The United States Virgin Islands

TERRITORIAL CORAL REEF MONITORING PROGRAM

ANNUAL REPORT

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A collaboration between:

The Center for Marine and Environmental Studies, University of the Virgin Islands

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MISSION STATEMENT

OUR VISION

To provide critical information on the status and threats to all the U.S. Virgin Island's coral reef ecosystems to increase management effectiveness and improve basic and applied coral reef research

OBJECTIVES

- **Monitor the status and trajectories of coral reefs across a variety 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 at a crossroads, facing the possibility of further dramatic decline. Management decisions made today will affect the goods and services that coral 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 in the USVI and has identified threats that influence 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 the 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).

EXECUTIVE SUMMARY

This event surpassed all known modern impacts from physical damage (storms and anchoring), ecosystem changes (fishing and disease), and pollution (terrestrial sediments and toxins). This thermal disturbance was compounded by record heat stress in the fall of 2023 and 2024, with widespread bleaching and community effects that are only now being investigated. Please see `Research Highlights` for more detailed information on this bleaching event.

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).

EXECUTIVE SUMMARY

Overfishing. There are clear indications that reefs of the USVI are suffering the effects of overexploitation of reef resources. 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 found a variety of species that are no longer encountered or are rare, including the black, Nassau, yellowfin, and tiger groupers, as well as the federally protected parrotfish species: blue, midnight, and rainbow. Rebuilding these fish stocks will require knowledge of 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. Signs of an increasing recruiting and spawning population of Nassau groupers in the northern USVI and more frequent sightings 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 (Grove et al. 2024; Heidmann et al. 2024). Therefore, for many species viable populations may still exist 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 (Ennis et al. 2016; 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 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) reefs with high populations of star corals (*Orbicella* spp.), which are 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 of 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, but not all, 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 in St. Thomas. 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. A red hind aggregation in the Red Hind Marine Conservation District (MCD) has dramatically rebounded (Nemeth 2005) and at the Grammanik Bank, a 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). About 875 individuals were seen in 2023-2024 (R. Nemeth, unpub. obs), contrasting the ~100 individuals seen spawning only a decade previously. Recruitment pulses of juvenile Nassau grouper in nearshore environments have also been noted in St. Thomas and St. John (R. Nemeth, unpub. data). Other territorial and federal closed areas in St. Croix, St. John, and St. Thomas are more recently established and thus may not show effects for several years. It has also been recently documented in the National Coral Reef Monitoring Program that larger and more numerous commercially important fish populations are found in deeper areas south of St. Thomas-St. John (Grove et al. 2024, Heidmann et al. 2024). 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. in prep.). Unpaved road segments have been implicated as the worst culprits in the production of sediment-laden run-off (Ramos-Scharrón and MacDonald 2007a, b) 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 23rd year of monitoring on reefs surrounding St. Croix, St. John, and St. Thomas, and includes surveys from annual sampling (2001-2023) as well as an additional post-bleaching benthic sampling that occurred in the early spring 2024. 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.

The TCRMP website describes the program and houses updated data

https://www.vitcrmp.org

Data can also be requested directly from the research team by contacting Erin Hollander at erin.hollander@uvi.edu

INTRODUCTION

Impacts, Scope, and Comparisons of the 2023 Coral Bleaching Event

In 2023, sea surface temperatures in the U.S. Virgin Islands reached unprecedented highs, inducing a mass coral bleaching event—the fourth to occur in the past 20 years (Fig. 2). Sea surface temperatures in the USVI have been increasing at a rate of about 0.2°C per decade since the 1980's (Fig. 3), increasing the incidence of coral thermal stress, measured as Degree Heating Weeks (DHW; Fig. 3). Major bleaching events since TCRMP monitoring began have occurred in 2005 (DHW=15.5), 2010 (DHW=9.8), and 2019 (DHW=8.9). In the fall of 2023, DHW peaked at 18.7, coinciding with observations of widespread bleaching and coral mortality, in both shallow and mesophotic environments. The waters were warmest during the months of September and October, when sea surface temperatures consistently stayed above 30.0˚C.

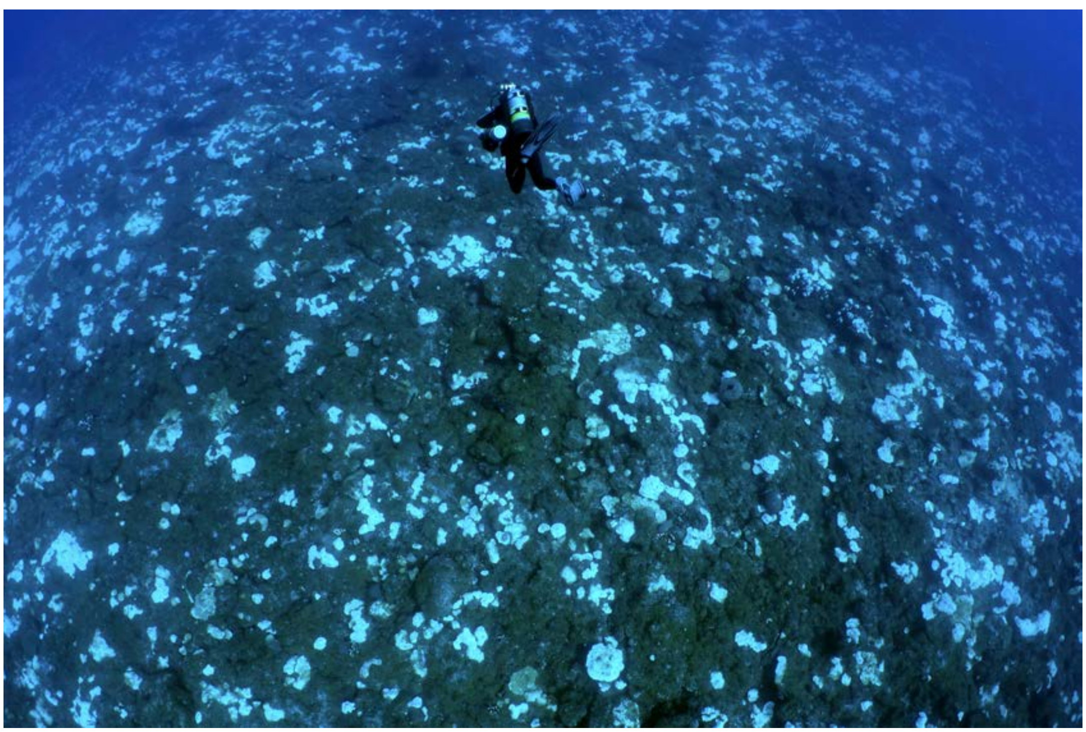

Figure 2. TCRMP diver surveys bleached corals during the peak of thermal stress at the Meri Shoal TCRMP monitoring site, December 7th, 2023 (photo credit: N. Krampitz)

INTRODUCTION

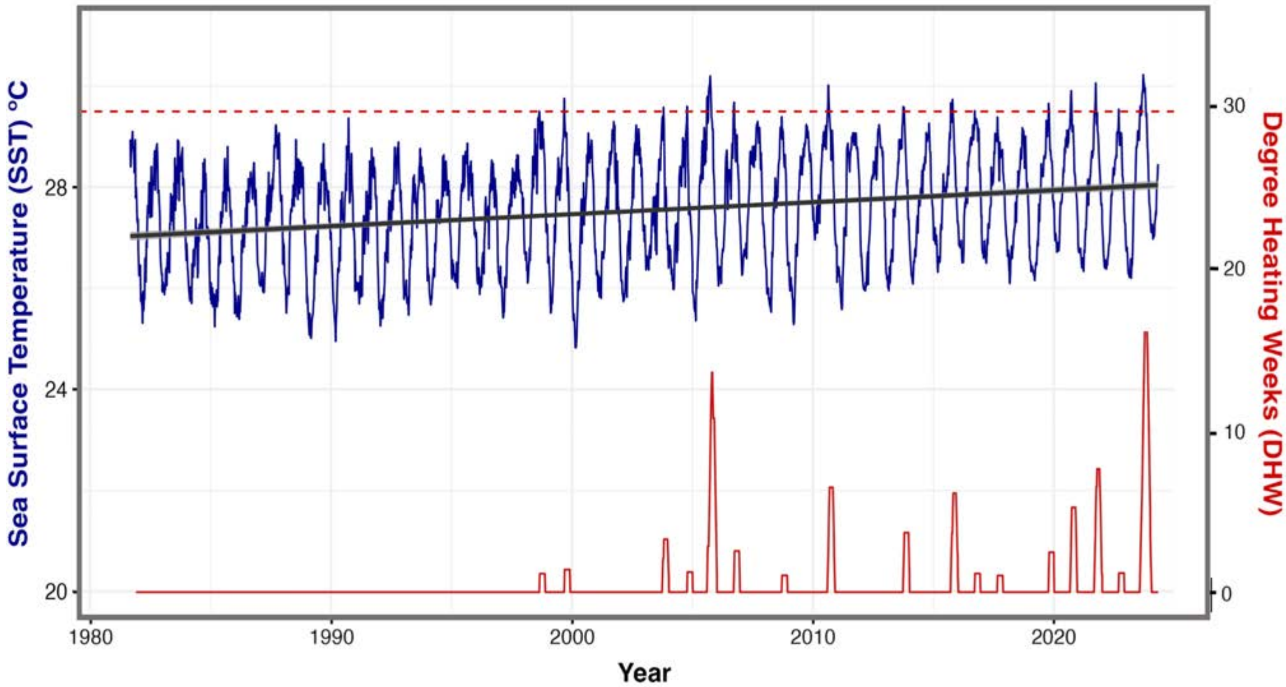

Figure 3. 1982 – 2023 sea surface temperature (left axis ,blue line), left vertical axis and degree heating weeks (right axis, red line) for the USVI.
The black line is a linear fit of the sea surface temperature showing a 0.024˚C increase in temperature per year (y = 0.0237/year * x - 26.75). Degree 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. (Optimum Interpolation Sea Surface Data, OISST, from https://psl.noaa.gov/data/gridded/data.noaa.oisst.v2.highres.html, accessed April 17, 2025 by Lauren Olinger Ph.D.)

Despite higher heat stress in 2023 compared to the 2005 bleaching event, it appeared that the bleaching response of corals in TCRMP locations was slightly stronger in 2005— both in bleaching extent and bleaching prevalence (Fig. 4). It should be noted, however, that not all sites were sampled during the peak of heat stress every year, which may underestimate bleaching responses for a given site. Furthermore, many of the deeper sites (e.g., Salt River Deep, Buck Island Deep) were not established until after 2005.

