EPA Public Access

Author manuscript

Environ Monit Assess. Author manuscript; available in PMC 2019 March 13.

About author manuscripts

Submit a manuscript

Published in final edited form as:

Environ Monit Assess. ; 190(4): 213. doi:10.1007/s10661-018-6562-1.

Assessing Land Use, Sedimentation and Water Quality Stressors as Predictors of Coral Reef Condition in St. Thomas, U.S. Virgin Islands

L.M. Oliver¹, W.S. Fisher¹, L. Fore², A. Smith³, and P. Bradley⁴

¹U.S. Environmental Protection Agency, Office of Research and Development, National Health and Environmental Effects Research Laboratory, Gulf Ecology Division, 1 Sabine Island Drive, Gulf Breeze, FL 32561-5299

²Puget Sound Partnership, 326 East D St., Tacoma, WA 98421

³*Former*, Student Services Contractor for the U.S. Environmental Protection Agency, Office of Research and Development, National Health and Environmental Effects Research Laboratory, Gulf Ecology Division, 1 Sabine Island Drive, Gulf Breeze, FL 32561-5299

⁴*Retired*, U.S. Environmental Protection Agency, Office of Research and Development, National Health and Environmental Effects Research Laboratory, Atlantic Ecology Division, 24 Tarzwell Drive, Narragansett, RI 02882

Abstract

Coral reef condition on the south shore of St. Thomas, U.S. Virgin Islands was assessed at various distances from Charlotte Amalie, the most densely populated city on the island. Human influence in the area includes industrial activity, wastewater discharge, cruise ship docks, and impervious surfaces throughout the watershed. Anthropogenic activity was characterized using a landscape development intensity index (LDI), sedimentation threat estimates (ST) and water quality impairments (WQ) in the near-coastal zone. Total 3-dimensional coral cover, reef rugosity and coral diversity had significant negative coefficients for LDI, as did densities of dominant species *Orbicella annularis*, *O. franksi*, *Montastraea cavernosa*, *O. faveolata*, and *Porites porites*. However, overall stony coral colony density was not significantly correlated with stressors. Positive relationships between reef rugosity and ST, coral diversity and ST, and coral diversity and WQ, were unexpected because these stressors are generally thought to negatively influence coral growth and health. Sponge density was greater with higher disturbance indicators ST and WQ, consistent with reports of greater resistance by sponges to degraded water quality compared to stony corals. The highest FoRAM (Foraminifera in Reef Assessment and Monitoring) indices indicating good water quality were found offshore from the main island and outside the harbor. Negative associations between stony coral metrics and LDI have been reported elsewhere in the Caribbean and highlight LDI potential as a spatial tool to characterize land-based anthropogenic stressor

Corresponding Author: oliver.leah@epa.gov, 850 934-2470.

The views expressed in this article are those of the author[s] and do not necessarily reflect the views or policies of the U.S. Environmental Protection Agency.

gradients relevant to coral reefs. Fewer relationships were found with an integrated stressor index but with similar trends in response direction.

Keywords

Coral reefs; landscape development intensity; sedimentation; impaired water quality

Introduction

Coral reefs of the U.S. Virgin Islands (USVI) are heavily impacted by multiple human influences at varying scales (Gardner et al. 2003; Smith et al. 2008; Rogers et al. 2008; Friedlander et al. 2013; Pait et al. 2013a; 2013; Pittman et al. 2013; Whittall et al. 2014). Global conditions such as elevated ocean temperature have led to coral bleaching and disease, causing stony coral mortality across broad reef areas of the Caribbean Sea. Local resource managers and decision-makers have little influence over global-scale anthropogenic pressures such as rising ocean temperature, ocean acidification, and elevated ultraviolet light levels. However, they do have some influence over local stressors, such as sediment, contaminants, nutrients and physical injuries, that can interact with global pressures in additive or synergistic ways (Knowlton and Jackson 2008; Wooldridge and Done 2009; Brown et al. 2013; Ban et al. 2014). Managing stressors that are within local authority can slow the cascade of negative impacts and better sustain reef-derived ecosystem services such as coastal protection from storm surge, habitat provision for commercial and recreational fisheries, and revenue from reef-related tourism (Moberg and Folke 1999; Principe et al. 2012). Effective management requires identifying which stressors are human-generated so that activities can be modified to reduce their adverse effects.

The Clean Water Act (CWA) provides a mechanism to set chemical, physical and biological criteria for the protection of aquatic ecosystems (Bradley et al. 2009; Fore et al. 2009; Bradley et al. 2010). As part of a process to establish biological water quality standards or biocriteria, the U.S. Virgin Islands government has explicitly identified coral reefs and reef functions as CWA designated uses, and defined biological integrity to lay the foundation for setting condition thresholds (biocriteria) for coral reefs (USVI 2010). This action was spawned in part by the demonstration of stony coral sensitivity to disturbance gradients using various measures of anthropogenic stress. For example, measures of coral reef condition including taxa richness, average colony size, and total and live topographic coral surface area showed significant positive relationships with increasing distance from human disturbance in St. Croix (Fisher et al. 2008). Negative relationships of coral condition with increased watershed land development were demonstrated in St. Croix (Oliver et al. 2011), Hawaii (Rodgers et al. 2012), St. Lucia (Bégin et al. 2016), and Thailand (Golbuu et al. 2008). Because the reefs around St. Thomas, USVI are located near areas of high human activity (Ennis et al. 2016), they present another opportunity to explore potential links between reef condition and anthropogenic activity.

The USVI economy depends heavily on tourism revenues with an estimated 60 – 80% of gross domestic product linked to tourism-related activities (Abt Associates 2016). Nearly 2

million cruise ship tourists visited the USVI annually from 2000–2005, and the majority of cruise ships visit St. Thomas (van Beukering et al. 2011). The area of shallow water coral reefs around St. Thomas cover approximately 42 km² and comprise mostly linear coral reefs and colonized pavement, with scattered patch reefs farther offshore (Kendall et al. 2001). Stony corals provide important ecosystem services due to the structural reef habitat which supports diverse fish and invertebrate species valued by important to divers, snorkelers, and recreational and commercial fishers, and provides shoreline protection from extreme weather events (van Beukering et al. 2011). Without stony corals, these ecosystem services would be lost or compromised, yet pressure from coastal development, cruise ship and recreational boat activity, and land-use change on the 74-km² island may threaten the very resource to which visitors are drawn.

Charlotte Amalie on the south shore is the largest city on St. Thomas with a population of approximately 18,481 (2010 US Census). The Charlotte Amalie watershed has 34% impervious surface and nearly no shoreline riparian buffer at either the St. Thomas Harbor or Inner Harbor where two cruise ship ports operate. With a hardened shoreline, presence of power and wastewater treatment plants, and multiple active cruise ship and shipping ports, this area was chosen to represent a center of human disturbance likely to adversely influence adjacent coral reef communities. We hypothesized that coral reef communities near this concentrated human disturbance would be degraded relative to reefs further away, and used 2 approaches to characterize anthropogenic disturbance. First, we developed three independent measures of potential stress to coral reefs; the watershed landscape development intensity index (LDI), modeled land-based sedimentation threat (ST), and impaired water quality index (WQ) based on proximity to impaired water bodies. These spatial proxies represent reef stressors that can be managed to some degree, and it would be valuable to discern their relative influence to assist managers in optimizing decisions to invest in the most effective remediation actions. Measures of density, size, and condition of benthic invertebrates including stony corals, gorgonians, sponges, macroinvertebrates, fishes, and foraminifera were assessed for potential responsiveness to these stressors. The reef metrics selected either showed a response to anthropogenic influence in previous studies, or are suitable candidates based on their connection to an ecosystem service (e.g. rugosity). Second, stressors were re-scaled from 0–1 and summed to generate an Integrated Stressor Index (ISI), allowing results from individual stressors and ISI to be compared for relative power to predict coral reef condition.

Materials and Methods

Survey design

Survey sites (n=13) were selected along the south shoreline of St. Thomas, and spanned approximately 10 km (Fig 1a). To control for the effect of natural site differences, we consistently selected sites on or near rocky points of land from Fortuna Bay in the west to Frenchman Bay in the east. All sites were identified as coral reef and colonized hardbottom habitat by Kendall (2001). We also targeted a narrow range of depth from 5.5 – 9.1 m (mean 6.9 m) to minimize habitat differences that could affect coral community composition

(Morelock et al. 2001). Based on field observations and knowledge of the area, we assumed Charlotte Amalie Harbor to be the greatest source of intense human disturbance in the area.

