The United States Virgin Islands

TERRITORIAL CORAL REEF MONITORING PROGRAM

ANNUAL REPORT

2021

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A collaboration between:

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The Division of Coastal Zone Management, USVI Department of Planning and Natural Resources

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MISSION STATEMENT

Mission

OUR VISION

To provide critical information on the status and threats to all Virgin Islands coral reef ecosystems in order to increase management effectiveness and improve basic and applied coral reef research

OBJECTIVES

- **Monitor the status and trajectories of coral reefs across a majority of habitats and threats, including land-based sources of pollution & thermal stress**
- **Link changes in coral reef health with specific stressors, indicating specific management interventions most effective for preserving reefs**
- **Integrate assessments of understudied mesophotic coral reef ecosystems and threatened species in the USVI**
- **Provide data, outputs, and advice to stakeholders and create a nexus of information for reef research**

EXECUTIVE SUMMARY

Executive Summary

Coral reefs in the Caribbean are facing a dramatic decline and are at a crossroads. Management decisions made today will affect the goods and services that reefs provide for decades to come. The government of the United States Virgin Islands (USVI), in coordination with the NOAA Coral Reef Conservation Program and the University of the Virgin Islands, implemented the **Territorial Coral Reef Monitoring Program** (TCRMP). The TCRMP has established baseline conditions and temporal trends of coral reefs and fish populations and has identified threats that will influence the future reef health and development.

A major focus of the TCRMP is to provide information that can lead to more effective management strategies that balance the immediate needs of the Virgin Island's population with preservation and sustainability of coral reefs and the renewable goods and services they provide.

The intent of this report is to distill monitoring data into actionable information that can guide management decisions and inform the public and policy-makers about areas that need further effort. This executive summary presents information on threats to USVI reefs that require management intervention/action as well as positive signs that can inform our understanding of sustainability.

CORAL REEFS OF THE VIRGIN ISLANDS: MANGEMENT ACTIONS NEEDED

The TCRMP data has identified threats to USVI coral reef ecosystems that need increased management attention if reef corals are to persist in a condition that is equal to or better than current conditions.

Coral Reef Bleaching. High thermal stress caused by climate change is currently the greatest threat to USVI coral reef ecosystems. The 2005 coral bleaching event caused the largest loss of coral in the documented history of the USVI, with a 50% decline in coral cover in shallow waters less than 25m/85' deep (Smith et al. 2013b; Smith et al. 2016a). This event surpassed all known modern impacts from physical damage (storms and anchoring), ecosystem changes (fishing and disease), and pollution (terrestrial sediments and toxins). Thermal stress and bleaching conditions are becoming more frequent.

While local management actions cannot remove impacts to reefs from global warming, reefs that are otherwise less stressed by land-based sources of pollution, fishing, and/or physical damage are known to recover more quickly from bleaching. Hence, local management actions that promote seascape-wide coral reef health offer the best strategy for sustainable reefs. We can also identify areas that are naturally more resistant to thermal stress and offer these areas further protection, since they offer insurance against the worst possible future outcomes for USVI reefs.

Figure 1. Partially bleached and recovering colony of *Siderastrea siderea* at Flat Cay, St. Thomas (Nov. 12, 2005).

Overfishing. There are clear indications that reefs of the USVI are suffering the effects of overexploitation of reef resources, although there are also positive signs. The entire district of St. Croix has an extremely low abundance of commercially important grouper species, including the threatened Nassau grouper (Kadison et al. 2017). In St. John and St. Thomas, many common species have completely or nearly disappeared from nearshore waters in the last 30 years. For example, a study conducted by Rogers et al. (1982) on the southwest coast of St. Thomas during the airport runway expansion (1979-1981) found a variety of species that are no longer encountered or are rare, including the black, Nassau, tiger, and yellowfin groupers, as well as the federally protected parrotfish species blue, midnight, and rainbow. A study by Randall (1963) also found high relative abundances of groupers and threatened parrotfish on the south coast of St. John. Rebuilding these fish stocks will require comprehensive life-history information for target species, a willingness to find strategies to rebuild stocks, and partnerships between commercial and recreational fishers, community stakeholders and managers. Positive signs of an increasing recruiting and spawning population of Nassau groupers in the northern USVI and more occurences at TCRMP sites in St. Croix suggests that management actions, including no-take restrictions and protection of spawning aggregation sites, can have tangible, positive effects (Kadison et al. 2010). In addition, recent work under the Deep Coral Reef Monitoring Program, an extension of the spatially randomized National Coral Reef Monitoring Program in depths between 30-50 m (100-165 feet), is finding higher abundances of commercially important species that are rare in shallow waters. Viable populations still exist as a base for rebuilding stocks.

Land-Based Source of Pollution. The steep hillsides of the USVI are natural conduits for run-off during heavy rain events and in many instances there is little interception of materials before they reach the sea and impact coral reefs. When tropical soils are naturally disturbed or altered through human activity, they can erode and release fine-grained silt and clay particles. In the USVI, these fine-grained particles are quickly transported to coral reefs where they can block sunlight, directly smother corals, or increase the growth of organisms that compete with corals for space. There is evidence from the TCRMP that terrestrial sediments are having large negative impacts on nearshore coral reefs by increasing mortality of ecologically important corals (Henderson et al. in prep.).

CORAL REEFS OF THE VIRGIN ISLANDS: POSITIVE SIGNS

Despite the incredible declines in reef health witnessed since the inception of the TCRMP in 2001, there are many positive signs for the USVI that should be highlighted. These successes offer lessons that can be applied to troubled reefs and may indicate refuge areas where we might "double-down" on current management strategies.

Reef Refuges. The USVI is blessed, perhaps uniquely for the Caribbean, with extensive areas of deep bank and slope reefs that may be buffered from the direct impacts of local pressures (Smith et al. 2019a, b). The mesophotic (pronounced: me-zo-photik; meaning; "middle-light") reefs of the USVI are the best developed in the Caribbean from what is currently known. Mesophotic Coral Ecosystem (MCE) bank reefs with high populations of star corals (*Orbicella* spp.), which have recently been listed as threatened on the United States Endangered Species List (NOAA 2014), form extensive tracts on the south shelf of St. John and St. Thomas, from the British Virgin Islands to Vieques, Puerto Rico. Well-formed, but patchier mesophotic boulder coral reefs also form on the Lang Bank, St. Croix and the northern Puerto Rican Shelf. The lower MCE consists mostly of lettuce corals (primarily *Agaricia undata*) and form a semi-continuous ring on steep slopes and walls at depths between 50-70m. These reefs are isolated from some local impacts, but may be susceptible to global climate change (see below).

In federal waters some of these areas are wholly or partly protected from fishing of ecologically important species that help maintain reef health. These include the Red Hind Marine Conservation District (est. 1999), the Grammanik Bank Seasonally Closed

Area (est. 2005), and the Lang Bank Red Hind Seasonally Closed Area (est. 1993). However, extensively developed mesophotic reef in unprotected territorial waters also exist near the island of French Cap and Sail Rock, St. Thomas District. It is important that these areas are identified, their threats assessed, and they are incorporated into the territorial and federal management planning process.

Rebounding Fisheries Species. There are positive signs of recovery for certain fish species in some areas. At the Grammanik Bank multi-species fish spawning aggregation site there have been increasing numbers of Nassau grouper during annual spawning (Kadison et al. 2010; Jackson et al. 2014), with about 800 individuals seen in 2021-2022 (R. Nemeth, unpub. obs). A red hind aggregation in the Red Hind Marine Conservation District (MCD) has dramatically rebounded (Nemeth 2005). Red hind caught in the fishery on the south side of St. Thomas are more numerous and larger (D. Olsen, pers. comm.; DCRMP unpub. data). In 2015 there was a recruitment pulse of juvenile Nassau grouper to shallow nearshore environments of St. Thomas and St. John. In Brewers Bay, St. Thomas over 70 juvenile Nassau grouper were recorded (R. Nemeth, unpub. data), increasing evidence that the reproductive population is contributing to the recovery of the species. It is also the impression of the authors that stocks of grouper and snapper are increasing in the MCD, although TCRMP measurements are confounded to some degree by the rotating array of aggregating fishes. Other territorial and federal closed areas in St. Croix, St. John, and St. Thomas are more recently established and may not show effects for several years. Although uncommon, Nassau grouper have been recorded at various TCRMP sites around St. Croix since 2011, a positive sign. For species that are completely protected from fishing (Nassau grouper and blue, midnight, and rainbow parrotfish), educational campaigns for recreational and commercial fisherman are critical, as awareness of regulations among the USVI residents appears to be lacking (Authors, unpub. obs.).

Land-based source of pollution. While development of steep island slopes has continued despite current regulations intended to prevent sediments from entering nearshore waters, research has identified key targets for restoration and some effective habitat restoration best-management practices. Results from TCRMP research suggest that there are certain levels of silt-laden terrestrial run-off that are damaging to corals, providing a target for reductions of sediment in the marine environment (Henderson et al. submitted). Unpaved road segments have been implicated as the worst culprits in the production of sediment-laden run-off (Ramos-Scharrón and MacDonald 2007b) and this provides a clear target for where management can be most effectively applied. Restoration of watersheds has shown that implementation of best-management practices and control structures can be effective in reducing sedimentation. For example, the American Recovery and Reinvestment Act project "USVI Coastal Habitat Restoration Through Watershed Stabilization" showed promising results (Virgin Islands Resource Conservation and Development Council; P.I. M. Taylor).

This report presents results of the 21st year of monitoring on reefs surrounding St. Croix, St. John, and St. Thomas (years 2001-2021). Monitoring sites were distributed across the insular platform in depths from 5 to 63 m (16 – 220') to capture the diversity of reef types present in the Virgin Islands. Long-term data is presented from 34 sites. While not exhaustive, the TCRMP is generally representative of the geographic areas and variety of reef types in the USVI. Digital video and diver surveys were used to quantify benthic cover and coral health at 15 permanent sites surrounding the island of St. Croix and 19 permanent sites on the Puerto Rican Shelf surrounding the island of St. John and St. Thomas. In addition, at 33 of these sites, sea urchin density and fish community structure were evaluated.

EXECUTIVE SUMMARY

The TCRMP website describes the program and houses updated data

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

Data can also be requested directly from the research team by contacting Rosmin S. Ennis at rosmin.ennis@uvi.edu

Updated: The Impact of Stony Coral Tissue Loss Disease in the USVI

Figure 2. The visual impacts of the stony coral tissue loss disease on coral diversity and large and old colonies. Clockwise from left: Active SCTLD lesions and recently dead colonies on a range of coral species (Flat Cay, February 2019), A large colony of *Colpophyllia natans* with active SCTLD lesions (Black Point, October 2019), A large colony of *C. natans* with active SCTLD. The area of regrowth of the colony after the 2005 bleaching event is evident (Flat Cay, February, 2019). (credit: Marilyn E. Brandt)

Stony coral tissue loss disease (SCTLD) is a rapidly spreading disease in the USVI that is causing dramatic loss of coral cover and declines in species diversity (Fig. 2). SCTLD was first sighted in the US Virgin Islands in January 2019 at the Flat Cay TCRMP monitoring site (Fig. 3). The disease was identified by its characteristic stony coral species-specific susceptibility characteristics. By March 2019 the disease had spread to the north side of St. Thomas, by January 2020 had spread eastward to the St. Thomas East End Reserves, and by April 2021 had spread to numerous locations around St. Croix.

