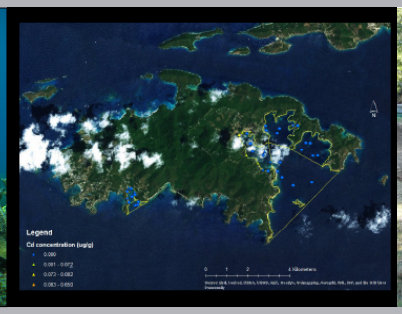
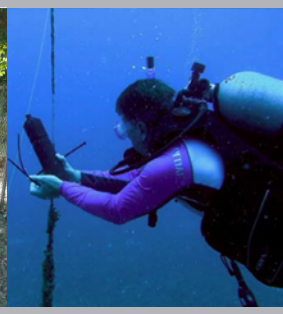
A Baseline Assessment of Coral and Fish Bays (St. John, USVI) in Support of ARRA Watershed Restoration Activities

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

Dave Whitall
Charlie Menza
Ron Hill





February 2014

N O A A T E C H N I C A L M E M O R A N D U M N O S N C C O S 1 7 8

NOAA NCCOS Center for Coastal Monitoring and Assessment

Citation
Whitall, D., C. Menza, and R. Hill. 2014. A Baseline Assessment of Coral and Fish Bays (St. John, USVI) in Support of ARRA Watershed Restoration Activities. NOAA Technical Memorandum NOS NCCOS 178. Silver Spring, MD. 74 pp.

Mention of trade names or commercial products does not constitute endorsement or recommendation for their use by the United States Government.

A Baseline Assessment of Coral and Fish Bays (St. John, USVI) in Support of ARRA Watershed Restoration Activities

Center for Coastal Monitoring and Assessment (CCMA)
National Centers for Coastal Ocean Science (NCCOS)
National Ocean Service (NOS)
National Oceanographic and Atmospheric Adminstration (NOAA)

Authors

David Whitall
NOAA Center for Coastal Monitoring and Assessment
Coastal and Oceanographic Assessment, Status and Trends Branch

Charles Menza
NOAA Center for Coastal Monitoring and Assessment
Biogeography Branch

Ron Hill
NOAA Southeast Fisheries Science Center
Galveston Laboratory

February 2014

NOAA Technical Memorandum NOS NCCOS 178

United States Department of Commerce

Rebecca Blank
Deputy Secretary

National Oceanic and Atmospheric Administration

Kathryn Sullivan
Acting Under Secretary

National Ocean Service
Holly Bamford

Assistant Administrator

About this document

This report provides a baseline environmental assessment of Coral Bay and Fish Bay in St. John, USVI in support of watershed restoration activities on the island. This project was funded by the Caribbean Coral Reef Institute (CCRI), NOAA's Coral Reef Conservation Program (CRCP), NOAA's National Centers for Coastal Ocean Science, Center for Sponsored Coastal Ocean Research (CSCOR), the National Park Service Natural Resource Preservation Program at Virgin Islands National Park, and NPS's South Florida/Caribbean Inventory and Monitoring Program. The historical data synthesized for this work work was collected as part of the National Oceanic and Atmospheric Administration's (NOAA) Caribbean Coral Reef Ecosystem Monitoring (CREM) project; a partnership effort between NOAA's National Centers for Coastal Ocean Science, Center for Coastal Monitoring and Assessment, NOAA Fisheries Southeast Fisheries Science Center (SEFSC), US Virgin Islands Department of Planning and Natural Resources – Division of Fish and Wildlife, US Geological Survey, National Park Service (NPS), the University of the Virgin Islands, and the University of Hawaii.

Related projects include:
Caribbean Coral Reef Ecosystem Monitoring
http://coastalscience.noaa.gov/projects/detail?key=57
Development of Reef Fish Monitoring Protocols to Support the National Park Service Inventory and Monitoring Program
http://coastalscience.noaa.gov/projects/detail?key=72

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

Characterization of Land-Based Sources of Pollution and Effects in the St. Thomas East End Reserves (STEER)
http://coastalscience.noaa.gov/projects/detail?key=19

Live hyperlinks to related products (indicated by blue text) are embedded throughout this report and are accessible when viewing this document as a PDF. For more information about this report and others like it, please visit the NCCOS web site, http://coastalscience.noaa.gov/, or direct comments to:

Dave Whitall, Ph.D.
Center for Coastal Monitoring and Assessment
NOAA National Centers for Coastal Ocean Science
Telephone: 301.713.3020 x160
E-mail: Dave.Whittal@noaa.gov

Table of Contents

Page

1. Background..... 1
2. A Synthesis of Ten Years of Biogeographic Data..... 7
3. Monitoring Coral Reef Changes Associated with Erosion Mitigation Projects in Fish Bay and Coral Bay, St. John, USVI 17
4. Contaminants in Surficial Sediments..... 29
5. Conclusions and Management Applications..... 62

Chapter 1: Introduction

Background

Coral reefs are among the most productive and diverse ecosystems in the world (Bryant et al. 1998), and provide a variety of goods and services ranging from commercial and subsistence fisheries, tourism and recreation, sources of new medicines, to natural protection against storms for communities and ports. The global value of coral reefs has been estimated at $375 billion/year (Costanza et al. 1997). Worldwide, coral reef ecosystems are declining at an alarming rate (Wilkinson 2004, Bellwood et al. 2004, Pandolfi et al. 2005) and the U.S. Virgin Islands (USVI) is no exception (Rogers and Beets 2001, Beets and Rogers 1997, Jeffrey et al. 2005, Rogers et al. 2008). Threats to coral reefs include pollution, disease, sedimentation, overfishing, global climate change, invasive species, ship groundings (Hughes 1994, Waddell et and Clark 2008), and possibly ocean acidification (Kleypas et al. 2006).

Caribbean coral reefs have changed dramatically in the past 40 years, with live coral cover estimated to have declined by 80% (Gardner et al, 2003) and many reefs exhibiting a new ecosystem steady-state dominated by macroalgae. To reduce stressors and improve the health of coral reefs, coastal managers in the USVI have implemented marine protected areas, fishing restrictions, reef monitoring programs, and land use management strategies (Waddell and Clarke 2008).

In 2010, over $2.7 million in NOAA Coastal and Marine Habitat Restoration funding under the American Recovery and Reinvestment Act of 2009 was awarded to the Virgin Islands Conservation and Development Council to conduct targeted watershed stabilization actions (www.recovery.gov; award 39857). This work complements the Coral Bay Community Council's (CBCC) Coral Bay Watershed Management Project and utilizes designs developed under CBCC's EPA CARE grant. The overarching themes of these projects were to improve coastal ecosystem condition in Coral Bay and Fish Bay, St. John through an immediate and long-term reduction in sediment loading to the bay, and to stimulate the local economy through the creation of jobs and infrastructure improvements. The main goal of the two-year USVI Coastal Habitat Restoration through Watershed Stabilization project is to reduce sediment loading rates into the coastal waters of three USVI watersheds (East End Bay on St. Croix and Coral & Fish Bays on St. John), by approximately 100 tons, by implementing erosion & sediment control practices to improve portions of foot trails and unpaved roads in each of the three sites.

Sedimentation from runoff is one of the biggest potential sources of reef degradation in the Caribbean (Dahl 1985, Rogers 1985, Rogers 1990). There is ample evidence to show terrestrial runoff and corollary increases of sedimentation, nutrient enrichment and turbidity in the water column can degrade coral reefs at local scales through impacts on coral growth and survival, reproduction and recruitment, and population interactions (Rogers 1990, many examples in Fabricius 2005). However, it has been difficult to show a direct link between increasing terrestrial runoff and reef degradation at regional scales, due to confounding disturbances, lack of historical data and natural variation (Fabricius 2005).

In addition to terrestrial runoff, there are other interrelated threats that may further contribute to declines in reef health. Around St. John, coral cover has declined over time due to hurricanes (Rogers et al. 1991), anchor damage (Rogers and Garrison 2001), disease (Miller et al. 2006, 2009), and loss of herbivores due to overfishing and disease (Rogers et al. 2008). Further, coral diseases and bleaching have played a major role in the degradation of coral reefs in the Caribbean, including those in the USVI. Although all of the diseases currently known from the Caribbean are found in the USVI, white plague and band diseases have had the most severe impact on the coral community (Rogers et al. 2008).

Long-term and consistent monitoring data is needed to critically assess the impacts of runoff and the differential effects from additional impacts (e.g. fishing, hurricanes, bleaching, anchoring). One of the greatest justifications for consistent monitoring is to differentiate natural fluctuations from human stresses. The magnitude and periodicity of disturbances

greatly affect the spatio-temporal patterns observed on coral reefs (Done et al. 1991, Connell 1997) and trajectories of these trends are determined by the synergistic effects of local and regional processes (Connell 1997). Monitoring needs to be conducted over time scales commensurate with the periodicity of these disturbance events in order to fully capture the impacts and changes over time.

This report provides a suite of data and analyses which will serve as an environmental baseline against which to measure future change in the ecosystem, including improvements resulting from watershed restoration activities. The suite includes:

1) An analysis of data acquired over ten years by the National Centers for Coastal Ocean Science in Coral Bay and Fish Bay on the island of St. John before watershed stabilization measures were undertaken in 2011. This data, collected as part of the NOAA's Caribbean Coral Reef Ecosystem Monitoring (CREM), will serve as an ecological baseline to assess the efficacy of watershed improvements on coral reefs within these bays. All CREM data presented here is available online free of charge at: http://www8.nos.noaa.gov/biogeo_public/query_main.aspx.
2) Contaminant analysis of surficial sediment samples in order to determine the pollutant stressors associated with sedimentation and land based sources of pollution. The contaminant levels are compared with other studies in the region, and against previously published sediment quality guidelines to put the levels of contamination in perspective. These data are available online free of charge at: http://egisws02.nos.noaa.gov/nsandt/
3) Measures of biological parameters expected to respond to changes in water quality from reductions in sediment inflow and runoff. The parameters, measured across a distance gradient from inshore sources of runoff, include relative abundance and biomass of macroalgae, composition and cover of scleractinian corals and other benthic organisms, as well as distribution and abundance of fish and macroinvertebrates.
4) Measures of water quality (e.g., nutrient levels, salinity, chlorophyll, temperature). Sample collection and analysis are on-going to characterize Fish, Coral and Lameshur Bays, St. John; work has been conducted by partners at the Univ. of the Virgin Islands, cooperatively with EPA and USVI Division of Environmental Protection. Data are highlighted briefly in this report but full results will be available separately. As with other baseline characterizations, these results, plus results of sedimentation studies, will be critical to future analyses of environmental conditions and an understanding of the biological findings of this work.

It is important to understand that this baseline represents a snap shot of the ecosystem at the time of the assessment, and is not indicative of the status of the system prior to human impacts.

Study sites

Virgin Islands National Park (VIIS) was established in 1956 to protect significant marine and terrestrial resources on the island of St. John (Figure 1.1). Submerged lands were added to the park in 1962 to further protect and preserve coral reefs and seascapes. The park consists of almost 30 square kilometers including approximately 2/3 of the island of St. John. The need to protect reefs from further degradation led to a Presidential Proclamation establishing the Virgin Islands Coral Reef National Monument (VICR) in January 2001, which designated approximately 50 square kilometers of federally owned submerged lands to be protected.

*Figure 1.1: Seargent Major (*Abudefduf saxatilis*) fish near a coral head (Photo credit: Baltimore Sun)*

The island of St. John is mostly covered by second generation forest growth. Almost the entire island was clear-cut to make way for sugar cane production during the colonial era and this had dramatic impacts to hydrology and soil composition of the island. Most of the vegetation on St. John today consists of native and nonnative species competing for space.

The seascape surrounding the island of St. John consists of coral reefs, seagrass beds, algal meadows and sand plains. It too has seen dramatic changes caused by anthropogenic and natural disturbances. Intensive fishing has caused the loss of several spawning aggregations, as well as severe declines in size and abundance of important fish species (Beets & Friedlander 1999; Beets & Rogers 2002). In addition to the effects of fishing, habitat degradation in the form of coral and seagrass habitat loss due to hurricanes and coral diseases has led to an ecosystem that is now dominated by macroalgae (Beets & Rogers 2002). VIIS and VICR were established in part to protect the coral reef ecosystem from anthropogenic disturbances.

From 2001 to 2011, the National Centers for Coastal Ocean Science monitored the coral reef ecosystem with annual surveys of fish and benthic habitats. Over 100 sites were visited every year and measurements of fish, corals, benthic habitats, and invertebrates were taken. Similar monitoring was conducted by NCCOS in Puerto Rico and St Croix, USVI. These data have been used to characterize the coral reef ecosystem over time and to assess protection measures. All of the data is freely available online.

Coral Bay

Coral Bay is a large bay on the eastern side of St. John. It is a specific region of management concern for St. John due to its proximity to human populations and unique physical and biological attributes. Different portions of the bay are within the limits of VIIS and VICR. Some areas are not within park boundaries.

Coral Bay is 13.3 km^2 and encompasses over 16 km of shoreline, including some of St. John's largest salt ponds, extensive mangrove habitat, sea grass beds and fringing reefs. The bay links to the largest catchment area on St. John draining into an individual bay. Coral Bay supports protected *Acropora* corals and sea turtle nesting areas and may be an important juvenile habitat for several commercially important fisheries species such as yellowtail snapper, schoolmaster snapper, and several species of parrotfishes (Friedlander et al in press, STJ tech report)

The watershed adjacent to Coral Bay is characterized by steep slopes (averaging 18%, with a large percentage over 35%), highly erodible soils, and high runoff volumes associated with average rain events. These factors, combined with a large percentage of dirt roads, active construction, and no existing storm water management, have been shown to contribute to excessive sediment loading to the bay (Devine et al. 2003, Ramos-Sharron 2005, Brooks et al, 2007). In addition, the watershed experienced an approximate 80% population increase between 1990 and 2000, making it the fastest growing area in the USVI. The population of the watershed is approximately 1200 people, with about half of the houses being vacation rental properties (CBCC 2012). Much is this development was done in the absence of infrastructure planning, leading to many unpaved, poorly maintained roads (CBCC, 2012). Erosion from these roads leads to not only sediments reaching the coastal system (via the ghuts), but also potentially pollutants associated with those sediments (e.g. PAHs, metals, pesticides). The Coral Bay Community Council, Inc. (CBCC), a local nonprofit watershed management association, identified erosion and bay sedimentation as priority issues threatening both marine ecosystem health and the community's quality of life.

Fish Bay

Fish Bay is a small bay on the south shore of St. John. Although significantly smaller than Coral Bay, Fish Bay is also an area of management concern and was identified as a high management priority by coral reef managers to achieve stable, sustainable coral reef ecosystems (USVI and CRCP 2010). Fish Bay represents a significant land surface area draining into an individual bay on St. John and includes extensive mangrove habitat, seagrass beds and coral reefs. A portion of the bay is within the limits of VIIS. The watershed adjacent to Fish Bay is characterized by steep slopes, highly erodible soils, and high runoff volumes associated with average rain events.

Goal

In this report, we provide baseline biological and pollutant data for Fish Bay and Coral Bay (St. John, USVI). These data will support future studies assessing the impact of new watershed stabilization improvements on erosion rates and the health of local coral reefs.

Chapter 2: A Synthesis of Ten Years of Biogeographic Data

Charles Menza*, Lisa Wedding, Randy Clark, Chris Caldow and Mark Monaco
National Oceanographic and Atmospheric Administration
National Ocean Service
National Centers for Coastal Ocean Science
Center for Coastal Monitoring and Assessment

EXECUTIVE SUMMARY

This report provides baseline biological data on fishes, corals and habitats in Coral and Fish Bays, St. John, USVI. A similar report with data on nutrients and contaminants in the same bays is planned to be completed in 2013.

Data from NOAA's long-term Caribbean Coral Reef Ecosystem Monitoring program was compiled to provide a baseline assessment of corals, fishes and habitats from 2001 to 2010, data needed to assess the impacts of erosion control projects installed from 2010 to 2011. The baseline data supplement other information collected as part of the USVI Watershed Stabilization Project, a project funded by the American Recovery and Reinvestment Act of 2009 and distributed through the NOAA Restoration Center, but uses data which is not within the scope of ARRA funded work.

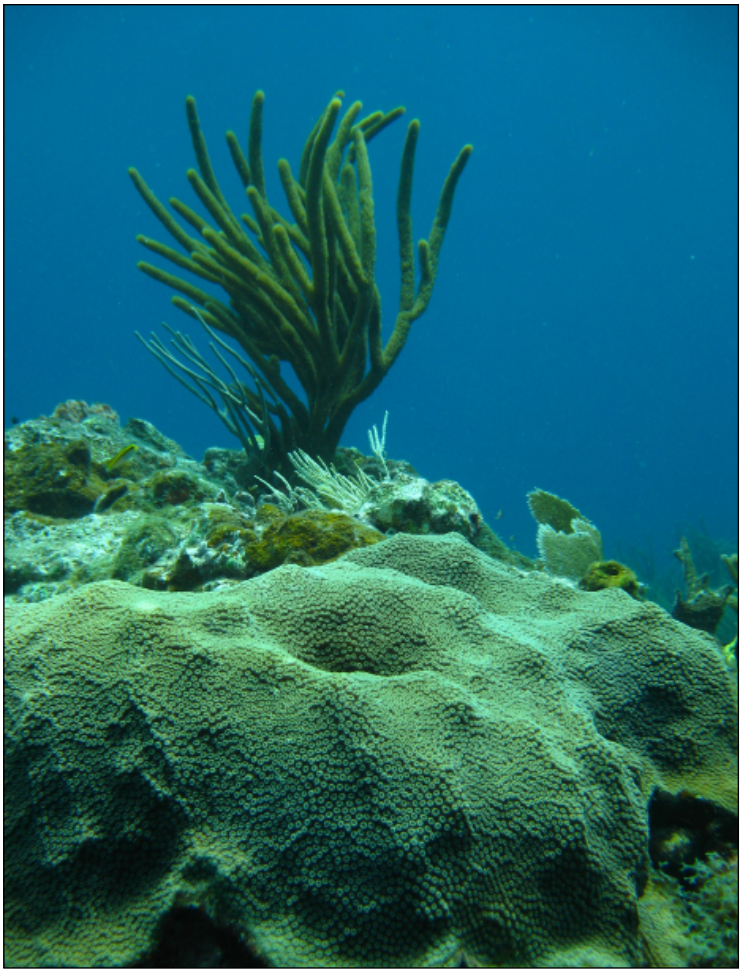


Figure 1. An image of a healthy Caribbean coral reef. Photo credit: NCCOS/CCMA/Biogeography Branch

We present data on 16 ecological indicators of fishes, corals and habitats. These indicators were chosen because of their sensitivity to changes in water quality noted in the scientific

literature (e.g., Rogers 1990, Larsen and Webb 2009). We report long-term averages and corresponding standard errors, plot annual averages, map indicator values and list inventories of coral and fish species identified among surveys. Similar data will be needed in the future to make rigorous comparisons and determine the magnitude of any impacts from watershed stabilization.

Over the course of ten years 30 distinct species of coral and 194 species of fish were observed in Coral Bay, and 21 species of coral and 84 species of fish were observed in Fish Bay. As is common in most coral reefs in the USVI, algae cover was one of the most abundant taxonomic components of the benthic communities in both bays and live coral cover was generally low. Plots of indicators over time and maps of indicators across space showed variation among fishes, corals and habitats across a range of spatial and temporal scales. Although many reefs in both bays were dominated by algae, pockets of coral refuges with very high coral cover were identified in Coral Bay and sites with healthy seagrass beds were found in Fish Bay (Figure 1). These observed patterns in the spatial occurrence and abundance of algae, live coral, and reef fishes were similar with and reflected the broader-scale spatial patterns observed by Friedlander et al. (2012) around the island of St. John.

The CREMP monitoring dataset investigated in this report offers a dataset which is well distributed in space and time for management strata similar in size to Coral Bay. For smaller strata like Fish Bay the monitoring program offered too little information to prepare rigorous baseline data. More data will be needed within Fish Bay in the future to better understand spatio-temporal heterogeneity.

Figure 2. Sediment plumes along coasts adjacent to reefs can introduce nutrients, toxins, pathogens, and sediment onto reefs, smothering and otherwise damaging reef ecosystems (left). Photo credit: NOAA Restoration Center. Sediment plume initiating from the mouth of a river in Puerto Rico enters a bay, impacting local reefs (right). Photo credit: Dave Burdick, http://coralreef.noaa.gov/threats/pollution/

Accurately defining the spatial and temporal heterogeneity in these bays is critical to assessing the baseline status among corals, fishes and habitats, and detecting any future impacts to coral reef communities resulting from watershed improvements. The variability described in this report underscores the need for sampling replicates throughout each bay and over time. These data can effectively be used in a BACI (Before, After, Control, Impact) design to conclusively assess impacts. Continued monitoring of Coral and Fish Bays will be needed to provide sufficient data to detect and understand changes among coral reefs.

BACKGROUND

Coral reefs are among the most productive and diverse ecosystems in the world (Bryant et al. 1998), and provide a variety of goods and services ranging from commercial and subsistence fisheries, tourism and recreation, sources of new medicines, to natural coastal protection against storms. Worldwide, coral reef ecosystems are declining at an alarming rate (Wilkinson 2004, Bellwood et al. 2004, Pandolfi et al. 2005) and the U.S. Virgin Islands (USVI) is no exception (Rogers and Beets 2001, Jeffrey et al. 2005, Rogers et al. 2008a, 2008b).

Caribbean coral reefs have changed dramatically in the past 40 years, with live coral cover estimated to have declined by 80% and many reefs exhibiting a new ecosystem steady-state dominated by macroalgae (Hughes 1994, Gardner et al. 2003). Numerous natural and anthropogenic stressors such as coral diseases (Miller et al. 2009), hurricanes (Rogers et al. 1991), anchor damage (Rogers and Garrison 2001), and loss of herbivores due to overfishing (Rogers et al. 2008a) have negatively impacted coral reefs. To reduce stressors and improve the health of coral reefs, coastal managers in the USVI have implemented marine protected areas, fishing restrictions, reef monitoring programs, and land use management strategies (see Waddell and Clarke 2008 for compilation).

In 2010, over $2.7 million was awarded through the American Recovery and Reinvestment Act (ARRA) of 2009 (www.recovery.gov; award 39857) and distributed through the NOAA Restoration Center to conduct targeted watershed stabilization projects in the USVI and reduce terrestrial runoff. Sediments in the water column resulting from terrestrial runoff are a key cause of coral reef degradation among many local reefs (Figure 2; Dahl 1985, Rogers 1990). There is ample evidence to show that terrestrial runoff and related increases in nearshore sedimentation, nutrient enrichment and turbidity can degrade coral reefs by impacting coral growth and survival, reproduction and recruitment, and population interactions (Rogers 1990, Richmond 1993, Fabricius 2005).

The USVI watershed stabilization project, funded by ARRA and led by the Virgin Islands Resource Conservation and Development Council (VIRCDC), has reduced sediment runoff through road stabilization and native plantings, and the reef monitoring sites were implemented to evaluate sediment reduction measures in Coral and Fish Bays on St. John. The project was designed using watershed management plans developed by the VIRCDC, the Virgin Islands Department of Planning and Natural Resources and other partners, and included community outreach and education. To assess the effectiveness of watershed stabilization, the project included plans to monitor sediment runoff (e.g., Ramos-Scharrón 2012), as well as ecological conditions in receiving bays.

This report provides supplemental information to the VIRCDC scope of work by presenting and analyzing ten years of monitoring data collected in Fish and Coral Bays, St. John, USVI (Figure 3). Most of the used data were collected as part of the NOAA's Caribbean Coral Reef Ecosystem Monitoring program (CREMP), and provide a critical long-term dataset before erosion control measures were completed in 2011. For example, as management actions are implemented to reduce sediment and other contaminant loads entering Coral Bay and Fish Bay, the rigorous assessment of fishes, corals and benthic habitats presented in this report, will provide a baseline against which future changes in benthic composition, habitats, reef fish assemblages, and coral community structure could be measured, and ultimately correlated with watershed improvements.