Data collection and calculation of reef indicators

Multiple-assemblage surveys were conducted during March 2009 as described by Fisher (2007) and Santavy et al. (2012). All surveys were conducted around a common 25 m transect line stretched along the sea floor. A fish survey was conducted first in a 4×25 m transect area (100 m² total), with divers identifying to species, counting and estimating the length of all fish >5 cm in length (Menza et al. 2006). Macroinvertebrates identified were queen conch, spiny and slipper lobsters, sea urchins (*Diadema* and others), and crabs. These were counted within either 1 or 2 m (due to diver inconsistency) on either side of the transect line for a total area of 50 or 100 m². Average reef rugosity index (RI) was calculated from 5 measurements made at the 0, 5, 10, 15, and 20 m marks along the transect line using the draped chain method: d/l , where d = linear chain distance when draped over coral reef and l = constant chain length of 6 m (Risk 1972). Stony coral colonies at least 10 cm in any dimension were counted along one side of the transect line using a 1 m stick placed parallel to the line as a guide. Colonies were measured for height, maximum diameter, and percent live in 10% increments, and identified to species. Sponges and gorgonians were counted in 5 – 1 m² quadrants located at 1, 5, 10, 15 and 20 m transect marks. Each colony was measured for height and diameter, and morphology recorded as one of nine possible shapes for gorgonians and nine for sponges (Santavy et al. 2012; 2013).

Stony coral three-dimensional (3D) colony surface areas (SA) were calculated using a modified hemisphere formula: $SA = m \pi R^2$, where m is a constant (1–4) based on relative morphological complexity and R is height-adjusted colony radius calculated as $((\text{diameter}/2) + \text{height})/2$ (Santavy et al. 2012). Regression equations developed for sponges and gorgonians were used to calculate colony SA (Santavy et al. 2013), except SA formula for a circle (πr^2) was used for encrusting sponges and in cases where very small colony measurements result in negative SA using regression equations. Average values for each station from five – 1 m² quadrats were calculated for SA and abundance of sponges and gorgonians.

Fish lengths were estimated in 5 cm increments. The median value of each increment was assigned to individual fish, for example 5–10 cm fish were assigned 7.5 cm length except for fish in the smallest size class (<5 cm) which were assigned 3 cm. Biomass was calculated using the formula $W = \alpha * L^\beta$ with species-specific constants α and β supplied by Froese and Pauly (2013). Invertebrate counts were standardized per m². Foraminifera samples were collected in triplicate at each site and results were averaged to calculate a FoRAM index based on the methods of Hallock et al. (2003).

Stressor Estimates

Land-Based Anthropogenic Activity—A Landscape Development Intensity Index (LDI) was calculated as a measure of land-based anthropogenic activity using a 2.4 m resolution land use layer produced for 2007 by NOAA Coastal Change Analysis Program (www.csc.noaa.gov/digitalcoast/data/ccapregional/index.html) and corresponding Emergency

coefficients (Brown and Vivas 2005; Oliver et al. 2011). Stations were assigned the LDI value of the adjacent Hydrologic Unit Code (HUC)14 watershed and for stations near a watershed border, the eastward watershed LDI value was used to account for the predominant coastal current (Nemeth and Nowlis 2001). Land use, station locations and adjacent watershed LDIs along the southern St. Thomas coastline are shown in Figs. 1a and 1b.

Sedimentation Threat (ST)—Benthic sediment threat values were obtained from World Resources Institute and National Oceanic and Atmospheric Administration Summit to Sea model (WRI and NOAA 2006). Summit to Sea is a GIS model that incorporates 30-m resolution land use / land cover data (NASA Geocover-LC 2000), soil type and associated erodibility factor (US Department of Agriculture / Natural Resources Conservation Service, Soil Service Geographic Database), slope (Digital Elevation Model) and precipitation (NOAA National Climate Data Center, maximum monthly precipitation for the 10-year period January 1990 – January 2001). Summit to Sea data sources and processing steps are available at: <https://data.nodc.noaa.gov/coris/data/NOAA/nos/Summit2Sea/USVirginIslands/>. In short, ST was calculated for USVI reef habitats from relative erosion potential for watersheds, adjusted for watershed size and modelled to correspond with pour points or “guts”. Dissipation of sediment into coastal areas employed ArcView v3.2 Point Density tool with a maximum distance from shore of 5 km.

Water Quality (WQ)

Water quality impairments reported by the USVI Department of Planning and Natural Resources (DPNR 2010) to the U.S. Environmental Protection Agency as required by the Clean Water Act include low dissolved oxygen, high enterococci bacteria, and elevated nutrient concentrations which may all negatively influence coral condition (Villanueva et al. 2006; Kaczmarek et al. 2005; Voss and Richardson 2006). However, specific loadings and stressor-coral interactions are not understood well enough to weight the individual stressors for relative importance, hydrologic transport to reefs is unknown, and bays are monitored for water quality exceedances only twice per year. Included in this water quality analysis from west to east were Perseverance Bay (PB), Brewer’s Bay (BrB), Lindberg Bay (LB), Krum Bay (KB), St. Thomas Harbor (ST), Inner Harbor (IN), and Frenchman Bay (FB) (Fig 1b). We assumed that only impaired water bodies east of a station would impact it due to the dominant westward current (Nemeth and Nowlis 2001), that impairment would dissipate with greater distance between station and waterbody, and that beyond 3 km no impact would occur. First, the distance between each station and mid-bay shoreline was measured using ArcMap v9.3, and if two bays were within 3 km of a station, distances were summed. Distance values were transformed to $(1 / \text{distance to bay})$ so that final WQ value increased with proximity to bays with WQ impairments. This transformation also achieved a non-linear function assuming that WQ contaminants dissipate rapidly from the source (Fig. 2), although in reality transport of runoff-borne pollutants to reef habitats is dependent on multiple factors such as precipitation, river density, storms, and wind (Warne et al. 2005).

Data analysis

Reef indicator values for stony corals, gorgonians, sponges, fish and macroinvertebrates were calculated for each station using SAS © (Cary, North Carolina), except Shannon-Weiner diversity index which was calculated using PRIMER© (Clark and Gorley 2006). Spearman's nonparametric correlation analysis was used to test for relationships between the 3 stressors. Distribution of reef indicators was evaluated using Shapiro-Wilke statistic, and either log10 or rank transformations were applied to correct for deviations from normal distribution. Stepwise regression was employed to discern whether LDI, ST or WQ were significant predictors of reef indicators with inclusion requirement of $p < 0.25$ and retention requirement of $p < 0.20$.

An Integrated Stressor Index (ISI) was calculated by first re-scaling station stressor data values from 0–1, then, assuming equal effects the re-scaled values were added. Spearman correlation analysis was conducted with reef metrics and ISI for comparison with stepwise regression results. No correction for multiple comparisons was applied, instead all results significant at $p < 0.05$ are reported. For all 3 stressors and ISI, quartile values were assigned for the purpose of comparing relative stressor levels at each station in the gradient.

Results

Station & Stressor Summary

Stations ranged in depth from 5.5 – 9.1 m and were within 0.1 km from land, with ten stations along the St Thomas shoreline, two near Water Island and one near Hassel Island (Fig 1a). Lowest LDI values were found in the western portion of the study area with the exception of uninhabited Hassel Island near Charlotte Amalie Harbor, which also had low LDI. The highest LDIs were in the center of the study area near Charlotte Amalie Harbor and the airport where impervious surface covers most of the land and numerous industries are present along the coast. Eastern stations were adjacent to an intermediate-LDI watershed. Some stations had common LDI values because they were adjacent to the same HUC14 watershed (Fig 1b).

Sedimentation threat was lowest at station 11 on Hassel Island, and ST was highest at station 10 near Charlotte Amalie Harbor, and second highest at station 13 on the eastern edge. A WQ impairment estimate based on distance from impaired bays ranged from 0 at four stations (which exceeded the maximum distance from an impaired bay under our assumptions), to high values at stations 5 and 6 in the center of the study area.

