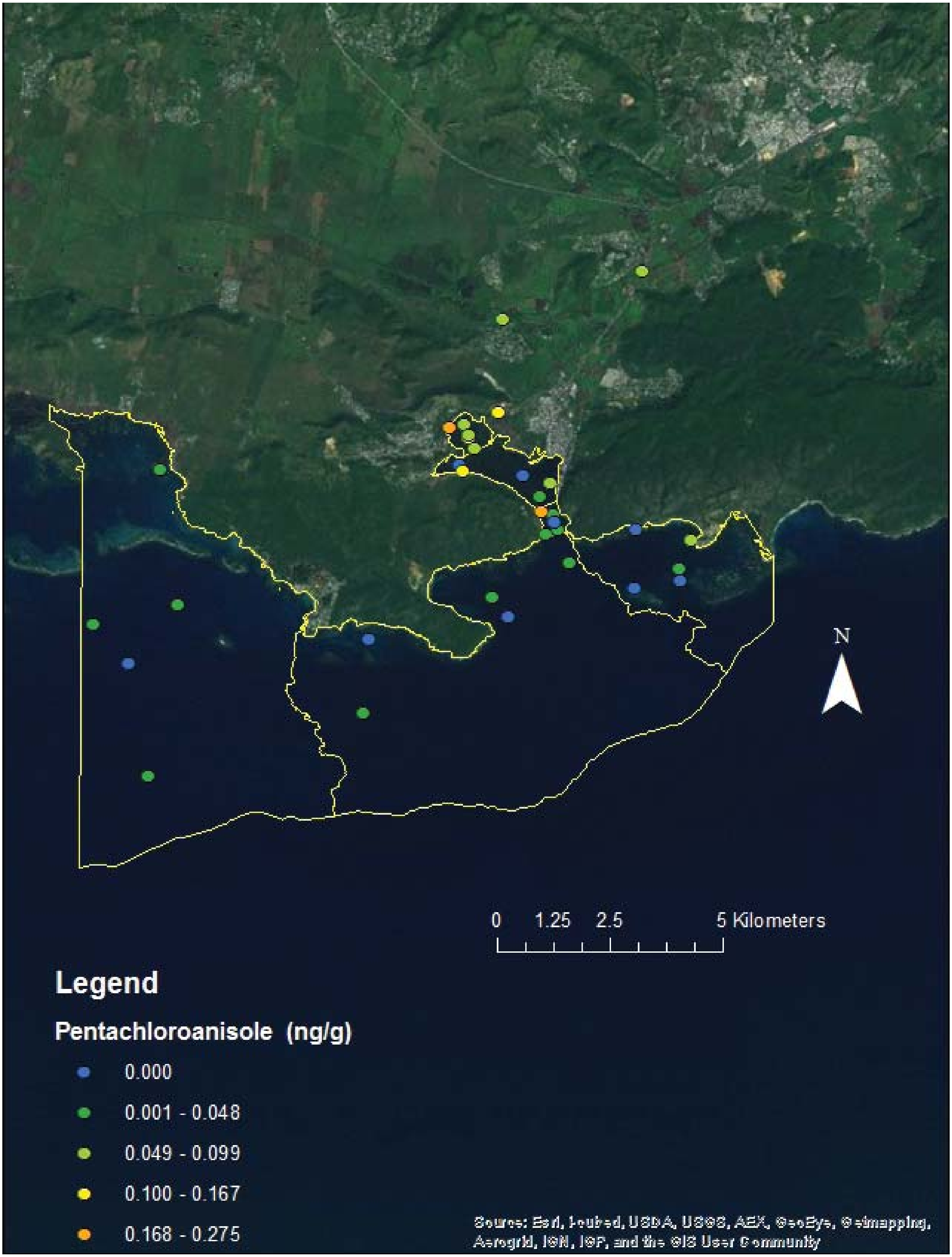

Figure B.7. Concentrations of pentachloroanisole (ng/g) in sediments.

Figure B.8. Concentrations of endosulfan II (ng/g) in sediments.

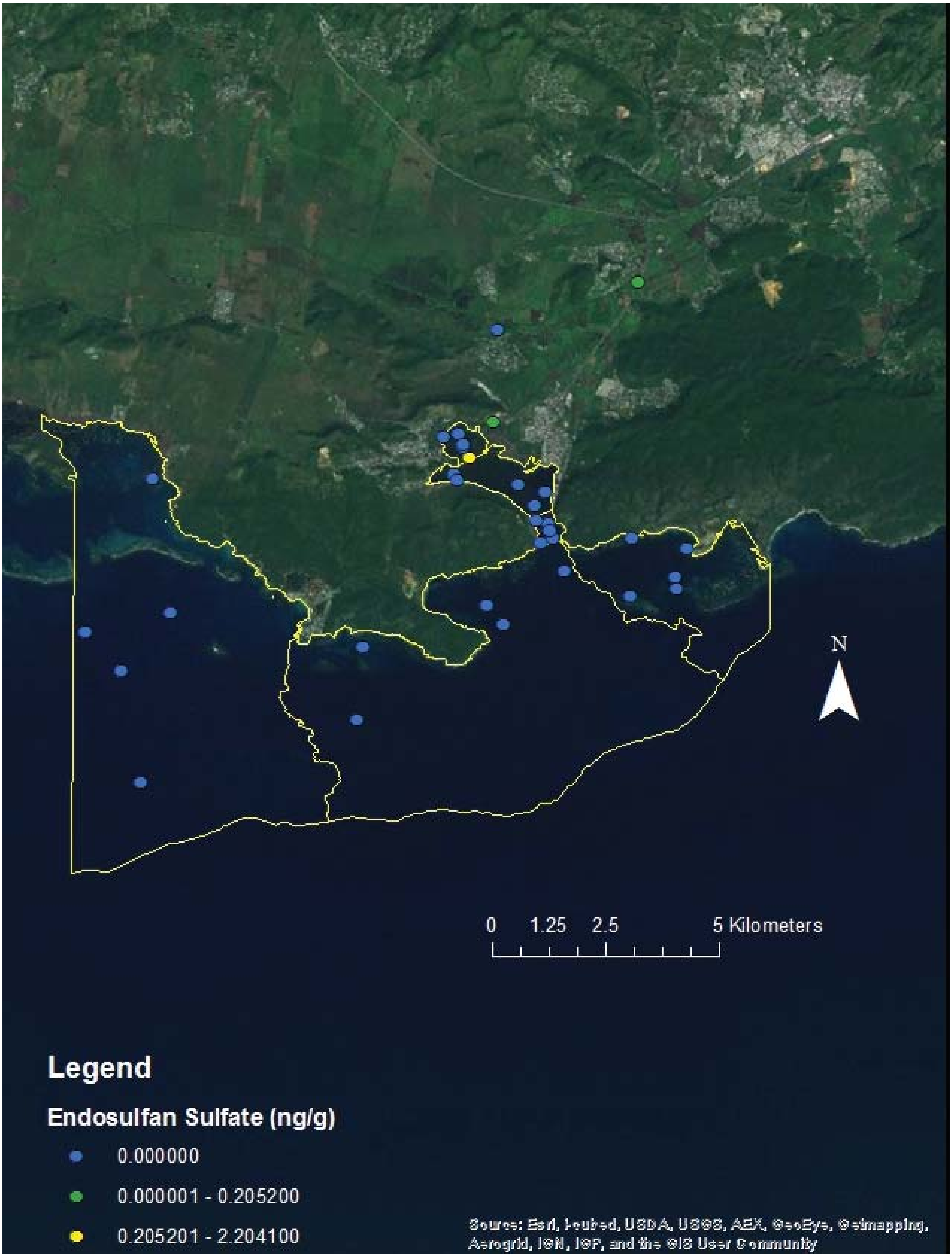

Figure B.9. Concentrations of endosulfan sulfate (ng/g) in sediments.

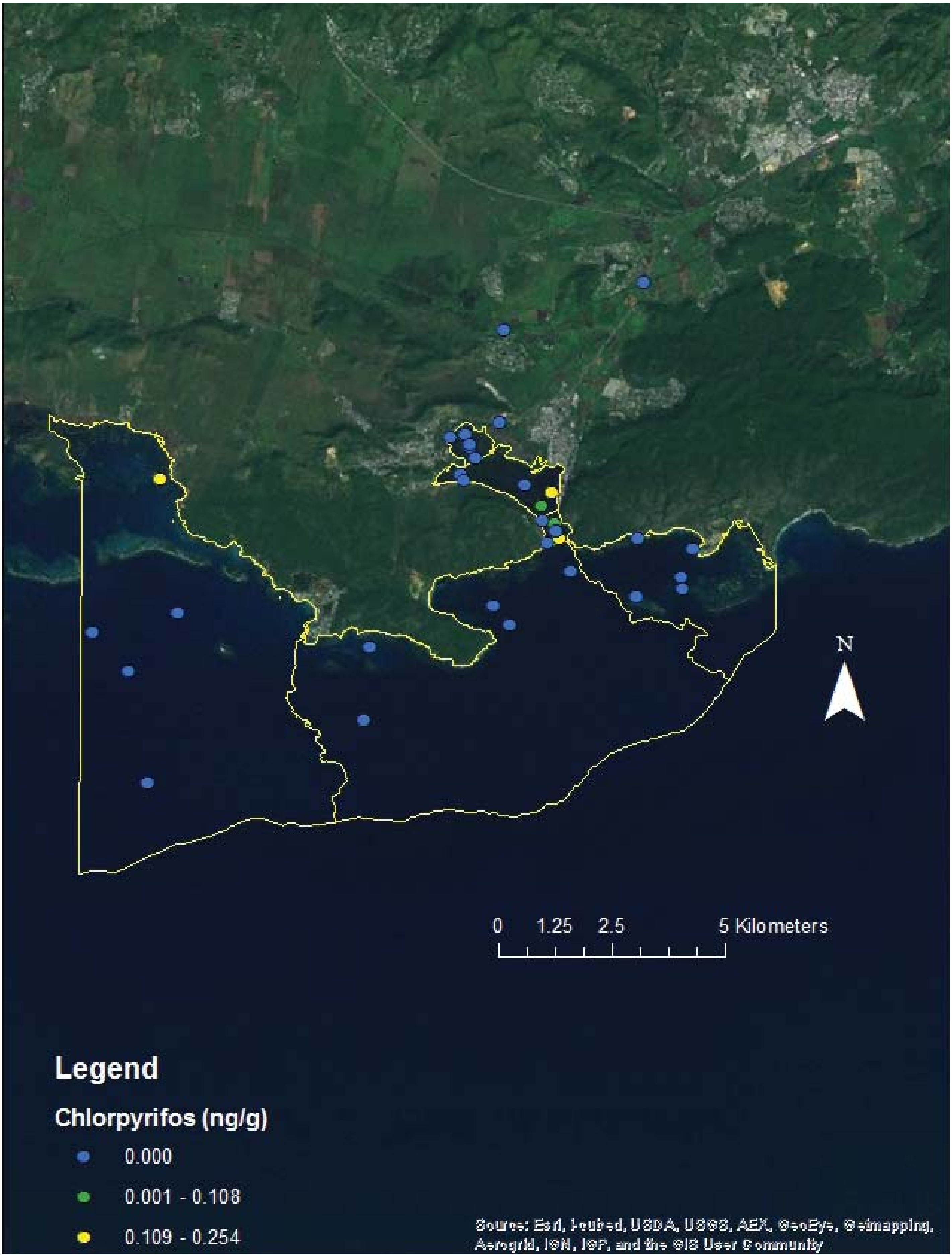

Figure B.10. Concentrations of chlorpyrifos (ng/g) in sediments.

Figure B.11. Concentrations of mirex (ng/g) in sediments.