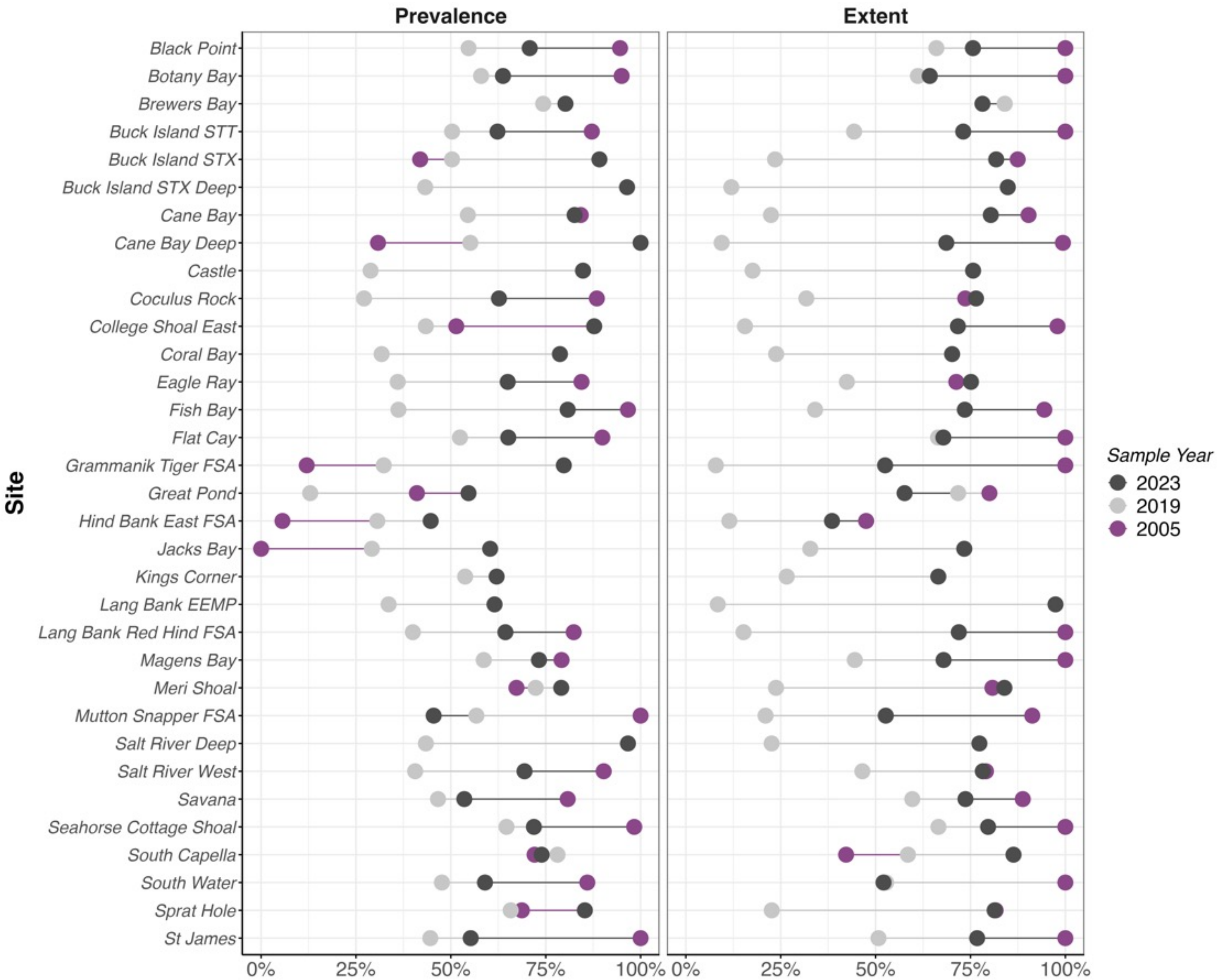

Figure 4. Mean coral bleaching prevalence and extent at each TCRMP site in 2005, 2019, and 2023. Bleaching prevalence is the proportion of the community showing some level of stark white bleaching. Bleaching extent is the mean proportion of the colony area affected by stark white bleaching. Sites are organized by increasing depth.

In 2005, bleaching prevalence averaged 70±6% across all sites with a dramatic extent (mean 82% of colonies' surface affected). In 2019, bleaching prevalence was lower (46±3%) as well as extent (36±4%). In this most recent year, bleaching prevalence was estimated to be near 71.5±2.5% across TCRMP locations with over 70% of each colony affected on average. All sites in 2023 were sampled between October 10th and December 12th, with peak heating stress estimated to be mid-October. Species most affected by the bleaching in terms of prevalence included *Agaricia* spp. (83%), *Orbicella* spp. (70%), *Millepora* spp. (68%), and branching *Porites* spp. (65%).

In mid-February 2024, a post-bleaching sampling began in response to the observed bleaching event during annual TCRMP sampling. Although bleaching prevalence on the reef had largely abated by early spring (~5% bleached colonies), paling presence was notably high (~30%), especially on deeper mesophotic reefs. Correspondingly, these deeper reefs also experienced slightly higher bleaching prevalence during the fall than their shallow water counterparts. Recent partial mortality was highly prevalent in post-bleaching surveys (~20% of corals affected) across all depths, and disease was occasionally noted, most frequently on *Orbicella* colonies at deeper depths. Changes in coral coverage during and after the bleaching event varied by site (Fig. 5), but there was an overall significant decline in relative coral coverage, with a mean relative loss of 21.8±4.3% in less than 6 months. Deeper sites, *Orbicella*-dominated sites, and those with higher initial coral coverage appeared to be most affected by coral mortality.

Sites with higher bleaching prevalence experienced significantly greater declines in coral cover, indicating a strong negative relationship between bleaching intensity and coral cover change (linear regression; $p < 0.01$). A significant relationship was also found between the projected accumulated DHW at individual sites and the observed extent of bleaching ($p = 0.02$). These results indicate for the 2023 bleaching event, the intensity (i.e., extent) of bleaching was highly correlated with DHW, but the impact of bleaching, notably measured by associated losses in coral coverage, was largely driven by prevalence of bleaching at a site. These findings underscore the importance of not only quantifying bleaching on a site level (prevalence), but also on a colony level (extent) and intensity (paling vs. bleaching) level to fully understand how corals are reacting to rising sea surface temperatures.

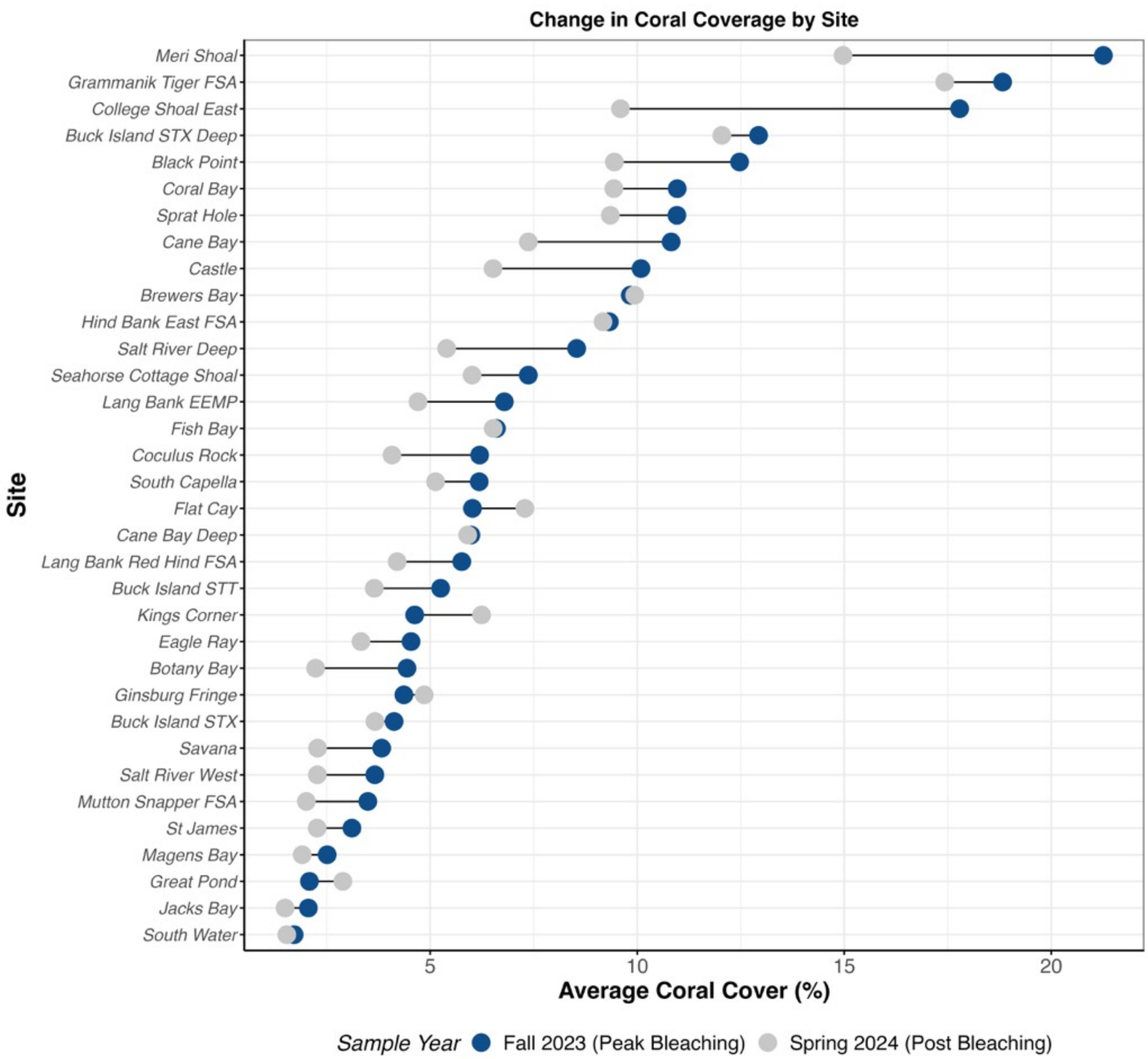

Figure 5. Mean coral cover at each TCRMP location during peak bleaching in the fall of 2023 and post-bleaching sampling in the fall of 2024.
Loss in coverage over this 4-to-6-month period was significant (paired t-test: $p<0.0001$).

INTRODUCTION

Introduction

The U.S. Virgin Islands consist 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, which are famous for white sand beaches that give way to clean, clear marine waters. The diverse marine life of the coral reefs and other habitats attract thousands of snorkelers 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 whelk, queen conch, 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.

INTRODUCTION

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; Brandt et al. 2021). 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). In addition, losses of the black spined sea urchin to an epizootic scutiociliate outbreak in 2022 have greatly reduced the already impacted populations of this key Caribbean herbivore (Hylkema et al. 2023; Hewson et al. 2023).

High thermal stress and coral bleaching events affected the northeastern Caribbean in 2005, 2010, 2012, 2019, and 2023. Prior to 2023, the most severe high sea surface temperature (SST) event occurred in 2005. Degree Heating Weeks (DHW) for this event peaked at 15.4 (NOAA Coral Reef Watch 2024; 5km satellite product) and was associated with severe coral bleaching and mortality (Miller et al. 2009). In the fall of 2023, DHW reached an all-time maximum of 18.7 (NOAA Coral Reef Watch 2024; 5km satellite product), coinciding with observations of wide-spread bleaching and coral mortality in shallow and mesophotic environments. (Authors, unpub. obs.) Recent research developed bleaching threshold temperatures for 24 of the 34 TCRMP monitoring sites dominated by star corals of the genus *Orbicella* (Smith et al. 2016a). This 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. This is further evidenced by the observed widespread (>45% of corals) bleaching that occurred at all mesophotic TCRMP sites in this most recent survey year (2023). Earlier events and the species-specific responses of Caribbean corals are summarized in Smith et al. (2013, b) for shallow corals and Smith et al. (2016, a) for shallow and mesophotic corals.

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; 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 2004-2006 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 are 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 while 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 currently sample reefs below 30m, and therefore misses the dominant habitat in the northern USVI. The Deep Coral Reef Monitoring Program (DCRMP) was instituted in 2019 to sample reef fishes and basic benthic attributes in depths of 30-50 m on hardbottom areas south of St. Thomas and St. John (Grove et al. 2024; Heidmann et al. 2024). However, this program has no long-term sustained funding and sampling ended in 2022.

This report presents monitoring results from 2001-2023 in St. Croix and from 2003-2023 in St. Thomas and St. John, as well results from a benthic-only post bleaching sampling that occurred at all sites in the spring of 2024.

