National Park Service
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Natural Resource Stewardship and Science

Natural Resource Condition Assessment

Salt River Bay National Historical Park and Ecological Preserve

Natural Resource Report NPS/SARI/NRR—2022/2407

ON THE COVER
View of Salt River Bay from Hemmer's Peninsula
Photo Credit: Anna Wachnika

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Executive Summary

Natural Resource Condition Assessments (NRCAs) provide managers with concise assessments for select focal resources within National Park Service (NPS) units. These assessments evaluate indicators of condition for a resource and determine status and trends over time for best management of the resources within a unit. Salt River Bay National Historical Park and Ecological Preserve (SARI) is a 1,015 acre unit located within the Caribbean, situated on the north side of the island of St. Croix. Consisting of both marine and terrestrial components, environments of SARI range in elevation from 200 m below mean sea level along the north edge of the Salt River Canyon to 83 m above sea level in the semi-deciduous tropical forest. Marine communities account for more than 60% of the area of the Park and include soft bottom habitats predominantly occupied by seagrasses and hardbottom habitats colonized by coral reefs. Terrestrial habitats are dominated by upland forests, mangroves, shrublands, and grasslands.

The SARI NRCA considers 11 focal resources within the Park categorized as either pertaining to the supporting environment or biological integrity. These include shoreline dynamics, water quality, and watershed dynamics in the framework category of supporting environment, and mangrove, semi-deciduous dry forest, coastal grassland, macroalgae, seagrass, corals, queen conch, and reef fish, in the framework category of biological integrity. Full assessments were conducted for all above-listed resources except for queen conch which was restricted to a limited assessment due to the data available. In each focal resource section, a discussion of threats, stressors, and data gaps relevant to the resource accompanies the assessment of condition. Resource issues relevant to all components within the Park are discussed separately and include impacts of hurricanes/tropical storms, land cover/land use changes, and human interactions related to boat traffic, marine debris, and poaching.

Assessment of the focal resources in the Park resulted in the majority, six of 11 (55%), considered as warranting moderate concern. Four focal resources warranted significant concern, and only one resource was considered to be in good condition. Trends in condition were nearly equally divided between improving, deteriorating, stable, and undetermined. The focal resources assessed in this report are a mix of marine and terrestrial resources. Terrestrial resources included condition assessments for three vegetation communities (mangroves, dry tropical forest, and coastal grassland) and two supporting environments (watershed condition and shoreline dynamics). With the exception of coastal grasslands (which warrant significant concern), the other terrestrial focal resources were considered to be of moderate concern or good condition, with improving trends or stable conditions. The marine focal resources of SARI were either of significant concern (reef fish, corals, and macroalgae) or moderate concern (conch, seagrass, and water quality) with deteriorating or stable trends. Taken as whole, the assessment suggests that the focal resources of SARI are experiencing degraded conditions compared to reference conditions for these resources and appear to be under a wide range of threats. Deteriorating conditions for corals and macroalgae combined with a lack of recovery of the reef fish communities are especially concerning. The current conditions for these resources appear to have resulted from the interaction of disturbance events and anthropogenic impacts, including extent of hurricane damage, increasing sea surface temperatures, contaminants, introduction of invasive species, and continued fishing pressure.

Shoreline dynamics was assessed as being in both good condition and having an improving trend attributable to increasing shoreline length and extent of the shoreline currently in vegetated cover. As a supporting resource, the differing character of the shoreline, vegetated vs. sandy or rocky, will undoubtedly benefit particular biological resources at the expense of others. We consider the overall land accretion that is happening at a steady pace, especially in the northeast area of the Park, to be an improving trend as it supports terrestrial resources, providing land for mangrove colonization on mudflats and shorebird use on sandy/gravel shorelines. Similarly, the condition of mangroves has improved over the time period assessed as mangroves have both recolonized areas in which they were lost following Hurricane Hugo (1989) and have expanded seaward onto accreted sediments and landward with rising sea level. Coastal grasslands are the third focal resource assessed with an improving trend in condition. The improving trend is a direct result of management actions that have reduced non-native invasive plant species, combined with a reforestation effort focused on mixed-dry grassland native woody species.

A moderate level of confidence was assigned to the majority of resources (6 of 11), with individual indicators assigned either low or medium confidence for those resources. A high level of confidence was assigned only for the coral reef focal resource and for shoreline dynamics. Three focal resources had low confidence in the assessment, including: coastal grassland, conch, and seagrass. Given that a minority of focal resources had high confidence in their assessments, assessments of condition are constrained by a lack of recent data, insufficient temporal or spatial coverage of datasets, or differences between survey methods for datasets compared in this assessment. Therefore, important information gaps, as well as protocols for future data acquisition and monitoring are suggested.

Recommendations for future monitoring include the following: 1) design of an integrated approach to monitoring and data collection of marine focal resources of SARI, incorporating metrics of water quality, coral health and abundance, seagrass cover, and the presence of non-native invasive species, 2) expansion of research on the use of the marine and terrestrial resources by visitors to estimate benefits from ecosystem services provided and amount of anthropogenic pressure on the resource, 3) expansion of a permanent plot network throughout the terrestrial vegetation focal resources to understand long-term changes in species assemblages and abundances, as well as tracking the distribution of invasive plant species, and 4) increased hydrological monitoring within the Salt River watershed, including establishment of a weather station, to quantify temporal frequency of the flow of water, nutrients, sediments, and contaminants from the terrestrial to the near-shore marine environment. Expansion of monitoring programs will add to the large body of research already conducted within the Park and will be invaluable for understanding changes to these resources resulting from future hurricane disturbance, rising seas, and increasing temperatures and changing rainfall patterns expected in a warming climate.

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Chapter 1. NRCA Background Information

Natural Resource Condition Assessments (NRCAs) evaluate current conditions for a subset of natural resources and resource indicators in national park units, hereafter "parks." NRCAs also report on trends in resource condition (when possible), identify critical data gaps, and characterize a general level of confidence for study findings. The resources and indicators emphasized in a given project depend on the park's resource setting, status of resource stewardship planning and science in identifying high-priority indicators, and availability of data and expertise to assess current conditions for a variety of potential study resources and indicators.

> ***NRCAs Strive to Provide…***
>
> - *Credible condition reporting for a subset of important park natural resources and indicators*
> - *Useful condition summaries by broader resource categories or topics, and by park areas*

NRCAs represent a relatively new approach to assessing and reporting on park resource conditions. They are meant to complement—not replace—traditional issue-and threat-based resource assessments. As distinguishing characteristics, all NRCAs:

- Are multi-disciplinary in scope;[1]
- Employ hierarchical indicator frameworks;[2]
- Identify or develop reference conditions/values for comparison against current conditions;[3]
- Emphasize spatial evaluation of conditions and GIS (map) products; [4]
- Summarize key findings by park areas; and [5]
- Follow national NRCA guidelines and standards for study design and reporting products.

Although the primary objective of NRCAs is to report on current conditions relative to logical forms of reference conditions and values, NRCAs also report on trends, when appropriate (i.e., when the underlying data and methods support such reporting), as well as influences on resource conditions. These influences may include past activities or conditions that provide a helpful context for

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[1] The breadth of natural resources and number/type of indicators evaluated will vary by park.

[2] Frameworks help guide a multi-disciplinary selection of indicators and subsequent "roll up" and reporting of data for measures ⇨ conditions for indicators ⇨ condition summaries by broader topics and park areas

[3] NRCAs must consider ecologically-based reference conditions, must also consider applicable legal and regulatory standards, and can consider other management-specified condition objectives or targets; each study indicator can be evaluated against one or more types of logical reference conditions. Reference values can be expressed in qualitative to quantitative terms, as a single value or range of values; they represent desirable resource conditions or, alternatively, condition states that we wish to avoid or that require a follow-up response (e.g., ecological thresholds or management "triggers").

[4] As possible and appropriate, NRCAs describe condition gradients or differences across a park for important natural resources and study indicators through a set of GIS coverages and map products.

[5] In addition to reporting on indicator-level conditions, investigators are asked to take a bigger picture (more holistic) view and summarize overall findings and provide suggestions to managers on an area-by-area basis: 1) by park ecosystem/habitat types or watersheds, and 2) for other park areas as requested.

understanding current conditions, and/or present-day threats and stressors that are best interpreted at park, watershed, or landscape scales (though NRCAs do not report on condition status for land areas and natural resources beyond park boundaries). Intensive cause-and-effect analyses of threats and stressors, and development of detailed treatment options, are outside the scope of NRCAs.

Due to their modest funding, relatively quick timeframe for completion, and reliance on existing data and information, NRCAs are not intended to be exhaustive. Their methodology typically involves an informal synthesis of scientific data and information from multiple and diverse sources. Level of rigor and statistical repeatability will vary by resource or indicator, reflecting differences in existing data and knowledge bases across the varied study components.

The credibility of NRCA results is derived from the data, methods, and reference values used in the project work, which are designed to be appropriate for the stated purpose of the project, as well as adequately documented. For each study indicator for which current condition or trend is reported, we will identify critical data gaps and describe the level of confidence in at least qualitative terms. Involvement of park staff and National Park Service (NPS) subject-matter experts at critical points during the project timeline is also important. These staff will be asked to assist with the selection of study indicators; recommend data sets, methods, and reference conditions and values; and help provide a multi-disciplinary review of draft study findings and products.

NRCAs can yield new insights about current park resource conditions, but, in many cases, their greatest value may be the development of useful documentation regarding known or suspected resource conditions within parks. Reporting products can help park managers as they think about near-term workload priorities, frame data and study needs for important park resources, and communicate messages about current park resource conditions to various audiences. A successful NRCA delivers science-based information that is both credible and has practical uses for a variety of park decision making, planning, and partnership activities.

Important NRCA Success Factors

- *Obtaining good input from park staff and other NPS subject-matter experts at critical points in the project timeline*
- *Using study frameworks that accommodate meaningful condition reporting at multiple levels (measures ⇨ indicators ⇨ broader resource topics and park areas)*
- *Building credibility by clearly documenting the data and methods used, critical data gaps, and level of confidence for indicator-level condition findings*

However, it is important to note that NRCAs do not establish management targets for study indicators. That process must occur through park planning and management activities. What an NRCA can do is deliver science-based information that will assist park managers in their ongoing, long-term efforts to describe and quantify a park's desired resource conditions and management targets. In the near term, NRCA findings assist strategic park resource planning[6] and help parks to report on government accountability measures.[7] In addition, although in-depth analysis of the effects of climate change on park natural resources is outside the scope of NRCAs, the condition analyses and data sets developed for NRCAs will be useful for park-level climate-change studies and planning efforts.

NRCAs also provide a useful complement to rigorous NPS science support programs, such as the NPS Natural Resources Inventory & Monitoring (I&M) Program.[8] For example, NRCAs can provide current condition estimates and help establish reference conditions, or baseline values, for some of a park's vital signs monitoring indicators. They can also draw upon non-NPS data to help evaluate current conditions for those same vital signs. In some cases, I&M data sets are incorporated into NRCA analyses and reporting products.

NRCA Reporting Products…

Provide a credible, snapshot-in-time evaluation for a subset of important park natural resources and indicators, to help park managers:

- *Direct limited staff and funding resources to park areas and natural resources that represent high need and/or high opportunity situations* ***(near-term operational planning and management)***
- *Improve understanding and quantification for desired conditions for the park's "fundamental" and "other important" natural resources and values* ***(longer-term strategic planning)***
- *Communicate succinct messages regarding current resource conditions to government program managers, to Congress, and to the general public* ***("resource condition status" reporting)***

Over the next several years, the NPS plans to fund an NRCA project for each of the approximately 270 parks served by the NPS I&M Program. For more information visit the NRCA Program website.

[6]An NRCA can be useful during the development of a park's Resource Stewardship Strategy (RSS) and can also be tailored to act as a post-RSS project.

[7] While accountability reporting measures are subject to change, the spatial and reference-based condition data provided by NRCAs will be useful for most forms of "resource condition status" reporting as may be required by the NPS, the Department of the Interior, or the Office of Management and Budget.

[8] The I&M program consists of 32 networks nationwide that are implementing "vital signs" monitoring in order to assess the condition of park ecosystems and develop a stronger scientific basis for stewardship and management of natural resources across the National Park System. "Vital signs" are a subset of physical, chemical, and biological elements and processes of park ecosystems that are selected to represent the overall health or condition of park resources, known or hypothesized effects of stressors, or elements that have important human values.

Chapter 2. Introduction and Resource Setting

2.1. Introduction

2.1.1. Enabling Legislation

The Salt River Bay National Historical Park and Ecological Preserve (SARI) is an archaeological and ecological preserve located on the island of St. Croix in the U.S. Virgin Islands. SARI "uniquely documents the human and natural Caribbean world from the earliest indigenous settlements in the central Caribbean to their clash with seven different colonial European powers to the present day" (NPS 2018a).

SARI was established by Congress in 1992, in order to "preserve, protect, and interpret for the benefit of present and future generations certain internationally significant historical, cultural, and natural sites and resources in the Virgin Islands" (Public Law 102-247, February 24, 1992).

Congress found that SARI and its resources are "worthy of a comprehensive preservation effort that should be carried out in partnership between the Federal Government and the Government of the U.S. Virgin Islands." Accordingly, Congress authorized the U.S. Secretary of the Interior to enter into cooperative agreements with the U.S. Virgin Islands for the preservation and management of SARI.

2.1.2. Geographic Setting

The Virgin Islands are part of the northerly Leeward Islands in the Caribbean, situated between the Greater Antilles and the Lesser Antilles. Politically, the islands fall into several jurisdictions: the British Virgin Islands, which are a British overseas territory, the Puerto Rican Virgin Islands, which is a territory of the United States, and the United States Virgin Islands (USVI), also a territory of the United States. The USVI consists of four larger islands: St. Croix, St. Thomas, St. John and Water Island, and some 50 smaller islets and cays. The total area of the USVI is 133 square miles.

The island of St. Croix, the largest island (with an area of 82.88 sq mi / 214.66 km^2) of the USVI, is located to the South of the string of islands that forms the Virgin Islands complex (USVI and BVI). St. Croix is a county and constituent district of the USVI. The highest point in the island is Mount Eagle with an altitude of 355 m (1,165 ft). The Salt River watershed is located in the central northern part of St. Croix (see Figure 2.1.2.1 and the map in Section 4.3). The Salt River National Historic Park and Ecological Preserve (SARI) is located at the northern end of the Salt River watershed. The park is five miles from Christiansted National Historic Site and can be reached by car via Rt. 75 from Christiansted, connecting to Route 80. SARI is unique for reasons of both history and nature (NPS 2018a).

The prehistoric complex at SARI is among the most important archaeological sites in the USVI. Within SARI, visitors can find several prehistoric sites dating back to A.D. 300, and other interesting sites locations such as the Columbus landing site. However, Salt River Bay's appeal to visitors is largely underwater. Divers can discover anchors belonging to sailing ships from the last 400 years. In addition, numerous marine lifeforms and environments attract the interest of visitors.at SARI. The Salt River Bay houses the largest stand of mangrove forest in the USVI. This lush estuarine area transitions into a barrier reef in about 35 feet of water, which precedes a spectacular submarine canyon that plunges down to over 600 feet. The proximity of this myriad of natural aquatic landscapes brings together a rich collection of marine species within a small area. The National Park Service and the Government of the U.S. Virgin Islands administer the park jointly (NPS 2019a).