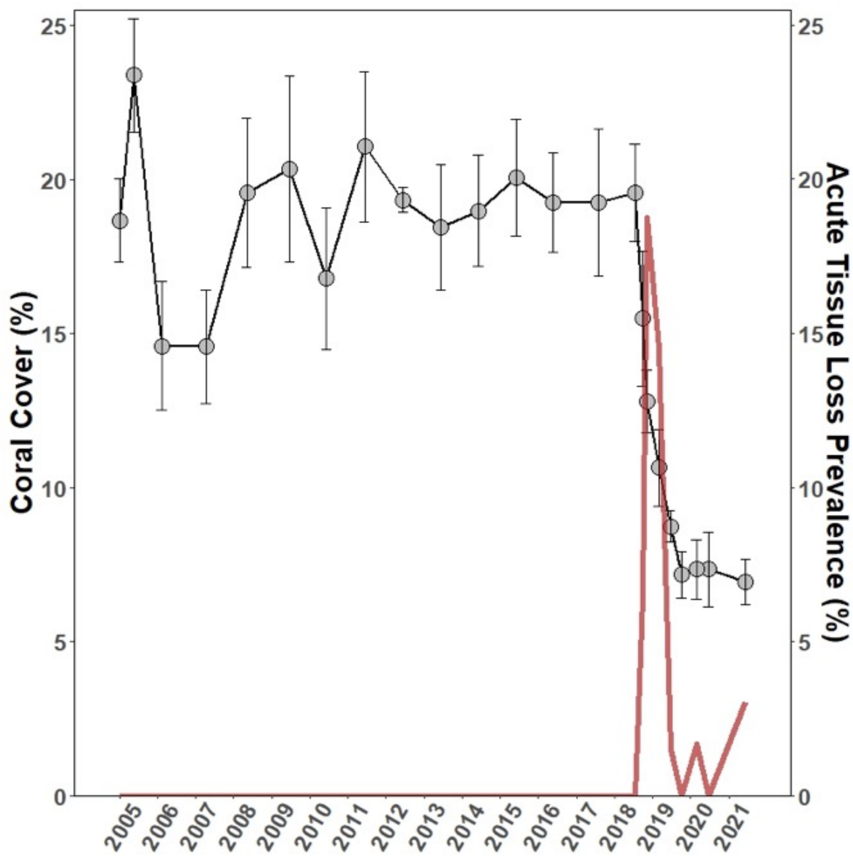

Figure 3. Mean coral cover (±SEM; gray circle) and prevalence of acute tissue loss (red line) at the Flat Cay monitoring location. Acute tissue loss includes any colony with an identified Stony Coral Tissue Loss Disease infection or rapid tissue loss with associated bleaching resulting in at least 10% recent mortality.

Numerous local teams at the University of the Virgin Islands, the Department Planning and Natural Resources, the National Park Service, the Caribbean Oceanic Restoration & Education (CORE) Foundation, and The Nature Conservancy have been coordinating monitoring, research, and response with national collaborators, such as the Florida

Department of Environmental Protection, Woods Hole Oceanographic Institution, University of Texas Arlington, Louisiana State University, Mote Marine Lab, Rice University, and the National Oceanic and Atmospheric Administration. Updated details concerning SCTLD can be found at the following website: https://www.vicoraldisease.org

Detailed impacts at TCRMP sites can be found in the site descriptions and are covered in more detail by Brandt et al. (2021). At the time of this report, all TCRMP locations, except Ginsburg Fringe, have become impacted and are experiencing losses in coral species and cover. The locations most adversely affected by SCTLD include Kings Corner, Seahorse Cottage Shoal, and Flat Cay with relative coral cover losses of 70%, 66%, and 63%, respectively, since before SCTLD arrived (Fig. 4).

Other accounts of SCTLD research in the territory can be found in Meiling et al. (2020), Costa et al. (2021), and Meiling et al. (2021). Studies based the biology of SCTLD from data taken in the US Virgin Islands can also be found in the literature cited.

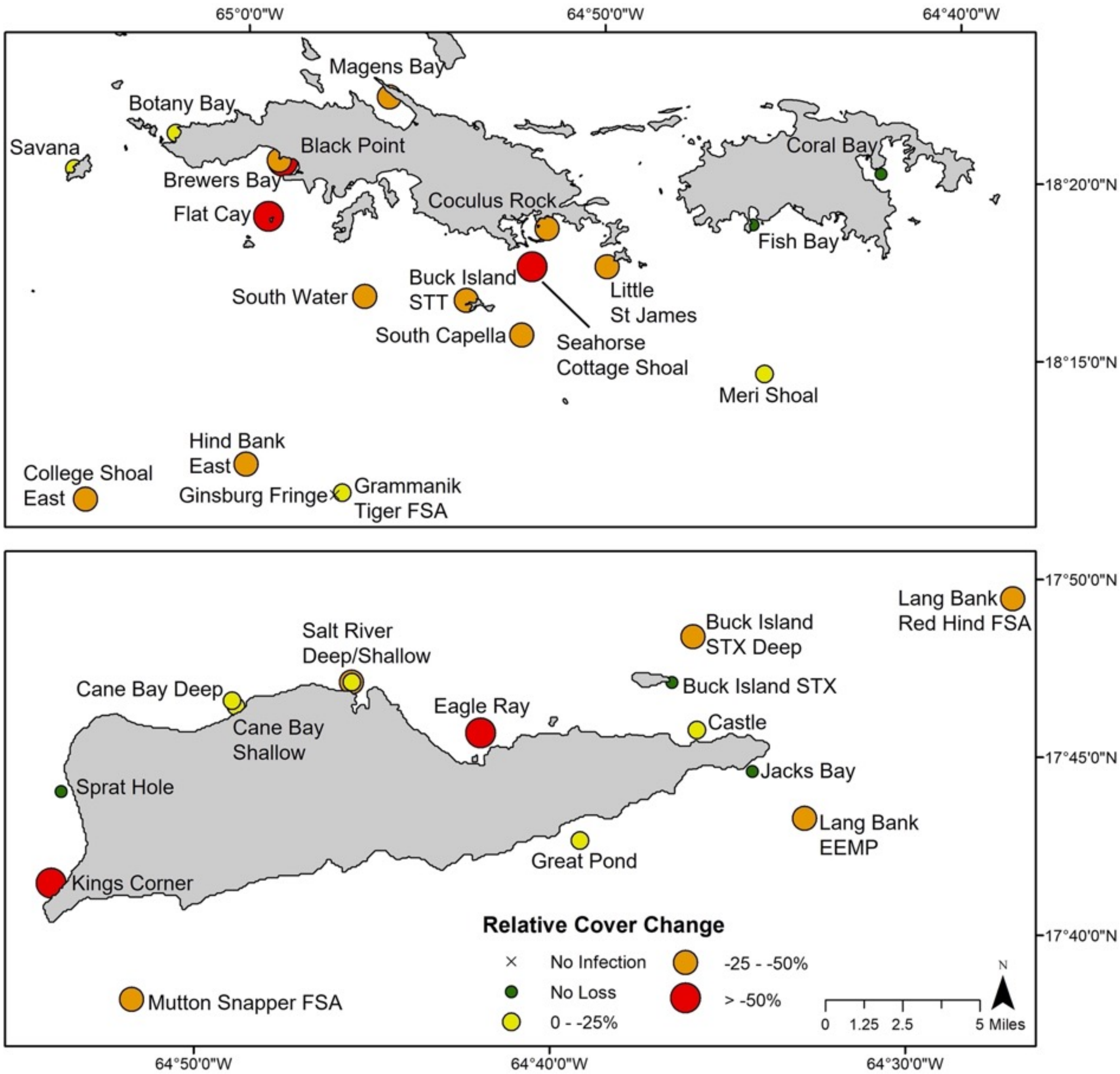

Figure 4. Relative coral cover loss at TCRMP locations in 2021 relative to pre-SCTLD monitoring.

Prior to the arrival of SCTLD, monitoring locations had relatively little change in coral cover since Hurricanes Irma and Maria (2017). Relative coral cover losses after initial SCTLD infection ranged from 8.2 – 13.6%, with differences likely driven by site-specific responses (Fig. 5). On average, locations infected in 2021 lost no coral cover and had a relative increase of 6.4% (Fig. 5). Monitoring locations with SCTLD infections present for multiple years have continued to lose coral cover, more than tripling their losses between the first and second year of infection; however, it appears that losses have stabilized at locations first infected in 2019 (Fig. 5). Coral species richness at locations infected in 2019 followed a similar pattern to coral cover with large relative losses in number of species following initial SCTLD infection (Fig. 6). However, monitoring locations infected in 2020 initially had relatively little change in species richness, with losses only occurring during the second year of infection (Fig. 6). Monitoring locations infected in 2021 had no change or a slight increase in coral species richness after infection (Fig. 6). Variability in coral species richness changes are likely due to the species composition specific to the monitoring locations that became infected during that year.

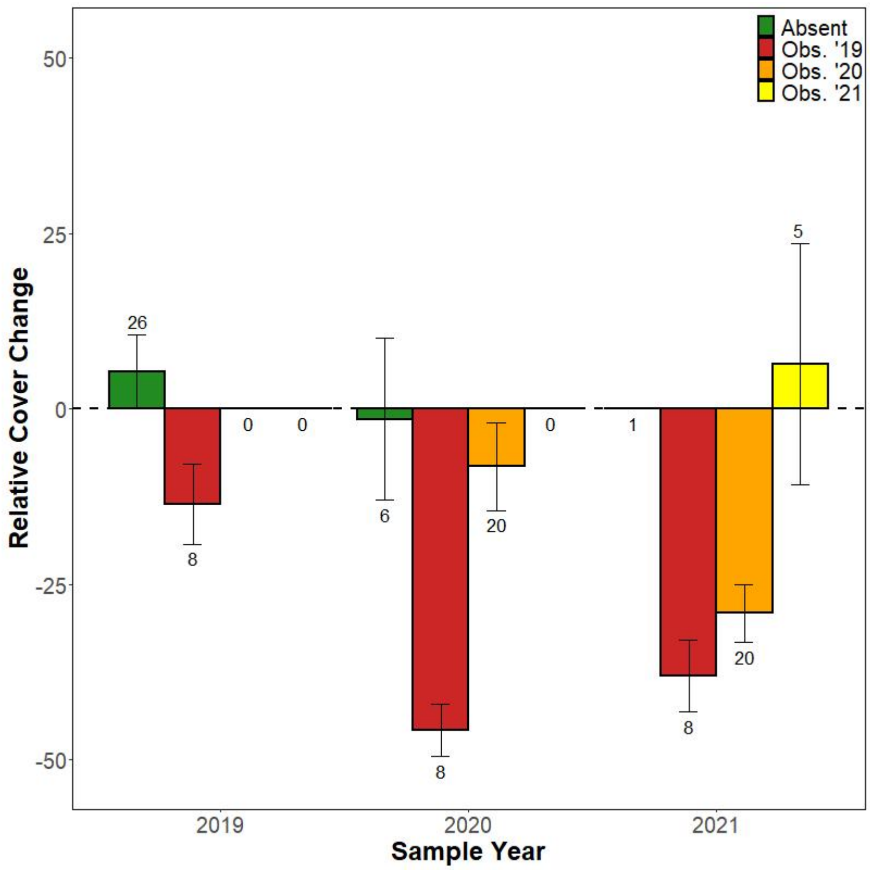

Figure 5. Relative coral cover change (±SEM) in the 2019, 2020, and 2021 monitoring periods relative to pre-SCTLD monitoring (2018). Value above bars represent the number of monitoring locations belonging to that category. "Absent" includes monitoring locations without observed SCTLD. "Obs. '19" refers to the following locations that were observed to have SCTLD in 2019: Black Point, Botany Bay, Brewers Bay, College Shoal East, Flat Cay, Savana, Seahorse Cottage Shoal, and South Water. "Obs. '20" refers to the following locations that were observed to have SCTLD in 2020: Buck Island STT, Buck Island STX Deep, Cane Bay, Cane Bay Deep, Coculus Rock, Coral Bay, Eagle Ray, Fish Bay, Grammanik Tiger FSA, Hind Bank East FSA, Kings Corner, Lang Bank EEMP, Lang Bank Red Hind FSA, Magens Bay, Meri Shoal, Mutton Snapper FSA, Salt River West, South Capella, Sprat Hole, and St. James. "Obs. '21" refers to the following locations that were observed to have SCTLD in 2021: Buck Island STX, Castle, Great Pond, Jacks Bay, and Salt River Deep.

