INTRODUCTION

Introduction

The U.S. Virgin Islands consists of three large islands, St. Thomas, St. John and St. Croix, and numerous smaller islands surrounded by a diverse, tropical marine environment that includes coral reefs, seagrass beds, and mangrove forests (Fig. 16). The islands of St. Thomas and St. John lie on the Puerto Rican Shelf, an extensive shallow water platform that connects them to Puerto Rico to the west and the British Virgin Islands to the east. St. Croix lies on an isolated platform sixty-five kilometers to the south of St. Thomas and St. John and separated by the 4000m deep Anegada Passage and the Virgin Islands Trough. This forms an effective barrier to the migration of adult coral reef fishes and invertebrates. The coral reefs of the Virgin Islands represent a wide range of characteristic coral reef habitats of the Caribbean, including patch reefs, fringing reefs, barrier reefs, shelf reefs, and extensive bank and slope mesophotic coral reef ecosystems. The area of a star coral bank mesophotic reef complex south of St. Thomas to Vieques covers more area than all of the shallow water coral reefs of the USVI combined (Smith et. al 2019a).

The economy of the US Virgin Islands is reliant to a large extent on maintenance of vibrant marine ecosystems. Tourism drives the economy of the Virgin Islands, famous for white sand beaches that give way to clean, clear marine waters. The diverse marine life of the coral reefs and other habitats attracts thousands of skin and scuba divers each year. Sport fishing on charter boats and private vessels also makes an important contribution to the economy. In addition, the coral reefs and other habitats in the Virgin Islands are essential to the lives of hundreds of thousands of species including economically important queen conch, whelk, spiny lobster, snapper, and grouper. Over three hundred full-time or part-time commercial fishermen work in territorial and federal waters surrounding all three islands (Tobias 1997). In tough economic times and after natural disasters, fishing is an important means of supplemental income or extra protein for many people.

Over the last few decades, major hurricanes, coral disease outbreaks, mass coral reef bleaching, and invasive species introductions have caused extensive coral mortality to the coral reefs surrounding the Virgin Islands (Gladfelter 1982; Edmunds and Witman 1991; Rogers et al. 1991; Rothenberger et al. 2008; Woody et al. 2008; Miller et al. 2009; Smith et al. 2013b). Recovery from these disturbances is hindered by a multitude of human impacts that affect coral reefs, such as overfishing of ecologically important species, physical damage to reef structure, and pollution (Hatcher 1984; Pastorok and Bilyard 1985; Rogers and Garrison 2001; Mumby 2006; Mumby et al. 2006; Mumby and Harborne 2010). Moreover, rapid development of steep island slopes has dramatically increased soil erosion and sedimentation into nearshore waters (Brooks et al. 2007; Gray et al. 2008; Smith et al. 2008), particularly below unpaved road surfaces (Anderson and Macdonald 1998; Ramos-Scharrón and MacDonald 2007a). Chronic sedimentation affects the abundance and diversity of corals and other reef organisms, increases coral stress and susceptibility to diseases and bleaching, and reduces the ability of corals and other reef organisms to recover and regenerate after natural disturbances such as hurricanes (Acevedo and Morelock 1988; Rogers 1990; Nemeth and Sladeck Nowlis 2001; Fabricius 2005; Sabine et al. 2015; Ennis et al. 2016). The first sightings of the invasive Indo-Pacific lionfish (*Pterois volitans*) occurred in the US Virgin Islands in 2009. This predator has the ability to dramatically alter coral reef fish community structure (Cote and Maljkovic 2010) and these alterations may have additional, indirect impacts on benthic communities (Albins and Hixon 2011). In addition, the invasive red alga *Ramicrusta* has increased in abundance at some locations and is killing coral tissue through competitive overgrowth (Eckrich and Engel 2013; Ballantine David et al. 2016).

INTRODUCTION

High thermal stress and coral bleaching events affected the northeastern Caribbean in 2005, 2010, and 2012, but these events had contrasting signatures in the United States Virgin Islands. These events and the species-specific responses of Caribbean corals are summarized in Smith et al. (2013) for shallow corals and Smith et al. (2016) for shallow and mesophotic corals. The year 2005 was the most severe high sea surface temperature (SST) event on record for the northeastern Caribbean (Eakin et al. 2010). In the Virgin Islands a peak of 10.25 Degree Heating Weeks (DHW) was registered from satellite SST records (NOAA, 2012) and a period of approximately 59 days above the local bleaching threshold of 29.5°C (Aug. 20 – Oct. 18); a level of thermal stress accumulation associated with severe coral bleaching and some mortality. The warm season of 2010 started as warm or warmer than 2005, with the bleaching threshold surpassed for 21 days between August 12 and September 2. In a clear example of ameliorative storm cooling (Manzello et al. 2007), the passing of the storm center of Hurricane Earl on August 30th, approximately 100 km to the northeast of the St. Thomas-St. John, caused a rapid decline in SST's below the bleaching threshold to 29.3°C, and then from October 5 - 8, the passage of Hurricane Otto caused windy and cloudy weather that further reduced SST below 29.1°C. Total DHW accumulated in 2010 began to decrease after the beginning of October, when it had reached 5.1 DHW (NOAA Coral Reef Watch, 50 km heritage product), a level associated with some bleaching and limited mortality. Recent research developed bleaching threshold temperatures for 24 of 33 TCRMP monitoring sites dominated by star corals of the genus *Orbicella* (Smith et al. 2016a). This research showed that mesophotic reefs bleached with shallow reefs in 2005 and then bleached when shallow reefs did not in 2012. The study concluded that mesophotic reefs of the USVI are unlikely to be long-term climate change refugia because they are not immune to high temperature thermal stress.

Most research around the Virgin Islands has focused on fringing reefs (5 – 30 m depth) located along the shoreline of the three main islands, St. Thomas, St. John, and St. Croix.

INTRODUCTION

In contrast, very little information exists for offshore and deeper reef systems, which can be quite extensive. These other reef systems include mid-shelf reefs (5 – 30 m depth) located 2 to 10 km from the shore of the main islands and mesophotic reefs (>30 m depth) located from 0.5 to 15 km offshore along the edge of the insular platform (Armstrong et al. 2002; Herzlieb et al. 2005; Armstrong et al. 2006; Armstrong 2007; Menza et al. 2007; Menza et al. 2008; Nemeth et al. 2008; Smith et al. 2010b; Smith et al. 2016b). Distance from shore may be a factor in the historical degeneration of coral reef systems in the Virgin Islands (Herzlieb et al. 2005; Calnan et al. 2008; Smith et al. 2008; Sabine et al. 2015; Ennis et al. 2016). A systematic approach to investigating these cross-shelf coral reef systems allows us to evaluate the variable impacts and synergistic effects of natural impacts and human-induced stress that influence the decline or recovery of Caribbean coral reef systems. The first two years of this project (2001 and 2002) concentrated on the fringing reefs surrounding St. Croix. In 2003, monitoring continued at St. Croix reefs and began at reef systems distributed across the insular platform surrounding St. Thomas. In 2004, 2005 and 2006 monitoring continued at reefs surrounding both islands, with additional reefs surrounding St. Thomas added in 2004, 2005, and 2011. Mesophotic coral reef monitoring sites were added to St. Croix during the 2008, 2009, and 2017 monitoring. In 2011, the TCRMP also expanded to include sites established under separate funding that will be continued in the core TCRMP monitoring activities funded by USVI DPNR and NOAA CRCP.

OBJECTIVES FOR MONITORING CORAL REEFS

Effective management is necessary to maintain the resources in the territorial and federal waters of the Virgin Islands in an ecologically and economically sustainable manner. Monitoring programs are essential for successful management because they provide managers with fundamental information with which to make and reinforce decisions. Standards for resource protection can be measured by comparison to baseline data established by monitoring. Monitoring also provides the means to assess the status and trends of ecological resources, allowing managers to determine the effectiveness of current management and to develop effective management plans for the future. The Territorial Coral Reef Monitoring Program monitors the condition of coral reefs throughout the U.S. Virgin Islands and provides key information to better manage these ecosystems. The TCRMP is complimentary to the National Coral Reef Monitoring Program (NCRMP) that started in 2013 and is co-coordinated in the USVI by the University of the Virgin Islands. TCRMP focuses on fixed sites and repeatedly samples the same corals to generate the most in-depth metrics of change over time. NCRMP uses a stratified-random sampling design to spread out samples and gain an understanding of change through time, with predictions that can be applied spatially. NCRMP does not sample reefs below 30m at this point, and therefore misses the dominant habitat in the northern USVI, which is only sampled in the TCRMP.

This report presents monitoring results from 2001-2018 in St. Croix and from 2003-2018 in St. Thomas and St. John. For both islands, temporal changes from year to year in the conditions of the reef communities are assessed.