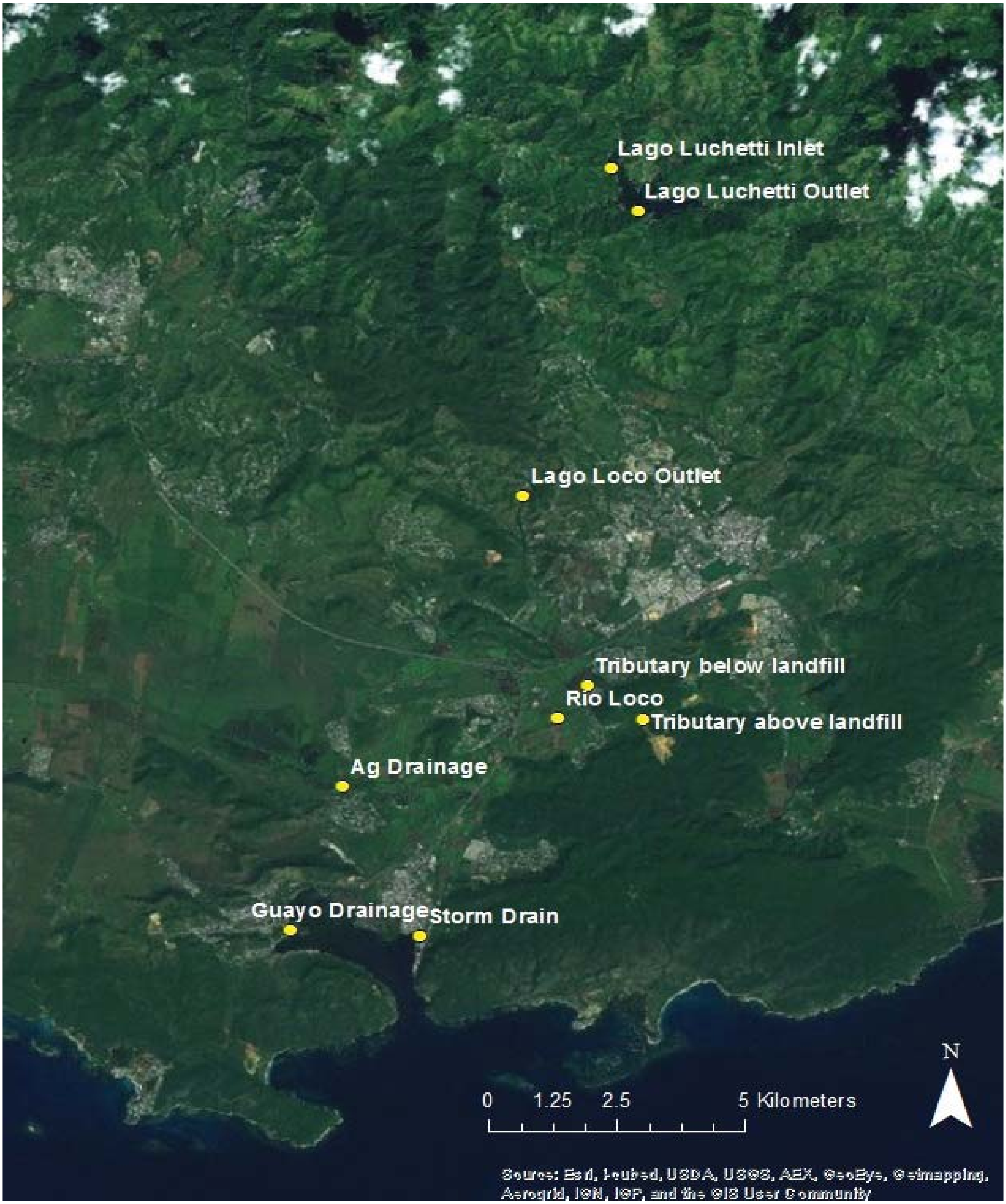

Figure B.12. Supplemental watershed sampling sites.

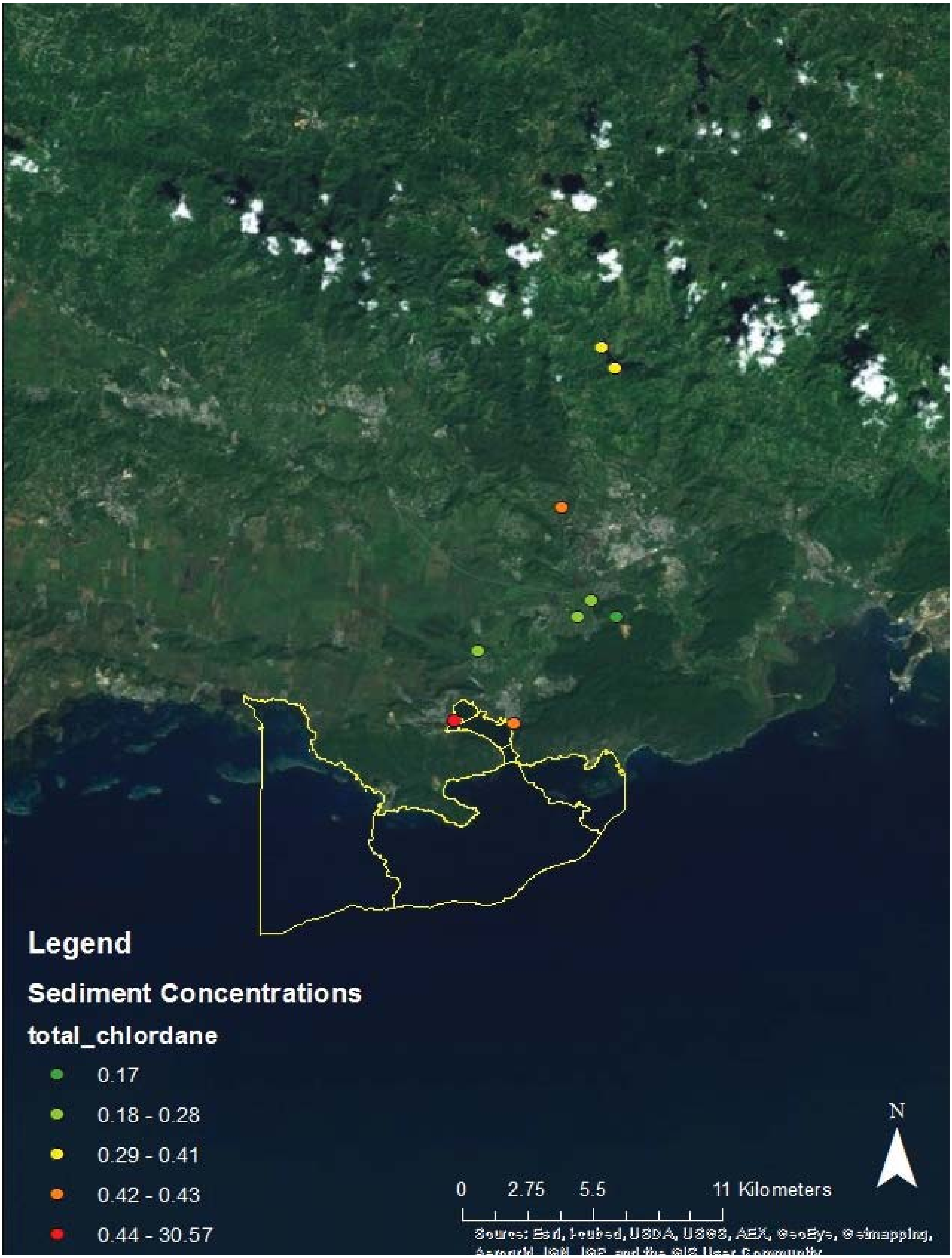

Figure B.13. Concentrations of total chlordane (ng/g) in watershed sediments.

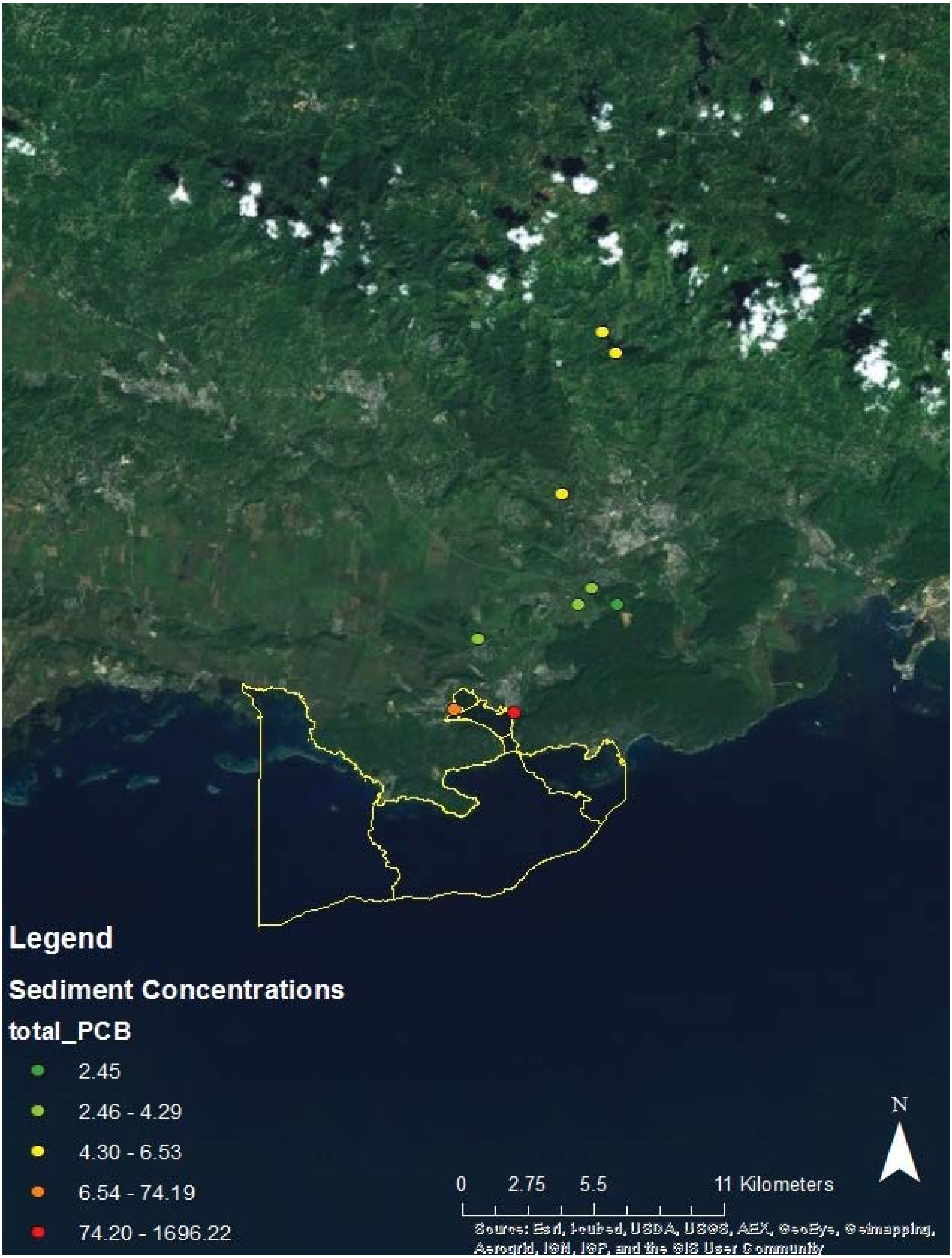

Figure B.14. Concentrations of total PCBs (ng/g) in watershed sediments.