Figure 2.1.2.1. Geographic location of the US Virgin Islands in the Caribbean (upper panel). Location of the Salt River National Historic Park and Ecological Preserve (SARI) within the island of St. Croix (lower panel right). Demarcation of the territory that encompasses SARI (lower panel left). (Delineation of SARI area from NPS Boundary, NPS 2019b).

2.1.3. Visitation Statistics

Visitation statistics for SARI are provided by the NPS and are calculated based on the following: 1) the number of visitors to the visitor center, 2) the number of individuals visiting as part of school groups, 3) the number of visitors to the beach, and 4) the number of visitors on kayak tours. From January 2006 to December 2019, SARI had 71,751 visitors to the park, with most visits occurring between the months of December and April (NPS 2019c) (Figure 2.1.3.1). The average number of recreational visitors to the park from 2006 to 2019 was 5,125 per year (NPS 2020). The number of visitors going to the park declined after the passage of Hurricanes Irma and Maria in September 2017 (the park closed for a several months), and the number of visitors began increasing in early 2018. The Salt River Bay is considered a living museum where history and nature blend on the island of St. Croix. Visitors can explore mangrove forests as well as coral reefs and a submarine canyon. Visitors can enjoy a variety of activities on the land and in the water, including swimming, snorkeling, scuba diving, sailing, kayaking, hiking, nature watching and archaeology (NPS 2019c). Additionally, there are culturally significant camping locations and visitors can also enjoy horseback riding. Guided tours within certain locations of the park are offered to visitors.

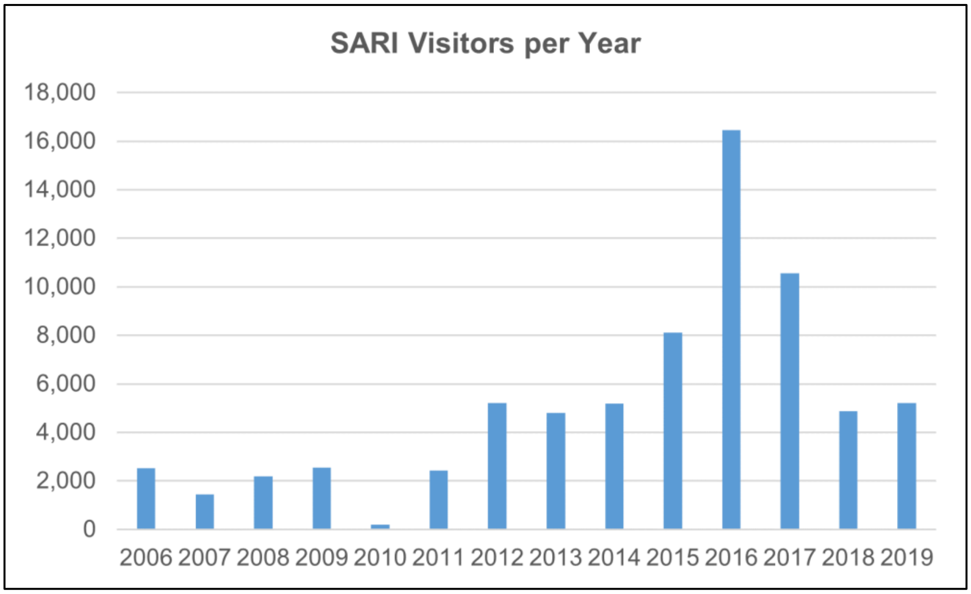

Figure 2.1.3.1. Annual Visitations in SARI during the period 2006–2019 (data from NPS 2020).

2.2. Natural Resources

2.2.1. Ecological Units and Watersheds

Salt River Bay National Historical Park and Ecological Preserve (SARI) is a mosaic of ecosystems that includes mangrove forests, a submarine canyon, coral reefs, seagrass beds, coastal forests, and some developed landscape elements (Figure 2.2.1.1). The park is located in the north coast of the island of St. Croix (NPS 2015).

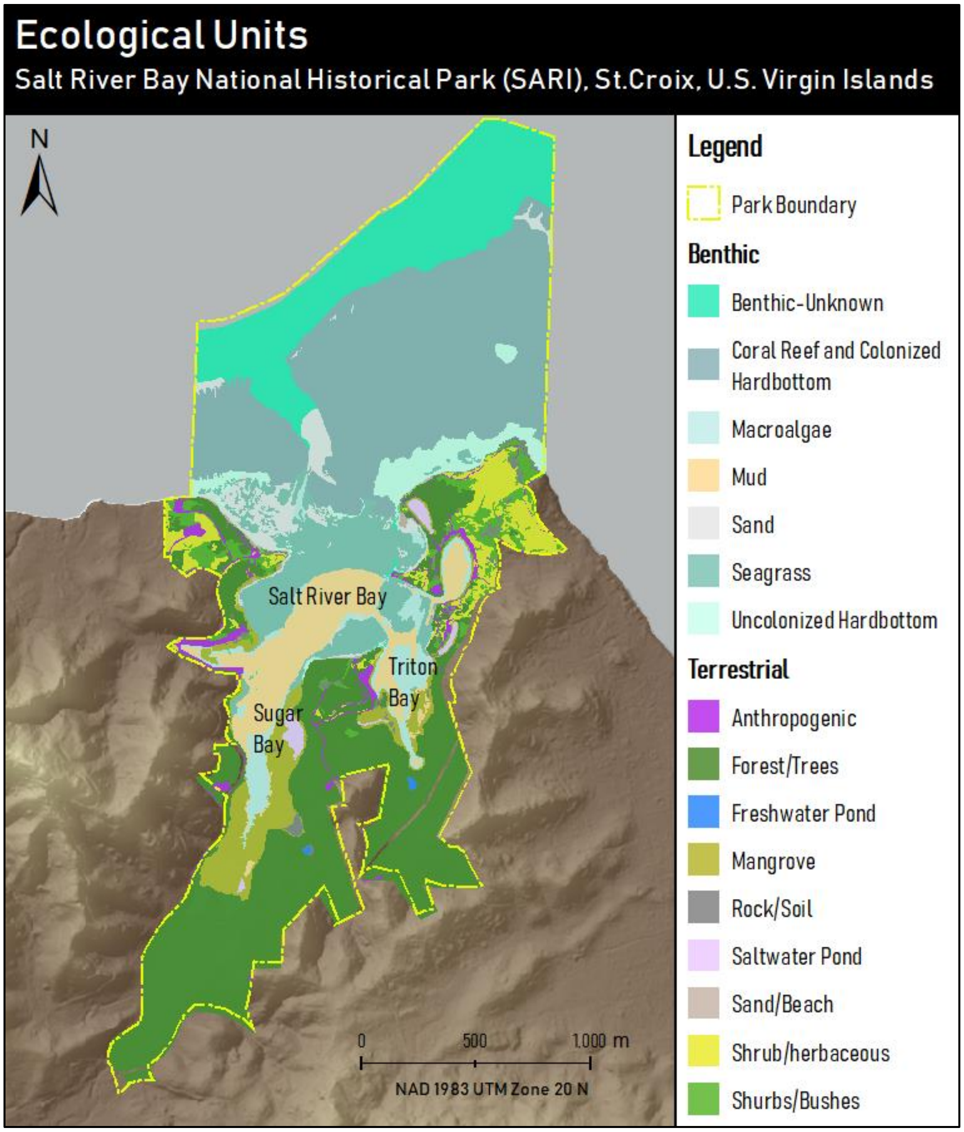

Figure 2.2.1.1. Ecological Units for Salt River Bay National History Park. Data collected in 2000 and published 2005 (Kendall et al. 2005; NCCOS 2019).

Ecological Units

Coral reefs and hardbottom (colonized and uncolonized) occupy about 116.3 ha (Table 2.2.1.1) up to 25 m deep. Based on Kendall et al. (2005), reefs are known to exist at deeper habitats but an assessment of these areas was not conducted at that time. More recent surveys have been monitoring corals as deep as 40 m (see Chapter 4.6.1). About 70% of the hardbottom habitat has been characterized as being colonized (Kendall et al. 2005). Previous studies on coral reef cover found that about 3% of SARI (hardbottom) is covered by corals (~ 41 species). However, these studies were mostly done on the canyon walls. In general, solidified substrates of the canyon tend to benefit live coral cover (Boulon 1978; Rogers et al. 1984). Nevertheless, the east and west walls present very different habitat conditions. For instance, the live coral optimal growth depth is about 18 m in the east while on the west wall its optimal growth depth is about 9 m (Rogers et al. 1984). In addition to the canyon, another important reef component are the linear reefs. The linear reefs serve as insulating barriers that protect the seagrass, algae, and mangrove habitats from wave action but also prevent deeper reefs from being impacted by terrestrial sediments and storm driven currents.

Table 2.2.1.1. Ecological Units for SARI. Data collected in 2000 and published 2005 (Kendall et al. 2005; NCCOS 2019).

Location	Ecological Unit	Area (ha)	% cover
Benthic	Benthic-Unknown	55.3	13.18
	Coral Reef and Colonized Hardbottom	104.8	24.96
	Macroalgae	18.6	4.44
	Mud	25.1	5.99
	Sand	11.4	2.70
	Seagrass	29.5	7.02
	Uncolonized Hardbottom	11.5	2.73
	Total Area	256.3	61%
Terrestrial	Forest/Trees	106.0	25.25
	Freshwater Pond	0.3	0.07
	Rock/Soil	2.7	0.63
	Saltwater Pond	2.4	0.58
	Sand/Beach	1.3	0.30
	Shrubs/Bushes	10.9	2.59
	Shrub/herbaceous	14.2	3.38
	Mangrove	18.6	4.44
	Total Area	156.4	37%
Artificial	Anthropogenic	7.3	1.73
Total Area	–	419.9	–

Seagrass and macroalgae ecological unit refers to areas where the vegetation cover is dominated by seagrass and algae. The seagrass species *Halophila decipiens* is found mostly in areas with unconsolidated sediment such as the canyon floor. Salt River Bay is dominated by *Thalassia testudinum*, *Syringodium filiforme*, and *Halodule wrightii*. The growth of the macrophytes is highly seasonal. The synergistic effect of reduced radiation, increased wave action, and sediment disturbance (from ephemeral river discharge and currents) during the winter season on these habitats tend to reduce growth and cover of seagrass. This is particularly evident in the shallow depths of the canyon floor, where light penetration is low. In general, the cover of seagrass and algae has been in decline in SARI since 1970, with Salt River Bay experiencing the greatest losses (Kendall et al. 2005).

The mangrove ecological unit is composed of mangrove forest stands of variable cover density. There are three major mangrove species: *Avicennia germinans* (black mangrove), *Laguncularia* *racemosa* (white mangrove), and *Rhizophora mangle* (red mangrove). Typically, these mangrove species present distribution patterns associated with the proximity to shore where red mangroves tend to be nearest to the shoreline, white mangroves most inland, and black mangroves in between. Prior to Hurricane Hugo, most of the mangrove cover was represented by black mangroves, followed by red mangroves, and a mixed mangrove forest also representing an important part of the landscape (see Ch. 4.4.1). A variety of organisms, such as fish (adults and juveniles), algae, sponges, tunicates, mollusks, and other invertebrates inhabit the mangrove forests (Tomlinson 1986; Hogarth 1999). Additionally, mangroves also serve a nursery function for reef fish (Nagelkerken 2009). Mangroves have experienced some of the highest losses in cover, especially after Hurricane Hugo. There have been restoration initiatives to increase mangrove cover; however, the success has been limited (Kendall et al. 2005).

Based on information collected in 2000 and published 2005 (Kendall et al. 2005), land use in SARI has had notable human influence, but at the time of the study less than 2% of the park presented permanent anthropogenic structures (Table 2.2.1.1). Dredging conducted in the 1960s (Pinckney 2014) to create an embayment for a hotel/marina development has been an important factor altering the salinity of the topsoil (spoil disposal) and the natural shape of the shoreline and bathymetry of the bays. Loam is the major component of the topsoil, which generally is not well suited for agriculture. Most of the vegetation cover consists of forest and shrubs/bushes of varying cover density. In general, the soils of the park, including the Salt River floodplain, are not suited for agriculture (NPS 1990; USDA 2000). As of the early 2000s, closed forest canopy covered most of the natural and semi-natural areas of the park (~25%), which are located in the southern portions of the park. The forests are dominated by dry forest types, which include semi-deciduous and gallery semi-deciduous forest. Shrubs and bushes concentrated in the northeastern and northwestern portions of the park account for ~6% of the vegetation cover of the park (Kendall et al. 2005). The most recent assessment of terrestrial vegetation in the park mapped 166 ha via interpretation of aerial orthophotos obtained in 2006–2007 (Moser et al. 2011). See the section on Terrestrial Communities in this chapter for more information.

Watershed

Although Salt River flows only during certain times of the year, the watershed drains approximately 1,360 ha (Figure 2.2.1.2), and presents an elevation gradient of about 300 m.a.s.l. The estimate of forest cover presented in Figure 2.2.1.2 for 2000 comes from a vegetation map produced as part of a Rapid Ecological Assessment by the Virgin Islands Conservation Data Center (University of the Virgin Islands 2001). This map was derived from aerial photographs that spanned the years 1994 to 2000. The analysis of the data used to produce the map showed that at the time, most of the area in the watershed was covered by natural or semi-natural vegetation (~81%). The remaining area (~19%) was occupied by human development (16.9%) or cropland (1.5%, Table 2.2.1.2). However, regardless of the small percentage of area developed, the main channel of the river has to traverse these areas before reaching the Salt River Gut[1], which can result in increased delivery of sediment to the bay, increasing turbidity of the water (Figure 2.2.1.2).