METHODS

Field survey methods

NOAA's Caribbean Coral Reef Ecosystem Monitoring program (CREMP) monitored coral reefs around the island of St. John using underwater visual surveys (Menza et al. 2006). Surveys began in 2001 and were conducted annually in the month of July. Divers or snorkelers surveyed fish, coral and benthic habitat along a 25 m long × 4 m wide belt transect (Figure 6). Fish were identified, counted, and sized. Benthic habitat measurements included: habitat type, physical (e.g., sand, rubble, reef) and biological (e.g., algae, coral, seagrass) cover, rugosity, and depth. Survey sites were selected using a stratified random sampling design incorporating hard and soft benthic habitat type strata derived from NOAA's nearshore benthic habitat maps (Kendall et al. 2001), and two management strata; inside VICR and outside VICR. Comprehensive descriptions of the measurement methods for both fishes and benthic habitats and the sampling design are available online at: http://ccma.nos.noaa.gov/ecosystems/coralreef/reef_fish/protocols.aspx.

Two distinct methodologies were used to acquire benthic information in Coral Bay: a compre-

hensive habitat assessment to evaluate the reef ecosystem and a rapid habitat assessment to measure the efficacy of marine protected areas. The comprehensive habitat assessment (CHA) used quadrats to increase precision of measurements and collected detailed species-level information. Alternatively, the rapid habitat assessment (RHA) collected data using visual estimation for the whole transect, and measured benthos in broad taxonomic categories (i.e., algae, stony coral, gorgonians, sponges, etc.). The comprehensive assessment was conducted around the entire island of St. John on hardbottom and softbottom habitats, while the RHA was implemented inside and adjacent to VICR and only on hardbottom sites. The RHA was implemented after several years of using the comprehensive assessment in order to decrease bottom time and increase the number of dives a survey team could perform during a sampling mission. Fish Bay was sampled only using the CHA and fishes were surveyed using the same protocol independent of the benthic methodology.

Although CREMP has surveyed over 1,000 sites while monitoring the coral reef ecosystem around St. John, this report examines only sites within the limits of Fish Bay and Coral Bay (Figures 4 and 5). Since 2001, 530 and 19 surveys have been conducted in Coral and Fish Bays, respectively. There are fewer surveys conducted in Fish Bay, because it is a much smaller bay. The 19 sites allocated in Fish Bay include five sites surveyed by NCCOS in 2010 which were outside of the normal monitoring program and were added for this report.

Data Analysis

This report focuses on 16 ecological indicators to assess the status of fishes, corals and benthic habitats (Table 1). These indicators were chosen because of their potential sensitivity to changes in water quality noted in the scientific literature (Rogers 1990, Larsen and Webb 2009). For each indicator we present long-term averages, corresponding standard errors and the probability of occurrence. We also plot annual averages, map indicator values and list inventories of all coral and fish species identified among surveys.

Percent cover estimates reflect the amount of a benthic component detected on a transect, and probability of detection is the probability a taxa or habitat type is detected on a transect. Species richness is defined as the number of species in a specific taxonomic category (i.e., corals, fish). Species diversity refers to the Shannon diversity index. Branching corals include: *Acropora cervicornis*, *Acropora palmata*, *Dendrogyra cylindrus*, *Porites porites*, *Madracis formosa*, *Madracis decactis*, and *Madracis mirabilis*. Groupers include species in the genera: Mycteroperca, Cephalopholis and Epinephelus. Total live coral cover estimates are taken from summing hard (stony), soft (octocoral) and hydrocoral taxa.

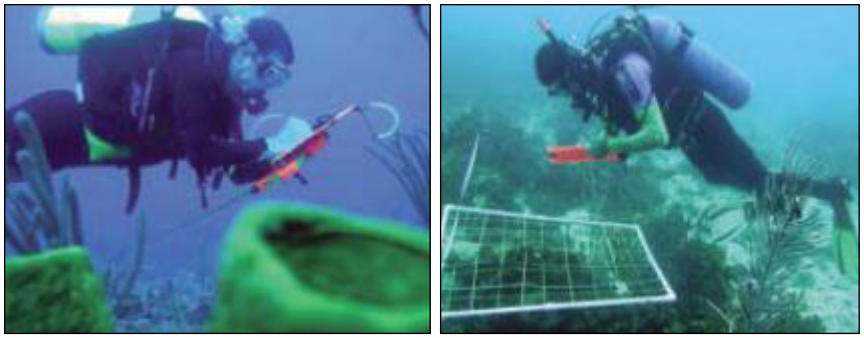


Figure 6. NOAA trained observers recording fish species abundance and body length along a timed belt transect (left); and benthic habitat composition recorded within a quadrat (right).

Analyzed data were extracted from the CREMP database by selecting sites within the limits of Coral Bay and Fish Bay (see Figures 4 and 5 for extracted sites in each bay). Summary statistics for benthic metrics were derived separately for transects using the comprehensive habitat assessment stratified by habitat and using the rapid habitat assessment. In Coral Bay, CHA and RHA habitat and coral data were not merged due to irreconcilable differences in precision, species detection and type of data collected. In Fish Bay, only CHA data was collected. Fish data collected among all sites for each bay were merged since the same protocol was used to assess the fish community. Habitat and fish data collected before 2003 were omitted from assessments of the long-term average and annual variation because of low sampling effort and changes to the sampling design. Measurements of coral bleaching began in 2006 and consequently only 2007-2010 data are presented. Species inventories reflect all species observed from 2001 to 2010. All summary statistics were calculated using JMP (SAS Inc., v. 9.0.0).

Interpolations for fish, corals, benthic cover, and other site specific data were accomplished using the ArcGIS 10.0 Spatial Analyst extension interpolation tool. The method utilized was Inverse Distance Weighted (IDW). IDW assumes that each measured point has a local influence that diminishes with distance and weights the points closer to the prediction location greater than those farther away. Barrier polygons representing the boundaries of study regions of Coral Bay and Fish Bay were used to limit the interpolations to areas where monitoring was conducted. IDW is a useful tool to visualize general spatial patterns in point data, but should be used with a full understanding that it only incorporates spatial correlation among sample locations, does not incorporate anisotropy, and is not an exact interpolator.

Table 1. Coral reef indicators used to report the baseline condition of corals, fishes and habitats in Fish and Coral Bays, St. John, USVI.

Coral Reef Indicators	
% Hard coral cover	Counts of All Fish
% Soft coral cover	Counts of Grunts
% Hydrocorals cover	Counts of Snappers
% Bleached coral cover	Counts of Parrotfish
% Branching coral cover	Counts of Groupers
Coral richness	Diversity of All Fish
% Algae cover	Biomass of All Fish
% Seagrass cover	Number of Fish Species

Table 2. Long-term averages for coral reef indicators in Coral and Fish Bays, St. John, USVI. Averages are provided separately for each habitat survey and bottom type combination. CHA = comprehensive habitat assessment. RHA = rapid habitat assessment. The long-term mean, standard error and probability of detection are represented by x, SE and P, respectively. The (-) symbol indicates the indicator was not measured or was not detected in sufficient quantities to make a valid estimate.

INDICATOR	Coral Bay									Fish Bay					
	CHA Softbottom Sites			CHA Hardbottom Sites			RHA Sites			CHA Softbottom Sites			CHA Hardbottom Sites		
	x	SE	P	x	SE	P	x	SE	P	x	SE	P	x	SE	P
% Hard coral cover	2.15	0.88	0.55	5.25	1.59	0.80	5.92	0.91	0.98	0	N/A	0.00	1.15	0.39	0.47
% Soft coral cover	1.36	0.56	0.49	3.00	0.86	0.96	11.9	1.73	0.93	0	N/A	0.00	1.02	0.40	0.42
% Hydrocorals cover	0.11	0.05	0.39	0.22	0.08	0.98	-	-	-	0	N/A	0.00	0.11	0.05	0.32
% Bleached coral cover	0.07	0.04	0.09	0.18	0.08	0.20	-	-	-	0	N/A	0.00	0.01	0.01	0.16
% Branching coral cover	0.14	0.06	0.34	0.43	0.12	0.68	-	-	-	0	N/A	0.00	0.06	0.03	0.21
Coral species richness	9.67	1.69	0.55	11.2	1.41	0.98	-	-	-	0	N/A	0.00	4.10	1.22	0.47
% Algae cover	37.4	7.68	0.98	21.8	4.82	0.98	32.6	1.02	0.99	20.7	14.3	1.00	46.8	10.0	1.00

RESULTS

Coral Bay

Over the course of ten years, 30 distinct species of coral and 194 species of fish were observed in Coral Bay (see Appendix C for species lists). Sampling was predominantly on hardbottom sites and where water visibility was greater than 2 meters. Large areas of Coral Bay, especially shallow bays with significant terrestrial runoff, such as Coral Harbor and the upper reaches of Hurricane Hole, were not sampled due to poor visibility. Areas deeper than 30 m were also not surveyed due to SCUBA diving constraints.

Table 3. Long-term averages for coral reef fish indicators in Coral and Fish Bays, St John, USVI (N=517 and N=19, respectively). The long-term mean, standard error and probability of detection of indicators are represented by x, SE and P, respectively. The (-) symbol indicates the indicator was not measured or was not detected in sufficient quantities to make a valid estimate.

INDICATOR	Coral Bay			Fish Bay		
	x	SE	P	x	SE	P
Counts of All Fish	228.26	41.98	1.00	202.58	7.94	0.89
Counts of Grunts	39.62	27.24	0.56	0.95	0.31	0.42
Counts of Snappers	3.90	0.87	0.63	2.68	0.77	0.58
Counts of Parrotfish	30.51	3.26	0.95	21.52	6.22	0.74
Counts of Groupers	1.89	0.20	0.46	0.32	0.17	0.21
Diversity of All Fish	1.78	0.15	-	2.18	0.01	-
Biomass of All Fish	4211.85	785.28	-	20933.00	2330.00	-

One of the principal reasons CREMP uses belt transects to collect data instead of point counts (i.e., visual surveys within a relatively large cylinder) is to gather as much data as possible even in areas of limited visibility. This choice has no doubt in-

creased the amount of data available for studies in areas like Coral Bay where turbidity in the water column can be high.

As is common in most coral reefs in the USVI, algae cover was one of the most abundant taxonomic components of the benthic community. Estimates of algae cover among hardbottom habitats ranged from 32.6% to 21.8%, depending on the method used to collect data, and among softbottom sites was 37.4% (Table 2). Corresponding estimates of total live coral were 17.9%, 8.5%, and 3.6%.

Estimates of algae and coral were quite variable in time (Appendix A) and in space (Appendix B) across Coral Bay. Among individual survey sites, estimates of live coral and algae cover varied from 0% to 90% and 1% to 90%, respectively. Together, comprehensive and rapid habitat assessments identified 148 sites where live coral cover was greater than 20% and 15 sites greater than 50% (N=544). Sites with relatively high coral cover may indicate refuge areas where stressors are low or where demographic processes have resulted in resilient populations. The reefs in the northeast of Coral Bay, specifically in Round Bay and south of Turner Point, tended to possess sites with the highest coral cover compared to other reefs in Coral Bay (Appendix B , see figure 3 for geographic locations).

Grunts and parrotfish were major components of the fish assemblage detected in Coral Bay (~17% and ~13%) (Table 3). Other investigated fish families tended to proportionally contribute much less to the long-term average fish community density estimate (< 2%). We found that most sites had few grunts, but several sites had much higher numbers and these schools had an enormous influence on the long-term average density estimate. The map of grunt density in Appendix B shows this heterogeneous distribution pattern well. Our data does not provide the information to know if these special sites arose from surveys which captured a large mobile school or surveyed an area which supported more fish. In contrast to grunts, parrotfish were more cosmopolitan and are found in moderate numbers among more sites. These distinct characteristics in spatial variability were mirrored in corresponding plots of temporal variability (Appendix A). For instance, parrotfish density was relatively similar among years and standard errors were small, and grunt density fluctuated greatly over time and standard errors were very large in some years.

Fish Bay

A total of 19 sites distributed over nine years were surveyed in Fish Bay. Across all sites, 84 species of fish and 21 species of coral were sighted (Appendix C). Sampling was distributed

throughout the Bay and included both hardbottom (N=14) and softbottom (N=5) habitats. Unlike Coral Bay, depth and visibility did not regularly constrain surveys.

Due to its size, in most years one or two sites were sampled in Fish Bay and due to random site placement around the island of St. John, sites were not placed within Fish Bay in 2006. We did not find it appropriate to calculate annual estimates from so few data in each year. Tables 2 and 3 show long-term averages of investigated community metrics calculated from combining all years of data in Fish Bay without stratification by year.

Algae covered 47% of hardbottom sites and 21% of softbottom sites and were the predominant benthic cover among all sites (Table 2). Algae covered more than 80% of the seafloor on four of the 14 hardbottom sites and exceed 13% on only one softbottom site. Seagrass covered 1.5% and 19.2% of hardbottom and softbottom sites, respectively, and wherever algae cover was low, seagrass cover was relatively high. Average total live coral cover was 2.3% among hardbottom sites and there weren't any sites with more than 9% total live coral cover. Approximately four species of corals were observed on average among hardbottom sites and no corals were found on softbottom sites. Branching corals, soft corals, and hydrocorals were uncommon in Fish Bay.

Although the cumulative number of fish species and individuals observed in Fish Bay were lower than in Coral Bay, average estimates of diversity and biomass per transect were higher in Fish Bay (Table 3). Tangs, wrasses and parrotfishes were the most commonly observed fishes, while groupers, snappers and grunts were rare. Parrotfish were the most common investigated taxonomic group, comprising about 10% of all observed fishes. Fish Bay has the undesirable distinction of being the first place a lionfish (*Pterois volitans*) was detected among CREMP sites. A single lionfish was observed in Fish Bay in 2010. Lionfish are an invasive species from the Pacific Ocean, which has spread throughout the Caribbean after first being sighted in 1985 off the coast of Florida (Whitfield et al. 2002).

DISCUSSION

Fish and Coral Bays have dynamic coral reef communities, exhibiting variation among fishes, corals and habitats across a range of spatial and temporal scales. Both bays showed characteristics of a degraded coral reef community (e.g., Hughes 1994) with low coral cover and high algae cover, yet surveys identified pockets of coral refuges with very high coral cover in Coral Bay and sites in Fish Bay with healthy seagrass beds.

Interestingly, Friedlander et al. (2012) found that parts of Coral Bay were among the areas with highest coral richness, coral cover, and structural complexity in St. John. In addition, these areas of high coral cover and richness in Coral Bay also correlated with hotspots of several fish assemblage metrics such as richness, numerical abundance, biomass, and diversity (Friedlander et al. 2012). Furthermore, the broader scale analyses by Friedlander et al. (2012) suggest that Coral Bay may be an important juvenile habitat for commercially important fisheries species such as Yellowtail Snapper, Schoolmaster Snapper, and several species of parrotfishes. These ecosystem attributes along with the nursery function of the Coral Bay area highlights the importance and need to mitigate known stressors through watershed improvements.

Accurately defining the spatial and temporal heterogeneity of natural resources in these bays is critical to assessing the baseline status among corals, fishes and habitats, and detecting changes to coral reef communities resulting from watershed improvements. Without information on spatial and temporal heterogeneity, any identified changes to the community could debatably correspond to differences in sampling effort, natural variation or unmeasured anthropogenic impacts. This is the critical issue with any study attempting to detect measurable changes due to some management action. Both baseline and monitoring data have to be completely comparable and of fine enough resolution that changes are apparent and measurable. The variability described in this report underscores the need for sampling replicates throughout each bay and throughout time.

Long-term and consistent monitoring data are needed to critically assess the impacts of runoff on coral reefs and the differential effects from other impacts (e.g., fishing, hurricanes, bleaching, anchoring). Coral reef ecosystems are dynamic and the magnitude and periodicity of disturbances greatly affect spatiotemporal patterns observed on coral reefs (Done et al. 1991, Connell 1997). To adequately identify and evaluate changes, monitoring needs to be conducted over time scales commensurate with the periodicity and spatial scale of disturbance events.

The CREMP monitoring dataset investigated in this report offers a dataset which is well distributed in space around the island of St. John. Hundreds of surveys were collected over ten years in Coral Bay, providing information on natural variation and long-term trends present before watershed improvements were put in place. These data can effectively be used in a BACI (Before, After, Control, Impact) design to conclusively assess impacts (Underwood 1994), and if needed allows for multiple control sites to be investigated and compared to impacted sites.

For smaller strata like Fish Bay, survey densities collected in CREMP are too low to offer a rigorous baseline. Too few data were collected to provide information on natural variation or long-term trends. In Fish Bay and other areas with similar amounts of data, only major community changes, such as phase shifts or catastrophes, will be generally detectable. A targeted survey with more sites in Fish Bay is needed to provide sufficient data to adequately measure spatial and temporal variation. Friedlander and Beets (2008) offer an alternative dataset, but is limited in spatial scope.

The data described in this report must be compared to similar data collected in the future to assess impacts of watershed improvements. We have presented information on key ecological attributes using a conventional framework to simplify the task of compiling and analyzing this information in future comparisons.

According to Rogers (1990) one might expect to see the following changes in coral communities if watershed improvements reduce sedimentation: higher species diversity and live coral cover; a smaller proportion of corals resistant to smothering from sediments, like branching corals or soft corals; larger coral colonies; higher recruitment and growth rates; and a downward shift in depth zonation. Much less is known about how fishes are affected by runoff and sedimentation.

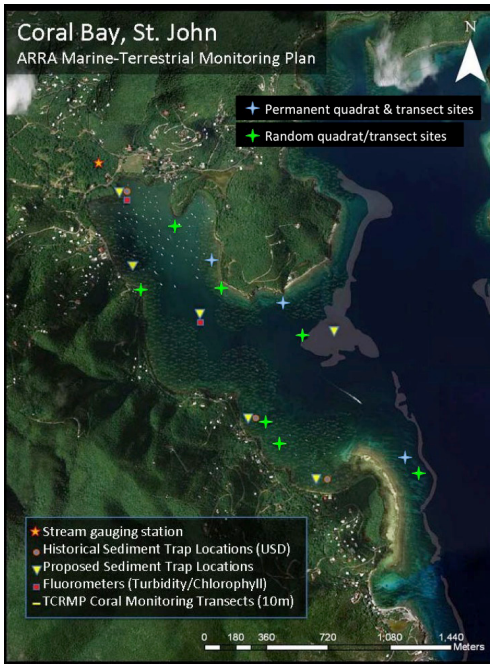
Changes to fishes, corals and benthic habitats related to changes in terrestrial runoff may not be clearly identifiable. Several studies have shown that different fish and coral species and habitats are affected by changes in runoff and sedimentation unequally and responses will vary over space and time (Rogers 1990, McClanahan and Obura 1997, Airoldi 2003). These differences are generally attributed to characteristics of the depositional environment, species life histories, the surrounding seascapes and historical patterns. A continuation of CREMP or a new well-planned survey program with spatiotemporal replicates and a broad taxonomic scope will maximize the probability of accurately assessing future community changes.

The indicators examined in this report were chosen to identify some of the expected changes by Rogers (1990), but not all possible changes can be measured using data collected by CREMP (e.g., coral recruitment). Other work by the University of the Virgin Islands and the Southeast Fisheries Science Center funded by ARRA and yet to be published will examine additional fish, coral and habitat indicators. Taken together this report and these future ARRA-funded reports will provide a more comprehensive assessment of the coral reef communities than either assessment on its own.

We present basic information on annual mean abundances over time and variability around the mean, but do not provide in-depth discussion of spatial patterns or of specific patterns in metrics, such as coral bleaching. Those interested in learning more about spatial patterns around St John and coral bleaching are encouraged to read Friedlander and Beets (2008) and Friedlander et al. (2012) for more information.

Acknowledgments

This work would not have been possible without the numerous contributors who shared their time and expertise. We thank Alan Friedlander, Kimberly Roberson, Simon Pittman, Caroline Rogers, Jeff Miller, Matt Kendall, Chris Jeffrey, Rob Warra, Andy Davis, Sarah Hile, Jim Beets, Carrie, Rafe Boulon, John Christensen, Erin Muller, and Kimberly Foley for collecting data. Contract support was provided by Consolidated Safety Service, Inc. under NOAA Contract No. DG133C07NC0616.



Chapter 3
Monitoring Coral Reef Changes Associated with Erosion Mitigation Projects in Fish Bay and Coral Bay, St. John, USVI

Ronald L. Hill1*, Jennifer C. Doerr1, Kimberly G. Ferran[2]

[1]NOAA Fisheries, Southeast Fisheries Science Center
[2]Florida Atlantic University

*Corresponding author: ron.hill@noaa.gov, 409-766-3519
Mailing address: 4700 Avenue U, Galveston,TX 77551

Abstract

Projects that reduce runoff of sediments and associated pollutants, including pesticides, herbicides, excess nutrients, and other exogenous materials have the potential to benefit coral reef ecosystems in downstream catchment basins. We undertook supplemental monitoring of the Virgin Islands Resource Conservation & Development Council, Inc's. (VIRC&D) ARRA-funded project: USVI Coastal Habitat Restoration through Watershed Stabilization in Fish Bay and Coral Bay, St. John, USVI. Our focus was on biological responses expected from erosion mitigation work; and, in lieu of before-and-after studies, our one-year project used distance from stream inflows as a proxy for sediment reduction expected over time. We found that biomass and dominant components of the macroalgal

assemblage varied with distance from stream inflows, as did percent cover of scleractinian corals. Additional analyses will place these results within the framework established by the primary monitoring efforts, focused on water quality and sediment flux, and prior sampling that has characterized fish and benthic communities. Based on our study results and successful reductions in runoff into these two bays, we would expect to see changes in inshore reef communities as algal assemblages change and coral cover increases.

Introduction

Many coral reef ecosystems, by their proximity to coastal development, are subjected to alterations in water quality and environment that change the nature of their linked communities (Edmunds 2005). Coastal development generally increases runoff, particularly during heavy rains, typical of the tropics. Increased runoff carries terrigenous sediments and other contaminants into downstream catchment basins where they can adversely affect habitats, including coral reef communities and seagrass beds in tropical settings. Excess sediments can smother corals, affecting relative abundances, since some species are more able to clear sediment from their surface tissues. Sediment also increases water turbidity, slowing coral growth, changing species composition and reducing diversity, abundance, cover, larval survival, fecundity, and reproductive outputs (Rogers 1990, Edmunds 2000, Fabricius 2005). Influx of terrigeneous sediments from developed or residential areas will generally introduce chemical pollutants that may also be detrimental to the organisms in the downstream ecosystems. Increased freshwater runoff tends to accelerate erosion depending on watershed slope, intensity of rainfall, soil conditions, and land use (Ramos-Scharrón and MacDonald 2005). Proper drainage controls can reduce the inflow and possibly reduce the impact to corals and other associated biological components.

The Virgin Islands Resource Conservation & Development Council, Inc. (VIRC&D) USVI Coastal Habitat Restoration through Watershed Stabilization Project was designed to reduce the runoff of sediments, nutrients, and other pollutants into coastal waters where they have changed the character of nearshore habitats, most notably macroalgal plains, seagrass beds, and coral reefs. Excess sediment and associated turbidity modifies conditions needed for effective maintenance of biotic integrity (Hubbard 1987, Philipp and Fabricius 2003, Steward et al. 2007). Our project was designed to evaluate some of the biological effects anticipated from a reduction in sediment, reduction in turbidity, and stabilization of water quality parameters. Our hypothesis was that measures of reef and macroalgal characteristics would reflect differences in sedimentation and pollution rates in the affected ecosystem and that spatial differences could approximate anticipated temporal changes.

In two of the bays, with erosion reduction projects, we assessed the effects of reduced sediment on corals, coral reef habitats, seagrasses, macroalgae, and associated fauna. We predicted that if sediment influx and associated turbidity were reduced, seagrasses would expand, macroalgal composition would change and macroalgal biomass in inshore areas would be reduced, corals would colonize and expand coverage, and fish assemblages would reflect habitat differences. Our methods are geared toward testing these predictions and establishing baselines for future monitoring.