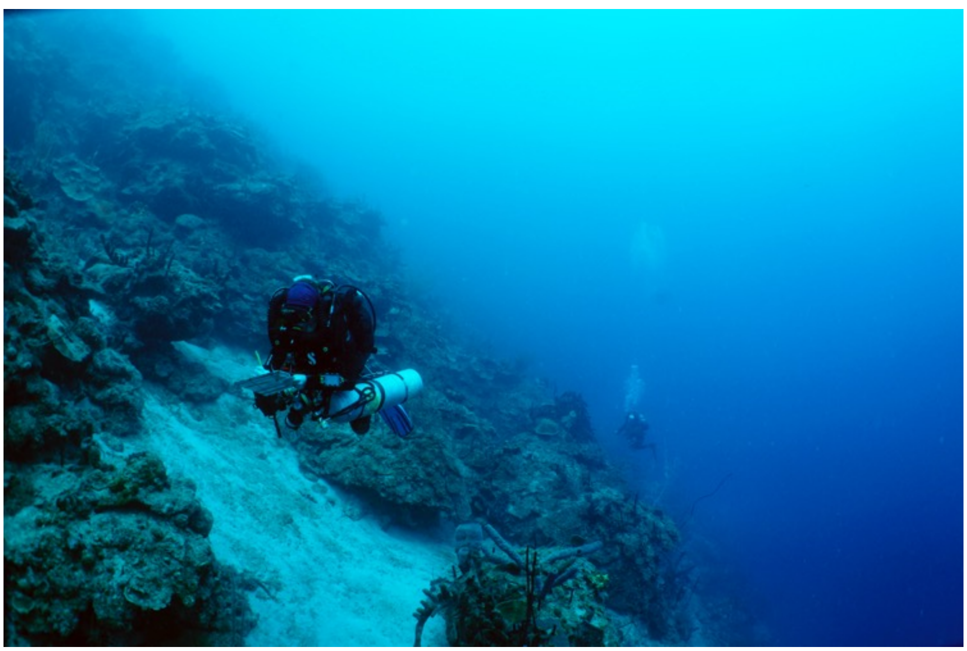

Figure 6. A TCRMP research diver (S. Heidmann) on closed circuit rebreather records a roving fish survey at the Salt River Deep wall site, 30m/100' depth (April 9, 2020; credit: S. Meiling).

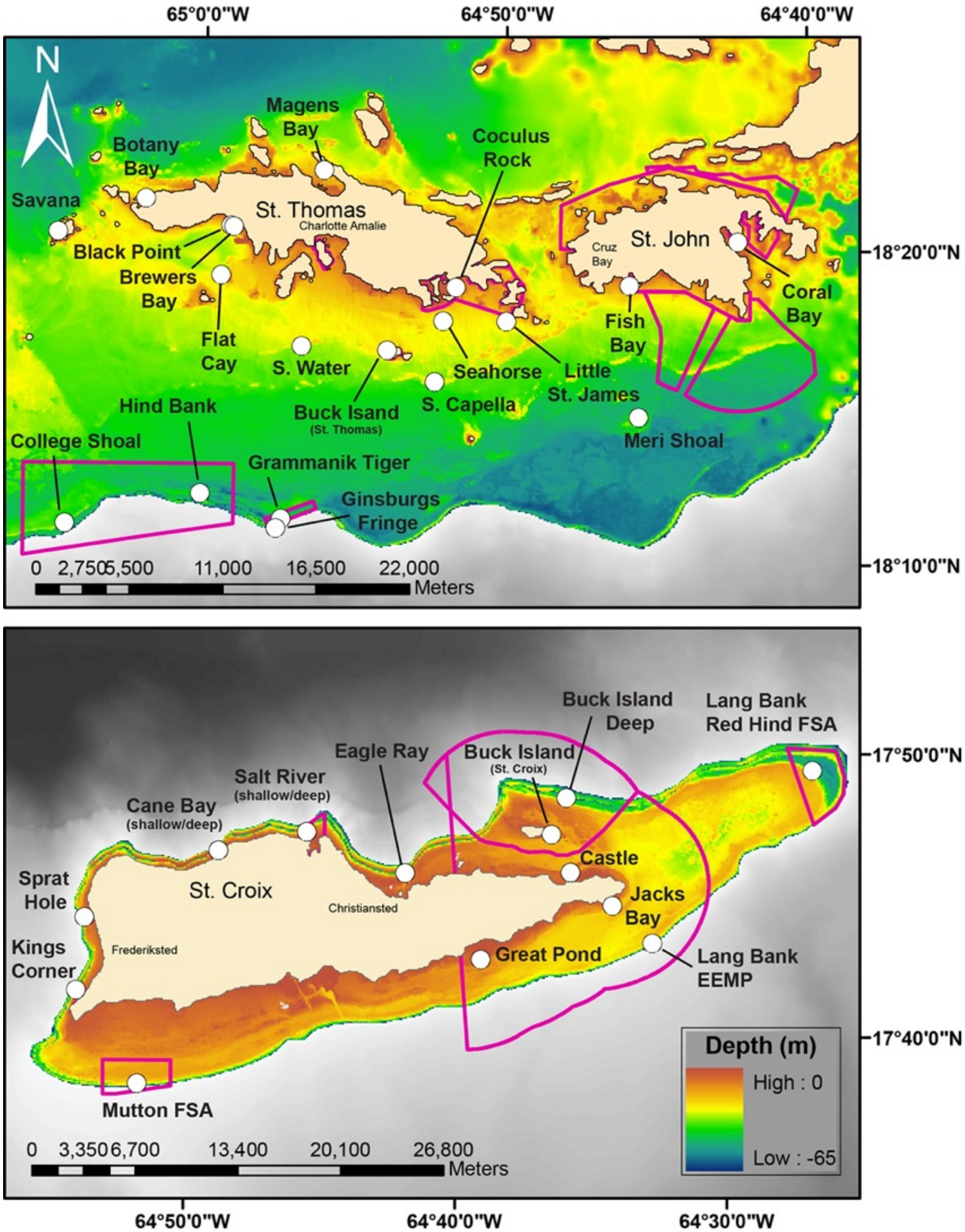

Figure 7. Locations of Territorial Coral Reef Monitoring Sites in the US Virgin Islands. Boundaries indicate federal and territorial marine protected areas.

METHODS

Methods

BENTHIC ASSESSMENTS

The University of the Virgin Islands determined the benthic composition at 34 long-term monitoring sites between 2001 and 2024 (Table 1; Table 2). All data is now available at the TCRMP website and updated annually after quality control:

https://www.vitcrmp.org

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) .

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, 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). Because of its deep depth, Ginsburgs Fringe at 60-66m was only sampled for benthic cover and occasional fish surveys.

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 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. 8). 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, 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.

Figure 8. 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	18.18901	-64.95630	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 sampling dates for fish, urchin, benthic cover, and coral health at each site. Where there are two dates an asterisk (*) indicates the full fish survey date for St. Thomas-St. John and clean up surveys for St. Croix (finished during post-bleaching surveys in 2024). Only benthic surveys were completed at Ginsburgs Fringe. Only benthic and health surveys were completed during the post-bleaching sampling season in spring of 2024.

Island	**Site**	**Annual Survey Sample Date** (2023)	**Post-Bleaching Sample Date** (2024)
St. Croix	Buck Island STX	11/9/23	3/13/24
	Buck Island STX Deep	11/9/23, 3/11/24*	3/11/24
	Cane Bay	11/12/23	3/13/24
	Cane Bay Deep	11/12/24, 3/13/24*	3/13/24
	Castle	11/13/23	3/11/24
	Eagle Ray	11/11/23	3/13/24
	Great Pond	11/14/23	3/12/24
	Jacks Bay	11/14/23	3/14/24
	Kings Corner	11/15/23	3/15/24
	Lang Bank EEMP	11/14/23, 3/16/24*	3/16/24
	Lang Bank Red Hind FSA	11/13/23	3/12/24
	Mutton Snapper FSA	11/10/23, 3/15/24*	3/15/24
	Salt River Deep	11/11/23	3/14/24
	Salt River West	11/11/23	3/14/24
	Sprat Hole	11/12/23	3/14/24
St. John	Coral Bay	8/25/23*, 10/13/23	2/16/24
	Fish Bay	8/25/23*, 10/13/23	2/16/25
	Meri Shoal	9/5/23*, 12/7/23	4/22/24
St. Thomas	Black Point	8/30/23*, 10/17/23	2/15/24
	Botany Bay	9/27/23*, 10/12/23	4/17/24
	Brewers Bay	8/30/23*, 10/17/23	4/11/24
	Buck Island STT	9/1/23*, 10/19/23	4/2/24
	Coculus Rock	8/25/23*, 10/13/23	4/2/24
	College Shoal East	9/22/23*, 12/6/23	5/14/24
	Flat Cay	8/30/23*, 10/17/23	2/15/24
	Ginsburgs Fringe	10/26/23	5/22/24
	Grammanik Tiger FSA	9/7/23*, 12/5/23	4/8/24
	Hind Bank East FSA	9/6/23*, 12/6/23	5/14/24
	Magens Bay	9/27/23*, 10/12/23	4/17/24
	Savana	9/27/23*, 10/30/23	4/11/24
	Seahorse Cottage Shoal	8/31/23*, 10/30/23	3/7/24
	South Capella	8/31/23*, 10/23/23	4/22/24
	South Water	9/1/23*, 10/23/23	4/26/24
	Little St James	9/25/23*, 10/19/23	4/26/24

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 and old. 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).

Diseases were conservatively categorized into recognized Caribbean scleractinian diseases and syndromes that included bleaching, black band disease, dark spots disease, white plague, stony coral tissue loss disease, 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.

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 (%), or extent of the colony affected was also estimated for each impairment category.

METHODS

FISH CENSUS

Fish abundance, size, and diversity were recorded using transects and 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 30 x 2 m in size. Starting in sampling year 2009 belt transects were enlarged to 25 m x 4 m, 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 (25 m x 4 m) 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 at that time (Menza et al. 2006; Friedlander et al. 2013). Before 2019 ten transects were conducted at each site. A retrospective analysis showed that nine transects were sufficient to capture within-site variability, and so the number of transects was reduced to nine starting in 2019.

All transects were begun at a random location on the site and were laid 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, etc. up to >150. This sampling strategy is not optimized for noting the presence and abundance of small (<5 cm) gobies and blennies, and the taxa were inconsistently recorded.

Roving diver survey (RDS) were also conducted at each site to estimate site fish diversity. Prior to 2016, RDS surveys in water depths less than 25 m were 30 minutes in duration and, because of restrictions in dive length, were 15 minutes in deeper water. Retrospective analysis showed that almost all diversity was captured in the first 15 minutes of the survey. Therefore, to make deeper sites comparable, the methods were switched to 15 minute RDS at all sites in 2016. In all surveys, all species encountered were recorded except blennies and gobies.

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 25 x 2 m belt transects corresponding to the return of the fish transects. The mean number of sea urchins per 100 m^2 was calculated for each site.

Data from all fish surveys were transcribed to Microsoft Excel and Access spreadsheets and analyzed for descriptive statistics of reef fish assemblage structure. Starting in 2021, data entry occurred in a customized data entry system with built-in quality control parameters. Data presented in individual site summaries represents data collected for the current report period only. Due to diving constraints, completion of all fish surveys at five sites in St. Croix could not be achieved during the annual sampling mission for the current survey year (2023). As a result, these surveys were 'picked up' during the spring 2024 mission. More details with specific dates and locations can be found in Table 2.

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. 9).

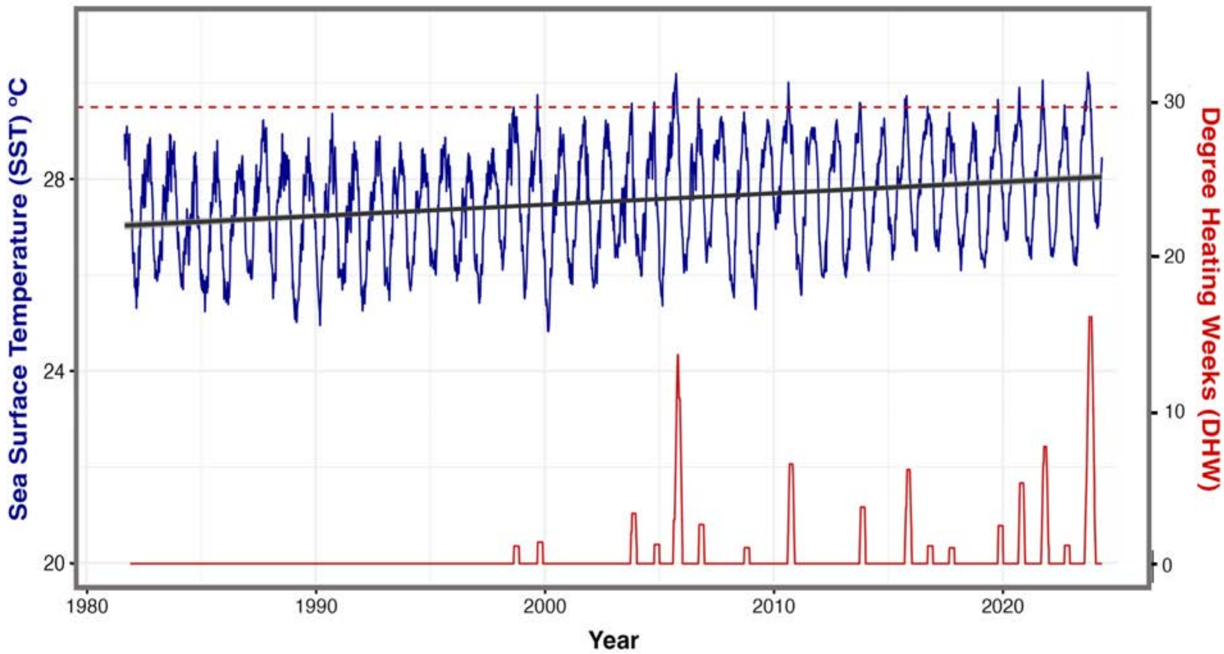

Figure 9. Sea surface temperatures and coral degree heating weeks of the US Virgin Islands from 1984 – 2024.
The black line is a linear fit of the sea surface temperature showing a 0.024˚C increase in temperature per year (y = 0.0237/year * x - 26.75). Degree heating weeks (DHW) are calculated as the 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. (Optimum Interpolation Sea Surface Data, OISST, from https://psl.noaa.gov/data/gridded/data.noaa.oisst.v2.highres.html, accessed April 17, 2025 by Lauren Olinger Ph.D.)