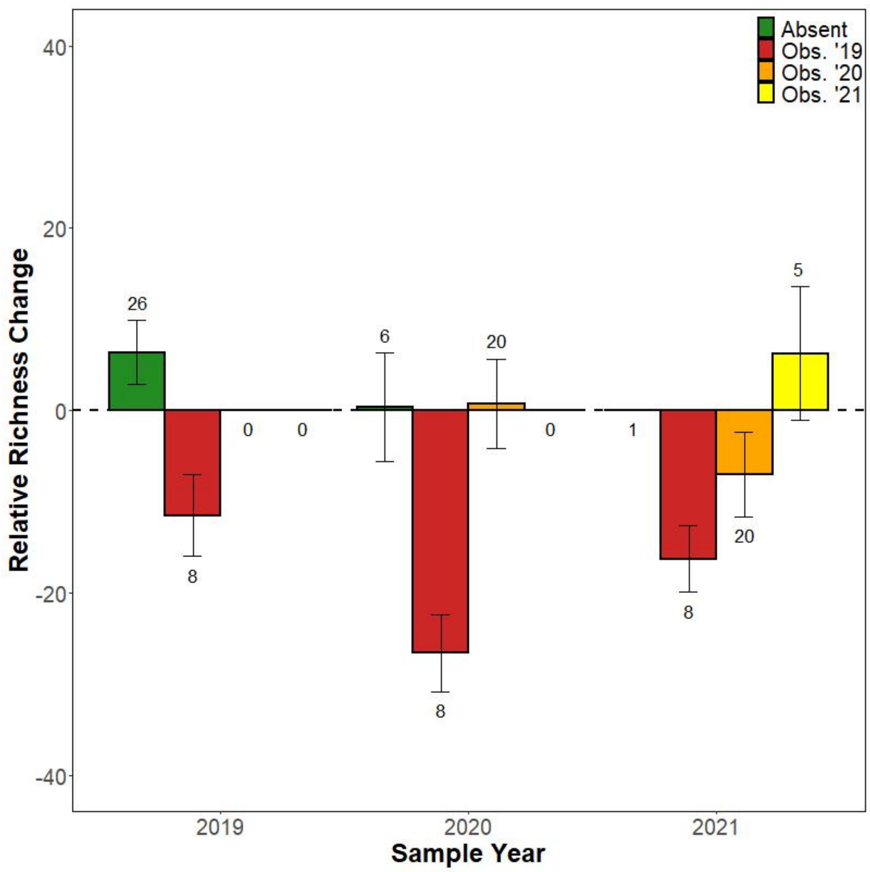

Figure 6. Relative coral species richness change (±SEM) in the 2019, 2020, and 2021 monitoring periods relative to pre-SCTLD monitoring (2018). Value above bars represent the number of monitoring locations belonging to that category. "Absent" includes monitoring locations without observed SCTLD. "Obs. '19" refers to the following locations that were observed to have SCTLD in 2019: Black Point, Botany Bay, Brewers Bay, College Shoal East, Flat Cay, Savana, Seahorse Cottage Shoal, and South Water. "Obs. '20" refers to the following locations that were observed to have SCTLD in 2020: Buck Island STT, Buck Island STX Deep, Cane Bay, Cane Bay Deep, Coculus Rock, Coral Bay, Eagle Ray, Fish Bay, Grammanik Tiger FSA, Hind Bank East FSA, Kings Corner, Lang Bank EEMP, Lang Bank Red Hind FSA, Magens Bay, Meri Shoal, Mutton Snapper FSA, Salt River West, South Capella, Sprat Hole, and St. James. "Obs. '21" refers to the following locations that were observed to have SCTLD in 2021: Buck Island STX, Castle, Great Pond, Jacks Bay, and Salt River Deep.

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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. 7). 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 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 can 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 aggressive and possibly introduced red alga *Ramicrusta textilis* has increased in abundance at many locations and is killing coral tissue through competitive overgrowth (Bramanti et al. 2017; Edmunds et al. 2019; Hollister et al. 2021; Williams and Edmunds 2021). Stony Coral Tissue Loss Disease (SCTLD), first observed at the Flat Cay monitoring location in early 2019, has led to widespread coral mortality and diversity losses throughout the Virgin Islands over the last several years (Brandt et al. 2021).

High thermal stress and coral bleaching events affected the northeastern Caribbean in 2005, 2010, 2012, and 2019 but these events had contrasting signatures in the United States Virgin Islands. Earlier 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; 50km satellite product) 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.

INTRODUCTION

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. 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 2001-2021 in St. Croix and from 2003-2021 in St. Thomas and St. John. For both islands, temporal changes from year to year in the conditions of the reef communities are assessed.

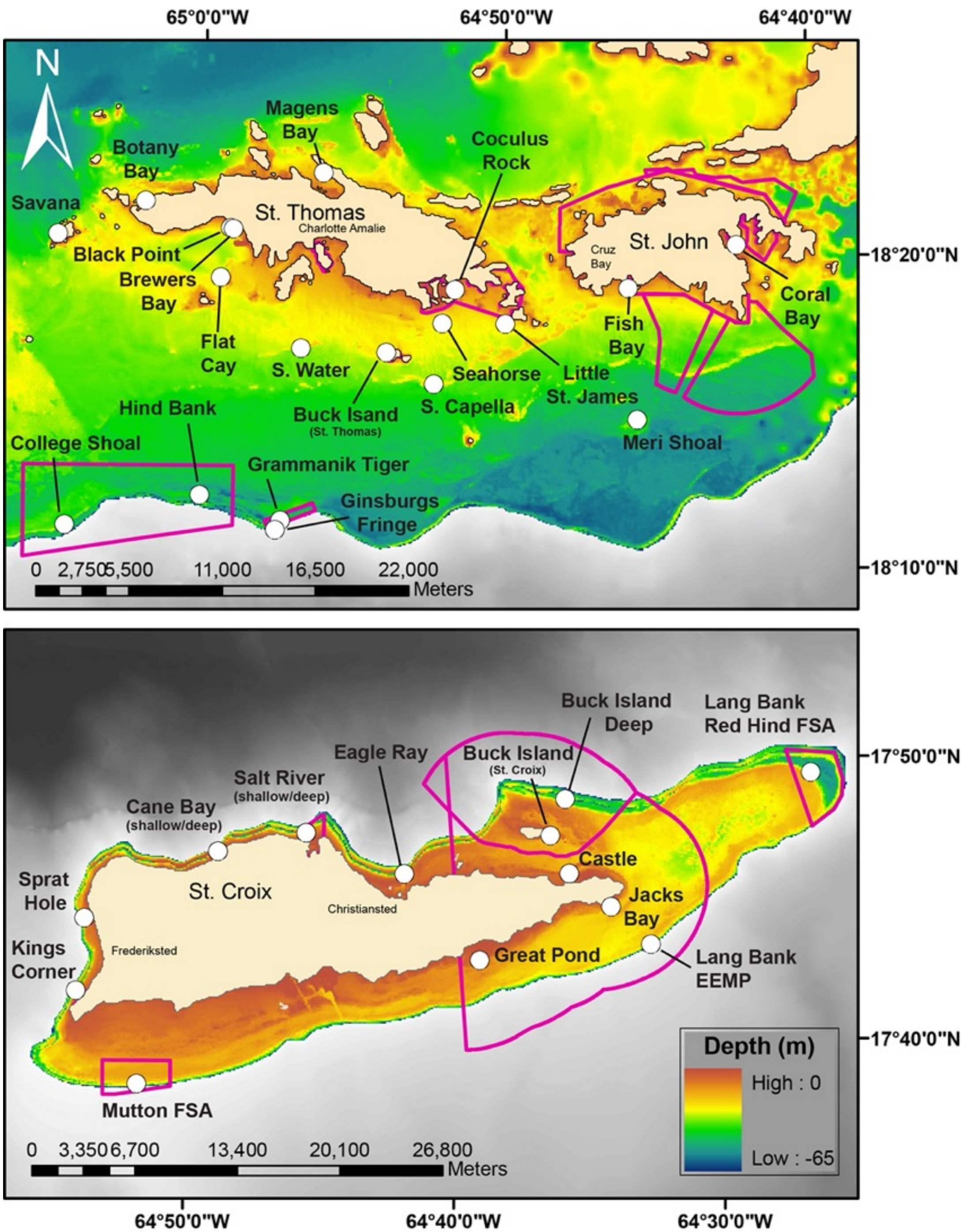

Figure 7. Locations of Territorial Coral Reef Monitoring Sites in the US Virgin Islands. Boundaries indicate federal and territorial marine protected areas.

Figure 8. A TCRMP research diver (S. Heidmann) on closed circuit rebreather records a roving fish survey at the Salt River Deep wall site, 30m/100' depth (April 9, 2020; credit: S. Meiling).

METHODS

Methods

BENTHIC ASSESSMENTS

The University of the Virgin Islands determined the benthic composition at 34 long-term monitoring sites between 2001 and 2021 (Fig. 8; Table 1; Table 2). All data is now available at the TCRMP website and updated annually after quality control:

https://sites.google.com/site/usvitcrmp/available-data

Around St. Croix the following 15 sites were assessed: Buck Island-St. Croix, Buck Island Deep, 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 and Buck Island Deep are within National Park Service/National Monument 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. 9). 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; prior to 2019) or R Studio (RStudio Team 2015; 2019 onward) 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 9. 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	7
	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
	Little St James	Offshore-Shallow	18.29459	-64.83238	17

Table 2. TCRMP site sampling date and type of sampling. Multiple dates indicate fish and benthic surveys were not completed on the same day and the date of fish survey is listed first. Surveys not completed are indicated by (-) while completed surveys are indicated by (X).