Methods

The methods we chose were intended to 1) detect differences along a presumed gradient of exogeneous sediment and nutrient inputs, 2) establish a baseline of conditions in the affected coral reef ecosystems that could detect early system responses to changes in flux of sediments and nutrients over time, and 3) conduct complementary monitoring that could contribute to project monitoring by UVI and partners. Results would be set within the framework of environmental monitoring of sediment (Gray et al. 2008), water quality, and contaminants for analysis.

Study locations include permanent and random sampling stations in both Fish Bay and Coral Bay (described in Chapter 1 Introduction) in areas selected to represent inner, middle, and outer zones relative to the primary locations of freshwater inflow (Fig 3.1). In each bay, streams or intermittent (seasonal) streams enter at the northern end of the bay. Site selection was limited by availability of reef sites of adequate size for transect surveys. Some consistency in depth was desirable and was achieved at all sites except inner Coral Bay (Table 3.1).

Table 3.1. Sampling stations and depths.

Bay – Station	Zone	Depth (m)
Fish – North	Inner	5
Fish – Cocaloba Point	Middle	6
Fish – Cocaloba Outer	Outer	9
Coral – Beach	Inner	1-2
Coral – North Point	Middle	4
Coral – Johnson's	Outer	9

Table 3.2. Algal functional groups.

FUNCTIONAL GROUP	EXAMPLE
1. Microalgae (single cell)	Cyanobacteria, diatoms, Schizothrix
2. Filamentous Algae (uniserate)	Cladophora, Wrangelia, Turf
3. Foliose Algae (single layer)	Ulva
3.5 Corticated Foliose Algae	Dictyota, Padina
4. Corticated Macrophytes (terete)	Laurencia, Acanthophora
5. Leathery Macrophytes	Sargassum
6. Articulated Calcareous Algae	Halimeda, Udotea,
7. Crustose Algae	Lithothamnion, Peyssonnelia

Monitoring Macroalgal Assemblages

Three sampling sites within each of the two bays were selected for permanent sampling stations. General reef areas were selected to represent inner, middle, and outer zones with the added condition that each area must be sufficiently large (minimum: 10m by 12m) to conduct benthic transect surveys. By ranging across the sediment/nutrient gradient, inshore sites were expected to show more influence of runoff and offshore sites were expected to show less influence (Fig. 3.1). At each permanent station, four "¼-meter quadrats" (25 x 25 cm) were set up for repeat monitoring. The ¼ m quadrat (made of ½-inch PVC) was thrown haphazardly from the surface onto the reef and the quadrat was moved to the nearest location where ¼ m of relatively flat reef could be found. Stainless steel screws (2 in.) were hammered into the reef substrate in each of the inner corners so the quadrat could be reliably repositioned for repeat surveys. The northeast corner screw of each quadrat also impaled a numbered plastic tag for identification. Screws were not put into any live coral tissue. Initial photographs and GPS coordinates were taken of each permanent quadrat. For assessment, macroalgal assemblage composition was assessed by identifying each organism within the quadrat to the lowest taxon possible and estimating percent cover for each species/taxon.

In addition to the permanent monitoring stations, random stations were sampled at locations interspersed among the permanent sites. Minimum requirements for the random sites were a sufficient area for sampling three quadrats in hardbottom/reef habitat. In Fish Bay, two random stations were sampled and in Coral Bay, seven random stations were sampled. Once an appropriate area was found, the quadrat was thrown haphazardly from the surface and the planar surface beneath or behind the frame was sampled. Initial photographs were taken with the quadrat frame in place. Macroalgal composition in random quadrats was assessed in the same way as permanent quadrats. Following visual assessment, all algae

were removed from the 25 x 25 cm surface area within the quadrat and placed into a fine mesh bag for later biomass analysis. Photos of the cleared quadrats were taken.

Fish and Benthic Surveys

At each of the permanent stations in Fish Bay and Coral Bay, we conducted paired fish and benthic surveys. A stationary point count method (Bohnsack and Bannerot (1986) method) was used for fish surveys. A diver descended to the bottom within the area where the subsequent coral survey would be performed and surveyed all non-cryptic fish species that

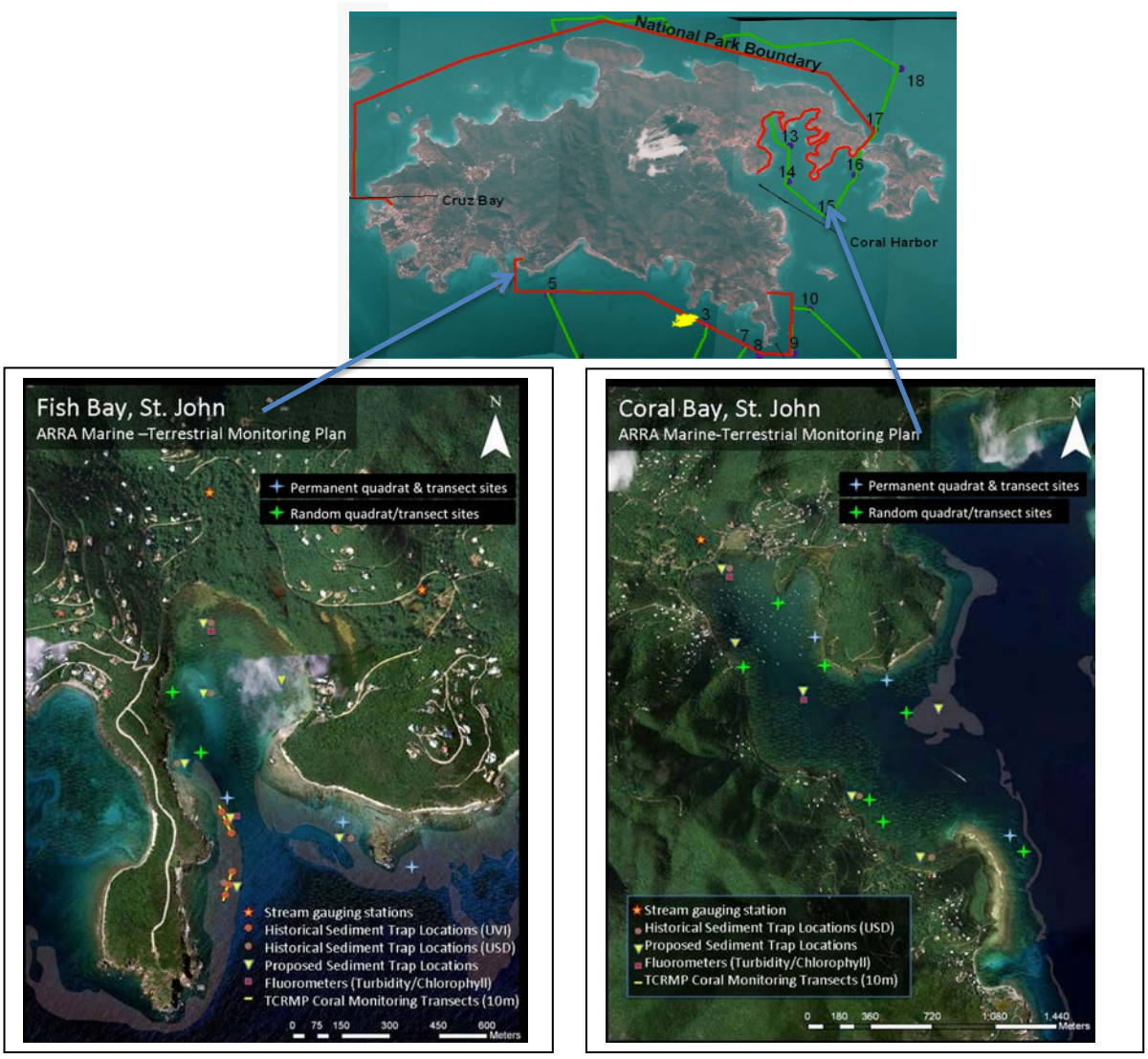


Figure 3.1. Study Sites at St. John, USVI: Fish Bay and Coral Bay study sites for permanent (blue stars) and random quadrat sites (green stars) displayed with project monitoring sites used by UVI and University of San Diego researchers. (Base images courtesy of T. Smith, UVI)

Table 3.3. Mean percent cover of dominant macroalgae species distributed along the inner, middle, and outer zones of Fish and Coral Bays. Species are listed in order of increasing complexity of their functional groups beginning with the simplest. Blank cells were left empty to more easily display patterns in macroalgal distributions by zone in each location, and do not necessarily indicate the absence of that species in that zone, simply low percentages.

Dominant Species	FISH BAY			CORAL BAY		
	Inner	Middle	Outer	Inner	Middle	Outer
Gelidiella spp.			27.50		30.00	40.00
Dictyota spp.	26.00	12.50	36.25		27.50	20.00
Lobophora spp.						28.33
Acanthophora spp.				32.50		
Caulerpa spp.				13.33		
Galaxaura spp.		20.50				
Jania spp.	55.00					
Halimeda spp.	41.25	35.25		15.50		
Crustose algae			18.33		16.00	20.00

passed within a 5m-radius cylinder, envisioned to extend from the benthos to the surface. All species were recorded to the lowest taxon possible (generally species level); their numbers and estimated fork lengths were noted. If large schools were encountered numbers were estimated by multiples of counts extrapolated over the spatial extent of the school. Mobile macro-invertebrates, i.e., queen conch, spiny lobster, and Diadema, were counted and recorded if found within the cylinder. Two non-overlapping surveys were conducted at each permanent station at each visit. The same diver (RLH) conducted all fish surveys.

Following the fish surveys, the benthic surveys were conducted. Benthic surveys followed the AGRRA (ver. 5.4) line point intercept method of sampling 6 parallel replicate transects, 10 m long. Each organism immediately below the tape at each 10 cm interval was noted. For the transect set-up, a 30m fiberglass transect tape with a weighted end was secured at a random starting point and a diver unreeled the tape while swimming in a straight line. After 10m of tape had been laid out, the diver wrapped the tape around a non-living portion of the reef, swam between 1-2 m perpendicular to the first transect, wrapped the tape around a non-living component of the reef and laid out a second transect parallel, but not closer than 1 m, to the first. At the transect's end, the tape was secured to the reef. A second tape was used to lay out the third and fourth transects in the same manner and after the survey was

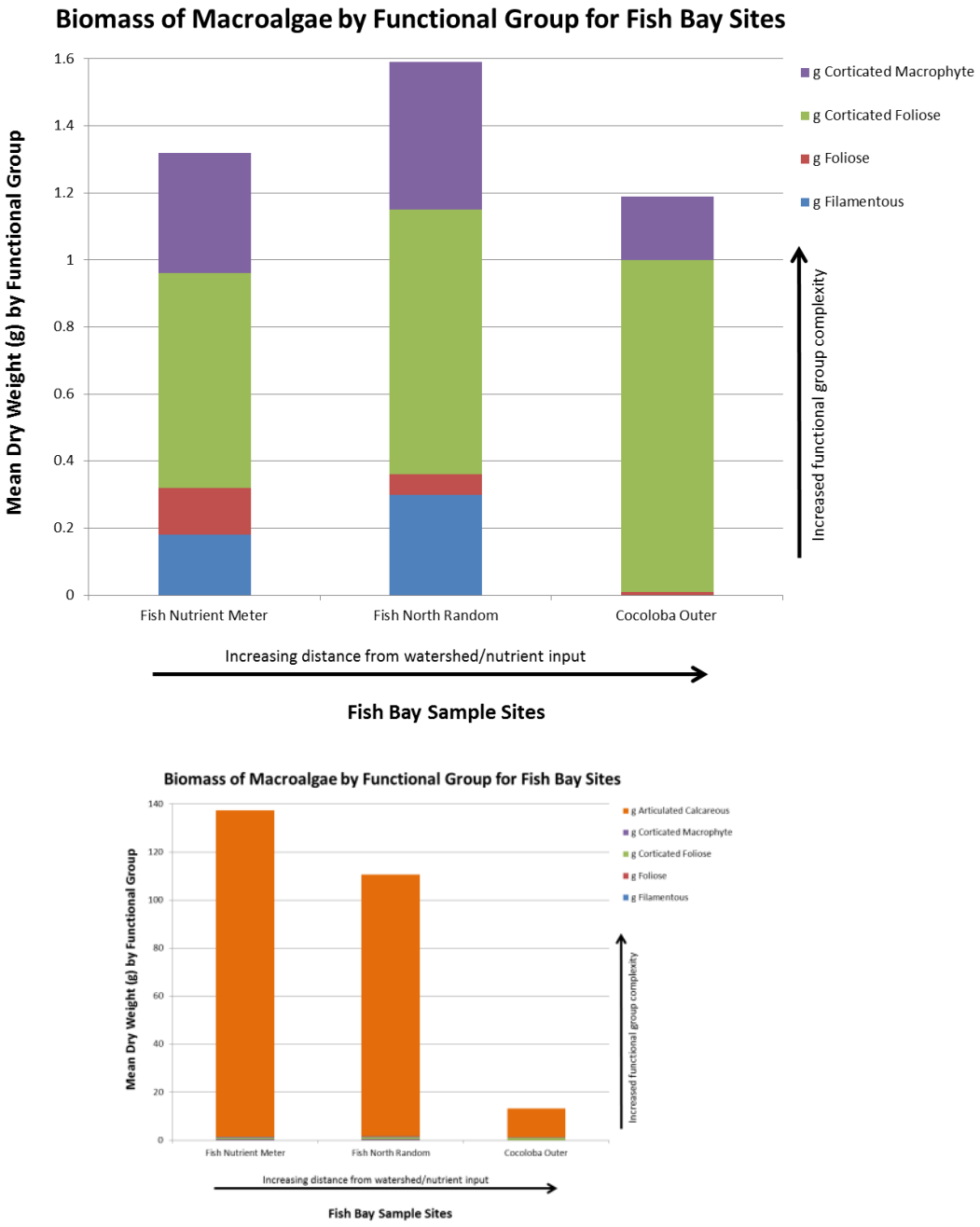


Figure 3.2. Mean algal biomass from Fish Bay sites measured in g dry weight by functional group. The small graph shows all groups and illustrates the dominance by articulated calcareous (AC) algae. This AC group is removed from the larger graphs to more clearly show the relationship between distance and dry weight for the other functional groups.

completed on the first tape, it was moved to make up the fifth and sixth transects. All benthic surveys were conducted by the same diver (KGF).

Benthic data from August 2012 surveys were analyzed by Analysis of Similarity [ANOSIM (Primer 6, ver. 6.1.15, Plymouth-E Ltd.)] comparing differences between inner, middle and

outer zones. Analysis was based on untransformed data (summed coral cover per transect) and Bray Curtis similarity matrices for each bay separately after ANOSIM showed them to be significantly different ($p<0.2\%$).

Algal Biomass Study

Macroalgae were cleared from the three replicate quadrats at each random station and from two of the four quadrats at each permanent station. Each quadrat's algae were placed into a fine mesh bag and maintained in seawater until time for analysis. For the analysis, each sample was sorted and weighed separately. Algae were rinsed and identified to the lowest taxon possible using magnifying lenses and then grouped into functional groups primarily based on life form (sensu Steneck and Dethier 1994).

Each functional group was weighed (blotted wet weight) in an aluminum weighing boat. All samples were dried in an oven at low temperature (170°F) for 8-9 hours. The first set of samples was weighed and then dried for additional time to test for additional weight loss. Eight to nine hours was sufficient for drying. Samples were held in the oven and allowed to cool before they were weighed to obtain a dry weight.

High Resolution Mapping and Habitat Boundaries

High resolution maps, using side-scan sonar and multibeam with groundtruthing are being used to generate high resolution maps of pertinent areas of Fish Bay and Coral Bay to delineate habitat boundaries (e.g., between mangroves, seagrass habitat, macroalgal beds, sand, mud, rocky shores, and coral reefs within the bay and immediately outside). Supplemental aerial images are available if needed for shallowest parts of the bays. Work is being done in conjunction with Polytechnic University of Puerto Rico and NMFS SERO Habitat Office and final products are expected by August 2013. One of the goals of the mapping is to assess changes in distribution of sediment type, seagrass, macroalgal beds, fish, macro-invertebrates and benthos over time. With appropriate groundtruthing we will delineate boundaries between seagrass, macroalgae, fine and coarse sediments, coral reefs and various hard substrates. Horizontal resolutions of 10-17 cm are expected. With changes in turbidity it is anticipated that boundaries between seagrass and macroalgae will shift, especially in the inner parts of the bays nearest to the terrestrial inputs. Animal distributions are also likely to change with water quality and benthic changes. Maps will provide a baseline for future comparisons.

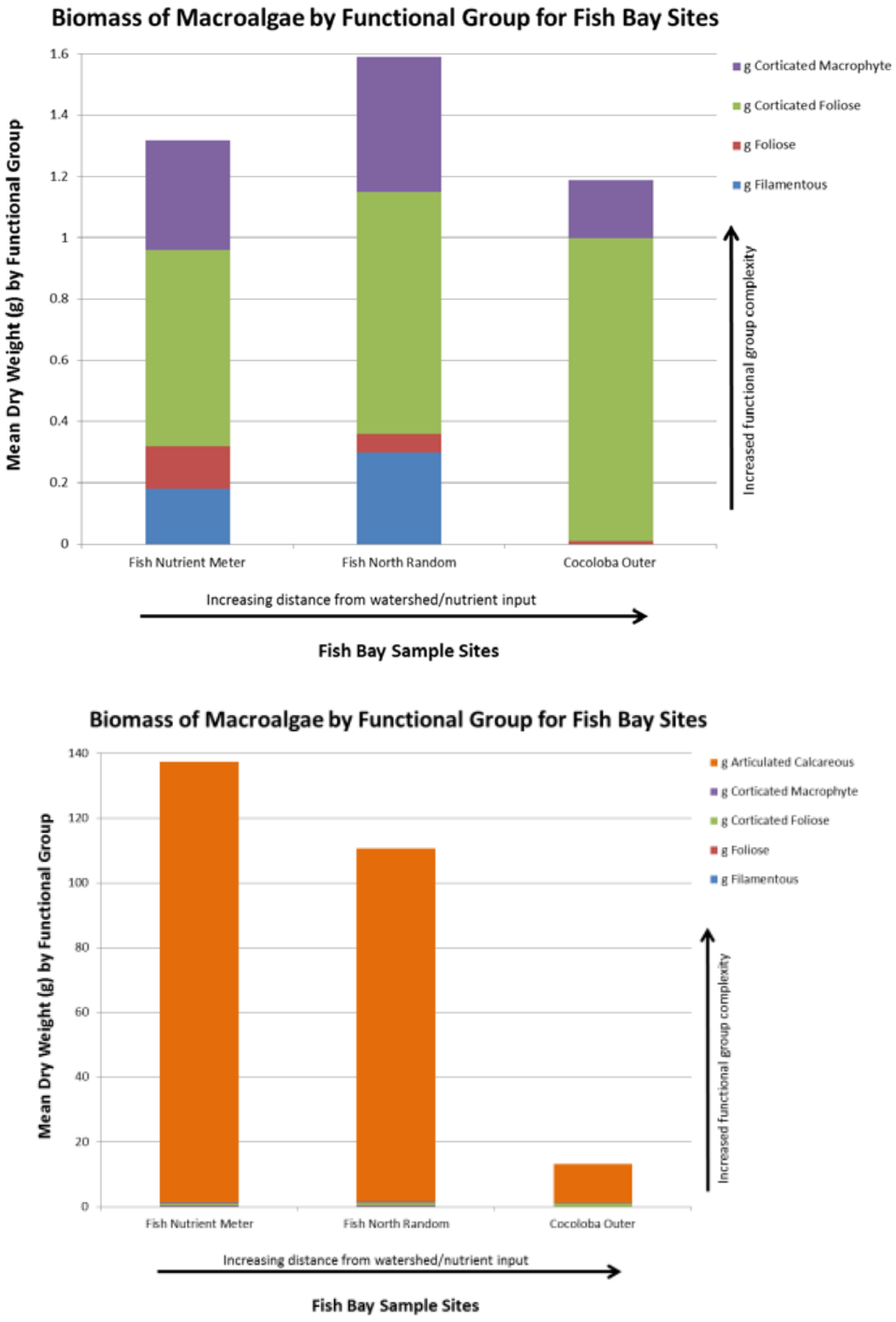


Figure 3.3. Mean biomass of algae from Coral Bay sites measured in g dry weight by functional group. The small graph shows all groups and illustrates the dominance by articulated calcareous (AC) algae. This group is removed from the larger graphs to more clearly show the relationship between distance and dry weight for the other functional groups.

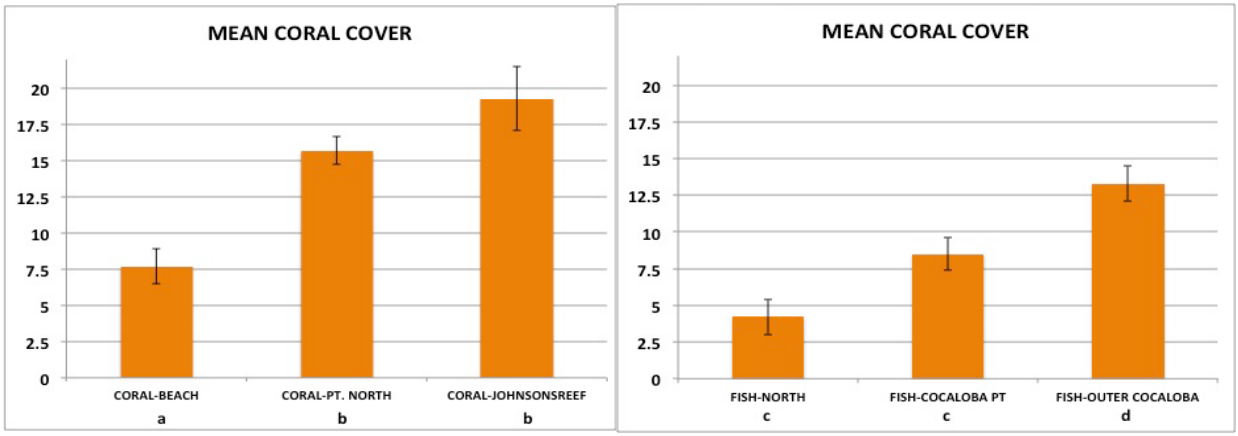


Figure 3.4. Mean coral cover of all scleractinian corals, as measured by LPI, from Coral Bay and Fish Bay sites in August 2012. Small case letters indicate significant differences between zones for each bay; different letters indicate differences at a p<5% level.

Coral Recruitment

Many coral species are known to fluoresce under proper lighting conditions and this property can be useful in identifying coral recruits before they would otherwise be detected by visual methods. Using a strobe filter and yellow eye shields, we investigated fluorescence in various species and sizes of coral colonies. We photographed algal quadrats to locate coral recruits but the results were inconclusive.

Coordination with project monitoringThe ARRA funding provided some basic monitoring to examine sediment flux and water quality in affected watersheds and expansion of existing efforts to characterize fish, corals, and other benthic components. These efforts are being coordinated/conducted by UVI (T. Smith). Under the current grant, results of additional research into the chemical contaminants in the bays are reported in this publication (Chapter 4). NOAA researchers (NOS and NMFS) have conducted fish and benthic monitoring since 2001 and since 2005 for queen conch and benthic composition. More detailed analyses will set our results within the environmental framework defined by those complementary studies.

Results and Discussion

We measured and characterized the macroalgal populations at 10 sites (3 permanent, 7 random) in Coral Bay and 5 sites in Fish Bay (3 permanent, 2 random) in June and August 2012. The overall dominant species by mean percent cover are listed in Table 3.3, with an additional 6 to 10 species identified at each site (e.g., Valonia spp., Udotea spp., Penicillus spp., Neomeris spp., Padina spp., Wrangelia spp., and cyanobacteria).