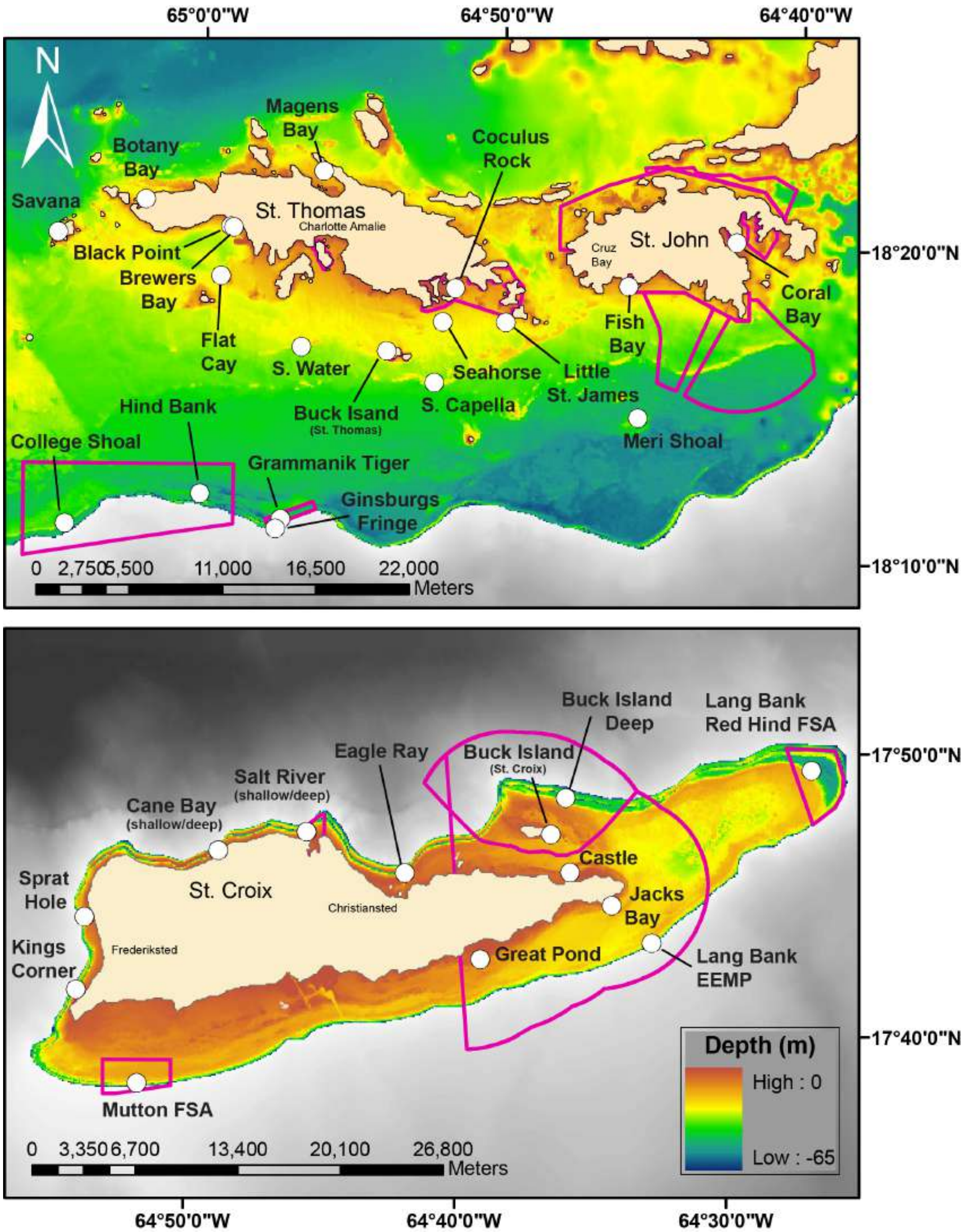

Figure 16. Locations of Territorial Coral Reef Monitoring Sites in the US Virgin Islands. Boundaries indicate federal and territorial marine protected areas.

Figure 17. A TCRMP research diver (T. Smith) on closed circuit rebreather records a fish transect at the lower mesophotic coral reef site Ginsburgs Fringe at 63m/220' depth (April 20, 2017; photo credit: V.W. Brandtneris).

METHODS

Methods

BENTHIC ASSESSMENTS

The University of the Virgin Islands determined the benthic composition at 34 long-term monitoring sites between 2001 and 2018 (Fig. 16). All data is now available at the TCRMP website and updated annually after quality control:

https://sites.google.com/site/usvitcrmp/home

Around St. Croix the following 15 sites were assessed: Buck Island-St. Croix, Cane Bay, Cane Bay Deep, Castle, Eagle Ray, Great Pond, Jacks/Isaacs Bay, Kings Corner, Lang Bank East End Marine Park (Lang EEMP), Lang Bank Red Hind Fish Spawning Aggregation (Lang Hind), Mutton Snapper, Salt River, Salt River Deep, and Sprat Hole. Four of these sites are within the St. Croix East End Marine Park boundary (Castle, Great Pond, Jacks Bay, Lang EEMP), two sites are in a territorially managed area associated with Salt River (Salt River West and Salt River Deep), Buck Island-St. Croix is within National Park Service boundaries, two sites are within federal fisheries marine protected areas (Lang Hind, Mutton Snapper), and five sites can be considered mesophotic coral reefs (Buck Island Deep, Cane Bay Deep, Lang Bank EEMP, Lang Hind, Salt River Deep; sensu Ginsburg 2007) . Salt River Deep transects 1-4 established at 40 m depth in April 2009 sampling were relocated upslope to 30 m in the January 2010 sampling due to low coral cover in the deeper transects.

Around St. John-St. Thomas the following 19 sites were assessed: Black Point, Botany Bay, Brewers Bay, Buck Island-St. Thomas, Coculus Rock, College Shoal East, Coral Bay, Fish Bay, Flat Cay, Ginsburgs Fringe, Grammanik Tiger FSA, Hind Bank FSA, Little St. James, Magens Bay, Savana Island, Seahorse Cottage Shoal (Seahorse), Meri Shoal, South Capella, and South Water Island. One site is the within the St. Thomas East End Reserve (Coculus Rock), four sites are within federal fisheries marine protected areas (College Shoal, Ginsburgs Fringe, Grammanik Tiger, Hind Bank), and five sites can be considered mesophotic coral reefs

(College Shoal, Ginsburgs Fringe, Grammanik Tiger, Hind Bank, Meri Shoal). Four sites were also part of the Ciguatera Fish Poisoning Monitoring Program and were surveyed monthly for benthic structure and coral health from 2010-2016 (Black Point, Coculus Rock, Flat Cay, Seahorse). Because of its deep depth, Ginsburgs Fringe at 60-66m was only sampled for benthic cover and some fish transects.

Benthic Cover. At each site benthic cover and coral health surveys were conducted along six 10 m long permanent transects marked with steel or brass rods. Video sampling consisted of one diver traversing each transect videotaping the benthic cover using a high definition standard definition digital video camera (prior to 2007) or a high definition digital camera (after 2007). TCRMP has attempted to continually upgrade video equipment through time to maintain the highest quality imagery possible for benthic analysis. The diver swam at a uniform speed, pointing the camera down and keeping the lens approximately 0.4 m above the substrate at all times. A guide wand or dropper weight attached to the camera housing was used to help the diver maintain the camera a constant distance above the reef. After taping, approximately 20 - 50 non-overlapping images per transect were captured and saved as JPEG files (Fig. 11). Captured images represented an area of reef approximately 0.31 m^2 (0.64 m x 0.48 m). Coral Point Count with Excel Extension software (Kohler and Gil 2006) was used to superimpose randomly located dots on each image. The number of points varied with the evolution of the video camera systems and was 10 points from 2001-2011, 15 points from 2012-2013, and 20 points from 2014 onwards. The substrate type located under each of the dots was then identified to the most descriptive level possible and entered into a database. Where multiple benthic cover categories fell under a single point, for example macroalgae over bedrock, the upper benthic category was assessed. For each transect, the percent cover of coral, epilithic algae (formerly called dead coral with turf algae), macroalgae, sponges, gorgonians, and sand/sediment were calculated by dividing the number of random dots falling on that substrate type by the total number of dots for that transect. Epilithic algae (sensu Hatcher and Larkum 1983) are diminutive turfs and filamentous algae without thallus structure that cover all rock surfaces of coral reefs not occupied by larger epibenthic organisms. They can also be considered to be grazed surfaces and are often an indicator of healthy grazing communities and high sessile animal cover.

Figure 18. A screen grab of benthic video used for the determination of percent cover of coral reef organisms and non-living substrate.

Table 1. TCRMP site reef complex type, location coordinates (decimal degrees; WGS 1984), and depths.

Island	Site	Reef Complex	Lat	Long	Depth (m)
St. Croix	Buck Island-St. Croix	Offshore-Shallow	17.78500	-64.60917	15
	Buck Island Deep-St. Croix	Offshore-MCE	17.80659	-64.59935	33
	Cane Bay	Nearshore	17.77388	-64.81350	10
	Cane Bay Deep	Offshore-MCE	17.77661	-64.81522	38
	Castle	Offshore-Shallow	17.76278	-64.59743	7
	Eagle Ray	Offshore-Shallow	17.76150	-64.69880	10
	Great Pond	Nearshore	17.71097	-64.65221	6
	Jacks Bay	Nearshore	17.74337	-64.57160	14
	Kings Corner	Nearshore	17.69116	-64.90008	17
	Lang Bank EEMP	Offshore-MCE	17.72145	-64.54706	27
	Lang Bank Red Hind FSA	Offshore-MCE	17.82372	-64.44943	33
	Mutton Snapper FSA	Offshore-Shallow	17.63660	-64.86240	24
	Salt River Deep	Offshore-MCE	17.78523	-64.75917	30
	Salt River West	Nearshore	17.78530	-64.75940	11
	Sprat Hole	Nearshore	17.73400	-64.89540	8
St. John	Coral Bay	Nearshore	18.33797	-64.70402	9
	Fish Bay	Nearshore	18.31417	-64.76408	6
	Meri Shoal	Offshore-MCE	18.24433	-64.75832	30
St. Thomas	Black Point	Nearshore	18.34450	-64.98595	9
	Botany Bay	Nearshore	18.35845	-65.03330	8
	Brewers Bay	Nearshore	18.34403	-64.98435	6
	Buck Island-St. Thomas	Offshore-Shallow	18.27883	-64.89833	14
	Coculus Rock	Nearshore	18.31257	-64.86058	7
	College Shoal East	Offshore-MCE	18.18568	-65.07677	30
	Flat Cay	Offshore-Shallow	18.31822	-64.99104	12
	Ginsburgs Fringe	Offshore-MCE	18.18770	-64.95998	63
	Grammanik Tiger FSA	Offshore-MCE	18.18885	-64.95659	38
	Hind Bank East FSA	Offshore-MCE	18.20217	-65.00158	39
	Magens Bay	Nearshore	18.37425	-64.93438	7
	Savana	Offshore-Shallow	18.34064	-65.08205	9
	Seahorse Cottage Shoal	Offshore-Shallow	18.29467	-64.86750	20
	South Capella	Offshore-Shallow	18.26267	-64.87237	20
	South Water	Offshore-Shallow	18.28068	-64.94592	20
	St James	Offshore-Shallow	18.29459	-64.83238	15

Table 2. TCRMP site sampling data (benthic/health) and type of sampling.