Island	Site	Sample Date	Benthic	Health	Fish + Urchin
St. Croix***	Buck Island STX	11/11/21	X	X	X
	Buck Island STX Deep	11/15/21	X	X	X
	Cane Bay	11/16/21	X	X	X
	Cane Bay Deep	11/16/21	X	X	X
	Castle	11/14/21	X	X	X
	Eagle Ray	11/15/21	X	X	X
	Great Pond	11/14/21	X	X	X
	Jacks Bay	11/11/21	X	X	X
	Kings Corner	11/12/21	X	X	X
	Lang Bank EEMP	11/14/21	X	X	X
	Lang Bank Red Hind FSA	11/22/21	X	X	X
	Mutton Snapper FSA	11/12/21	X	X	X
	Salt River Deep	11/13/21	X	X	X
	Salt River West	11/13/21	X	X	X
	Sprat Hole	11/12/21	X	X	X
St. John	Coral Bay	10/19/21	X	X	X
	Fish Bay	10/19/21	X	X	X
	Meri Shoal	10/13/21, 11/9/21	X	X	X
St. Thomas	Black Point	10/4/21, 10/22/21	X	X	X
	Botany Bay	10/15/21, 10/18/21	X	X	X
	Brewers Bay	10/4/21, 10/22/21	X	X	X
	Buck Island STT	10/8/21, 11/5/21	X	X	X
	Coculus Rock	11/24/21, 10/19/21	X	X	X
	College Shoal East	10/14/21, 12/3/21	X	X	X
	Flat Cay	10/6/21, 10/22/21	X	X	X
	Ginsburgs Fringe	-	-	-	-
	Grammanik Tiger FSA	10/14/21, 11/29/21	X	X	X
	Hind Bank East FSA	10/12/21, 12/3/21	X	X	X
	Magens Bay	10/15/21, 10/18/21	X	X	X
	Savana	10/18/21	X	X	X
	Seahorse Cottage Shoal	11/30/21, 11/5/21	X	X	X
	South Capella	10/8/21, 11/4/21	X	X	X
	South Water	10/6/21, 11/4/21	X	X	X
	Little St James	11/24/21, 11/4/21	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 abundance, size and diversity were recorded along transects and in roving diver surveys. There have been changes in protocol through the years to make initial methods more comparable to other regional sampling programs, to increase detection of rare, commercially important species, and to streamline logistics. Fish species identity, abundance, and size were assessed along at each site conducted over 15 minutes per transect. Prior to 2009 belt transect were 30x2m in size. Starting in sampling year 2009 belt transects were enlarged to 25m x 4m, increasing the survey area however from 2009 through 2011, two surveys were done per replicate, the first counting all non-site attached fish and the return survey along the transect line surveying smaller, site attached fish. In 2012, transects were again modified, and single surveys (25m x4m) were conducted per transect replicate, counting both site attached and non-site attached fish on a single survey. This change brought TCRMP assessment methodologies in line with NOAA Biogeography Branch methods (Menza et al. 2006; Friedlander et al. 2013). Before 2019 ten transects were conducted at each site. A retrospective analysis of running means of fish abundance showed that nine transects were sufficient to capture within-site variability, and so the number of transects was reduced to nine starting in 2019, which greatly assisted with field logistics.

All transects were begun at a random location on the site and were swum in a random direction. Fish were sized by fork length recorded into size-bin categories (cm): 1-5. 6-10, 11-20, 21-30, 31-40, 40+. This sampling strategy is not optimized for noting the presence and abundance of small (<5 cm) gobies and blennies and the species were inconsistently recorded.

Roving diver survey (RDS) were also conducted to estimate site fish diversity. Prior to 2016 RDS surveys in water depths less than 25m were 30 minutes in duration and, because of restrictions in dive length, were 15 minutes in deeper water. Retrospective analysis of species accumulation curves showed that almost all diversity was captured in the first 15 minutes of the survey. Therefore, 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. Data presented in individual site summaries represents data collected for the current report period only.

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.

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).

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. 10).

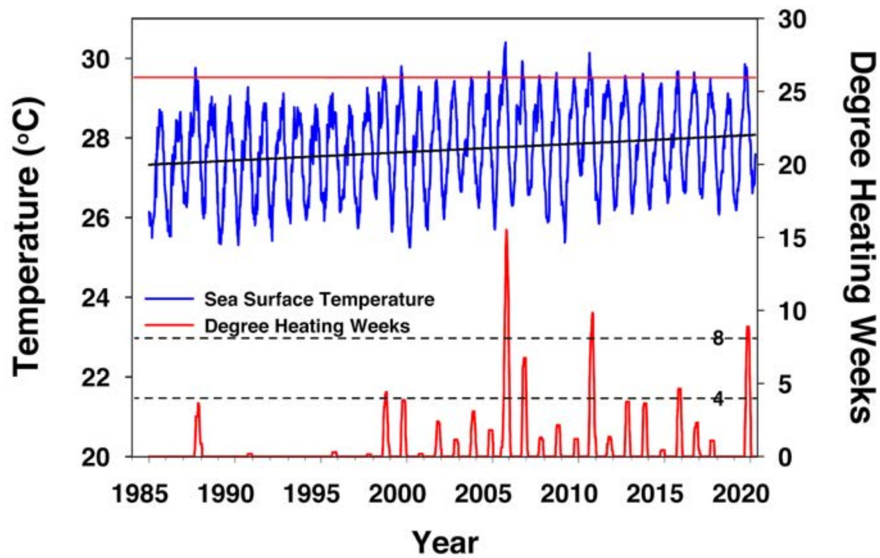

Figure 10. Sea surface temperatures and coral degree heating weeks of the US Virgin Islands from 1984 – 2020.

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 April 7, 2020 by Doug Wilson Ph.D..

TCRMP MONITORING SUMMARY

BENTHIC COMMUNITIES AND CORAL REEF HEALTH

Benthic cover was monitored at 33 monitoring sites and coral health was monitored at 33 sites in 2021. 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, 2012, and 2019 coral bleaching events (Fig. 14). 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. Most sites had apparently level coral cover with recovery potentially inhibited by disease and increased interactions with other organisms. However, prior to impacts of SCTLD, slow and irregular upward trajectories were 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.

In December 2018 stony coral tissue loss disease (SCTLD) was first sited at the Flat Cay monitoring site. It has now spread to impact nearly all reefs in the TCRMP, which is evident in many of the coral cover traces for individual sites.

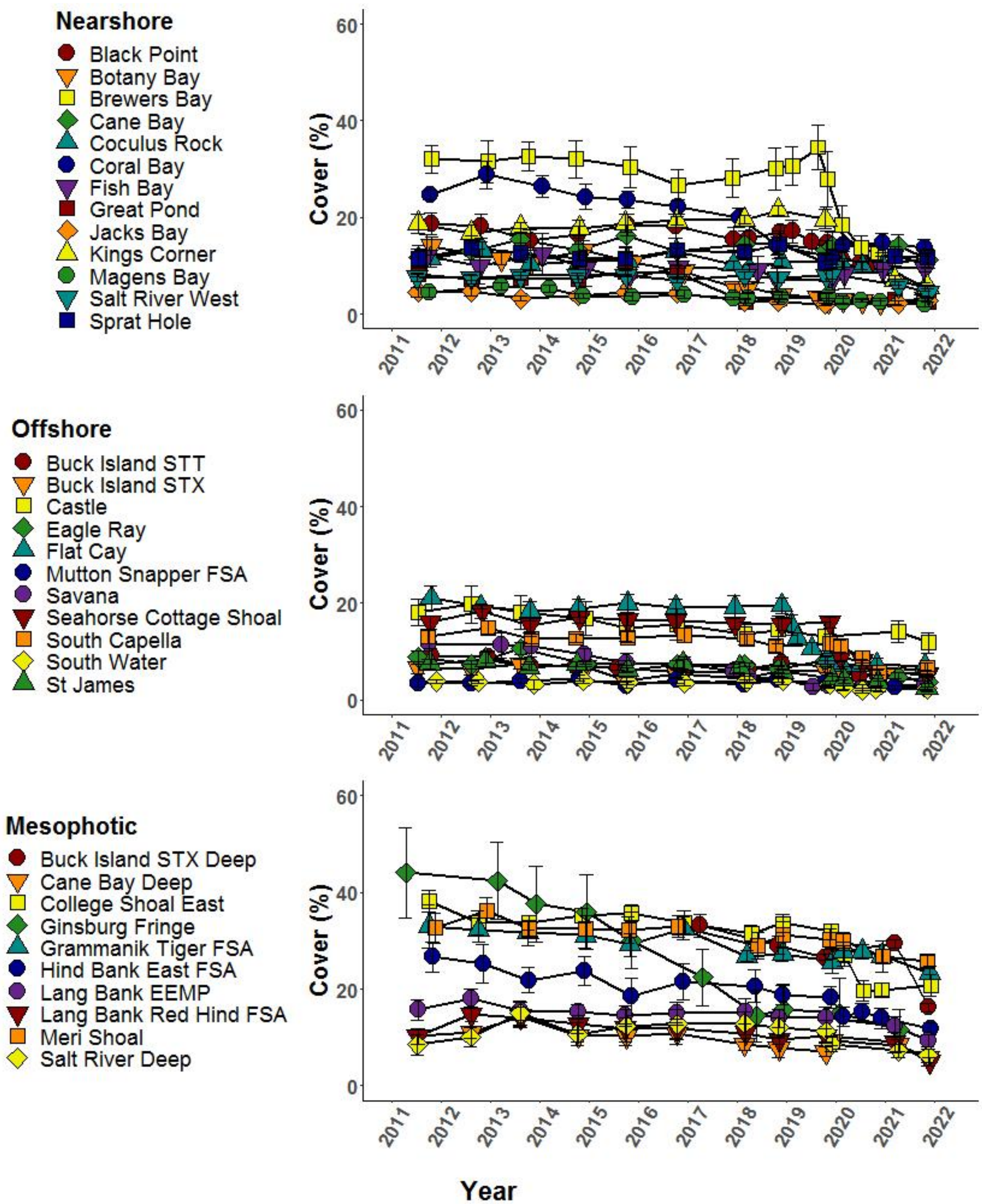

Figure 11. Coral cover (±SE) across TCRMP monitoring sites from 2011 – 2021.

Epilithic Algal Community Cover

Algae show the highest inter-annual variability of any group of benthic organisms (Fig. 15). 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.