The ISI ranged from 0.40 – 2.37 with an average of 1.23. The lowest values were found at the western edge of the study area and at Hassel Island, and highest values at stations 5, 6 and 10 (Table 1). Station stressor values grouped into quartiles show the most consistent estimates of high human impact were at stations nearest the harbor and at the eastern edge, and most consistent low impact was at western stations and Hassel Island. However, no stations were in the lowest quartile for all stressor estimates. Station values for these three stressors were not significantly correlated with one another.

Stony Corals

A total of 1,330 stony coral colonies representing twenty-six taxa were observed and measured. Of these, fifty-two colonies were not identifiable to species and were eliminated from taxa richness counts and calculation of the Shannon-Weiner diversity index. Most colonies (81.8%) had hemispherical morphology; 15.7% were branching and 2.5% were flat, and no living *Acropora* colonies were observed. Two colonies (*O. annularis* and *O. faveolata*) observed at station 13, and one *O. faveolata* colony at Station 3 exceeded the average (live + dead) stony coral SA + 3 standard deviations, so were considered outliers. These colonies were excluded from indicator calculations and subsequent analyses. Eight stony coral species comprised 82% of colony density (Table 2).

Coral density varied from 0.6 / m² at station 9 near Water Island to 9.1 at stations 1 and 5 (Table 3). Despite low colony density at station 9, those corals present were large with highest CSA_Av, LCSA_Av (See Table 3 for variable names and abbreviations). Total 3D coral cover (3DTC includes dead coral) ranged from 0.12 – 0.99 m² coral/m² sea floor, and 3DLC ranged from 0.05 – 0.59 (Table 3). Due to high correlation ($r^2 = 0.794$, $p < 0.0001$) between 3DTC and 3DLC, only 3DTC was included in regression models and correlation tests with ISI. Total structure of the reef at all five stations west of Charlotte-Amalie was largest with higher total cover (3DTC) and live cover (3DLC), and stony coral taxa richness (SC_Tx) was also high at these stations relative to most others. Both total and live average colony size varied little except for relatively high values at stations 9 and 3. Average coral LT ranged from lowest values around 55% at stations 3, 8 and 9 to highs over 80% at stations 2, 5, 11 and 13. The CSA_CV of total coral colony size was lowest at Stations 8 and 10, and highest at stations 2 and 13.

Increased watershed LDI was negatively related to stony coral reef structure measured by 3DTC and rugosity (Table 4). LDI was also negatively associated with coral diversity (Shannon-Weiner diversity index, SC_H'). Sedimentation threat was significant with reef rugosity and SC_H' with positive relationships, and WQ had positive relationship with SC_H'. While not significant for total colony density, LDI was negatively related to colony density of five of eight dominant coral species (Table 5). Water quality had significant positive relationship with *Porites porites* density (Table 5).

The composite ISI did not correlate with any population-level stony coral metrics. Colony densities of *Orbicella annularis* and *O. franksii* were negatively correlated with ISI with r values = - 0.502 and - 0.49, and $p < 0.08$ and 0.09, respectively.

Gorgonians and Sponges

Gorgonians were observed at seven of the thirteen stations (Table 6) with a total of forty-seven colonies exhibiting eight morphologies. The circle SA formula was applied to two sea plumes < 26 cm in height. Sea rods were dominant, accounting for about 60% of the abundance and 79% of the SA of gorgonians. Highest gorgonian density and SA were found at Station 1 (Table 6). Regression analysis detected no significant relationships between any stressors and average gorgonian surface area or gorgonian density. Correlation analysis with ISI revealed no significant relationships with gorgonian metrics.

Sponges were found at all thirteen stations (Table 6) with a total of 139 colonies exhibiting seven different morphologies. The most common sponge morphologies were rods (56), mounds (39) and encrusting (20) types, with rods contributing 96% of total sponge SA. Three “vase” sponge colonies <10 cm in height yielded negative SA using regressions, so the formula for a circle was used to estimate SA for these exceptions.

Sedimentation threat and WQ were positively significantly related to sponge colony density (Table 4). Sponge colony density was positively correlated with ISI, $r^2 = 0.395$, $p < 0.02$.

Fish

A total of 1968 fish belonging to twenty-five families were observed and eight families comprised 92% of total abundance (highest abundance to lowest): Labridae, Scaridae, Pomacentridae, Acanthuridae, Gobiidae, Haemulidae, Serranidae and Lutjanidae. These dominant eight families comprised 88% of total biomass (highest biomass to lowest): Scaridae, Serranidae, Acanthuridae, Pomacanthidae, Haemulidae, Pomacentridae, Lutjanidae, and Labridae. From sixty-seven different species, the ten species comprising 70% of total abundance were (from highest to lowest) bluehead, masked/glass goby, striped parrotfish, ocean surgeonfish, blue tang, redband parrotfish, cocoa damselfish, French grunt and bridled goby. Fish abundance ranged from 45 – 231 in the 100 m² survey area. Piscivores were rare and constituted less than 1% of abundance. Zooplanktivores constituted 16%, invertivores 49% and herbivores 45%.

Station metrics for total fish abundance, total biomass, taxa richness, and Shannon diversity index are presented in Table 6. No fish metrics were related to individual stressors and no significant correlations with ISI were found.

Foraminifera and Macroinvertebrates

The FoRAM index was lowest (2.83) at station 10 near Charlotte Amalie Harbor. Stations 7 and 9 near Water Island had the highest FoRAM values of 6.48 and 4.88, and remaining station values ranged from 3.06 – 3.84 (Table 6). Sea urchins of various species comprised 98% of macroinvertebrates observed at all stations. Of all macroinvertebrates, 67% were found at Stations 1–5, west of Lindberg Bay (Table 6). Only seven spiny lobsters (five at station 10) were found, and no slipper lobsters, juvenile queen conch, or adult conch were observed at all. Neither the FoRAM index nor invertebrate density showed significant relationships in regressions with individual stressors, or in correlation analysis with ISI.

Discussion

Human disturbance on the south shore of St. Thomas was characterized using three independent spatial measures of stressors known to adversely impact coral reefs. The stressors harm coral reefs either directly (Loya 1976; Rogers 1979; Rogers 1990; Fabricius 2005) or synergistically through interaction with other factors (Carilli et al. 2009; Vega Thurber et al. 2013). High stressor values at stations closest to Charlotte Amalie Harbor validated selection of the area as an anthropogenic disturbance center where numerous human impacts occur. Our approach included comparing predictive power of individual stressors with a composite stressor index, the latter assuming additive effects as used to

model reef stressors on a worldwide scale (Halpern et al. 2008). Land-based human activity represented by watershed LDI (Brown and Vivas 2005) had the most significant relationships with reef rugosity, coral diversity, and coral taxa richness exhibiting consistent negative coefficients in regression models. This supported the hypothesis that increased intensity of human development on land (high LDI) is associated with poor condition of coral reefs adjacent to high-LDI watersheds, as demonstrated in other Caribbean areas (Rogers 1979; Smith et al. 2008, Oliver et al. 2011), Hawaii (Rodgers et al. 2012) and Indonesia (Golbuu 2008). The capability of LDI to predict ecosystem condition in wetlands and nutrient loading in watersheds (Brown and Vivas 2005) extends to predicting characteristics of coral reef condition such as stony coral colony size, species colony density, species diversity, topographic reef complexity, and benthic habitat composition at multiple watershed scales (Pittman et al. 2013). Some of these coral indicators have been related to distance to disturbance or other stress proxies (Fisher et al. 2008; Smith et al. 2008; Rodgers et al. 2012), and our results reinforce their application in stressor-response characterizations.