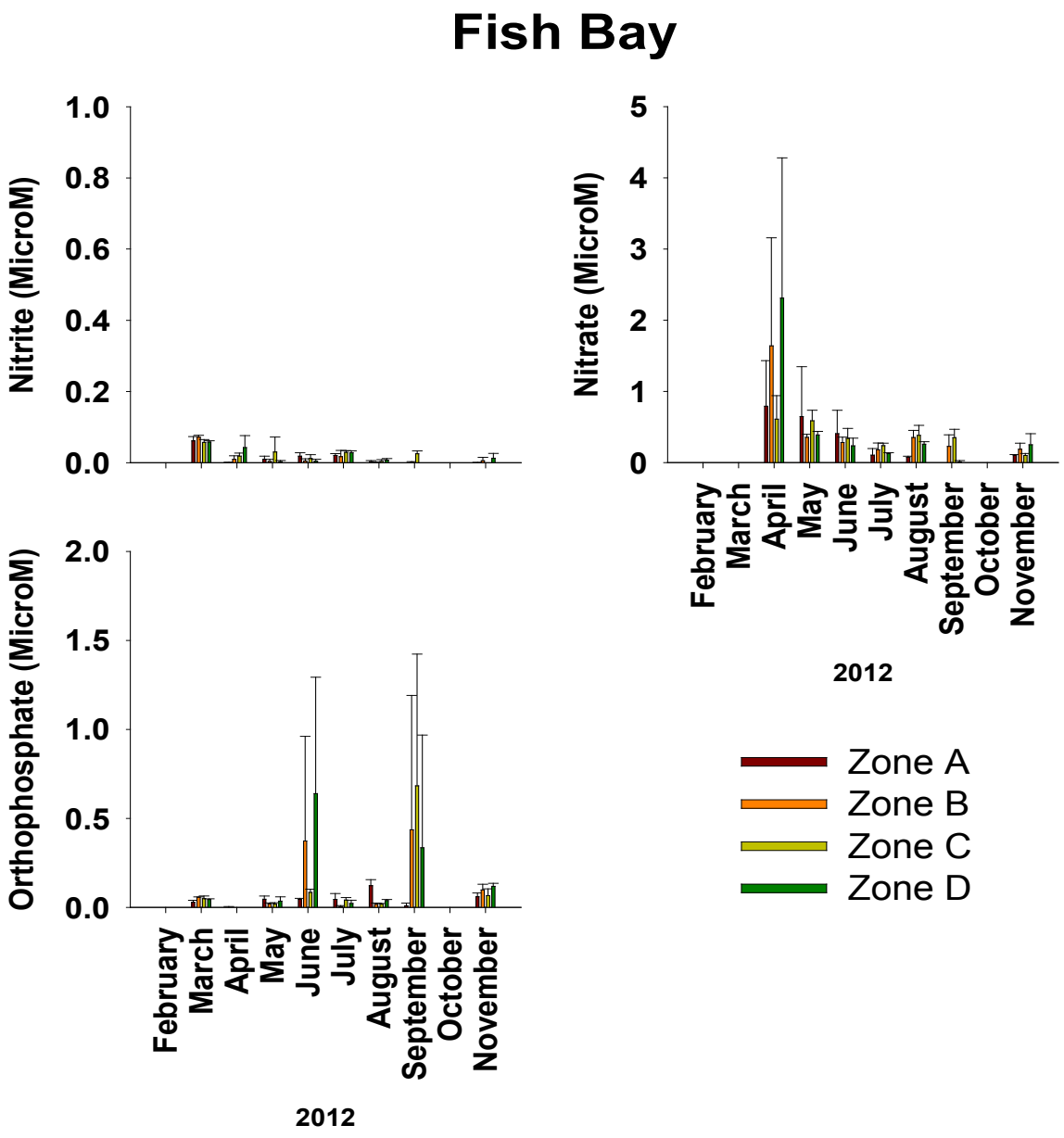


Figure 3.5. Nitrite, nitrate, and orthophosphate concentrations (mean +/- SD) for sampling months between February and November 2012 (Courtesy of T. Smith, from Section 106, Report 3).

There was no clear pattern in percent cover or species composition with increasing distance from the major watersheds or nutrient inputs in either Fish or Coral Bay (Figs. 3.2 and 3.3). In Fish Bay, a decreasing trend from inner to outer (middle) stations is not continued onto the outermost, Cocaloba Outer, site. However, when species were examined by major functional groups and graphed by increasing complexity, there was a trend toward increased cover of the most simple and most complex groups and decreased cover of the mid-range groups with increasing distance from the watershed or nutrient input in both Fish and Coral Bay (Figs. 3.4 and 3.5). Sites in the inner to middle zones in each location showed increased

cover for the mid-range groups with decreased cover by the most simple and complex functional groups.

Macroalgal biomass was dominated at each site by the presence of calcareous algae, particularly Halimeda spp., although dry weight generally decreased with increased sample distance from inshore watersheds and nutrient inputs. Fish Bay sites showed a decrease in articulated calcareous, filamentous, and foliose algae while corticated foliose and corticated macrophytes remained nearly the same across sample zones (Fig. 3.6). In Coral Bay biomass of articulated calcareous and filamentous algae decreased similarly across the distance gradient, but no trend appeared in the biomass of the other functional groups (Fig. 3.7).

Figure 3.6. Mean algal biomass from Fish Bay sites measured in g dry weight by functional group. The small graph shows all groups and illustrates the dominance by articulated calcareous (AC) algae. This AC group is removed from the larger graphs to more clearly show the relationship between distance and dry weight for the other functional groups.

Fluorescence, as a means for detecting coral recruits, was investigated in both Fish and Coral Bays and in shallow areas of Hurricane Hole. Photographs of each permanent and random quadrat were taken to document the presence of small corals. Future opportunities for field assessments would allow us to document coral growth and/or survival, as well as algal growth and change. Fluorescence photographs were also taken and have allowed us to examine quadrats for potential recruits. While this analysis is still on-going, an example of the results is shown in Fig. 3.10 where a small, previously undetected, colony was found in post-processing. Literature citing the use of this technique is unclear on some of the parameters affecting reliability of the method; coral size and species are mentioned as variables. In order to test the applicability, we photographed various sizes and species and results will be documented for future work. An example of the photographs is presented in Fig. 3.11, where a small Diploria colony was the target of the photo but a small colony, possibly a recruit, can be seen just to the right of the colony.

The line point intercept benthic surveys were completed at each of the permanent stations within Fish Bay and Coral Bay in June and August 2012; here we present data from August surveys. Coral cover, in general, was considered low [mean cover: 11.5%; +/- 1.0 (SE)] with an increasing trend from inshore to offshore (Figure 3.8). In Fish Bay [mean cover: 8.7%; +/- 1.1 (SE)] coral cover ranged from 4.2 to 13.3% and in Coral Bay [mean cover: 14.2%; +/- 1. (SE)] cover ranged from 7.7 to 19.3%. These values do not include fire corals or octocorals. Some key species also show spatial variation; Montastraea species covered 2.5, 3.0, and

4.3% on Fish Bay inner, middle, and outer reef sites, respectively. They covered 0.0, 3.8, and 10.1 on Coral Bay inner, middle, and outer sites, respectively. As indicated in Fig. 3.8, pair-wise comparisons demonstrated significant differences between zones within each bay. In Fish Bay, the inner site differed from the middle site only at the $p<8.0\%$ level but the outer site differed significantly from the middle site ($p<3.0\%$) and the inner site ($p<0.2\%$). In Coral Bay, the inner site differed significantly from both middle ($p<0.2\%$) and outer ($p<0.4\%$) sites but the middle and outer sites differed only at $p<9.1\%$.

Water quality measurements have been taken by project partners at UVI in areas of Coral Bay, Lameshur Bay, and Fish Bay. Samples are taken on a regular basis in order to examine seasonal and storm-induced changes in water quality in inner and outer zones of each bay. An example of the data collected is presented in Figure 3.9. Analysis of benthic, algal, and chemical contaminant data will use the water quality data to define the spatial framework of the bays.

Conclusions

Excess sediment input into coastal waters has been shown to affect habitats and habitat constituents, such as seagrasses, macroalgae, and corals (Smith et al. 2008). From our research we have documented the aspects of these systems that will enable detection of change over time:

- Surveys of macroalgae confirm that a gradient, defined by species, relative abundances, and biomass exists in Fish Bay and Coral Bay.
- A functional group approach to understanding macroalgal distributions and responses to environmental change may be an appropriate way to document spatio-temporal change.
- Surveys of scleractinian corals confirm that a gradient, defined by percent cover and species composition exists in Fish Bay and Coral Bay. Additional analysis will be needed to fully elucidate differences and expectations for change in response to changes in environmental conditions.
- Surveys of coral recruits demonstrate the feasibility of the approach. Expansion of sample size and temporal range are required to document scientific robustness.

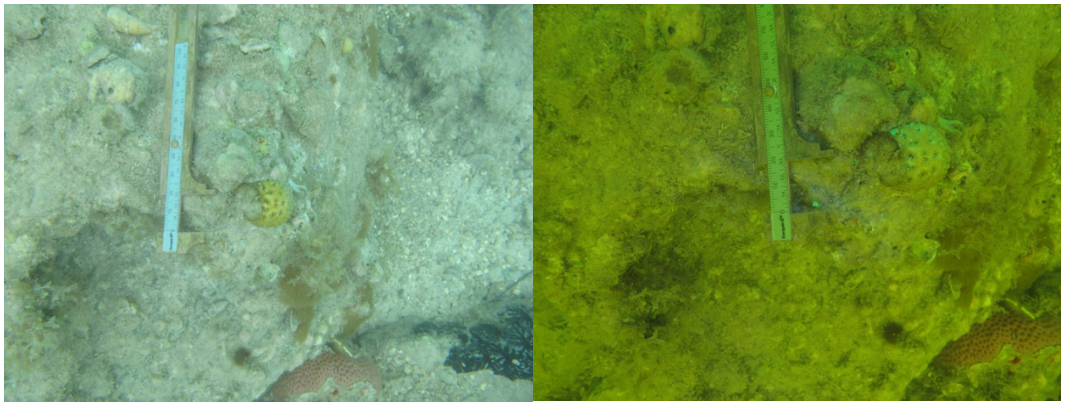


Figure 3.6. Photos of small Favia fragum colony showing fluorescence. A small colony, possibly a recruit can be seen glowing in the photo on the right just inside the jaws of the calipers.

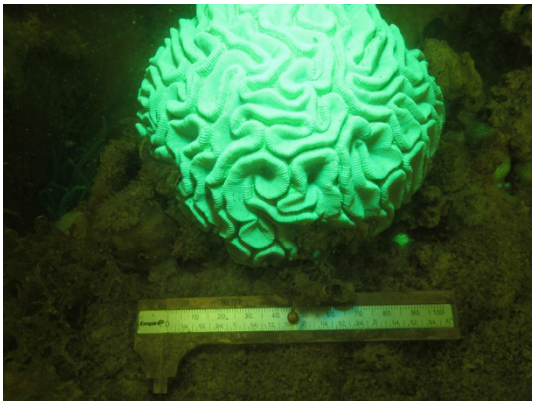


Figure 3.7. Example of fluorescence of small Diploria colony with likely recruit just to the right.

Chapter 4
Contaminants in Surficial Sediment of Coral and Fish Bays, St. John USVI

David Whitall*, Anthony Pait, S. Ian Hartwell

National Oceanographic and Atmospheric Administration

National Ocean Service

National Centers for Coastal Ocean Science

Center for Coastal Monitoring and Assessment

*Corresponding author: dave.whitall@noaa.gov, 301-713-3028x138,

Mailing address: N/SCI 1, 1305 East West Hwy, Silver Spring, MD 20910

Background

In their report entitled “America’s Living Oceans”, the Pew Oceans Commission cited both point and nonpoint sources of pollution as major threats to the oceans (Pew Oceans Commission, 2003). Waddell et al. (2005) have described some of the threats posed by pollution to coral reefs of the U.S. and Freely Associated States. The U.S. Coral Reef Task Force identified pollution as a focus area for priority action in the LAS for Puerto Rico, the Virgin Islands and Southeast Florida (FDEP 2004). Although pollution is frequently cited as impacting coral reef health, the concentration of chemical contaminants present in coral reefs is not well characterized, and typically even less is known regarding linkages between contaminants and coral condition. Downs et al. (2005) concluded that coral decline in a section of the northern Florida Keys is likely related to chemical contaminant exposure, and noted that an analysis of contaminants present would greatly increase the power of determining the impact of this stressor. Developing an understanding of how and to what extent contaminants affect the health of corals and coral reefs would help focus management efforts.

Contaminant Background

Chemical Contaminants Analyzed

Since 1986, NOAA’s National Status and Trends Program (NS&T) 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 reef ecosystems represents a relatively recent expansion of NS&T activities.

The suite of chemical contaminants routinely analyzed by NS&T in sediment samples for this project is shown in Table 4.1. 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 (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.

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 oil seeps. The PAHs analyzed in this study are two to six ring aromatic compounds and their substituted analogs.

Environmental Effects of PAHs. Although an extensive amount of research has been done on the accumulation and effects of PAHs on aquatic organisms, comparatively 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 (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 carcinogens (USDHHS 1995). The carcinogenic potential of PAHs is associated with their metabolic breakdown which generates reactive epoxides which 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. 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 PCB's 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 *Diploria strigosa* and *Madracis decatis*, however, depuration proceeded at a slow rate; after 275 days nearly 33 percent 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 samples (Table 4.1). 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 **d**ichloro**d**iphenyl**t**richloroethane (DDT). Organochlorine pesticides, including DDT, are still of environmental concern due to their environmental persistence, potential to bioaccumulate, and toxicity to non-target organisms. These concerns led to their ban in the United States in 1972, but because of their persistence and heavy use in the past, residues of many organochlorine pesticides can be found in the environment, including biota.

Environmental Effects of Organochlorine Pesticides.

Organochlorine pesticides primarily act on biota as neurotoxins. Both DDT and PCBs have also been shown to be endocrine disruptors. DDT and its metabolite dichlorodiphenyldichloroethylene (DDE) have been specifically linked to eggshell thinning in birds, particularly raptors (Lundholm 1997). A number of organochlorine pesticides are also toxic to aquatic life including crayfish, shrimp and fish (DeLorenzo 2001). Herbicides have also been shown to have negative impacts on early coral life stages (Negri et al 2005) and significantly impact coral metabolism and photosynthetic efficiency of coral zooxanthellae (Owen et al 2002, 2003, Jones and Kerswell 2003, Raberg et al 2003).

Butyltins

Butyltins (mono-, di-,tri- and tretabutyltins) have a range of uses from biocides to catalysts to glass coatings. In the 1950s, tributyltin or TBT was first shown to possess 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 is 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 be 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 2 - 4 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 percent of the original TBT concentration, successful settlement of coral larvae in the laboratory was inhibited. TBT also reduces coral recruitment by inhibiting fertilization (Negri and Heyward 2001), and has been shown to significantly reduce protein synthesis and skeletal deposition of protein in corals (Allemand et al 1998). TBT exposure can cause mortality to corals but was found to have limited effect on photosynthesis, suggesting TBT may have a greater impact to coral hosts than their associated zooxanthellae (Smith et al 2003). Tin is also bioaccumulated in coral skeletons, which may offer the potential to evaluate changes in TBT use/exposure over time (Inoue et al 2004).

Major and Trace Elements

A total of 16 trace and major elements were measured in sediments for this project (Table 4.1). 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 elemental (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 Al, Fe and Si; 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 impede 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 through atmospheric deposition. Effects of mercury on copepods include reduced growth and reproductive rates (Eisler 1987). 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 boat antifouling paints, wood preservatives, heat exchangers in power plants, electrical wires, coinage, and in agriculture. Although an 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 brain coral *Goniastrea aspera*. At copper concentrations at or above 75 µg/L, fertilization success was reduced to one percent or less. Fertilization success was also significantly reduced in the coral *Acropora longicyathus* at 24 µg/L, a similar concentration level to when effects were observed in *G. aspera*. Zinc and cadmium may be less toxic to coral gametes than copper (Reichelt-Brushett and Harrison 1999). High levels of metals such as Cd and Cu can cause coral mortality (Mitchelmore et al 2007), though there is some evidence that corals have some potential to acclimate to metal exposure over time (Harland and Brown 1989). Trace concentrations of many metals are incorporated into coral skeletal material, and are commonly used as proxies for oceanographic processes (e.g. Matthews et al 2008, Shen 1986) or indications of anthropogenic impacts (e.g. Shen and Boyle 1987, Guzman and Jimenez 1992, Bastidas and Garcia 1999) .

Ancillary 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.

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.

For sediment samples, six geographic strata were initially articulated based on natural geographic breaks within the system (e.g. harbors): four within Coral Bay (Main Bay (MB), Hurricane Hole (HH), Round Bay (RB) and Coral Harbor (CH)) and two within Fish Bay (Fish Bay North (FBN), and Fish Bay South (FBS)). In each of these six strata, 5 sites were randomly selected (Figure 4.1). 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. Sediment strata were constructed from existing benthic habitat maps and included all non-hard bottom areas. Due to weather issues, only 3 sites were sampled in Fish Bay South. Additionally, 12 targeted sediment sites (4 in Fish Bay, 8 in Coral Bay) were sampled (Figure 4.1). These targeted sites are co-located with sediment trap sites sampled by researchers from the University of the Virgin Island and the University of San Diego.

Field and Laboratory Methods

Sampling was conducted from June 14th to June 17th 2010 aboard a small boat using a GPS programmed with the station coordinates. Sediment samples were collected using standard NOAA National Status and Trends (NS&T) Program protocols (Apeti et al. 2012a). Briefly, a Ponar sediment grab was deployed to collect the sediment samples. Rocks, 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 a site randomly selected from a list of predetermined alternate sites.

To avoid contamination of samples by equipment and cross contamination between sites, the equipment was rinsed with acetone followed by site water just prior to use. Field personnel handling the samples also 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 WhirlPak® bag, sealed and placed on ice in a cooler. After returning from the field each day, sediment samples were frozen (-15°C) and the WhirlPak® bags for grain size analysis were refrigerated (4°C), to avoid altering the grain size structure of the sediment that could occur during freezing. A suite of water parameters (dissolved oxygen, temperature, and salinity) were measured at each site using a YSI® salinity/conductivity/temperature meter. The instrument probe was submerged to a depth of approximately 0.5 m for surface measurements and approximately 0.5 m from the bottom for measurements at depth. At extremely shallow sites, measurements at depth were not taken. At the end of the mission, samples were shipped overnight to the NS&T contract laboratory (TDI Brooks, International) for analysis.

PAHs were analyzed using gas chromatography/mass spectrometry in the selected ion monitoring (SIM) mode (Kimbrough et al. 2006). PCBs and chlorinated 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).

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, a=0.05) was used to evaluate difference between strata. Data from the targeted sites were included in

the summary statistics as representative of the entire study area, but were not included in the analysis of differences between strata.

Providing Context for Results

In addition to comparing contamination results between strata, there are two primary ways to evaluate the relative level of contamination of Coral and Fish Bays and the surrounding reef ecosystems. First, and most simply, these findings can be compared to the contaminant concentrations from a similar study in St. Thomas, USVI (Pait et al 2013). Second, the degree of sediment contamination in Coral and Fish 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). A NOAA SQG value has not been defined for all analytes; existing ERL and ERM values are presented in Table 4.2. 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). Finally, when SQG are not available for a given pollutant, the values from this study can be placed in a national context by comparing the results to a national contaminant monitoring program, such as NOAA's National Status and Trends (NS&T) Program, which includes sediment chemistry data from over 3000 coastal sites throughout the United States.

Sediment Contaminant Results and Discussion

Organics

PAHs

Concentrations of total PAHs (sum of 58 PAHs measured in this study) in sediments ranged from 2.94 ng/g to 199.08 ng/g (Figure 4.2), with a mean of 31.65 ng/g (Table 4.3). The PAH concentrations measured in this study were slightly lower than PAH values measured in sediments in St. Thomas (Table 4.5). When comparing measured concentrations to published sediment quality guidelines, no sites exceeded the ERL. PAH concentrations in Coral Harbor were significantly higher than in Fish Bay South (Dunn Method, p=0.0126), reflecting more boat traffic in that lobe of Coral Bay. 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 of oil. P/A ratios less than 10 are more indicative of pyrogenic sources; F/P ratios greater than 1 are also thought to be associated with pyrogenic sources. In this study, the F/P ratio

(Appendix 4B, Table B4.1) suggests that most sites have a pyrogenic source of PAHs. The P/A ratio, however, suggests that a petrogenic source is more important. This seemingly conflicting information may be a result of generally low PAH concentrations confounding this comparison technique.

PCBs

Concentrations of total PCBs (sum of 38 PCB congeners analyzed) in sediments ranged from 0.16 ng/g to 1.95 ng/g (Figure 4.3) with a mean of 0.68 ng/g (Table 4.3). This is markedly lower than sediment concentrations measured in St. Thomas (Table 4.5). These concentrations did not exceed any sediment quality guidelines. Statistically, total PCB concentrations did not vary by strata (Dunn Method, a=0.05).

DDT

Concentrations of total DDT (sum of parent compound and its degradation products, DDD and DDE) in sediments ranged from below detection limits to 0.64 ng/g (Figure 4.4), with a mean of 0.03 ng/g (*Table 4.3*). These observed concentrations are similar to what has been observed in St. Thomas (Table 4.5). When comparing measured concentration to published sediment quality guidelines, total DDT did not exceed any NOAA sediment quality guidelines. Statistically, total DDT concentrations did not vary between strata (Dunn Method, a=0.05). 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. Parent material was only detected at two sites of the targeted sites and made up a low percentage (<25%) of the total DDT measured. This suggests that the relatively low DDT levels measured in this system are due to the environmental persistence of this compound, rather than any new inputs into the system (e.g. from illegal applications or a leaking storage container).

HCH

HCH was not observed in the sediments in Coral or Fish Bays, i.e. all samples were below limits of detection (Figure 4.5). This is similar to what was observed in a similar study in St. Thomas (Table 4.5), where HCH was detected only at one site at a low concentration.

Chlordane

Concentration of total chlordane ranged from below detection levels to 0.06 ng/g (Figure 4.6) with a mean of 0.01 ng/g (Table 4.3). This is similar to values previously observed in St.

Thomas (Table 4.5). Two targeted sites in Coral Bay (within Coral Harbor) slightly exceeded the ERL of 0.05 ng/g, suggesting that sediment toxicity is possible at these two sites, but the study area as a whole had low chordane concentrations. There are no statistically significant differences between the strata (Dunn Method, a=0.05). Although restricted in 1983 and banned in 1988, chlordane is environmentally persistent, and historical use as an insecticide is likely to explain the two slightly elevated nearshore sites in Coral Bay.

Other Pesticides

Other pesticides or pesticide degradation products that were detected in sediments included: aldrin, dieldrin, heptachlor, hexachlorobenzene, pentachloroanisole, Endosulfan I, mirex and chlorpyrifos. Spatial distribution of these contaminants are show in Figure 4.A1 to 4.A8 in the Appendix. With one exception (discussed below), these detections were limited to a few sites, and at relatively low concentrations.

Chlorpyrifos was only detected at three sites, but two of these sites were above the NS&T national mean (0.15 ng/g). Both of these sites were in Coral Harbor. Chlorpyrifos was banned for home use in 2001, but it can still be used as an agricultural insecticide (EPA 2006). Because there is not significant agriculture in the Coral Bay watershed, these elevated values are likely due to legacy use in home applications.