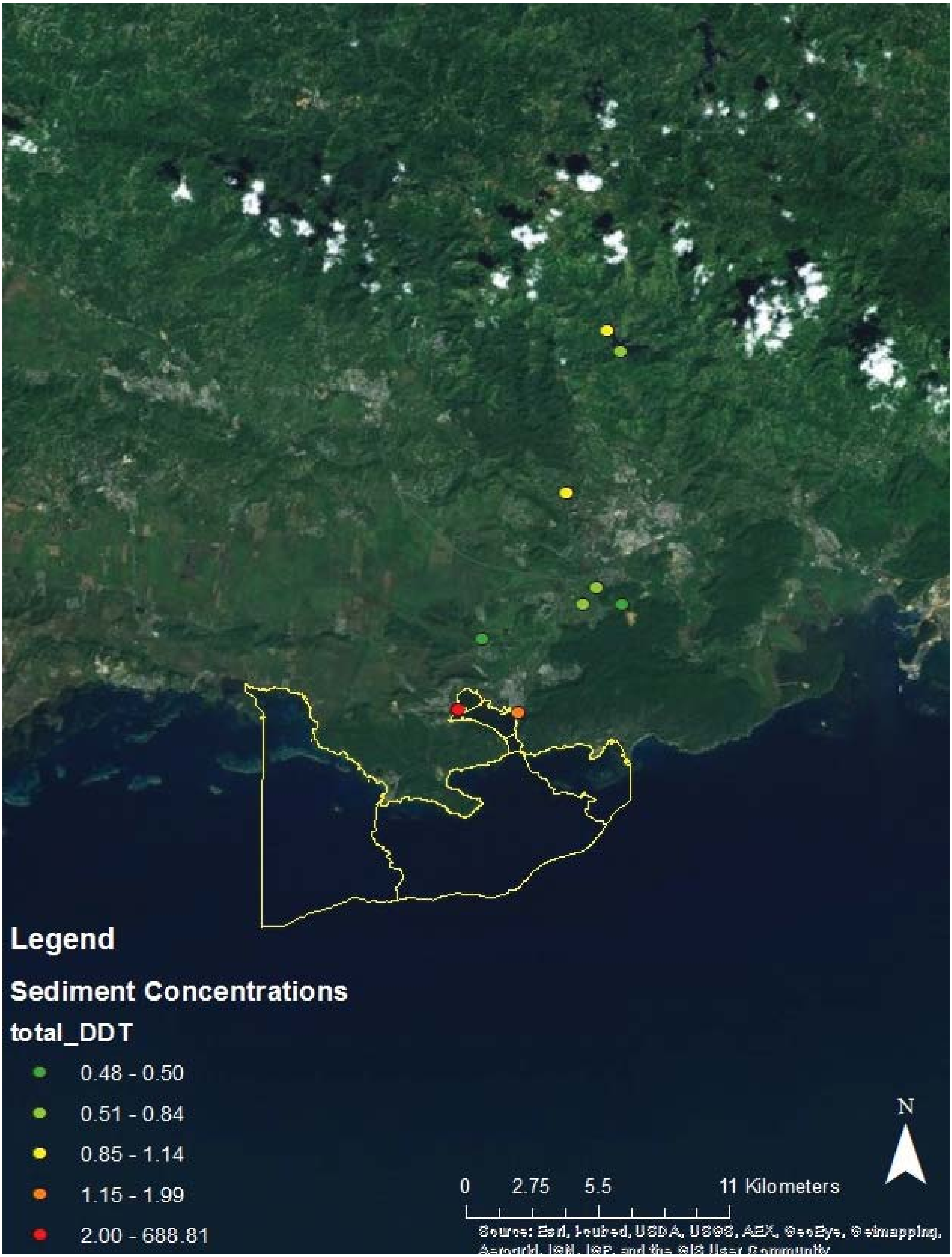

Figure B.15. Concentrations of total DDTs (ng/g) in watershed sediments.

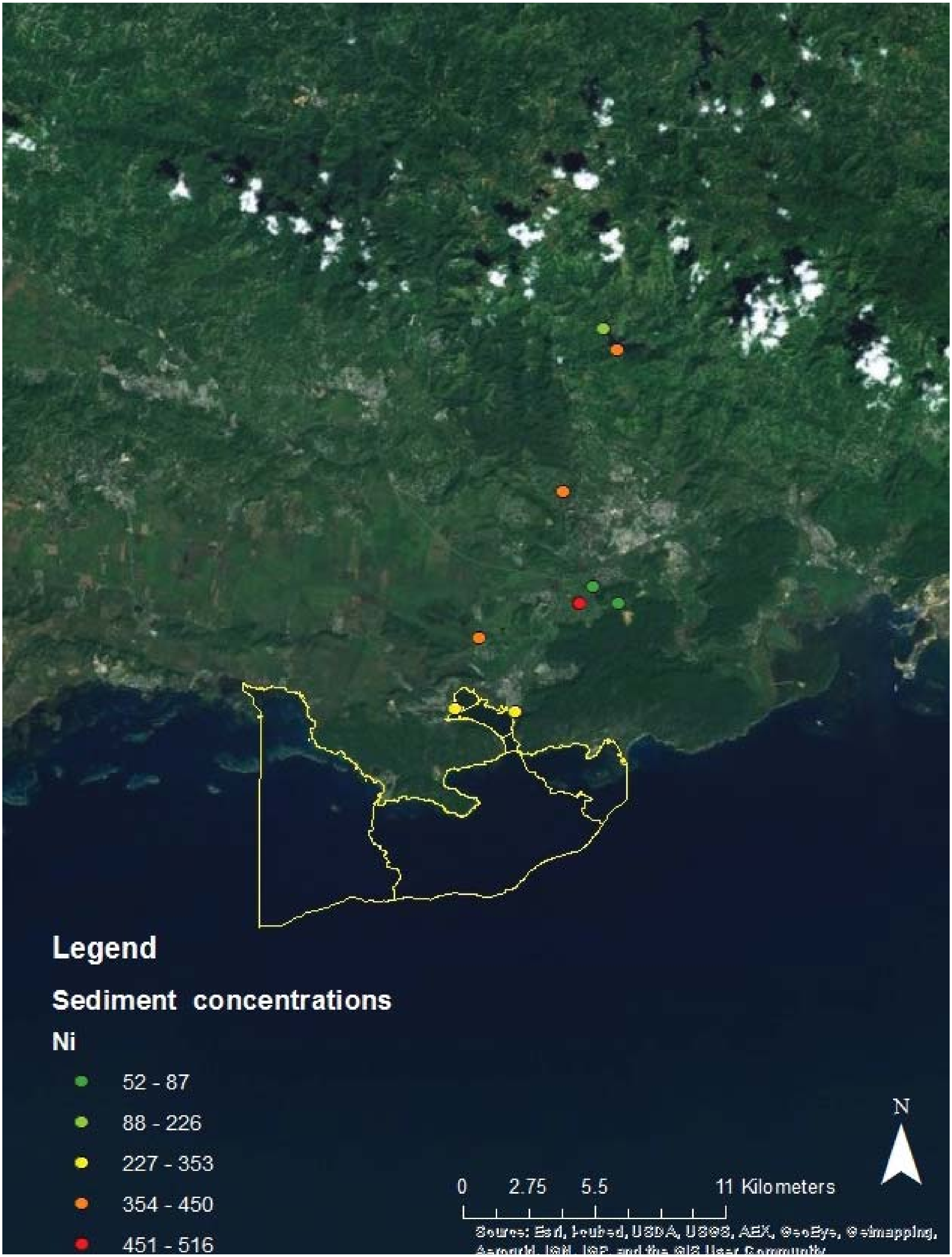

Figure B.16. Concentrations of nickel (μg/g) in watershed sediments.

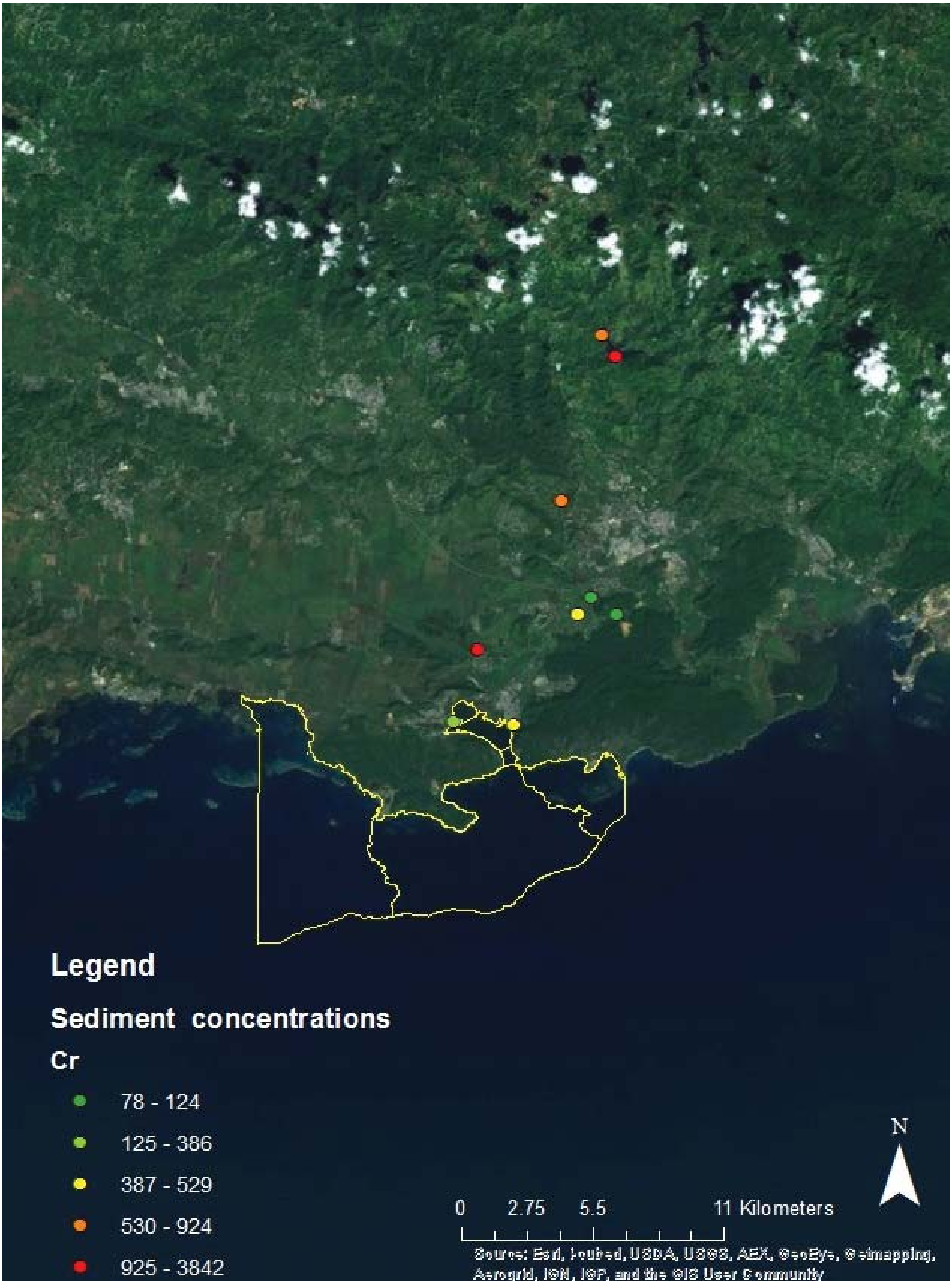

Figure B.17. Concentrations of chromium (µg/g) in watershed sediments.

