TCRMP SUMMARY

Territorial Coral Reef Monitoring Summary

BENTHIC COMMUNITIES AND CORAL REEF HEALTH

Benthic cover was monitored at 34 monitoring sites and coral health was monitored at 33 sites in fall 2024 and a post bleaching season in spring 2025. Raw data for benthic coverage, coral health, algae heights, and benthic temperature can be found at https://www.vitcrmp.org/data-and-methods .

Coral Cover

TCRMP sites are affected by a myriad of stressors, both stochastic and chronic, that drive changes in coral cover. The most impactful events include bleaching from thermal stress events and disease outbreaks. During the bleaching event in 2005, hard coral cover declined at most sites and subsequent recovery was limited, particularly at mesophotic depths. In contrast, coral cover losses associated with the 2010, 2012, and 2019 bleaching events were less pronounced. Direct impacts of these recent bleaching events are likely obscured by overall chronic decline of coverage regardless of the temperature conditions. Finally, responses to the consecutive bleaching events in 2023 and 2024 were substantial but highly variable across TCRMP sites, highlighting differences in site-level vulnerability and recovery potential.

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.

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 (Figure 12; Figure 13). 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 contribute to increases in coral cover.

The deepest site, Ginsburgs Fringe, has lost site lost 89% of its coral cover in the past decade, in what appears to be a steady continuous decline (Figure 12). Multiple co-occurring stressors are apparent at this TCRMP site, including invasive lionfish and high cover of the macroalgae *Lobophora variegata*. 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 (Eckrich and Engel 2013; Hollister et al. 2021).

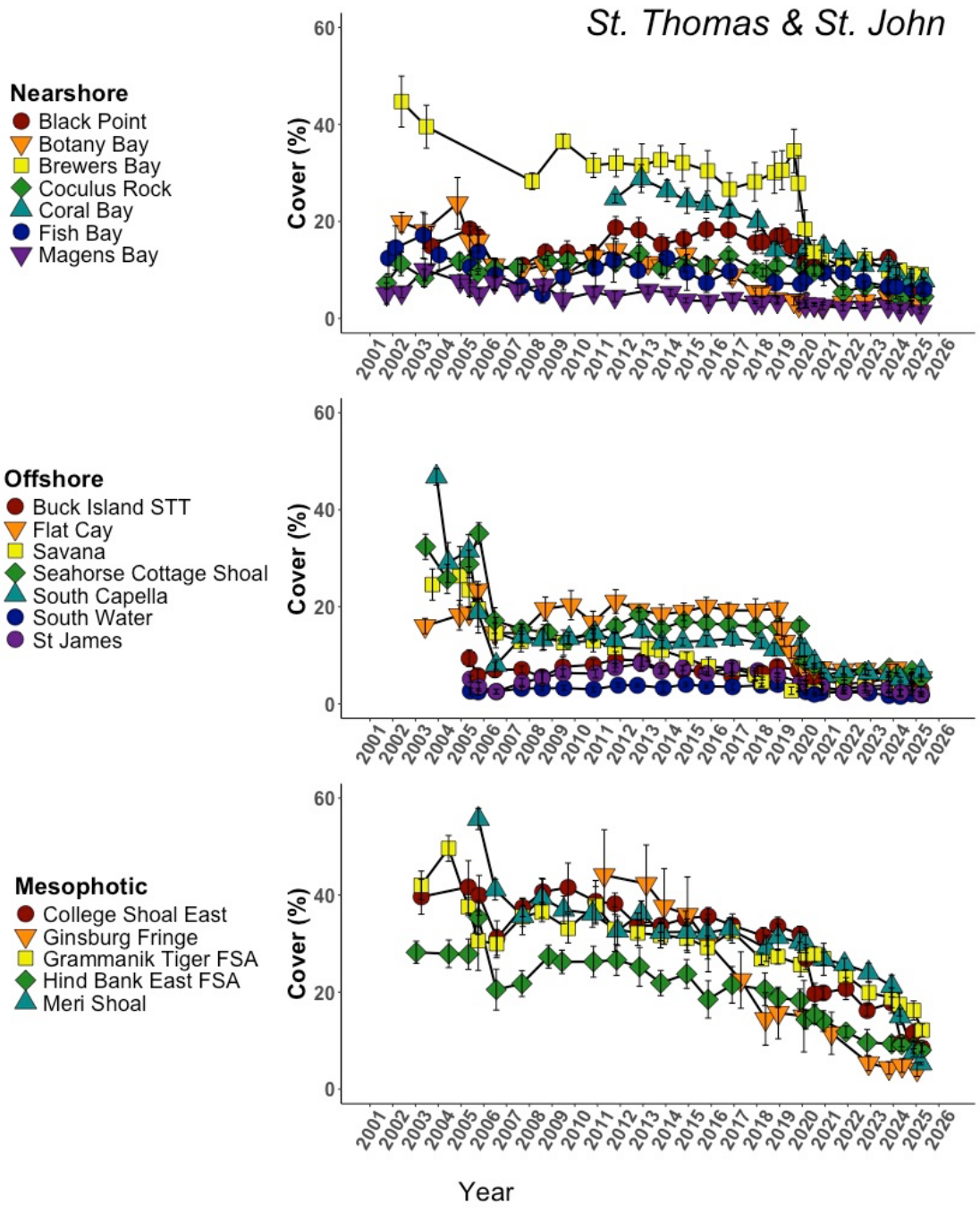

Figure 12. Coral cover (±SE) across St. Thomas and St. John TCRMP monitoring sites from 2002 – spring 2025.

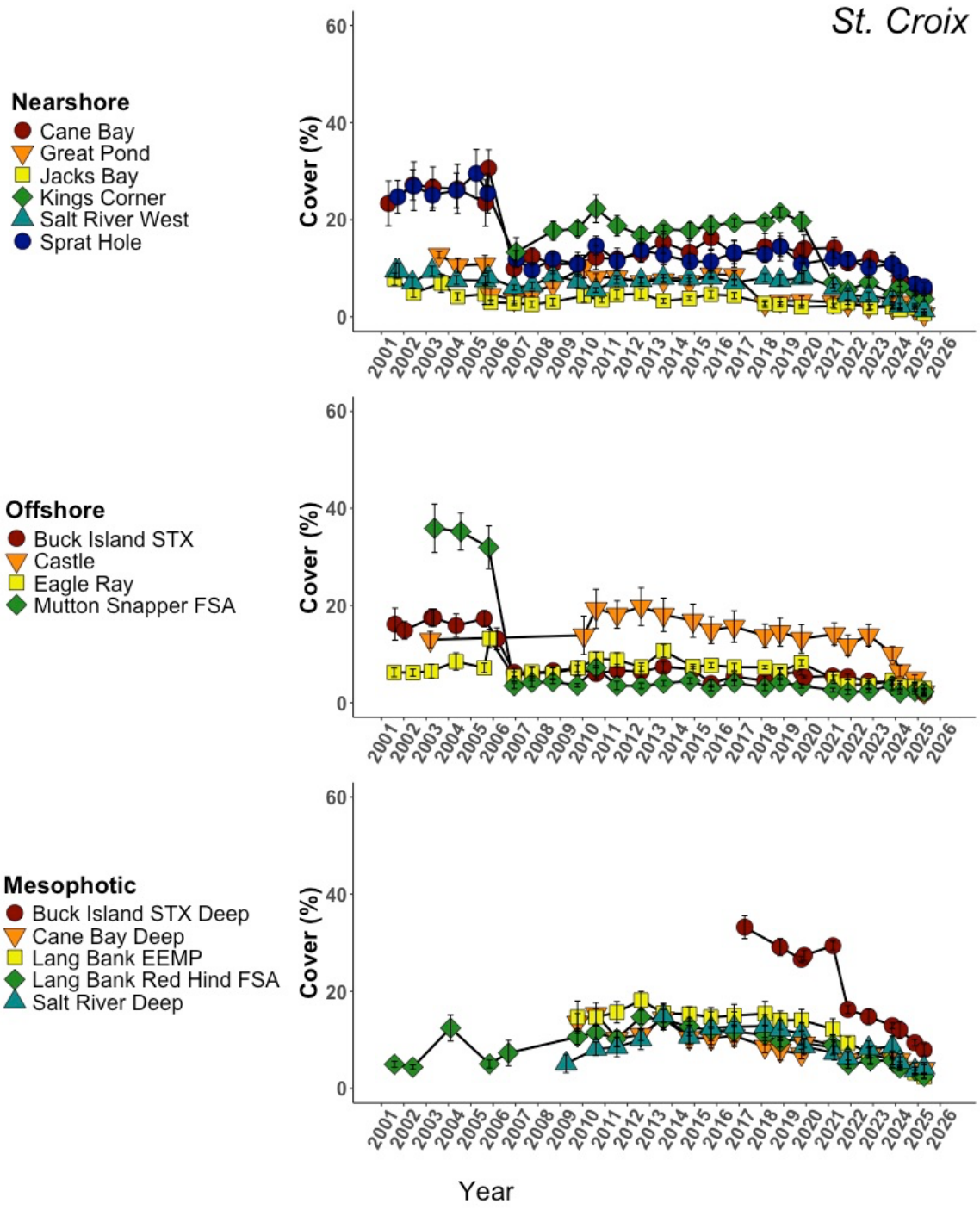

Figure 13. Coral cover (±SE) across St. Croix TCRMP monitoring sites from 2001 – spring of 2025.