Island	Site	Date Sampled	Benthic	Health	Fish/Urchin
St. Croix	Buck Island STX	11/1/18	x	x	x
	Buck Island Deep STX	11/1/18	x	x	x
	Cane Bay	11/5/18	x	x	x
	Cane Bay Deep	11/5/18	x	x	x
	Castle	11/2/18	x	x	x
	Eagle Ray	11/2/18	x	x	x
	Great Pond	11/7/18	x	x	x
	Jacks Bay	11/2/18	x	x	x
	Kings Corner	11/3/18	x	x	x
	Lang Bank EEMP	11/6/18	x	x	x
	Lang Bank Red Hind FSA	11/6/18	x	x	x
	Mutton Snapper FSA	11/3/18	x	x	x
	Salt River Deep	11/4/18	x	x	x
	Salt River West	11/4/18	x	x	x
	Sprat Hole	11/5/18	x	x	x
St. John	Coral Bay	10/25/18	x	x	x
	Fish Bay	10/25/18	x	x	x
	Meri Shoal	12/7/18	x	x	x
St. Thomas	Black Point	11/21/18	x	x	x
	Botany Bay	11/24/18	x	x	x
	Brewers Bay	10/11/18	x	x	x
	Buck Island STT	11/26/18	x	x	x
	Coculus Rock	11/25/18	x	x	x
	College Shoal East	12/5/18	x	x	x
	Flat Cay	11/28/18	x	x	x
	Ginsburgs Fringe	12/12/18	x		x
	Grammanik Tiger FSA	12/4/18	x	x	x
	Hind Bank East FSA	12/5/18	x	x	x
	Magens Bay	11/27/18	x	x	x
	Savana	11/27/18	x	x	x
	Seahorse Cottage Shoal	11/28/18	x	x	x
	South Capella	10/11/18	x	x	x
	South Water	12/4/18	x	x	x
	St James	12/7/18	x	x	x

METHODS

Coral Health. Coral health assessments followed methodologies outlined in Calnan et al. 2008, Smith et al. 2008, and Smith et al. 2013 and are briefly described here. All coral colonies located directly under the transect lines were assessed in situ for signs of mortality and disease following a modified Atlantic and Gulf Rapid Reef Assessment protocol (Kramer et al. 2005). Starting in 2008 all colonies were assessed, regardless of size, in contrast to previous years where only colonies greater than 10 cm in maximum linear dimension were assessed. Partial mortality of coral colonies was broken into two categories. Recent partial mortality was characterized visually as skeleton not eroded (fine corallite structure still intact) and bare or with a thin veneer of sheeting or filamentous algae. Recent partial mortality is typically visible for up to three months following tissue loss. Old partial mortality was characterized as skeleton eroded and covered with turf or macroalgae. Old partial mortality is a transition from recent mortality and typically lasts up to 1–6 years (Smith et al. 2008, also see http://www.agrra.org/method/methodcor.html).

Diseases were conservatively categorized into recognized Caribbean scleractinian diseases and syndromes that included bleaching, black band disease, dark spots disease, white plague, and yellow band (blotch) disease (following Bruckner 2007). Acroporid corals were extremely rare at the study sites; thus, their associated diseases (white band and white pox) are not presented. Bleaching was assessed as abnormal paling of the colony, and, when present, the severity of the bleaching (paling or total whitening) and the area of the colony affected were assessed. A major bleaching event occurred between September and December 2005 affecting all sites monitored that year, a mild bleaching event occurred September and October 2010 affecting only shallow sites, and a mild bleaching event occurred in October and November 2012 and affected only mesophotic sites (Smith et al. 2013b; Smith et al. 2016a). On St. Croix, a subset of sites were assessed during the 2010 coral bleaching event, and included Cane Bay, Cane Bay Deep, and Jacks Bay.

For each transect, the prevalence of coral impairment categories was calculated as the number of colonies with partial mortality, disease, or bleaching divided by the number of colonies assessed. Also, for affected colonies in each transect the average three-dimensional surface area (%) of the colony affected was also estimated for each impairment category.

FISH CENSUS

Fish surveys were conducted on 17 sites in the northern USVI in 2017 before the two hurricanes, Irma and Maria, passed in September 2017 (Table 2). Unfortunately, the building housing the raw data was destroyed during the storms, before data entry, and all data sheets for two sites, Flay Cay and St. James were lost. Hind Bank East FSA was sampled post-storm in 2018 and nine sites were at least partially resampled. These sites included Buck Island STT, Botany Bay, Black Point, Coculus Rock, Magens Bay, Savana, Coral Bay and St. James. Several of these sites were sampled in December 2017, and again in March or April 2018. Fish surveys were conducted at 14 sites around St. Croix in early 2018. Ten replicate belt transects and three replicate roving dive surveys (RDS) were conducted at each site on St. Croix and during pre-storm sampling in the northern USVI. During post-storm sampling as many replicates were conducted as possible given time constraints (1-10). Belt transects were 25m x 4m and conducted in 15 minutes per replicate following protocols established by the NOAA Biogeography Branch (Menza et al. 2006; Friedlander et al. 2013). All transects were begun at a random location on the site and were swum in a random direction. RDS replicates were 15 min in duration. In previous years 30 minute RDS surveys were conducted in depths less than 20m. Because almost all diversity was captured in the first 15 minutes, for easier diving logistics, and to make deeper sites comparable, the methods were switched to 15 minute RDS at all sites in 2016. In addition to relative abundance data, specific total length estimates were made for each large grouper, large snapper, or hogfish (*Lachnolaimus maximus*) encountered. In all surveys, all species encountered were recorded except blennies and gobies. Data were transcribed to Microsoft Excel and Access spreadsheets and were analyzed for descriptive statistics of reef fish assemblage structure.

Divers also counted the number of *Diadema antillarum* sea urchins within 1 m on either side of a transect. From 2001 – 2008 this occurred along the 6 – 10 m long benthic transects. Starting in 2009, urchins were assessed along 25x2m belt transects corresponding to the return of the 10 fish transects. The mean number of sea urchins per 100 m^2 was calculated for each site.

TCRMP MONITORING SUMMARY

Territorial Coral Reef Monitoring Summary

TEMPERATURE

The general sea surface temperature for the USVI is presented here as background for overall coral condition and site-specific temperatures presented in the "Site Summaries" section (Fig. 19).

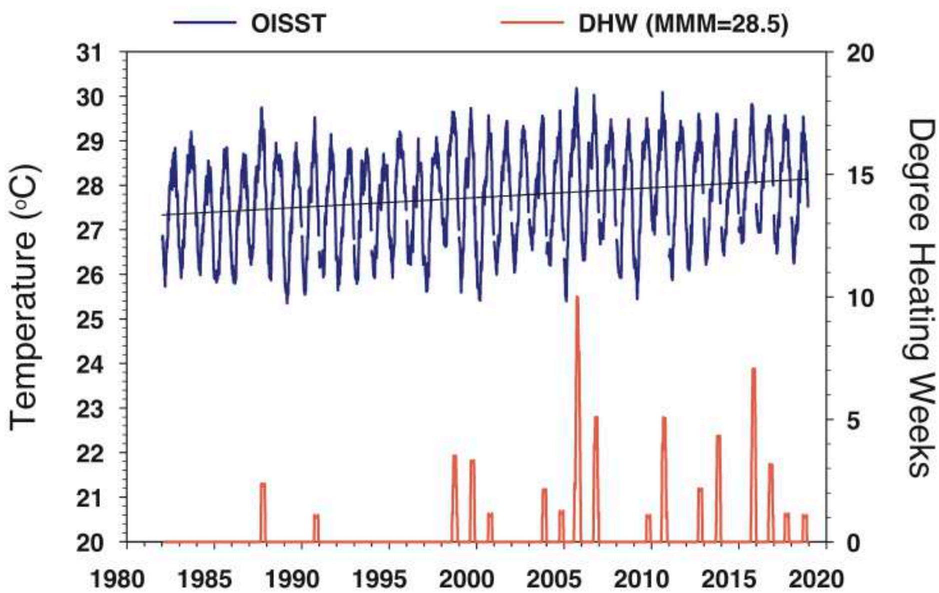

Figure 19. Sea surface temperatures and coral degree heating weeks of the US Virgin Islands from 1982 - 2019

Optimum Interpolation Sea Surface Temperature (OISST; blue line, left vertical axis) and degree heating weeks (red line, right vertical axis) for the USVI. The black line is a linear fit of the OISST data and shows about 0.007°C increase in temperature per year (y = 0.000669/year*x – 25.545). Degree heating weeks (DHW) are calculated as the 12 week rolling sum of temperatures exceeding 1°C over the monthly maximum mean temperature, which is estimated at 28.5°C for the USVI (NOAA 2006). DHW values above 4 are associated with the onset of bleaching, and above 8 with the onset of mass bleaching and coral mortality. OISST values averaged from coordinates 17.5N/65.5W, 17.5N/64.5W, 18.5N/65.5W 18.5N/64.5W from https://www.ncdc.noaa.gov/oisst; Accessed June 6, 2019.

TCRMP MONITORING SUMMARY

BENTHIC COMMUNITIES AND CORAL REEF HEALTH

Benthic cover was monitored at 34 monitoring sites and coral health was monitored at 33 sites in 2018. Benthic cover raw data is presented in electronic Appendix I. Coral health raw data is presented in electronic Appendix II. In addition, updated benthic cover for each site individually is presented in the "Site Summaries" section.

Coral Cover

The cover of hard corals decreased at most sites immediately after the 2005 coral bleaching event, but showed little or no change as the results of the 2010 and 2012 coral bleaching events (Fig. 20). Shallow (<25 m depth), nearshore and offshore sites with greater than about 20% coral cover showed declines in cover, but there was extreme variability in the degree of cover change. For example, the relative loss in coral cover of shallow star coral (*Orbicella* spp.) dominated reefs ranged from 87% at the Mutton Snapper site to less than 20% at the Brewers Bay site. Mesophotic coral monitoring sites that were sampled before and after the 2005 coral bleaching event showed lower relative losses of coral cover compared with shallow site. Losses ranged from 5.4% (Grammanik Tiger) to 36.0% (Meri Shoal).

Sites that had low coral cover prior to 2005 lost far less relative cover as the result of bleaching. While part of this may be due to detectability at coral cover values nearer to 0, it is also true that these sites tend to be dominated by small massive species that are more resistant to bleaching and disease related mortality (Smith et al. 2013b). A few sites showed no change, and included Buck Island STT, Coculus Rock, Jacks Bay, and South Water. However, the Buck Island STT site may be anomalous since transects were not permanently placed until 2007 and unprecedented prevalence of white disease was seen at this site in 2006.