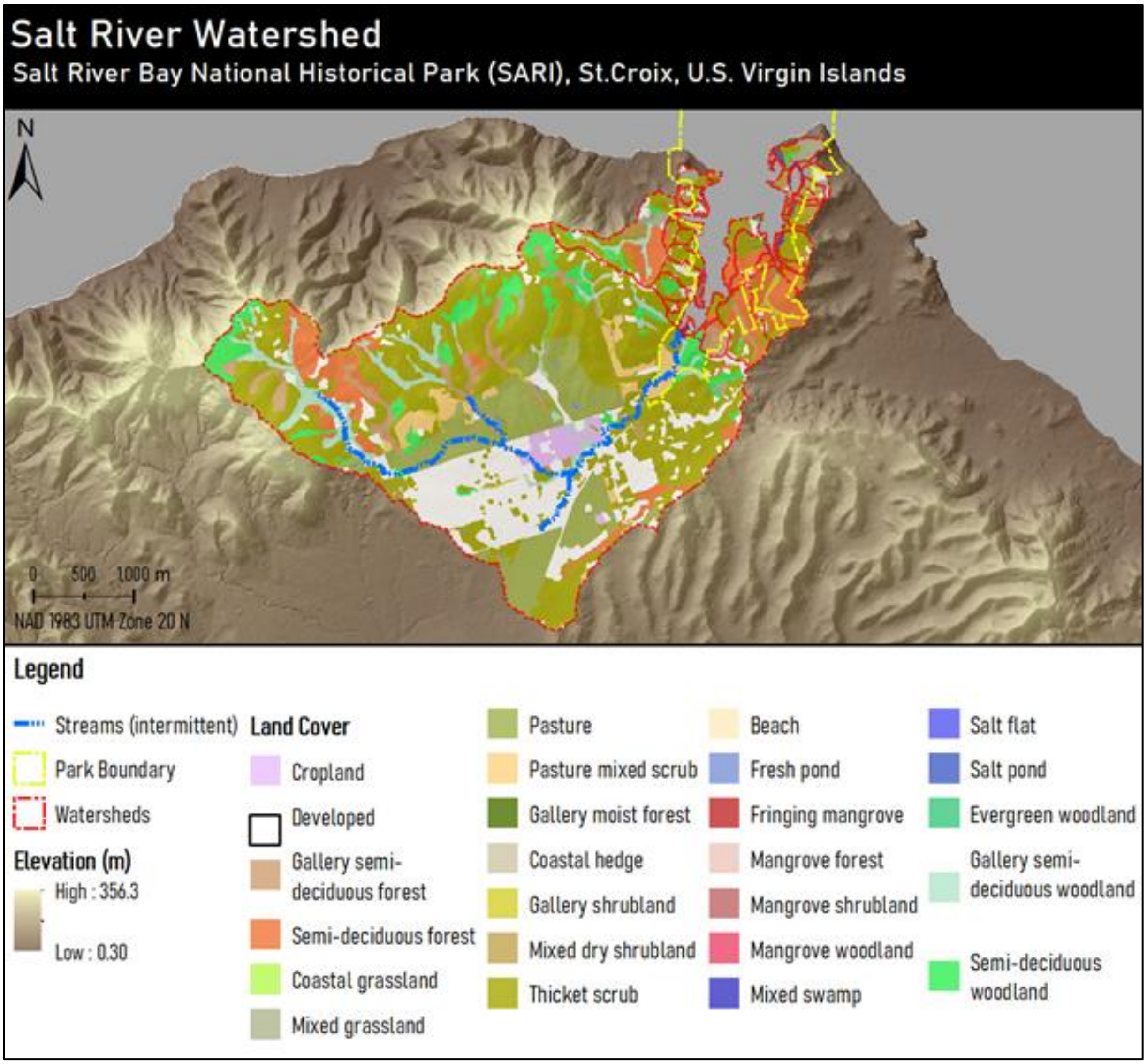

Figure 2.2.1.2. Land use for the Salt River watershed (University of the Virgin Islands 2001). Salt River watershed and smaller catchments were delineated in ArcGIS using Saint Croix DEM (St. Croix, U.S. Virgin Islands Coastal Digital Elevation Model – CKAN (data.gov)).

Based on the resulting land cover analysis, it was observed that within natural areas of the basin, shrubland was the major cover of the watershed (~40%, Table 2.2.1.2). Thicket scrub was the most abundant vegetation type (37.9%), with several other shrubland classes constituting the remaining 2%. Herbaceous areas covered about 16.6% of the watershed, with pasture being the most important

[1] In the Virgin Islands, gut or ghut, is a local term used to refer to a watercourse or stream of either permanent or intermittent flow (Gardner et al. 2008)

cover (12.32%). Dry tropical forest (~12%) and woodland (~11%) were and are present but confined to areas such as river guts where higher soil moisture over longer periods are encountered. Wetlands in the watershed cover a very small percentage (1%). The majority of the area classified as wetland was occupied by mangrove cover and adjacent freshwater wetland (Table 2.2.1.2).

Table 2.2.1.2. Land Use for the Salt River Watershed (University of the Virgin Islands 2001).

Land Use	Cover type	Area (Ha)	%
Cropland	Cropland	20.94	1.54
Developed	Developed	230.13	16.94
Dry Forest	Gallery semi-deciduous forest	43.77	3.22
	Semi-deciduous forest	118.46	8.72
Herbaceous	Coastal grassland	0.02	0.00
	Mixed grassland	21.91	1.61
	Pasture	167.39	12.32
	Pasture mixed scrub	36.74	2.70
Moist Forest	Gallery moist forest	0.01	0.00
Shrubland	Coastal hedge	0.24	0.02
	Gallery shrubland	8.68	0.64
	Mixed dry shrubland	29.37	2.16
	Thicket scrub	514.78	37.90
Sparse Vegetation	Beach	0.25	0.02
Wetland	Fresh pond	3.63	0.27
	Fringing mangrove	1.59	0.12
	Mangrove forest	1.82	0.13
	Mangrove shrubland	3.78	0.28
	Mangrove woodland	0.02	0.00
	Mixed swamp	0.88	0.06
	Salt flat	0.91	0.07
	Salt pond	1.49	0.11
Woodland	Evergreen woodland	0.64	0.05
	Gallery semi-deciduous woodland	69.97	5.15
	Semi-deciduous woodland	80.88	5.95
Total	–	1358.3	–

The condition of the Salt River watershed is analyzed in Section 4.3.1 using a data set obtained from the National Oceanic and Atmospheric (NOAA 2002, 2007, 2012). Temporal trends in condition metrics are evaluated using the land use/cover change for the referred period 2002–2012.

2.2.2. Resource Descriptions

Coastal Dynamics

Similar to other areas in the U.S. Virgin Islands, winds are the dominant force controlling the currents in Salt River Bay and along the bay mouth. Easterly wind direction is responsible for an east to west current throughout the year alongshore. This dominant east-west current is the major drivers of the shelf sediment transport into the Salt River Canyon (Kendall et al. 2005). In the eastern section of the park, the Judith's Fancy headland formation functions as a shield that reduces the wave action generated by the easterly winds that transports water over the reef crest all the way into the Salt River Bay (Kendall et al. 2005). As a result, the east shelf is a major source of sediment for the Salt River Canyon (Hubbard 1989, 1992).

Sediment type and distribution is heavily influenced by depth and proximity to terrestrial sources (Kendall et al. 2005). Sediment of the deep areas of the bays is very fine and originates from terrestrial erosion and runoff that is carried into the bays by ephemeral streams during periods of heavy rain (Gerhard and Petta 1974). While in the shallow areas of the bays, the sediment is primarily carbonate derived from calcareous algae *Halimeda* (Kendall et al. 2005). Hubbard (1989, 1992) also suggested that outside the bays, bioerosion of corals is the main source of carbonated sediments.

Sediment input and export of Salt River Canyon is a process in which sediment accretion is a gradual, slow and long term process while erosion is brief and intense (Williams 1988). For instance, there are records of heavy rain removing the equivalent of 5–10 years of sediment accumulation on a single day. Also, during strong storms such as hurricanes, large volumes of water can enter the bays causing large storms surges (1–1.5 m, Kendall et al. 2005) and result in flooding and erosion of the coastal areas. Given the sediment dynamics in SARI, erosion is likely to increase with an increase in strong hurricane frequency (Kendall et al. 2005).

An important aspect of the coastal dynamics, in particular inside of the bays, is the role of mangroves in coastal stabilization and water quality of the bays (Kendall et al. 2005). Mangroves serve as a barrier reducing the effect of storm surge and flooding against the coast preventing erosion. Additionally, the outlet of the Salt River Gut flows through a mangrove forest which buffers seagrass beds, coral reefs, and other benthic habitats from effects of terrestrial runoff (Kendall et al. 2005).

Coastal Geomorphology

The coastal geomorphology of Salt River Bay is heavily influenced by the Salt River watershed (Kendall et al. 2005). Two main formations underlie the SARI area. The Miocene Kingshill Formation (mostly limestone) underlies the south part of SARI and most of the river drainage. On the north side, the Cretaceous Judith's Fancy Formation is a mixture of volcanoclastics, sandstone, and mudstone with exposed bedrock around the shoreline of SARI (Figure 2.2.2.1, Kendall et al. 2005).

The Salt River, Sugar and Salt River Bays consist of eroded course and fine (terrestrial silt and clay) grain sediments (Kendall et al. 2005). Sediment in Sugar and Salt River Bay consists of two distinct types: carbonate and terrestrial sediments. Carbonate sediments originate from calcareous algae and other benthic organisms (mollusks, *foraminifera*, and echinoids), while terrestrial sediments originate from upland runoff and are transported to the bays by the Salt River (Gerhard and Petta 1974). To the east, Triton Bay has lower presence of terrestrial sediments since it is less influenced by the Salt River Gut. However, the carbonate composition is likely to be similar to that of the other two bays with calcareous algae *Halimeda* being the main source of calcareous sediments (Kendall et al. 2005).

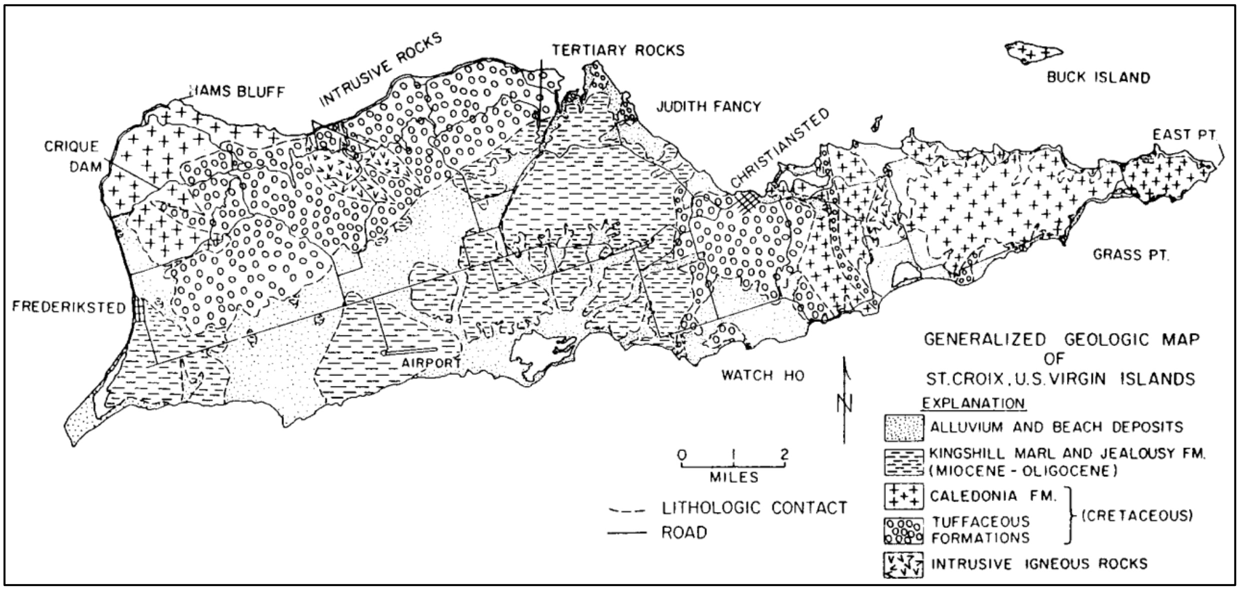

Figure 2.2.2.1. Generalized geologic map of St. Croix. After Whetten (1974) (in Nagle and Hubbard 1989).

<u>Shoreline</u>

The shoreline in SARI can be divided into the west and east shores of the Salt River Bay and shores that surround Sugar and Triton Bays. Shores in the Salt River Bay west and east walls are mostly influenced by longshore and wind generated currents, while tidal dynamics and the Salt River mouth deposition mostly affect shores in the other bays. The length of the shores of the west of the Salt River Bay in general have been relatively stable since 1956. However, the east shores have changed significantly, especially as a result of manmade modifications which included dredging of a saltwater pond (Table 2.2.2.1, Figure 2.2.2.2). However, changes in the length of the shoreline only partially describe the dynamics of the coast. Coastal habitat changes, such as loss or gain of sandy beach areas, can have an important impact for local wildlife (see Section 4.1.1).

The northwest area of the park contains mostly sandy beaches with some rocky shores. While the shores closer to the Salt River mouth are mostly covered by vegetation with some artificial structures and sandy/gravel beaches. The northeast area of the park presents mostly cliffs and rocky beaches with some areas populated by mangroves. The shoreline east of Salt River Bay presents mostly sandy beaches with some rocky shores.

Table 2.2.2.1. Coastal shoreline length for 1954 and 2019. Length of the shoreline in meters was determined by digitization of the landward vegetated boundary using aerial photography (see Data and Methods of Section 4.1.1).

Section	1954	2019
East	1342	2652
West	1431	1302

In general, erosion in the SARI is not a major concern, especially in the western shores where, although there exists movement of the shoreline in some areas, the majority of the shoreline is relatively stable. Conversely, the eastern shoreline has experienced a big change from 1954. However, this change has been mostly a result of man-made modifications. Interestingly, natural processes such as longshore sediment transport are moving sediment from the northeast corner southward (Figure 2.2.2.2, see Section 4.1.1).

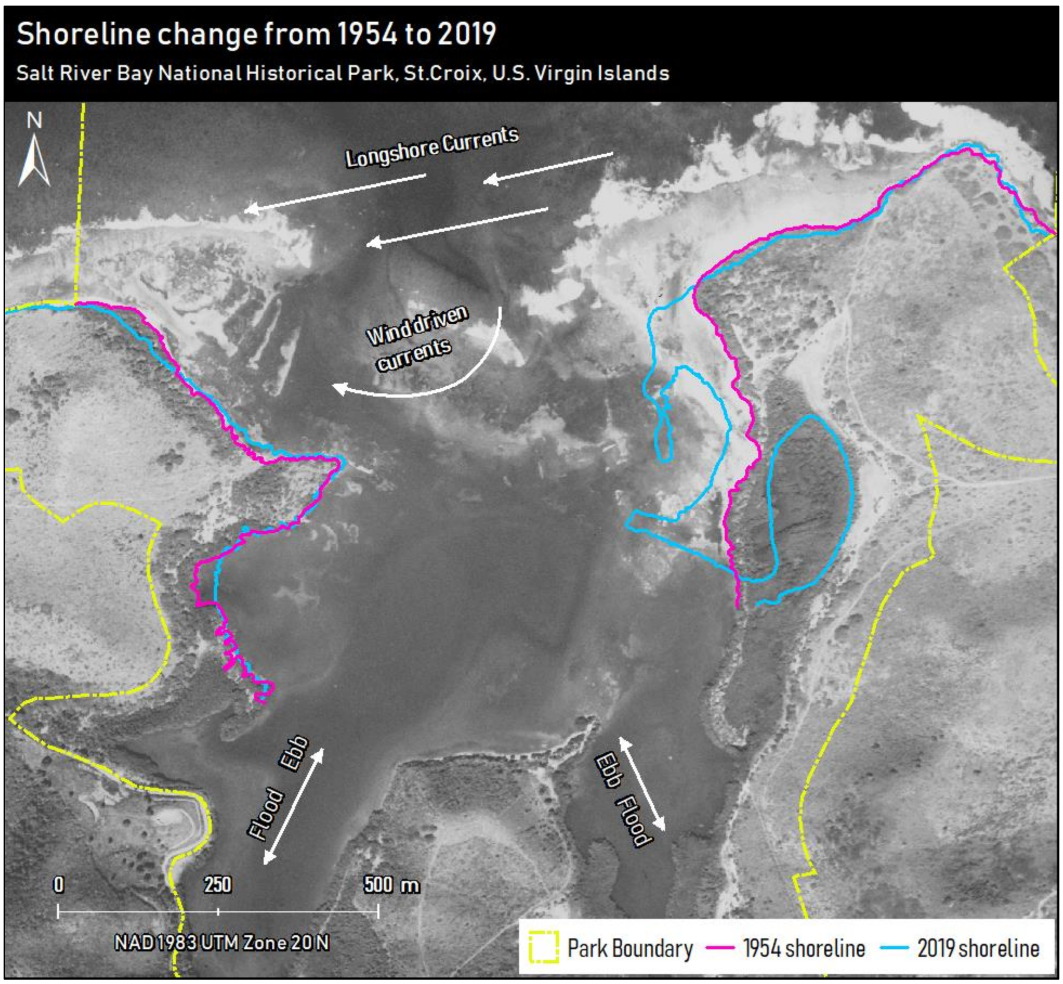

Figure 2.2.2.2. Shoreline for the northern and central shorelines for the period between Salt River Bay for 1956 (pink line) and 2019 (blue line). Park boundary in hatched yellow and currents modeled after Kendall et al. (2005).