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 (Figure 14). 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 nearshore sites, surpassing all other macroalgal coverage in most years (Figure 14, Figure 15).

Macroalgal Cover

Macroalgal cover has been increasing at most TCRMP sites, particularly where coral cover has declined (Figure 14). 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 a large increase in macroalgal cover was due to the expansion of the encrusting red alga *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. Recent research on *R. textilis* indicates that this invasive encrusting red alga competitively overgrows coral species (Hollister et al. 2021) and remineralizes coral tissue (Gretta 2025), although some coral species can resist or counteract this overgrowth.

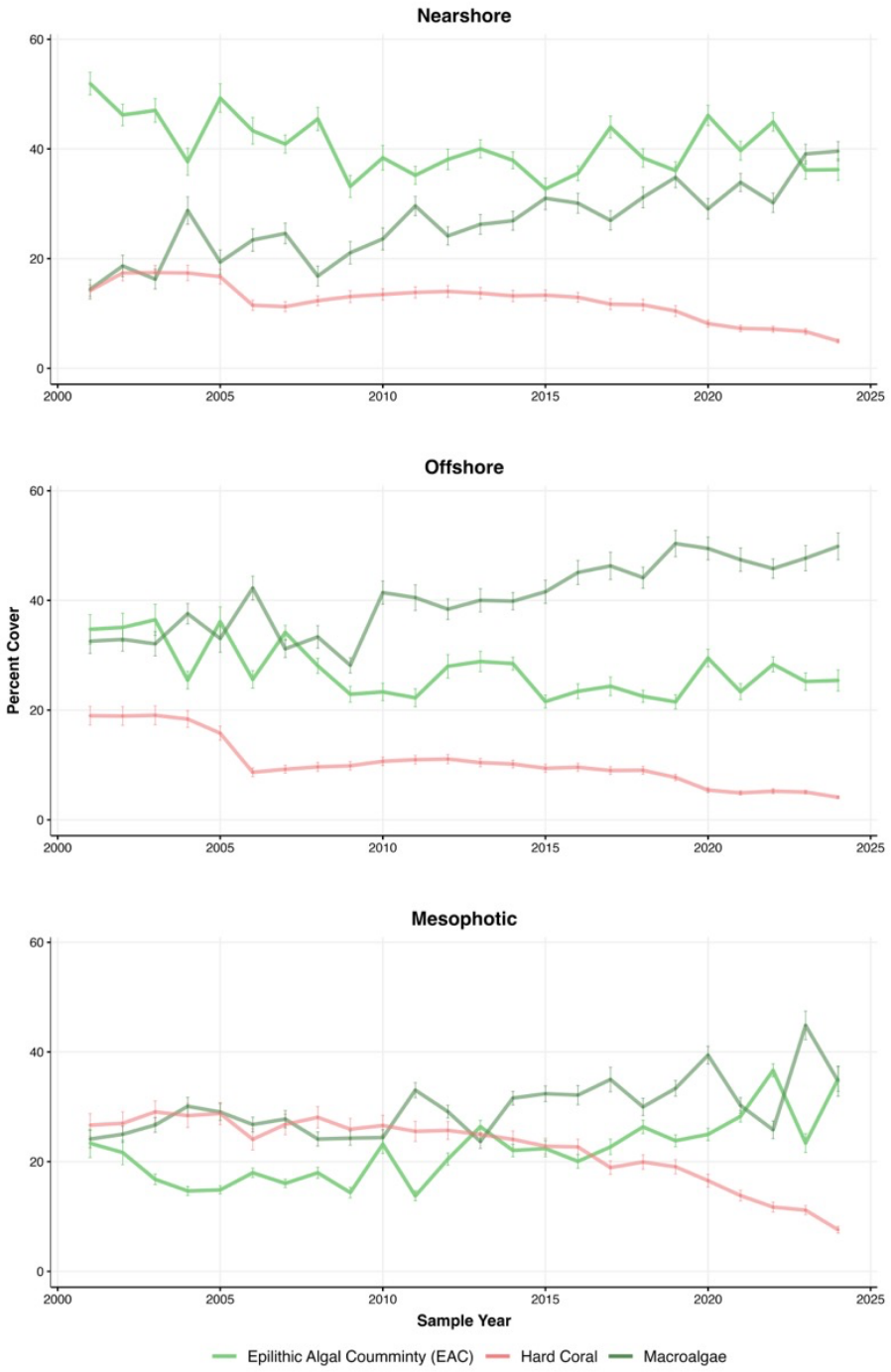

Figure 14. Mean hard coral, macroalgal (encrusting and erect), and epilithic algal community (EAC) coverage (±SEM) across all TCRMP sites during sampling from 2002-2024.
Data is split into mesophotic (≥27 m), offshore (7-24 meters; ≥2 km offshore), and nearshore (6-17 meters; <2 km offshore).

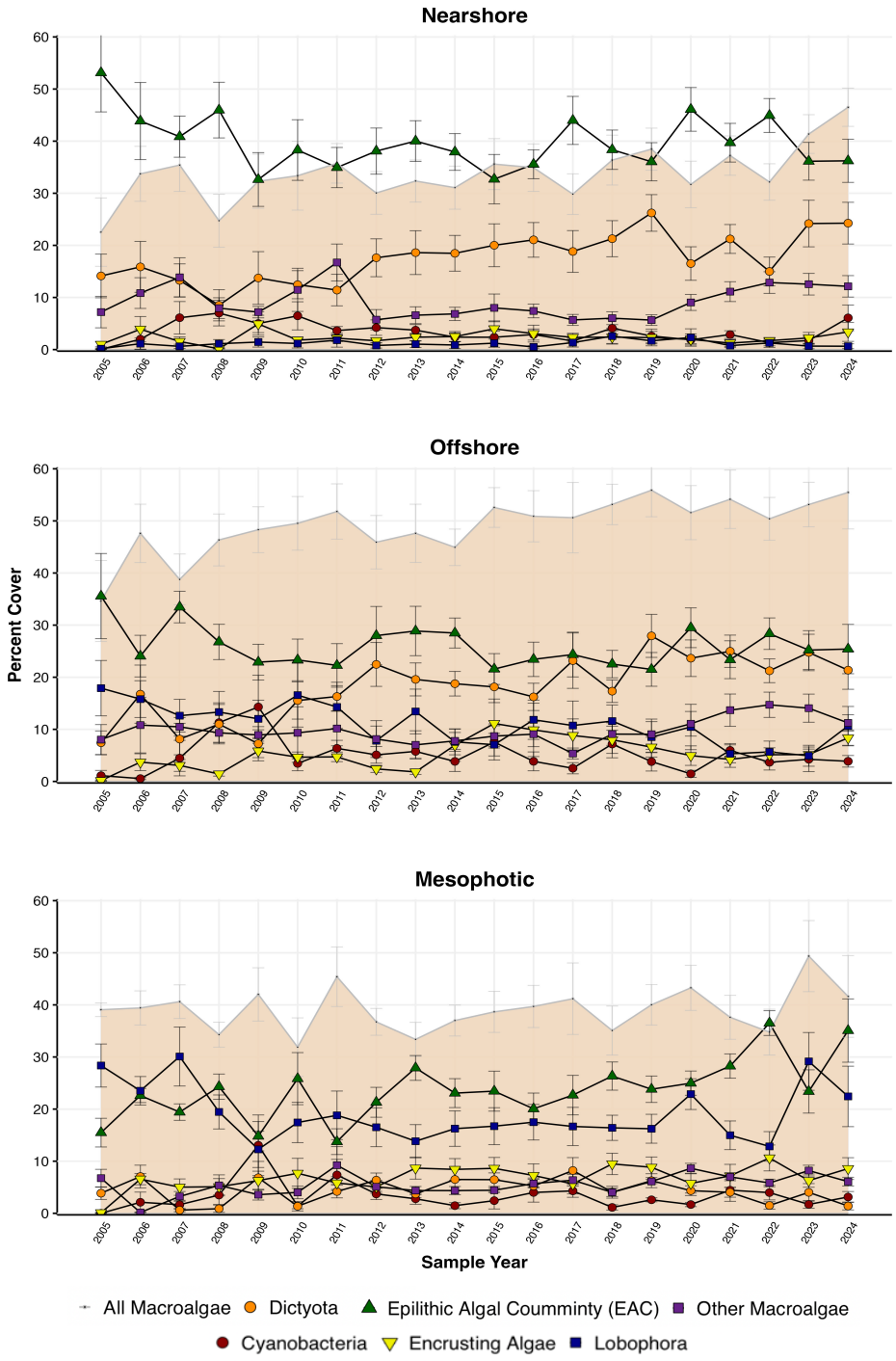

Figure 15. Mean algal coverage (±SEM) across TCRMP monitoring sites from 2005-spring of 2024. All macroalgae (grey line/orange shading) includes all erect and encrusting algae combined, as well as cyanobacteria. Other macroalgae (purple) includes all erect macroalgae except *Dictyota* (orange) and *Lobophora* (blue)

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 (

Figure 15). 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 and juvenile 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.