Stations near high-LDI watersheds had lower stony coral 3D cover and reef rugosity, and reduced colony density for 5 dominant reef-building species (Tables 4, 5). Total coral colony density and taxa richness were not statistically associated with LDI, but stations adjacent to highest-quartile LDI watersheds tended to have fewer colonies and fewer taxa compared to those near low-quartile LDI watersheds. Also, coral communities at the high LDI stations were dominated by “weedy” species like *P. astreoides* and *S. siderea* (Table 6), which have shown greater resistance to stress in Puerto Rico and other Caribbean areas (Torres and Morelock 2002, Green et al. 2008, Smith et al. 2013), but do not provide the habitat complexity of branching species (e.g. *Acropora*) nor the reef structure of massive corals (e.g. *Orbicella* and *Montastraea*). There were significantly fewer *Orbicella* sp. at stations with higher LDI, a finding that highlights a conclusion similar to Smith et al. (2008) and Ennis et al. (2016), who reported that reduced coral cover near human influence was driven by fewer *O. annularis*, *O. franksii* and *O. faveolata*. There is particular concern about loss of prime reef-building species (Morelock et al. 2001) and the compromise of multiple ecosystem services that ultimately depend on reef structure (Principe et al. 2012; Alvarez-Filipi et al. 2009; Burke and Maidens 2004). *Montastraea cavernosa*, another massive reef-building coral is thought to be relatively resistant to stress but was also more abundant with lower LDI. These results underscore the importance of species-specific coral monitoring methods to detect community shifts caused by varied tolerance to stressors. Multivariate methods to detect changes across reef assemblages have also been applied to characterizing reef-stressor relationships (Jupiter et al. 2008). However well-designed the methods, detecting stressor-response relationships requires as broad a range of biological conditions possible (Karr and Chu 1997) and locating USVI reefs representative of “reference condition for biological integrity” (or zero human impact) is challenging as most USVI coral reefs are probably affected in some way by human influence (Rothenberger et al. 2008; Friedlander et al. 2013; Pait et al. 2013a, 2013b; Pittman et al. 2013; Whitall et al. 2014; Jackson et al. 2014).

There was a significant, but weak, negative relationship of modeled ST with average stony coral colony size. There were also significant but positive relationships of ST with reef rugosity and coral diversity index, countering a common premise that high sediment loading adversely affects stony coral condition (Rogers 1979; Fabricius 2005). However, all of these

results may have little interpretive potential because the range of ST values in this study was relatively small. In addition to testing a wide range of biological responses, the stressor gradient ideally incorporates multiple stressor levels from near-pristine to severe human disturbance levels (Karr and Chu 1997). But here, the highest ST was 54,359 (station 10), which is relatively small compared to the maximum Summit to Sea ST in the region of 251,714. Thresholds of sediment exposure are not well understood (Rogers 1990), and, even if they were, the ST model does not convert to actual sediment concentrations. A positive coefficient found for ST and sponge density is consistent with other studies showing some sponge species to be relatively tolerant to stress from excess sediments (Powell et al. 2014). If so, this could indicate a possible phase-shift in impacted environments from coral-dominated habitats to sponge-dominated, and a loss of many ecosystem services (Ward-Paige et al. 2005; Norström et al. 2009).

For coral reefs near Charlotte-Amalie, a more relevant source of sediment exposure for corals might be pulsed resuspension of sediments associated with cruise ship traffic in and out of ports in Crowne Bay and Charlotte-Amalie Harbor (Kisabeth et al. 2014). Ship traffic creates plumes of resuspended sediment which substantially increase turbidity and total suspended solids compared to undisturbed levels (Kisabeth et al. 2014; Jones et al. 2015). This daily recurring activity is not captured in the ST model, which relies on landscape features, precipitation, and soil erodibility. It may be necessary to adjust sediment-delivery models like Summit to Sea to account for factors like ship traffic to improve predictive capabilities. Many other nuances of sediment effects on corals not included in ST are grain size of the sediment, with finer grains generally causing greater harm (Acevedo et al. 1989; Weber et al. 2006), and whether exposure occurs in pulsed or steady sediment regimes (Browne et al. 2015).

As proxy for impaired water quality, a spatial estimate based on distance to disturbance (Fisher et al. 2008) was developed using data reported to the US EPA under the Clean Water Act 305(b) and 303(d) programs (DPNR 2010). Assuming no effect beyond 3 km, three stations were too far away from impaired bays to be affected, and stations 5 and 6 located in the center of the study area had potential impact from three and two bays, respectively, where impaired water quality was reported. Significant positive relationships between WQ and coral diversity index, and WQ and *P. porites* colony density did not indicate that proximity to bays with WQ issues had negative effects on coral reefs. Estimates of WQ stress to benthic organisms could be improved with additional field data to better define potential relationships between specific WQ stressors like turbidity, TSS (total suspended solids), and low DO on corals. Field measurements such as TSS and nutrients are needed to validate remotely-sensed satellite products which can more accurately characterize exposure to WQ conditions over time compared with occasional measurements (Brodie et al., 2010). Fine-scale data on water currents would inform predictions of transport of materials to reef habitats and exposure of coral reefs to WQ stressors. Our distance-based WQ approach detected positive relationship with sponge colony density; more sponge colonies were present closer to bays with WQ impairments. This is similar to results from a study of sponges and land-based nutrient inputs in the Florida Keys (Ward-Paige et al. 2005) and is consistent with the concept of phase shifts from stony corals to sponges with increasing anthropogenic stress.

Modeling environmental stressor gradients is inherently difficult for several reasons. Co-occurrence of stressors in areas of high human disturbance is common, which makes it difficult to discern which stressors are responsible for reduced biological condition. The LDI itself is one type of composite index on a watershed basis, integrating inputs of nonrenewable energy to land use parcels (Odum 1996). Watershed management for coral health is not a new concept, for example many Pacific island cultures practice “ridge-to-reef” stewardship which assumes a connection between land-based human activities and health of adjacent marine communities (Richmond et al. 2007). Yet, management options may not be obvious using LDI as a predictor unless specific land use types associated with reduced biological condition can be identified (Le et al. 2015). Characterizing all operative stressors regarding a system of interest is likely impossible. Models that incorporate multi-stressor data across broad scales (Halpern et al. 2008; Burke et al. 2011) may provide more comprehensive stressor information but these scales make the data less useful for small islands like St. Thomas. For some areas of the Caribbean, data about coral reef stressors such as fishing pressure (Shivlani and Koenike 2010), chemical contaminants (Whitall et al. 2013), and ocean temperature anomalies (Novoa et al. 2012; Liu et al. 2014) could improve forecasting and clarify management options. Development of spatially-explicit, comprehensive data on multiple stressors impacting coral reefs and accounting for potential synergistic, additive or antagonistic effects would best serve reef management needs.

Three westward stations adjacent to sparse human population and undisturbed lands had consistent low stressor estimates below 50% of the mean and among the highest total and live stony coral cover, taxa richness and colony density. Watersheds adjacent to these stations had high coverage of forest and other natural lands (Fig. 1a). Progressing to the east, different patterns were seen depending on the stressor. For example, Station 4 is adjacent to a low LDI watershed but fits the 4th (highest) quartile for ST due in part to adjacent bare landscape and increased likelihood for runoff. Stressor values for station 13 located farthest east from the center of disturbance indicate greater human influence than might be assumed by its distance from Charlotte Amalie. This station had second-highest LDI and ST values and may combine with another human activity source to the east. Influences such as an unlined landfill located 0.1 km from shore in Bovani Bay (Horsely Witten Group 2013) are not represented in our stressor estimates and could have a large impact. Station 8 with highest LDI values also had the lowest stony coral cover and low percent live tissue. In contrast, station 11 located near uninhabited Hassel Island had low LDI and intermediate coral indicator values. Although uninhabited, Hassel Island is close to the harbor and probably receives resuspended sediment.

Although non-significant in regressions and correlations, the FoRAM Index was highest (best condition) at stations 7 and 9, located away from the coast and human influences and the lowest at station 10 near Charlotte Amalie Harbor. The FoRAM has shown to be responsive to eutrophication (Hallock et al. 2003) and distance to disturbance (Oliver et al. 2014), and may have value in reef monitoring protocols. Fish populations may associate with reef structure but are not expected to associate directly with stressors because fish are mobile; a lack of significant findings with stressors are not surprising.

Healthy coral reefs benefit the people and economy of St. Thomas by providing ecosystem services. Although causality cannot be directly determined, a human impact signature was evident from LDI associations with reef condition indicators at thirteen stations, showing that reefs around Charlotte Amalie are proportionally affected by human development activities in adjacent watersheds. Management steps to reduce human disturbances can lessen direct impacts on corals, as well as additive or synergistic effects from tropical storms and hurricanes, global-scale ocean temperature anomalies (Hughes and Connell 1999; Maina et al. 2011; Gurney et al. 2013) and ocean acidification (Wooldridge and Done 2009; Ban et al. 2014).