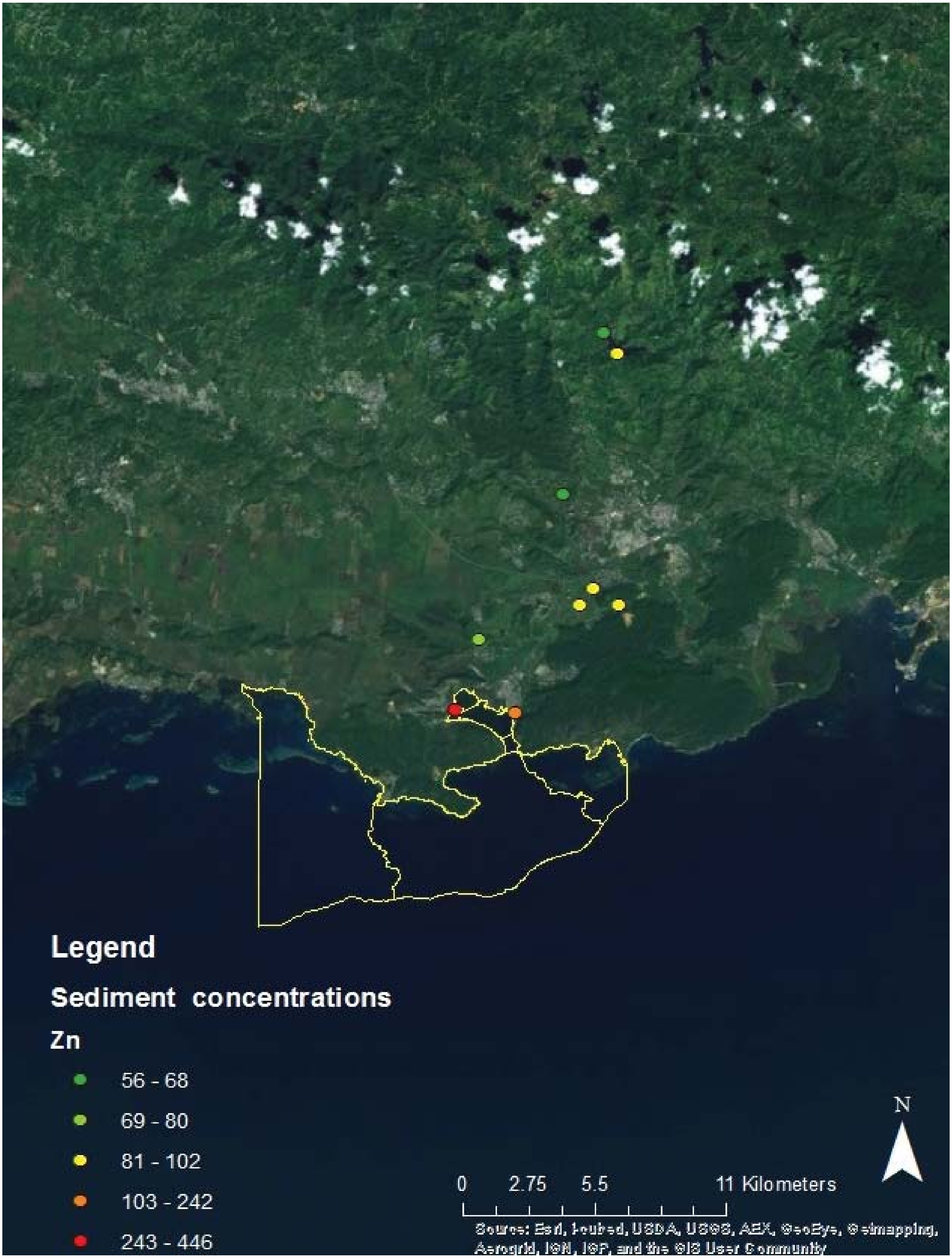

Figure B.18. Concentrations of zinc (μg/g) in watershed sediments.

Appendix C

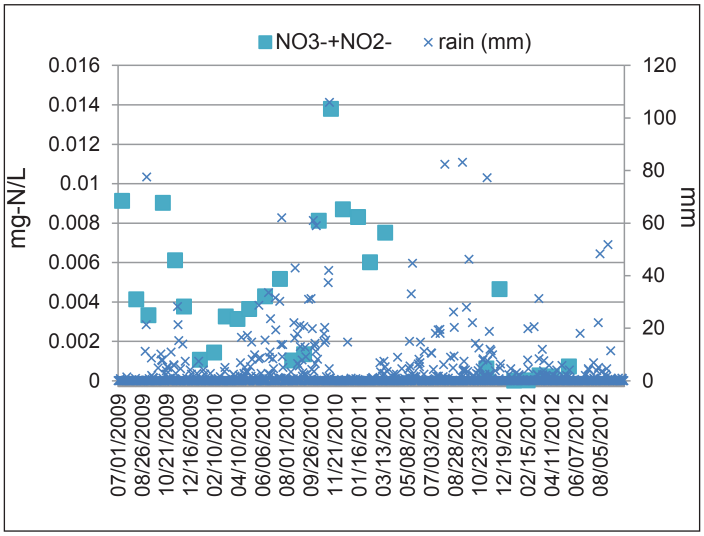

Figure C.1. Time series of precipitation (Lajas) and oxidized nitrogen (nitrate plus nitrite) concentrations from N13 (Offshore East strata).

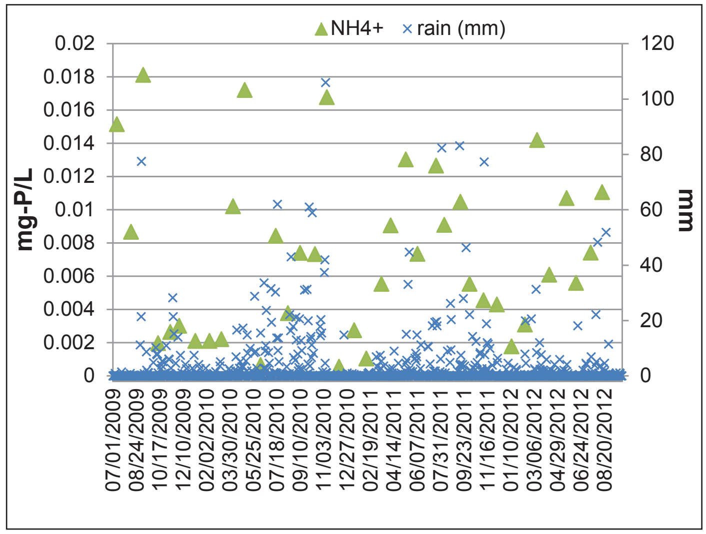

Figure C.2. Time series of precipitation (Lajas) and ammonium concentrations from N13 (Offshore East strata).

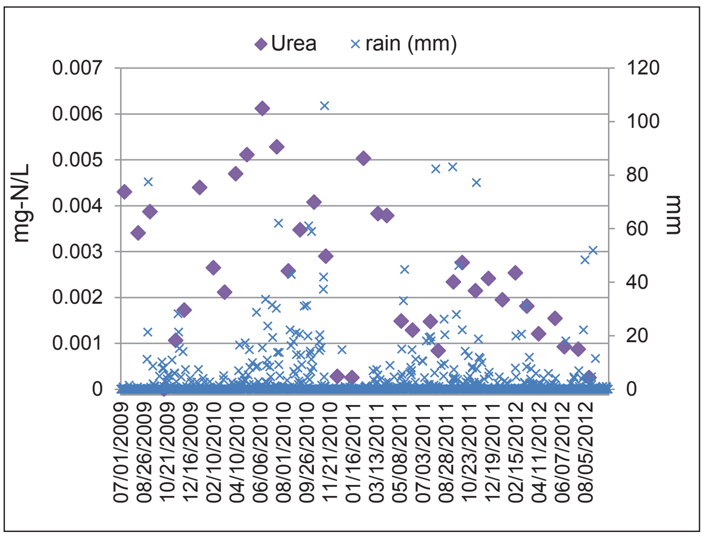

Figure C.3. Time series of precipitation (Lajas) and urea concentrations from N13 (Offshore East strata).

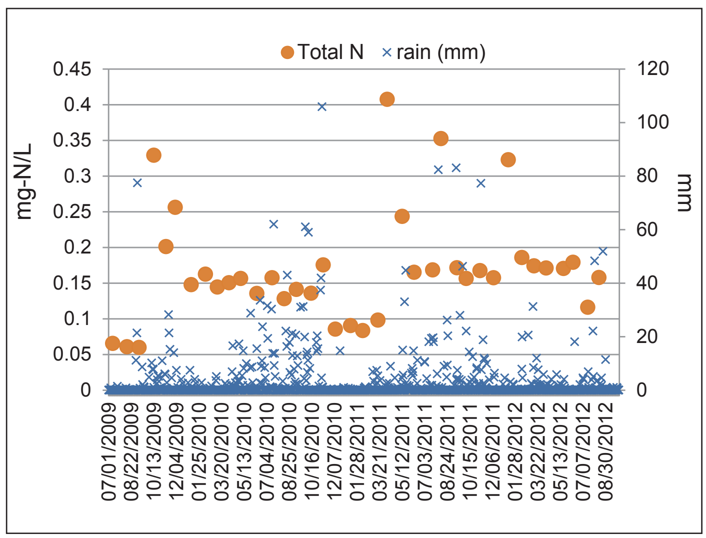

Figure C.4. Time series of precipitation (Lajas) and total nitrogen concentrations from N13 (Offshore East strata).

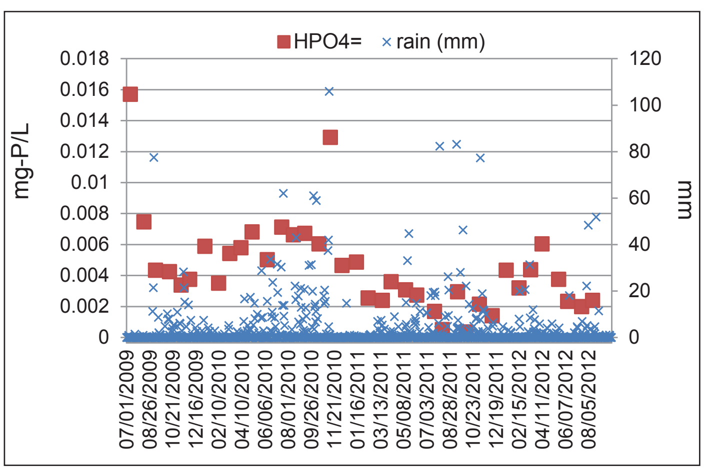

Figure C.5. Time series of precipitation (Lajas) and orthophosphate concentrations from N13 (Offshore East strata).

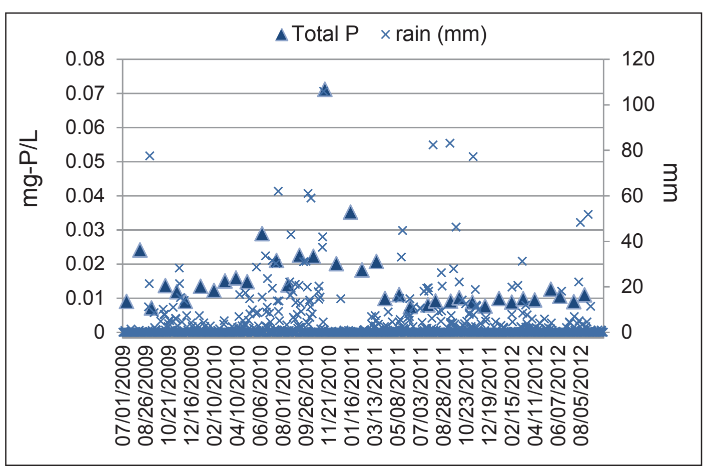

Figure C.6. Time series of precipitation (Lajas) and total phosphorus concentrations from N13 (Offshore East strata).