BENTHIC COMMUNITIES AND CORAL REEF HEALTH

Benthic cover was monitored at 34 monitoring sites and coral health was monitored at 33 sites in 2023. Raw data for benthic coverage, coral health, algae heights, and benthic temperature can be found at https://www.vitcrmp.org/data-and-methods . 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 following the 2005 coral bleaching event but losses in coral cover following the 2010, 2012, 2019, and 2023 bleaching events were less evident (Fig. 10). Recovery following bleaching in 2005 was marginal, especially in mesophotic sites. Direct impacts of the more recent bleaching events (2019 and 2023) are likely obscured by overall chronic decline of coverage regardless of the temperature conditions. Sites that had low coral cover to start generally lost far less relative cover from bleaching stress. While part of this may be attributed to the difficulty of detectability as coral cover values nearer to 0, it is also true that these sites tend to be dominated by small species that are more resistant to bleaching and disease related mortality (Smith et al. 2013b).

However, prior to impacts of SCTLD in 2019, 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 (Fig. 11; Fig. 12). Generalities that might indicate why these sites are recovering are difficult to pinpoint, but the coral communities in these reefs are all diverse. This diversity may contribute to recovery as thermally sensitive, but fast-growing species, such as *Agaricites* spp. and *Porites porites,* may push increases in coral cover.

The deepest site, Ginsburgs Fringe, has lost site lost 87% of its coral cover in the past decade, in what appears to be a steady continuous decline (Fig. 11). Multiple co-occuring stressors are apparent at this TCRMP site, including invasive lionfish and high cover of the macroalgae *Lobophora variegate*. However, the most obvious cause of disturbance is anchoring on the reef (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. These 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)). 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.

Additionally, there are a few shallow sites that have experienced chronic degradation. Magens Bay is highly impacted by sedimentation since it is largely enclosed, surrounded by steep hillsides under constant development (sediment run-off), and is susceptible to strong winter swells (Rothenberger et al. 2008). 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 textilis*), which has been competing for benthic space and slowly decreasing coral cover by overtopping colony margins.

In January 2019 stony coral tissue loss disease (SCTLD) was first observed at the Flat Cay monitoring site. Over the span of a few years, it spread across all reefs in the U.S. Virgin Islands, causing significant tissue loss and mortality, especially to highly and moderately susceptible species (most brain corals and bouldering corals; Brandt et al. 2021). During its spread, numerous organizations in the USVI coordinated response teams to attempt to mitigate and slow the spread of the disease. However, dramatic declines in species diversity and coral coverage still occurred. The TCRMP locations most adversely affected by SCTLD include Kings Corner, Flat Cay, and Brewer’s Bay with relative coral cover losses of 67%, 64%, and 59%, respectively, in the three years following disease introduction.

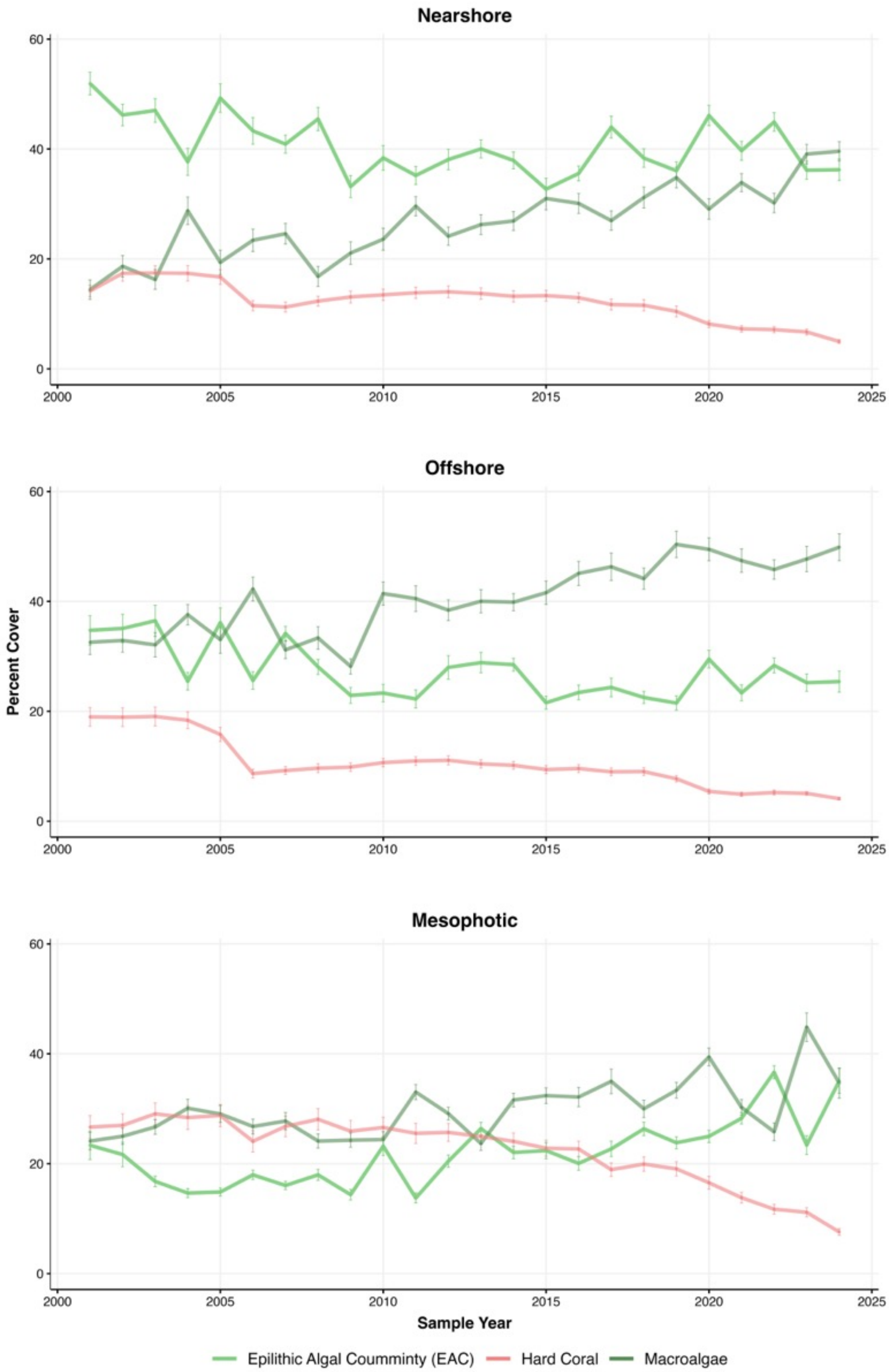

Figure 10. Mean hard coral, macroalgal (encrusting and erect), and epilithic algal community (EAC) coverage (±SEM) across all TCRMP sites during sampling from 2002-2024.
Data is split into mesophotic (≥27 m), offshore (7-24 meters; ≥2 km offshore), and nearshore (6-17 meters; <2 km offshore).

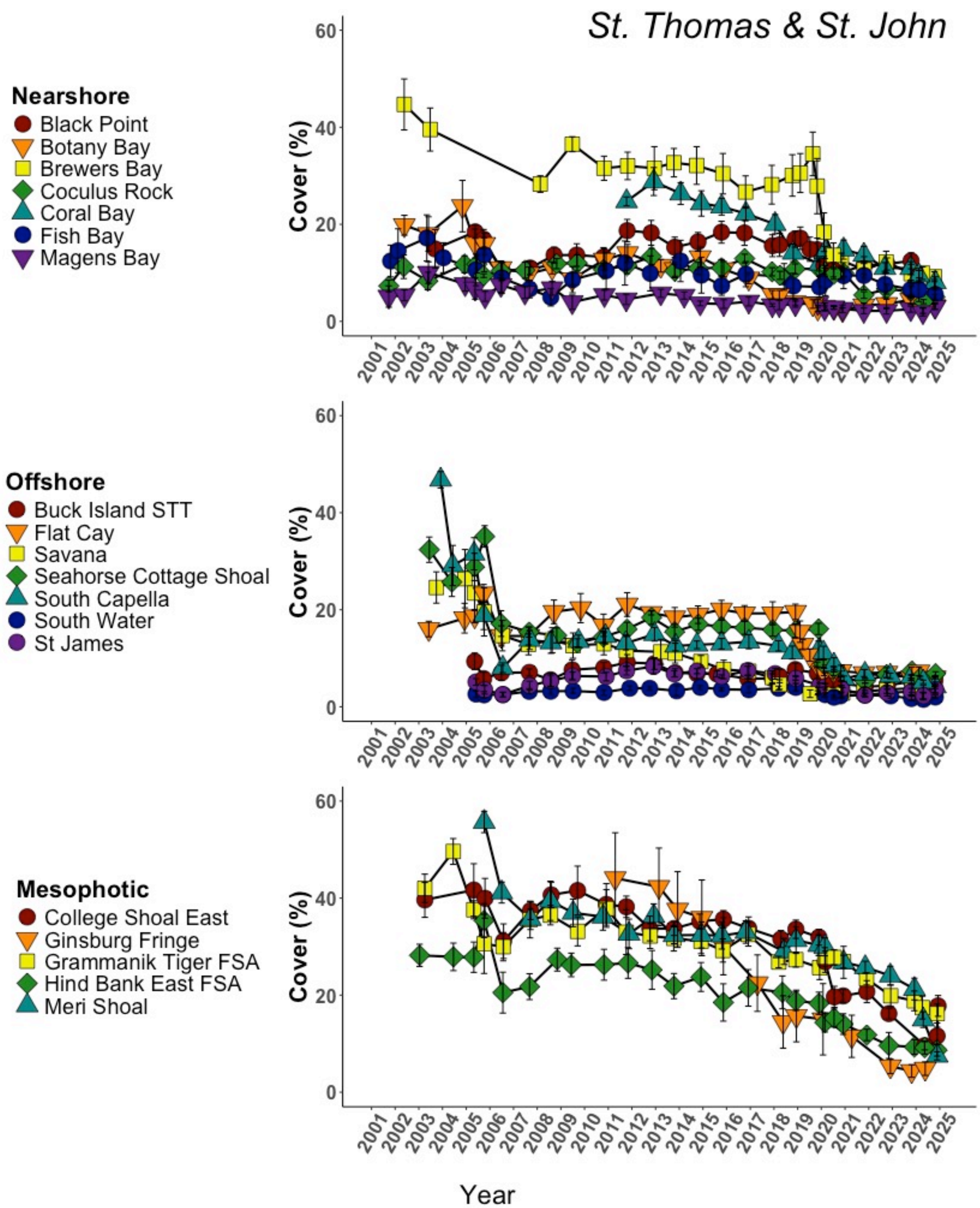

Figure 11. Coral cover (±SE) across St. Thomas and St. John TCRMP monitoring sites from 2001 – spring 2024.