Cost-effective mitigation options to reduce sedimentation for e.g., have been demonstrated in the USVI (Ramos-Scharrón 2012) and effective spatial mapping of reef stressors can prioritize where such actions are most needed. A balanced approach to coastal management decision making, which incorporates available scientific research to better characterize stressors and understand their influence, can help protect coral reef ecosystems and the economic, ecological and aesthetic benefits they provide.

Acknowledgements

Outstanding field assistance in conducting coral reef surveys was provided by Jed Campbell, Peggy Harris, Becky Hemmer, Robert Quarles and Sherry Vickery (US EPA Gulf Ecology Division), Charles LoBue and Danny Rodriguez (US EPA Region 2), Mel Parsons (USEPA Region 4), Aaron Hutchins (The Nature Conservancy, St. Croix), Rich Henry (US Fish & Wildlife Service, Environmental Response Team), Alan Humphrey (USEPA, Environmental Response Team), Jon McBurney and Scott Grossman (Lockheed Martin, Scientific, Engineering, Response & Analytical Services (*SERAS*) Program). The crew of the OSV *Bold*, Captain Jere Chamberlain provided excellent support in all aspects of field data collection and logistics. Kent Bernier (U.S. Virgin Islands, Department of Planning and Natural Resources) provided small boat support for the survey teams.

References

- Abt Associates, Inc. (2016). Describing the ocean economies of the U.S. Virgin Islands and Puerto Rico. <https://coast.noaa.gov/digitalcoast/training/econ-usvi-pr.html>
- Acevedo RJ, Morelock J, & Olivieri RA (1989). Modification of coral reef zonation by terrigenous sediment stress. *Palaos* 4(1), 92–100.
- Alvarez-Filip L, Dulvy NK, Gill JA, Côte IM, & Watkinson AR (2009). Flattening of Caribbean coral reefs: region-wide declines in architectural complexity. *Proceedings of the Royal Society B* 276, 3019–302. [PubMed: 19515663]
- Ban SS, Graham NAJ, & Connolly S (2014). Evidence for multiple stressor interactions and effects on coral reefs. *Global Change Biology* 20, 681–697. [PubMed: 24166756]
- Bégin C, Schelten CK, Nugues MM, Hawkins J, Roberts C, Côté IM (2016). Effects of protection and sediment stress on coral reefs in Saint Lucia. *PLoS ONE* 11(2), e0146855. doi:10.1371/journal.pone.0146855 [PubMed: 26845451]
- Bradley P, Fisher W, Bell H, Davis W, Chan V, LoBue C, & Wiltse W (2009). Development and implementation of coral reef biocriteria in U.S. jurisdictions. *Environmental Monitoring & Assessment* 150, 43–51. [PubMed: 19052888]
- Bradley P, Fore L, Fisher W, & Davis W (2010). Coral Reef Biological Criteria: Using the Clean Water Act to Protect a National Treasure. U.S. Environmental Protection Agency, Office of Research and Development, Narragansett, RI EPA/600/R-10/054, 7 2010.
- Brodie J, Schroeder T, Rohde K, Faithful J, Masters B, Dekker A, Brando V, & Maughan M (2010). Dispersal of suspended sediments and nutrients in the Great Barrier Reef lagoon during river-discharge events: conclusions from satellite remote sensing and concurrent flood-plume sampling. *Marine and Freshwater Research* 61(6), 651–664.

- Brown MT, & Vivas MB (2005). Landscape development intensity index. *Environmental Monitoring & Assessment* 101, 289–309. [PubMed: 15736887]
- Brown CJ, Saunders MI, Possingham HP, & Richardson AJ (2013). Managing for interactions between local and global stressors of ecosystems. *PLoS ONE* 8(6), e65765. doi:10.1371/journal.pone.0065765 [PubMed: 23776542]
- Browne NK, Tay J, & Todd PA (2015). Recreating pulsed turbidity events to determine coral-sediment thresholds for active management. *Journal of Experimental Marine Biology and Ecology* 466, 98–109.
- Burke L, & Maidens J (2004). *Reefs at risk in the Caribbean*. World Resources Institute, Washington, D.C. 81 pp.
- Burke L, Reynter K, Spalding M, & Perry A (2011). *Reefs at risk revisited*. World Resources Institute, Washington, D.C.
- Carilli JE, Norris RD, Black BA, Walsh SM, & McField M (2009). Local stressors reduce coral resilience to bleaching. *PLoS One* 4(7), e6324. [PubMed: 19623250]
- Clark KR, & Gorley RN (2006). *PRIMER v6: User Manual/Tutorial*, Plymouth, UK.
- DPNR, (2010). *USVI Integrated Water Quality Monitoring & Assessment Report*. Department of Planning and Natural Resources, Division of Environmental Protection Water Quality Management & Planning Program. 165 pp. Retrieved 2018, 10 February from: http://www.horsleywitten.com/stx-east-endwatersheds/pubs/2010IntegratedReport_b.pdf
- Ennis RS, Brandt ME, Wilson Grimes KR, & Smith TB (2016). Coral reef health response to chronic and acute changes in water quality in St. Thomas, United States Virgin Islands. *Marine Pollution Bulletin* 111, 418–427. [PubMed: 27499526]
- Fabrizius K (2005). Effects of terrestrial runoff on the ecology of coral reefs: review and synthesis. *Marine Pollution Bulletin* 50, 125–146. [PubMed: 15737355]
- Fisher WS (2007). *Stony coral rapid bioassessment protocol* U.S. Environmental Protection Agency, Office of Research and Development, Gulf Breeze, FL EPA/600/R-06/167, 7 2007.
- Fisher WS, Fore LS, Hutchins A, Quarles RL, Campbell JG, LoBue C, & Davis WS (2008). Evaluation of stony coral indicators for coral reef management. *Marine Pollution Bulletin* 56, 1737–1745. [PubMed: 18715598]
- Fore LS, Karr JR, Fisher WS, Bradley P, & Davis WS (2009). Heeding a call to action for US coral reefs: The untapped potential of the Clean Water Act. *Marine Pollution Bulletin* 58, 1421–1423. [PubMed: 19615697]
- Friedlander AM, Jeffrey CFG, Hile SD, Pittman SJ, Monaco ME & Caldwell C, eds. (2013). *Coral reef ecosystems of St. John, U.S. Virgin Islands: Spatial and temporal patterns in fish and benthic communities (2001– 2009)*. NOAA Technical Memorandum 152 Silver Spring, MD 150 pp.
- Froese R, & Pauly D, eds. (2013). *FishBase*. World Wide Web electronic publication www.fishbase.org, version (12/2013).
- Gardner TA, Côte IM, Gill JA, Grant A, & Watkinson AR (2003). Long-term region-wide declines in Caribbean reef corals. *Science* 301, 958–960. [PubMed: 12869698]
- Golbuu Y, Fabrizio K, Victor S, & Richmond RH (2008). Gradients in coral reef communities exposed to muddy river discharge in Pohnpei, Micronesia. *Estuarine, Coastal and Shelf Science* 76, 14–20.
- Green DH, Edmunds PJ, & Carpenter RC (2008). Increasing relative abundance of *Porites astreoides* on Caribbean reefs mediated by an overall decline in coral cover. *Marine Ecology Progress Series* 359, 1–10.
- Gurney GG, Melbourne-Thomas J, Geronimo RC, Aliño PM & Johnson CR (2013). Modelling coral reef futures to inform management: can reducing local-scale stressors conserve reefs under climate change? *PLoS One* 8(11), e80137. [PubMed: 24260347]
- Halpern BS, Walbridge S, Selkoe KA, Kappel CV, Micheli F, D'Agrosa C, Bruno JF, Casey KS, Ebert, C, Fox HE, Fujita R, Heinemann D, Lenihan HS, Madin EMP, Perry MT, Selig ER, Spalding M, Steneck R, & Watson R (2008). A global map of human impact on marine ecosystems, *Science* 319, 948–952. [PubMed: 18276889]