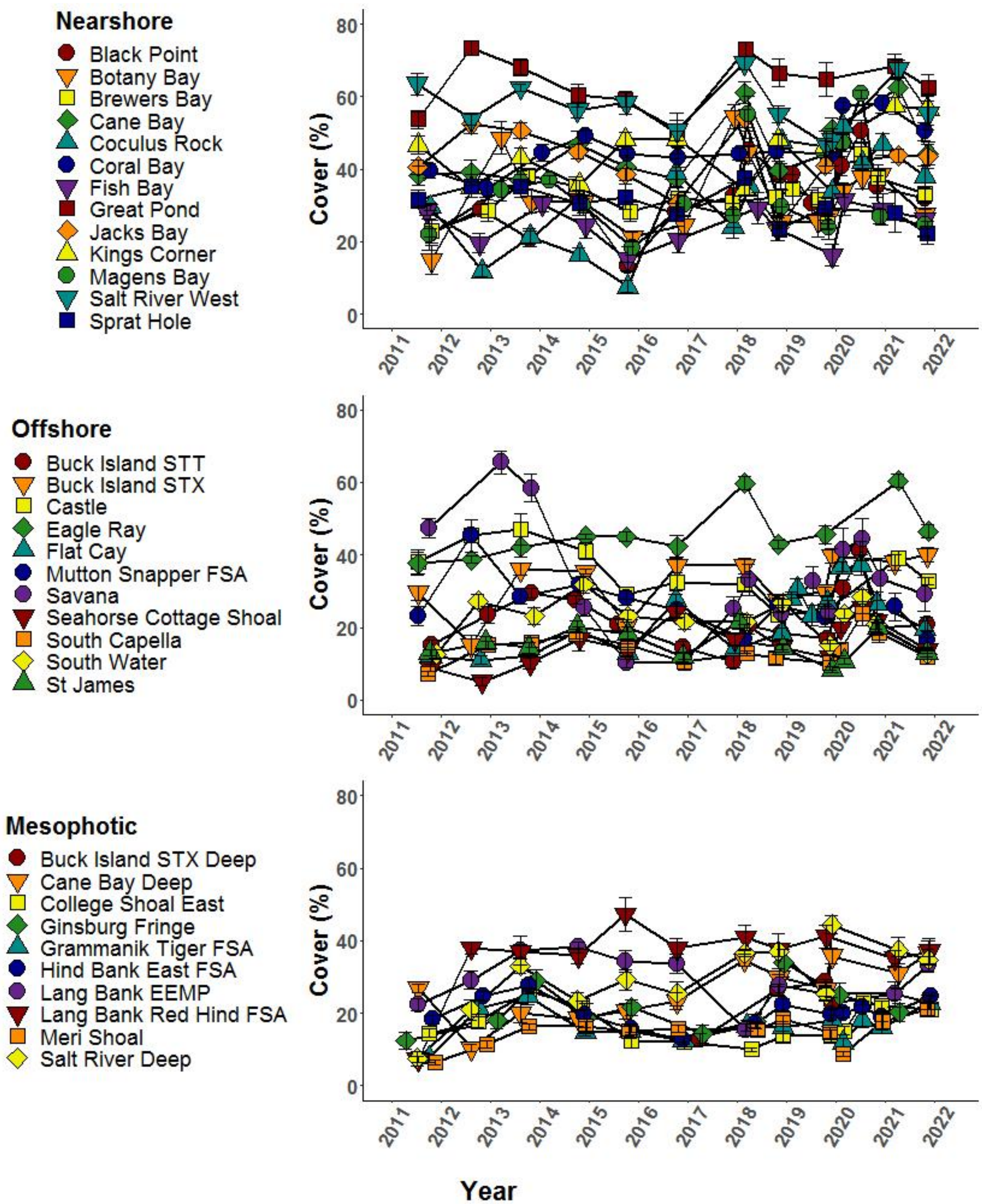

Figure 12. Epilithic Algal Community cover (±SE) across TCRMP monitoring sites from 2011 – 2021

Macroalgal Cover

Macroalgae have been increasing at many reefs, particularly after the 2005 bleaching event (Fig. 16). 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.

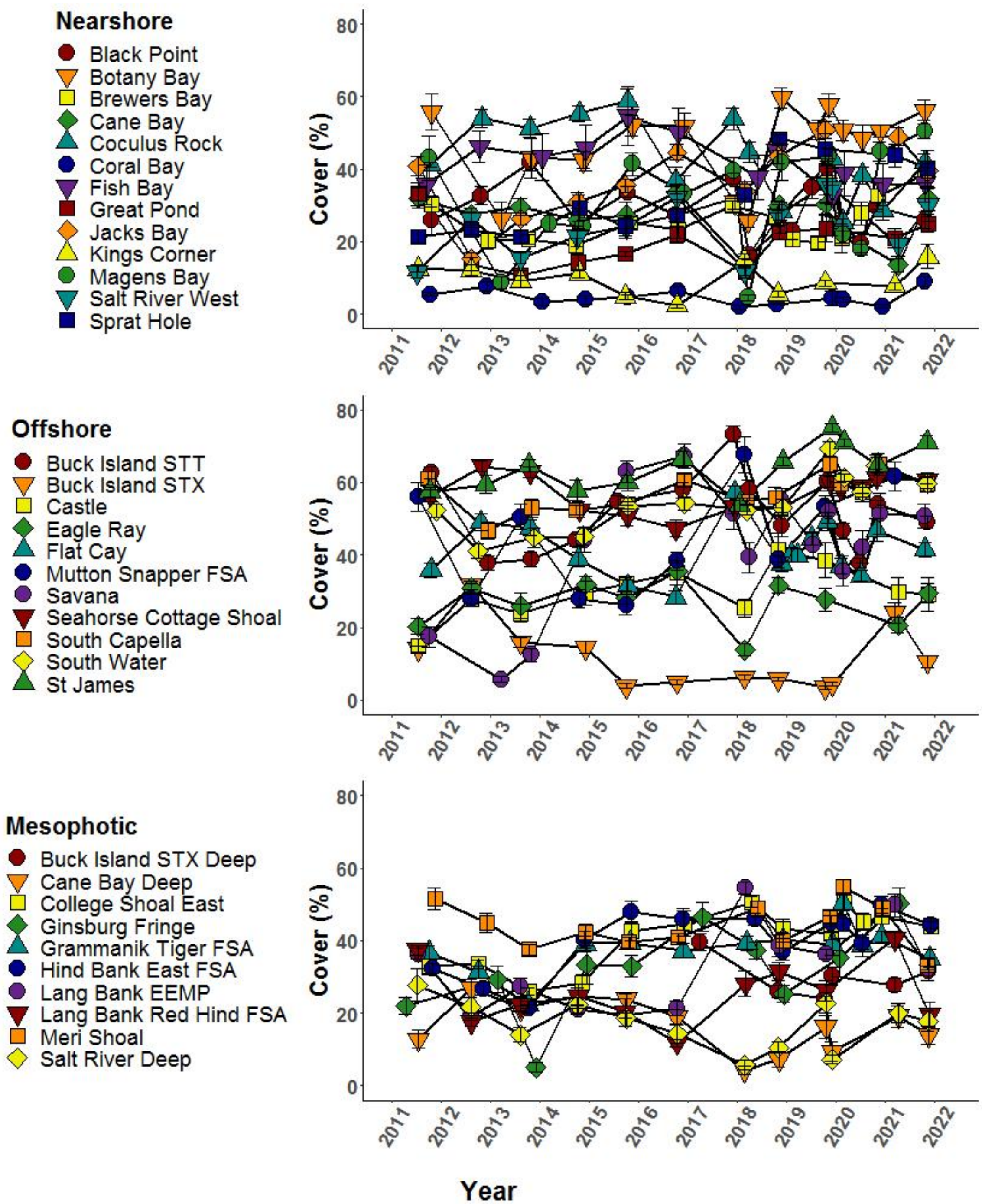

Figure 13. Macroalgae cover (±SE) across TCRMP monitoring sites from 2011 – 2021.

Filamentous Cyanobacteria

Filamentous cyanobacteria cover has been increasing at many sites in the TCRMP since the 2005 coral bleaching event (Fig. 17). 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.

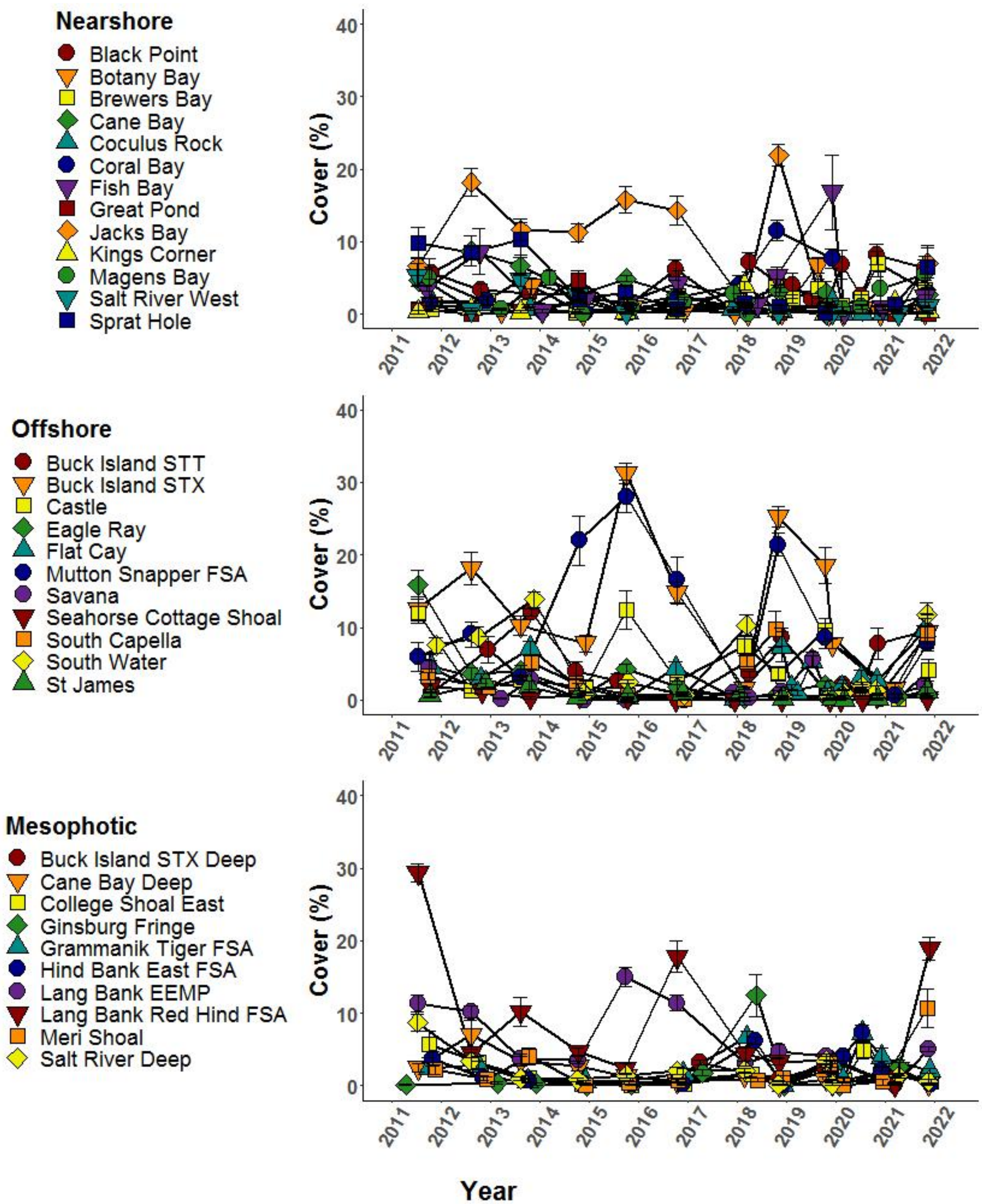

Figure 14. Filamentous cyanobacteria cover (±SE) across TCRMP monitoring sites from 2011 – 2021.