Recovery since bleaching in 2005 was marginal at most sites. The majority of sites had apparently level coral cover with recovery potentially inhibited by disease and increased interactions with other organisms. However, slow and irregular upward trajectories are notable at some sites, including Black Point, Botany Bay, Cane Bay, Fish Bay, Lang Hind, Salt River West, Salt River Deep, Seahorse, and St. James. Generalities that might indicate why these sites are recovering are difficult, but the coral communities in these reefs are all diverse. This diversity may contribute to recovery as fast growing species, such as *Agaricites* spp. and *Porites porites* may lead increases in coral cover.

Some sites also showed degradation since 2007, when direct impacts of the 2005 bleaching abated. This was indicated by declines in coral cover and the sites include College Shoal, Ginsburgs Fringe, Grammanik Tiger, Magens Bay, Meri Shoal, and Savana. Four of six sites that were declining are mesophotic coral reefs, which may reflect the impact of generally higher prevalence of white diseases at high coral cover deep sites and a mild bleaching event that occurred in 2012.

In addition, the deep Ginsburgs Fringe site lost 62% of its coral cover between 2011 and 2018, in what appears to be a continuous decline. While lionfish are frequent at this site and there is high cover of the macroalgae *Lobophora variegata*, the most obvious cause of disturbance is anchoring (Smith et al. 2019b). A derelict reef claw anchor with at least 30m of polypropylene line was seen embedded in the monitoring site in 2014. Since damage has been recurrent it is likely that one or a few people are repeatedly anchoring on the edge to fish the Grammanik Bank. The activities have broken large plates and overturned portions of a large section of the large *Agaricia* spp. colonies that compose this reef. Ginsburgs Fringe is just along the border of the Grammanik Bank Federal Fisheries Managed Area and the site of a multi-species spawning aggregation, including Nassau grouper and yellowfin grouper (Kadison et al. 2006; Nemeth et al. 2006; Nemeth and Kadison 2013). Anchoring was likely for the purpose of fishing within the seasonal closed area, as there is little other obvious reason for anchoring at the shelf edge in deep water. Impacts to the corals and other essential fish habitat at this site may indirectly harm fishing in the US Virgin Islands.

Two nearshore shallow sites that are degrading since 2007 may be impacted for different reasons. Magens Bay is highly impacted by sedimentation since it is largely enclosed, is surrounded by steep hillsides under constant development (sediment run-off), and is susceptible to strong winter swells (Rothenberger et al. 2008). Degradation at this site may primarily be the result of sediment impacts. On the other hand, Savana is an offshore and uninhabited island next to the typically clear waters of the Virgin Passage. Degradation at this site can be largely attributed to encrusting alga (*Ramicrusta* sp.), which has been competing for benthic space and slowly decreasing coral cover by overtopping colony margins.

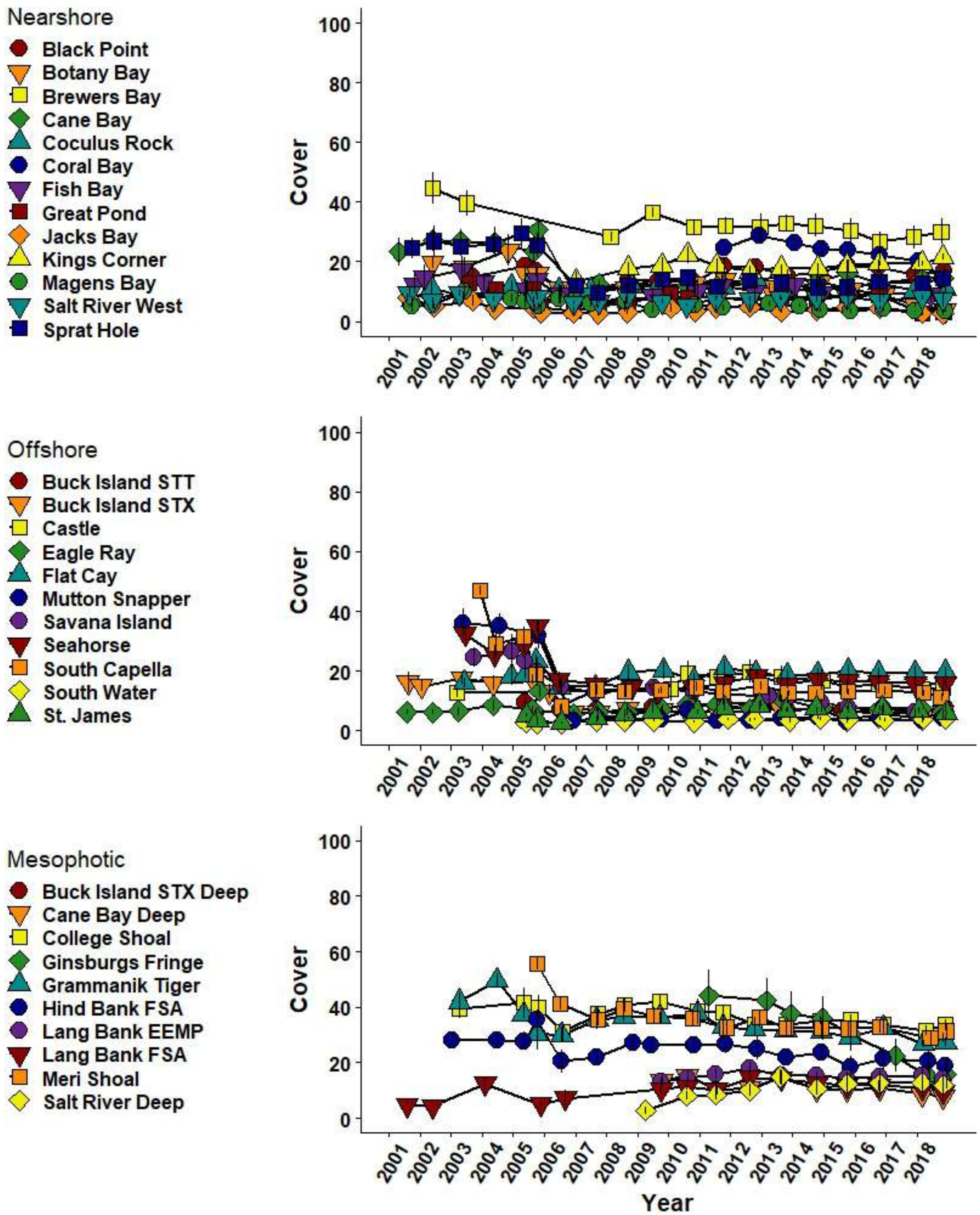

Figure 20. Coral cover (±SE) across TCRMP monitoring sites over time.

Epilithic Algal Community Cover

Algae show the highest inter-annual variability of any group of benthic organisms (Fig. 21). This is largely due to seasonality. The cover of epilithic algae is no exception, since it tends to negatively covary with more ephemeral macroalgae. Epilithic algae is important as it can indicate substrates grazed by herbivores and therefore open to the settlement of sessile epibenthic animals, including coral. Therefore, declines in the cover of epilithic algae (or increases in the cover of macroalgae and filamentous cyanobacteria) could be an early indication of declining herbivory at sites.

Some offshore sites, such as Eagle Ray, Buck Island-St. Croix, and Savana, appear to have a declining abundance of epilithic algae over the extent of the monitoring. Large recent declines in epilithic algae at Savana are due to increases in the "macroalgae" *Ramicrusta* (see next section). Nearshore and mesophotic sites typically showed little inter-annual trend in epilithic algal cover, although interannual variability was high at many nearshore monitoring sites.

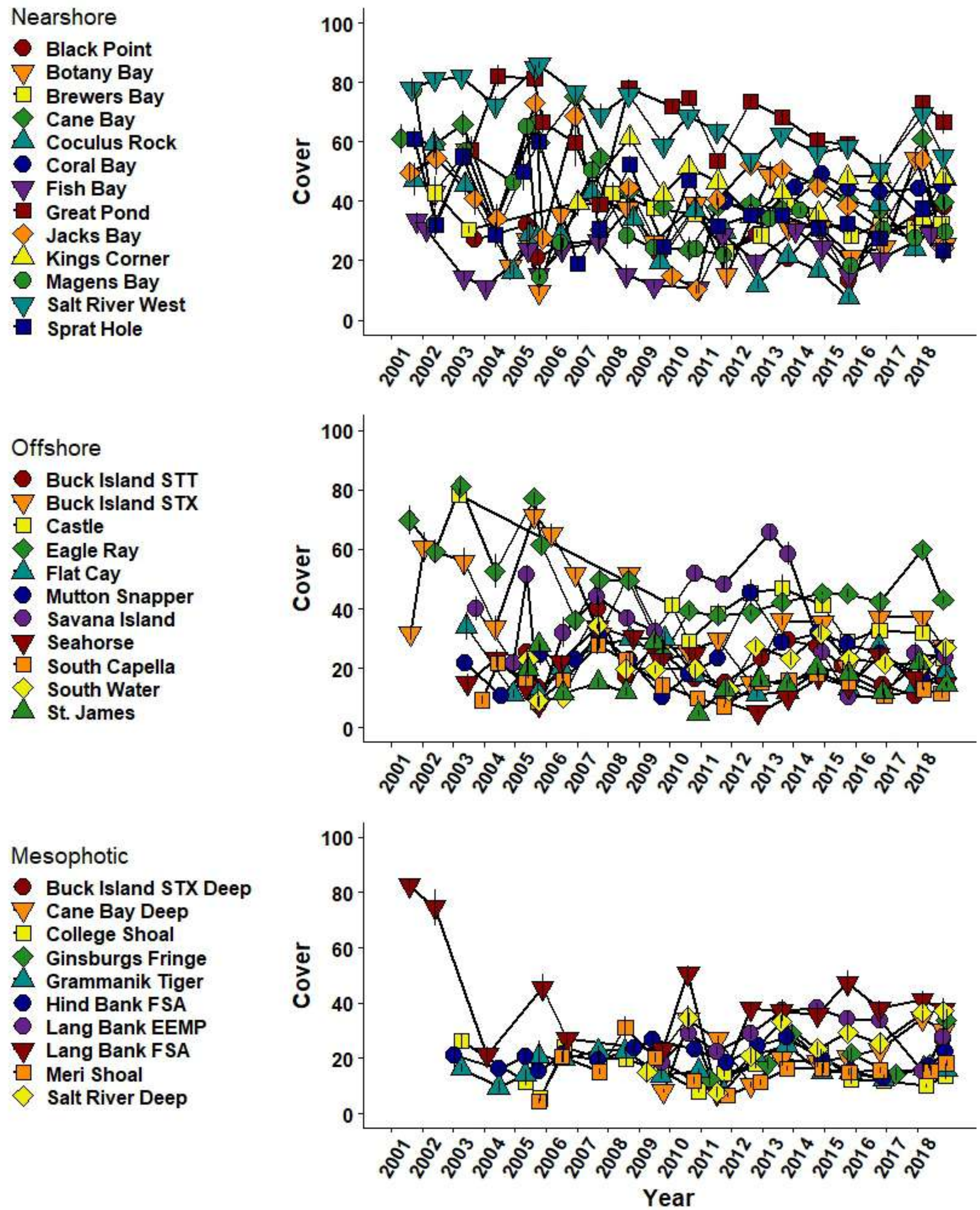

Figure 21. Epilithic Algal Community cover (±SE) across TCRMP monitoring sites over time.