Gorgonian and Antipatharian Cover

The cover of gorgonians and antipatharians has been consistent or increasing at most monitoring sites throughout the years of monitoring (Figure 16). 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 and 2024 (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 (Figure 16). 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).

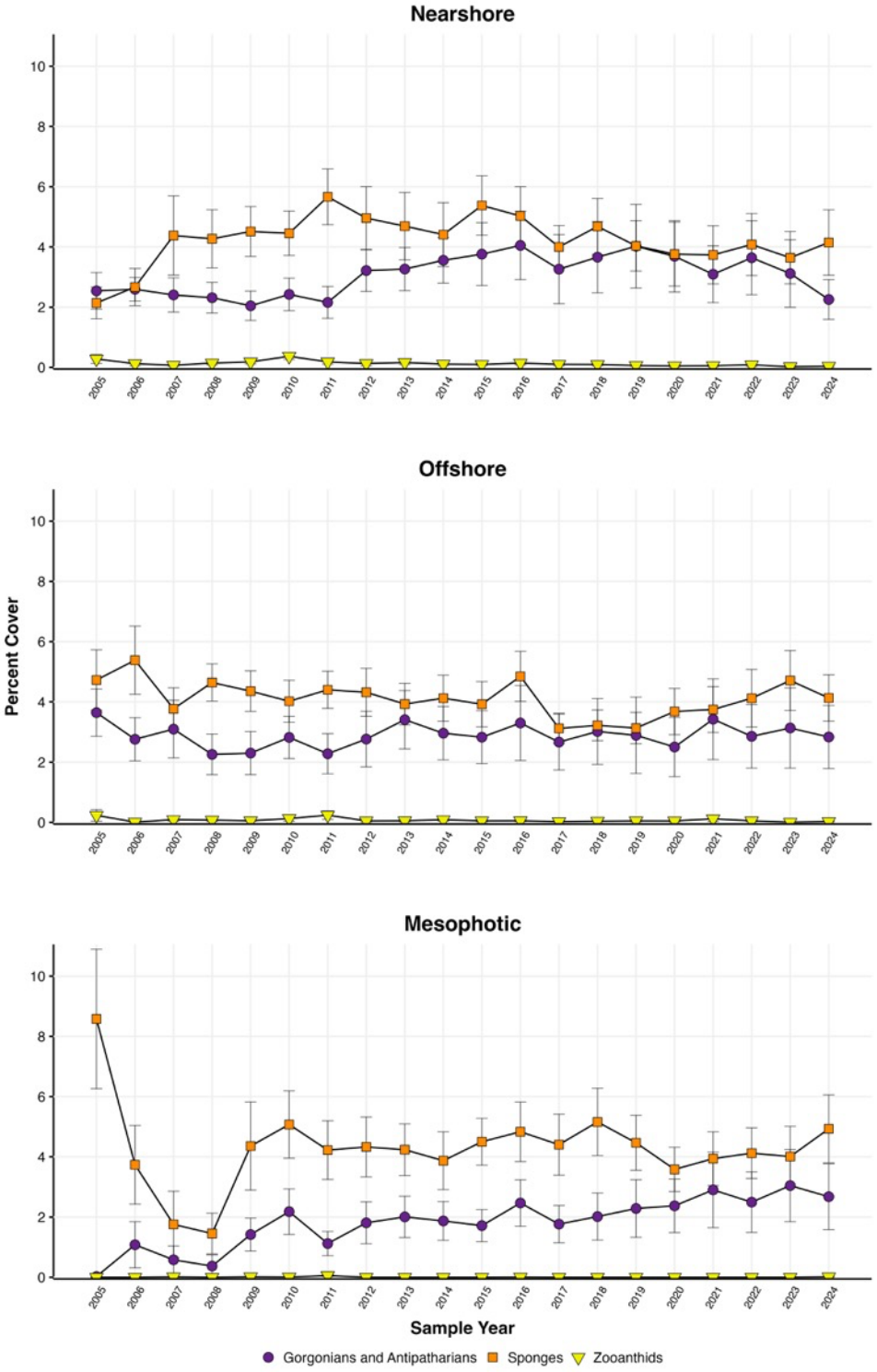

Figure 16. Mean benthic cover (±SEM) of gorgonians, sponges, and zooanthids from 2005-2024 at TCRMP locations.

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 by 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 with more intense fishing pressure, has likely led to greater depletion of commercially important species on St. Croix (Kadison et al. 2017).

Northern USVI and St Croix Abundance and Biomass

Northern USVI: Transect surveys conducted in 2024 on 18 northern USVI locations documented 141 fish species across 39 different families. A total of 37,905 individuals were recorded with a mean density of 234 ± 12.6 fish $100m^{-2}$ across all sites. The biomass of fish surveyed across all sites totaled 1,920 kg with a mean of 12 ± 3 kg $100m^{-2}$ at each site.

As shown in Table 5, fish diversity and abundance was greatest in the offshore sites, with 114 species documented and a mean density of 253 ± 21 fish $100m^{-2}$ across the belt transects. Mesophotic sites supported substantially higher biomass, averaging 27 ± 7 kg per 100 m^2, which is more than double the biomass observed at either nearshore (6 ± 1 kg per 100 m^2) or offshore (9 ± 1 kg per 100 m^2) sites and is line with previous analyses of TCRMP data (Kadison et al. 2019) and randomized surveys across the northern USVI shelves (Heidmann et al. 2024).

The top five biomass-contributing families of the mesophotic were Lutjanidae (snappers; 38.7%), Carangidae (jacks, scad, pompanos, runners; 10.4%), Carcharhinidae (sharks; 9.6%), Labridae (wrasses; 8.3%), and Scaridae (parrotfish; 5.8%). Offshore reef biomass was dominated by Lutjanidae (16.0%), Haemulidae (grunts; 15.7%), Scaridae (14.7%), Acanthuridae (surgeonfish and tangs; 11.5%), and Pomacentridae (damselfish; 7.6%). Finally, nearshore reefs supported biomass largely of Scaridae (33.7%),

Pomacentridae (12.6%), Carangidae (12.3%), Lutjanidae (10.1%), and Acanthuridae (9.8%).

Table 5: Northern USVI (St Thomas and St John) fish belt data across habitat zones nearshore, offshore, and mesophotic

	Number of species	Number of families	Total abundance	Density (fish $100m^{-2}$)	Total biomass (kg)	Average biomass density (kg $100m^{-2}$)
Nearshore	100	31	14133	224±23	382	6±1
Offshore	114	33	15965	253±21	563	9±1
Mesophotic	104	31	7807	216±14	976	27±7

St. Croix: The fish transect surveys conducted in 2024 on the 15 sites in St. Croix documented 126 species across 36 different families. A total of 23,992 individuals were recorded with a mean abundance across all sites of 178 ± 21 fish $100m^{-2}$. The calculated biomass of fish surveyed across all sites totaled 1,803 kg with a mean of 13.4 ± 3 kg $100m^{-2}$ at each site.

In St. Croix, fish diversity was comparable across nearshore, offshore, and mesophotic reefs, with 94 to 97 species and 28 to 31 families observed (Table 6). Mesophotic sites had the lowest fish abundance and density along transects, but as in the northern USVI sites, supported the greatest average biomass across the three reef habitats. However, mesophotic reefs in St Croix had lower biomass than mesophotic reefs in St Thomas and St John, indicating that fish assemblages at depth in St Croix were comprised of smaller fish.

Based on biomass metrics, St Croix mesophotic reef fish assemblages are comprised of Carcharhinidae (54.2%), Carangidae (18.0%), Labridae (4.7%), Balistidae (triggerfish; 4.6%), and Scaridae (4.0%). Offshore reefs supported Carcharhinidae (19.4%), Carangidae (14.0%), Holocentridae (squirrelfish 9.5%), Acanthuridae (8.8%), and Acanthuridae

(8.8%). Finally, nearshore St Croix reefs supported Scaridae (25.8%), Acanthuridae (17.4%), Carangidae (10.3%), Carcharhinidae (8.8%), and Ginglymostomatidae (8.3%).