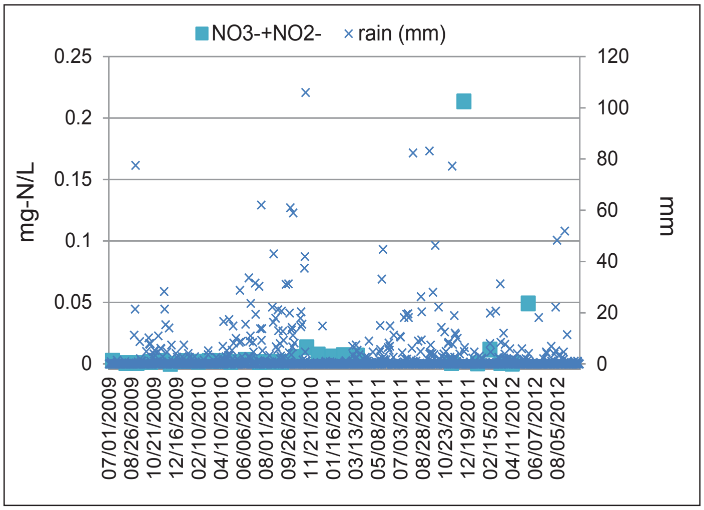

Figure C.7. Time series of precipitation (Lajas) and oxidized nitrogen (nitrate plus nitrite) concentrations from N17 (Offshore West strata).

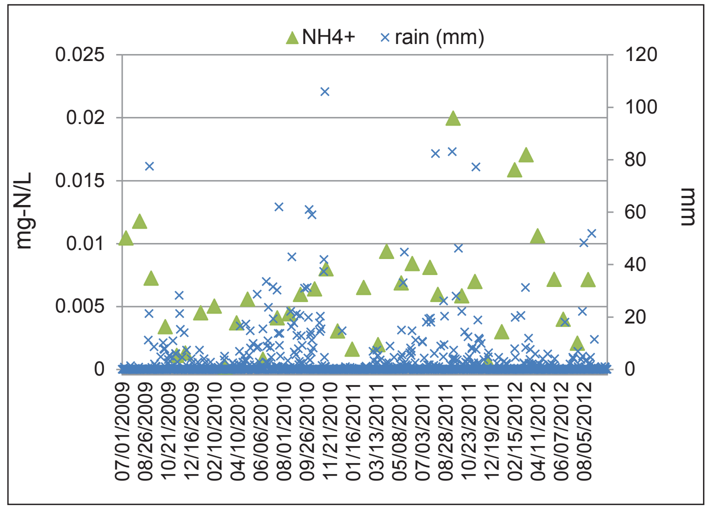

Figure C.8. Time series of precipitation (Lajas) and ammonium concentrations from N17 (Offshore West strata).

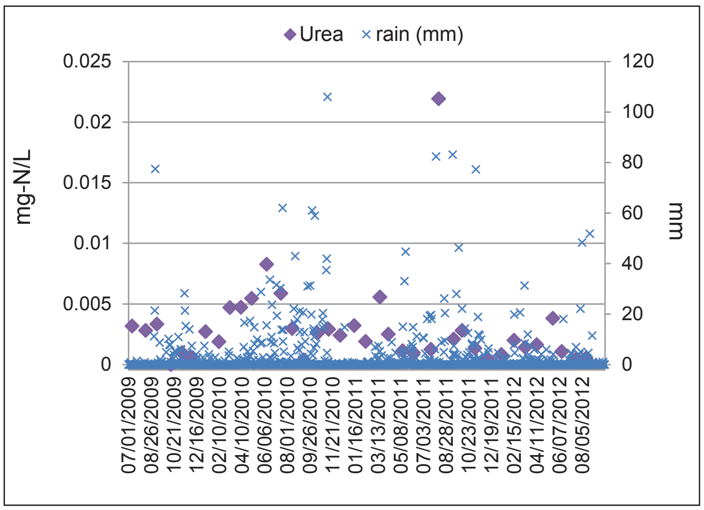

Figure C.9. Time series of precipitation (Lajas) and urea concentrations from N17 (Offshore West strata).

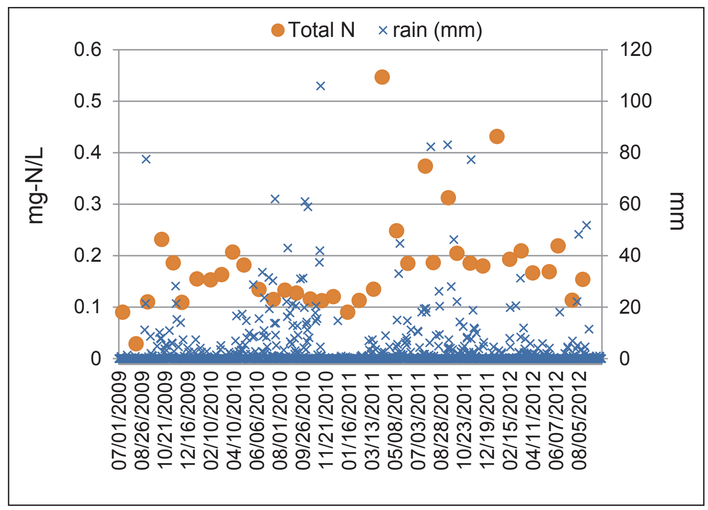

Figure C.10. Time series of precipitation (Lajas) and total nitrogen concentrations from N17 (Offshore West strata).

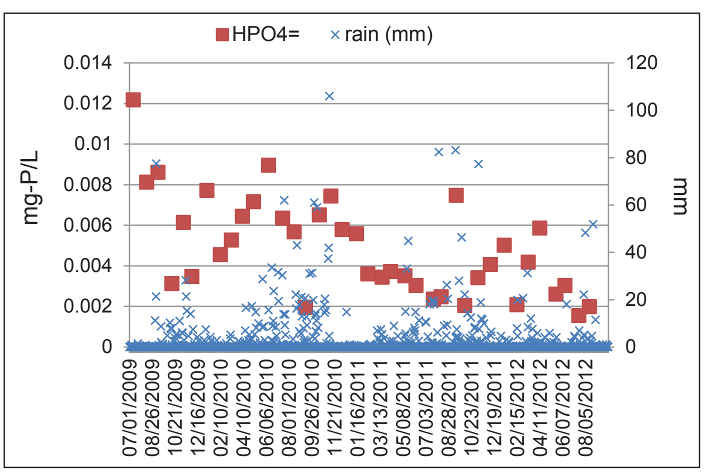

Figure C.11. Time series of precipitation (Lajas) and orthophosphate concentrations from N17 (Offshore West strata).

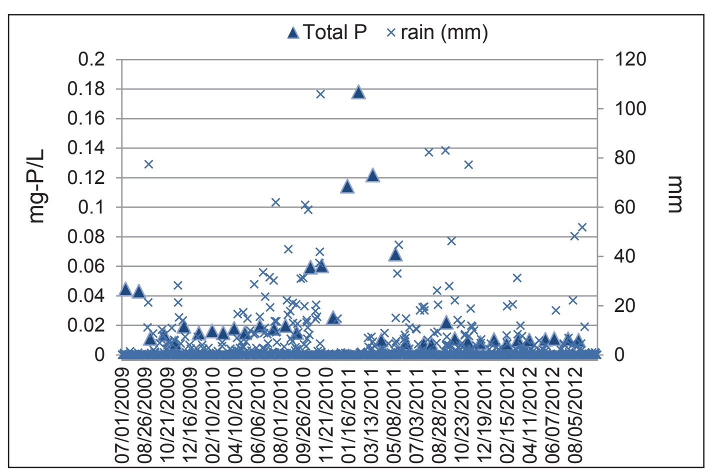

Figure C.12. Time series of precipitation (Lajas) and total phosphorus concentrations from N17 (Offshore West strata).

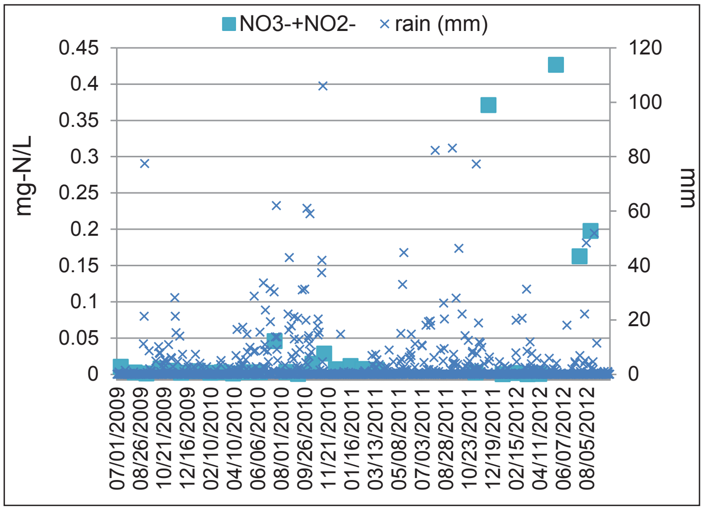

Figure C.13. Time series of precipitation (Lajas) and oxidized nitrogen (nitrate plus nitrite) concentrations from N28 (Offshore Central strata).

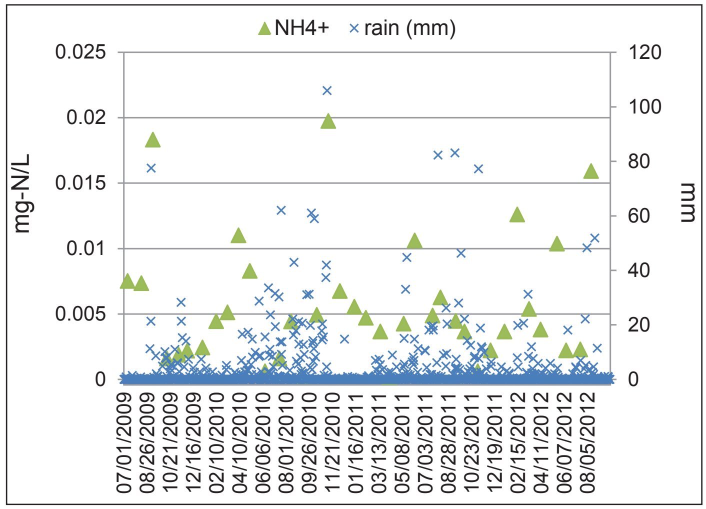

Figure C.14. Time series of precipitation (Lajas) and ammonium concentrations from N28 (Offshore Central strata).

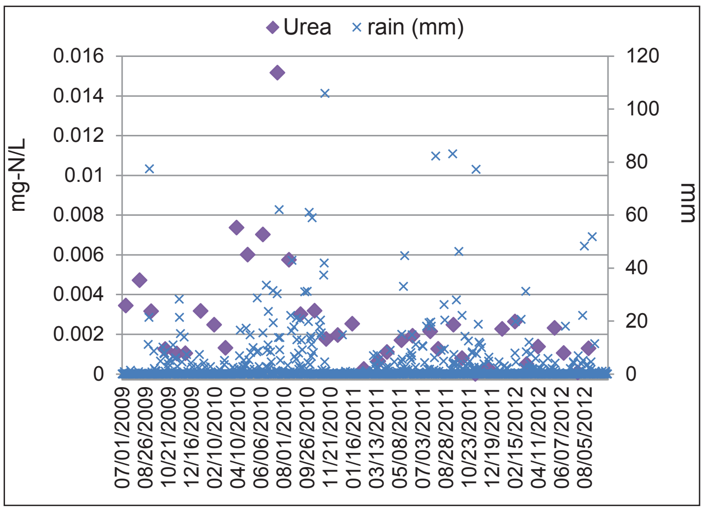

Figure C.15. Time series of precipitation (Lajas) and urea concentrations from N28 (Offshore Central strata).