Butyltins

Tributyltin (TBT) in sediments ranged from below limits of detection to 10.47 ng/g (Figure 4.7), with a mean of 1.01 ng Sn/g. TBT breakdown products (dibutyltin and monobutyltin) were also detected in similar concentrations (Table 4.3). This is orders of magnitude lower than what has been observed in St. Thomas (Table 4.5). Monobutyltin was significantly higher in Coral Harbor when compared to the Main Bay (Dunn Method, p=0.0034) and Fish Bay South (p=0.0221). Dibutyltin is significantly higher in Coral Harbor when compared to the Main Bay (p=0.0046), Round Bay (p=0.0046) and Fish Bay South (p=0.0277). Similarly, concentrations of TBT are significantly greater in Coral Harbor when compared with Main Bay (Dunn Method, p=.0076), Hurricane Hole (p=0.0076) and Fish Bay South (p=0.0409). Tetrabutyltin, a byproduct of TBT production, was detected at one site in Coral Harbor, where the highest BT concentrations were observed.

Metals

Sixteen trace and major elements were analyzed in sediments collected from Coral and Fish Bays. A summary of the means and standard errors for the elements are shown in Table 4.4. The highest mean concentrations of all trace and major elements are those of silicon (321,000 µg/g), aluminum (49,300 µg/g), and iron (25,700 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 St. Thomas, USVI (Pait et al 2013), 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 ten elements are also provided.

Arsenic

Concentrations of arsenic in the sediments in Coral and Fish Bays ranged from 0.92 µg/g to 7.1 µg/g (Figure 4.8), with a mean of 2.28 µg/g (Table 4.4), which is within the range of values observed for St. Thomas (*Table 4.5*). No sediments analyzed in this study exceeded the ERL for arsenic (8.2 µg/g).

Statistically, when comparing the strata, the Coral Harbor had higher arsenic concentrations than the Main Bay stratum (Dunn Method, p=0.0199).

Chromium

The mean concentration of chromium found in the sediments of Coral and Fish Bays ranged from 2.7 µg/g to 35.4 µg/g (Figure 4.9) with a mean of 10.90 µg/g (Table 4.4). This is similar to what has been measured in other sites in St. Thomas (Table 4.5). No sites exceeded the ERL for chromium (81 µg/g). Statistically, there were no differences between the strata for chromium (Dunn Method, a=0.05).

Copper

Copper concentrations in sediments ranged from 0 µg/g to 38.7 µg/g (Figure 4.10), with a mean concentration of 8.79 µg/g (Table 4.4), which is slightly lower than what has been observed in a similar study in St. Thomas (Table 4.5). Three sites exceeded the ERL (34 µg/g). No sites exceeded the ERM. Two of these sites were in Coral Harbor (at near shore targeted sites) and the third was in Fish Bay North (random site). Statistically, Fish Bay North had higher copper concentrations than Hurricane Hole (Dunn Method, p=0.0498).

Copper can enter the environment from a variety of sources (see discussion above). Anti-fouling paints associated with boat traffic in Coral Harbor may explain high copper concentrations there, although there is heavy boat use across Coral Bay. Fish Bay North is quite shallow and has less boat traffic, so there is likely another source of copper to this system as well.

Mercury

Detected concentrations of mercury from Coral and Fish Bay sediments ranged from 0.001 µg/g to 0.031 µg/g (Figure 4.11), with a mean of 0.006 µg/g (Table 4.4) which is similar to what has been measured another study in St. Thomas (Table 4.5). No sites exceeded the ERL (0.15 µg/g).

Statistically, the Coral Harbor had higher mercury concentrations than Fish Bay South (Dunn Method, p=0.0346).

Nickel

Concentrations of nickel detected in the sediments of Coral and Fish Bays ranged from 0 to 6.09 µg/g (Figure 4.12), with a mean of 0.425 µg/g (Table 4.4), which is an order of magnitude lower than Ni sediment values observed in St. Thomas (Table 4.5). No sites exceeded the ERL (20.9 µg/g). Statistically, there were no differences between strata (Dunn Method, a=0.05).

Zinc

Detected concentrations of zinc in the sediments of Coral and Fish Bays ranged from 0 µg/g to 145 µg/g (Figure 4.13), with a mean concentration of 17.94 µg/g (Table 4.4), which is lower than observed values in a similar study in St. Thomas (Table 4.5). No sites exceeded the ERL (150 µg/g). Statistically, there are no differences between the strata (Dunn Method, a=0.05).

Aluminum

The highest concentration of aluminum detected in this study was in Coral Harbor (49,300 µg/g, Figure 4.14), and the mean was 12,215 µg/g (Table 4.4) which lower than what has been observed at a separate study in St. Thomas (Table 4.5). Statistically, there were no differences between the strata (Dunn Method, a=0.05).

Antimony

Concentration of antimony in Coral and Fish Bay sediments is ranged from below limits of detection to 0.615 µg/g (Figure 4.15), with a mean of 0.095 µg/g (Table 4.4), which is similar to what has been observed in St. Thomas (Table 4.5). Statistically, there were no differences between the strata (Dunn Method, a=0.05).

Cadmium

Concentrations of cadmium in sediments from Coral and Fish Bays ranged from below detection limits to 0.65 µg/g (Figure 4.16), with a mean of 0.023 µg/g (Table 4.4), which is

slightly higher than values observed in St. Thomas (Table 4.5). No sediment sites exceeded any thresholds or guidelines for cadmium. There are no significant differences between strata for cadmium (Dunn Method, a=0.05).

Iron

The highest concentration of iron detected in this study is in Coral Harbor (25,700 µg/g, Figure 4.17), and the mean was 6,260.7 µg/g (Table 4.4) which is similar to what has been observed in a separate study in St. Thomas (Table 4.5). Statistically, there were no differences between strata (Dunn Method, a=0.05).

Lead

The highest concentration of lead detected in the sediments of Coral and Fish Bays is 12.6 µg/g found in Coral Harbor (Figure 4.18). The mean concentration of lead was 2.08 µg/g (Table 4.4), which is within the range of value reported in St. Thomas (Table 4.5). No sediment sites from Coral and Fish Bays exceeded any thresholds or guidelines for lead. Statistically, Coral Harbor had higher lead concentrations than Fish Bay North (Dunn Method, p=0.0374).

Manganese

The highest concentration of manganese detected is 437 µg/g in Coral Harbor (Figure 4.19). The mean concentration of manganese in the sediments of Coral and Fish Bays is 101.58 µg/g (Table 4.4) which is similar to what was observed at a separate study in St. Thomas (Table 4.5). Statistically, there are no difference between the strata (Dunn Method, a=0.05).

Selenium

The highest concentration of selenium detected in sediments from Coral and Fish Bays was 0.916 µg/g in Coral Harbor (*Figure 4.20*). The mean concentration of selenium is 0.200 µg/g (Table 4.4) which is similar to what has been observed in St. Thomas (Table 4.5). Selenium concentrations did not differ between strata (Dunn Method, a=0.05).

Silicon

The highest concentration of silicon detected is 321,000 µg/g at a site in Coral Harbor (Figure 4.21). The mean concentration of silicon in the sediments of Coral and Fish Bays is 62,730 µg/g (Table 4.4), which is slightly higher than what has been observed in a similar study in St. Thomas (Table 4.5). Statistically, there were no differences between sites (Dunn Method, a=0.05).

Silver

Silver concentrations ranged from below limits of detection to 0.081 µg/g (Figure 4.22), with a mean concentration of 0.012 µg/g (Table 4.4). Silver was not detected in sediments in a similar study in St. Thomas (Table 4.5). No sediment sites in this study exceeded any thresholds or guidelines for silver. Silver was significantly higher in the Main Bay (strata) than in Fish Bay North or Hurricane Hole (Dunn Method, a=0.05).

Tin

The highest concentration of tin detected in this study is 1.33 µg/g at a site in Coral Harbor (Figure 4.23). The mean concentration of tin in Coral and Fish Bays sediments is 0.196 µg/g (Table 4.4), which is similar to what has been observed in St. Thomas (Table 4.5). Statistically, there were no differences between the strata (Dunn Method, a=0.05). Although relatively low, Sn concentrations are positively correlated with mono-, di- and tributyltin (a=0.05), with Spearman r values of 0.90, 0.74 and 0.70 respectively. This suggests that Sn levels in these Bays are at least partially driven by the degradation of TBT, which was historically used in anti-foulant boat paints.

Potential Point Sources of Pollutants in the Watershed

A search of the USEPA National Pollution Discharge Elimination System (NPDES) database shows that there is only one permitted discharger in the study area, and this facility in the Coral Bay watershed does not appear to be actively discharging effluent (EPA 2013). Other sources of pollutants to the bays include drain pipes (Image 4.1), several ghuts (i.e. stream channels, Image 4.2), runoff from the town dump site and an abandoned gas station. There is also an informal chandlery on the north shore of Coral Harbor that could be a source of boat related pollutants (T. Smith, pers. comm.).

Relationship of Contaminants with Grain Size and Total Organic Carbon

None of the analytes measured in this study were well correlated (i.e. statistically significant with a Spearman r value greater than 0.7) 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. Similarly, no contaminants were well correlated with grain size, expressed as percent fines (clay and silt). This is also surprising because the increased surface area in fine grained sediments offers more binding sites for contaminants. This may be a function of relatively low contaminant concentrations for most analytes.

Relating Contaminant Levels to Crustal Erosion

Another way to potentially examine the nature of the source of these metals is to look at the ratio of each metal to Al, the primary element in the Earth's crust and generally not considered to be a pollutant. If a metal is well correlated with Al, it is more likely to have a natural (erosional) source (see also, Apeti et al. 2012b). The following trace elements were well correlated ($r > 0.70$) with Al (Spearman, $a=0.05$): As, Cr, Cu, Fe, Mn, Pb, Si, Sn, and Zn. Elements which make up large portions of the Earth's crust (e.g. Si, Mn and Fe) are especially well correlated with Al (i.e. r values of 0.95 or greater). Cu, which exceeded sediment quality guidelines at 3 sites, was also well correlated with Al, although sites with disproportionately high concentrations (relative to Al) were observed (Figure 4.24). These "outliers", located in Coral Harbor and Fish Bay North (red points in Figure 4.10) may suggest that anthropogenic sources of these metals are prevalent at those sites.

Other Water Quality Data

There was very little variability in temperature, salinity and dissolved oxygen measurements (Table 4.6), and nothing to suggest that hypoxia is a problem at any of the sampling sites. The water column does not appear to be stratified, even at the deeper sites. It should be noted that these water column data represent a one time "snapshot" of conditions and are likely to change much faster (i.e. on the scale of hours) than the sediment data presented earlier. Therefore, caution should be used when interpreting these data as this sampling window may not have captured important water quality issues (e.g. hypoxia) that might exist intermittently.

Ecological Significance of Findings

With the exception of copper and chlordane, no contaminants quantified in this study exceeded published sediment quality guidelines, suggesting that the likelihood of toxicity to sediment infauna within the study area is low. It should be noted, however, that not all analytes have guidelines. Furthermore, these guidelines do not consider additive or synergistic effects of multiple toxicants. In order to definitively address sediment toxicity concerns, sediment toxicity assays could be conducted in the future in order to assess potential toxicity in areas where multiple pollutants are highest (e.g. Coral Harbor). These pollutants also have the potential to impact other trophic levels, such as corals and fish. Previous studies in the Caribbean (Pait et al. 2009, Pait et al. 2010, Whitall et al. 2011) have demonstrated that a wide variety of contaminants do accumulate in coral tissues, although the ecological significance of this is not well understood. 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.
Uptake of pollutants by fishes may be of concern not only from an ecological perspective, but also from a seafood safety/human health issue. Future studies could include fish body burden analysis in both commercial and recreational species.

Conclusions

Overall, the chemical contamination of Coral and Fish Bays is fairly low. High levels of pollution are generally limited to the inner portions of bays, especially in Coral Harbor. This data set is an important baseline against which to measure change. It should be noted that watershed management activities may have unintended consequences. For example, in order to reduce erosion a number of dirt roads were paved. However, newly paved roads may cause higher PAH fluxes to the Bays in the future, from PAHs contained in the paving materials or as a result of increased vehicle miles traveled.
There are a variety of potential sources of pollution to Coral and Fish Bays 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.

Image 4.1: Outflow pipe in Fish Bay North near a targeted sampling site (Photo Credit: S. Ian Hartwell, NOAA)

Image 4.2: Ghut draining into Coral Harbor (Photo Credit: S. Ian Hartwell, NOAA)

Table 4.1: 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
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

Table 4.2: Sediment Quality Guidelines (Long and Morgan 1990).

Contaminant	ERL	ERM
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

Sediment quality guideline have not been developed for all analytes monitored by the NOAA's National Status and Trends Program.

Table 4.3: Summary statistics of surficial sediment organic contaminant data for Coral and Fish Bays

	Mean	Median	Max	Min
total PAHs	31.65	12.99	199.08	2.94
total PCBs	0.68	0.62	1.95	0.16
total DDT	0.03	0.00	0.64	0.00
total chlordane	0.01	0.01	0.06	0.00
total HCH	0.00	0.00	0.00	0.00
Monobutyltin	1.38	0.00	10.39	0.00
Dibutyltin	0.88	0.00	8.62	0.00
Tributyltin	1.01	0.00	10.47	0.00
Tetrabutyltin	0.01	0.00	0.22	0.00

Table 4.4: Summary statistics of surficial sediment inorganic contaminant data for Coral and Fish Bays (random and targeted sites).

	Min	Max	Mean	Median
Ag	0	0.081	0.012	0
Al	493	49300	12215.7	6535
As	0.921	7.1	2.826	2.28
Cd	0	0.65	0.023	0
Cr	2.7	35.4	10.907	9.005

Table 4.5: Comparison of St. John, USVI (this study) with a previous study in St. Thomas, USVI (Pait et al. 2013)

	St. Thomas			St. John		
	Mean	Maximum	Std Dev	Mean	Maximum	Std Dev
Total PAHs	142	1131	285	31.65	199.08	46.22
Total Chlordane	0.04	0.33	0.07	0.01	0.06	0.02
Total DDT	0.05	0.61	0.12	0.03	0.64	0.11
Total PCBs	1.00	7.2	1.58	0.68	1.95	0.36
Tributyltin	1.85	31.1	6.38	1.01	10.47	2.34
Ag	0.00	0.00	0.00	0.01	0.08	0.03
Al	13596	63800	19019	12215.7	49300	13553.07
As	2.74	12.4	3.50	2.83	7.1	1.67
Cd	0.03	0.26	0.08	0.02	0.65	0.10
Cr	14.1	35.7	8.62	10.91	35.4	6.51
Cu	21.0	155.0	36.6	8.79	38.7	11.74
Fe	8547	40900	12347	6260.7	25700	6583.85
Hg	0.02	0.11	0.03	0.01	0.03	0.01
Mn	89.0	338	99.8	101.58	437	99.91
Ni	6.53	15.1	2.81	0.42	6.09	1.37
Pb	5.87	31.0	9.32	2.08	12.6	2.39
Sb	0.12	0.82	0.25	0.10	0.615	0.15
Se	0.24	0.93	0.22	0.20	0.961	0.24
Si	48340	181000	55143	62729.75	321000	77208.15
Sn	0.61	3.95	1.15	0.20	1.33	0.35
Zn	37.3	159	52.3	17.94	145	27.68

Table 4.6: Sonde (YSI) data for surface and bottom water. Note: shallow sites did not have data from bottom, so bottom data are biased towards deeper sites.

	Surface				Bottom			
	Mean	Median	Min	Max	Mean	Median	Min	Max
Temp (degrees C)	30.48	30.50	29.60	31.60	30.00	29.70	29.50	31.00
Salinity (PSU)	35.16	35.34	31.16	35.46	35.34	35.35	35.25	35.43
DO (mg/L)	6.78	6.56	5.63	8.89	6.48	6.46	6.00	7.44
Depth (m)	surface	surface	surface	surface	9.27	5.49	0.61	23.16

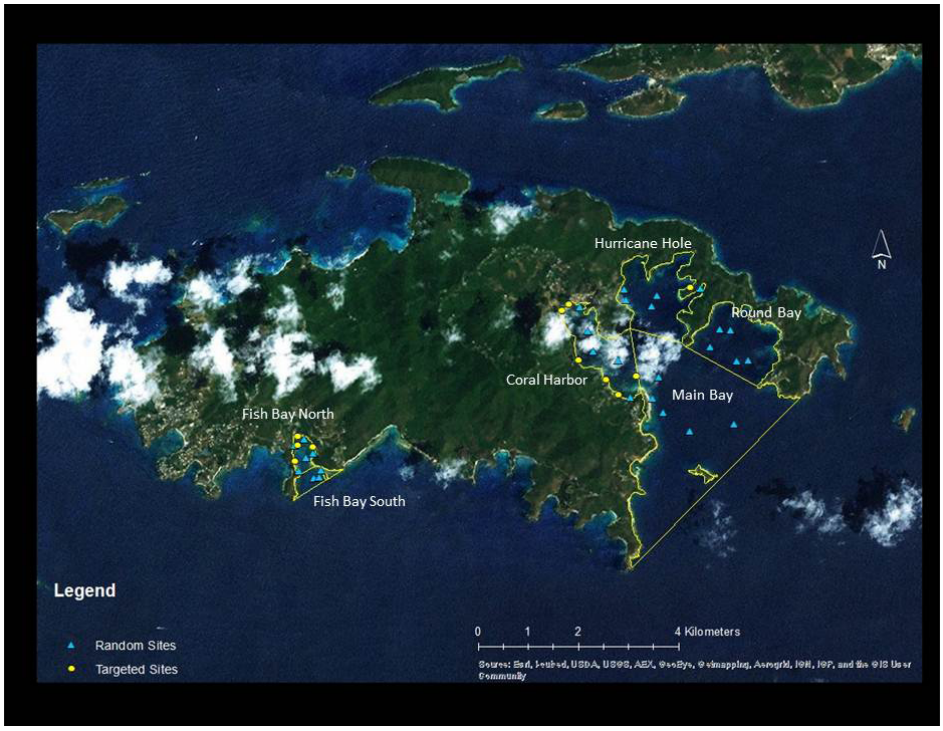


Figure 4.1: Sediment sampling sites from June, 2010. Randomly selected sites (round dots) were selected in six geographic strata (yellow polygons) in a stratified random design. Additionally, 12 targeted sites (diamonds) were sampled which were co-located with sediment trap locations.

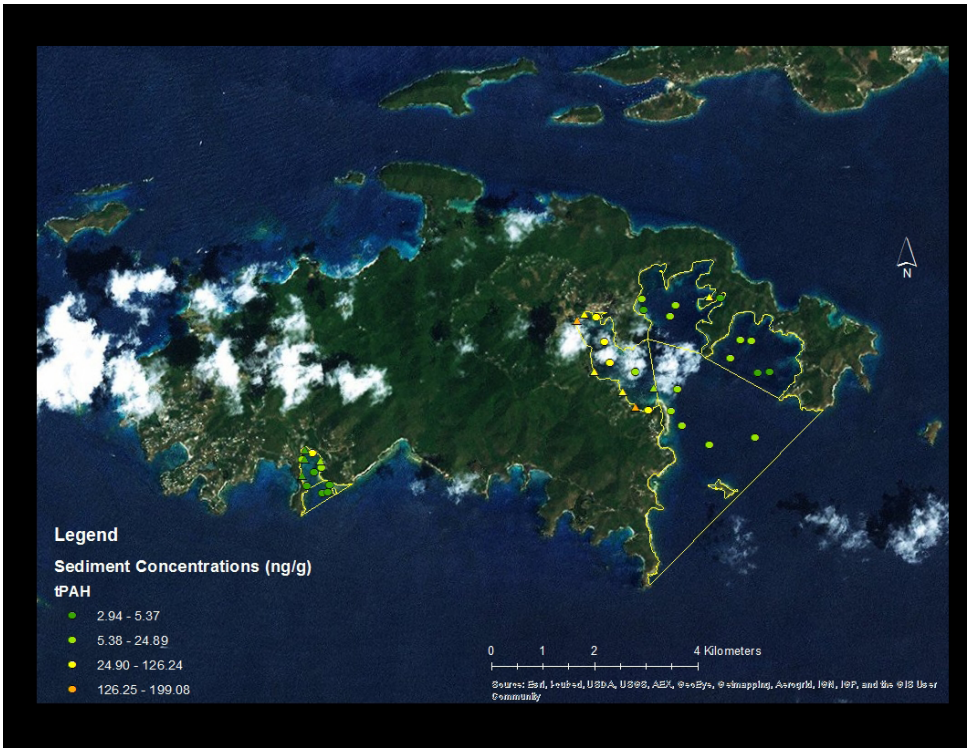


Figure 4.2: Concentrations of total PAHs in sediments.

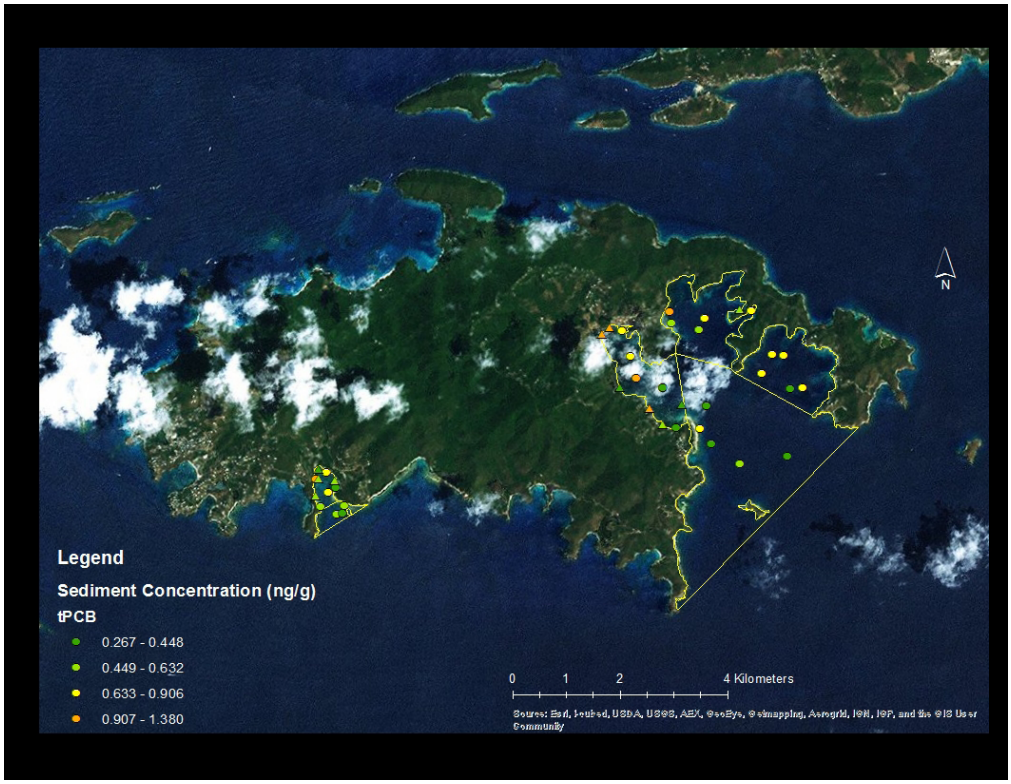


Figure 4.3: Concentrations of total PCBs in sediments.