Contrary to the northern USVI, offshore and nearshore reefs supported a higher biomass of the higher trophic level fishes, such as sharks and jacks. This was likely driven by the short shelf edge in St Croix, where nearshore reefs are not as geographically separated from the mesophotic reefs, as seen in Cane Bay shallow and Cane Bay Deep, and Salt River West and Salt River Deep. Furthermore, likely due to the higher fishing pressure in St Croix, the density and average biomass density in mesophotic reefs was considerably lower in St Croix than in St Thomas, supporting findings of Kadison et al. 2017.

Table 6: St Croix fish belt data across habitat zones nearshore, offshore, and mesophotic

	Number of species	Number of families	Total abundance	Density (fish/ $100m^{-2}$)	Total biomass (kg)	Average biomass density (kg $100m^{-2}$)
Nearshore	95	30	10862	201±40	475	9±3
Offshore	97	31	7185	200±37	430	12±4
Mesophotic	94	28	5945	132±23	898	20±5

Species Composition

Northern USVI: Based on abundance metrics, the most common species observed were striped parrotfish (Scarus iseri; 12.9% of total fish observed), bluehead wrasse (Thalassoma bifasciatum; 12.7%), blue chromis (Chromis cyanea; 9.2%), creole wrasse (Clepticus parrae; 8.4%), and bicolor damselfish (Stegastes partitus; 6.8%). However, while most abundant, these five species only comprised 10.2% of the fish biomass observed in the northern USVI. Other species that contributed significantly to overall biomass in St. Thomas and St. John include cubera snapper (Lutjanus cyanopterus; 11.8% of total biomass), schoolmaster snapper (Lutjanus apodus; 6.3%), horse-eye jack (Caranx latus; 5.2%), Caribbean reef shark (Carcharhinus perezi; 4.9%), and yellowtail snapper (Ocyurus chrysurus; 4.8%). Apart from yellowtail snapper — which is

abundant across nearshore, offshore, and mesophotic reefs — the four other high biomass contributing species are associated with mesophotic reefs, which explains why biomass is significantly higher in mesophotic zones compared to nearshore and offshore reefs.

St. Croix: The five most abundant species observed in St. Croix were similar to the northern USVI, however, their relative distributions varied. In descending order of relative abundance on St Croix reefs: bicolor damselfish (14.6% of total fish observed), bluehead wrasse (14.6%), blue chromis (12.5%), creole wrasse (7.9%), and bar jack (*Carangoides ruber*; 5.1%). These top five abundant species only comprised 12.9% of total biomass on St Croix reefs. Instead, similar to the northern USVI, the bulk of biomass came from the Caribbean reef shark (33.9% of total biomass), horse-eye jack (8.8%), and bar jack (5.6%).

Across the USVI, the sites with the greatest fish diversity (number of species observed) were Seahorse Cottage Shoal, Flat Cay, Hind Bank East FSA, Kings Corner, and Savana, all offshore or mesophotic sites (Table 7). The sites with the lowest fish diversity were Coral Bay, Salt River West, Salt River Deep, and Cane Bay Deep, all sites with high sedimentation (Figure 17).

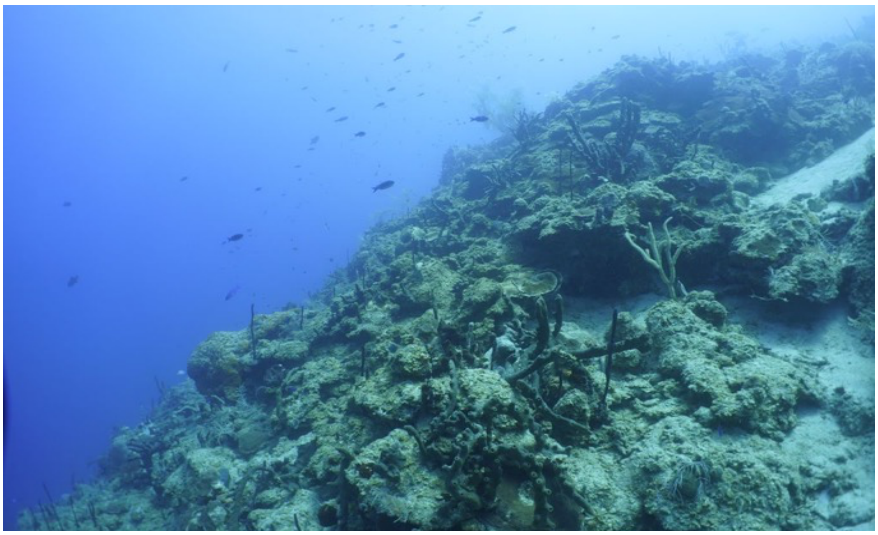

Figure 17. A representative photo of the mesophotic wall sites on St. Croix, characterized by sloping walls, high silt loads, *Agaricia* spp. coverage, and low fish abundance, biomass, and richness (photo credit: L. Henderson).

Table 7: The 2024 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	Black Point	59	22.7±1.2	50
	Botany Bay	59	22.7±1.2	61
	Brewers Bay	53	20.1±0.9	56
	Cane Bay	62	26.4±1.4	42
	Castle	49	23.9±1.3	49
	Coculus Rock	53	23.6±1.1	37
	Coral Bay	38	19.2±0.9	41
	Fish Bay	52	22.4±2.7	49
	Great Pond	47	16.8±1.2	39
	Jacks Bay	54	20.3±1.2	60
	Magens Bay	53	23.9±1.6	44
	Salt River West	39	21.8±1.9	42
	Sprat Hole	50	24.2±0.7	55
Offshore	Buck Island, St. Croix	54	23.3±1.3	52
	Buck Island, St. Thomas	60	28.1±0.9	66
	Eagle Ray	58	21.3±1.5	56
	Flat Cay	66	22.6±1.2	41
	Kings Corner	65	26.3±1.4	50
	Mutton Snapper FSA	51	22.7±1.2	60
	Savana Island	65	27.9±1.9	61
	Seahorse Cottage Shoal	69	26.4±1.1	61
	South Capella	50	24.0±1.5	55
	South Water	51	22.6±1.3	49
	St. James	54	23.4±2.1	55
Mesophotic	Buck Island STX Deep	48	18.7±1.4	53
	Cane Bay Deep	45	14.9±0.9	41
	College Shoal East	54	21.9±0.9	56
	Ginsburg's Fringe	-	-	-
	Grammanik Tiger FSA	62	23.4±1.2	57
	Hind Bank East FSA	65	25.4±1.0	60
	Lang Bank EEMP	61	22.4±1.1	52
	Lang Bank Red Hind FSA	53	18.9±1.7	54
	Meri Shoal	60	18.8±0.9	48
	Salt River Deep	44	17.1±1.2	43

Fish Abundance

Total yearly fish abundances across nearshore, offshore, and mesophotic sites are shown in Figure 18 and Figure 19. 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 slightly higher than the previous year (~38,000 vs. ~36,600 individuals) however St Croix total abundance was lower in 2024 than 2023 (~24,000 vs. ~26,400 fish, respectively).

Sites with the highest overall fish abundances in 2024 were Great Pond (St Croix), Botany Bay (St Thomas), Buck Island St Thomas, Savana (St Thomas), and Kings Corner (St Croix). 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*), bicolor damselfish (*Stegastes partitus*), and striped parrotfish (*S. iseri)* observed during the surveys. The sites with the lowest fish abundances were Salt River Deep (St Croix), Castle (St Croix), Buck Island St Croix, and Cane Bay Deep (St Croix).