- Hallock P, Lidz BH, Cockey-Burkhard EM, & Donnelly KB (2003). Foraminifera as indicators coral reef assessment and monitoring: The Foram Index. *Environmental Modeling & Assessment* 81, 221–238.
- Horsely Witten Group, Inc, (2013). St. Thomas East End reserves watershed management plan. Report to NOAA coral Reef Conservation Program, USVI DPNR, and The Nature Conservancy 73 pp.
- Hughes TP & Connell JH (1999). Multiple stressors on coral reefs: A long-term perspective, *Limnology & Oceanography* 44(3), 932–940.
- Humann P, & DeLoach N (2002). Reef coral identification: Florida Caribbean Bahamas. New World Publications, Jacksonville, FL.
- Jackson JBC, Donovan MK, Cramer KL, & Lam VV, (editors), (2014). Status and Trends of Caribbean Coral Reefs: 1970–2012. Global Coral Reef Monitoring Network, IUCN, Gland, Switzerland.
- Jupiter S, Roff G, Marion G, Henderson, Schrameyer V, McCulloch M, Hoegh-Guldberg O (2008). Linkages between coral assemblages and coral proxies of terrestrial exposure along a cross-shelf gradient on the southern Great Barrier Reef. *Coral Reefs* 27(4), 887–903.
- Kaczmarek LT, Draud M, & Williams EH (2005). Is there a relationship between proximity to sewage effluent and the prevalence of coral disease? *Caribbean Journal of Science* 41(1), 124–137.
- Karr JR, & Chu EW (1997). Biological Monitoring and Assessment: Using Multimetric Indexes Effectively. EPA 235-R97-001. University of Washington, Seattle.
- Kendall MS, Kruer CR, Buja KR, Christensen JD, Finkbeiner M, & Monaco ME (2001). Methods used to map the benthic habitats of Puerto Rico and the U.S. Virgin Islands. Silver Spring, MD: National Oceanic and Atmospheric Administration, NCCOS Biogeography Program <http://biogeo.nos.noaa.gov/projects/mapping/caribbean/startup.htm>
- Knowlton N, & Jackson JBC (2008). Shifting baselines, local impacts, and global change on coral reefs. *PLoS Biology* 6(2), 215–220.
- Loya Y (1976). Effects of water turbidity and sedimentation on the community structure of Puerto Rican corals. *Bulletin of Marine Science* 26(4), 450–466.
- Liu G, Heron SF, Eakin CM, Muller-Karger FE, Vega-Rodriguez M, Guild LS, De La Cour JL, Geiger EF, Skirving WJ, Burgess TFR, Strong AE, Harris A, Maturi E, Ignatov A, Sapper J, Li J, & Lynds S (2014). Reef-scale Thermal Stress Monitoring of Coral Ecosystems: New 5-km Global Products from NOAA Coral Reef Watch. *Remote Sensing* 6(11): 11579–11606, doi:10.3390/rs6111579.
- Maina J, McClanahan TR, Venus V, Ateweberhan M, & Madin J (2011). Global gradients of coral exposure to environmental stresses and implications for local management. *PLoS One* 6(8), e23064. [PubMed: 21860667]
- Menza C, Ault J, Beets J, Bohnsack J, Caldwell C, Christensen J, Friedlander A, Jeffrey C, Kendall M, Luo J, Monaco M, Smith S, & Woody K, (2006). A guide to monitoring reef fish in the National Parks Service's South Florida / Caribbean Network. NOAA Technical Memorandum NOS NCOOS 39 169 pp.
- Moberg F, & Folke C (1999). Ecological goods and services of coral reef ecosystems. *Ecological Economics* 29, 215–233.
- Morelock J, Ramirez WR, Bruckner AW, & Carlo M (2001). Status of coral reefs in southwest Puerto Rico. *Caribbean Journal Science Special Publication* No 4.
- Nemeth RS, & Nowlis JS (2001). Monitoring the effects of land development on the near-shore reef environment of St. Thomas, USVI. *Bulletin of Marine Science* 69(2), 759–775.
- (NHD) National Hydrography Dataset, (2005). National Hydrography Dataset, U.S. Geological Survey in cooperation with the U.S. Environmental Protection Agency, the U.S. Department of Agriculture Forest Service, and other Federal, State and local partners Reston, VA.
- Norström AV, Nyström M, Lokrantz J & Folke C (2009). Alternative states on coral reefs: Beyond coral-macroalgal phase shifts. *Marine Ecology Progress Series* 376, 295–306.
- Novoa S, Chust G, Froidefond JM, Petus C, Franco J, Orive E, Seoane S, & Borja A (2012). Water quality monitoring in Basque coastal areas using local chlorophyll-a algorithm and MERIS images. *Journal of Applied Remote Sensing* 6(1), 063519. doi: 10.1117/1.JRS.6.063519
- Odum HT (1996). *Environmental Accounting: Emergy and Environmental Decision Making*, John Wiley and Sons, New York, 370 pp.

- Oliver LM, Lehrter JC, & Fisher WS (2011). Relating landscape development intensity to coral reef condition in the watersheds of St. Croix, US Virgin Islands. *Marine Ecology Progress Series* 427, 293–302.
- Oliver LM, Fisher WS, Dittmar J, Hallock P, Campbell J, Quarles RL Harris P & LoBue C (2014). Contrasting responses of coral reef fauna and foraminiferal assemblages to human influence in La Parguera, Puerto Rico. *Marine Environmental Research* 99, 95–105. [PubMed: 24840256]
- Pait AS, Hartwell SI, Mason AL, Galdo FR, Jr., Warner RA, Jeffrey CFG, Hoffman AM, Apeti DA, & Pittman SJ (2013a). An assessment of chemical contaminants detected in passive water samplers deployed in the St. Thomas East End Reserves (STEER) NOAA Technical Memorandum NOS/NCCOS 157 Silver Spring, MD 22 pp.
- Pait AS, Hartwell SI, Mason AL, Warner RA, Jeffrey CFG, Hoffman AM, Apeti DA, Galdo FR, Jr., & Pittman SJ (2013b). 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 70 pp.
- Pittman SJ, Dorfman DS, Hile SD, Jeffrey CFG, Edwards MA & Caldow C (2013). Land-Sea Characterization of the St. Croix East End Marine Park, U.S. Virgin Islands NOAA Technical Memorandum NOS NCCOS 170. Silver Spring, MD 119 pp.
- Principe P, Bradley P, Yee S, Fisher W, Johnson E, Allen P & Campbell D (2012). Quantifying Coral Reef Ecosystem Services U.S. Environmental Protection Agency, Office of Research and Development, Research Triangle Park, NC EPA/600/R-11/206, January 2012, 147 pp.
- Powell A, Smith DJ, Hepburn LJ, Jones T, Berman J, Jompa J, & Bell JJ (2014). Reduced diversity and high sponge abundance on a sedimented Indo-Pacific reef system: Implications for future changes in environmental quality. *PLoS ONE* 9(1), e85253. doi:10.1371/journal.pone.0085253. [PubMed: 24475041]
- Ramos-Scharrón CE (2012). Effectiveness of drainage improvements in reducing sediment production rates from an unpaved road. *Journal of Soil and Water Conservation* 6(2), 87–100.
- Richmond RH, Rongo T, Golbuu Y, & Victor S (2007). Watersheds and coral reefs: conservation science, policy, and implementation. *Bioscience* 57, 598–607.
- Risk M (1972). Fish diversity on a coral reef in the Virgin Islands. *Atoll Research Bulletin* 153, 1–6.
- Rodgers KS, Kido MH, Jokiel PL, Edmonds T, & Brown EK (2012). Use of integrated landscape indicators to evaluate the health of linked watersheds and coral reef environments in the Hawaiian Islands. *Environmental Management* 50, 21–30. [PubMed: 22538320]
- Rogers CS (1979). The effects of shading on coral-reef structure and function. *Journal of Experimental Marine Biology & Ecology* 41, 269–288.
- Rogers CS (1990). Responses of coral reefs and reef organisms to sedimentation. *Marine Ecology Progress Series* 62, 185–202.
- Rogers CS, Miller J, Muller EM, Edmunds P, Nemeth RS, Beets JP, Friedlander AM, Smith TB, Boulon R, Jeffrey CFG, Menza C, Caldow C, Idrisi N, Kojis B, Monaco ME, Spitzack A, Gladfelter EH, Ogden JC, Hillis-Starr Z, Lundgren I, Schill WB, Kuffner IB, Richardson LL, Devine BE, & Voss JD (2008). Ecology of Coral Reefs in the US Virgin Islands In: Riegl B, & Dodge R, editors. *Coral Reefs of the USA*, Springer Netherlands 1: 303–373.
- Rothenberger P, Blondeau J, Cox C, Curtis S, Fisher WS, Garrison V, Hillis-Starr Z, Jeffrey C, Kadison E, Lundgren I, Miller WJ, Muller E, Nemeth R, Paterson S, Rogers C, Smith T, Spitzack A, Taylor M, Toller W, Wright J, Wusinich-Mendea D, & Waddell J (2008). The state of coral reef ecosystems of the U.S. Virgin Islands In: Wadell JE, & Clarke AM (editors), *The State of coral reef ecosystems of the United States and Pacific Freely Associated States: 2008*. NOAA Center for Coastal Monitoring and Assessment's Biogeography Team, Silver Spring, MD, p 29–73.
- Santavy DL, Fisher WS, Campbell JG, & Quarles RL (2012). Field Manual for Coral Reef Assessments. U.S. Environmental Protection Agency, Office of Research and Development, Gulf Ecology Division, Gulf Breeze, FL EPA/600/R-12/029. 4 2012.
- Santavy DL, Courtney LA, Fisher WS, Quarles RL, & Jordan SJ (2013). Estimating the surface area of marine gorgonians and sponges in the field. *Hydrobiologia* 707, 1–16.
- Shivlani M, & Koeneke R (2010). Spatial characterization of artisanal fisheries in Puerto Rico: Geographic Information Systems (GIS) approach for assessing the regional effort and landings.