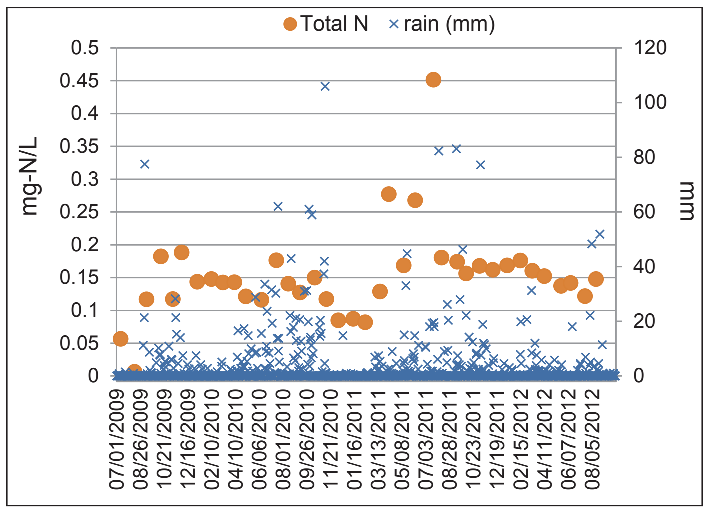

Figure C.16. Time series of precipitation (Lajas) and total nitrogen concentrations from N28 (Offshore Central strata).

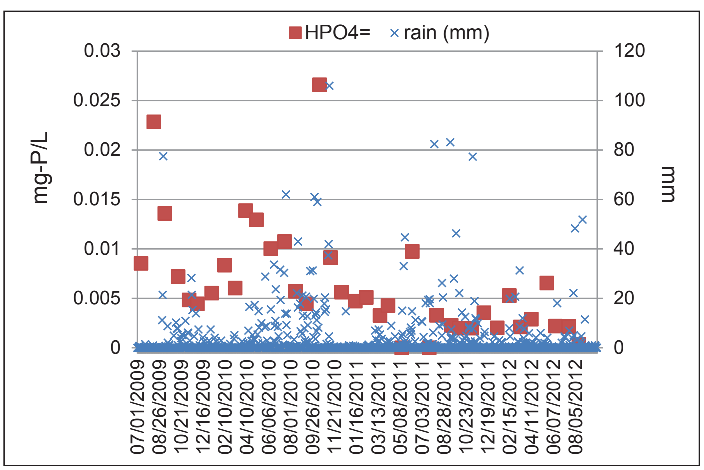

Figure C.17. Time series of precipitation (Lajas) and orthophosphate concentrations from N28 (Offshore Central strata).

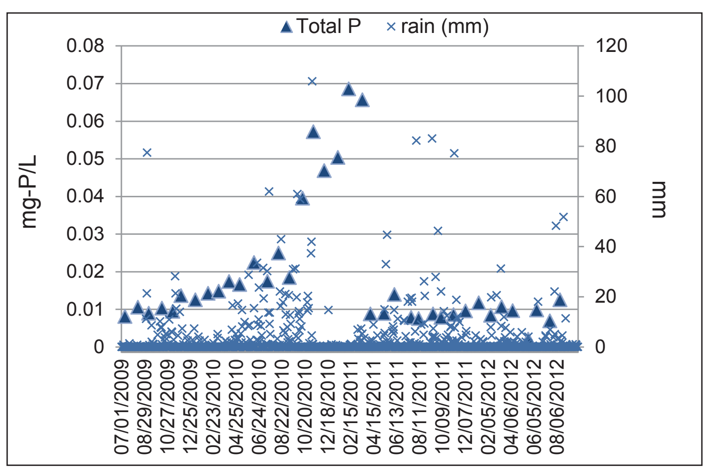

Figure C.18. Time series of precipitation (Lajas) and total phosphorus concentrations from N28 (Offshore Central strata).

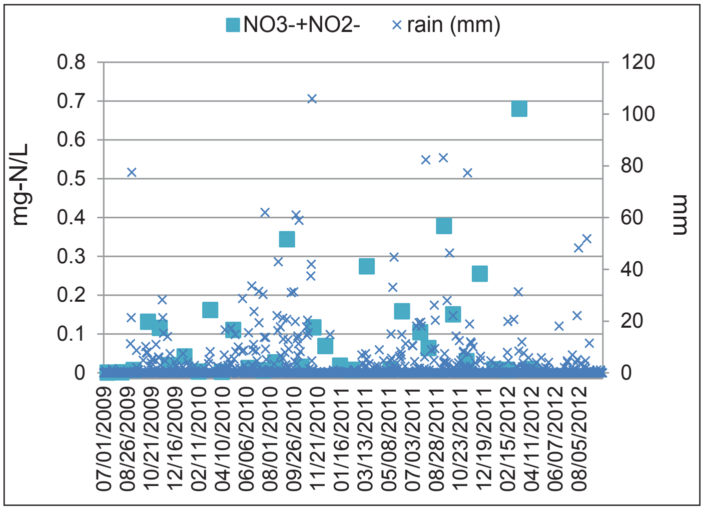

Figure C.19. Time series of precipitation (Lajas) and oxidized nitrogen (nitrate plus nitrite) concentrations from N18 (North Bay strata).

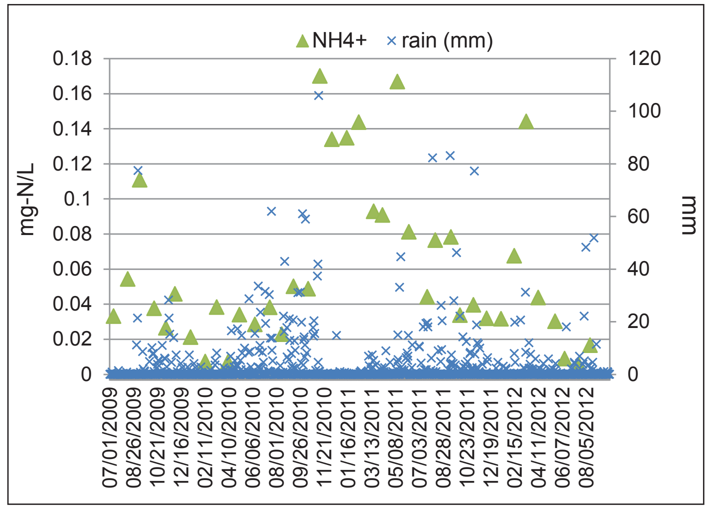

Figure C.20. Time series of precipitation (Lajas) and ammonium concentrations from N18 (North Bay strata).

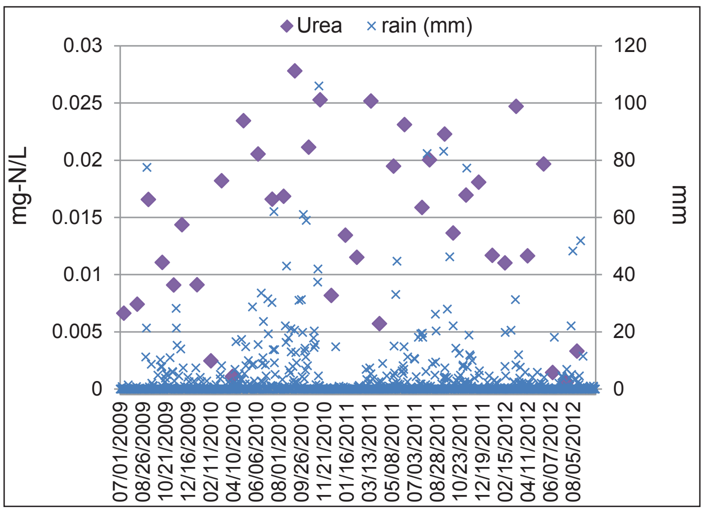

Figure C.21. Time series of precipitation (Lajas) and urea concentrations from N18 (North Bay strata).

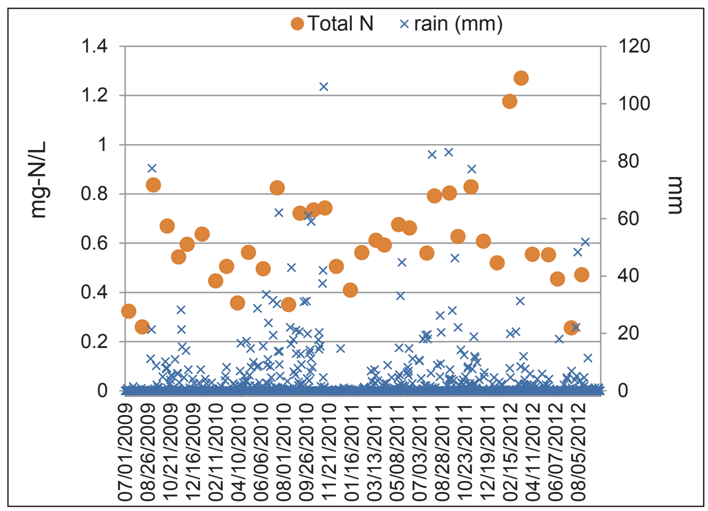

Figure C.22. Time series of precipitation (Lajas) and total nitrogen concentrations from N18 (North Bay strata).

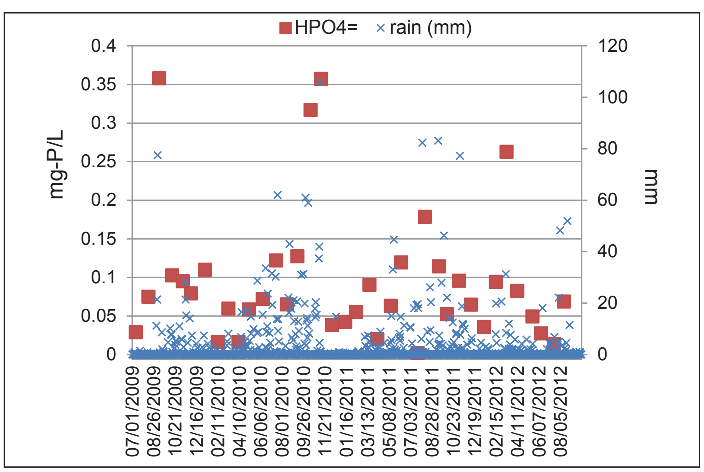

Figure C.23. Time series of precipitation (Lajas) and orthophosphate concentrations from N18 (North Bay strata).

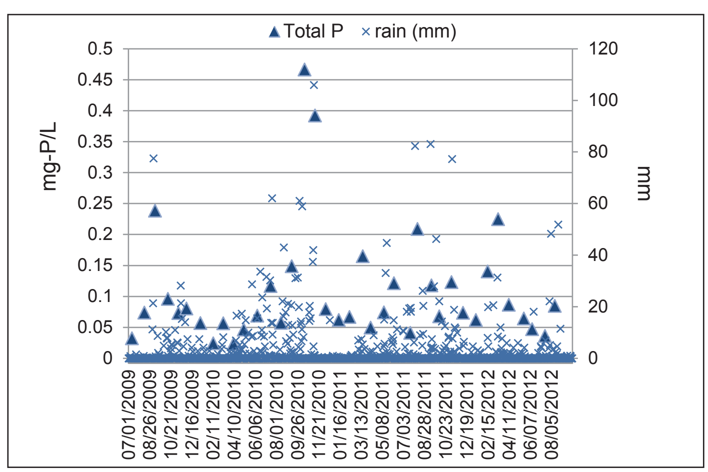

Figure C.24. Time series of precipitation (Lajas) and total phosphorus concentrations from N18 (North Bay strata).