Gorgonian and Antipatharian Cover

The cover of gorgonians and antipatharians has been constant to increasing at most monitoring sites throughout the years of monitoring (Fig. 18). 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 makes them less detectable in planar imagery. At Coral Bay and Fish Bay on the south side of St. John the cover of gorgonians has been increasing through the monitoring time series. This has also been shown by a separate research group for Lameshur Bay, in between these TCRMP sites on south St. John (Tsounis and Edmunds 2017). Magens Bay also showed increasing gorgonian cover, but this has reversed somewhat in later monitoring years. 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).

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).

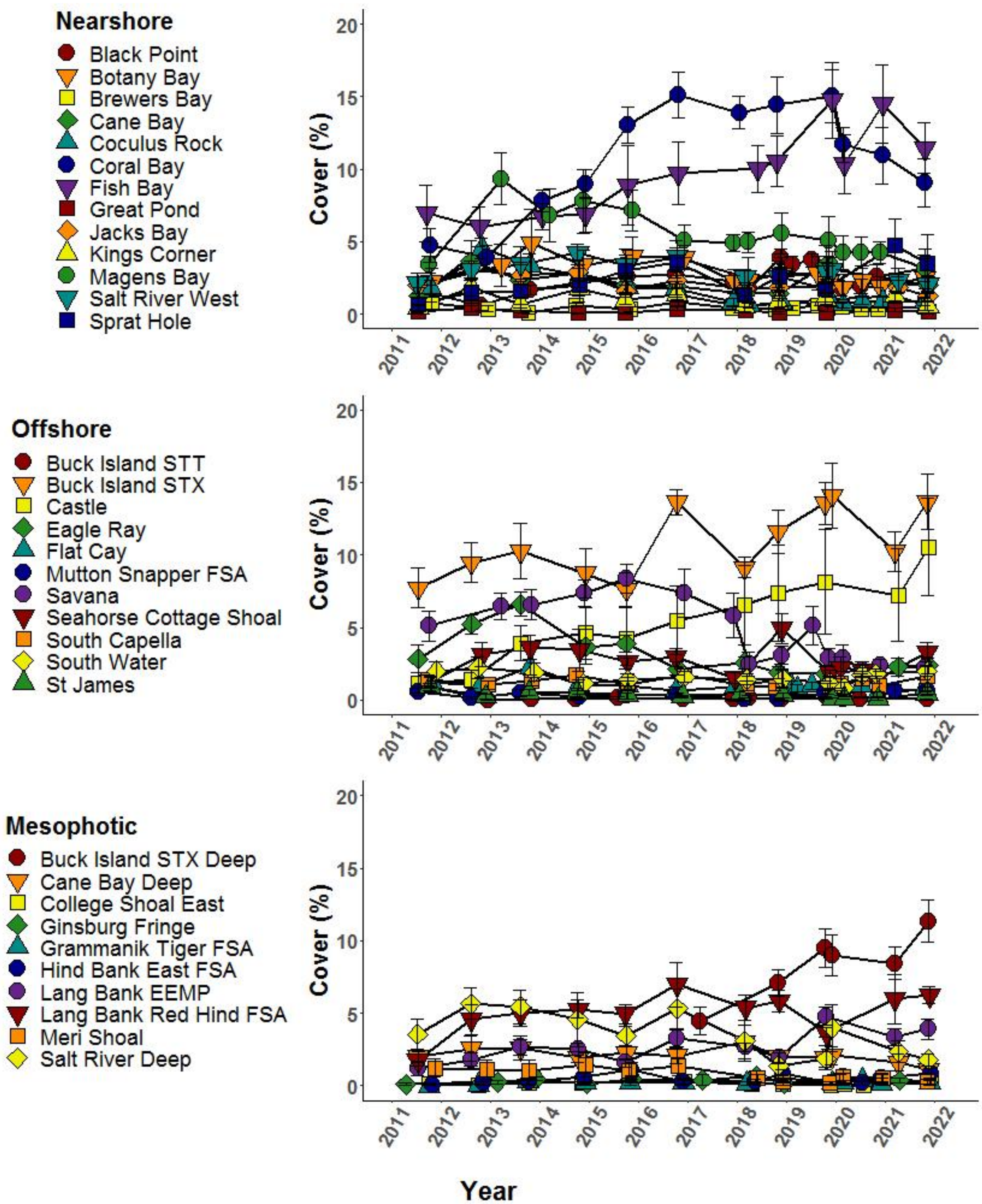

Figure 15. Gorgonian and Antipatharian cover (±SE) across TCRMP monitoring sites from 2011 – 2021.

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. 19). 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 (Gochfeld et al. 2020).

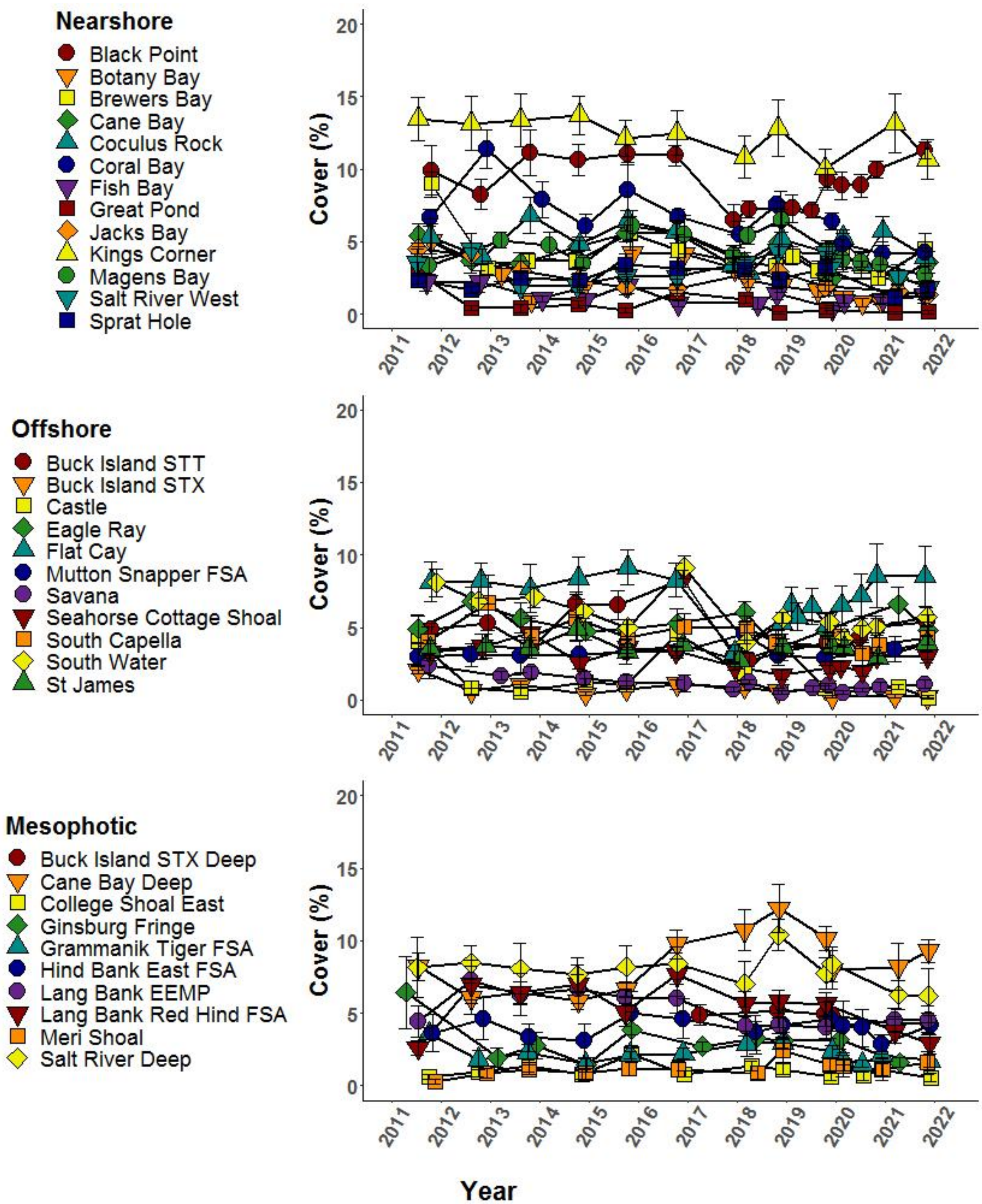

Figure 16. Sponge cover (±SE) across TCRMP monitoring sites from 2011 – 2021.

TCRMP MONITORING SUMMARY

FISH COMMUNITIES

Northern USVI: The fish transect surveys conducted in 2021 on the 18 sites in the northern USVI's represented 155 species and 43 families. A total of 38,438 individuals were recorded with a mean abundance across all sites of 237.27 ± 42.04 $100m^{-2}$. The calculated biomass of fish surveyed across all sites totaled 2290.10 kg with a mean of 14.16 ± 6.50 kg $100m^{-2}$.

On the seven nearshore sites in the northern USVI, a total of 14,133 fish were counted (mean= 224.33 ± 44.71 fish), representing 107 species and 34 families. The total biomass of fish recorded on nearshore reefs in the northern USVI was 459.3 kg (mean 7.29 ± 2.19 kg $100m^{-2}$). On seven offshore sites of the northern USVI both diversity and abundance were higher than on nearshore reefs; 108 species representing 34 families were recorded and a total of 14,842 fish were counted (mean=235.59 ± 35.60 fish $100m^{-2}$). The total fish biomass estimation on the offshore sites was 772.29 kg (mean = 12.26 ± 4.64 kg $100m^{-2}$). On mesophotic reefs diversity was lower; 115 species and 33 families were recorded. This represented 9463 individuals (mean = 262.86± 47.29 fish $100m^{-2}$) and a total fish biomass of 1058 kg (mean = 29.40 ± 10.52 kg $100m^{-2}$). Although mean fish biomass was much higher across offshore and mesophotic sites, variance was also very high. This represented the patchy occurrence of schools of larger reef associated and pelagic fishes on those sites.

Several ubiquitous species were found on all sites the northern USVI including all three surgeonfishes: the blue tang (*Acanthurus coeruleus*), ocean surgeon (*A. bahianus*), and doctorfish (*A. chirurgus*); two wrasses: bluehead wrasse (*Thalassoma bifasciatum*) and yellowhead wrasse (*Halichoeres garnoti*); four parrotfishes: striped parrotfish (*Scarus iserti*), redband parrotfish (*Sparisoma aurofrenatum*), stoplight parrotfish (*Sparisoma viride*) and princess parrotfish (*Scarus taeniopterus*); the French grunt (*Haemulon flavolineatum*), long spined squirrelfish (*Holocentrus rufus*), yellowtail snapper *(Ocyurus chrysurus*), graysby (*Cephalopholis cruentata*) and four eyed butterflyfish (*Chaetodon capistratus*).