Macroalgal Cover

Macroalgae have been increasing at many reefs, particularly after the 2005 bleaching event (Fig. 22). At sites where there was no loss of coral cover, increased macroalgae may be due to declining grazing, such as at Eagle Ray. At other sites where coral cover dropped after 2005, space opened for algal colonization by coral die-off may have been taken by macroalgae. This might occur where resident herbivores communities are already at the threshold of maximum grazing rates (Williams et al. 2001). This process could be enhanced where herbivores numbers are falling due to fishing. These reefs include: the Buck Islands (St. Thomas and St. Croix), Cane Bay, Meri Shoal, South Capella, and Sprat Hole. The same explanation may also apply for filamentous cyanobacteria (see following section). At Savana the large increase in macroalgae in 2014 and continuing to 2015 was due to a large increase in encrusting *Ramicrusta*, which was mentioned above as a cause of declining coral cover (*Ramicrusta* is classified with macroalgae in TCRMP data summaries despite its largely encrusting habitat). This increase in *Ramicrusta* sp. was also at the expense of epilithic algae.

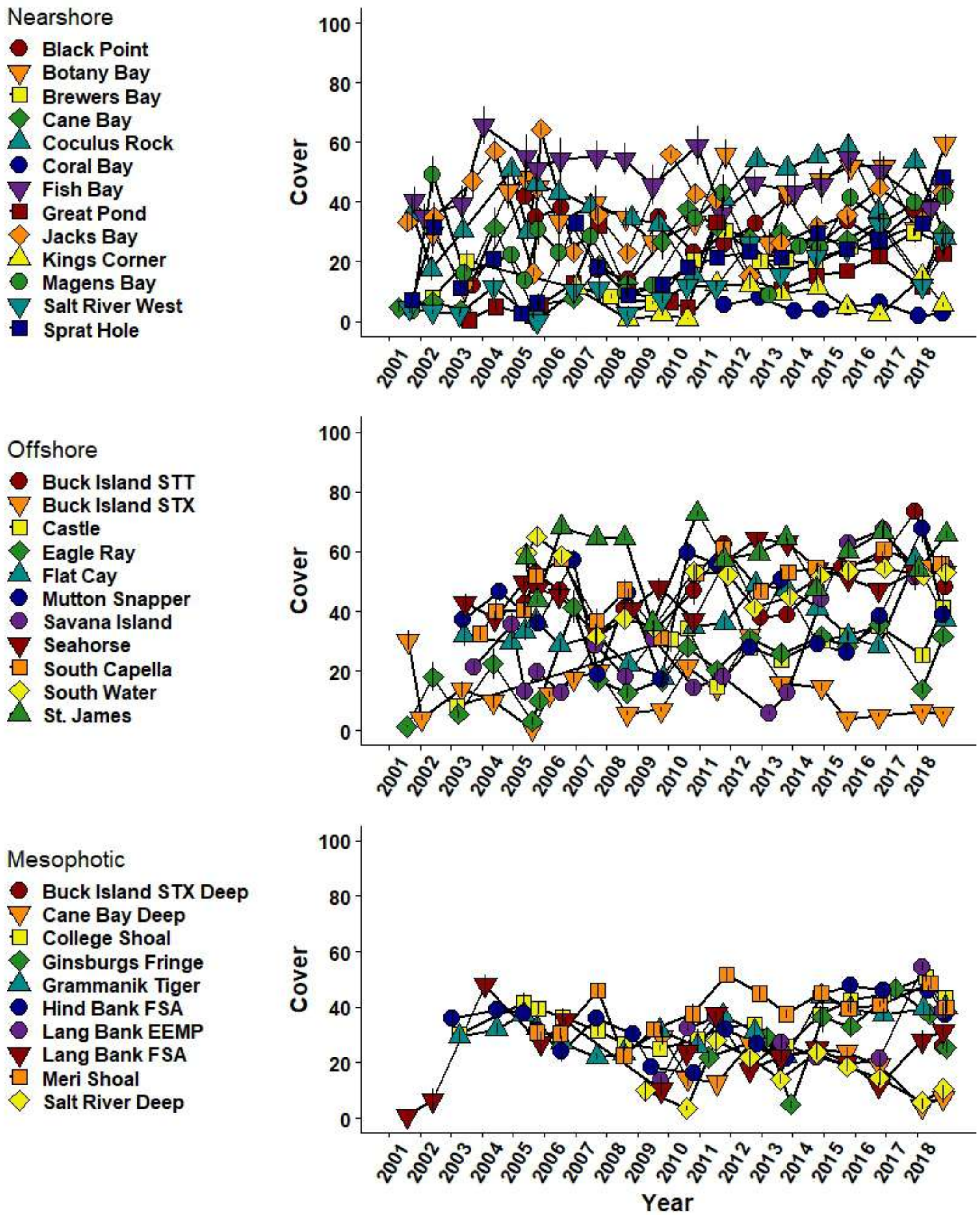

Figure 22. Macroalgae cover (±SE) across TCRMP monitoring sites over time.

Filamentous Cyanobacteria

Filamentous cyanobacteria cover has been increasing at many sites in the TCRMP since the 2005 coral bleaching event (Fig. 23). In many cases this was a multi-year peak that has abated, but at some sites high cover relative to baseline has persisted until 2014. This is particularly true at many sites on St. Croix. For example, Salt River West, Jacks Bay, Cane Bay, Sprat Hole, Mutton Snapper, Buck Island-St. Croix, Eagle Ray, Lang Bank EEMP, and Lang Bank Hind have all seen cover of filamentous cyanobacteria from 10 – 60%, with 2009 as a particularly high abundance year for offshore sites.

The increased incidence of filamentous cyanobacteria can be an indication of disturbance, increased nutrient inputs, and insufficient grazing (Fong and Paul 2011). In addition, filamentous cyanobacteria can promote increases in palatable macroalgae in coral reefs by coating and protecting algae with secondary metabolites that deter grazing (Fong et al. 2006; Smith et al. 2010a). Filamentous cyanobacteria can inhibit the recruitment of coral larvae (Kuffner et al. 2006) and have been observed interacting at the borders of adult coral (TCRMP, unpub. data). Monitoring the trends of filamentous cyanobacteria in USVI reef systems will be increasingly important in future years in an effort to understand the factors influencing bloom formation and which reefs are most vulnerable.

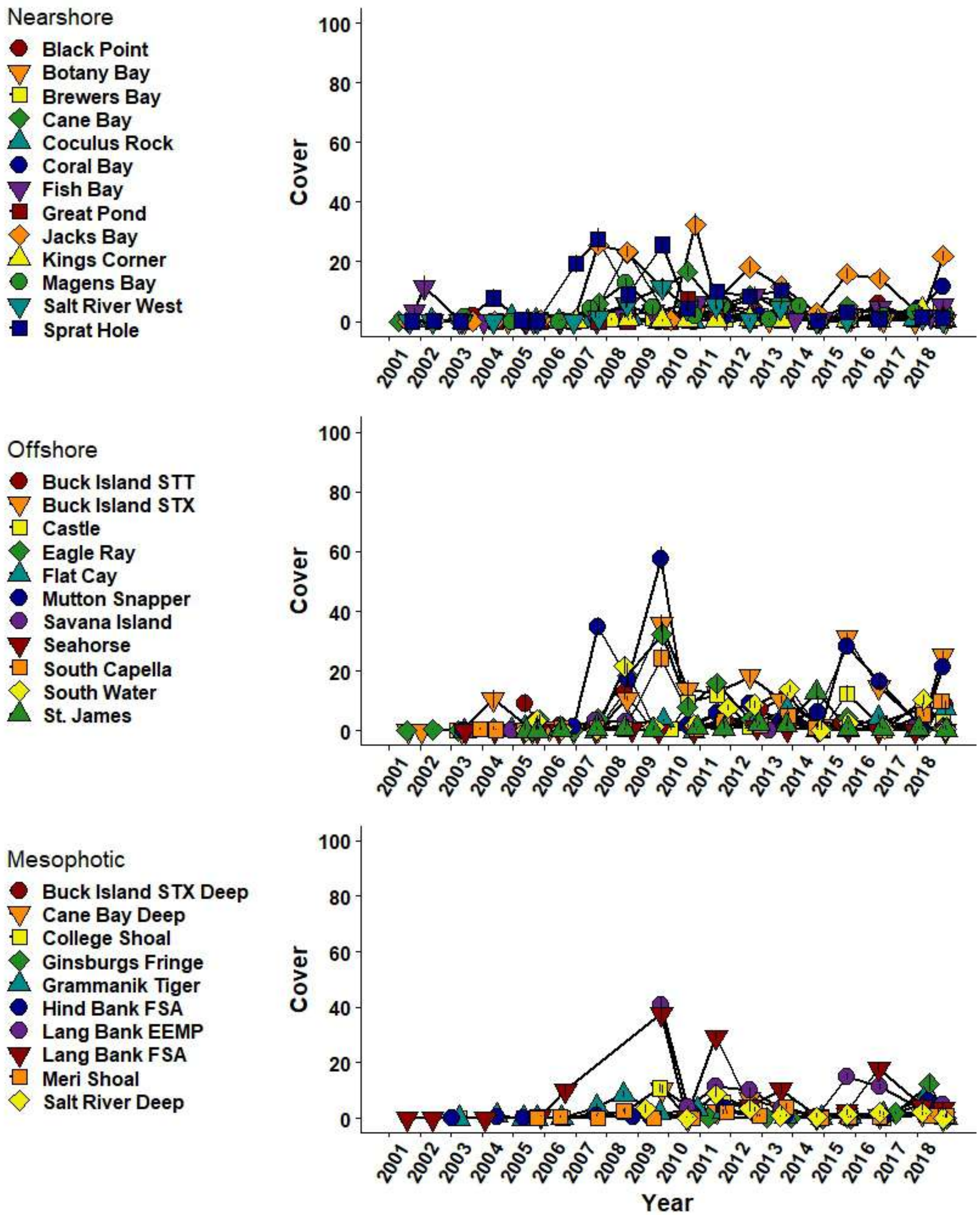

Figure 23. Filamentous cyanobacteria cover (±SE) across TCRMP monitoring sites over time.