Bathymetry assessments for the SARI show that most areas in the Salt River Bay range in depth between 0 to 6 m depth below mean lower low water (MLLW) (Figure 2.2.2.3). These shallow waters play an important role in preventing shoreline erosion by reducing wave energy. SARI also includes areas of deep waters, which are mostly in the northern area of the park along the Salt River canyon. The northeastern shelf, which plays a major role in the sediment transport east-west, presents shallow waters between 0 to 4 m below MLLW.

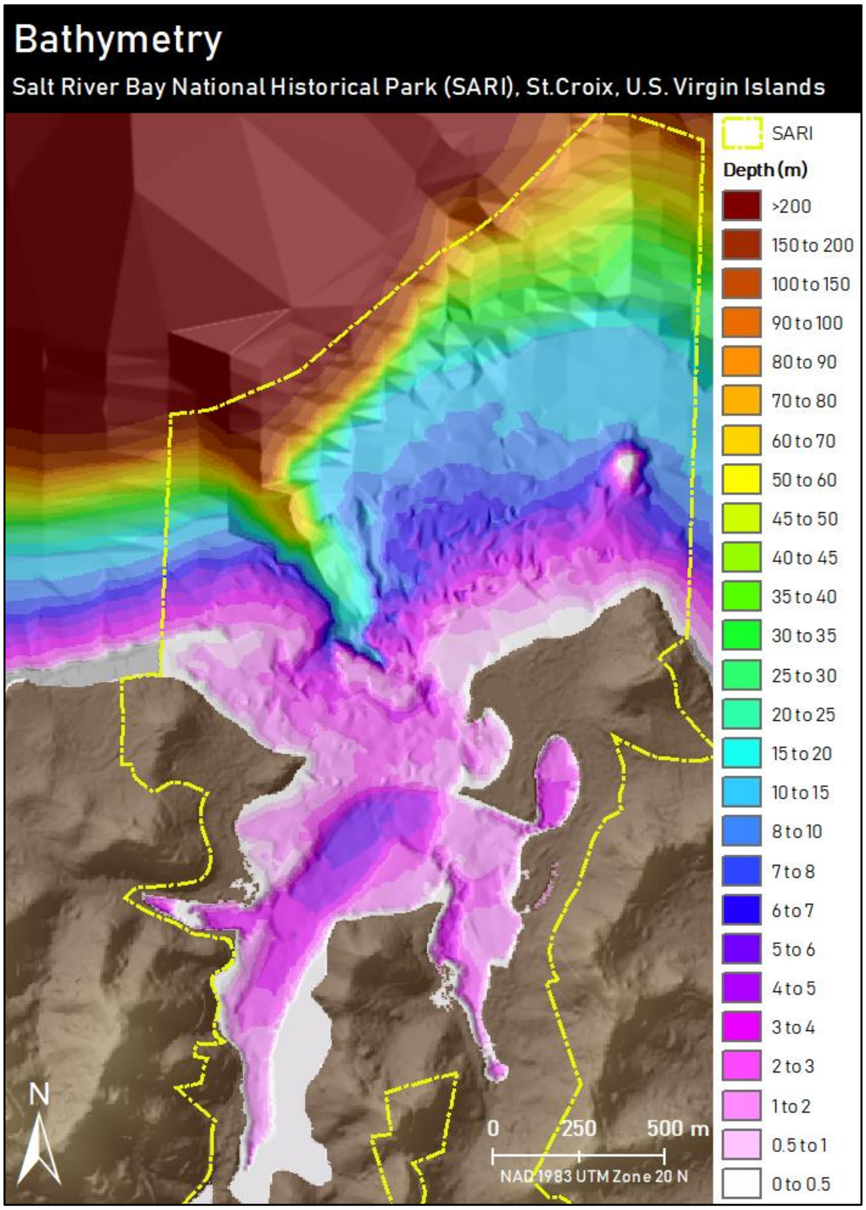

Figure 2.2.2.3. Bathymetry for Salt River National Historic National Park (Battista 2015). Park boundary in hatched yellow.

Bathymetry data (from Battista 2015) were plotted using a density distribution (similar to a histogram) to quantify the most prevalent depths within the protected areas. The plots were constructed using density function in ggplot2 (RStudio, version 1.2.1335). The range in water depths for the park is large, ranging from 0 to −200 m relative to MLLW. However, the majority of park presents shallow water with depths ranging from 0 to −10 m MLLW (Figures 2.2.2.3 & 2.2.2.4).

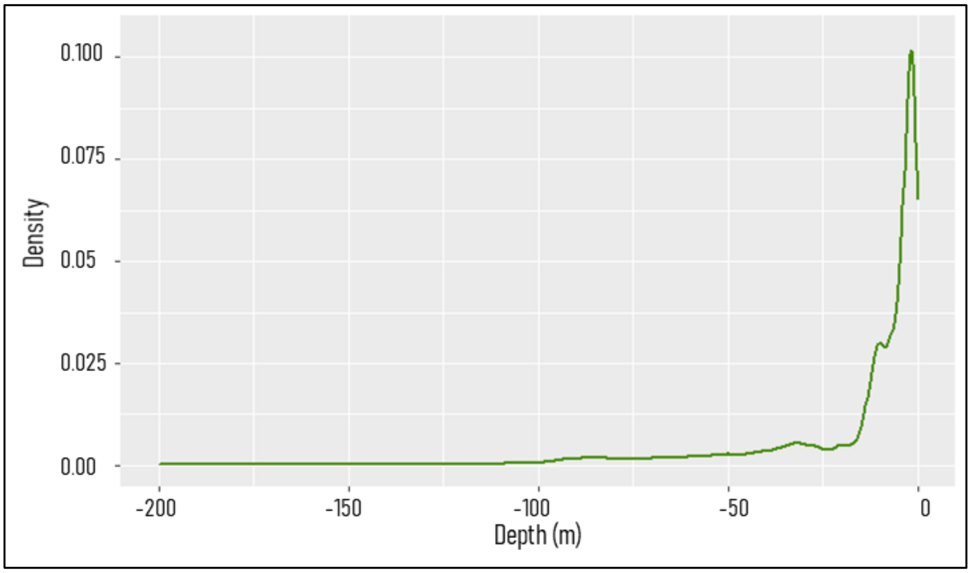

Figure 2.2.2.4. Density distribution for bathymetry estimates for Salt River National Historic National Park (SARI). Higher density values represent higher occurrence. Depth is referenced to MLLW. (https://coast.noaa.gov/data/Documents/Metadata/Lidar/harvest/usvi2011_bathy_m1394_metadata.xml)

Chemical / Physical Conditions

Water Quality

The marine waters around the Salt River Canyon are very popular for SCUBA diving, noted for periods of exceptional water clarity. Turbidity can increase during periods of intense northern swell and sediment resuspension. Waters inside the barrier reef and towards the lagoons are much more estuarine, with some areas near a marina complex and boat chandlery and other areas farthest from the canyon exhibiting low dissolved oxygen and high turbidity (Kendall et al. 2005). Values in these areas occasionally exceed guidelines of the Division on Environmental Protection (DEP) of the USVI Department of Planning and Natural Resources (DPNR) for Class B waters designated for contact recreation and aquatic life use support (Kendall et al. 2005). Land-based sources of pollution are likely the largest threat to water quality in SARI. Section 4.2.1 shows a map of monitoring sites for water quality.

Weather and Climate

The climate in the Virgin Islands is tropical. In SARI, the average high temperature ranges between 84°F and 89°F, with lows between 73°F and 80°F (23°C to 27°C). The temperatures of 98°F (37°C) and 51°F(11°C) are respectively the maximum and minimum temperatures registered for the period March 1951 to December 2019 at the Christiansted Hamilton Field Airport located on St. Croix (NOAA 2020). The coolest months in the year occur from December to March. Average temperatures in the winter are around 73°F (23°C). August through October is the hottest time of the year, with average high temperatures in the upper 80s and low 90s (29°C to 32°C) (data from NOAA 2020).

The rainy season extends from May to December, with a short dry spell in June and July, while the dry season goes from January through April. The months with least precipitation are February and March, while the wettest period is from September to November. The total annual precipitation is of the order of 1,000 millimeters (mm) to 1,200 mm (40 to 47 inches) per year and is generally slightly more abundant on the northern slopes of Buck Island. The maximum 24 hour rainfall registered for the period March 1951 to December 2019 at the Christiansted Hamilton Field Airport is St. Croix was 457 mm (about 18 in) (Figure 2.2.2.5). This precipitation was recorded during the passage of Hurricane Frederick in early August, 1979. NOAA (2020) daily rainfall records show that during the passage of Hurricane Maria on September 20, 2017, the precipitation reached over 130 mm (5 inches) prior to the instrument being damaged by strong winds. Thus, the total amount of rainfall associated with the storm was not recorded. Major rain episodes are commonly linked to hurricanes events. Hurricane season in the region starts officially on June 1 and extends until November 30, with peak months for storms from August to October. A detailed discussion of hurricanes can be found in Section 2.2.3.

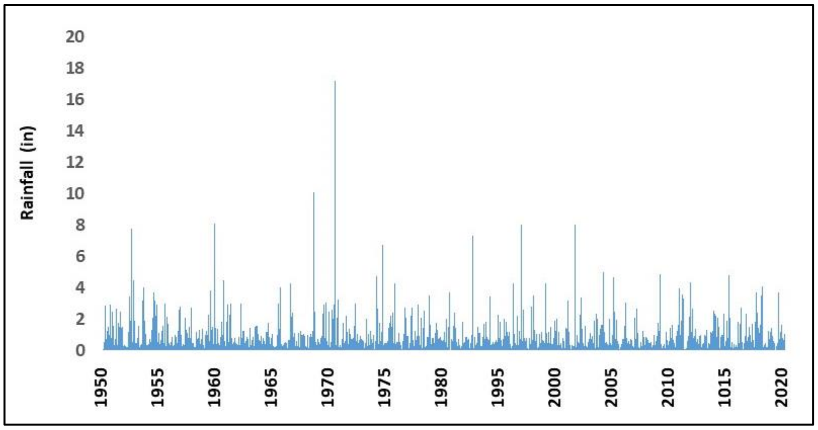

Figure 2.2.2.5. Maximum daily rainfall registered for the period March 1951 to December 2019 at the Christiansted Hamilton Field Airport in the Island of St. Croix (NOAA 2020).

The weather in the Caribbean is also modulated by the trade winds (easterlies) blowing east to west. The strong easterlies can sometimes bring clouds of African dust from the Sahara; millions of tons of dust can be transported each year, affecting air quality and potentially affecting marine life, including coral reefs. The intensity of the winds in the Virgin Islands vary, but the strongest wind episodes not linked to hurricanes occur from December to February and correspond to systems with winds from the north, aka Christmas Winds. Maximum average daily wind speed is 27.74 miles per hour (mi/hr), and the fastest 2 minute wind speed, registered for the period August 2000 to December 2019, was 61 mi/hr, as recorded at the Christiansted Hamilton Field Airport in St. Croix (NOAA 2020).

Data for weather parameters presented in this chapter were obtained from the NOAA GHCN (Global Historical Climatology Network)-Daily database. GHCN-Daily is a composite of climate records from numerous sources that are merged and then subjected to a suite of quality assurance reviews (Menne 2012). The archive includes over 40 meteorological parameters, including temperature daily maximum/minimum, temperature at observation time, precipitation, snowfall, snow depth, evaporation, wind movement, wind maximums, soil temperature, cloudiness, and more (NOAA 2020). The Caribbean region has undergone relatively consistent seasonal rainfall periods, small annual temperature fluctuations, and a variety of extreme weather events, such as hurricanes, tropical storms, and droughts. Notwithstanding, these patterns are changing and are projected to be increasingly altered due to climate change.

Climate change is anticipated to add to the stresses of coastal environments by modifying temperature and precipitation patterns, increasing the likelihood of extreme precipitation events, and accelerating rates of sea level rise. Changing climate and weather patterns interacting with human activities, are affecting land use, air quality, and resource management and are posing growing risks to food security, the economy, culture, and ecosystems services. Some coral reefs in the Caribbean are already experiencing transformational changes (USGCRP 2018).

Climate variations due to these large-scale patterns directly impact water resources in the U.S. Caribbean because the islands largely rely on surface waters and consistent annual rainfall to meet freshwater demands. According to recent studies (Campbell et al. 2011; Henareh et al. 2016), the Caribbean is envisaged to have longer dry seasons and wetter rainy seasons. Extended dry seasons are expected to increase the stress on already scarce and vulnerable water resources. Dependable and safe water supplies for U.S. Caribbean communities are threatened by drought, flooding, and saltwater contamination due to sea level rise (Cashman et al. 2010). Air and seawater temperatures are predicted to rise. Rising air and water temperatures along with changes in precipitation are intensifying droughts.

St. Croix, like so many other islands in the Caribbean, is among the Earth's most vulnerable places to the impacts of climate change, particularly sea-level rise. Sea level rise, combined with stronger wave action and higher storm surges, will worsen coastal flooding and increase coastal erosion, likely leading to diminished beach area, loss of storm surge barriers, decreased tourism, and negative effects on livelihoods and well-being (USGCRP 2018).

The NOAA-developed Sea Level Rise (SLR) and Coastal Flooding Impacts Viewer can be used to visualize the impact of high tide flooding and sea level rise. This viewer presents coastal managers and scientists with a preliminary look at SLR and coastal flooding impacts and helps gauge trends and prioritize actions for different scenarios. The viewer is a screening-level tool that uses nationally consistent datasets and analyses presented in a Web mapping application format using ESRI's ArcServer and Adobe's FLEX technology (http://www.csc.noaa.gov/digitalcoast/tools/slrviewer/). Figure 2.2.2.6 shows a simulation of the extent of flooding in SARI during high tide.