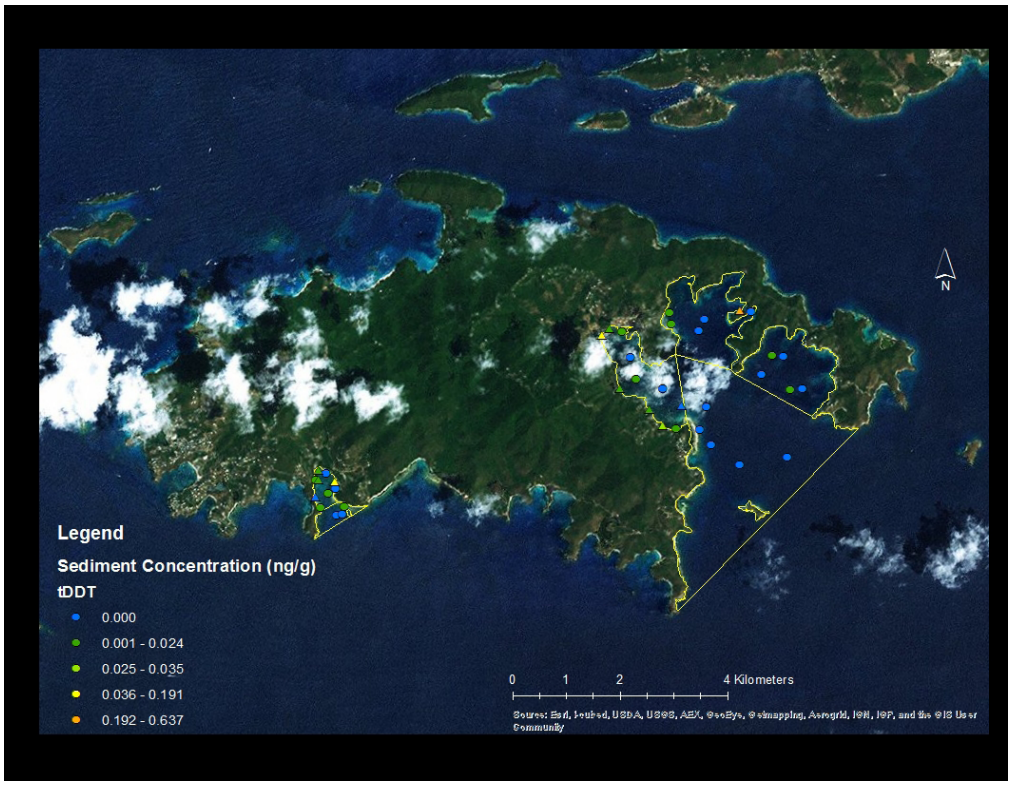


Figure 4.4: Concentrations of total DDT in sediments.

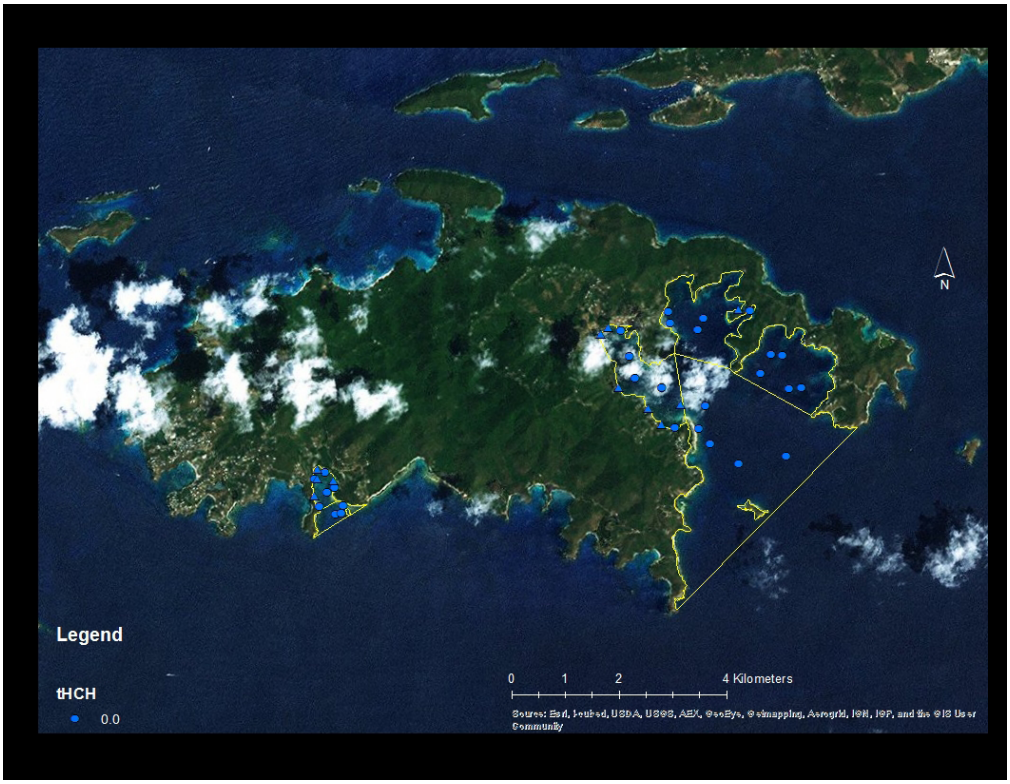


Figure 4.5: Concentrations of total HCH in sediments.

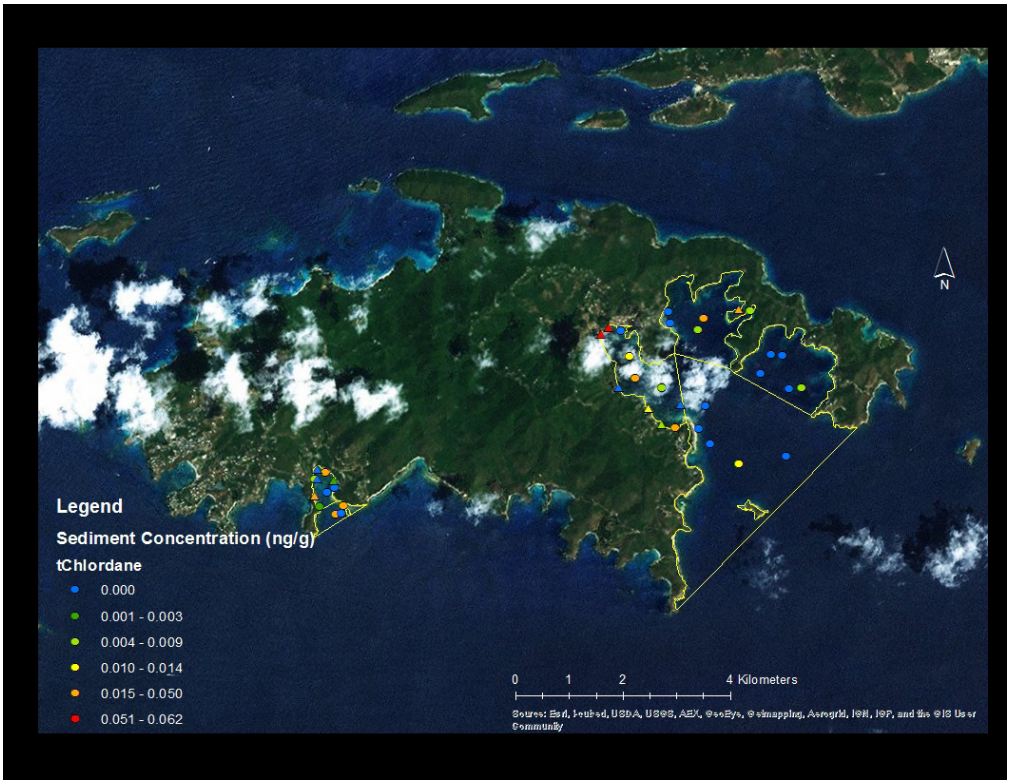


Figure 4.6: Concentrations of total chlordane in sediments. Red indicates an exceedance of the ERL (0.05 ng/g).

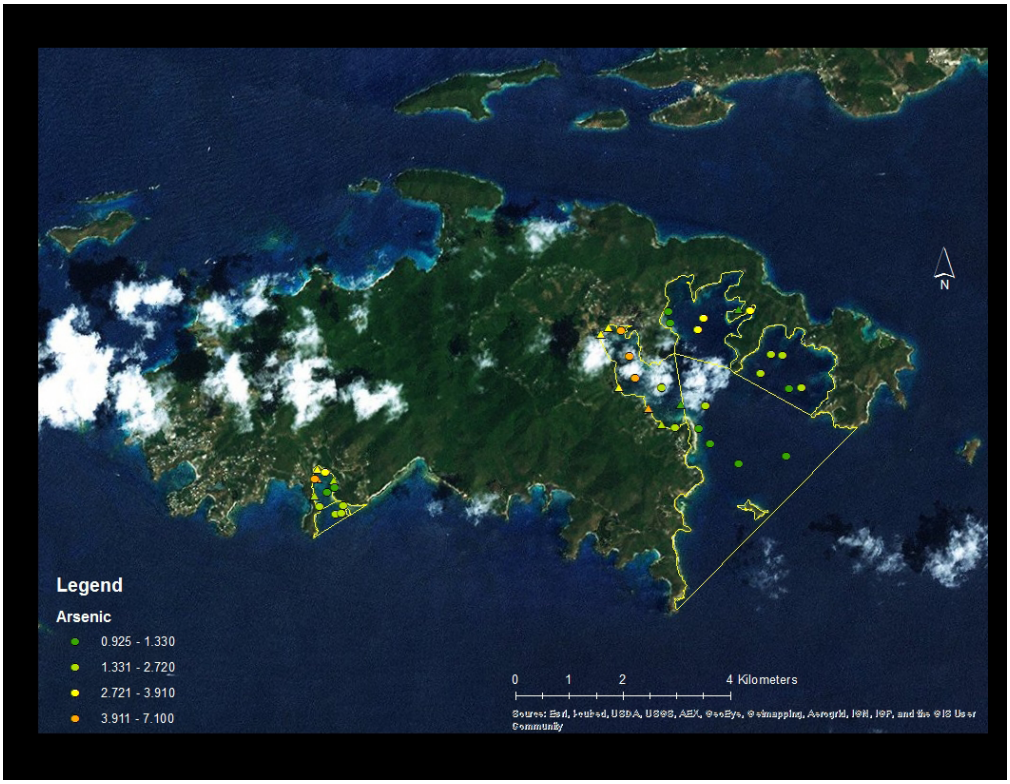


Figure 4.7: Concentrations of TBT in sediments.

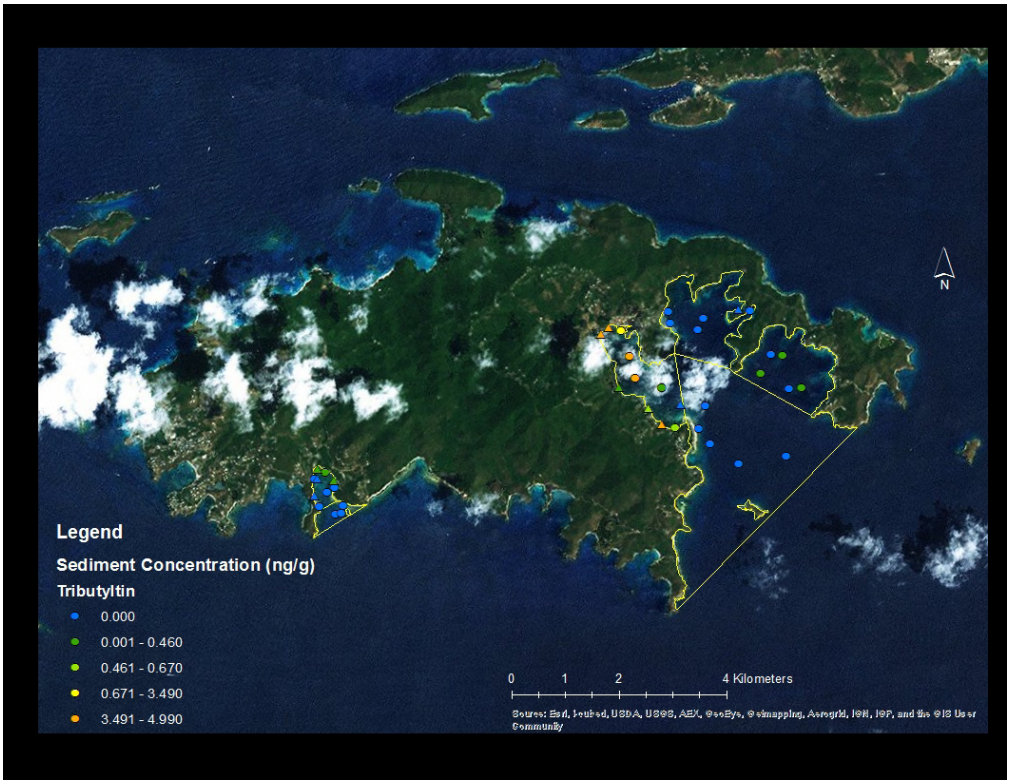


Figure 4.8: Concentrations of arsenic in sediments

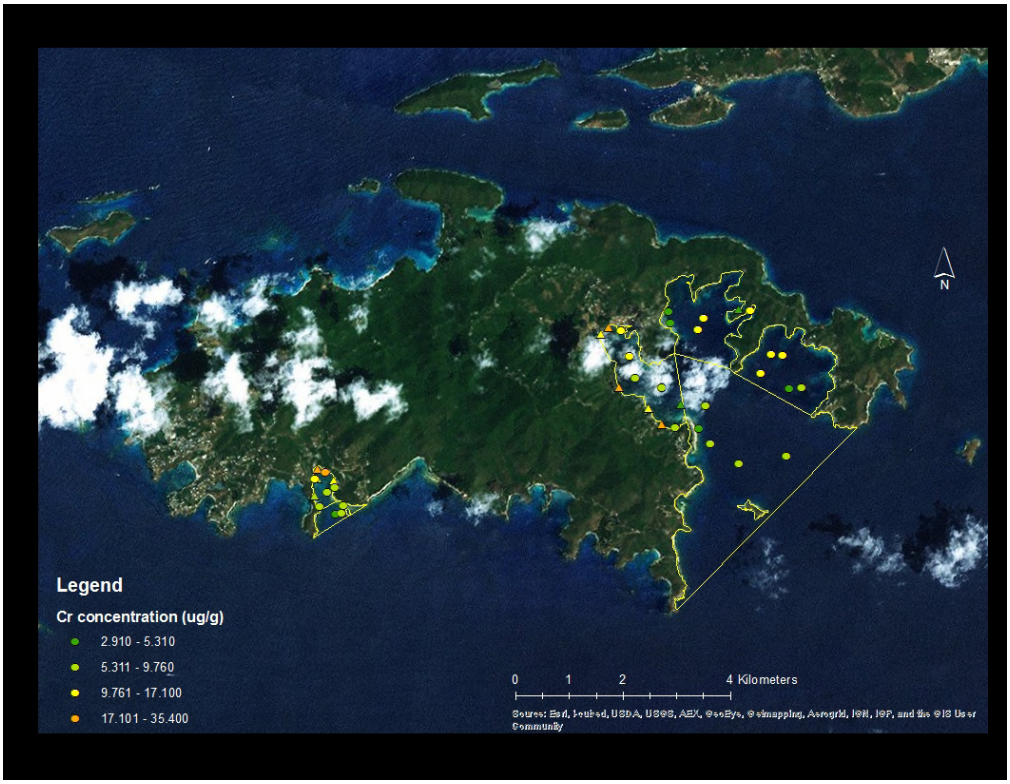


Figure 4.9: Concentrations of chromium in sediments.

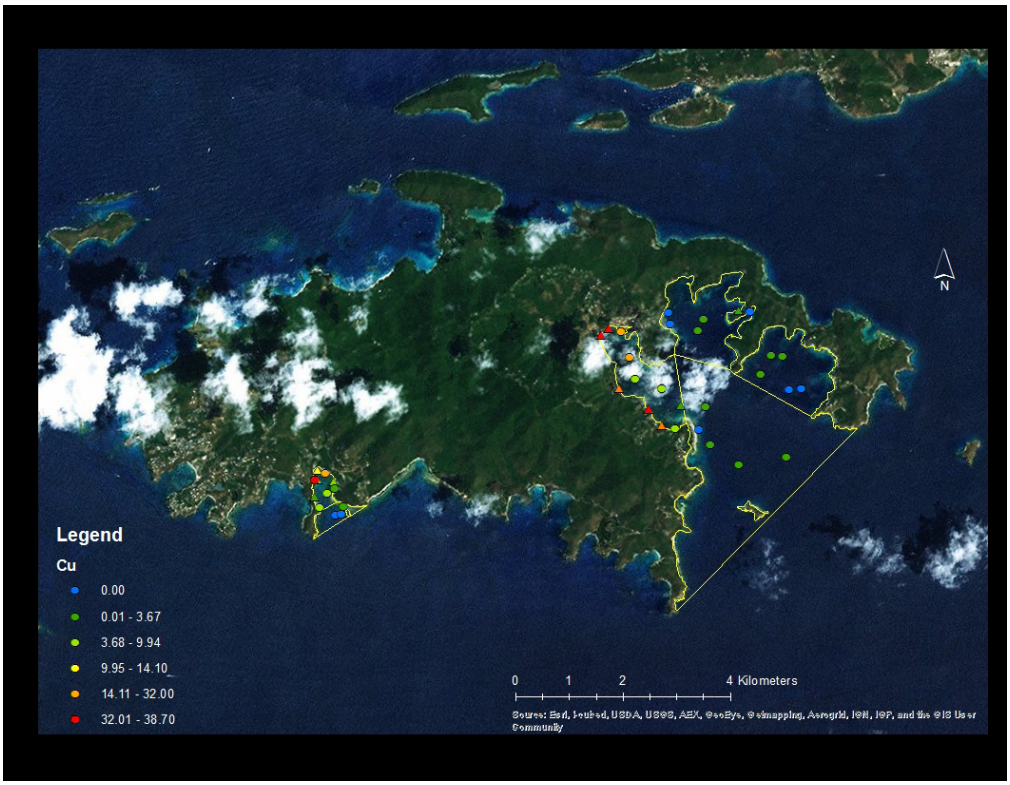


Figure 4.10: Concentrations of copper in sediments. Red indicates an exceedance of the ERL (34 ug/g).

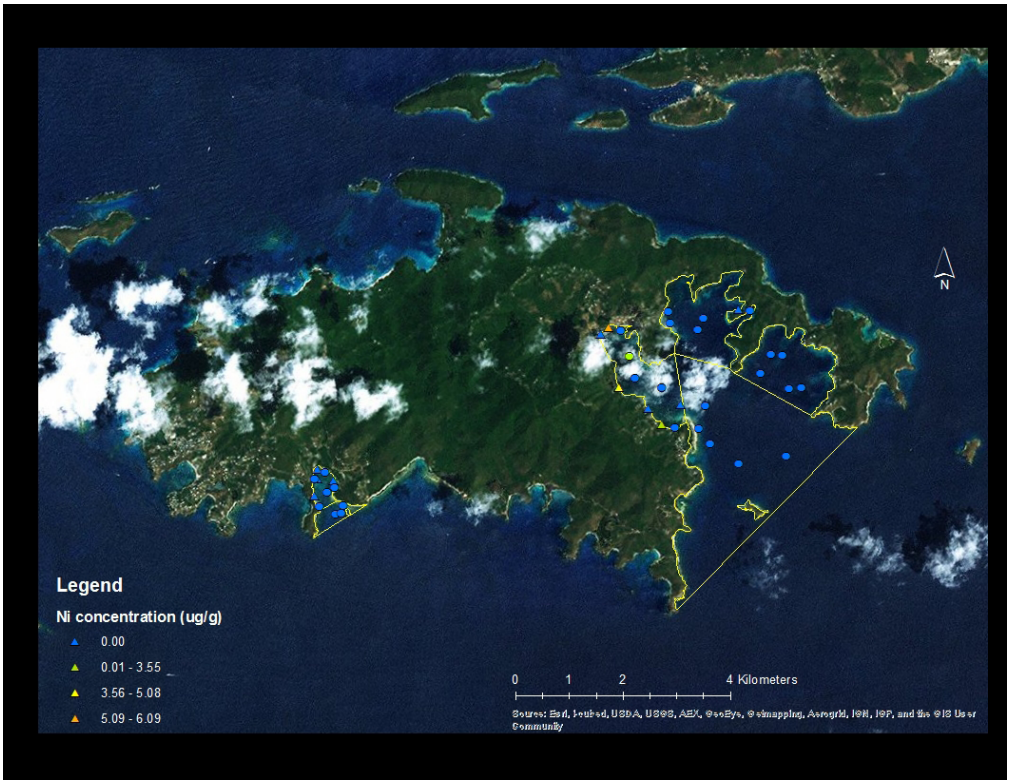


Figure 4.11: Concentrations of mercury in sediments.

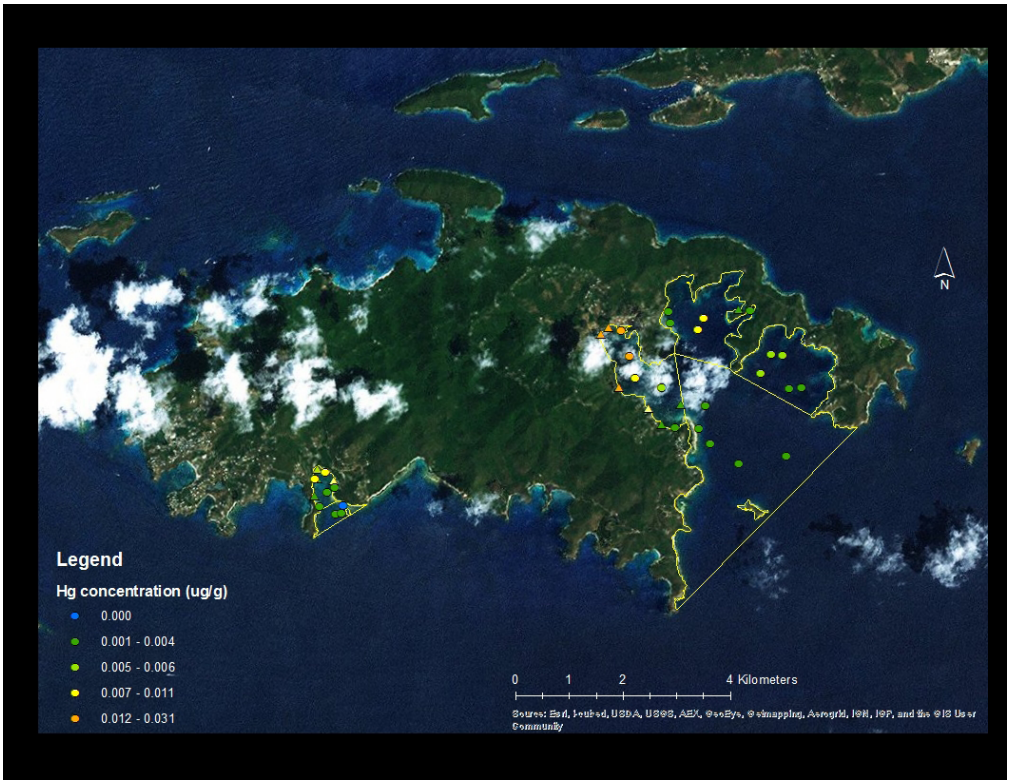


Figure 4.12: Concentrations of nickel in sediments.