Gorgonian and Antipatharian Cover

The cover of gorgonians and antipatharians has been fairly constant at most monitoring sites throughout the years of monitoring (Fig. 24). These groups did not seem sensitive to the thermal stress events in 2005, 2010, and 2012 (Tsounis and Edmunds 2017). In most cases they are a relatively minor component of cover because of their upright growth form and small branches. At Coral Bay, Fish Bay, and Magens Bay the cover of gorgonians has been increasing through the monitoring time series. These sites are known to have water quality issues and a high influx of terrestrial sediments. It is possible that inputs of nutrients from terrestrial run-off and poor sewage disposal are stimulating pelagic primary productivity (Furnas et al. 2005) and increasing the abundance of gorgonians that can feed heterotrophically on water column resources (De'ath and Fabricius 2010). Increasing abundance of octocorals has also been detected at permanent monitoring sites in Lameshur Bay, St. John under investigation for 30 years (Tsounis and Edmunds 2017).

Note that Black Corals (antipatharians) are rare and when they occur tend to be more prominent in deep monitoring sites. For many gorgonians species their abundance tends to peak in shallow water where there is constant swell (benthic orbital turbulence).

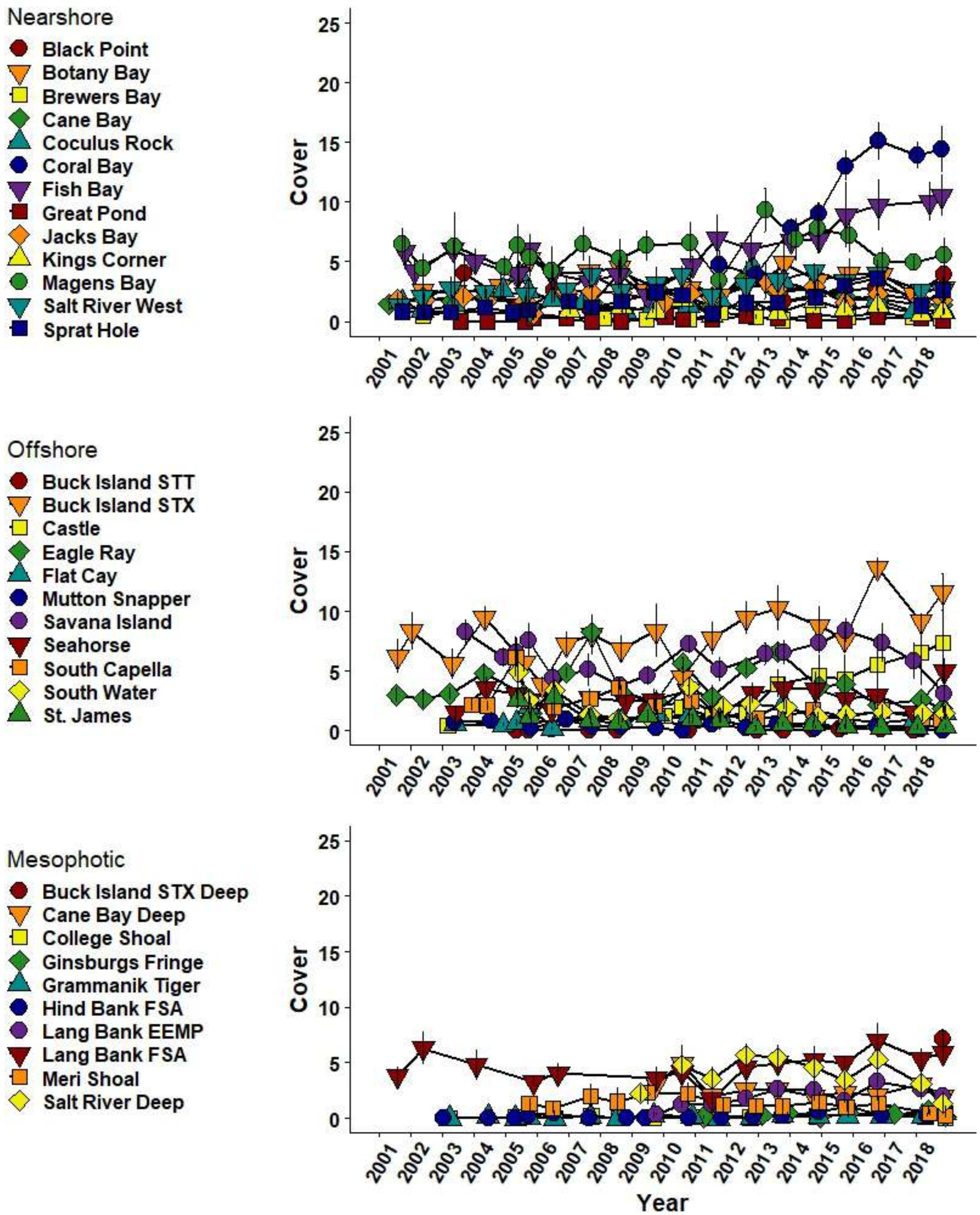

Figure 24. Gorgonian and Antipatharian cover (±SE) across TCRMP monitoring sites over time.

Sponge Cover

Sponge cover has been constant or variable at many offshore and mesophotic sites, but there is an indication of slightly increasing sponge cover at some nearshore sites (Fig. 25). Nearshore increases were most pronounced at Black Point, Coral Bay, and Magens Bay. This increase in epibenthic and boring sponges may indicate increasing supplies of food, such as bacteria and small eukaryotes, in nearshore environments. This may be a consequence of increasing nearshore nutrient pollution. Further study needs to be done to establish this linkage. Sites such as Black Point and Flat Cay showed declines in sponge cover in the 2017 monitoring year, possibly due to the impacts of Hurricane Irma and Hurricane Maria.

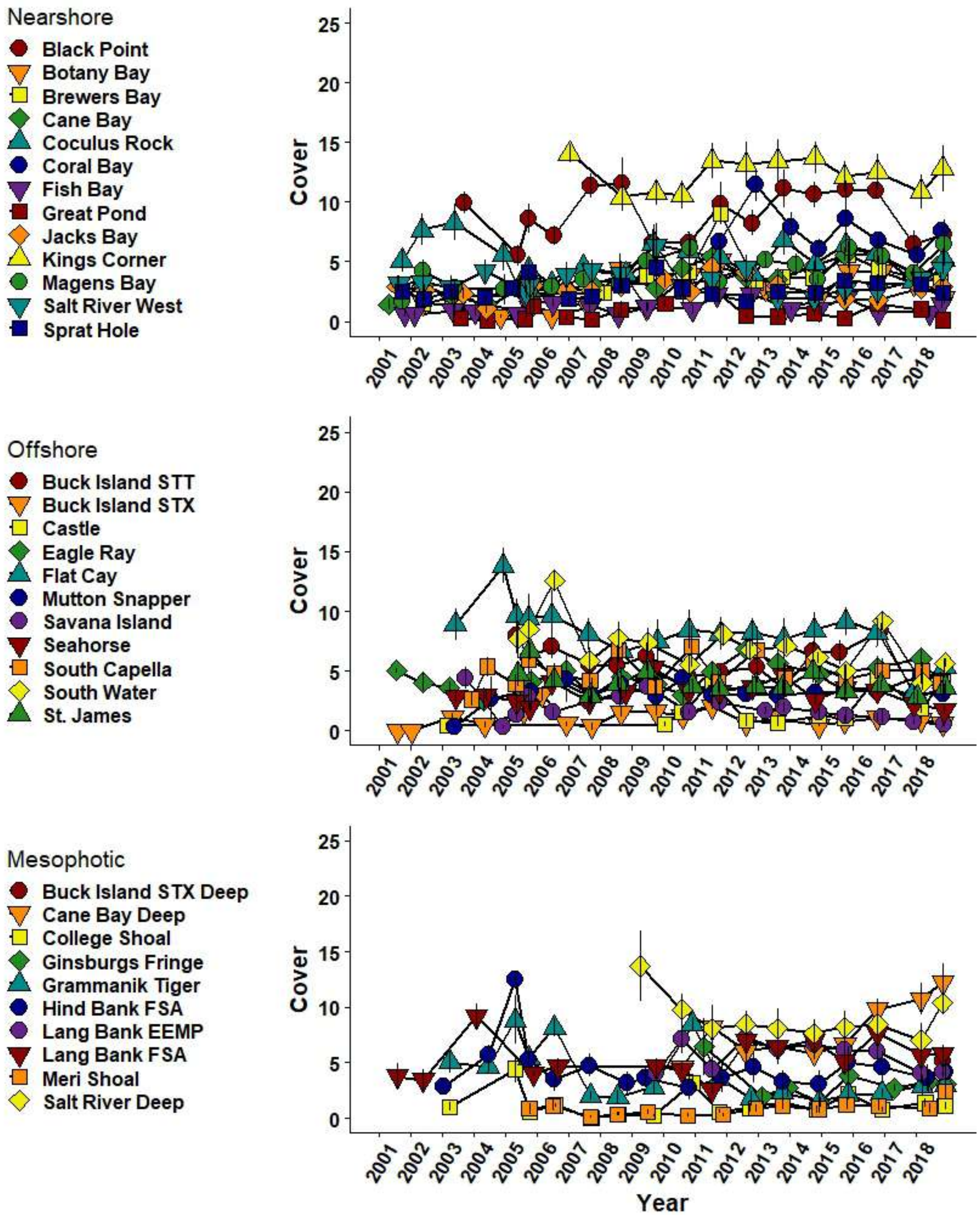

Figure 25. Sponge cover (±SE) across TCRMP monitoring sites over time.