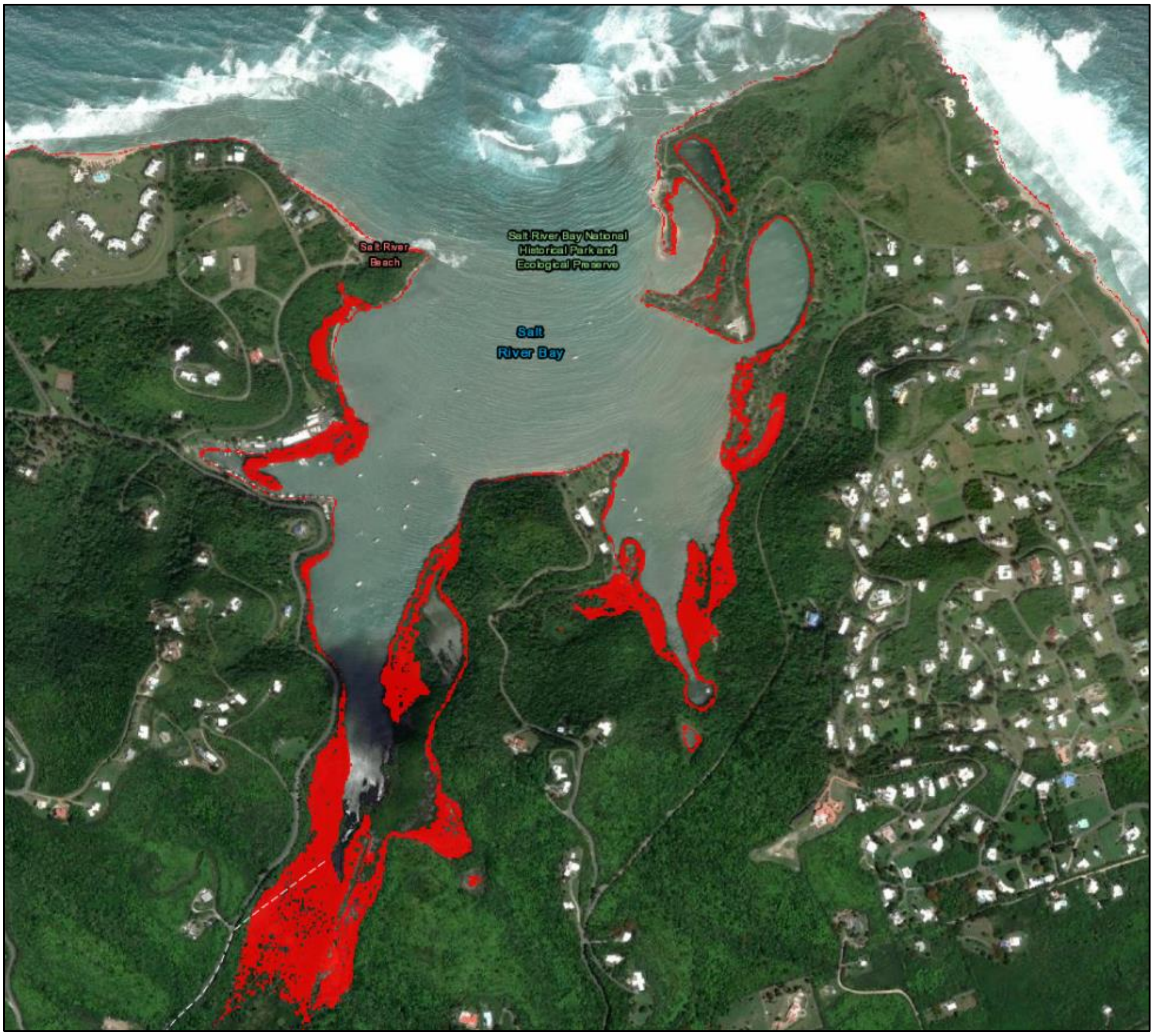

Figure 2.2.2.6. High Tide flooding simulation for SARI. Red marking depicts the flooded areas during Mean High Water (MHW) Image derived using the NOAA SLR and Coastal Flooding Impacts Viewer (https://coast.noaa.gov/slr/#/layer/slr/).

Figure 2.2 2.7 shows the impact on of a 1.2 m (4 ft) sea level rise above mean higher high water (MHHW) in SARI. In the graphic display provided by the viewer, areas that are hydrologically connected (according to the digital elevation model used) are shown in shades of blue that represent depth of inundation. Low-lying areas, displayed in green, are hydrologically "unconnected" areas that may flood. These are determined solely by how well the elevation data capture the area's hydraulics (NOAA 2011). Water levels are shown as they would appear during Mean Higher High Water (MHHW) and do not take into consideration future erosion, subsidence, or man-made alterations of the shoreline.

In addressing climate change, it is important to be aware that the islands have unique issues related to data availability and the capacity to develop datasets comparable to those available for the continental United States. For example, the small size of the islands, particularly the USVI, affects the availability and accuracy of downscaled climate data and projection.

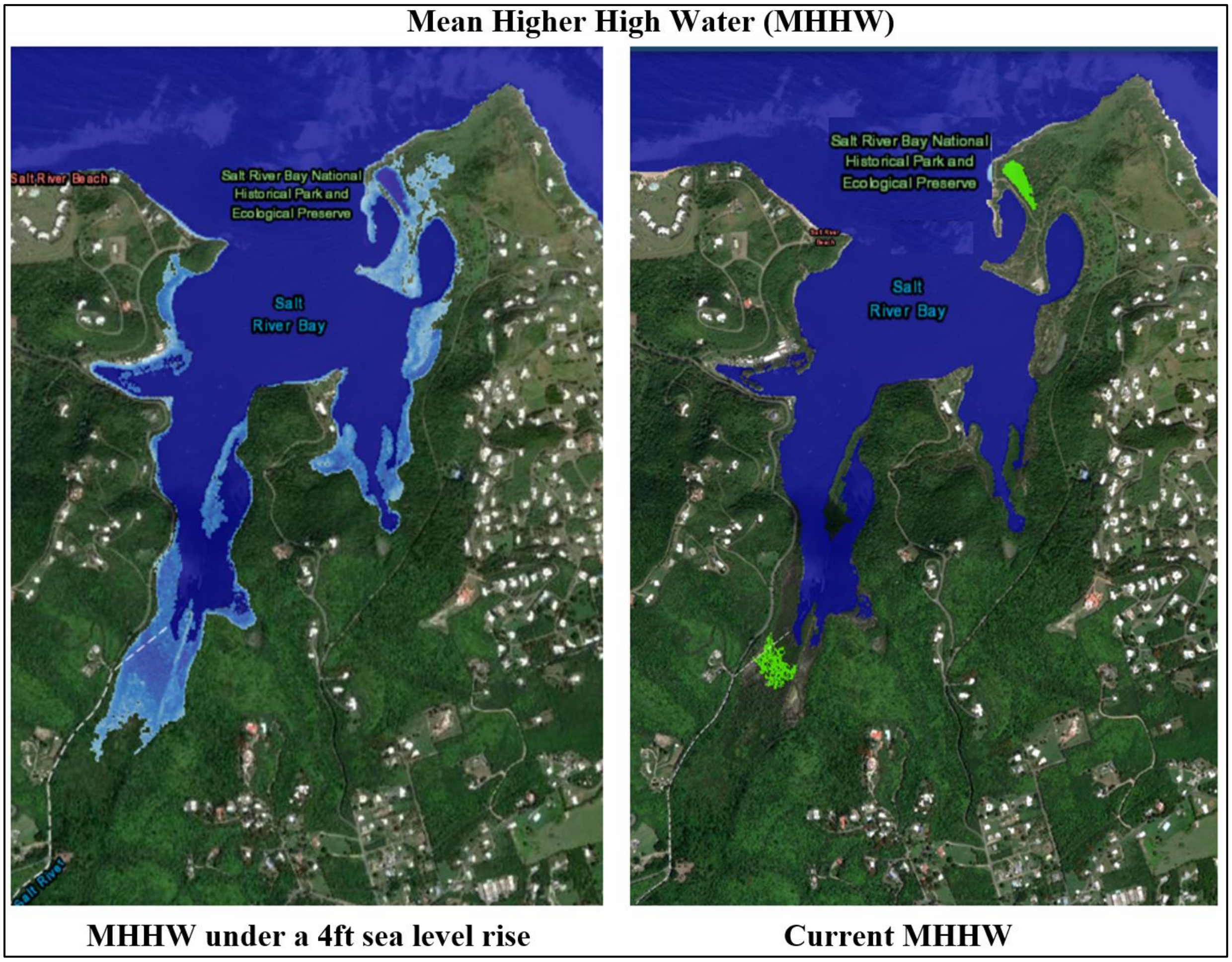

Figure 2.2.2.7. Sea level rise simulation. SARI. Right panel: SARI present coastline. Left panel: SARI coastline for a 4 feet rise corresponding to the estimated sea level in 2080. Low-lying areas, displayed in green, are hydrologically “unconnected” areas that may flood. Graphic display under this scenario derived using the NOAA SLR and Coastal Flooding Impacts Viewer (https://coast.noaa.gov/slr/#/layer/slr/).

Air Quality

The National Park Service participates in several national, multi-agency air quality monitoring networks. These networks focus on ozone, visibility, particulate matter, and atmospheric deposition of nitrogen, sulfur, and mercury. The trade winds blowing across the tropical Atlantic Ocean bring millions of tons of dust from the Sahara and Sahel regions of Africa to the Caribbean every year. The dust that reaches the Caribbean limits visibility and research indicates that this dust also contains viable bacteria, viruses, and fungi, nutrients, metals, and persistent organic pollutants (e.g., pesticides, PAHs, PCBs) (Kellogg and Griffin 2003; Garrison et al. 2011). During the periods of high wind blown dust concentration, known as dust pulses, the number of microbes present in the air can be as much as ten times higher than during normal times. This condition represents a hazard to the health of humans and ecosystems. For example, a particular soil fungus detected, *Aspergillus sydowii*, causes sea fan disease and results in widespread coral mortality (Kellogg and Griffin 2003).

Certain chemicals transported by the wind may also have harmful effects on surface waters, marine environments, and vegetation similar to those found in SARI. Nitrogen and sulfur can contribute to ocean acidification. Ocean acidification, caused by greenhouse gas emissions, may contribute to the degradation of coral communities (Sullivan et al. 2011).

African dust or human-caused haze from fine particles of air pollution may also affect visibility. There is an air quality permanent monitoring site for air quality on SARI. Observations made in this station indicate a reduction of the average natural visual range from about 120 miles (without pollution) to about 65 miles on days with pollution. During high pollution days, the visual range can be reduced to below 40 miles (NPS 2019d) (Figure 2.2.2.8).

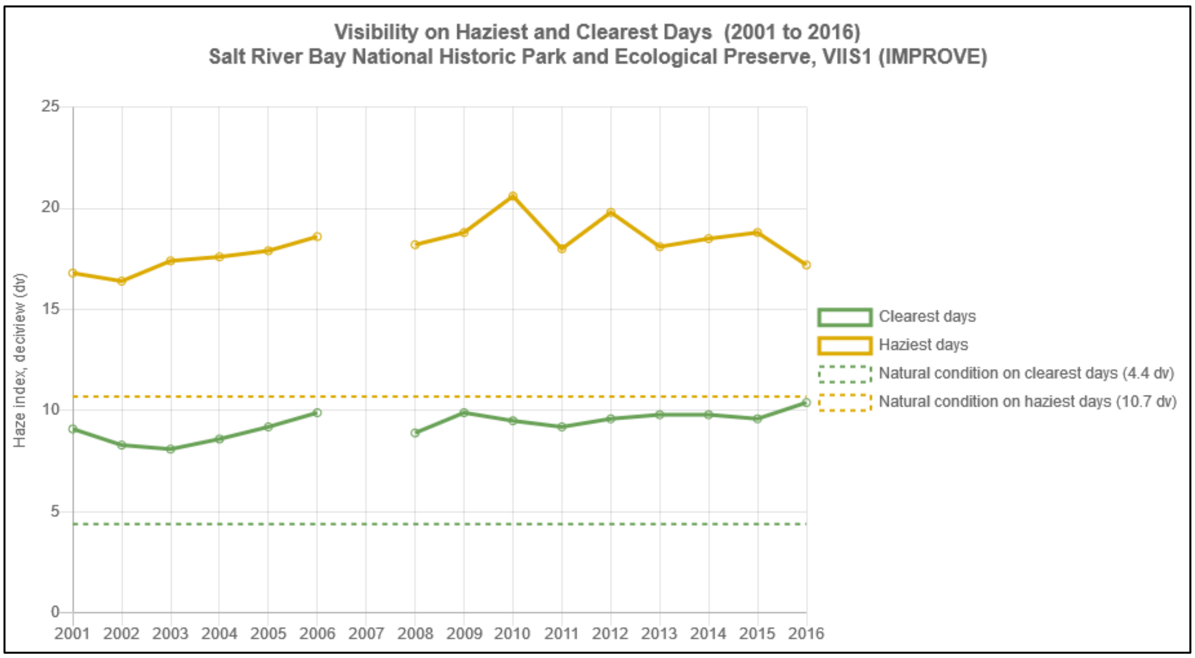

Figure 2.2.2.8. Visibility on haziest and clearest days at the Salt River National Historic and Ecological Preserve during the period 2001–2016 (NPS 2019d)

Surface Hydrology

There are no rivers or permanent streams in St. Croix. However, precipitation associated with tropical storms and hurricanes can be significant and last for several days. Over the period from August to December, very intense rains can fall within very short periods. During such episodes, water runoff can collect in streambeds and turn into strong temporary streams (Rogers et al. 2008). Stormwater runoff can cause considerable erosion which in turn can have profound effects on local marine sedimentation (Hubbard et al. 1981; KellerLynn 2011). The Salt River is not a perennial river, but it flows during certain periods of the year. The Salt River watershed (see Section 4.3.1) drains an area of approximately 1,360 ha (hectares). The Salt River flows into the Salt River Bay traversing the southern lands of SARI. More detail on the hydrology of the Salt River basin is provided in Section 4.3.

Ocean Currents

A characteristic feature of the oceanography of the Caribbean Sea is the exchange of water with the Atlantic Ocean, which takes place through a number of passages between the islands and the shallow plateaus (Figure 2.2.2.9).

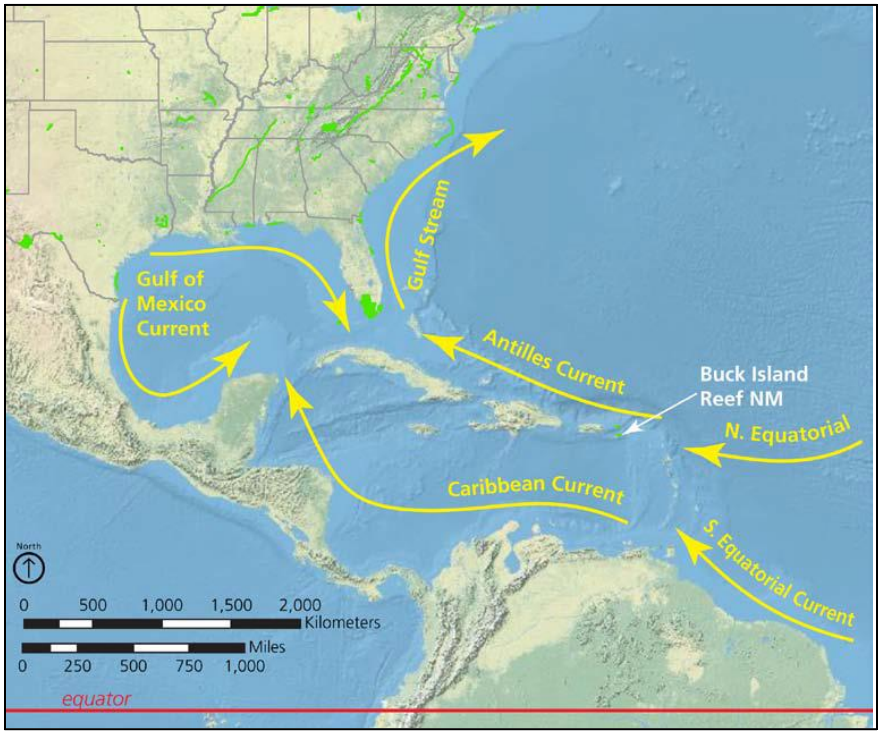

Figure 2.2.2.9. Major oceanographic currents. Global circulation around the equator drives oceanographic currents in the Caribbean. Currents around the Island of St. Croix flow from east to west. Current directions after Hubbard (1989). Aerial imagery from ESRI Arc Image Service, USA Prime Imagery, compiled by Jason Kenworthy (NPS Geologic Resources Division). Image and caption from KellerLynn 2011).