The most common species numerically found on near shore sites in the northern VI's included the striped parrotfish (*Scarus iserti*) 24% and bluehead wrasse (*T. bifasciatum*) 20%. Blue chromis (*Chromis cyanae*), three spot damselfish (*Stegastes planifrons*) and blue tang (*A. coeruleus*) were the next most common fish. These five species (three of which are herbivores) represented over 59% of all fish recorded on nearshore sites. Species contributing the most biomass on nearshore sites were primarily herbivores as well and included: stoplight parrotfish (*S. viride*) 10%, blue tang (*A. coeruleus*) 9%, French grunts (*H. flavolineatum*) 9%, ocean surgeonfish (*A. bahianus*) 6%, striped parrotfish (*S. iserti*) 6%, and redband parrotfish (*S. aurolineatum*) 6%.

The most common fish numerically on the offshore sites in the northern VI's included: blue chromis (*C. cyanae*) 13%, bluehead wrasse (*T. bifasciatum*) 12%, bicolor damselfish (*S. partitus*) 12%, creole wrasse (*Clepticus parrae*) 8%, and striped parrotfish (*S. iserti*) 7%. These species together represented 53% of all individuals recorded. The species contributing the highest biomass to fish communities on the offshore sites included: yellowtail snapper *(O. chrysurus*) 11%, creole wrasse (*C. parrae*) 10%, blue striped grunts (*Haemulon sciurus*) 6%, and Caribbean reef sharks (*Carcharhinus perezii*) 6%. Caribbean reef sharks were observed in transects on two offshore sites, Buck Island STT and South Water.

The most common species numerically on the mesophotic sites in the northern VI's were creole wrasse (*Clepticus parrae*) 23%, blue chromis (*C. cyanae*) 16%, and bicolor damselfish (*S. partitus*) 7%. These three planktivorous species represented over 45% of all fish recorded on mesophotic sites. Species contributing the most biomass to the mesophotic sites in the northern USVI included five pelagic species: ocean triggerfish (*Canthidermis sufflamen*) 16%, blue runners (*C. crysos*) 9%, horseeye jacks (*C. latus*) 9%, creole wrasse (*C. parrae*) 6%, and bar jacks (*C. ruber*) 5%.

On the 15 sites surveyed off St. Croix, an additional 30,267 fish representing 130 species and 44 families were recorded. The mean fish abundance across all sites off St. Croix was 174.69 ± 38.34 fish 100m^{-2}. The calculated total biomass across all sites was 961.66kg with a mean biomass of 7.12 ± 2.39 kg 100m^{-2}. This is approximately half of the mean biomass per transect of fish surveyed on the northern VI sites.

The sites across St. Croix showed much less variability in terms of fish abundance and biomass across depth strata thon those in St. Thomas. Nearshore sites in St. Croix, had a total of 8373 fish counted (mean= 174.44 ± 33.24 fish 100m^{-2}), representing 95 species and 30 families. The calculated total biomass of fish recorded on nearshore reefs in St. Croix was 317.73 kg (mean 5.88 ± 1.96 kg 100m^{-2}). On four offshore sites a total of 5763 fish were counted (mean=180.09 ± 41.76 fish 100m^{-2}) representing 93 species and 30 families. The total fish biomass estimation on the offshore sites was 303.64 kg (mean = 7.12 ± 2.39 kg 100m^{-2}). On mesophotic reefs 106 species in 32 families were recorded. This represented 7029 fish (mean = 175.73± 39.53 fish 100m^{-2}) and a total fish biomass of 340.29 kg (mean = 7.56 ± 2.48 kg 100m^{-2}). High fish abundance and biomass were not seen on mesophotic reefs in St. Croix as in the northern USVI, and variance was not as high.

The most common species numerically found on nearshore sites in St. Croix included: bluehead wrasse (*T. bifasciatum*) 24%; blue tang (*A. coeruleus*) 12%, bicolor damselfish (*S. partitus*) 9%, blue chromis (*C. cyanae*) 8%, and creole wrasse (*C. parrae*) 8%. These five species represented 62% of nearshore fishes recorded numerically. Unlike the most common species found on nearshore reefs in the northern USVI, which were primarily herbivores, the most abundant species on St. Croix shallow reefs were planktivorous. This probably reflects the type food availability on the narrow shelf of the St. Croix insular platform. Species contributing the most biomass to nearshore sites in St. Croix included: blue tang (*A. coeruleus*) 10%, stoplight parrotfish (*S. viride*) 9%, black durgeon (*Melichthys niger)* 8%, bar jacks (*C. ruber*) 6%, and yellow goatfish (*Mugil martinicus*) 5%. These fish represented a variety of trophic groups.

The most common fish numerically on the offshore sites in St. Croix included: bluehead wrasse (*T. bifasciatum*) 18%, bicolor damselfish (*S. partitus*) 13%, blue chromis (*C. cyanae*) 13%, and brown chromis (*C. multilineata*) 5%. These four planktivorous species represented 49% of individuals recorded on offshore reefs in St. Croix. Unlike previous years, the creole wrasse, (*C. parrae*) was missing from the most common fish. The species contributing the highest biomass to fish communities on the mid-shelf sites included: blackbar soldierfish (*Myripristis jacobus*) 13%, redband parrotfish (*S. aurolineatum*) 11%, yellow goatfish (*M. martinicus*) 10%, black durgeon (*Melichthys niger)* 8% and stoplight parrotfish (*S. viride*) 4%.

The most common species numerically on the mesophotic sites in St. Croix included: blue chromis (*C. cyanae*) 25%, creole wrasse (*Clepticus parrae*) 15%, bicolor damselfish (*S. partitus*) 11%, bluehead wrasse (*T. bifasciatum*) 5%, and princess parrotfish (*S. taeniopterus*) 5%. These five species (four of which are planktivores) represented 61% of all fish recorded on those sites. Princess parrotfish also made up 4% of fish abundance on northern USVI mesophotic reefs and were the most common herbivore. Species contributing the most biomass to the mesophotic sites in St. Croix included: Caribbean reef shark (*Carcharhinus perezii*) 22%, black durgeon (*Melichthys niger)* 10%, spadefish (*Chaetodipterus faber*) 7% and creole wrasse (*C. parrae*) 5%. Except for the Caribbean reef shark, and similarly to northern USVI, the highest biomass on mesophotic reefs was seen in planktivores.

Table 3. The 2021 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	72	31.9±0.9	75
	Great Pond	52	12.1±1.2	40
	Jacks Bay	64	28.6±2.2	64
	Kings Corner	63	27.8±1.2	65
	Salt River West	47	17.8±1.6	48
	Sprat Hole	69	29.0±0.6	71
	Coculus Rock	41	18.0±0.8	40
	Black Point	61	23.2±2.4	59
	Brewers Bay	70	21.3±1.0	52
	Botany Bay	74	29.6±1.5	73
	Coral Bay	35	15.2±0.8	39
	Fish Bay	59	23.3±1.3	69
	Magens Bay	62	26.8±1.0	68
Offshore	Eagle Ray	60	26.8±1.2	63
	Buck Island, St. Croix	51	29.9±1.2	52
	Castle	53	19.6±1.2	52
	Mutton Snapper FSA	57	26.4±1.4	55
	Buck Island, St. Thomas	74	29.9±1.3	64
	Seahorse Cottage Shoal	67	26.8±0.6	57
	South Capella	65	29.2±2.8	60
	South Water	57	23.6±1.4	54
	Flat Cay	58	25.9±1.1	59
	Savana Island	70	28.8±1.4	64
	St. James	53	23.3±1.8	64
Mesophotic	Buck Island STX Deep	47	19.1±1.5	52
	Cane Bay Deep	35	13.4±0.7	33
	Lang Bank EEMP	66	27.8±1.4	70
	Lang Bank Red Hind FSA	54	22.8±1.3	56
	Salt River Deep	50	13.5±1.1	48
	College Shoal East	68	26.3±0.5	67
	Ginsburg’s Fringe	-	-	-
	Grammanik Tiger FSA	72	28.3±1.1	67
	Hind Bank East FSA	71	28.0±1.3	70
	Meri Shoal	62	29.6±1.2	61

Fish Abundance

Total fish abundances across nearshore, offshore, and mesophotic sites and years are shown in Fig. 17. As in previous, years total fish abundance was highly variable across sites and strata.

The sites with the highest overall fish abundance in 2021 were Botany Bay, Great Pond and Grammanik Tiger FSA. Fish abundance at Botany Bay was influenced strongly by very high numbers of bluehead wrasse (*T. bifasciatum*), blue chromis (*C. cyanae*) and newly settled parrotfish, less than 10cm in length. The relatively high fish abundance at Great Pond was due to large numbers of wrasses (*T. bifasciatum* and *Halichoeres* spp.), and at the Grammanik Tiger FSA to large schools of creole wrasse (*C. parrae*) and boga (*H. vittatum*). The site with the lowest fish abundance was (as in years past) Salt River Deep. This mesophotic site, defined by agariciid corals and high silt loads, also had the expected low species richness and biomass. A general decline in fish abundance in several offshore sites was noted, possibly due to the effects of bleaching and disease to those vulnerable midshelf reefs over time.

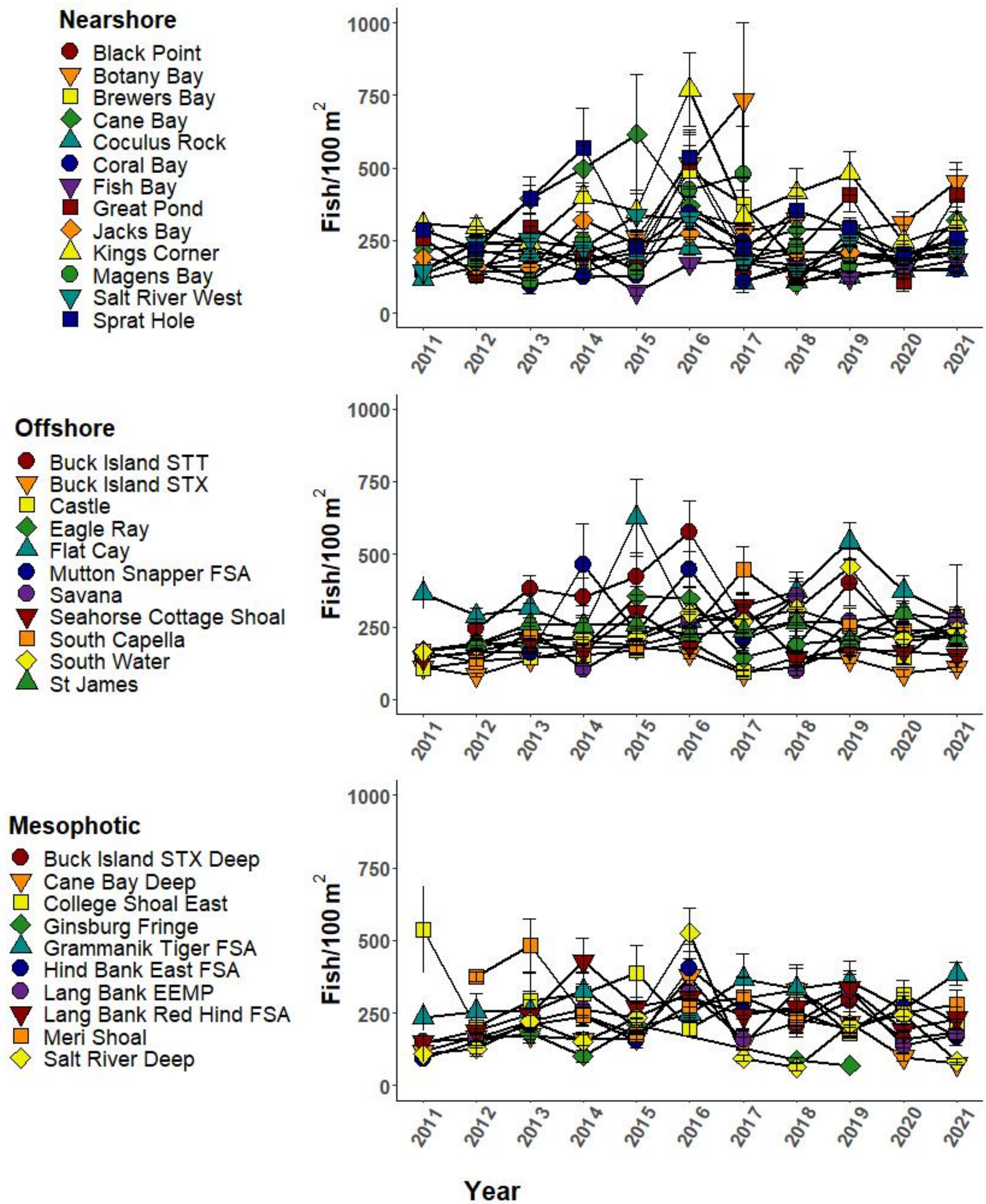

Figure 17. Fish abundance (±SE) across TCRMP monitoring sites over time.