TCRMP MONITORING SUMMARY

FISH COMMUNITIES

In 2018 a total of 58,044 fish representing 148 species and 38 families were recorded over 183 belt transects across 19 sites off the northern USVI. Total calculated biomass on sites of the northern USVI was 3219.04kg. An additional 33,984 fish representing 128 species and 45 families were recorded over 151 transects across all 15 sites off St. Croix. The calculated biomass was 1,198.54 kg. Using roving diver surveys (RDS) 141 species representing 36 families were observed in 2018 off the northern USVI and 126 species representing 38 families off St. Croix in 2018. Species richness was variable between sites but was similar between survey methods (transects and RDS). No differences in species richness were apparent between nearshore, offshore, and mesophotic sites (Table 3). As in previous years, sites with notably high species diversity in the northern USVI were Hind Bank East FSA, South Capella, and St. James (29.3 ±2.1, 30.9 ± 1.6 and 28.7 ± 3.3 species transect^{-1}, respectively). Cane Bay Shallow and Sprat Hole off St. Croix had the highest species richness of sites on the St. Croix shelf (27.2 ± 1.1 and 29.0 ± 1.4 species transect^{-1}, respectively). On St. Croix, the sites with the lowest species richness was the mesophotic site Salt River Deep (13.2 ± 3.4 species transect^{-1}), and the nearshore site Great Pond (17.4 ± 1.4 species transect^{-1}). This site is shallow with low living coral cover. Likewise, the shallow, low coral cover site in Coral Bay had the lowest species richness on the northern USVI shelf (18.0 ± 0.8 species transect^{-1}). The 65m site, Ginsberg's Fringe, also had a low species richness (17.5 ± 0.7). Only two transects were conducted on Ginsberg's Fringe.

Overall fish size distribution in both the northern USVI and St. Croix followed trends seen in earlier years. Over 42% of all individuals counted off the northern USVI and 53% in St. Croix were less than 5cm TL. Over 75% were less than 10cm TL. Large fish (> 40cm TL) constituted 0.5% of the numeric total in the northern USVI (265 fish), and 0.1% in St. Croix (35 fish). Numerically the most dominant fish in the northern USVI were the creole wrasse (*Clepticus parrae*), striped parrotfish (*Scarus iserti*), bluehead wrasse (*Thalassoma bifaciatum*), blue chromis (*Chromis cyanae*), and brown chromis (*C. multilineatum*). These species made up 50% of the numeric total of all fish reported. Four of these species were also the most abundant in the 2017 monitoring report. On St Croix reefs, blue chromis, creole wrasse, bicolored damselfish (*Stegastes partitus*), bluehead wrasse, and brown chromis contributed 63% to the numeric total of all sites. They species are ubiquitous and occurred on all sites. The four species that contributed most to biomass in the northern USVI included the creole wrasse, yellowtail snapper (*Ocyurus chrysurus*), and horseeye jack (*Caranx latus*). These made up 21% of total biomass. In St. Croix the fish contributing most to biomass included the creole wrasse, blackbar soldierfish (*Myripristis jacobus*) and Caribbean reef shark (*Carcharhinus perezi*). These species made up 24% of the biomass recorded.

Table 3. The 2018 species richness for belt transects and roving diver surveys (RDS). Sites are divided into nearshore, offshore, and mesophotic sites as described in the text.

		Belt Transects (25x4)		RDS
		Total Number of Species	Mean species per transect (±SE)	Total Number of Species
Nearshore	Cane Bay	70	27.2±1.1	59
	Great Pond	47	17.4±1.4	38
	Jacks Bay	66	24.9±2.0	56
	Kings Corner	72	29.0±2.6	50
	Salt River West	44	17.8±1.6	37
	Sprat Hole	75	29.0±1.3	62
	Coculus Rock	45	23.5±1.1	65
	Black Point	66	23.8±2.1	66
	Brewers Bay	63	24.4±150	55
	Botany Bay	72	27.1±1.3	76
	Buck Island, St. Thomas	60	23.7±0.8	72
	Coral Bay	54	18.0±1.7	38
	Fish Bay	58	20.7±3.5	57
	Magens Bay	57	27.2±1.2	71
Offshore	Eagle Ray	62	22.1±1.7	50
	Buck Island, St. Croix	61	21.8±1.9	50
	Castle	66	22.2±1.4	53
	Mutton Snapper FSA	65	23.9±1.7	51
	Seahorse Cottage Shoal	61	23.2±1.1	49
	South Capella	71	25.7±1.3	60
	South Water	63	24.1±1.3	55
	Flat Cay	74	24.6±1.0	65
	Meri Shoal	54	19.9±1.0	57
	Savana Island	78	26.8±1.6	69
	St. James	71	28.7±3.3	61
Mesophotic	Buck Island STX Deep	58	22.4±1.4	48
	Cane Bay Deep	61	21.3±1.9	47
	Lang Bank EEMP	63	24.1±2.3	57
	Lang Bank Red Hind FSA	65	23.9±2.3	51
	Salt River Deep	49	13.2±1.1	30
	College Shoal East	58	25.2±1.1	57
	Ginsburg's Fringe	24	17.5±0.7	-
	Grammanik Tiger FSA	65	27.7±2.0	72
	Hind Bank East FSA	69	29.3±2.1	64

Fish Abundance

Total fish abundances across nearshore, offshore, and mesophotic sites and years are shown in Fig. 26. As in previous, years total fish abundance was highly variable across sites and strata with no obvious patterns over time or space noted.

The sites with the highest overall fish abundance in 2018 were Black Point and Botany Bay. Black Point has high numbers of very small parrotfish and wrasse, raising the mean abundance of that site. Botany Bay is a site with high habitat diversity that supports many species in relatively high numbers. It is a reef that acts as juvenile habitat for grunts, which recruit there in very large schools. Other sites with relatively high fish abundance were Buck Island STT and Mutton Snapper. Sites with the lowest abundance included Salt River Deep and Ginsberg's Fringe. These mesophotic sites, dominated by agariciid corals, also had low species richness and biomass. Hurricanes of 2017 did not appear to affect the fish communities in a dramatic way. In general, mesophotic sites had a lower biomass than offshore or nearshore sites. Since mesophotic sites are not juvenile habit for many species, they lack the schools of smaller fishes like striped parrotfish. Wrasses are also notably less common on mesophotic reefs.

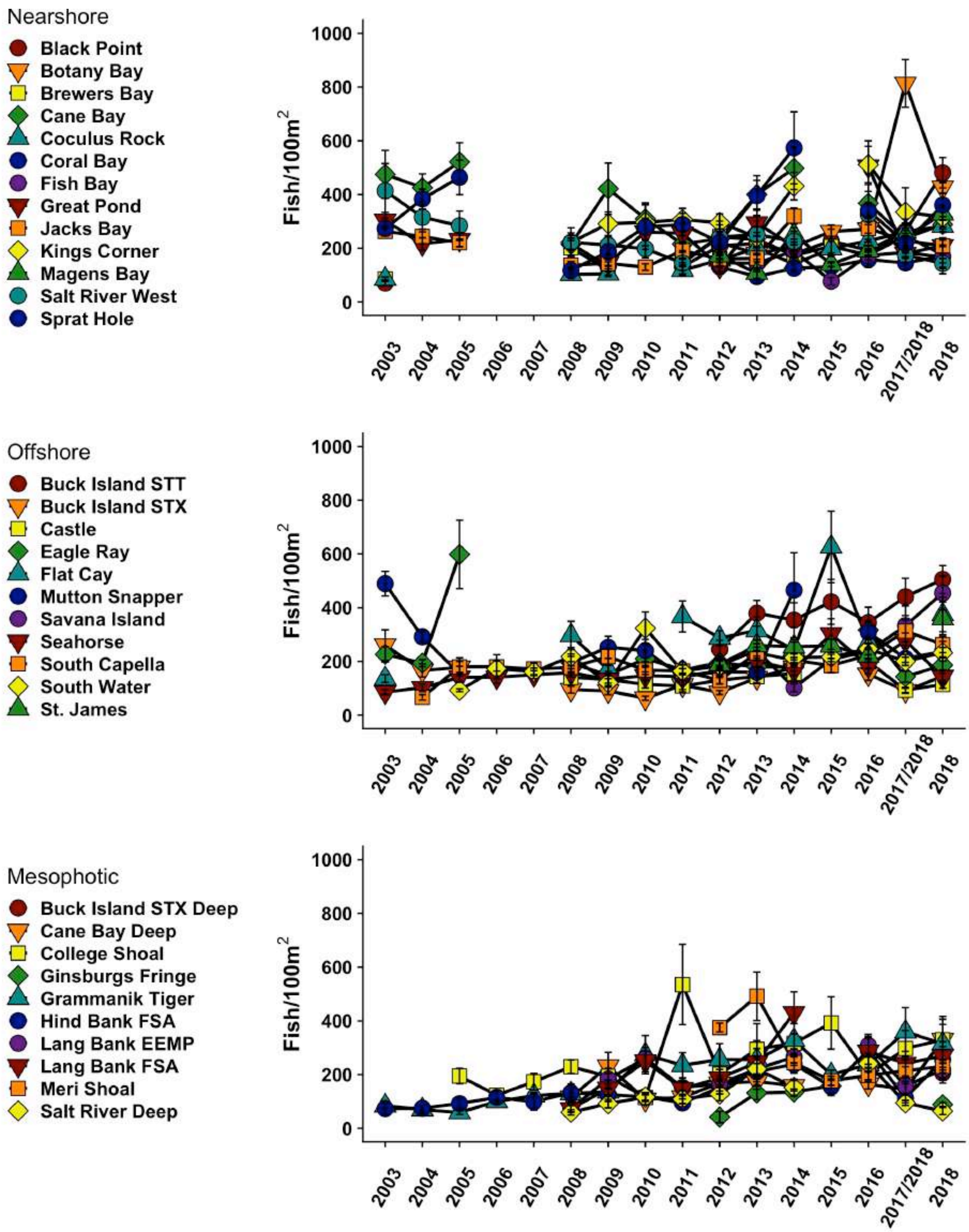

Figure 26. Fish abundance (±SE) across TCRMP monitoring sites over time. St. Croix sites are to the left and northern USVI to the right on the x-axis.