The major surface and near-surface exchange with the Caribbean occurs through the eastern passages. Surface flow is fed into de Caribbean by the Guinea[2] and the Atlantic North Equatorial

[2] The Atlantic South Equatorial Current (SEC) flows westward toward the Brazilian shelf, and or splits at Cabo de Sao Roque, near 16°S with one branch, the stronger of the two, heading northwards as the North Brazil Current (NBC) and the other, weaker southwards branch, as the Brazil Current. The NBC flows north along the northeastern coast of South America, it reaches French Guiana, where part of it separates from the coast and turns to join the North Equatorial Counter Current moving eastward. The rest of the NBC continues flowing northwestward to form the Guiana Current. The Guiana (Guyana) Current has been previously referred to as the South Equatorial Current, the North Brazil Coastal Current, and the North Brazilian Current. The confusion surrounding its name is due partly to the seasonal change in flow of nearby currents (https://oceancurrents.rsmas.miami.edu/atlantic/atlantic.html)

Current (Watlington and Donoso 1996). The Caribbean Current flows at an average rate of 35 to 45 cm (13 to 18 inches) per second in a westward direction and is modulated by the annual migration of the Intertropical Convergence Zone (ITCZ) (Donoso 1990). Upon flowing into the Gulf of Mexico, the current enters a clockwise loop, and ultimately moves out of the Gulf south of Florida (Keller Lynn 2011). Part of the Atlantic North Equatorial Current that has flowed on the eastern side of the Antilles as the Antilles Current merges with the with the Florida Current which issues from the Gulf through the Florida Straits to form the initial portion of the Gulf Stream system.

In the vicinity of SARI, the speed of the longshore ocean current is 5 cm to 10 cm (2 in to 4 in) per second (Wust 1964; Donoso 1990; Kendall el al. 2005). These currents are not as intense as those in the central portions of the Caribbean or in the western side of St. Croix, where much stronger currents are observed. The SARI shoreline is divided in three sections, namely the northern, central and southern sections. The coastal current in the northern section is associated with the ocean current, whereas the central area is modulated mostly by wind-drive currents. The direction of the flow in the southern section is driven by the tides and runoff entering the bays from the neighboring land areas. In the area of the Salt River Canyon, flow rate down the canyon is 10 to 15 cm/s and during ebb tides it can reach 20 cm/s (Kendall et al. 2005).

Marine Communities

Marine Plants

Seagrass

A mostly continuous seagrass meadow covers the mouth of Salt River Bay from south of Columbus Landing into East Cove with patchy (less dense) seagrass extending out to the fringing coral reefs and a smaller meadow adjacent to the beach at the end of the central peninsula. *Thalassia testudinum*, *Syringodium filiform* and *Halodule wrightii* (to a lesser extent) constitute these meadows while the more depth tolerant *Halophila decipiens* is found seasonally on the canyon floor (Kendall et al. 2005). A more detailed assessment of the condition of seagrass is presented in Section 4.5.1.

Macroalgae

Macroalgae is found in shallow areas within Sugar Bay, the outlet for the Salt River, along the eastern coast of Triton Bay and Salt River Bay, as well as in coral reef and hardbottom habitats in SARI. Ground truthing of benthic habitats for mapping in 2000 revealed muddy bottom in water deeper than 2 m, and patchy macroalgae in areas shallower than 2 m (Kendall et al. 2005). Correspondingly, calcareous macroalgae are the primary contributors to the sediments found in shallow areas in the northern parts of the bay. Extent of macroalgae in Salt River Bay as mapped from 2000 aerial photography is shown in Figure 2.2.2.10 and the area (ha) in three cover classes (patchy and continuous) is shown in Table 2.2.2.2.

Within Salt River Canyon rhizophytic algae can be found in depths up to 100 ft (30.5 m) (Kesling 1990). After Hurricane Hugo caused damage to corals and sponges along the walls of Salt River Canyon, video monitoring observed ~5% of the dead coral was covered with macroalgae and 70–80% was covered with turf algae (Nemeth et al. 2003). One study found that macroalgae may be responsible for a large percentage of the total primary production of the reefs (Rogers and Salesky 1981).

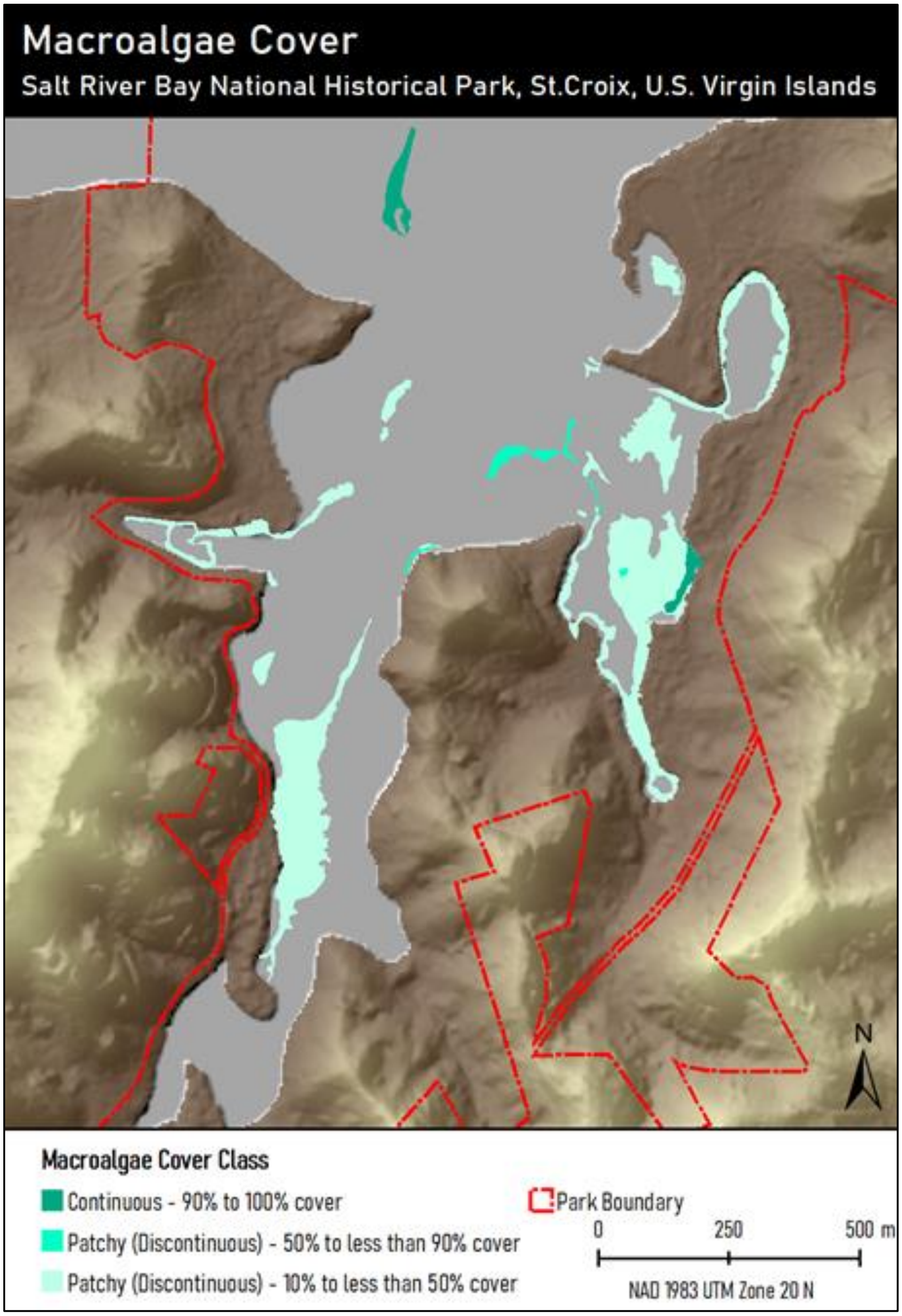

Figure 2.2.2.10. Macroalgae cover classification within SARI as derived from 2000 aerial imagery (Kendall et al. 2005). Percent cover is depicted in three cover classes.

Table 2.2.2.2. Number of polygons and total area (ha) by cover class of macroalgae in 2000 in Salt River Bay (Kendall et al. 2005, Table 2.2).

Cover Class	# of polygons	Area (ha)
10–49.9%	15	10.9
50–89.9%	6	0.4
90–100%	2	0.8

Microalgae

The dinoflagellate primarily responsible for bioluminescence (Figure 2.2.2.11) in Salt River Bay is *Pyrodinium bahamense* var. *bahamense* supported by the shallow water, mangrove-lined habitat (Zimberlin 2013). Mangrove Lagoon in the southern Bay (locally referred to as Bio Bay) has a small inlet and is considered a biobay because of the regular bioluminescence. As an important draw for ecotourism groups, continued monitoring of abiotic (e.g., dissolved oxygen, temperature) and biotic (e.g., phytoplankton biomass) factors and environmental assessments are recommended as nearby development continues.

Figure 2.2.2.11. Bioluminescence created by dinoflagellates nearshore, Photo credit: iStock

Marine Invertebrates

Corals

On the outer portions of SARI, near and within the canyon, stony corals (Order Scleractinia) are the most important habitat forming species, supporting the highest diversity of plants and animals. Hardbottoms and coral reefs cover approximately half of the benthic habitat of Salt River. These reefs support about 30 species of stony corals including the US Endangered Species Act listed species: elkhorn coral (*Acropora palmata*), staghorn coral (*Acropora cervicornis*), pillar coral (*Dendrogyra cylindrus*), rough cactus coral (*Mycetophyllia ferox*), lobed star coral (*Orbicella annularis*), mountainous star coral (*Orbicella faveolata*), and boulder star coral (*Orbicella franksi*). These communities have suffered from bleaching events, but many species present are more resistant to bleaching related mortality. The deeper canyon walls support mesophotic corals reefs (30 m depth) composed almost predominantly of lettuce corals (*Agaricia lamarcki* and *Agaricia grahamae*). These communities extend to about 65 m, where they transition to black corals (antipatharia), octocorals, and sponges. Mesophotic communities have been impacted by thermal stress and bleaching, but have shown a relatively higher degree of recovery (see Section 4.6.1). Section 4.6.1 presents a map of marine habitats and monitoring sites for coral reefs of SARI.

Long spine urchins

The long spined sea urchin (*Diadema antillarum*) was one of the most important grazing herbivores in SARI due to its ability to intensively overgraze reef surfaces keeping them free of coral competing species, such as macroalgae, and promoting coral recruitment (Edmunds and Carpenter 2001). The urchins were decimated by a Caribbean-wide epizootic of unknown cause in the early 1980s (Lessios 1988). Typical abundances on shallow coral reefs prior to the die-off were greater than 100 urchins per 100 m^2. Between 2002 to 2017, abundance of urchins was always less than one urchin per 100 m^2 at both a deep and shallow monitoring site, with 87% of sampling periods recording no urchins. Figure 2.2.2.12 presents the density of long-spined sea urchins in SARI. There appeared no trend of increase compared to historical abundances. Descriptions of the long-term sites are provided in Section 4.6.1. Note that deep and shallow sites were always sampled on the same day, so data points overlap at zero urchins. The deep site was not sampled until 2009.

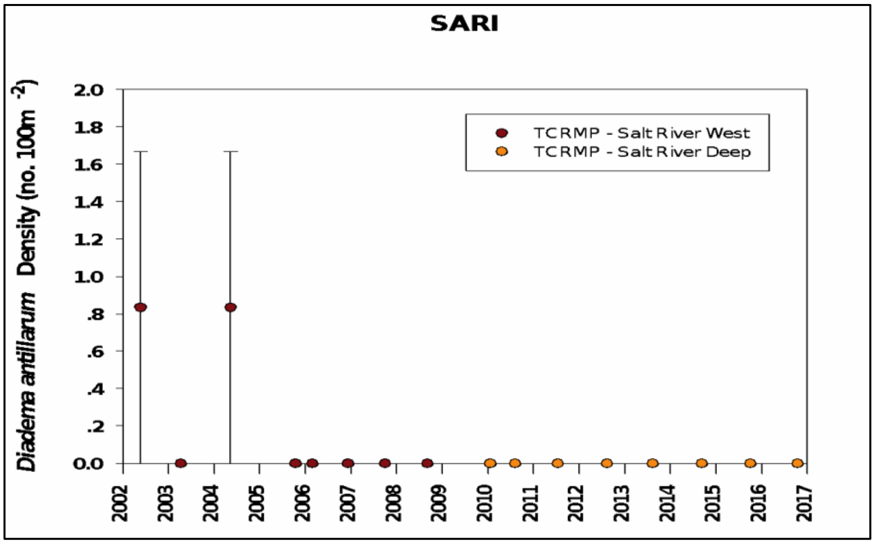

Figure 2.2.2.12. Density of the long-spined sea urchin (*Diadema antillarum*) at coral reef monitoring sites in SARI from USVI Territorial Coral Reef Coral Monitoring Program (Ennis et al. 2019).

Queen conch and spiny lobster

Caribbean spiny lobster (*Panulirus argus*) and queen conch (*Lobatus gigas*) have historically been important fisheries species in the USVI. Fish and shellfish population declines in the 1960s–1970s prompted fishing regulations to be signed into law in 1972 (Virgin Islands Code). Several amendments in the following years established further restrictions on lobster and queen conch, such as minimum size requirements and seasonal closures. Additionally, in 1995 terrestrial and marine organisms became protected within park boundaries. The Salt River Bay National Historic Park and Ecological Preserve provides a unique semi-enclosed protected habitat due to the presence of the nearshore submarine canyon. Although there have been no studies within the boundaries of the SARI that focus on lobster populations (Richter et al. 2018), the biennial National Coral Reef Monitoring Program (NCRMP) found lobster densities to be very low over several sampling periods (2012–2017; Richter et al. 2018), with only a single lobster being recorded during the 2017 sampling. Figure 2.2.2.13 depicts lobster and conch densities from the 2017 surveys.