- Proceedings 63rd Gulf and Caribbean Fisheries Institute, November 1–5, 2010, San Juan, Puerto Rico.
- Smith TB, Nemeth RS, Blondeau J, Calnan JM, Kadison E, & Herzlieb S (2008). Assessing coral reef health across onshore to offshore stress gradients. *Marine Pollution Bulletin* 56, 1983–1991. [PubMed: 18834601]
- Smith TB, Brandt ME, Calnan JM, Nemeth RS, Blondeau J, Kadison E, Taylor M & Rothenberger P (2013). Convergent mortality responses of Caribbean coral species to seawater warming. *Ecosphere* 4(7):87 <http://dx.doi.org/10.1890/ES13-00107.1>
- Torres JT, & Morelock J (2002). Effect of terrigenous sediment influx on coral cover and linear extension rates of three Caribbean massive coral species. *Caribbean Journal of Science* 38(3–4), 222–229.
- USVI (U.S. Virgin Islands), (2010). United States Virgin Islands 2010 Revisions to: Water Quality Standards Title 12, Chapter 7, Government of the Virgin Islands, Department of Planning and Natural Resources, Division of Environmental Protection, 22 pp.
- van Beukering P, Brander L, Van Zanten B, Verbrugge E, Lems K (2011). The Economic Value of the Coral Reef Ecosystems in the United States Virgin Islands. In: Edmunds PET (ed.), *The Economic Value of Coral Reefs*, IVM Report number: R-11/06
- Vega Thurber RL, Burkepile DE, Fuchs C, Shantz AA, McMinds R, & Zaneveld JR (2013). Chronic nutrient enrichment increases prevalence and severity of coral bleaching. *Global Change Biology* doi: 10.1111/gcb.12450.
- Voss JD, & Richardson LL (2006). Nutrient enrichment enhances black band disease progression in corals. *Coral Reefs* 25, 569–576.
- Villanueva RD, Yap HT, & Montaña NE (2006). Survivorship of coral juveniles in a fish farm environment. *Marine Pollution Bulletin* 51, 580–589.
- Ward-Paige CA, Risk MJ, Sherwood OA, & Jaap WC (2005). Clionid sponge surveys on the Florida Reef Tract suggest land-based nutrient inputs. *Marine Pollution Bulletin* 51, 570–579. [PubMed: 15946702]
- Weber M, Lott C, & Fabricius KE (2006). Sedimentation stress in a scleractinian coral exposed to terrestrial and marine sediments with contrasting physical, organic and geochemical properties. *Experimental Marine Biology & Ecology* 336, 18–32.
- Warne AG, Webb RMT, & Larsen MC (2005). Water, Sediment, and Nutrient Discharge Characteristics of Rivers in Puerto Rico, and their Potential Influence on Coral Reefs: U.S. Geological Survey Scientific Investigations Report 2005–5206, 58 pp.
- Whitall D, Menza C, & Hill R (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.
- Wooldridge SA, & Done TJ (2009). Improved water quality can ameliorate effects of climate change on corals. *Ecological Applications* 19(6), 1492–1499. [PubMed: 19769097]
- World Resources Institute (WRI), & NOAA (2006). Land-based sources of threat to coral reefs in the U.S Virgin Islands Washington, DC.

Fig. 1.

a Land use / land cover (LULC) and coral reef stations sampled in St. Thomas, USVI around Charlotte Amalie. Numbered black circles indicate sampling stations. Ports are indicated by anchor symbols. **b** HUC14 watersheds are indicated by black lines and their corresponding LDI values are indicated within watershed polygons. Letters show locations of water bodies which exceeded chemical or physical criteria as reported to USVI DPNR: FoB = Fortuna Bay, PB = Perseverance Bay, BrB = Brewers Bay, LB = Lindberg Bay, KB = Krum Bay, CB = Crowne Bay ST, ST = St. Thomas Harbor, FB = Frenchman Bay. Triangular symbols grade from light to dark blue, indicating increasing average number of impaired conditions from 2000–2010. Black circles indicate sampling stations

Fig. 2.
Water quality (WQ) function used to estimate stress from proximity to St. Thomas bays which are listed as impaired under the Clean Water Act 303(d) program

Table 1

Stressor values and quartiles for landscape development intensity (LDI), sedimentation threat (ST) and impaired water quality (WQ) for 13 stations along the south coast of St. Thomas, USVI. ISI is the sum of individual stressors which were re-scaled from 0–1

Station	LDI	Quartile	ST	Quartile	WQ	Quartile	ISI	Quartile
1	1.93	2	14844.3	2	0	1	0.486	1
2	1.93	2	14844.3	2	0.395	2	0.7	1
3	1.44	1	12670	1	1.193	3	0.882	2
4	1.44	1	21868.8	3	1.449	4	1.19	3
5	3.4	3	24166	3	1.843	4	2.305	4
6	3.72	4	8055.6	1	1.583	4	2.007	4
7	2.23	2	20708.1	2	0	1	0.728	2
8	3.72	4	22381.4	3	1.08	3	1.998	3
9	2.23	2	0.4	1	0	1	0.346	1
10	3.72	4	54359	4	0.685	2	2.371	4
11	1.58	1	2364.3	1	0.538	2	0.397	1
12	2.75	3	27719	4	0	1	1.086	2
13	2.75	3	31225.6	4	0.813	3	1.591	3

Table 2

Stony coral species and density (# / m² sea floor) counted at 13 stations along the southern St. Thomas coast (unidentifiable / dead colonies not presented or included in total). Stations in 1st (lowest) LDI quartile have no shading, 2nd LDI quartile in bold font, 3rd LDI quartile shaded light gray, and 4th LDI (highest) quartile shaded dark gray