Fish Biomass

Total fish biomass for all sites and years is shown in Fig. 18. As with abundance, biomass is highly variable across strata, sites and years. No temporal pattern is obvious, and differences appear to be seasonal or natural variation. College Shoal, the mesophotic site in the northern USVI, has over time shown the highest fish biomass values in the TCRMP sampling series. The other mesophotic sites off the northern USVI (Grammanik Tiger FSA, and Hind Bank FSA) have also consistently shown relatively high biomass values through the years. Although large pelagic fishes primarily influence this biomass, occasional large reef fish including snappers and groupers are also present on the protected northern USVI reef surveys. Biomass on three mesophotic sites on St. Croix: Lang Bank Red Hind FSA, Lang Bang EEMP, and Cane Bay Deep, was higher than all nearshore and offshore sites on St Croix except Kings Corner. Kings Corner, on the western end St. Croix, consistently produces the highest biomass of sites on St. Croix, and the highest biomass of all nearshore reefs in the TCRMP sampling series. Kings Corner receives a regular sea current and water mixing due to its location at the western point of the island. A variety of hard bottom, coral reef and sand habitats support larger numbers of sub-adult and adult parrotfishes, doctorfishes, yellowtail snapper and pelagic jacks. Sites with the lowest biomass in 2021 included Brewers Bay, Coculus Rock, Buck Island STX Deep and Salt River Deep.

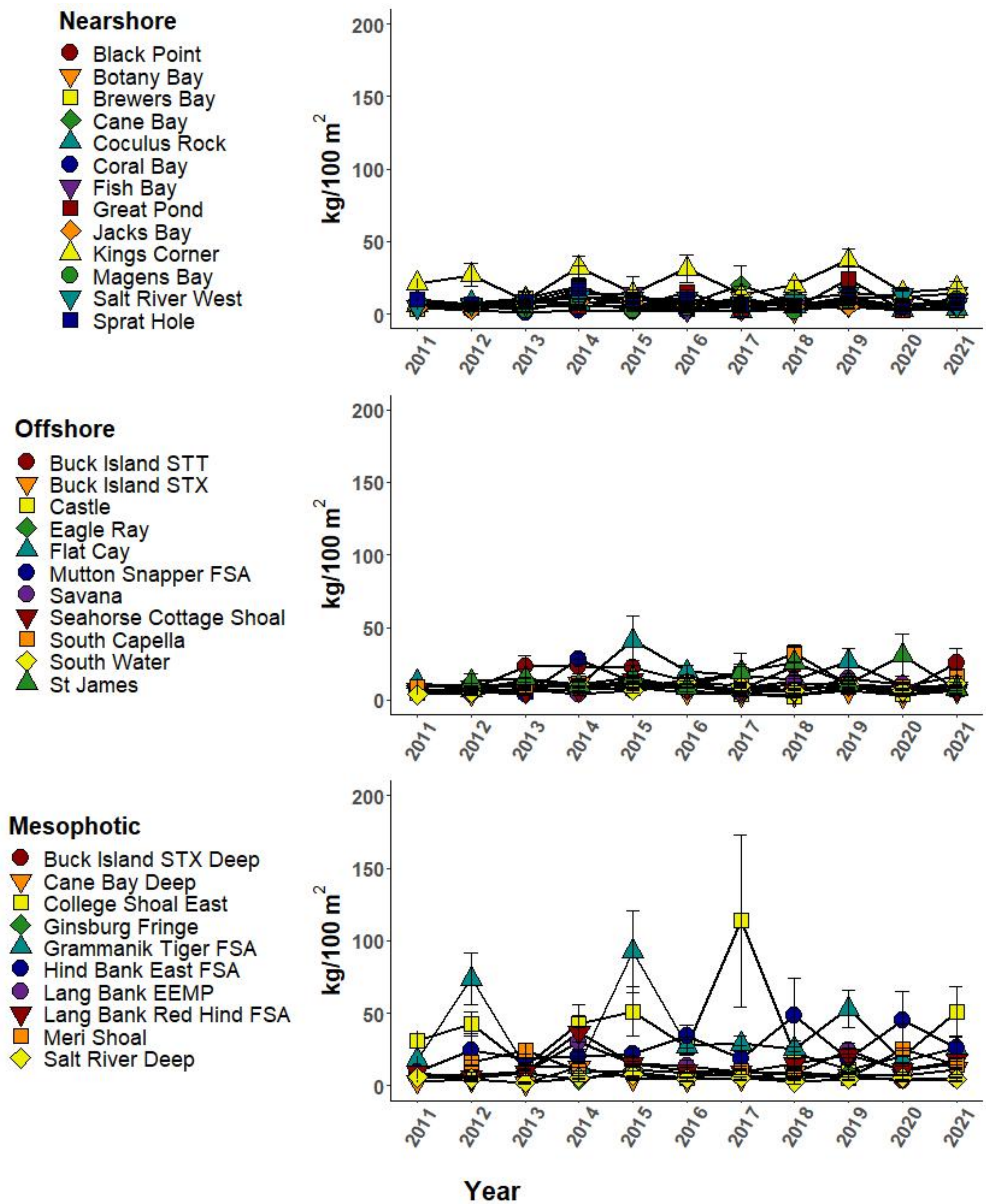

Figure 18. Mean fish biomass (±SE) across TCRMP monitoring sites over time.

BLACK SPINED SEA URCHIN *DIADEMA ANTILLARUM*

The abundance of the black spined sea urchin *Diadema antillarum* shows tremendous site-to-site variability (Fig. 19). 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 was a high abundance of *Echinometra* spp. that has not been quantified, but the abundance seems to have declined after 2017. 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.

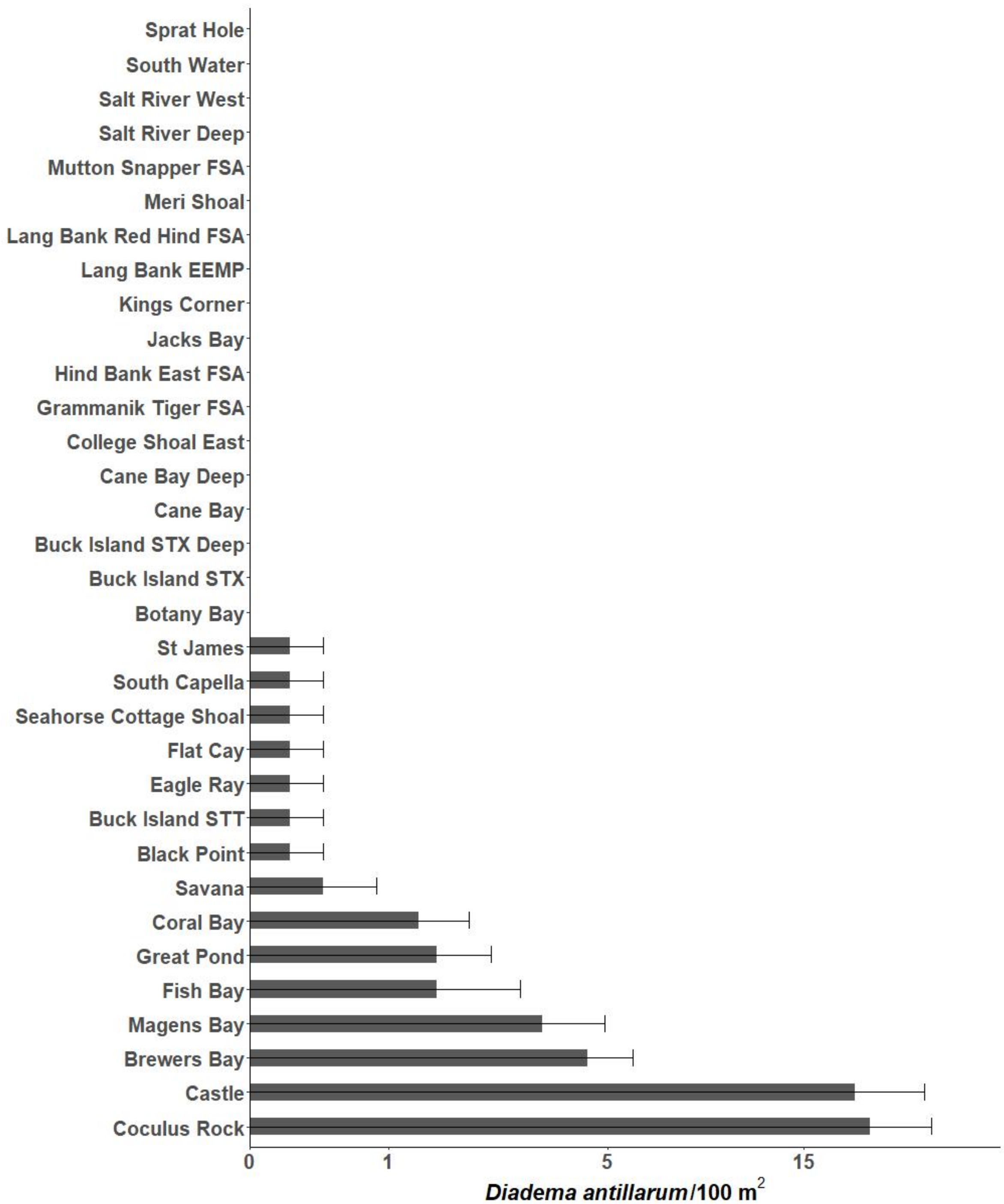

Figure 19. Average abundance (±SEM) of the black spiny sea urchin (*Diadema antillarum*) at TCRMP monitoring sites in 2021. Note the log scale.