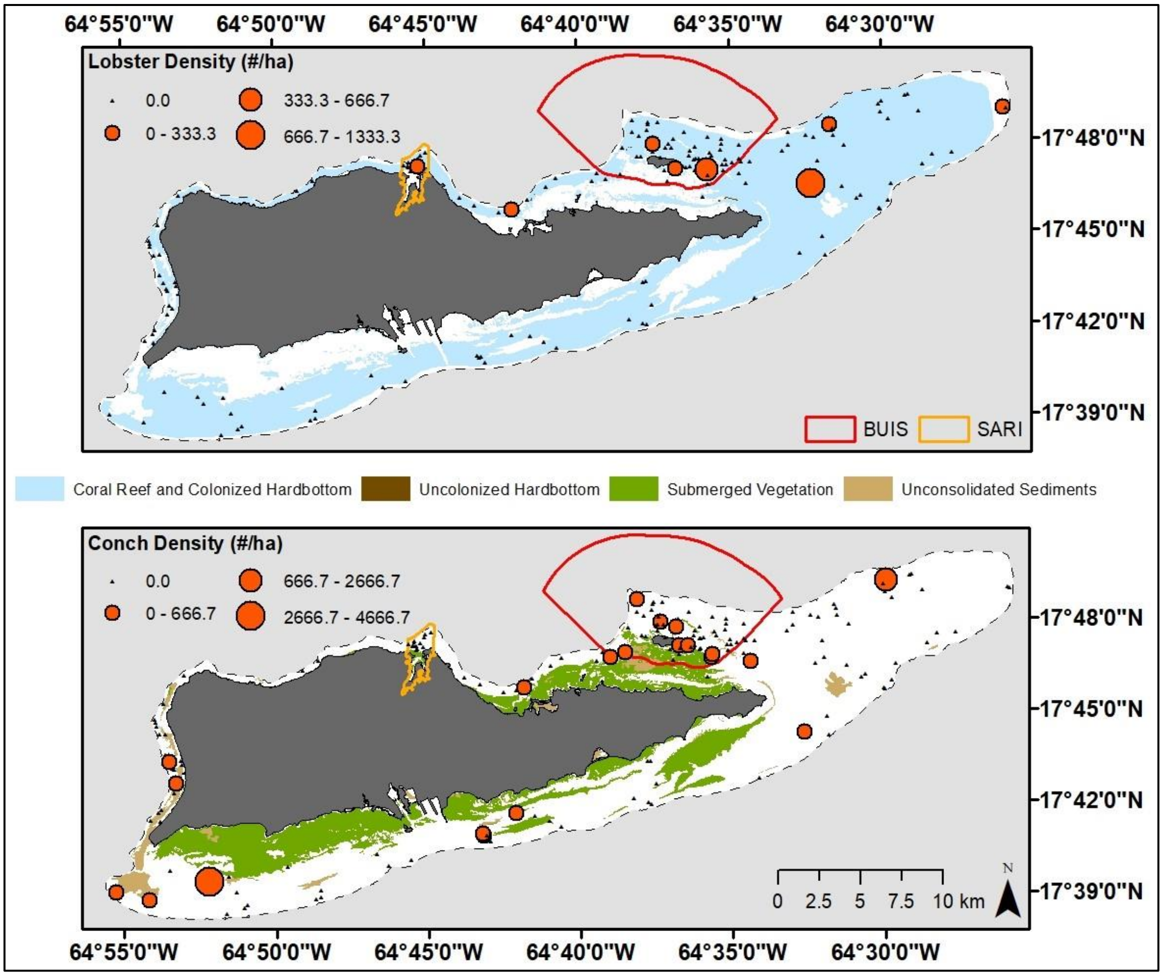

Figure 2.2.2.13. Lobster (top) and queen conch (bottom) densities (#/ha) calculated from the most recently completed National Coral Reef Monitoring Program (NCRMP) sampling (2017). Boundaries of Buck Island Reef National Monument (BUIS) in red and SARI in orange.

Early studies of queen conch populations found those within the deeper water of the SARI to be more abundant than those in shallow water, and it was suggested that the dense deep water populations should be protected (Coulston et al. 1987). The NCRMP recorded no conch within the SARI in the most recent sampling period (2017); however, populations could be underestimated since the protocol only samples on hardbottom habitat.

Marine Vertebrates

<u>*Reef Fish*</u>

Diverse fish assemblages are supported by the multiple habitats within SARI (NPS 2015). The majority of studies done on reef fish and their populations in the park have been conducted within Salt River Canyon (Figure 2.2.2.14) and along the canyon walls during saturation missions from NOAA's Hydrolab between 1977 and 1985. In 2011, Dorfman & Battista performed a gap analysis of ecosystem data and found that SARI does not have regular fish surveys and the limited available data were provided by the UVI-CMES reef fish census. Kendall et al. (2005) compiled a list of nearly

200 reef fishes observed in SARI's coral reefs and pointed out the need to study the rest of the park's shallow reefs. From 2012 to 2019, several surveys have been conducted by National Park Service (NPS), National Oceanic Atmospheric Administration, and the University of Virgin Islands (UVI), referred to as National Coral Reef Monitoring Program (NPS-NCRMP-UVI). Analyses of the NCRMP dataset are discussed in Section 4.7.1

Figure 2.2.2.14. Research diver descends the Salt River Canyon wall among schools of planktivorous fish (Photo credit: Sonora Meiling).

Pelagic Fish

Salt River Canyon extends beyond park boundaries providing access to pelagic fish. The variety of shallow and relatively protected habitats of SARI provide nursery habitat for some pelagic fish species (e.g., Figure 2.2.2.15) and is therefore recognized as an ecological link between the shallower habitats within SARI and open ocean habitats (Kendall et al. 2005). Pelagic fish (e.g., bar jack (*Caranx ruber*), horse-eye jack (*Caranx latus*), and cero (*Scomberomorus regalis*)) are often observed during reef fish surveys along the canyon walls.

Figure 2.2.2.15. Two species of pelagic fish caught near Salt River Bay on a commercial fishing charter, Left: Atlantic tarpon (*Megalops atlanticus*), Right: bonefish (*Albula Vulpes*), Photo credit: Captain Colt Cook, Captain Cook Charters, www.stcroixfishingadventures.com

<u>Sea Turtles</u>

Historically, sea turtles in the USVI have played an important role in the local culture (e.g., as food and inspiring art) and economy (e.g., through the sale of green turtle meat and hawksbill jewelry). Throughout the USVI, sea turtle populations have declined because of habitat loss, hunting to meet the demand of restaurants, and nest predation by non-native mongooses and dogs (Nellis and Small 1983). The USVI prohibited the take of hawksbill (*Eretmochelys imbricata*) and leatherback (*Dermochelys coriacea*) turtles in 1972 prior to the 1973 U.S. Endangered Species Act that added protection for green turtles *(Chelonia mydas*) (Platenberg and Boulon 2006). SARI offers diverse habitat for sea turtles (Figure 2.2.2.16 and 2.2.2.17), however, no targeted research has been conducted on sea turtles in SARI. The Territory does not conduct nesting beach surveys at Columbus Landing, which is the only beach that might support sea turtle nesting (C. Pollock 2021, personal communication).

REEF (2018) catalogs fish surveys conducted by both novice and expert observers on SCUBA. From 92 surveys conducted between May 1996 and February 2017, 10 turtle sightings were recorded (Table 2.2.2.3). One hawksbill turtle (*Eretmochelys imbricata*) was sighted in each of the three surveys conducted on the West Wall of Salt River Canyon. Observers reported sighting a hawksbill in four surveys and a green turtle (*Chelonia mydas*) in each of three surveys of the East Wall of Salt River Canyon.

Figure 2.2.2.16. Hawksbill turtle as seen in Great Lameshur Bay, St. John. Photo credit: Caroline Rogers, NPS.

Figure 2.2.2.17. Green turtle as seen in Leinster Bay, St. John. Photo credit: Caroline Rogers, NPS.

Table 2.2.2.3. Turtle sighting locations, data, and species. Zones names are retained from data provided by REEF (2018).

Zone	Latitude	Longitude	Date	Species
Salt River (Inner East Wall)	17 47.162	−64 45.469	6/26/2005	Hawksbill
Russ' Rock/Salt River Canyon East Wall	17 47.09	−64 45.26	8/16/2007	Hawksbill
Russ' Rock/Salt River Canyon East Wall	17 47.09	−64 45.26	10/22/2007	Hawksbill
Salt River (Inner West Wall)	17 47.024	−64 45.480	2/26/2009	Hawksbill
Salt River (Inner East Wall)	17 47.162	−64 45.469	2/9/2010	Green
Shooters (West Wall)	17 47.114	−64 45.605	10/14/2010	Hawksbill
Shooters (West Wall)	17 47.114	−64 45.605	2/5/2012	Hawksbill
Salt River (Inner East Wall)	17 47.162	−64 45.469	6/16/2012	Green
Salt River (Inner East Wall)	17 47.162	−64 45.469	2/11/2014	Hawksbill
Salt River (Inner East Wall)	17 47.162	−64 45.469	2/6/2016	Green

Sharks and Rays

The connection to deep water habitat through Salt River Canyon and the high biomass of fish along the reefs often attract large marine fauna such as sharks and rays (Figure 2.2.2.18). Dive shops often promote the likelihood of seeing hammerheads, blacktip sharks and eagle rays to their potential customers, and sharks and rays are often recorded during fish surveys conducted by novice and expert personnel (REEF 2018). During 92 surveys completed from 1996 to 2016, observers recorded 18 requiem sharks (n = 16 Caribbean reef shark, *Carcharhinus perezi*, n = 2 blacktip sharks, *Charcharhinus limbatus*) and 24 southern stingrays (*Dasyatis americana*). No targeted research has been done on sharks and rays in SARI.

Figure 2.2.2.18. (Top) Blacktip shark in front of a Caribbean reef shark visits Salt River Canyon, photo credit: http://www.gotostcroix.com/st-croix-blog/dive-the-salt-river-canyon-sites/ (Bottom) A Caribbean reef shark patrolling reefs in SARI (photo credit, Tessa Code/NPS)

Marine mammals

Both dolphins and whales have been spotted along the north shore of St. Croix. One manatee and calf, likely displaced by Hurricanes Irma and Maria, were observed in late 2017 and early 2018 (C.

Pollock 2021, personal communication). No targeted research has been done on marine mammals within SARI.

Terrestrial Communities

Terrestrial communities occupy 166 ha and range from mangroves and mudflats at the water's edge to semi-deciduous dry forests at the park's highest elevation, 83 meters above sea level (Moser et al. 2011). There is a long history of human occupation of the Salt River Bay area, stretching at least 1600 years with evidence of prehistoric habitation by Igneri, Taino, and Carib peoples (Island Resources Foundation 1993). Extensive land clearing for agriculture, specifically sugar cane and cotton commenced in the 1730s, leading to the cultivation of all flat land on the island of St. Croix (Lewisohn 1970). The impacts of this history on the terrestrial landscape include the presence of secondary forests and numerous non-native plant and animal species. Guinea grass, *Urochloa maxima*, and tan-tan, *Leucaena leucocephala*, are the most problematic non-native plant species, found extensively throughout the park. Today, the largest percentage of the terrestrial land area is covered by semi-deciduous dry forest (~45%), followed by mangrove habitats accounting for another 11%, and coastal grassland covering 10% (Figure 2.2.2.19) (Moser et al. 2011). The following sections describe in detail the dominant terrestrial plant community types found in the park, as well as the terrestrial flora and fauna (birds, herpetiles, mammals, and invertebrates).

Terrestrial Plants

A minimum of 165 species in 57 families have been documented in SARI as part of several surveys and inventories conducted over the span of several decades (Kendall et al. 2005; Moser et al. 2011; NPS 2017b) (Appendix A). However, a complete floristic inventory has not been conducted within SARI and the actual plant diversity is likely much higher. For comparison, nearby Buck Island has ~250 species documented in an area encompassing 71 ha. Several locally threatened and endangered species occur in SARI, including an agave, *Agave eggersiana;* three species of tree, 1) lignum vitae, *Guacium officinale*, stingingbush, *Malpighia infestissima*, and cow-itch, *Malpighia woodburyana;* the wooly nipple cactus, *Mammilaria nivosa*; and two orchids, *Epidendrum ciliare* and *Psychilis bifidum* (Kendall et al. 2005). Found only on St. Croix, *A. eggersiana* was designated as federally endangered under the U.S. Endangered Species Act in 2014. Several *A. eggersiana* individuals were planted at the SARI Visitor Center in 2008 as part of a native restoration program. After these plants matured, they were out-planted to the east side of SARI on Hemmer's Peninsula. Unfortunately, the majority of planted individuals have not survived and it is suspected that the agave snout weevil, *Scyphophorus acupunctatus*, is responsible for the die off (K. Ewen 2021, personal communication).

Control of invasive plant species began in 2009 as a collaboration between the South Florida Caribbean Network (SFCN), the Florida and Caribbean Exotic Plant Management Team (FLC-EPMT), and SARI resource management staff (Moser et al. 2011). The targeted species included those covering large areas within the park—guinea grass, tan-tan, and Madagascar rubber vine, *Cryptostegia madagascariensis*—as well as ginger Thomas, *Tecoma stans*, Spanish bayonet, *Yucca alofolia*, seaside mahoe, *Thepesia populnea*, beach naupaka, *Scaevola sericea*, and coconut palm, *Cocos nucifera*. In 2012 and 2014, the aforementioned invasive exotic species were treated using a combination of mechanical and herbicide treatments across ~70 acres of semi-deciduous forest, woodland, shrubland, and coastal grassland (NPS 2013; NPS 2014). Reintroduction of native hardwood species in coastal shrubland commenced in 2012 (NPS 2012; NPS 2015).