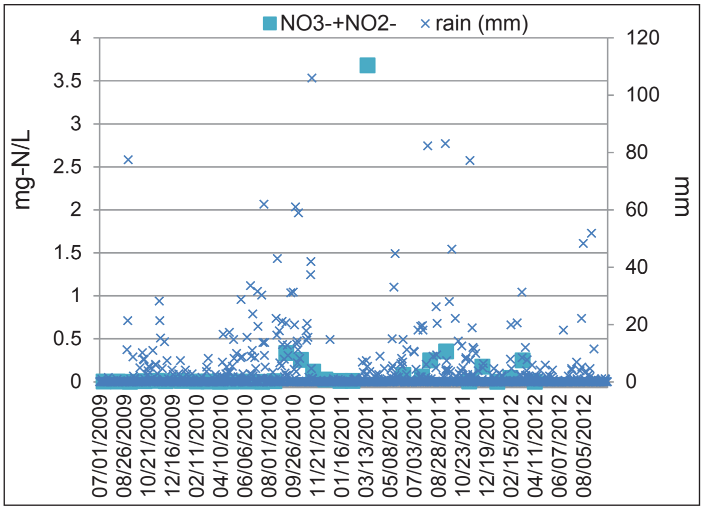

Figure C.25. Time series of precipitation (Lajas) and oxidized nitrogen (nitrate plus nitrite) concentrations from N23 (Central Bay strata).

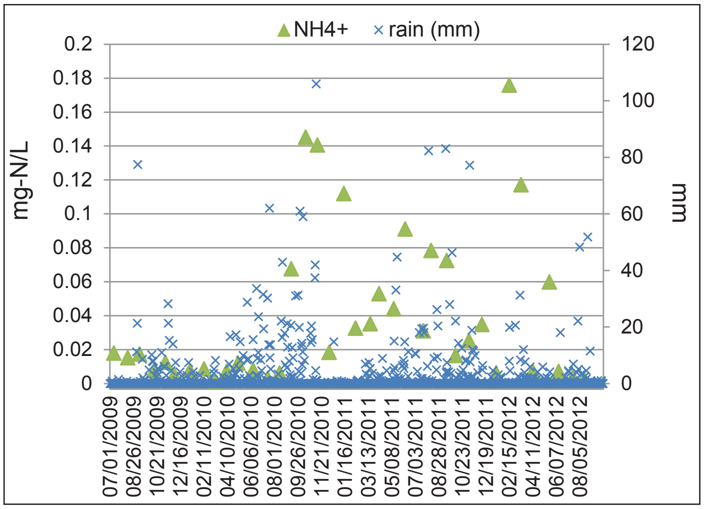

Figure C.26. Time series of precipitation (Lajas) and ammonium concentrations from N23 (Central Bay strata).

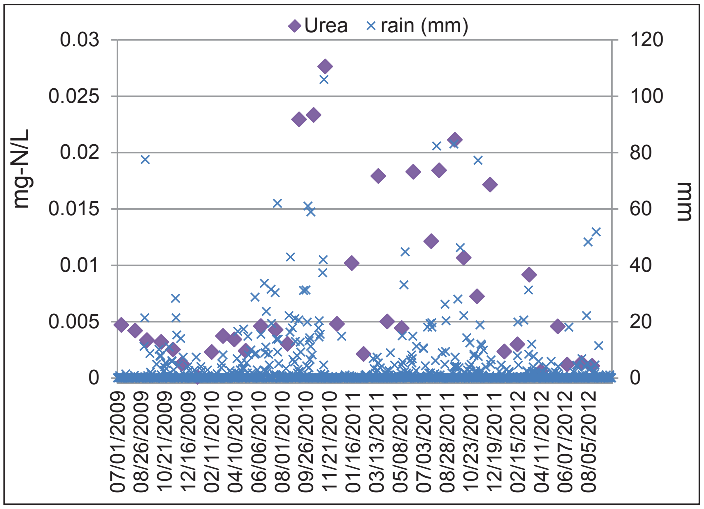

Figure C.27. Time series of precipitation (Lajas) and urea concentrations from N23 (Central Bay strata).

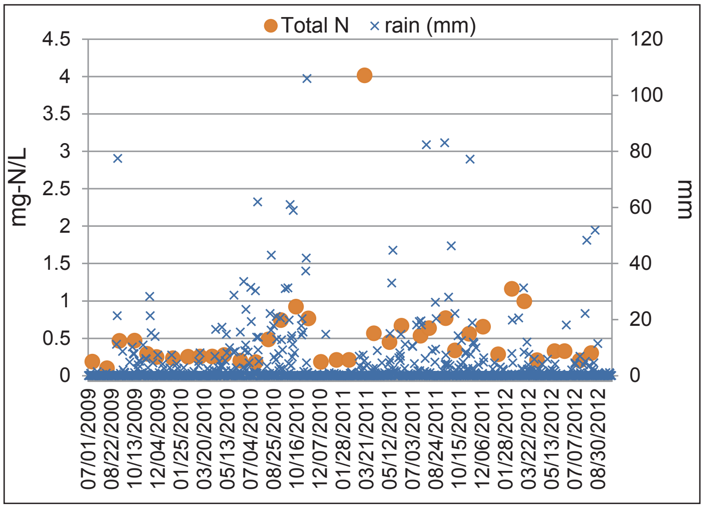

Figure C.28. Time series of precipitation (Lajas) and total nitrogen concentrations from N23 (Central Bay strata).

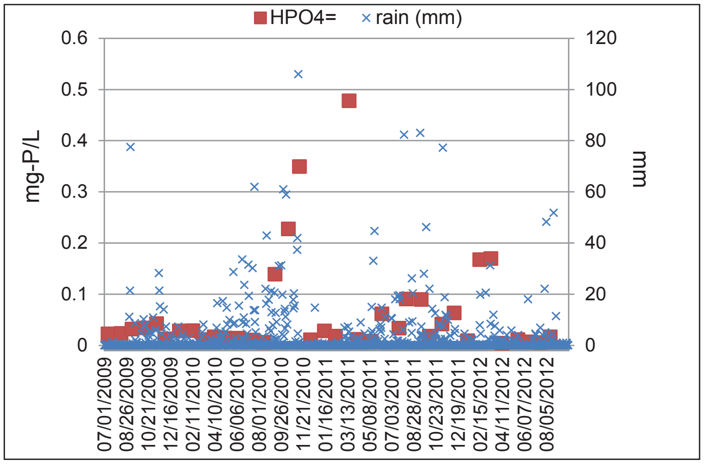

Figure C.29. Time series of precipitation (Lajas) and orthophosphate concentrations from N23 (Central Bay strata).

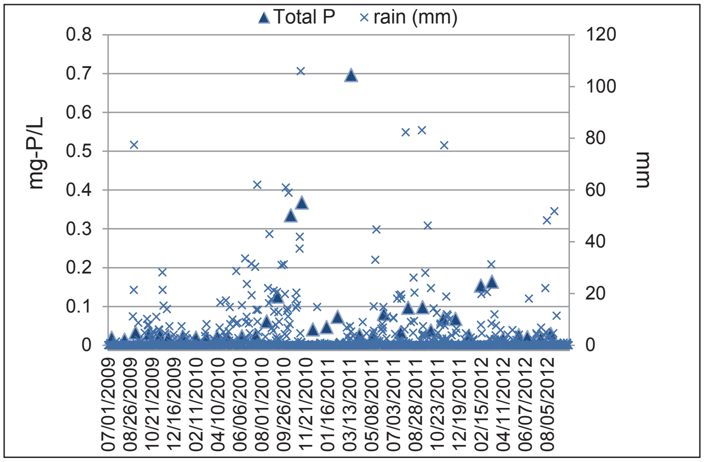

Figure C.30. Time series of precipitation (Lajas) and total phosphorus concentrations from N23 (Central Bay strata).

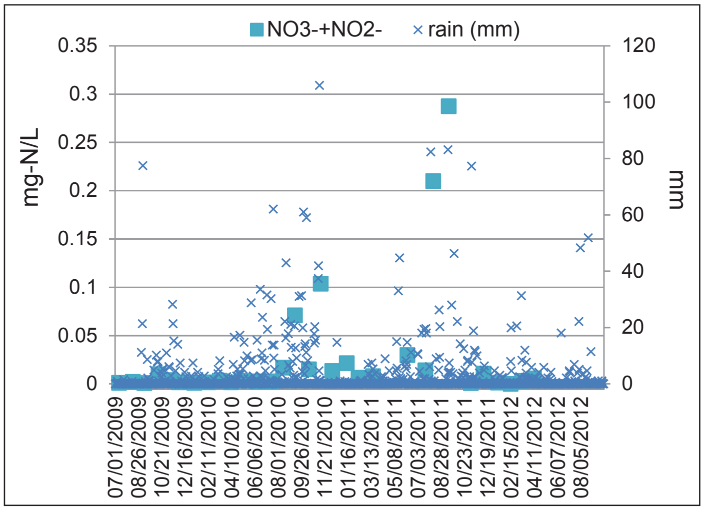

Figure C.31. Time series of precipitation (Lajas) and oxidized nitrogen (nitrate plus nitrite) concentrations from N24 (South Bay strata).

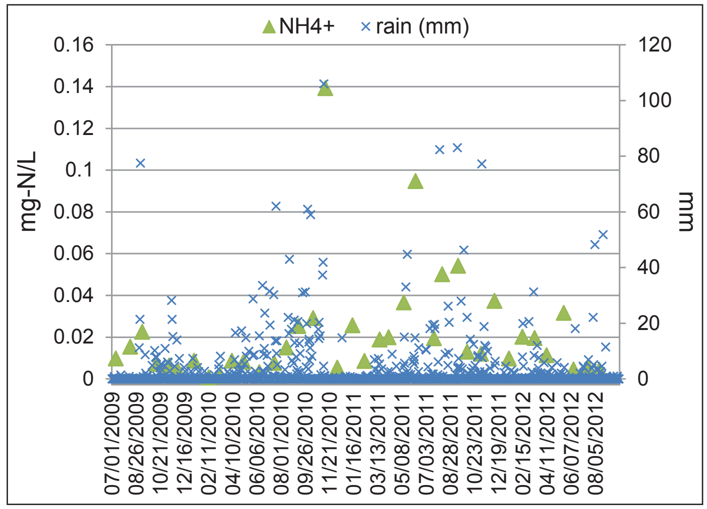

Figure C.32. Time series of precipitation (Lajas) and ammonium concentrations from N24 (South Bay strata).

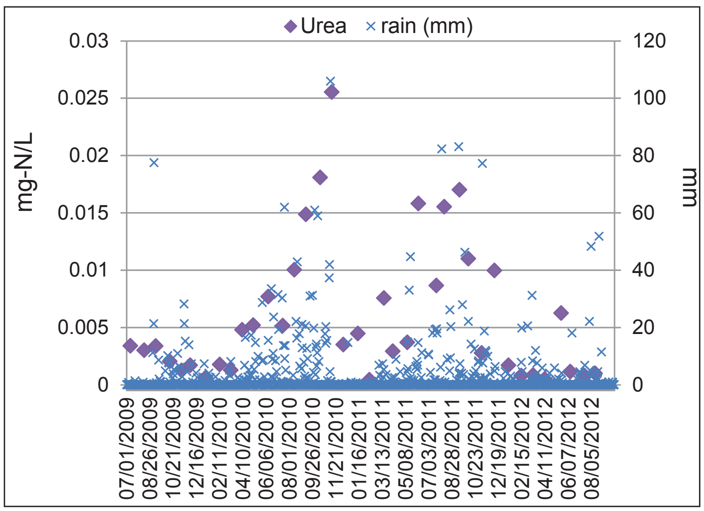

Figure C.33. Time series of precipitation (Lajas) and urea concentrations from N24 (South Bay strata).