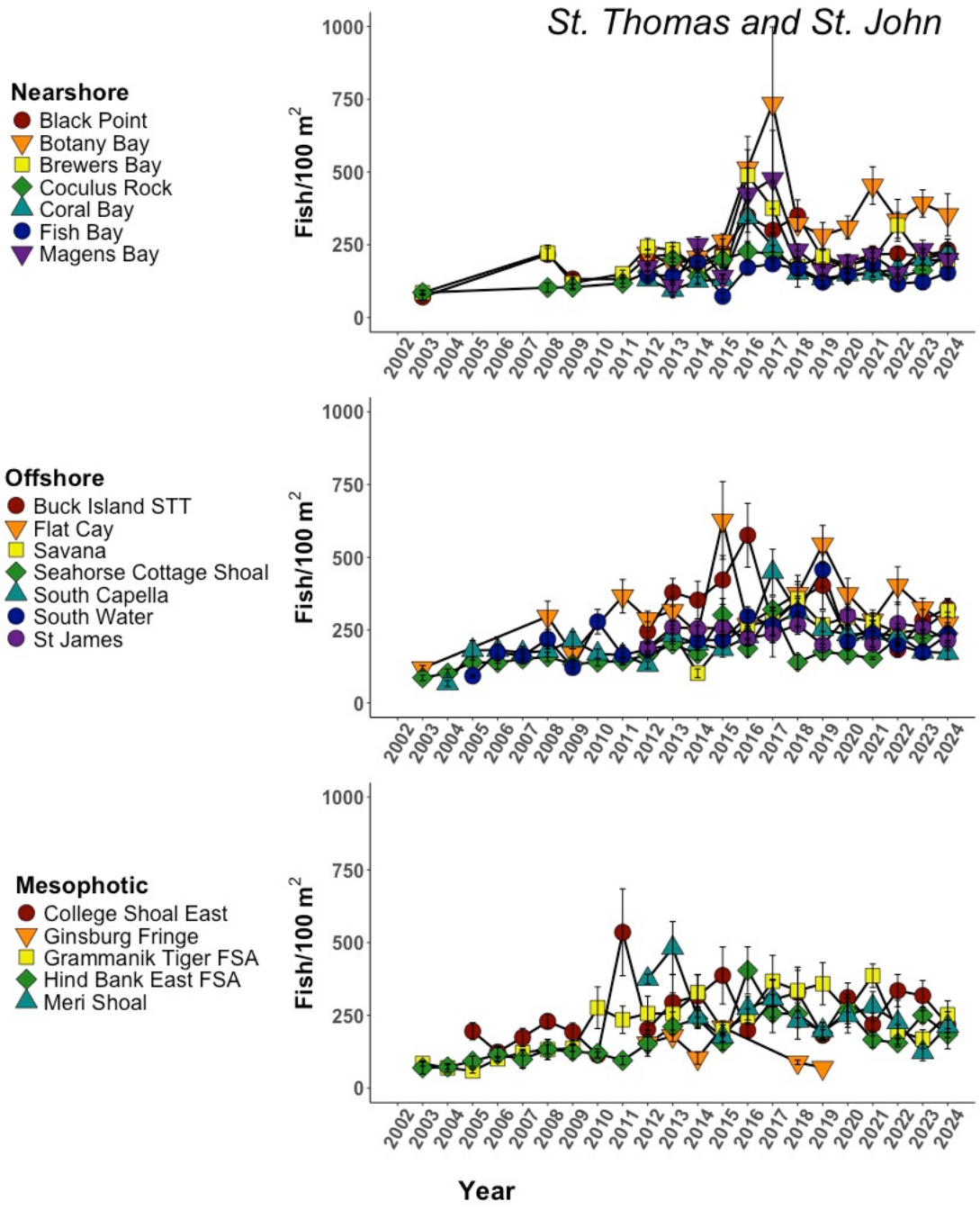

Figure 18. Fish abundance (±SE) across St. Thomas and St. John TCRMP monitoring sites from 2003-2024.

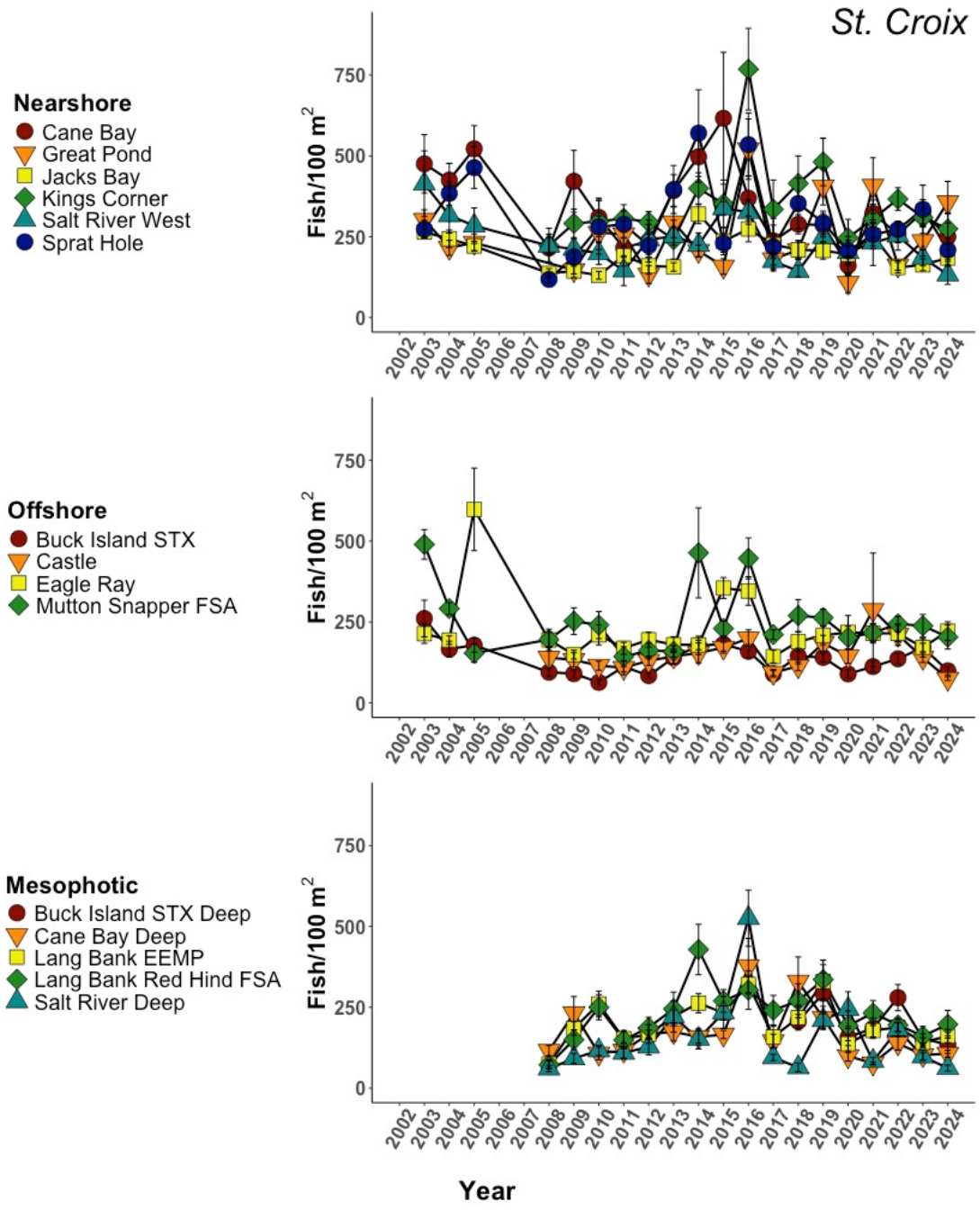

Figure 19. Fish abundance (±SE) across St. Croix TCRMP monitoring sites from 2003-2024.

Fish Biomass

Total fish biomass for all sites and years is shown in Figure 20 and Figure 21. 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 2024 (4,774 kg vs 3,723 kg, respectively), however this was likely attributed to the high biomass observed at Grammanik Tiger FSA in 2023 when sampling coincided with cubera snapper (*Lutjanus cyanopterus*) spawning (August; Biggs and Nemeth, 2016).

The five sites with the highest biomass were all mesophotic sites: Grammanik Tiger FSA (9.2% of total biomass), College Shoal East (8.9%), Salt River Deep (7.4%), Cane Bay Deep (7.0%), and Lang Bank EEMP (6.4%). These sites often have large pelagic species such as sharks, as well as larger groupers and snappers. Similar to the fish diversity metrics, the sites with the lowest biomass of fish coincided with sites that often have high sedimentation or siltation rates: Magens Bay, Coral Bay, Salt River West, and Castle.