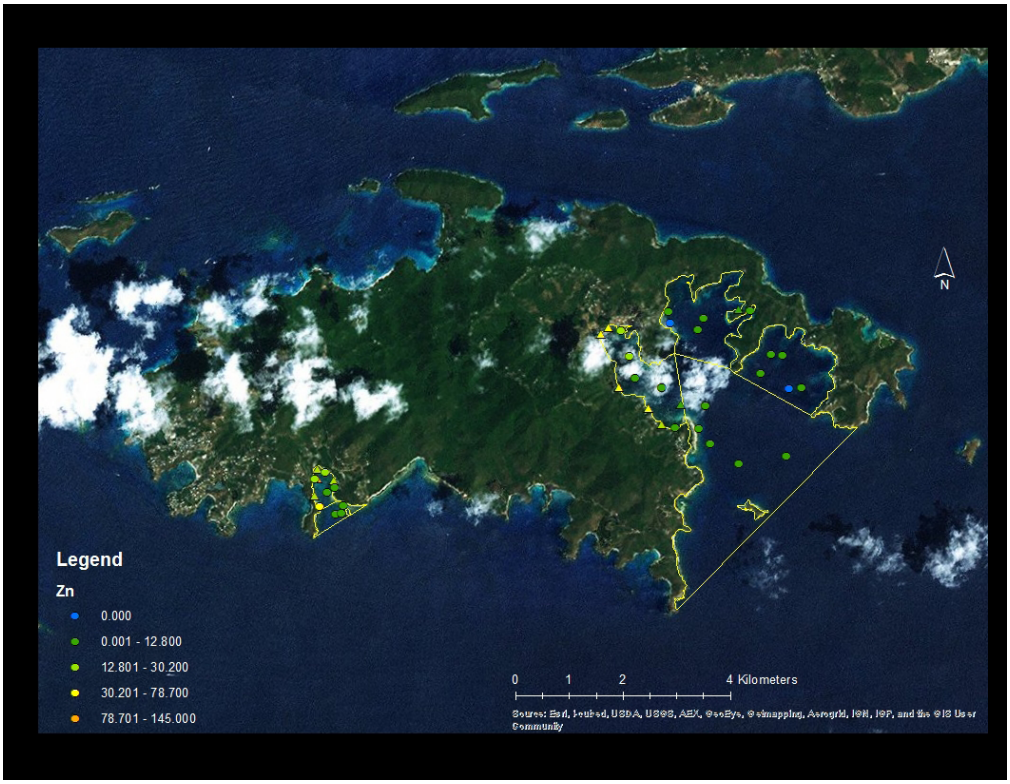


Figure 4.13: Concentrations of zinc in sediments

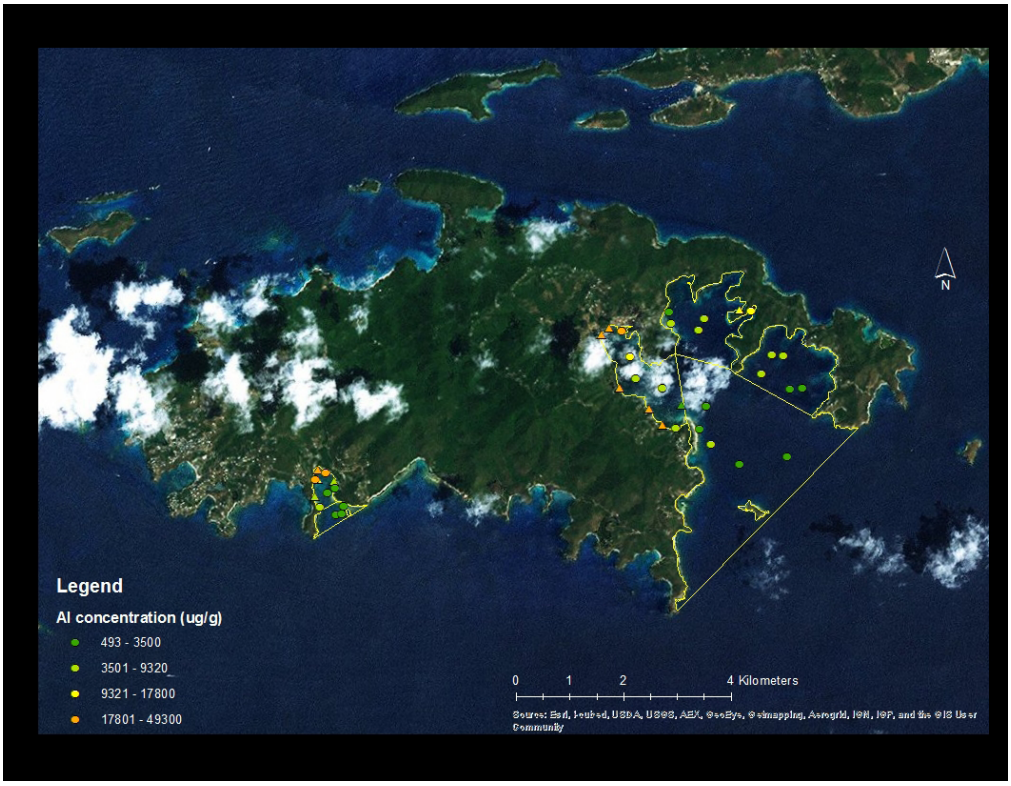


Figure 4.14: Concentrations of aluminum in sediments.

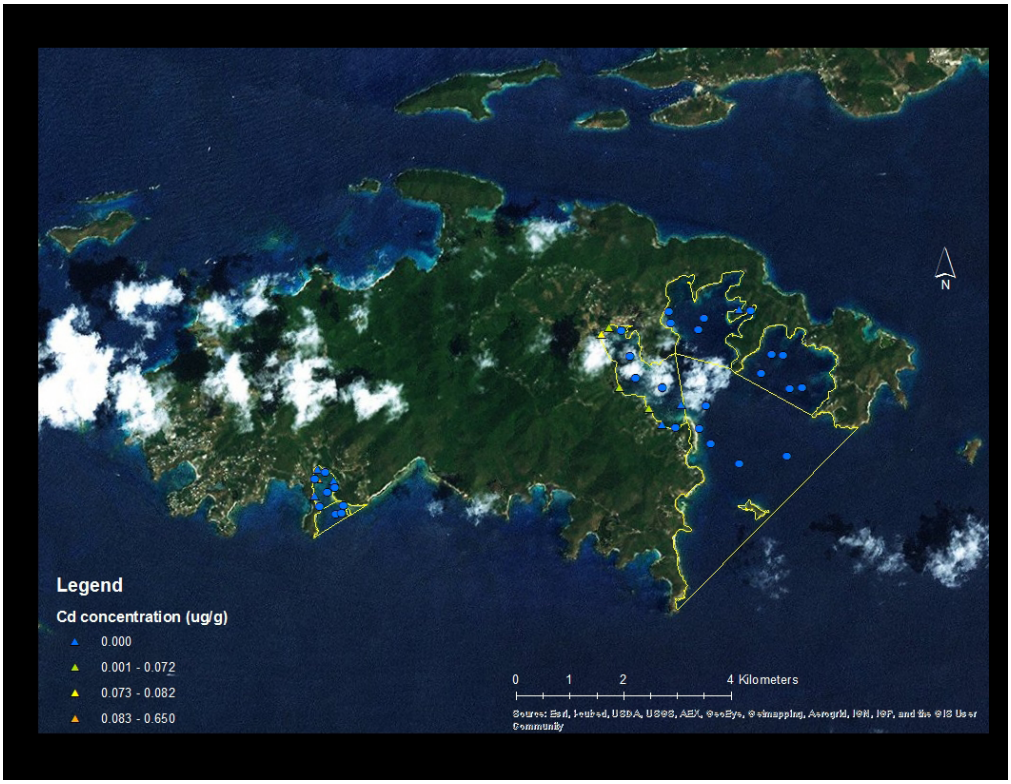


Figure 4.15: Concentrations of antimony in sediments.

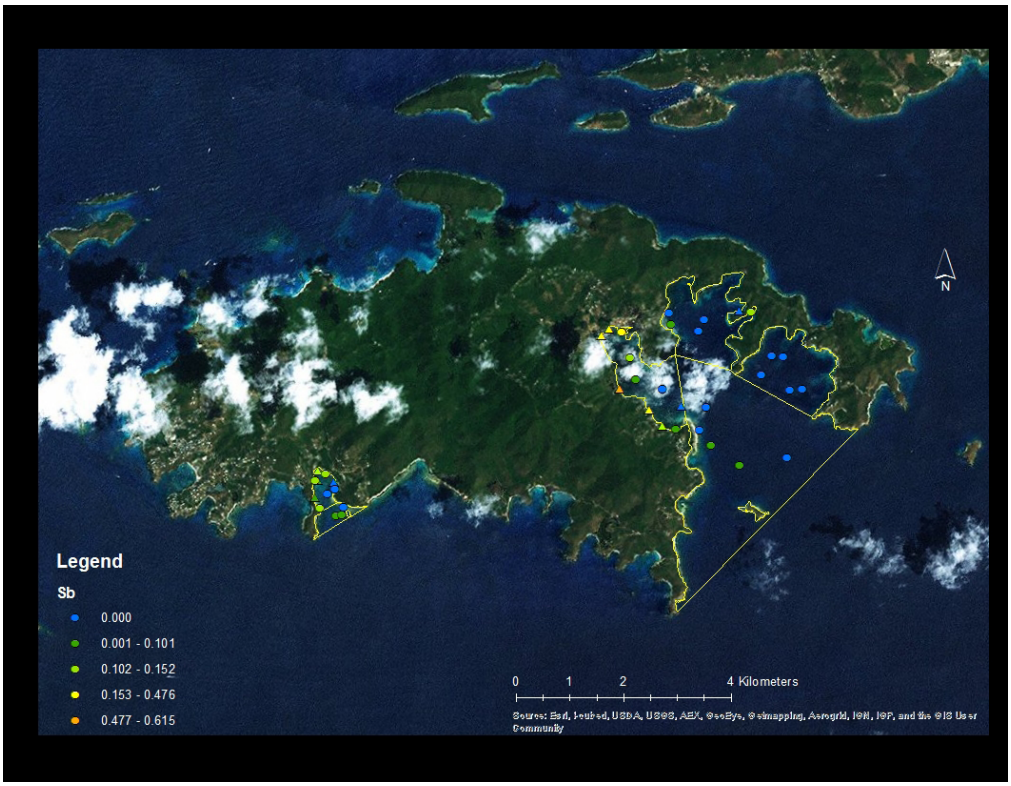


Figure 4.16: Concentrations of cadmium in sediments.

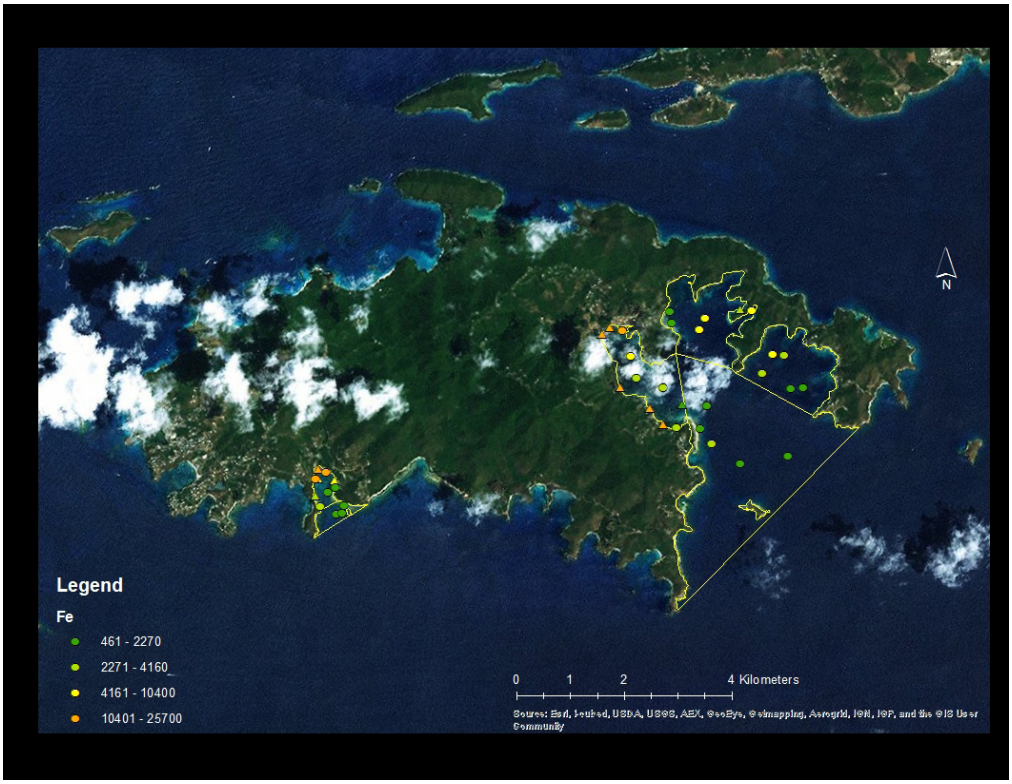


Figure 4.17: Concentrations of iron in sediments.

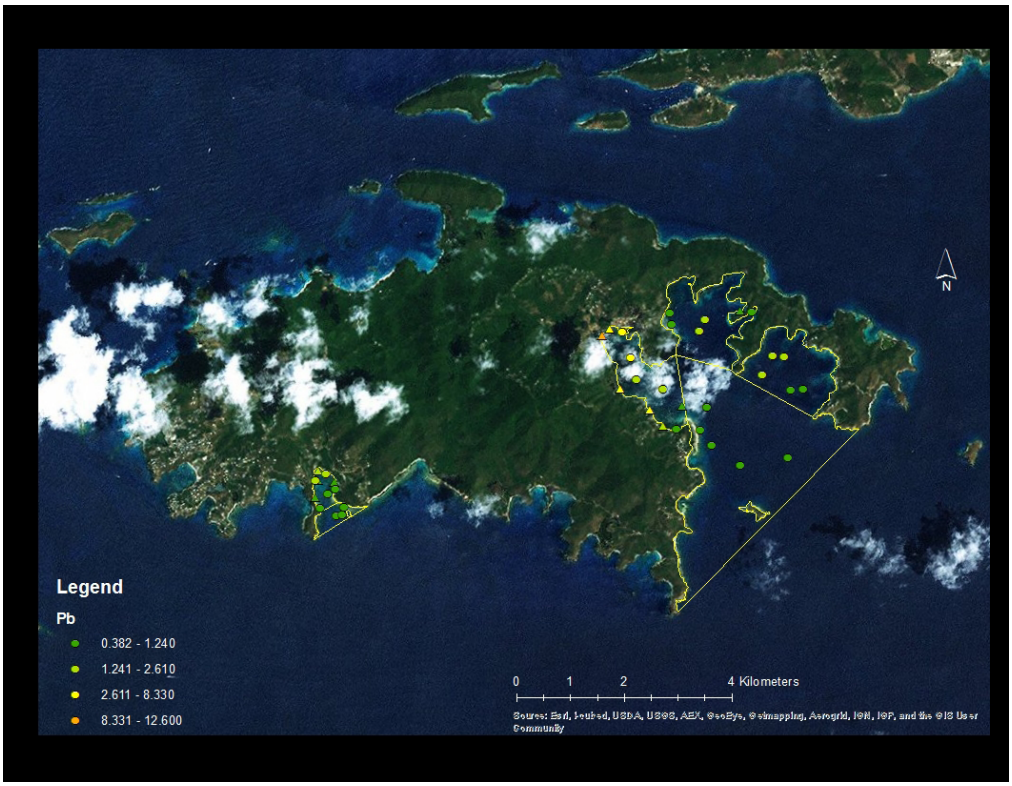


Figure 4.18: Concentrations of lead in sediments.

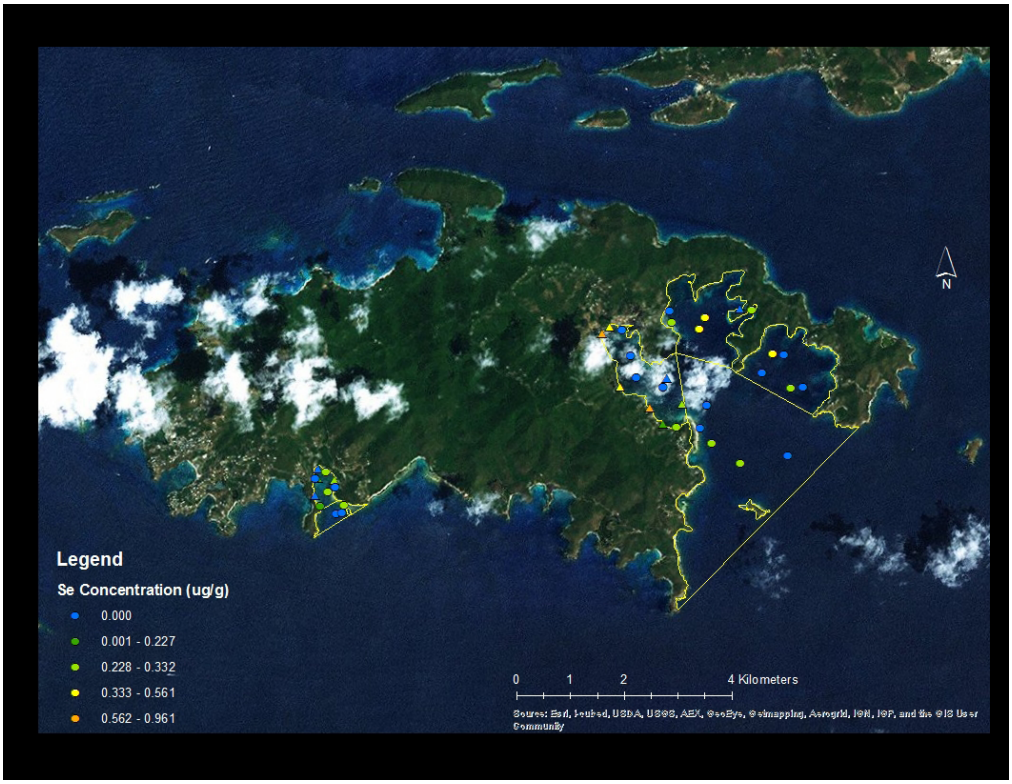


Figure 4.19: Concentrations of manganese in sediments.

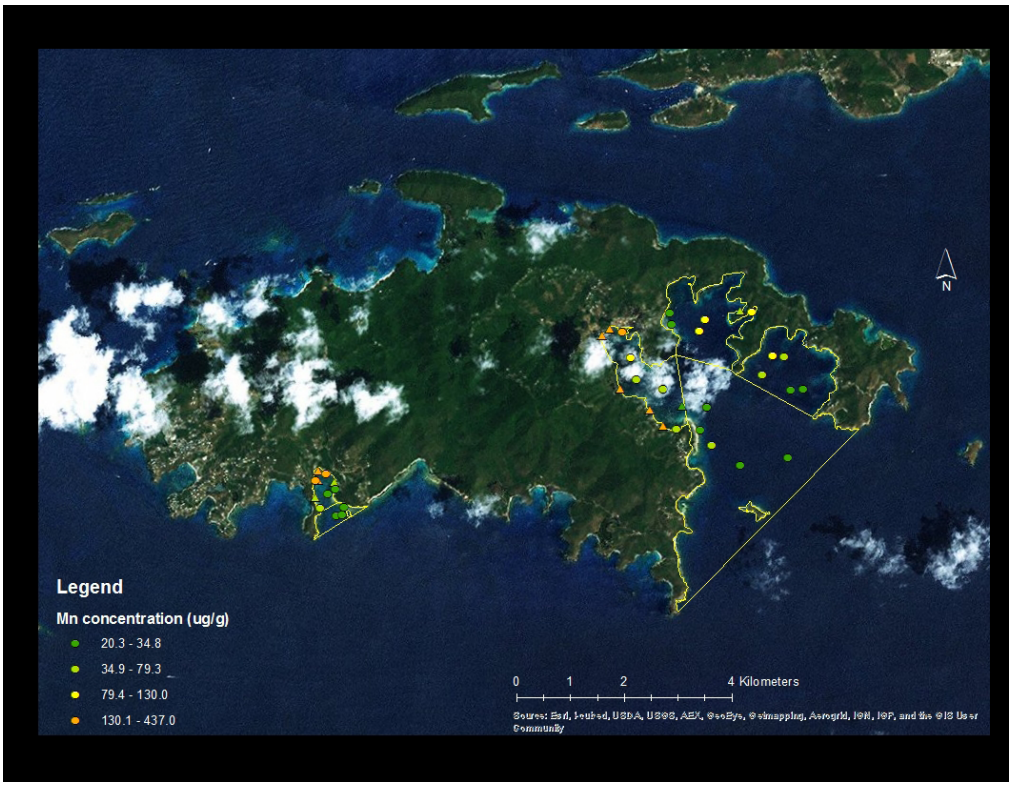


Figure 4.20: Concentrations of selenium in sediments.

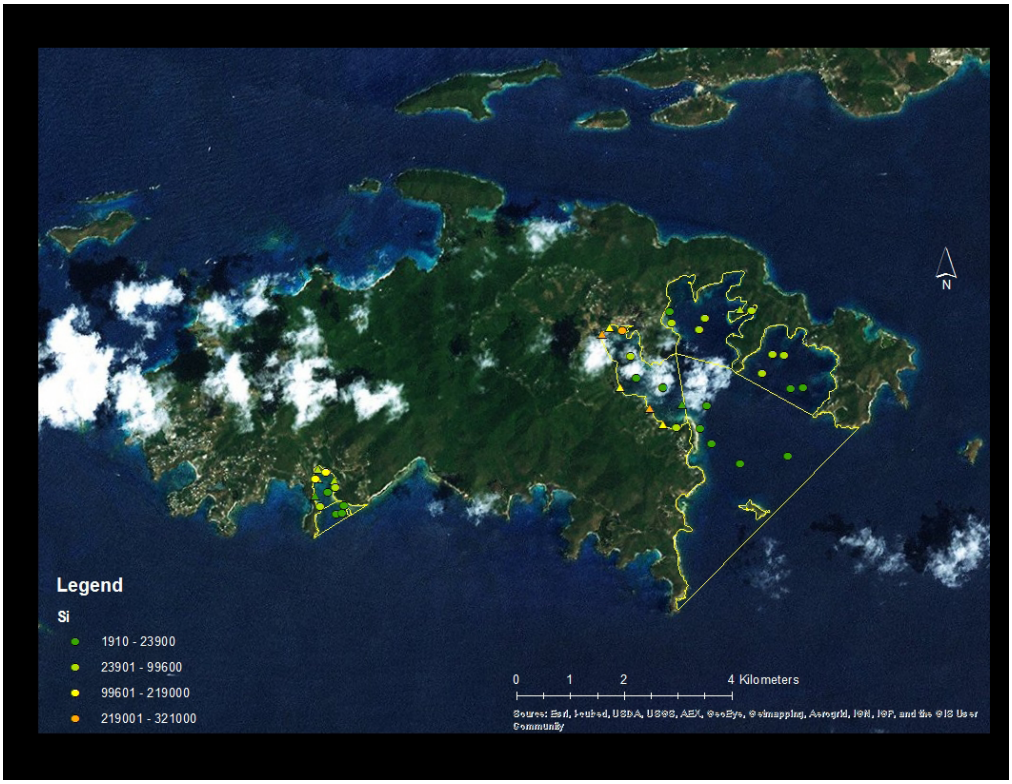


Figure 4.21: Concentrations of silicon in sediments.

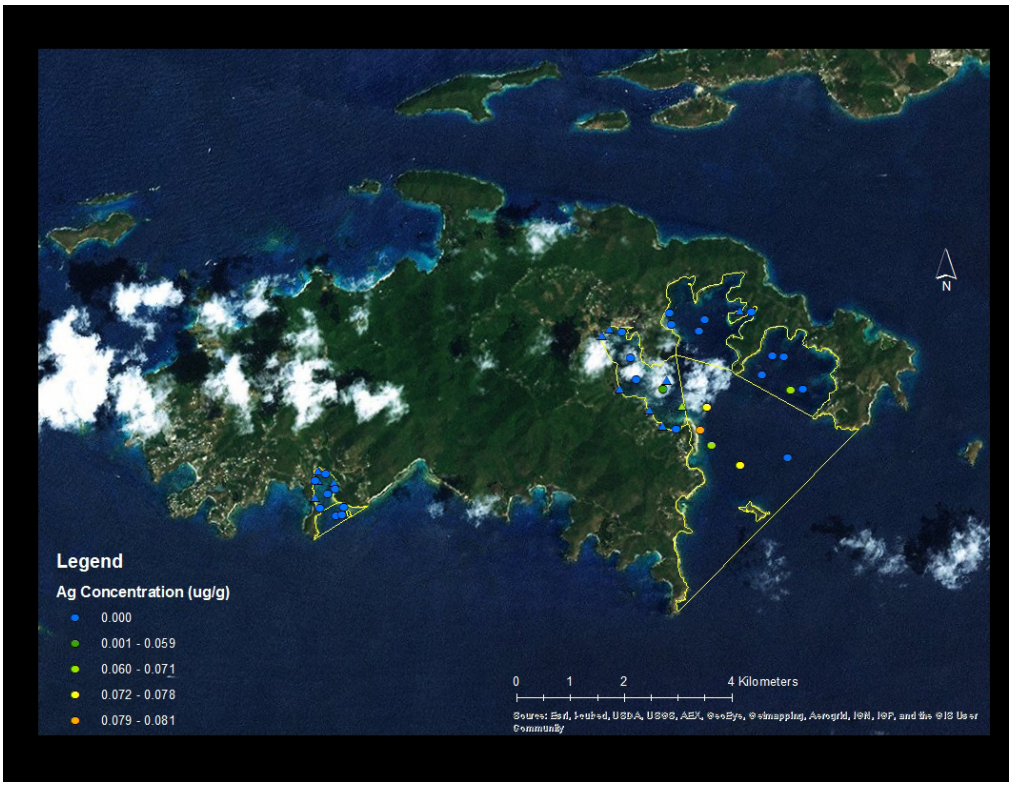


Figure 4.22: Concentrations of silver in sediments.