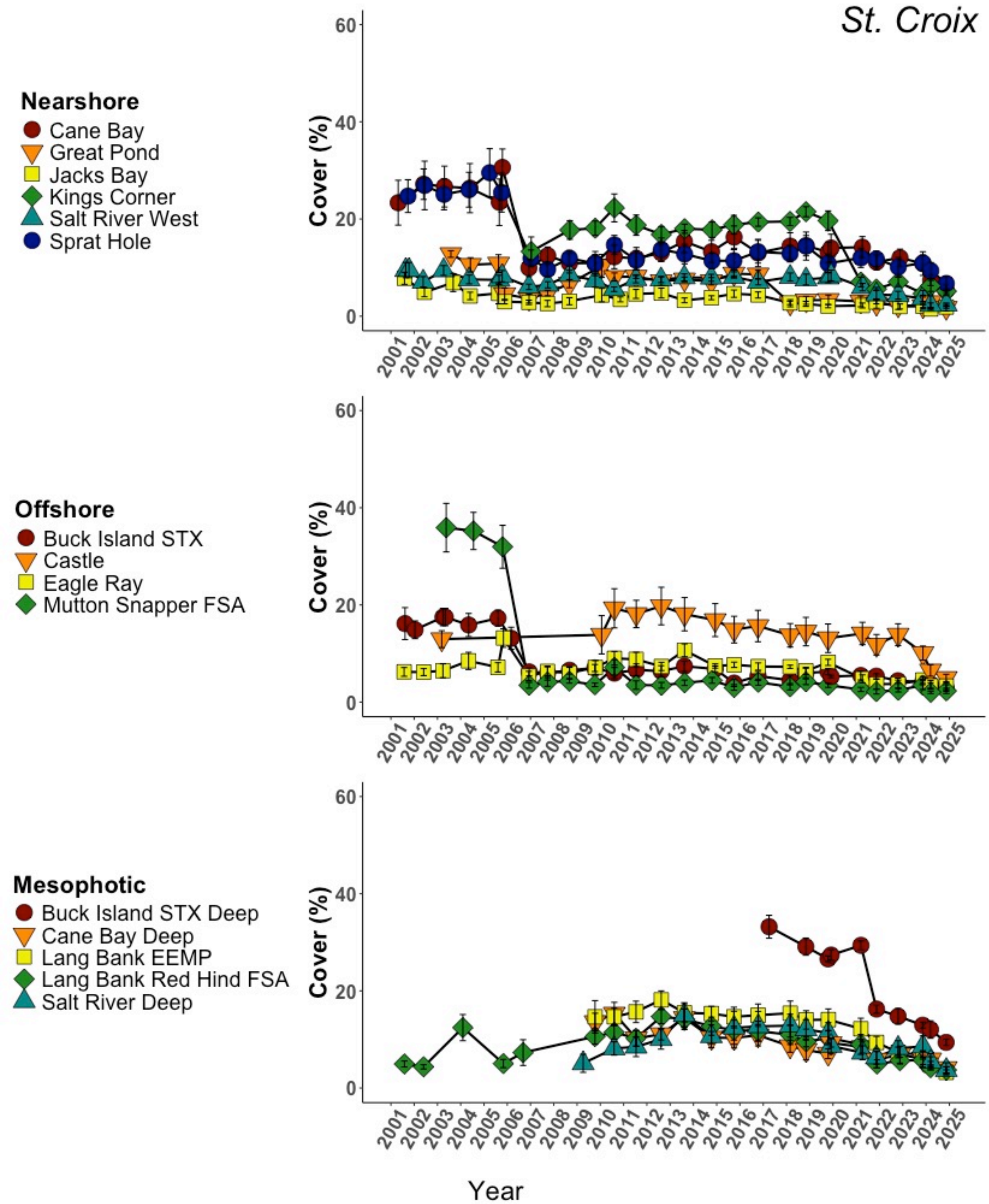

Figure 12. Coral cover (±SE) across St. Croix TCRMP monitoring sites from 2001 – spring 2024.

ALGAL COVER

Epilithic Algal Community Cover

Algae show the highest inter-annual variability of any group of benthic organisms and is largely due to seasonality. The cover of epilithic algae is no exception since it tends to negatively covary with more ephemeral macroalgae (Fig. 10). 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 *Ramicrusta*. Epilithic algae coverage appears to be highest at mesophotic sites, surpassing all other macroalgal coverage in most years (Fig. 13).

Macroalgal Cover

Macroalgal cover has been increasing at most TCRMP sites, particularly where coral cover has declined (Fig. 10). At sites where there was no loss of coral cover, increased macroalgae may be due to declining grazing, such as at Eagle Ray, the Buck Islands (St. Thomas and St. Croix), Cane Bay, Meri Shoal, South Capella, and Sprat Hole. Additionally, this might occur when 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. At Savana the large increase in macroalgae in was due to the expansion in encrusting *Ramicrusta textilis*, (*Ramicrusta* is classified with macroalgae in TCRMP data summaries despite its largely encrusting morphology). This increase in *Ramicrusta* was also at the expense of epilithic algae.

Filamentous Cyanobacteria

Filamentous cyanobacteria cover has been increasing at many sites in the TCRMP since the 2005 coral bleaching event. 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 (Fig. 13). This is particularly true at many sites on St. Croix.

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 often have secondary metabolites that deter grazing on the cyanobacteria and on palatable macroalgae coated with cyanobacteria (Fong et al. 2006). Filamentous cyanobacteria can inhibit the recruitment of coral larvae (Kuffner et al. 2006) and has 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 to understand the factors influencing bloom formation and which reefs are most vulnerable.

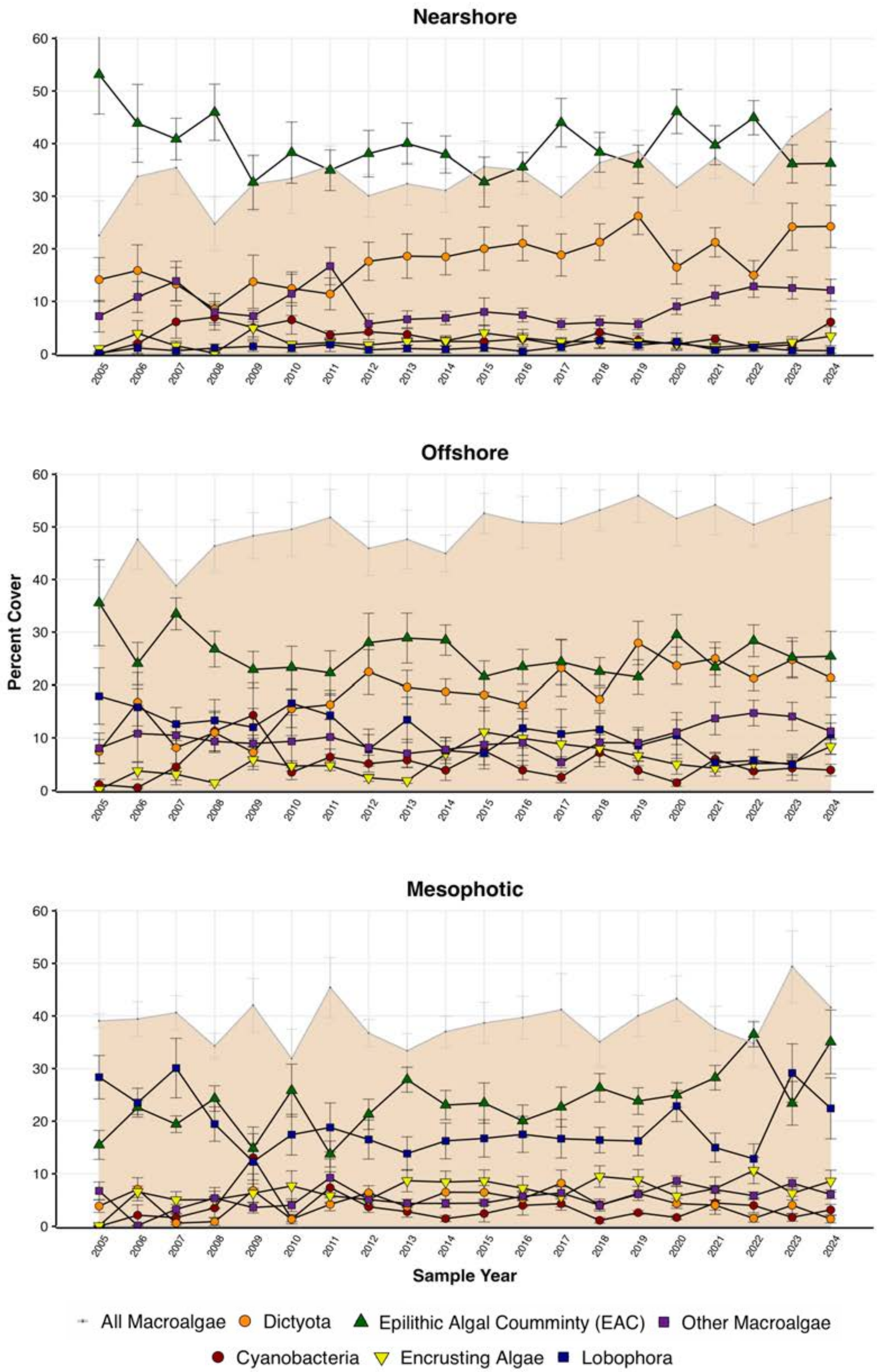

Figure 13. Mean algal coverage (±SEM) across TCRMP monitoring sites from 2005-spring of 2024. All macroalgae (grey line/orange shading) includes all erect and encrusting algae combined, as well as cyanobacteria. Other macroalgae (purple) includes all erect macroalgae except *Dictyota* (orange) and *Lobophora* (blue)

OTHER BENTHIC COVER

Gorgonian and Antipatharian Cover

The cover of gorgonians and antipatharians has been consistent or increasing at most monitoring sites throughout the years of monitoring (Fig. 14). Black Corals (antipatharians) are rare and when they occur tend to be more prominent in deep monitoring sites. For many gorgonian species their abundance tends to peak in shallow water where there is constant swell (benthic orbital turbulence). Both gorgonians and antipatharians did not seem sensitive to the thermal stress events in 2005, 2010, and 2012 (Tsounis and Edmunds 2017), however bleaching was observed on many gorgonians in 2023 (author, pers. obs). In most cases, gorgonians are a relatively minor component of cover because of their upright growth form and small branches which 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, which has also been observed by a separate research group in the same area (Tsounis and Edmunds 2017). Magens Bay had previously shown 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).

Sponge Cover

There is an indication of slightly increasing sponge cover at select nearshore and offshore sites—but general sponge coverage appears steady (Fig. 14). Increases were most pronounced at sites around St. Thomas, including Black Point, Buck Island, Flat Cay, and Magens Bay. Sites such as Black Point, South Water, 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).