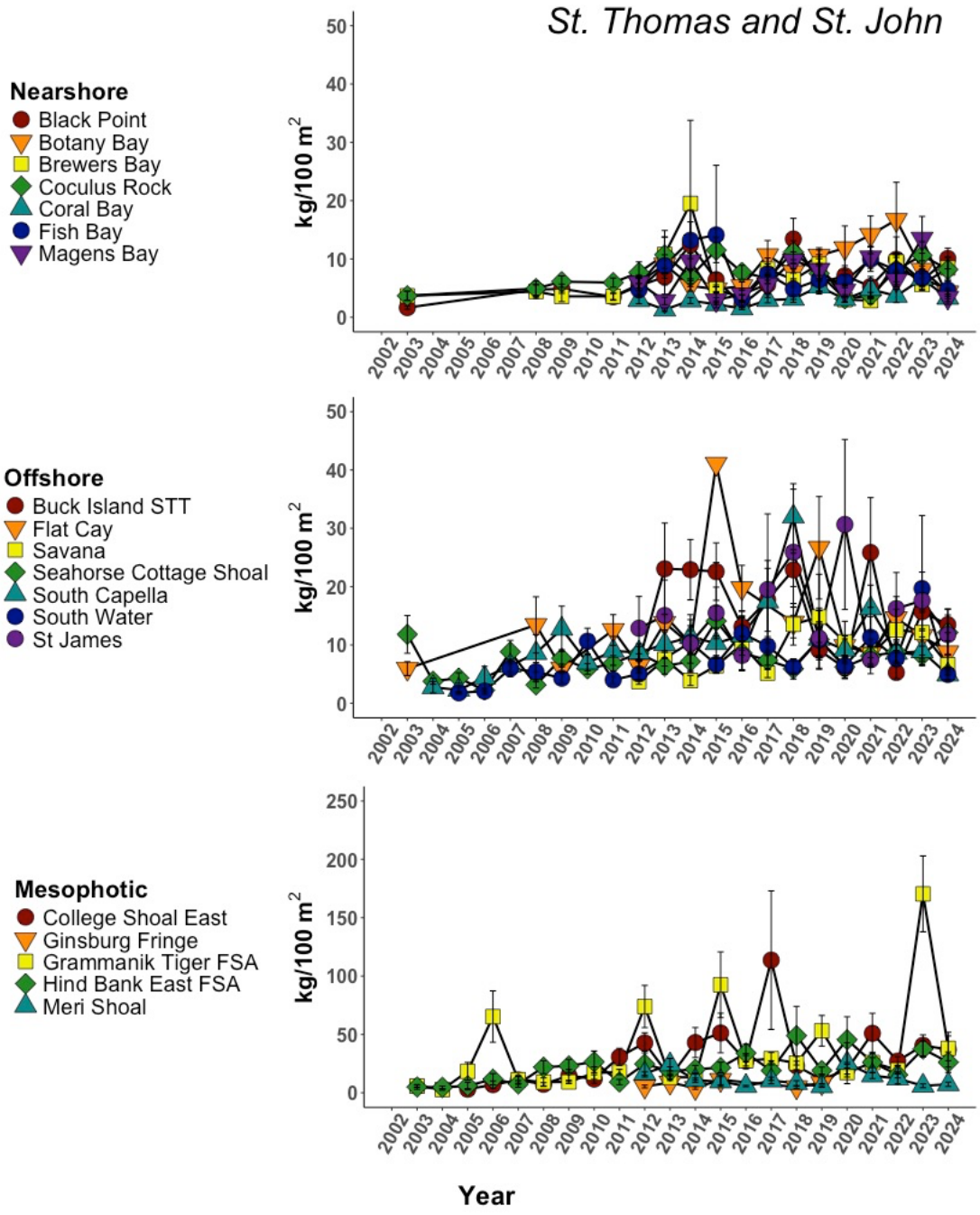

Figure 20. Mean fish biomass (±SE) across St. Thomas and St. John TCRMP monitoring sites from 2003-2024.

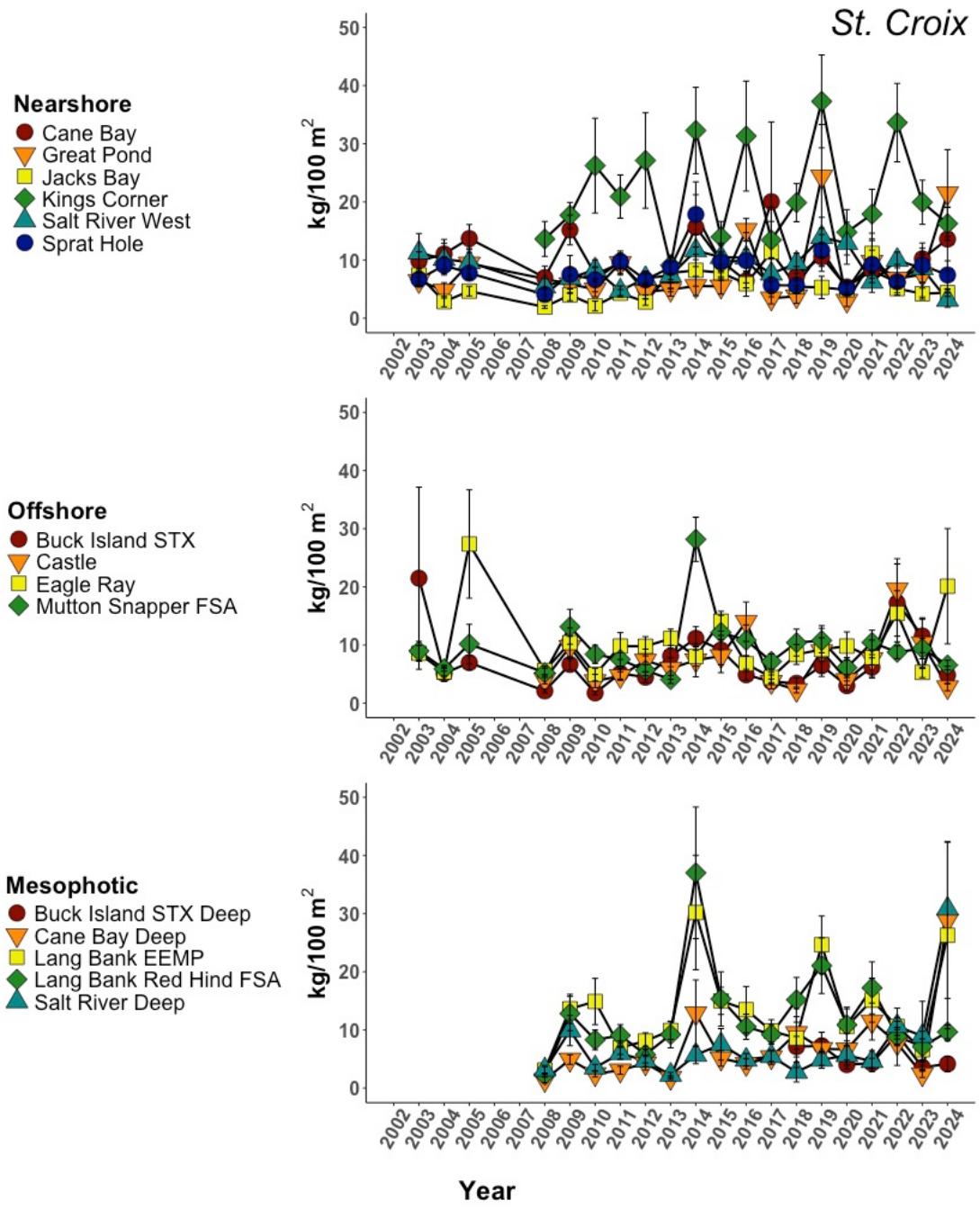

Figure 21. Mean fish biomass (±SE) across St. Croix TCRMP monitoring sites from 2003-2024.

SPECIAL INTEREST FISH: NASSAU GROUPER AND INVASIVE LIONFISH

Nassau grouper: The International Union for Conservation of Nature (IUCN) has focused on Nassau grouper (*Epinephelus striatus*) as early as 1996 when the species was officially classified as Threatened (IUCN 1996). Since then, the species' conservation status has escalated: it was listed as Endangered on the IUCN Red List in 2003 (Cornish and Eklund 2003), uplisted to Critically Endangered in 2018 (Sadovy et al 2018), and listed as Threatened under the U.S. Endangered Species Act in 2016 (NOAA Fisheries 2016). This species was once abundant in the USVI, but numbers dwindled with a heavy fishing presence in the territory. The spawning aggregation site at Grammanik Bank south of St. Thomas remains protected by a seasonal fishing closure (February 1–April 30) implemented through the Caribbean Fishery Management Council and NOAA Fisheries. Since the seasonal closure was enacted in 2005 (NOAA Fisheries 2005), TCRMP data has shown an increase in the abundance density of Nassau grouper observed during annual surveying (Figure 23). From 2003 through 2014, Nassau grouper were observed in survey transects at a maximum of 3 TCRMP sites. In 2024, Nassau were observed within transects at 13 TCRMP sites, with the maximum observed of 7 at the Grammanik Bank and 5 at Hind Bank East FSA.

Figure 22. Nassau grouper (*Epinephelus striatus)* observed while sampling at Grammanik Tiger FSA (photo credit: N.Krampitz).