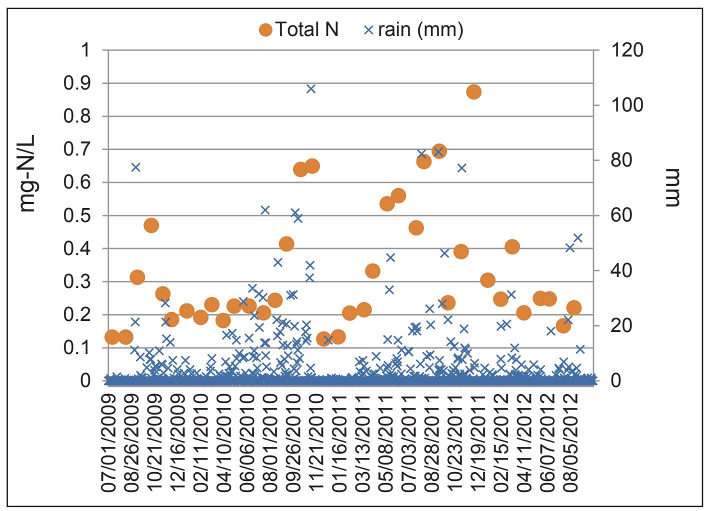

Figure C.34. Time series of precipitation (Lajas) and total nitrogen concentrations from N24 (South Bay strata).

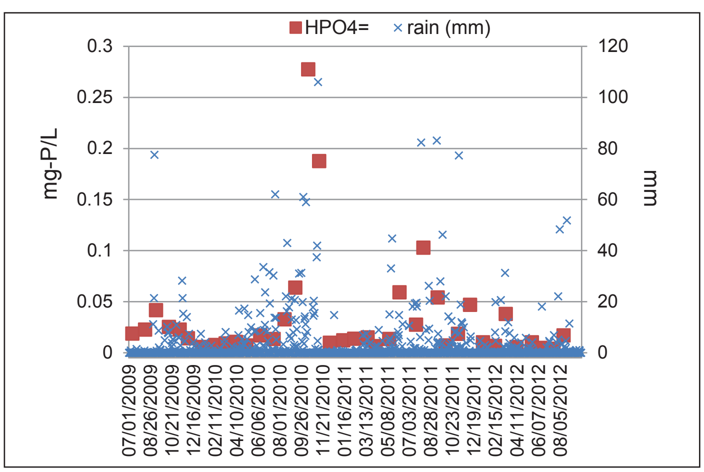

Figure C.35. Time series of precipitation (Lajas) and orthophosphate concentrations from N24 (South Bay strata).

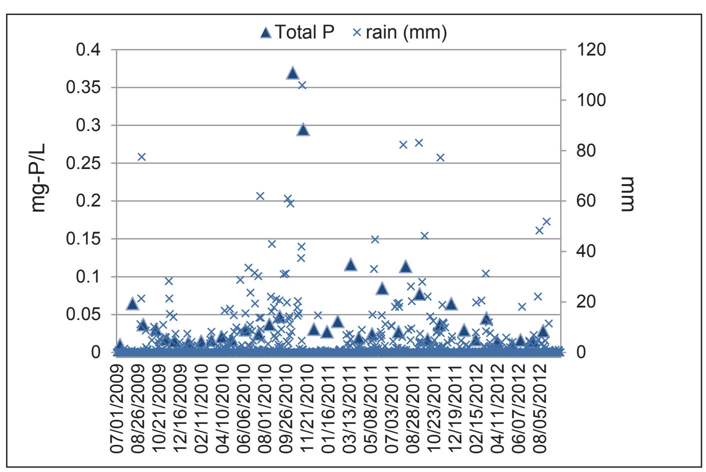

Figure C.36. Time series of precipitation (Lajas) and total phosphorus concentrations from N24 (South Bay strata).

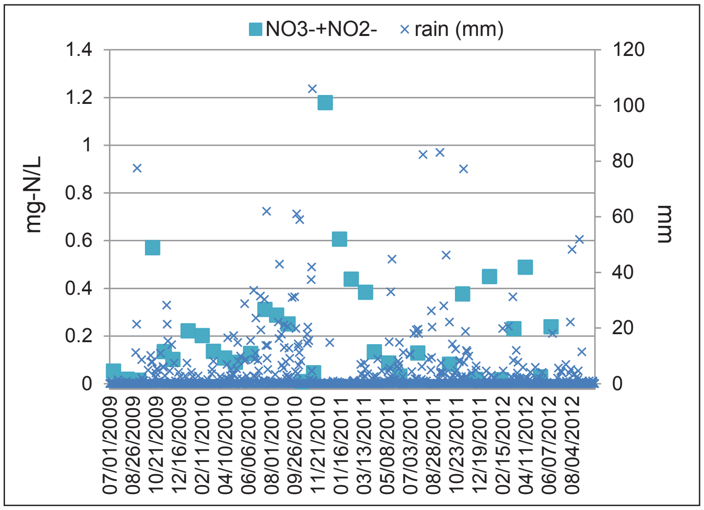

Figure C.37. Time series of precipitation (Lajas) and oxidized nitrogen (nitrate plus nitrite) concentrations from N33 (Watershed Ag Canal strata).

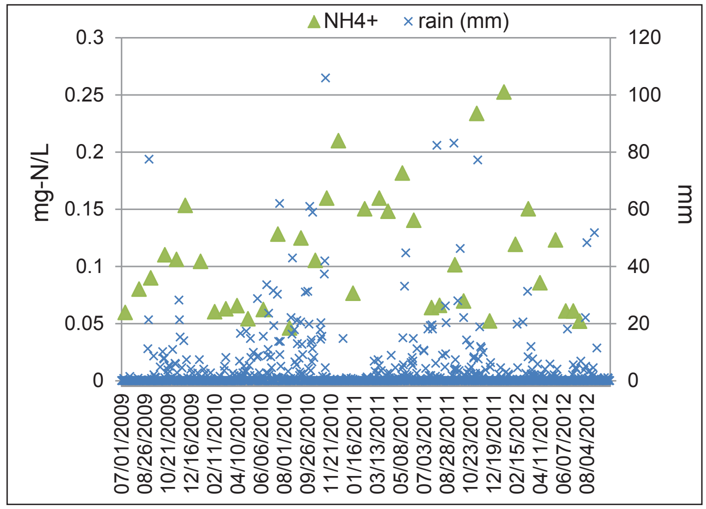

Figure C.38. Time series of precipitation (Lajas) and ammonium concentrations from N33 (Watershed Ag Canal strata).

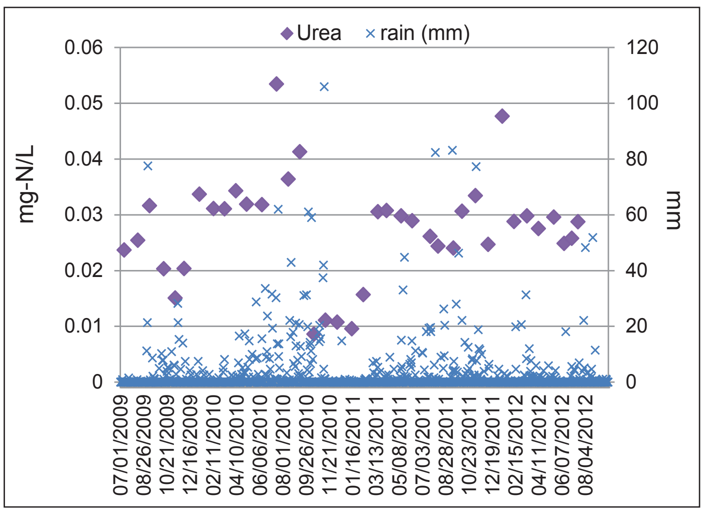

Figure C.39. Time series of precipitation (Lajas) and urea concentrations from N33 (Watershed Ag Canal strata).

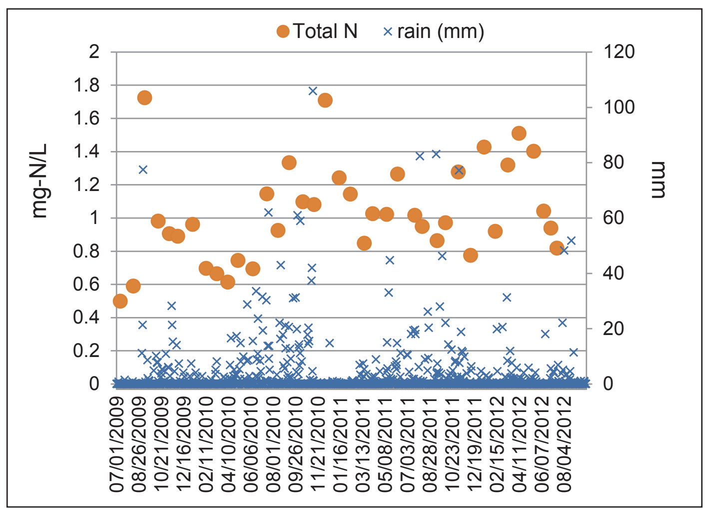

Figure C.40. Time series of precipitation (Lajas) and total nitrogen concentrations from N33 (Watershed Ag Canal strata).

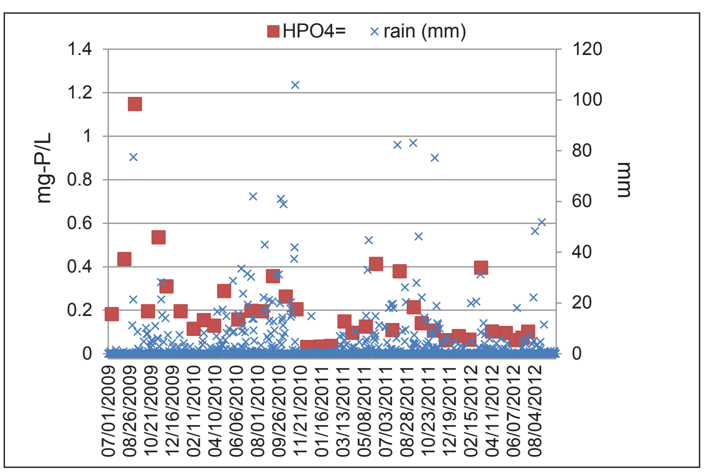

Figure C.41. Time series of precipitation (Lajas) and orthophosphate concentrations from N33 (Watershed Ag Canal strata).

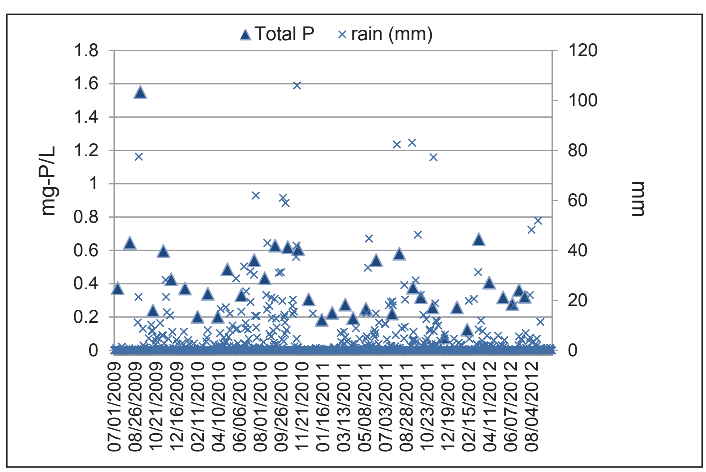

Figure C.42. Time series of precipitation (Lajas) and total phosphorus concentrations from N33 (Watershed Ag Canal strata).

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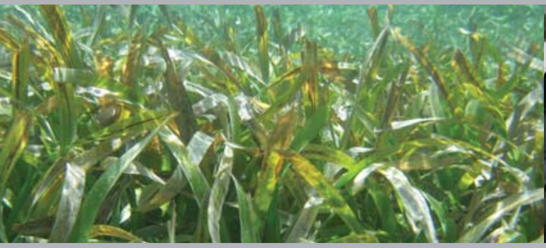

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