Fish Biomass

Total fish biomass for all sites and years is shown in Fig. 27. As with abundance, biomass is highly variable across strata, sites and years. No temporal pattern is obvious, and differences in time appear to be seasonal or natural variation. The biomass of fish at mesophotic sites off the northern USVI (Grammanik Bank FSA, Hind Bank FSA, and College Shoal) have had the highest biomass values throughout the TCRMP sampling series. These are protected reefs on the insular shelf edge, and fish spawning occurs on both the Grammanik Bank FSA and Hind Bank FSA. Although TCRMP sampling occurs outside of the spawning season, higher numbers of large fish may inhabit these sites due to their roles as aggregation areas. College Shoal does not host spawning events however, and biomass is also generally high on this site. In 2017 total biomass was higher on College Shoal than on any other site sampled. This was due primarily to large numbers of bar jacks, horseeye and black jacks (*C. lugubris*) that were present. Biomass on the mesophotic sites on St. Croix were low as in past years. Although now protected from fishing, Lang Bank does not seem to attract the large pelagics that are present on the southern edge of the northern USVI shelf. As with abundance, biomass appeared to be slightly lower on most St. Croix sites, possibly due to the passing of Hurricane Maria five months earlier. The St. Croix nearshore site, Kings Corner, generally has high biomass when compared to the other nearshore and offshore sites, however in 2018 biomass was very low. Northern USVI nearshore sites, Coral Bay, Magens Bay, Fish Bay and Botany Bay all had usual low fish biomass in 2017. These sites are along shorelines of developing residential areas and sustain high turbidity levels. They are also far from the deeper water near the shelf edge, and so these sites appear to support only juvenile fishes, and small species and at very low abundances.

Fish biomass at nine sites resampled after the two major hurricanes is shown in Fig 26. Like fish abundance, biomass dropped at most of the sites after the storms, but rebounded to pre-storm levels when sites were sampled six or seven months after the events.

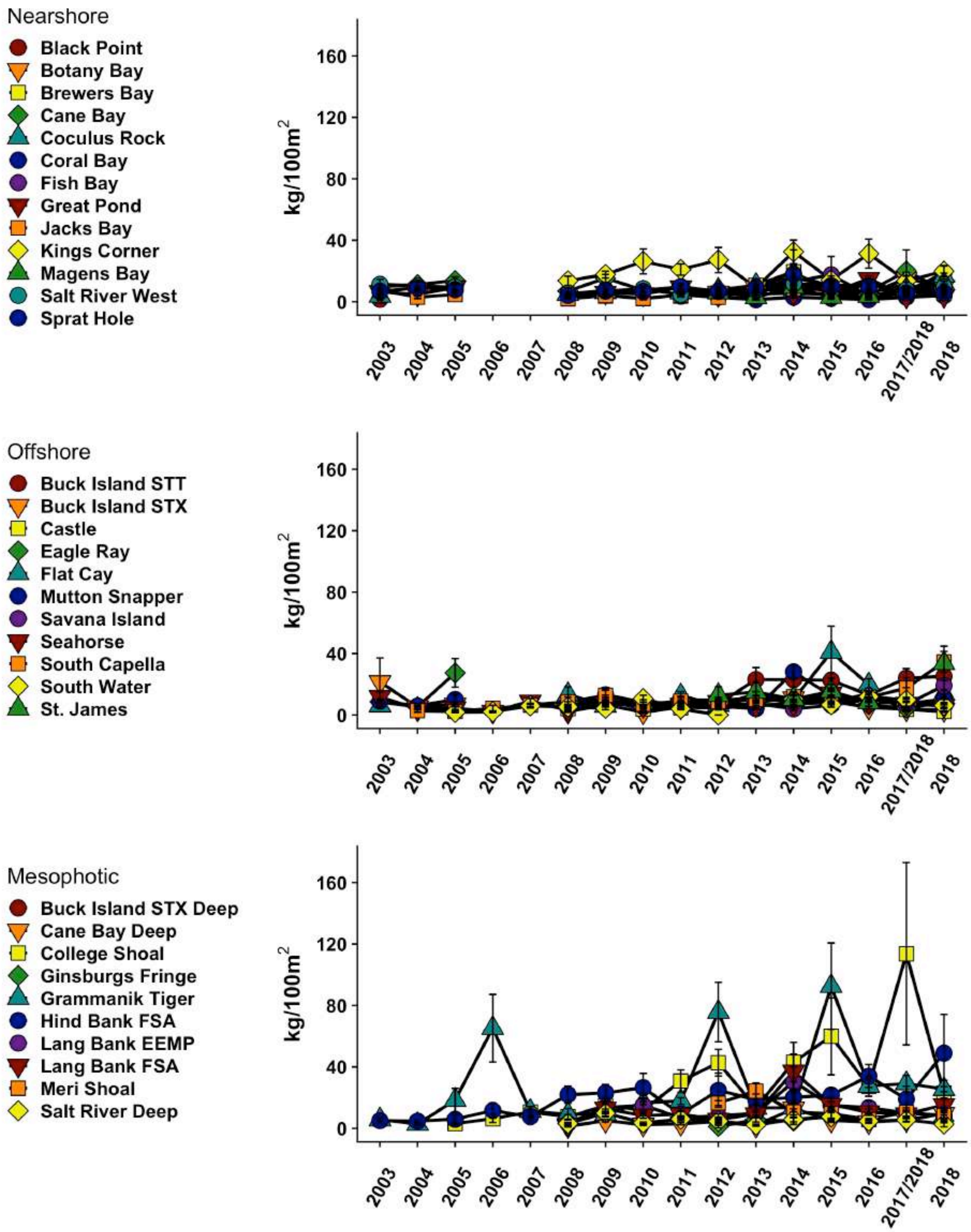

Figure 27. Mean fish biomass (±SE) across TCRMP monitoring sites over time. St. Croix sites are to the left and northern USVI to the right on the x-axis.

BLACK SPINED SEA URCHIN *DIADEMA ANTILLARUM*

The abundance of the black spined sea urchin *Diadema antillarum* shows tremendous site-to-site variability (Fig. 28). In general the shallowest sites, e.g., Great Pond, support the greatest abundance of *D. antillarum*. Trends are not presented here by year, as variability is generally low. At Coral Bay there is a high abundance of *Echinometra* spp. that has not been quantified. This species seems to be the dominant grazer and effectively removes most macroalgal cover, but also contributes apparently high bioerosion (gnawed coral bases). Future monitoring might consider targeted monitoring of these species at certain sites.

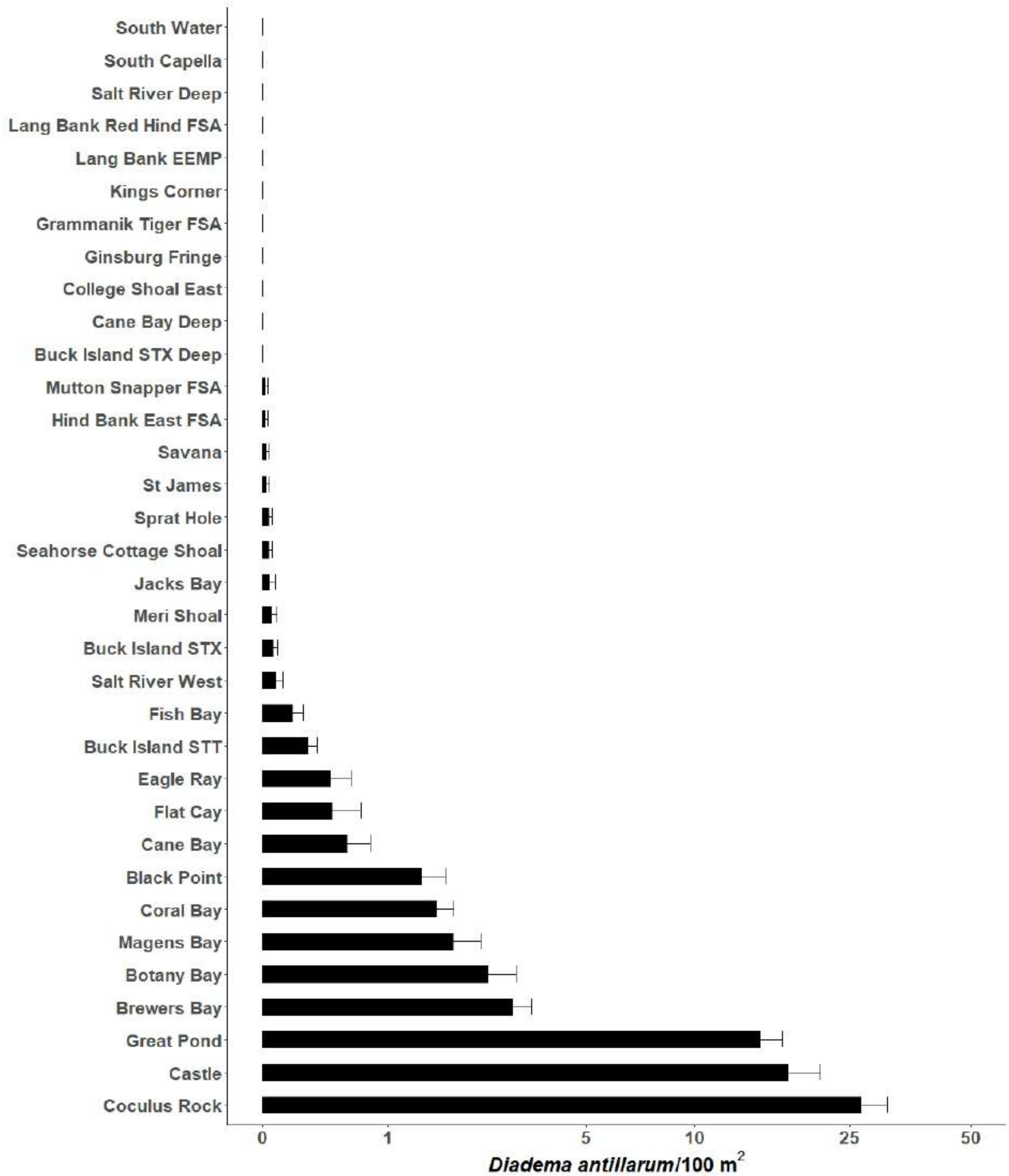

Figure 28. Average abundance (±SEM) of the black spiny sea urchin (*Diadema antillarum*) at TCRMP monitoring sites across all years. Note the log scale.