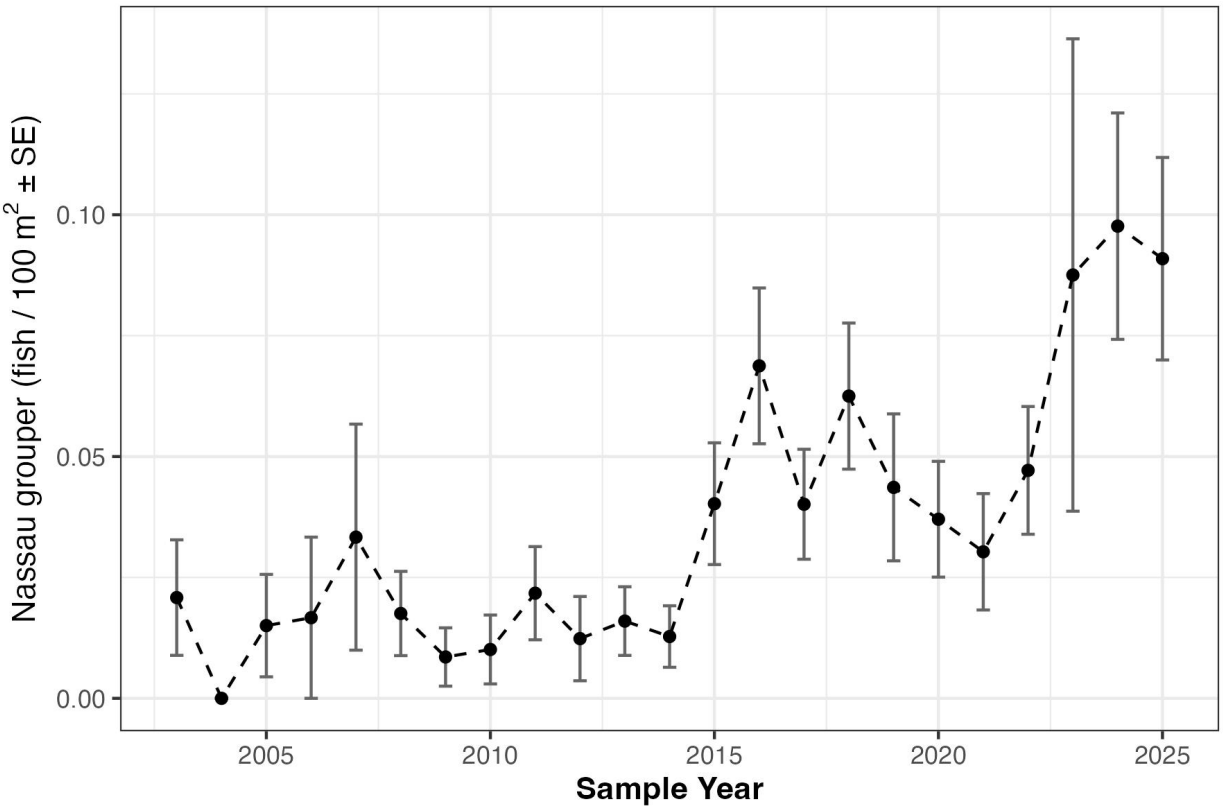

Figure 23: Density of Nassau grouper (*Epinephelus striatus*) per 100m^2 across all TCRMP sites, 2003 – 2025.

Lionfish: Lionfish (*Pterois volitans*) are an invasive piscivore that were introduced to the Caribbean in the early 2000's (Figure 24) (Schofield 2009). This species is a generalist predator and consume a wide range of local species, including ecologically and commercially important species (Green et al. 2012). In addition to their diverse palate, lionfish reproduce quickly, have no known natural predators, and can inhabit a wide range of reefs from shallow nearshore to deeper mesophotic depths. Lionfish therefore pose a significant threat to native fish species and reef communities. TCRMP first documented lionfish in fish transect surveys in 2011, and since then, lionfish abundance and density on TCRMP surveys have remained variable through the years (Figure 25). At TCRMP sites, lionfish are present across nearshore, offshore, and mesophotic reefs, but are most abundant at mesophotic depths (Blincow et al. 2025). In 2024, 69 lionfish were observed within belt transects, and 40 of these lionfish were at mesophotic sites. These mesophotic

sites may function as a refuge for lionfish because culling efforts are often concentrated at shallower sites due to site accessibility and diver and gas restrictions. Mesophotic lionfish have also been proposed to be a source population for seeding shallower sites; lionfish at mesophotic depths comprise the highest proportion of actively spawning females, and shallower sites contain a greater proportion of immature individuals (Andradi-Brown et al 2017).

Figure 24. Lionfish (*Pterois volitans*) observed while sampling at Flat Cay, St Thomas (photo credit: E Hollander)

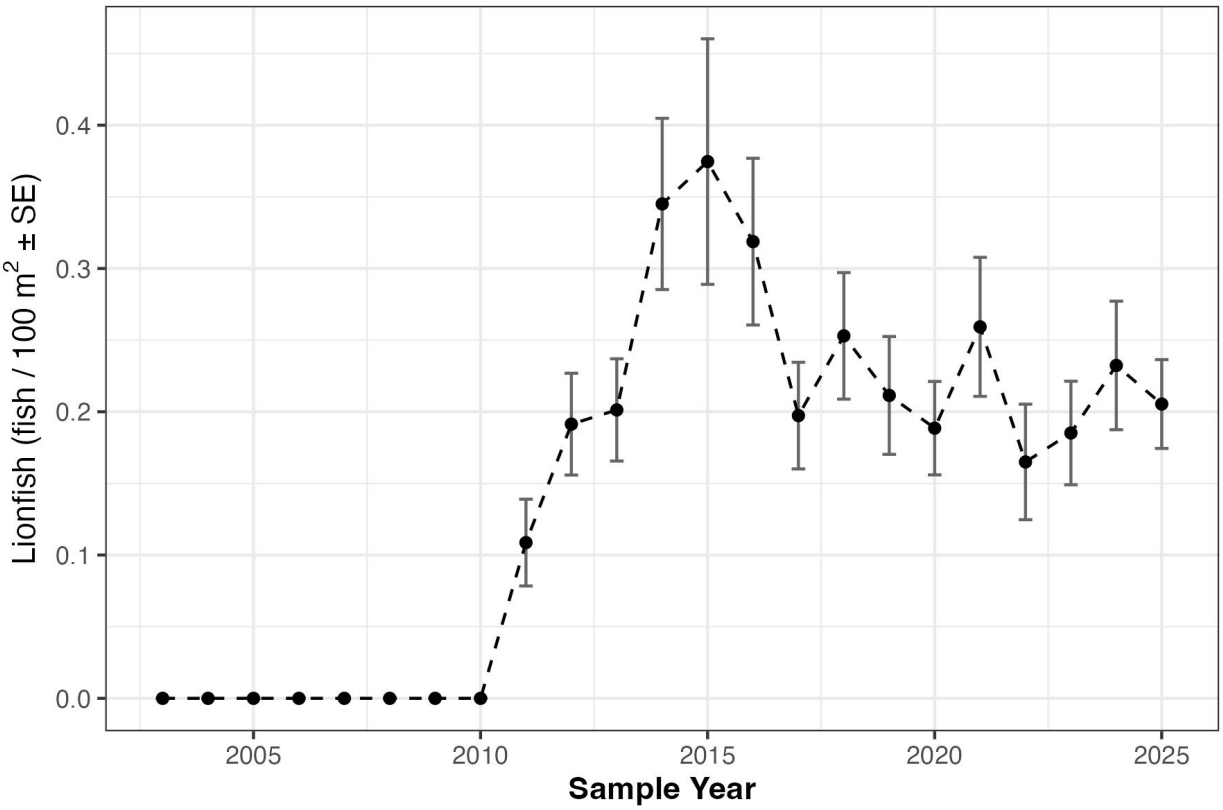

Figure 25: Density of lionfish (*Pterois volitans*) per 100 m^2 across all TCRMP sites from 2003 - 2025.

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* (Figure 26). The overall abundance of *Diadema* at TCRMP sites has remained somewhat variable over the years (Figure 27). Exceptions to this are major declines seen in 2017 following 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. In 2023, the overall density of *Diadema* in survey transects increased from 0.23 ± 0.15 urchins/100m^{-2} in 2022 to 1.00 ± 0.68 urchins/100m^{-2}. However, in 2024, the average density of Diadema decreased to 0.28 ± 0.10 urchins/100m^{-2}. The distribution of TCRMP sites where *Diadema* were observed has also shifted over time; in 2021, *Diadema* were recorded at 15 TCRMP sites, compared to 7 sites

in 2023 and 8 sites in 2024 (Figure 27). Furthermore, between 2023 and 2024, only 3 sites from 2023 maintained *Diadema* into 2024. The four new sites in 2024 only observed 1 *Diadema* each, with the exception of Magens Bay which had two. These sites last recorded *Diadema* in 2020 at Sprat Hole, 2021 at Eagle Ray, Savana, and Castle, and in 2022 at Magens Bay.

In addition to a decreased density of urchins at TCRMP sites, test size has also slightly decreased since 2023 from 7.69 ± 2.21 cm to 5.50 ± 1.62cm (Figure 28). Fewer and smaller urchins on the reef indicates a reduction in grazing power as well as a lower reproductive output.