Station:	1	2	3	4	5	6	7	8	9	10	11	12	13
<i>Agaricia agaricites</i>	0.48	0.16	0.12	0.04	0.04		0.04				0.04	0.04	
<i>Agaricia fragilis</i>			0.04		0.04						0.08		
<i>Agaricia lamarki</i>	0.04												0.04
<i>Colpophyllia natans</i>	0.04		0.04		0.08								
<i>Dendrogyra cylindrus</i>	0.04	0.04			0.08		0.04						
<i>Dichocoenia stokesi</i>					0.08								
<i>Pseudodiploria labyrinthiformis</i>	0.56	0.48		0.08	0.32	0.12			0.04		0.04		
<i>Diploria strigosa</i>	1.24	0.44	0.04		0.08		0.04			0.08	0.24	0.16	
<i>Eusmilia fastigiata</i>			0.04	0.2				0.16		0.04			
<i>Madracis decactis</i>		0.04			0.08			0.04		0.08			
<i>Madracis mirabilis</i>										0.08			
<i>Manicina areolata</i>								0.04					
<i>Meandrina meandrites</i>	0.04	0.04	0.04	0.2	0.36	0.04	0.04			0.16	0.04		0.04
<i>Millepora complanata</i>	0.2	0.12			0.52		0.24		0.12			0.04	
<i>Orbicella annularis</i>	0.44	0.48	0.6	0.36	0.16	0.08	0.36		0.2	0.04	0.04		0.08
<i>Montastraea cavernosa</i>	0.24	0.32	0.16	0.28	0.12	0.04	0.04	0.08		0.04	0.64	0.28	0.04
<i>Orbicella faveolata</i>	0.44	0.72	0.36	0.04	0.24	0.04	0.04	0.04	0.04	0.04	0.16		0.08
<i>Orbicella franksi</i>	0.48	0.2	0.12	0.16	0.2	0.04	0.36	0.08		0.04	0.44		
<i>Porites astreoides</i>	1.48	1.08	0.52	0.36	2.32	0.68	0.28	0.2		1.08	0.12	0.44	0.36
<i>Porites colonensis</i>		0.12	0.04								0.04		
<i>Porites divaricata</i>			0.12	0.2		0.04				0.08	0.04		
<i>Porites furcata</i>	0.04	0.04	0.16	0.12		0.04		0.04		0.20	0.08		
<i>Porites porites</i>	0.12	0.2	0.24	0.56	0.04	0.20	0.12	0.28			0.16	0.04	0.24
<i>Siderastrea siderea</i>	2.36	0.36	0.52	1	3.28	1.04	0.12	0.12	0.12	0.76	1.08	1.68	0.24
<i>Solenastrea bournoni</i>												0.12	
<i>Stephanocoenia intersepta</i>	0.32	0.16			0.04			0.04		0.20			0.08

Station:	1	2	3	4	5	6	7	8	9	10	11	12	13
Total Abundance:	8.56	5	3.16	3.6	8.08	2.36	1.72	1.12	0.52	2.92	3.24	2.8	1.2

Table 3

Station depths and mean stony coral indicators: density (SC_Den; #colonies/m²), colony surface area (CSA_Av; m² coral), live colony surface area (LCSA_Av; m² coral), percent live tissue (LT; %), total cover (3DTC; m² coral/m² seafloor), live cover (3DLC; m² coral/m² seafloor), coefficient of variation of colony surface area, (CSA_CV), number of taxa (SC_Tx; # taxa/m²), Shannon index of diversity (SC_H'), and reef rugosity (RI)

Station	Depth (m)	SC_Den	CSA_Av	LCSA_Av	LT	3DTC	3DLC	CSA_CV	SC_Tx	SC_H'	RI
1	7.62	9.12	0.11	0.06	75.57	0.975	0.557	241.5	0.72	2.24	1.66
2	7.01	8.24	0.09	0.07	82.43	0.757	0.542	321.6	0.68	2.02	1.45
3	9.14	3.32	0.30	0.09	58.43	0.986	0.310	149.1	0.68	2.36	1.66
4	6.71	3.60	0.11	0.05	68.22	0.378	0.186	194.3	0.52	2.22	1.57
5	6.71	9.12	0.10	0.06	81.43	0.869	0.589	198.0	0.76	1.94	1.41
6	7.01	2.44	0.07	0.03	74.92	0.162	0.073	246.7	0.48	1.61	1.35
7	7.62	3.60	0.10	0.05	73.22	0.360	0.182	208.7	0.52	1.94	1.54
8	6.71	1.12	0.11	0.04	54.46	0.121	0.046	124.0	0.44	2.14	1.34
9	6.40	0.64	0.54	0.13	52.50	0.345	0.082	182.8	0.24	1.44	1.28
10	6.40	2.92	0.09	0.07	78.49	0.264	0.199	167.6	0.56	1.93	1.50
11	6.71	4.32	0.09	0.05	81.39	0.373	0.219	287.1	0.60	2.03	1.32
12	5.49	3.52	0.08	0.03	74.20	0.273	0.123	178.7	0.36	1.47	1.45
13	6.10	1.12	0.17	0.07	82.14	0.196	0.074	288.5	0.36	1.89	1.59

Table 4

Results of stepwise regression testing relationships between coral reef indicators and stressors: LDI (Landscape Development Intensity Index), ST (Sedimentation Threat), and WQ (Water Quality) with inclusion $p < 0.25$, retention $p < 0.20$. Transformation indicated to meet requirement for normal distribution. 3DTC: total (live + dead) three-dimensional coral cover, RI: rugosity index, SC_H': Shannon-Weiner diversity index, Sp_Den: density of sponge colonies. Equation coefficients show direction of relationship, ns: not significant

Indicator	Transformation	LDI		ST		WQ		Equation
		p value	Partial r^2	p value	Partial r^2	p value	Partial r^2	
3DTC	log10	0.08	0.252	ns	ns	ns	ns	3DTC = (LDI $\times$ - 0.05054)
RI	n/a	0.012	0.156	0.015	0.392	ns	ns	RI = (LDI $\times$ - 0.1088) + (ST $\times$ 0.00000659)
SC_H'	n/a	0.083	0.224	0.187	0.122	0.125	0.185	SC_H' = (LDI $\times$ - 0.24242) + (ST $\times$.0000082) + (WQ $\times$ 0.21111)
Sp_Den	log10	ns	ns	0.01	0.471	0.194	0.088	Sp_Den = (ST $\times$ 0.00001022) + (WQ $\times$ 0.09406)

Table 5

Results of stepwise regression testing relationships between colony density of dominant stony coral species and stressors (see Table 4 and text) with inclusion $p < 0.25$, retention $p < 0.20$. Transformation indicated to meet requirement for normal distribution. Equation coefficients show direction of relationship, ns: not significant

Stony Coral Species	Transformation	LDI		ST		WQ		Equation
		p value	Partial r^2	p value	Partial r^2	p value	Partial r^2	
<i>Orbicella annularis</i>	n/a	0.0089	0.4773	ns		ns		$LDI \times -4.02511$
<i>Montastraea cavemosa</i>	n/a	0.0436	0.3208	ns		ns		$LDI \times -2.86174$
<i>Orbicella faveolata</i>	log10	0.1469	0.1812	ns		ns		$LDI \times -0.18755$
<i>Orbicella franksi</i>	n/a	0.0866	0.2409	ns		ns		$LDI \times -2.33885$
<i>Porites astreoides</i>	n/a	ns		ns		ns		ns
<i>Siderastrea siderea</i>	n/a	ns		ns		ns		ns
<i>Pseudodiploria strigosa</i>	log10	ns		ns		ns		ns
<i>Porites porites</i>	n/a	0.0415	0.2822	ns		0.0258	0.2016	$(LDI \times -2.39872) + (WQ \times 3.61828)$

Table 6

Station indicators FoRAM Index (FI), invertebrate density (In_Den; # invertebrates/m² sea floor), gorgonian density (Gorg_Den; # gorgonians/m²), average gorgonian surface area (GSA_Av; m² gorgonian SA), sponge density (Sp_Den; # sponges/m²), average sponge surface area (SSA_Av; m² sponge SA), fish abundance (F_Abun; # fish/transect), fish biomass (F_Biom; total g/transect), F_Tx (number of fish taxa/transect), Shannon index of diversity for fish (F_H')

Station	FI	In_Den	Gorg_Den	GSA_Av	Sp_Den	SSA_Av	F_Abun	F_Biom	F_Tx	F_H'
1	3.38	0.8	5.4	0.19	0.6	0.03	213	796.1	21	1.87
2	3.15	3	0.6	0.2	0.8	0.08	134	2501.8	20	2.14
3	3.22	0.11	0	0	2.2	5.85	109	316.8	16	2.29
4	3.06	0.37	0.8	0.05	2	0.04	231	3046.7	19	1.98
5	3.72	0.31	1.2	0.06	3.4	0.18	151	2810.3	24	2.56
6	3.46	0.16	0	0	1.6	0.01	148	1644.8	21	2.47
7	6.48	0.04	0	0	1.6	0.01	132	1476.8	24	2.53
8	3.84	0.02	0	0	2	0.4	156	3509.4	19	2.47
9	4.88	0.09	0	0	0.4	0.1	176	2440.2	25	2.82
10	2.83	0.12	0.2	0.07	7.4	0.69	166	2664.6	22	2.25
11	3.34	0.02	0	0	2	0.1	153	5799.7	21	2.52
12	3.69	1.78	0	0	3	0.09	45	367.1	14	1.99
13	3.65	0.01	1.2	0.11	0.8	0.06	154	638	25	2.31