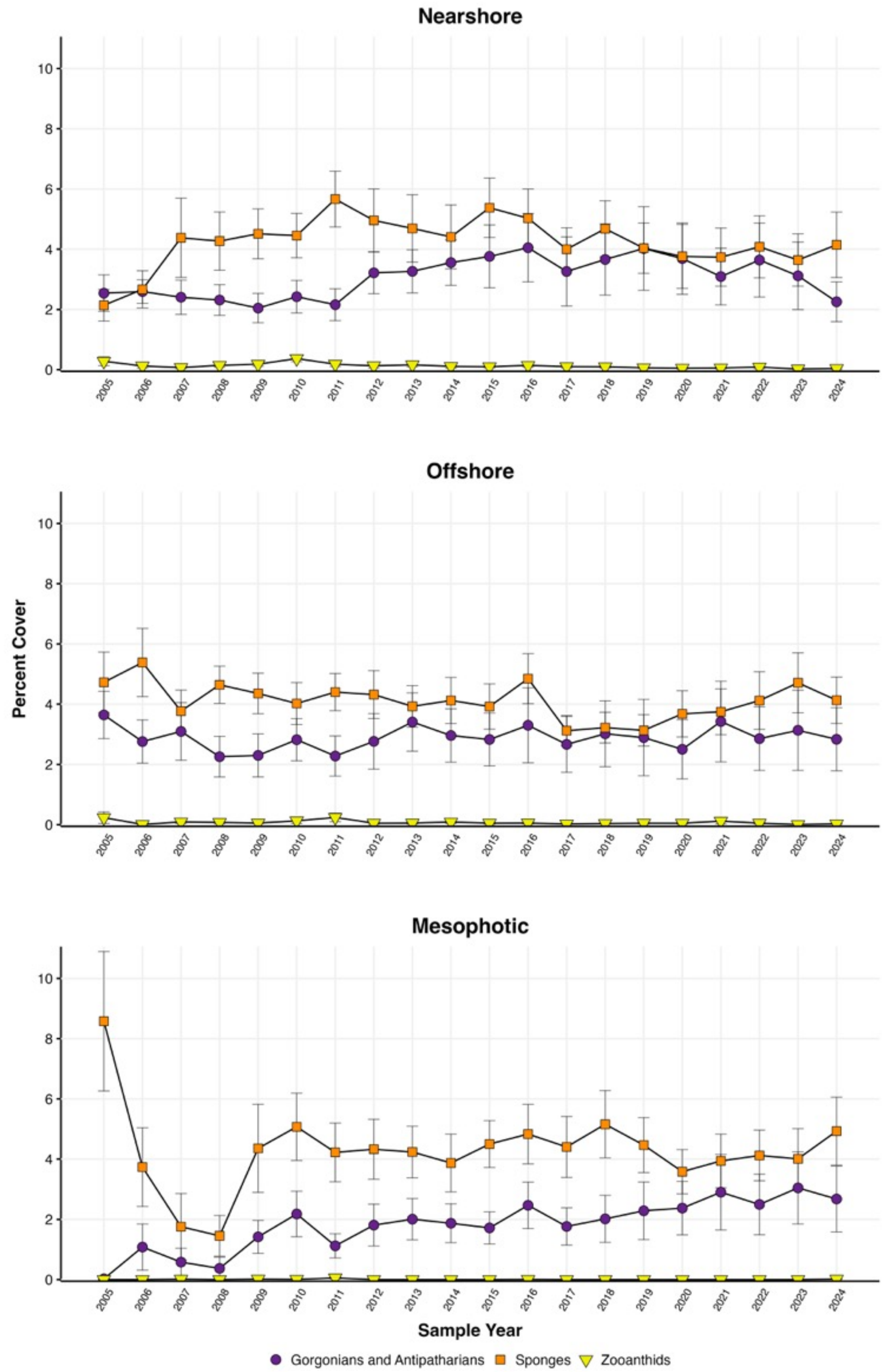

Figure 14. Mean benthic cover (±SEM) of gorgonians, sponges, and zooanthids from 2005-spring of 2024 at TCRMP locations.

FISH COMMUNITIES

Raw data from fish census can be found at: https://www.vitcrmp.org/data-and-methods. The following metrics are separated by northern USVI (St. Thomas & St. John) and St. Croix due to the differing assemblages of fish present at these locations. These dissimilarities are primarily driven differences in topography across the islands: St. Croix's shelf is much shorter and has a steeper edge, resulting in less fish habitat compared to the abundant Orbicellid banks in the northern UVSI (Smith et al. 2019). This limited habitat, combined more intense fishing pressure, has likely led to greater depletion of commercially important species (Kadison et al. 2017).

Abundance and Biomass

Northern USVI: The fish transect surveys conducted in 2023 on the 18 sites in the northern USVI's documented 144 species and 40 families. A total of 37,108 individuals were recorded with a mean density of 229 ± 32 fish $100m^{-2}$ across all sites. The biomass of fish surveyed across all sites totaled 3,590 kg with a mean of 22 ± 5 kg $100m^{-2}$ at each site.

Fish diversity was highest in offshore sites, with 115 species noted across 11 sites. In 2023, fish density was very similar across nearshore and offshore sites, with an average of 233 ± 27 $100m^{-2}$ fish seen at each site. Mesophotic sites had a slightly lower average density, with only 215 ± 43 individuals $100m^{-2}$ observed. Conversely, biomass was much higher in mesophotic sites (64 ± 36 kg $100m^{-2}$), whereas biomass in nearshore and offshore environments averaged only 11 ± 1 kg $100m^{-2}$. This large discrepancy was primarily driven by occurrence of schools of larger pelagic fishes on shelf edge sites and by large serranids and lutjanids observed at one site (Grammanik Tiger FSA) as sampling occurred during peak cubera snapper (*Lutjanus cyanopterus*) spawning (August; Biggs and Nemeth, 2016). Average biomass observed in each transect at Grammanik was 170 ± 32.5 kg $100m^{-2}$, thirteen times higher than the average biomass across all other sites (13 ± 0.1 kg $100m^{-2}$).

St. Croix: The fish transect surveys conducted in 2023 on the 15 sites in St. Croix documented 142 species and 43 families. A total of 25,926 individuals were recorded with a mean abundance across all sites of 192 ± 25 fish $100m^{-2}$. The calculated biomass of fish surveyed across all sites totaled 1,182 kg with a mean of 9 ± 3 kg $100m^{-2}$ at each site. This is much lower than the estimated biomass in northern USVI (22 ± 5 kg 100m^{-2}), likely due to higher fishing pressure removing more of the large-bodied species (Kadison et al. 2017).

In St. Croix, fish diversity was lowest in mesophotic sites, and higher in nearshore and offshore sites, which both observed 107 unique species. Fish abundance per site ranged from ~330 fish per transect (Cane Bay) to ~100 fish per transect (Salt River Deep). Overall, density was highest in nearshore sites (244 ± 32 fish 100m^{-2}) followed by offshore sites (200 ± 26 fish 100m^{-2}). As with northern USVI, mesophotic sites had the lowest abundance, with an average of 131 ± 13 individuals observed per transect. Contrasting high biomass in northern USVI mesophotic sites, deeper sites in St. Croix had the lowest biomass estimates, with only 5.7 ± 1.2 kg 100m^{-2}, indicating smaller fish on average. Nearshore and offshore environments had almost double the biomass of mesophotic sites, with an average of 10.3 ± 2.2 kg 100m^{-2}.

Species Composition

Northern USVI: The most common species observed were striped parrotfish (*Scarus iseri*; 15.7% of total fish observed), bluehead wrasse (*Thalassoma bifasciatum*; 12.9%), blue chromis (*Chromis cyanea*; 10.8%), bicolor damselfish (*Stegastes partitus*; 7.2%), and creole wrasse (*Clepticus parrae*; 6.0%). These five species comprised 19% of the fish biomass observed in the northern USVI. Other species contributed significantly to overall biomass in St. Thomas and St. John include stoplight parrotfish (*Sparisoma viride*; 5.7% of total biomass), redband parrotfish (*Sparisoma aurofrenatum*; 5.0%), and cubera snapper (*Lutjanus cyanopterus*; 4.9%) – most of which were all observed at Grammanik.

St. Croix: The five most abundant species observed in St. Croix were the same five as in the northern USVI in 2023, however their relative distributions varied from the northern USVI's. Blue chromis (*Chromis cyanea*; 17.7% of total fish observed) was most common, followed by bluehead wrasse (*Thalassoma bifasciatum*; 17.1%), bicolor damselfish (*Stegastes partitus*; 10.1%), creole wrasse (*Clepticus parrae*; 5.8%), and striped parrotfish (*Scarus iseri*; 5.5%). These five species comprised 16.3% of the fish biomass observed in St. Croix. Other significant contributions to St. Croix's biomass came from the black durgeon (*Melichthys niger*; 5.8% of total biomass), redband parrotfish (*Sparisoma aurofrenatum*;

5.0%), blue tang (*Acanthurus coeruleus*; 5.0%), and ocean surgeonfish (*Acanthurus bahianus*; 4.2%).

Table 3. The 2023 species richness for belt transects and roving diver surveys (RDS).

		Belt Transects (25x4)		RDS
		Total Number of Species	Mean species per transect (±SE)	Total Number of Species
Nearshore	Cane Bay	68	26.3±1.4	51
	Great Pond	38	16.8±1.2	46
	Jacks Bay	58	20.3±1.2	54
	Castle	52	23.8±1.2	48
	Salt River West	57	21.8±2.0	48
	Sprat Hole	55	24.2±0.7	62
	Coculus Rock	54	23.6±1.1	57
	Black Point	55	22.7±1.2	60
	Brewers Bay	46	20.1±1.3	43
	Botany Bay	56	22.6±1.1	56
	Coral Bay	41	19.0±0.9	42
	Fish Bay	65	22.4±2.7	60
	Magens Bay	53	23.6±1.5	52
Offshore	Eagle Ray	53	21.3±1.5	43
	Buck Island, St. Croix	59	23.2±1.3	51
	Kings Corner	61	26..2±1.4	57
	Mutton Snapper FSA	55	22.4±1.2	56
	Buck Island, St. Thomas	68	28.1±0.9	61
	Seahorse Cottage Shoal	72	26.4±1.1	52
	South Capella	66	24.0±1.5	52
	South Water	53	22.6±1.3	58
	Flat Cay	53	22.6±1.2	59
	Savana Island	64	27.9±1.9	58
	St. James	56	23.2±2	46
Mesophotic	Buck Island STX Deep	49	18.7±1.4	44
	Cane Bay Deep	49	14.9±0.9	39
	Lang Bank EEMP	62	22.4±1.1	55
	Lang Bank Red Hind FSA	53	18.9±1.7	49
	Salt River Deep	53	17.1±1.2	48
	College Shoal East	52	21.9±0.9	43
	Ginsburg's Fringe	-	-	-
	Grammanik Tiger FSA	61	23.4±1.2	52
	Hind Bank East FSA	62	25.4±1.0	59
	Meri Shoal	47	18.8±0.9	43

Fish Abundance

Total yearly fish abundances across nearshore, offshore, and mesophotic sites are shown in Figures 16 and 17. As in previous years, total fish abundance showed high variability across sites and strata. In deeper reefs, these changes can be attributed to the presence or absence of the prolific small pelagic fishes, (most often the creole wrasse, *Clepticus parrae*) that vary annually and seasonally. Total fish abundance in the northern USVI was similar to the previous year (~36,000 individuals) however St. Croix total abundance was lower in 2023 than 2022 (25,656 vs. 29,280 fish respectively).

Sites with the highest overall fish abundance in 2023 were Botany Bay, Sprat Hole, and Cane Bay. High fish abundance at these sites was influenced by large schools of small bluehead wrasse (*T. bifasciatum),* blue and brown chromis (*Chromis cyanea* and *Chromis multilineata*), and striped parrotfish (*S. iseri)* observed during the surveys. The sites with the lowest fish abundance were Salt River Deep and Cane Bay Deep. These mesophotic wall sites, defined by agariciid corals and high silt loads, continue to also have low species richness values and low biomass each year during sampling (Fig. 15).

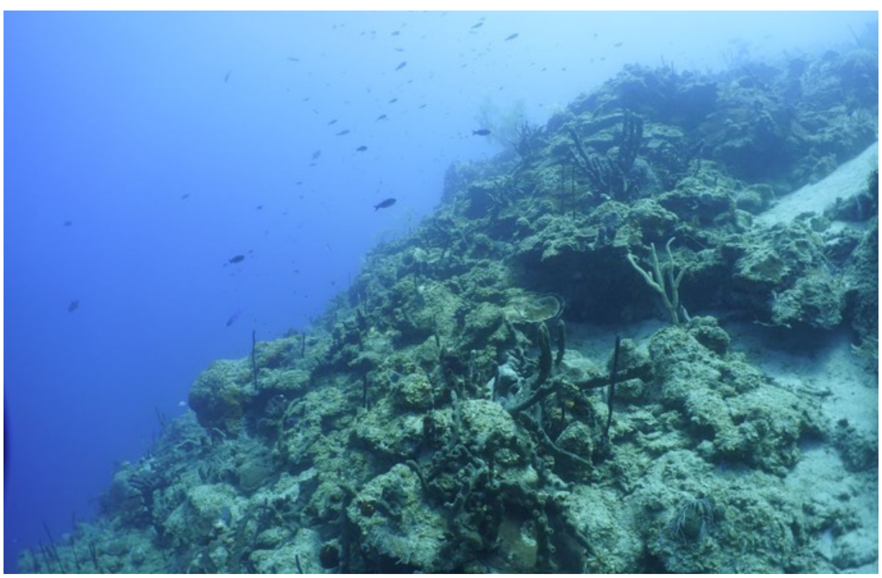

Figure 15. A representative photo of the mesophotic wall sites on St. Croix, characterized by sloping walls, high silt loads, *Agarica* spp. coverage, and low fish abundance, biomass, and richness (photo credit: L. Henderson).