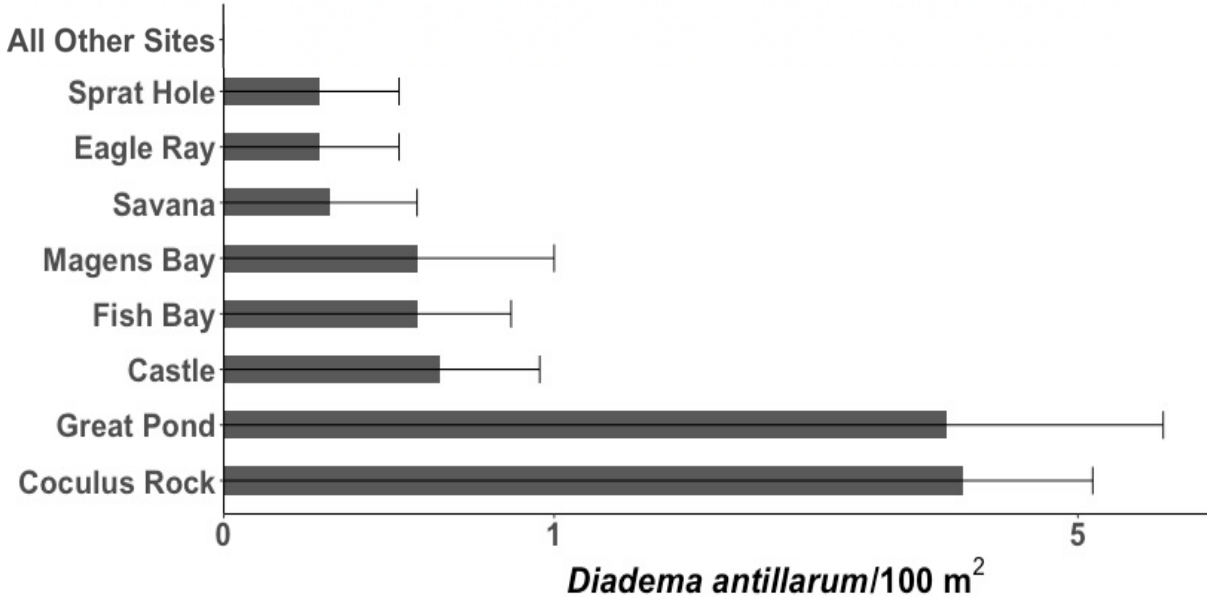

Figure 26. Average density (±SE) of the black spined sea urchin (*Diadema antillarum*) at 33 TCRMP monitoring sites in 2024.

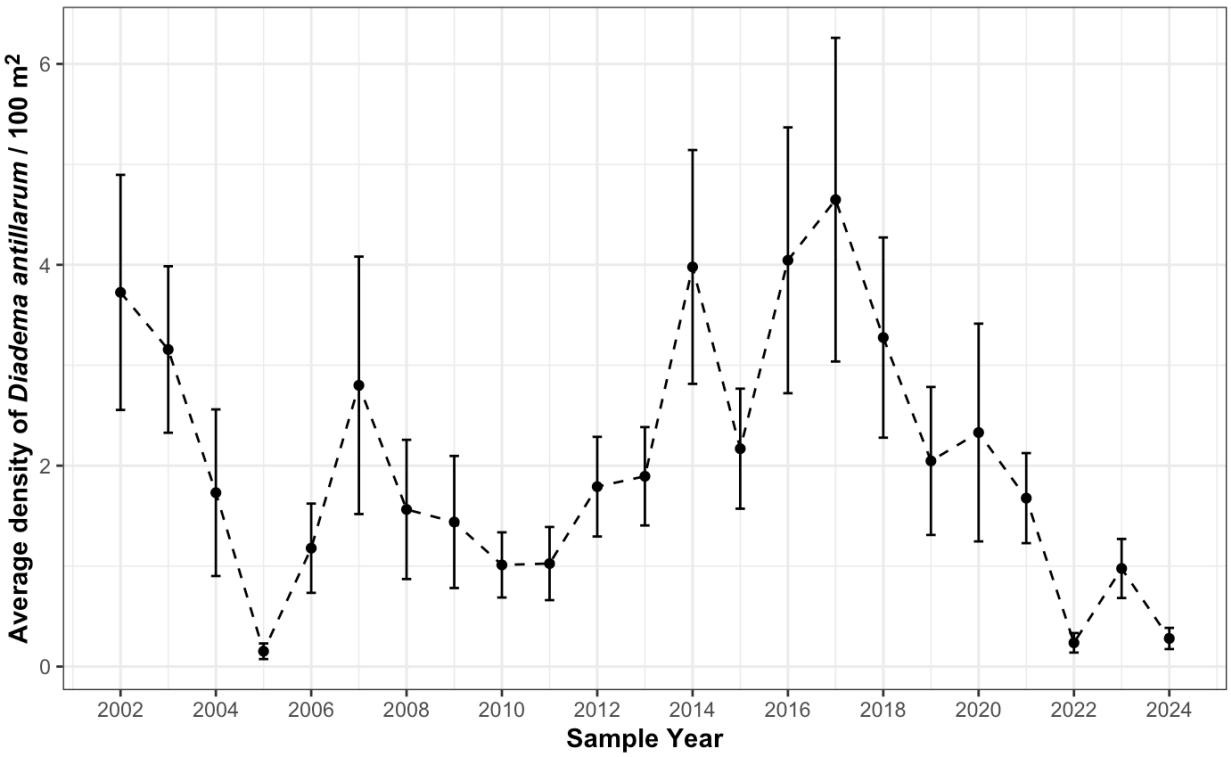

Figure 27. Average density ± SE of *Diadema antillarum* (*Diadema* / 100m^2) at TCRMP sites from 2002 – 2024.

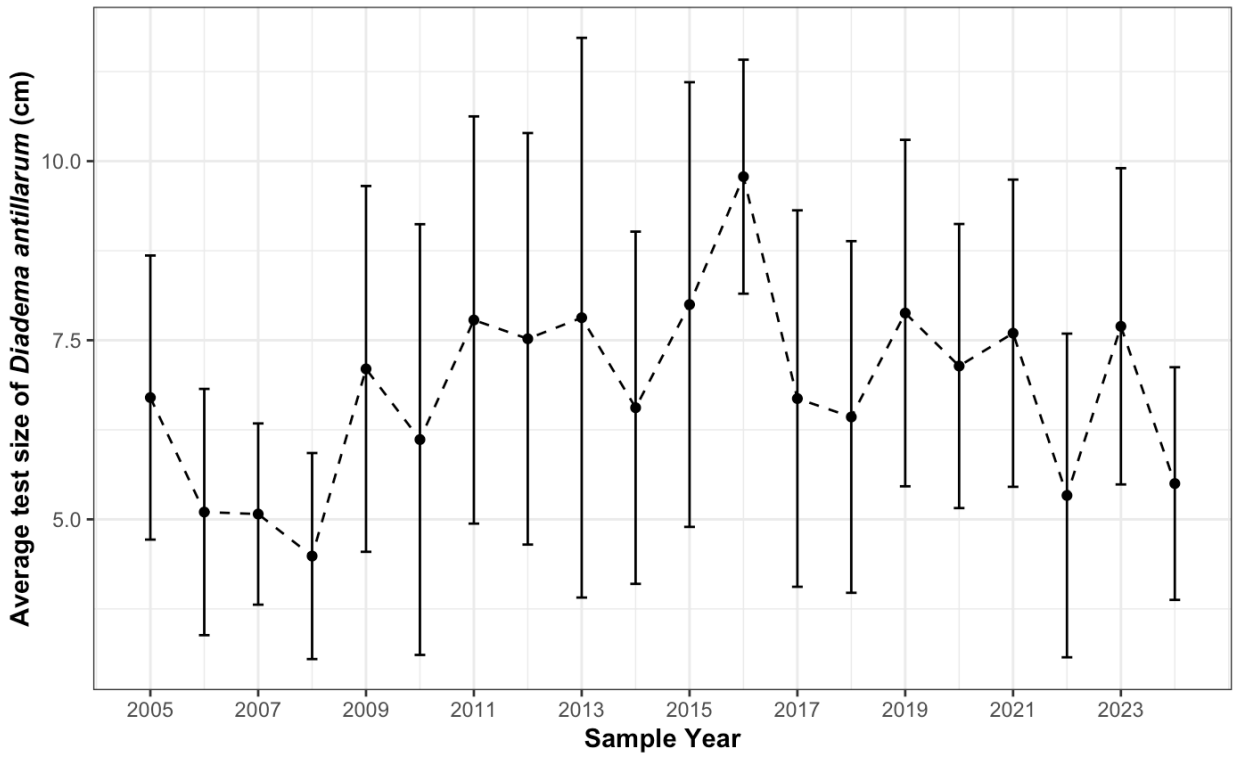

Figure 28. Average test size of Diadema antillarum (cm) ± SE at TCRMP sites from 2005 – 2024

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