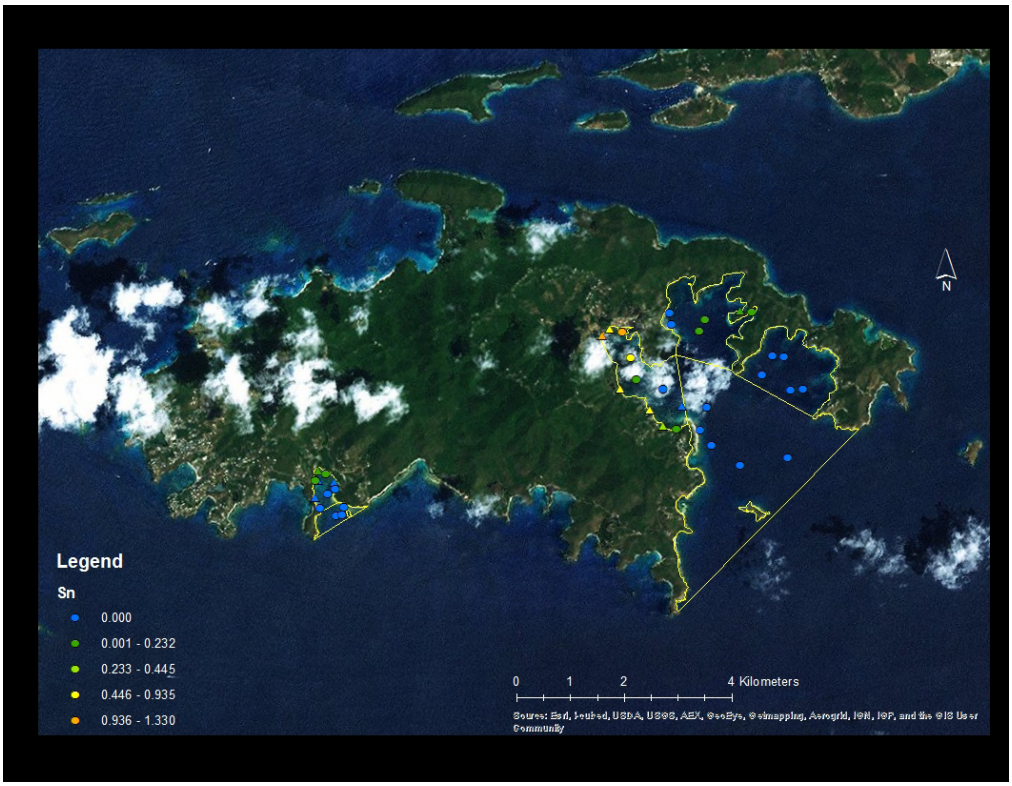


Figure 4.23: Concentrations of tin in sediments.

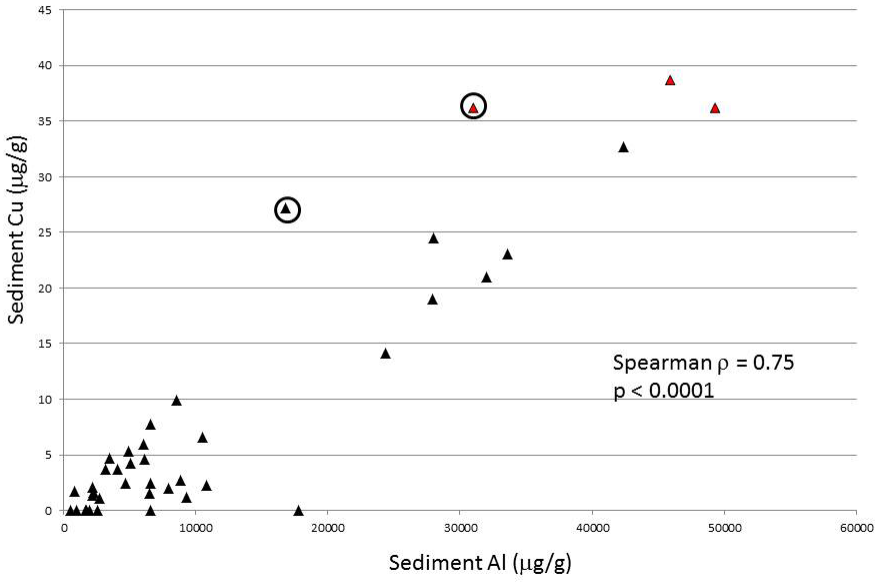


Figure 4.24: Correlation between sediment concentrations of Cu and Al. Red points are values which exceed the ERL (indicating possible sediment toxicity). Circled points show higher Cu concentrations that would be expected from crustal erosion alone (i.e. these concentrations are likely anthropogenically enhanced) ρ value is Spearman rho (ά=0.05).

Chapter 5
Conclusions and Management Applications

Overall Conclusions

This study represents an interdisciplinary data set which enhances our understanding of the ecosystems of Coral and Fish Bays and 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.

The Coral Reef Ecosystem Monitoring (CREM) biological data set provides good spatiotemporal coverage and will be useful for detecting change in Coral and Fish Bays. In Coral Bay, while estimate of algal and coral cover varied over time, portions of Coral Bay could be considered coral “hot spots.” Fish Bay generally had lower coral cover and more algae than Coral Bay.

Distance from point source watershed discharge, and therefore presumably a gradient in sedimentation, was correlated with changes in both macroalgae and scleractinian coral metrics. This suggests that as sedimentation is reduced, biological change may result.

Sediment contamination of Coral and Fish Bays was generally low, but Coral Bay had three sites for copper and two sites for chlordane had sufficiently high concentrations to suggest potential toxicity to sediment infauna. Because contaminant threshold values do not exist for coral, it is unclear what effect the observed contaminant levels might have on coral health. Future sampling of coral or fish tissues would be informative in assessing ecosystem effects of these pollutants, and, in the latter case, assess potential human health issues associated with local fish consumption.

This suite of environmental data (biological and stressors) represent an important baseline against which to measure future change, e.g. 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 times scales ranging from relatively short term responses in sediment loading to potentially decadal long recovery processes for reef systems.

Management Applications

The desired societal condition that is being managed towards is improved coral reef ecosystem health. In order to track progress towards meeting this goal, a starting point, or baseline, is necessary against which to measure change. This study provides that critical piece of information that allows for coastal managers to evaluate the effectiveness of watershed management activities. It is important to remember that this baseline represents status quo at the time of this study, not the natural status of this system.

Managing towards improved coral reef ecosystem health is a complex and difficult task. In addition to land based sources of pollution (sediments, nutrients, contaminants), reef ecosystems may also be stressed by overfishing, physical damage to reefs from boats or divers and climate change. Furthermore, coastal development pressures in the watersheds have the potential to offset gains through active watershed restoration. Managing these stressors piecemeal may result in a failure to achieve the desired ecological endpoint. This points to the need for an integrated coral reef ecosystem management plan that involves the cooperation of local (e.g. Coral Bay Community Council), territorial (e.g. USVI Department of Planning and Natural Resources), and federal (e.g. U.S. Environmental Protection Agency).

In order to assess the efficacy of management actions, ongoing monitoring and future assessments are needed. Managers must construct their criteria for success with understanding that some metrics of coral reef ecosystem health may respond quickly (e.g. sediment load) and others may require many years to recover (e.g. percent coral cover).

Acknowledgments

This work was funded by the Caribbean Coral Reef Institute, with personnel support from NOAA's National Centers for Coastal Ocean Science, and Southeast Fisheries Science Center. The authors would like to thank Greg Piniak, Suzanne Bricker, Andrew Mason (NOAA) and Tyler Smith and Renata Platenberg (UVI) for their helpful reviews and editorial comments.

References

Chapter 1

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

Beets, J. and C.S. Rogers. 2002. Decline of fishery resources in marine protected areas in the US Virgin Islands: the need for marine reserves. Proc 9th International Coral Reef Symposium:449-454.

Bellwood, D.R., T.P. Hughes, C. Folke, and M. Nystrom. 2004. Confronting the coral reef crisis. Nature 429:827833.

Brooks, G.R., B. Devine, R.A. Larson, and B.P. Rood. 2007. Sedimentary Development of Coral Bay, St. John, USVI: A Shift From Natural to Anthropogenic Influences. Caribbean Journal of Science 43(2):226-243.

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, D.C. 56 pp.

Connell, J.H. 1997. Disturbance and recovery of coral communities. Coral Reefs 16(Suppl):S101-S113.

Coral Bay Community Council (CBCC). 2012. Coral Bay, St. John – Our Future based on Balancing Environmental Protection & Development. http://www.aia.org/aiaucmp/groups/aia/documents/pdf/aiab098288.pdf

Dahl, A.L. 1985. Status and conservation of South Pacific coral reefs. Proc. 5th int. Coral Reef Congr. 6:509513.

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

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

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.

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

Hughes, T. 1994. Catastrophes, phase shifts and large-scale degradation of a Caribbean coral reef. Science 265:1547–1551.

Jeffrey, C.F.G., U. Anlauf, J. Beets, S. Caseau, W. Coles, A. Friedlander, S. Herzlieb, Z. Hillis-Starr, M. Kendall, V. Mayor, J. Miller, R.S. Nemeth, C.S. Rogers, and W. Toller. 2005. The State of Coral Reef Ecosystems of the U.S. Virgin Islands. pp. 45-88. In: J.E. Waddell (ed.). The state of coral reef ecosystems of the United States and Pacific Freely Associated States: 2005. NOAA Technical Memorandum NOS NCCOS 11. Silver Spring, MD. 522 pp.

Miller, J., R. Waara, A. Atkinson, E. Muller, and C.S. Rogers. 2006. Coral bleaching and disease combine to cause extensive mortality on reefs in US Virgin Islands. Coral Reefs 25(3):418-425.

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

Ramos-Scharrón, C.E. and L.H. MacDonald. 2005. Measurement and prediction of sediment production from unpaved roads, St. John, U.S. Virgin Islands. Earth Surface Processes and Landforms. 30:1283–1304.

Ramos-Scharrón, C.E. 2012. Effectiveness of drainage improvements in reducing sediment production rates from an unpaved road. Journal of Soil and Water Conservation 67(2):87-100.

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

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

Rogers, C.S. and J. Beets. 2001. Degradation of marine ecosystems and decline of fishery resources in marine protected areas in the US Virgin Islands. Environ. Conserv. 28 (4):312-322.

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

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

Rogers, C.S., J. Miller, and E.M. Muller. 2008b. Coral diseases following assive bleaching in 2005 cause 60 percent decline in coral cover and mortality of the threatened species, Acropora palmata, on reefs in the U.S. Virgin Islands. USGS Fact Sheet 2008-3058.

Waddell, J.E. and A.M. Clarke (eds.) 2008. The State of Coral Reef Ecosystems of the United States and Pacific Freely Assocaited States: 2008. NOAA Technical Memorandum NOS NCCOS 73. NOAA/NCCOS Center for Coastal Monitoring and Assessment's Biogeography Team. Silver Spring, MD. 569 pp.

Wilkinson, C. 2004. Status of coral reefs of the world: 2004, Vol. 1. Global Coral Reef Monitoring Network.

Chapter 2

Airoldi, L. 2003. The effects of sedimentation on rocky coastal assemblages. Oceanography and Marine Biology Annual Review 41:161–236.

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

Beets, J. and C.S. Rogers. 2002. Decline of fishery resources in marine protected areas in the US Virgin Islands: the need for marine reserves. Proc 9th International Coral Reef Symposium:449-454.

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

Boulon, R.H. 1992. Use of mangrove prop roots habitats by fish in the northern U.S. Virgin Islands. Proc. Gulf Carib. Fish. Inst. 41:189-204.

Brooks, G.R., B. Devine, R.A. Larson, and B.P. Rood. 2007. Sedimentary Development of Coral Bay, St. John, USVI: A Shift From Natural to Anthropogenic Influences. Caribbean Journal of Science 43(2):226-243.

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, D.C. 56 pp.

Connell, J.H. 1997. Disturbance and recovery of coral communities. Coral Reefs 16(Suppl):S101-S113.

Dahl, A.L. 1985. Status and conservation of South Pacific coral reefs. Proc. 5th int. Coral Reef Congr. 6:509-513.

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

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

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.

Friedlander, A.M. and J. Beets. 2008. Temporal Trends in Reef Fish Assemblages inside Virgin Islands National Park and around St. John, U.S. Virgin Islands, 1988-2006. NOAA Technical Memorandum 70. Silver Spring, MD. 60 pp.

Friedlander, A.M., C.F.G. Jeffrey, S.D. Hile, and S.J. Pittman. 2012. Coral reef ecosystems of St. John, U.S. Virgin Islands: Spatial and temporal patterns in fish and benthic communities (2001-2009). NOAA Technical Memorandum 152. Silver Spring. MD.

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

Hughes, T. 1994. Catastrophes, phase shifts and large-scale degradation of a Caribbean coral reef. Science 265:1547–1551.

Jeffrey, C.F.G., U. Anlauf, J. Beets, S. Caseau, W. Coles, A. Friedlander, S. Herzlieb, Z. Hillis-Starr, M. Kendall, V. Mayor, J. Miller, R.S. Nemeth, C.S. Rogers, and W. Toller. 2005. The State of Coral Reef Ecosystems of the U.S. Virgin Islands. pp. 45-88. In: J.E. Waddell (ed.). The state of coral reef ecosystems of the United States and Pacific Freely Associated States: 2005. NOAA Technical Memorandum NOS NCCOS 11. Silver Spring, MD. 522 pp.

Kendall, M.S., C.R. Kruer, K.R. Buja, J.D. Christensen, M. Finkbeiner, and M.E. Monaco. 2001. Methods Used to Map the Benthic Habitats of Puerto Rico and the U.S. Virgin Islands. NOAA/NOS Biogeography Program Technical Report. Silver Spring, MD. 45 pp.

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

Larsen, M.C. and R.M.T. 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.

McClanahan, T.R. and D. Obura. 1997. Sedimentation effects on shallow coral communities in Kenya. Journal of Experimental Marine Biology and Ecology 209(1-2):103-122.

Menza, C., J. Ault, J. Beets, J. Bohnsack, C. Caldow, J. Christensen, A. Friedlander, C. Jeffrey, M. Kendall, J. Luo, M. 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. 169 pp.

Miller, J., R. Waara, A. Atkinson, E. Muller, and C.S. Rogers. 2006. Coral bleaching and disease combine to cause extensive mortality on reefs in US Virgin Islands. Coral Reefs 25(3):418-425.

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

Ramos-Scharrón, C.E. and L.H. MacDonald. 2005. Measurement and prediction of sediment production from unpaved roads, St. John, U.S. Virgin Islands. Earth Surface Processes and Landforms. 30:1283–1304.

Ramos-Scharrón, C.E. 2012. Effectiveness of drainage improvements in reducing sediment production rates from an unpaved road. Journal of Soil and Water Conservation 67(2):87-100.

Richmond, R.H. 1993. Coral reefs: present problems and future concerns resulting from anthropogenic disturbance. American Zoologist. 33(6):524–536.

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

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

Rogers, C.S. and J. Beets. 2001. Degradation of marine ecosystems and decline of fishery resources in marine protected areas in the US Virgin Islands. Environ. Conserv. 28 (4):312-322.

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

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

Rogers, C.S., J. Miller, and E.M. Muller. 2008b. Coral diseases following assive bleaching in 2005 cause 60 percent decline in coral cover and mortality of the threatened species, Acropora palmata, on reefs in the U.S. Virgin Islands. USGS Fact Sheet 2008-3058.

Underwood, A.J. 1994. On beyond BACI: sampling designs that might reliably detect environmental disturbances. Ecological Applications 4:3-15.

Waddell, J.E. and A.M. Clarke (eds.) 2008. The State of Coral Reef Ecosystems of the United States and Pacific Freely Assocaited States: 2008. NOAA Technical Memorandum NOS NCCOS 73. NOAA/NCCOS Center for Coastal Monitoring and Assessment's Biogeography Team. Silver Spring, MD. 569 pp.

Whitfield, P.E., T. Gardner, S.P. Vives, M.R. Gilligan, W.R. Courtenay Jr, G.C. Ray, and J. Hare. 2002. Biological invasion of the Indo-Pacific lionfish (Pterois volitans) along the Atlantic coast of North America. Marine Ecology Progress Series 235:289-297.

Wilkinson, C. 2004. Status of coral reefs of the world: 2004, Vol. 1. Global Coral Reef Monitoring Network.

Chapter 3

Edmunds, P.J. 2000. Patterns in the distribution of juvenile corals and coral reef community structure in St. John, US Virgin Islands. Marine Ecology Progress Series. 202: 113-124.

Edmunds, P.J. 2005. Long-term dynamics of coral reefs in St. John, US Virgin Islands. Coral Reefs. 21: 357-367.

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

Gray, S.C., K. Gobbi, P.V. Narwold, M. Fox, and K. Miller. 2008. Comparison of sedimentation in bays and reefs below developed vs. undeveloped watersheds, St. John, US Virgin Islands. 11th International Coral Reef Symposium, July 7th-11th, Fort Lauderdale, Florida.

Hubbard DK. 1987. A General Review of Sedimentation as it Relates to Environmental Stress in the Virgin Islands Biosphere Reserve and the Eastern Caribbean in General. Biosphere Reserve Report No. 20, Virgin Islands Resource Management Cooperative, St Thomas.

Philipp, E., and K. Fabricius. 2003. Photophysiological stress in scleractinian corals in response to short term sedimentation. Journal of Experimental Biology and Ecology. 287: 57-78.

Ramos-Scharrón, C.E. and L.H. Macdonald. 2005. Measurement and prediction of sediment production from unpaved roads, St John, US Virgin Islands. Earth Surface Processes and Landforms. 30: 1283-1304.

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: 1983-1991.

Steneck, R.S., and M.N. Dethier. 1994. A functional group approach to the structure of algal-dominated communities. Oikos. 69: 476-498.

Steward, J.S., and W.C. Green. 2007. Setting load limits for nutrients and suspended solids based upon seagrass depth-limit targets. Estuaries and Coasts. 30(4): 657-670.

Chapter 4

Apeti, D.A., S.I. Hartwell, W.E. Johnson and G.G. Lauenstein. 2012a. National Status and Trends Bioeffects Progam: Field Methods. NOAA National Centers for Coastal Ocean Science, Center for Coastal Monitoring and Asssessment. NOAA NCCOS Technical Memorandum 135. Silver Spring, MD.27 pp.

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

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

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

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-search.html.

DeLorenzo M., Scott G., Ross P. 2001. Toxicity of pesticides to aquatic microorganisms: a review. Environ Toxicol Chem. 2001 Jan;20(1):84-98.

Downs, C.A., J.E. Fauth, C.E. Robinson, R. Curry, B. Lanzendorf, J.C. Halas, J. Halas, and C.M. Woodley. 2005. Cellular diagnostics and coral health: declining coral health in the Florida Keys. Marine Pollution Bulletin. 51: 558-569.

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. 1998. 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.

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). 2006. Interim Reregistration Eligibility Decision for Chlorpyrifos. http://www.epa.gov/oppsrrd1/REDs/chlorpyrifos_ired.pdf

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. NPDES Facility Registry. Facility Detail Report for Bordeaux Mountains. http://iaspub.epa.gov/enviro/fii_query_dtl.disp_program_facility?pgm_sys_id_in=VIGSA0034&pgm_sys_acrnm_in=NPDES

Florida Department of Environmental Protection (FDEP). 2004. Southeast Florida coral reef initiative: a local action strategy. FDEP, Office of Coastal and Aquatic Managed Areas, Coral Reef Conservation Program. Miami, FL. 28pp.

Gibbs, P.E. and G.W. Bryan. 1996. TBT-induced imposex in neogastropod snails: masculinization to mass extinction. In: S. J. de Mora, editor, 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.

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. Silver Spring, MD. NOAA Technical Memorandum NOS NCCOS 29. 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. 137 pp.

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

Long, E.R., Robertson, A., Wolfe, D.A., Hameedi, J. and Sloane, G.M. 1996. Estimates of the spatial extent of sediment toxicity in major U.S. estuaries. Environmental Science and Technology 30(12):3585-3592.

Long, E.R., L.J. Field and D.D. MacDonald. 1998. Predicting toxicity in marine sediments with numerical sediment quality guidelines. Environmental Toxicology and Chemistry 17(4): 714-727.

Long, E.R., and D.D. MacDonald. 1998. Recommended uses of empirically derived, sediment quality guidelines for marine and estuarine ecosystems. Human and Ecological Risk Assessment 4(5): 1019-1039.

Lundholm, C.E. 1997. “DDE-Induced eggshell thinning in birds”. Comp Biochem Physiol C Pharmacol Toxicol Endocrinol **118** (2): 113–28

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. Silver Springs, MD. NOAA Technical Memorandums NOS NCCOS 28. 17 pp.

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

Neff, J.M., Stout, S.A., and Gunster, D.G. 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.

Pait, A.S., Whitall, D.R., Jeffrey, C.F.G., Caldow, C., Mason, A.L., Christensen, J.D., Monaco, M.E., Ramirez, J. 2007. An assessment of chemical contaminants in the marine sediments of southwest Puerto Rico. NOS NCCOS 52. Silver Spring, MD. NOAA/NOS/Center for Coastal Monitoring and Assessment, 116pp. Available at: http://www.ccma.nos.noaa.gov/publications/southwestpuertorico.pdf.

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

Pait, A.S., Jeffrey, C.F.G, Caldow, C., Whitall, D.R., Hartwell, S.I., Mason, A.L. and Christensen, J.D. 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.

Pait, A.S., Mason, A.L.,Whitall, D.R., Christensen, J.D. and Hartwell, S.I. 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.

Pait, A.S., S.I. Hartwell, A.L., Mason, R.A. Warner, C.F.G. Jeffrey, A.M. Hoffman, D.A. Apeti, F.R. Galdo Jr., and S.J. Pittman. 2013. An assessment of chemical contaminants, toxicity and benthic infauna in sediments from the St. Thomas East End Reserves (STEER). NOAA Technical Memorandum NOS NCCOS 156. Silver Spring, MD. 62 pp.

Peachey, R.L. and Crosby, D.G. Phototoxicity in tropical reef animals. Mar. Environ. Res. 1996, 42: 359-362.

Pew Oceans Commission. 2003. America's Living Oceans: Charting a Course for Sea Change. A Report to the Nation. Pew Oceans Commission, Arlington, Virginia. 144pp.

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 Wallace, G. 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.

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.

Waddell, J.E. (ed.) 2005. The state of coral reef ecosystems of the United States and Pacific Freely Associated States: 2005. NOAA Technical Memorandum NOS NCCOS 11. NOAA/ NCCOS Center for Coastal Monitoring and Assessment's Biogeography Team.
Silver Spring, MD. 522pp.

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.

Windom, H.L., Schropp, S.J., Calder, F.D., Ryan, J.D., Smith Jr., R.G., Burney, L.C., Lewis, F.G. and Rawlinson, C.H. 1989. Natural trace metal concentrations in estuarine and coastal marine sediments of the southeastern United States. Environmental Science and Technology 23: 314-320.

U.S. Department of Commerce
Rebecca Blank, *Deputy Secretary*

National Oceanic and Atmospheric Administration
Kathryn Sullivan, *Acting Under Secretary for Oceans and Atmosphere*

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

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