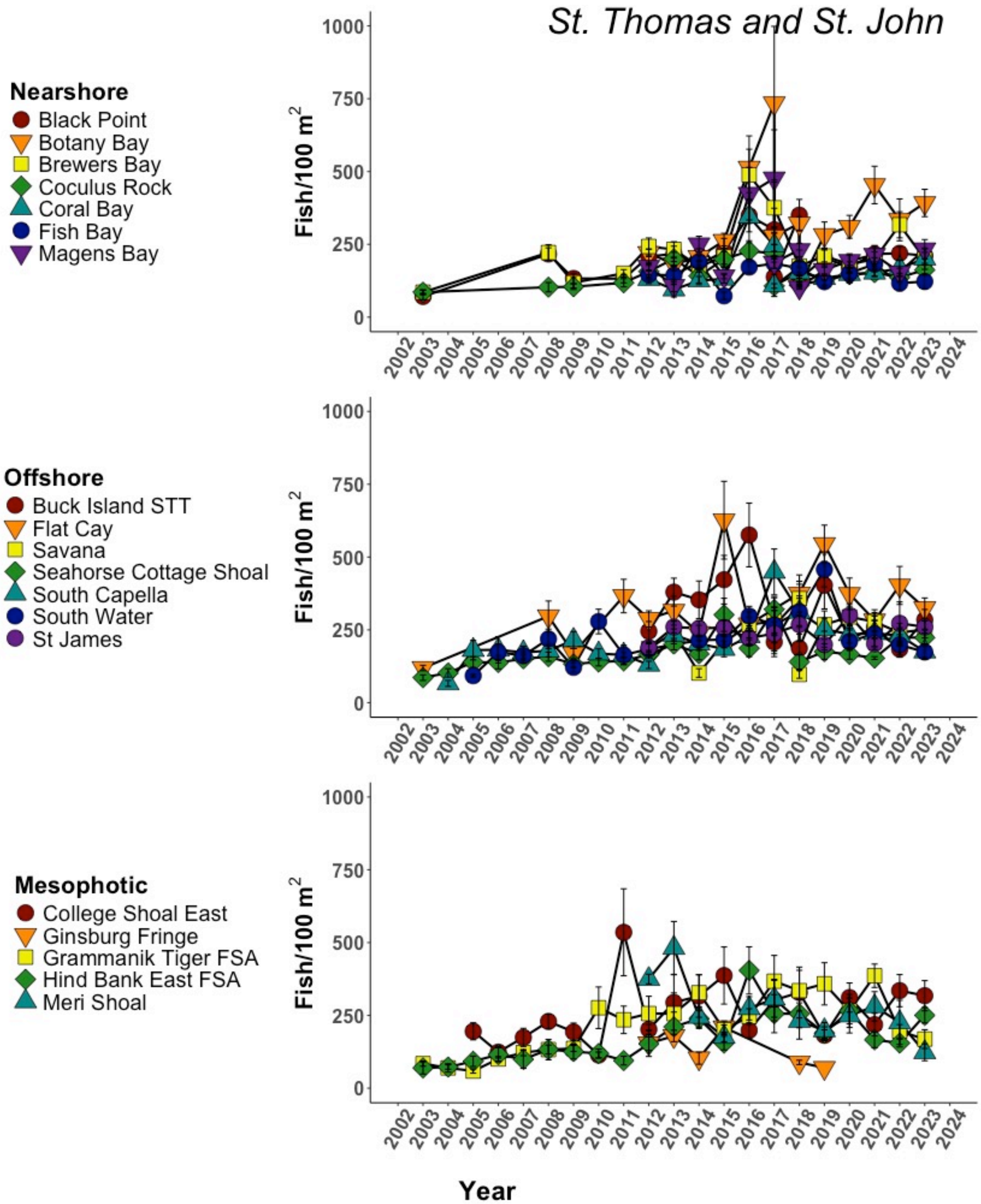

Figure 16. Fish abundance (±SE) across St. Thomas and St. John TCRMP monitoring sites from 2003-2023.

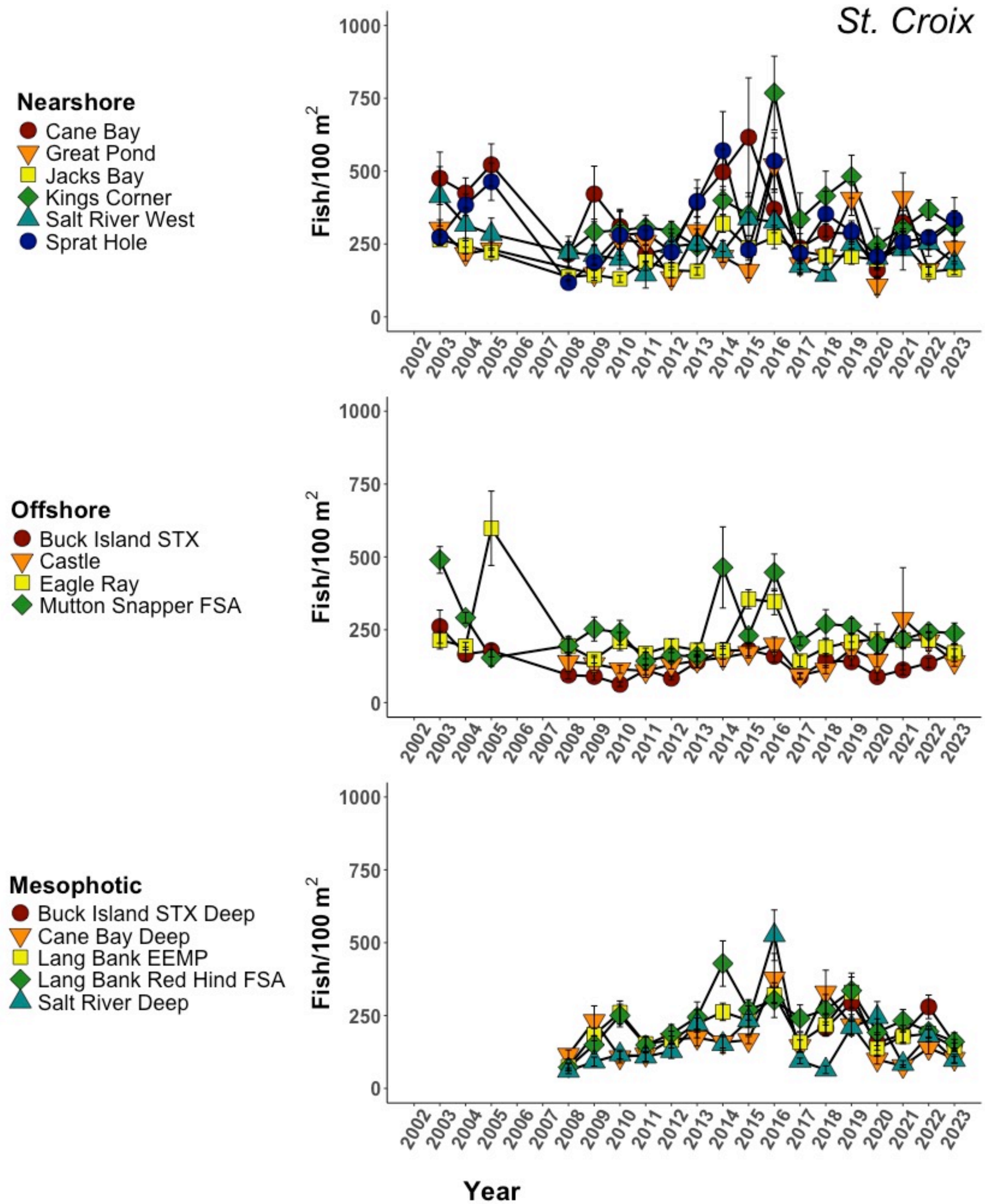

Figure 17. Fish abundance (±SE) across St. Croix TCRMP monitoring sites from 2003-2023.

Fish Biomass

Total fish biomass for all sites and years is shown in Figures 19 and 20. As with abundance, biomass was highly variable across strata, sites, and years. No temporal pattern is obvious, and differences appear to be seasonal or natural variation. Overall average biomass was higher in 2023 than in 2022, however this was almost entirely driven by the very high (>150 kg 100m^{-2}) biomass observed at Grammanik Tiger FSA. Sampling this year occurred during cubera snapper (*Lutjanus cyanopterus*) spawning (August; Biggs and Nemeth, 2016) and biomass this species accounted for 89% of the total biomass estimated at Grammanik. Other sites with high biomass included the two other sites on the St. Thomas' shelf edge (College Shoal East and Hind Bank East FSA), as well as King's Corner, which regularly boasts high fish biomass, abundance, and diversity each year. All three mesophotic sites on the northern VI shelf edge vary seasonally due to the annual or seasonal presence/absence of large pelagic fishes which serve as the primary influence on biomass values. The same influx of spawning large pelagic fishes has not been noted on any of the mesophotic St. Croix sites, although they are usually sampled a couple months later, outside of spawning season for these species. In fact, the sites with two lowest biomass values in 2023 were Cane Bay Deep and Buck Island STX Deep. Jack's Bay and Eagle Ray also had notably lower biomass values; all four of these sites are in St. Croix.

Figure 18. Nassau grouper (*Epinephelus striatus)* observed while sampling at Grammanik Tiger FSA (photo credit: N.Krampitz).

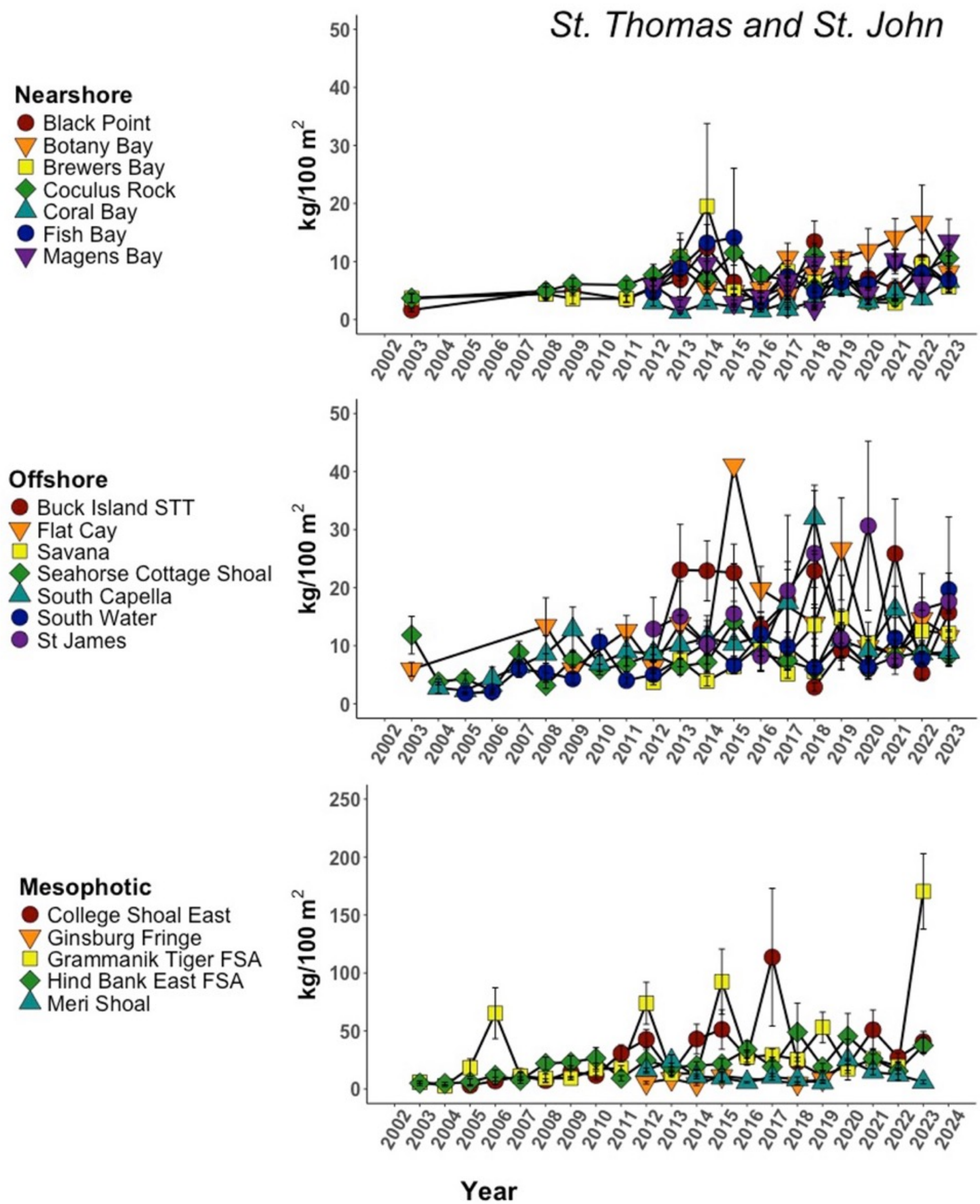

Figure 19. Mean fish biomass (±SE) across St. Thomas and St. John TCRMP monitoring sites from 2003-2023.

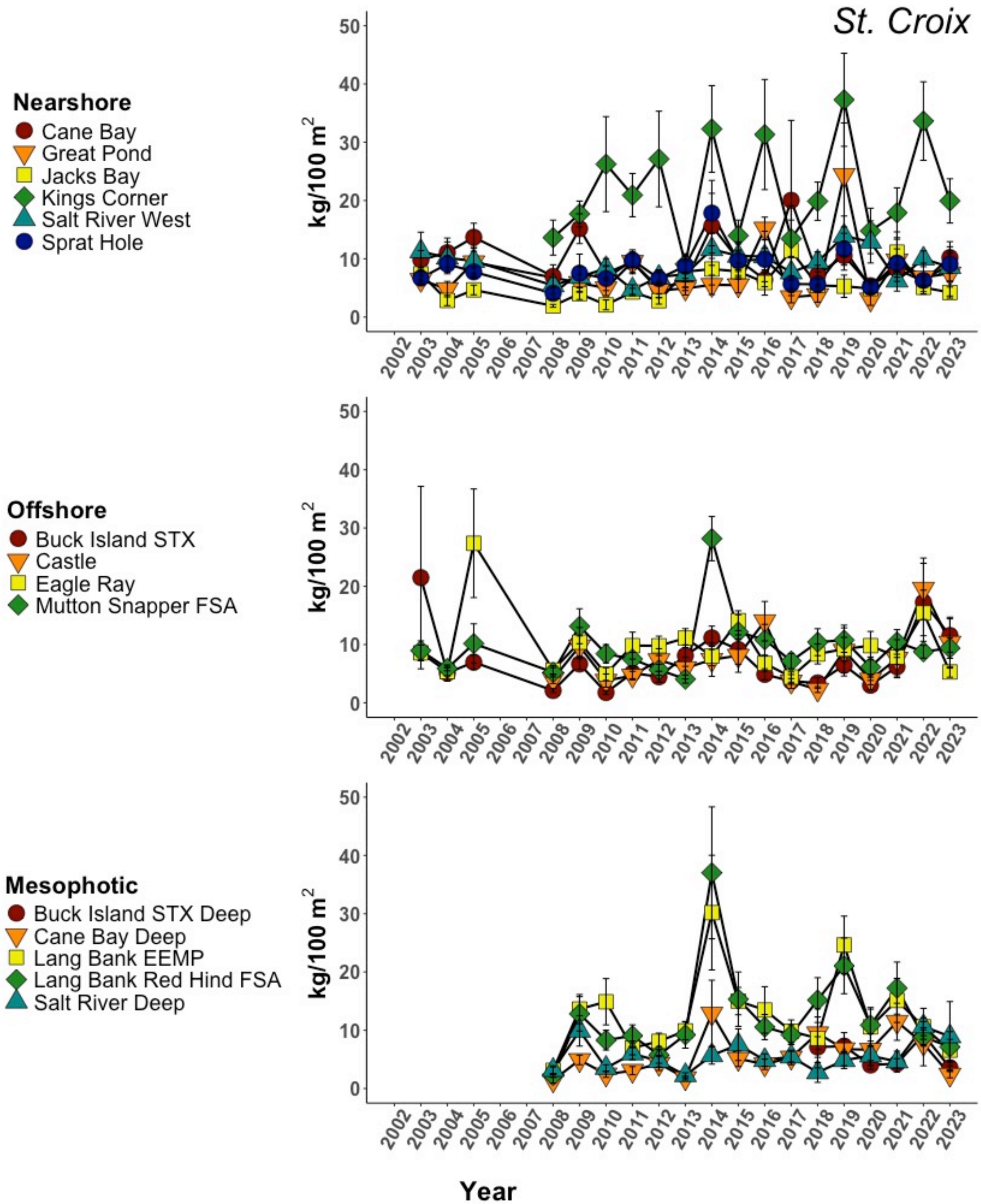

Figure 20. Mean fish biomass (±SE) across St. Croix TCRMP monitoring sites from 2003-2023.

BLACK SPINED SEA URCHIN *DIADEMA ANTILLARUM*

In general, the shallowest sites, e.g., Great Pond and Coculus Rock, support the greatest abundance of *Diadema antillarum,* hereafter referred to as *Diadema* (Fig. 21). 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.

The overall abundance of *Diadema* at TCRMP sites has remained somewhat variable over the years but has generally increased since 2010 (Fig. 22). Exceptions to this are major declines seen in 2017, following due to two Category 5 hurricanes, and 2022, following a mass mortality event (Hewson et al. 2023). Despite significant declines in *Diadema* density in early 2022, many juvenile *Diadema* began to reappear on many reefs by the end of the year, suggesting population numbers are growing instead of declining further. Where they occurred, overall density of *Diadema* increased from 0.23 ± 0.15 urchins/100m^{-2} in 2022 to 1.00 ± 0.68 urchins/100m^{-2} in 2023. However, *Diadema* were only observed at 7 TCRMP locations in 2023 (vs. 15 in 2021); and of those urchins that were observed, most were small or juveniles, whose functionality as a key herbivore in controlling algal growth would be limited.

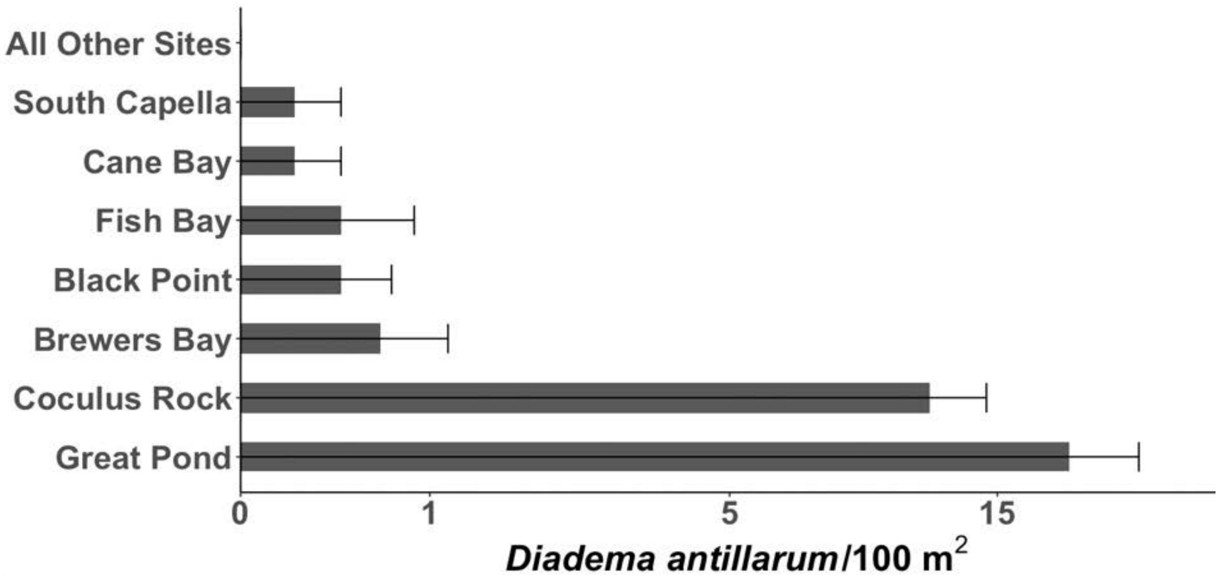

Figure 21. Average density (±SE) of the black spined sea urchin (*Diadema antillarum*) at 33 TCRMP monitoring sites in 2023. Note the log scale

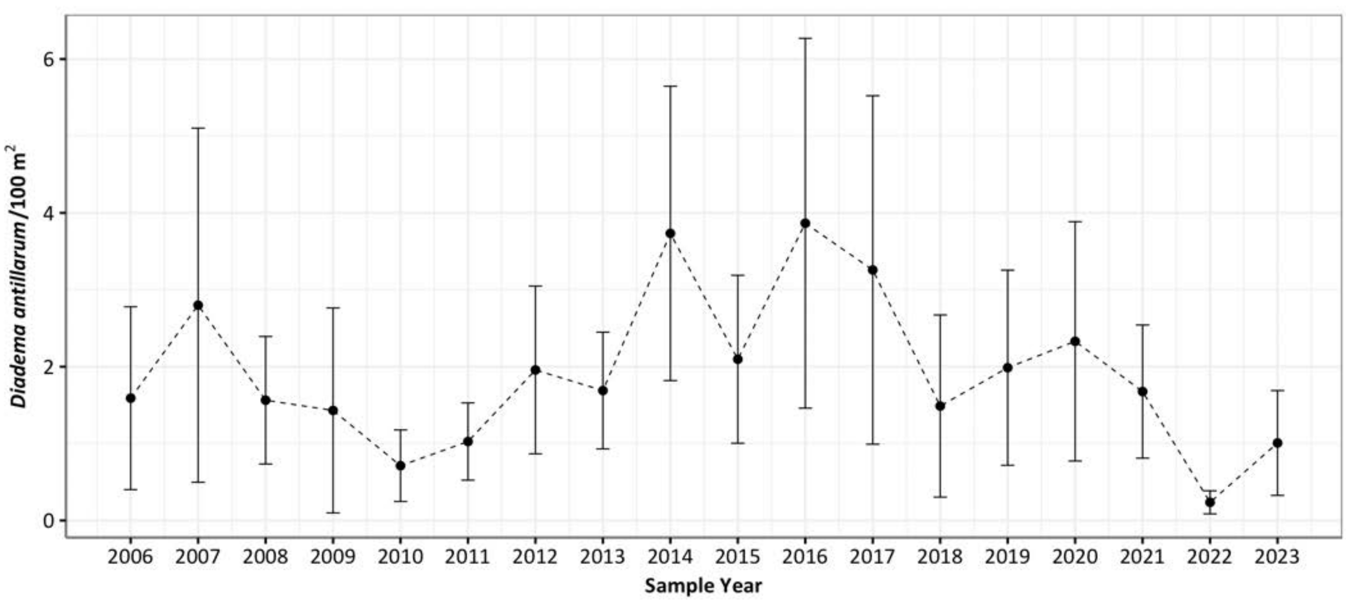

Figure 22. Average density (±SE) of *Diadema antillarum* at TCRMP sites from 2012-2023.