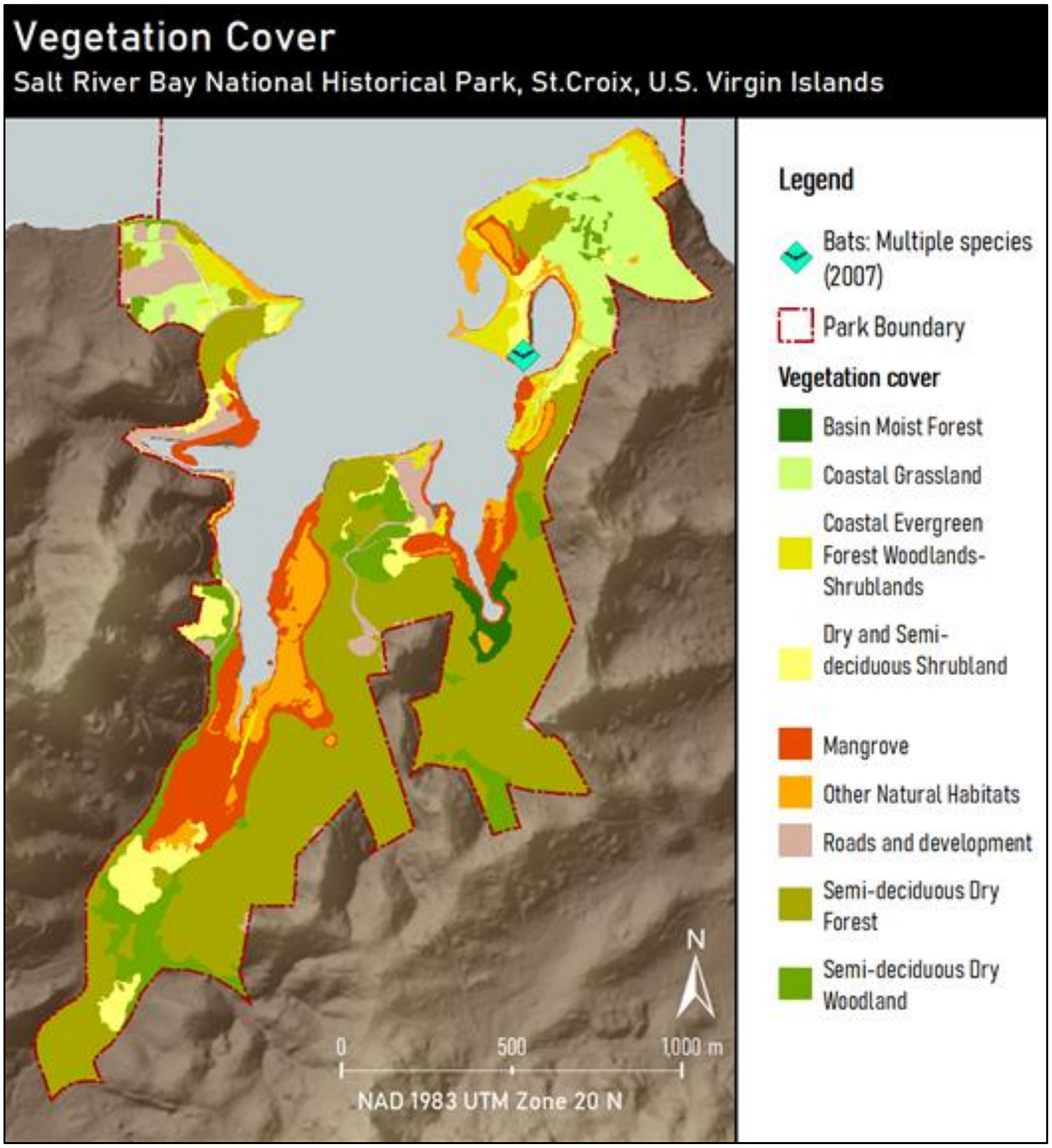

Figure 2.2.2.19. Land cover classification of major vegetation types mapped in Salt River Bay National Historic Park and Ecological Reserve. Classes aggregated from Moser et al. (2011). Bat sightings from Fly By Night, Inc. (2017).

Mangroves

Mangrove communities within the park include 18 ha of forests, woodlands, and shrublands located along the Salt River Bay and are comprised of three species of mangrove: black, *Avicennia germinans*, red, *Rhizophora mangle*, and white mangrove, *Laguncularia racemosa* (Figure 2.2.2.19) (Moser et al. 2011). This constitutes the largest area of mangroves remaining in the Territory (USVI DPNR 1992). Prior to colonization by Europeans, the bay was likely fringed with mangroves along is entirety (Gerhard and Bowman 1975). The island of St. Croix has lost over 50% of its original mangrove cover; making the remaining mangroves in Salt River Bay a vital resource (Island Resources Foundation 1993). Impact from Hurricane Hugo in 1989, greatly damaged the mangroves, especially in Sugar Bay (Island Resources Foundation 1993). However, limited recovery of the forest was observable by 1992 and red mangrove propagules planted as part of a restoration project that began in 1999 saw an estimated 80% survival rate (Kendall et al. 2005). Changes in the extent of mangrove forest in SARI are discussed in Section 4.4.1. Mangrove ecosystems in SARI provide nesting habitat for up to 26 of the 44 bird species known to breed on the island (Sladen 1988) and mangrove forest of Sugar Bay provides critical habitat for Nearctic-Neotropical migrant parulids, both during migration and over-wintering periods (Wauer and Sladen 1992; Yntema et al. 2017).

Semi-deciduous Dry Forest

Semi-deciduous dry forests cover 75 ha of the park, accounting for just under half of terrestrial area of SARI and occur primarily at interior, landward locations (Figure 2.2.2.19) (Moser et al. 2011). The three Virgin Islands sub-formations of this forest type mapped on SARI by Moser et al. (2011) include: 1) gallery semi-deciduous forest, 2) semi-deciduous forest, and 3) semi-evergreen forest. As they fall hierarchically under the Lowland tropical/subtropical semi-deciduous forest of the U.S. Virgin Islands community classification (Gibney et al. 2000), we consider these three communities as constituting the semi-deciduous dry forest habitat type here. The semi-deciduous forest sub-formation (2) accounts for more than 80% of that described habitat. Pigeon berry, *Bourreria succulent,* and white stopper, *Eugenia monticola* are the dominant tree species found within semi-deciduous forest in SARI (NPS 2017a). Gallery semi-deciduous forest is restricted to riparian corridors, including guts and intermittent streams. In SARI, this sub-formation is dominated by the non-native genip tree, *Melicoccus bijugatus* (NPS 2017a). The semi-evergreen forest sub-formation in SARI is found in one location on a northwest-facing slope, and is community is typified by comparatively greater number of evergreen species. See Section 4.4.2 for a discussion of the condition of SARI’s dry forest habitat.

Coastal Grasslands

Coastal grasslands are found in the northeast and northwestern coastal regions of the park and cover 16.7 ha (Moser et al. 2011). We included two Virgin Islands sub-formations as constituting this community type: coastal grassland and the mixed dry grassland. The coastal grassland sub-formation includes grasses adapted to conditions of high wind, salt spray, and low moisture (Gibney et al. 2000). In SARI, this habitat type consists entirely the *Urochloa-maxima* coastal grassland alliance and is found both in the northeastern and northwestern portions of the park. Mixed dry grasslands are dominated by grass species, but have greater than 25% shrub, tree and/or herbaceous species associated with selective grazing (Gibney et al. 2000). Within this sub-formation, *Urochloa maxima-Cryptostegia madagascariensis* association is most prevalent within SARI, and it is dominated by the aforementioned non-native invasive grass and shrub, respectively. As coastal grasslands as a general category in the park are overwhelmingly dominated by non-native invasive species, management action to eradicate *U. maxima* and *C. madagascariensis* and restore native woody species to the mixed grassland was initiated in 2009 (Moser et al. 2011). See Section 4.4.3 for a discussion of condition of the coastal grasslands in SARI.

Terrestrial Vertebrates and Invertebrates

Terrestrial vertebrate species include ~80 birds, four bats, eight non-native mammals, six amphibians, and 12 reptiles. Species lists of terrestrial vertebrates are included as tables within the text or appendices. Invertebrates are numerous, but a species list has not been created.

Birds

While a comprehensive species list of avian fauna is not yet available for SARI, we compiled data from online sources and published literature to arrive at a tentative total of 82 bird species documented as occurring within the park (Wauer and Sladen 1992; McNair et al. 2005; McNair 2008; McNair et al. 2008; ebird 2017; Yntema et al. 2017) (Appendix B). This constitutes approximately 40% of the total number of species recorded for the entire island of St. Croix. Included in this list are seabirds, shorebirds, marshbirds, waterfowl, landbirds and near-arctic migrants. Meanwhile, 135 species have been recorded during the 35 years (1972–2016) of the annual Audubon Christmas Bird Count (CBC) on western side of the island of St. Croix (circle VISC; National Audubon Society 2010). As SARI is included within the VISC CBC 7.5 mile radius circle, the above total of 82 is likely an under-estimate.

Over a dozen of the species that spend some part of the year within SARI are considered USVI territorially endangered, threatened, or of special concern (Watson 2003; Platenberg et al. 2005). The Caribbean brown pelican, *Pelecanus occidntalis occidentalis*, federally de-listed in 2009, does not nest, but actively forages within the park (Watson 2003). Important bird habitats within SARI include a heron rookery in the red mangroves adjacent to the marina, a tern nesting beach on the northeast side of the park, two freshwater ponds, beaches, mud flats, and intertidal foraging habitats (Kendall et al. 2005). Least terns nest on SARI Spit, an area on the northeast side of the park composed of rock, coral rubble and dredge spoils (Kendall et al. 2005). Twenty-five nests with 25 brooding pairs were observed in June 2013 (data provided by Zandy Hillis-Starr). Avian inventories, while not comprehensive, have been conducted over the past several decades throughout the park. Of particular note are the inventories of Wauer and Sladen (1992), who conducted 12 surveys from 1986–1987 within the Sugar Bay mangrove forest and documented 35 species. A re-survey of the mangrove forest area in the early 2000s to assess the impact and recovery from Hurricane Hugo (1989) on bird populations found declines in the number of individuals and species of Nearctic-Neotropical migrants and in increase in number and species of waterbirds (McNair 2008). The results of this work highlighted the importance of changing habitat structure, with Neotropical-Nearctic migrants requiring mature forest compared to open mudflat habitats preferred by many waterbirds. Results of analysis of three decades of avian survey data from the entire island of St. Croix suggest an increase in confirmed nest and occurrence records for some species of birds, which the authors related to greater amounts of precipitation received during the 2000s (Yntema et al. 2017).

Herpetofauna

Twelve reptiles and six amphibians have been documented on St. Croix (Table 2.2.2.4). To our knowledge, herpetofaunal inventories have not been conducted specifically within SARI. However, it is likely that many of the species present in St. Croix occur within the park. Endemic species to St. Croix include the St. Croix racer, *Alsophis sanctaecrucis*, presumed extinct (Philobosian and Yntema 1977), the St. Croix anole, *Anolis acutus*, the St. Croix dwarf gecko, *Sphaerodactylus beattyi*, and the

St. Croix ground lizard, *Pholidoscelis polops*, now restricted to 4 cays surrounding the island of St. Croix (Platenberg and Boulon 2006).

Table 2.2.2.4. Amphibians and reptiles occurring on the island of St. Croix (Platenberg and Boulon 2006).

Category	Scientific Name	Common Name	Status
AMPHIBIANS	*Eleutherodactylus antillensis*	Antillean coqui	native
	Eleutherodactylus coqui	common coqui	introduced
	Eleutherodactylus lentus	mute coqui	native
	Leptodactylus albilabris	Caribbean white-lipped frog	native
	Osteopilus septentrionalis	Cuban treefrog	invasive
	Rhinella marina	cane toad	invasive
REPTILES	*Amphisbaena fenestrata*	Virgin Islands worm lizard	native
	Anolis acutus	St. Croix anole	endemic
	Antillotyphlops richardi	Richard's blind snake	native
	Borikenophis sanctaecrucis	St. Croix racer	presumed extinct
	Chelonoidis carbonarius	red-footed tortoise	introduced
	Hemidactylus mabouia	Afro-American house gecko	introduced
	Iguana	common green iguana	introduced
	Pholidoscelis exsul	Puerto Rican ground lizard	introduced
	Pholidoscelis polops	St. Croix ground lizard	endangered, restricted to cays
	Sphaerodactylus beattyi	St. Croix dwarf gecko	endemic
	Sphaerodactylus macrolepis	common dwarf gecko	native
	Thecadactylus rapicauda	fat-tailed gecko	likely introduced

<u>*Terrestrial Invertebrates*</u>

A complete inventory of invertebrate fauna has not been conducted to date within the park and a species list is not available. In general, the invertebrate fauna of the entirety of the USVI remain poorly inventoried (Platenberg et al. 2005). Two endemic butterfly species occur within the SARI: the Cassius Blue, *Leptotes cassius catalina*, and the Polydamas Swallowtail, *Battus polydamas thyamus* (NPS 1990). Important invertebrates include the land crab, *Cardisoma guanhumi*,which are a locally harvested species, ghost crab, *Ocypode* spp., fiddler crab, *Uca pugnax rapax*, and rock crab, *Graspus* sp. and soldier crab, *Coenobita clypeatus* (NPS 1990).

<u>*Mammals*</u>

Non-native mammals are numerous on the island (Table 2.2.2.5), with several of these species having negative impacts on the natural resources (Patterson et al. 2008). Mongoose and tree rats have negative impacts on native flora and fauna throughout the Caribbean. Grazing animals, including goats, sheep, and horses, have direct impacts on flora which can result in soil erosion and ultimately sedimentation into the bay. Native mammals include three species of frugivorous bats and the fishing bat, *Noctilo leporinus*, all documented during Anabat surveys in July 2007 along the east side of Salt River Bay (Fly By Night. Inc. 2017) (Figure 2.2.2.19*)*.

Table 2.2.2.5. Mammal species occurring inside or adjacent to SARI (Patterson et al. 2008; Fly By Night, Inc. 2017).

Scientific Name	Common Name(s)	Status
Brachyphylla cavernarum	Antillean fruit-eating bat	Native
Molossus	Pallas' free-tailed bat, Pallas's mastiff bat	Native
Noctilio leporinus	Greater bulldog bat	Native
Tadarida brasiliensis	LeConte's free-tailed bat	Native
Canis familiaris	feral dog	Non-native
Felis catus	feral cat	Non-native
Herpestes javanicus	Indian mongoose, Javan mongoose, small Asian mongoose	Non-native
Mus musculus	house mouse	Non-native
Odocoileus virginianus	white-tailed deer	Non-native
Rattus norvegicus	Norway rat	Non-native
Rattus	black rat	Non-native
Sus scrofa	feral hog	Non-native

Other Resources

<u>*Sound scape*</u>

Noise levels in the park have increased as a result of development both within the park boundary, nearby commercial activity, boats, and generators (NPS 2015). Acoustical monitoring is needed to quantify the impact both underwater and on land.

<u>*View scape*</u>

Scenic views extend across the bay out to the Caribbean Sea and can be observed from either side of the entrance to the Salt River Bay, as well as from the visitor station on the western side of the watershed. Both historical and ecological aspects are conveyed within the maritime viewshed, which fortunately, remains largely intact and unobstructed (NPS 2015). On a clear day, Puerto Rico is visible, 90 miles away. In contrast, the viewshed from the bay landward has been negatively impacted by private development in inholdings and outside of the park boundary. Night skies are an important resource within the park, providing a place to experience nighttime scenery and starry skies away from light pollution (NPS 2015). Ongoing and future development, which includes the building of private homes, impacts the viewshed and creates light pollution impinging on night skies and the Biobay experience. Airborne pollution from regional and global sources decrease visibility. Strengthening and enforcing existing regulations and zoning within the legislative boundary is needed. Data needs include a visual resource inventory, viewshed analysis, and night sky inventory (NPS 2015).

2.2.3. Resource Issues Overview

Resource condition threats or stressors identified as being "of concern" in terms of potential risk or harm to important park resources are explored in more detail in Chapter 4. Some have already been mentioned in Section 2.2 of this chapter. This section provides a brief introduction to other threats and stressors that are impacting or could potentially compromise the adequate condition of SARI's resources.

<u>Human Interactions</u>

The Foundation Document (NPS 2015) of the Salt River Bay National Historical Park and Ecological Preserve (SARI) in St. Croix, U.S. Virgin Islands states that SARI "preserves, protects, studies, and interprets internationally significant historical and cultural sites that encompass more than 2,000 years and human use of the diverse tropical, marine, and terrestrial ecosystems that comprise the Salt River watershed". Consequently, it is indisputable that human interactions occur and will continue to occur in the premises and vicinity of the park. SARI provides a number of valued resources and services to visitors. As per the SARI Conceptual Model (Patterson et al. 2008; NPS 2018b), coral reefs are a resources of particular aesthetic value which in turn provide a highly productive habitat for fish and invertebrates. Equally productive are mangroves and seagrass beds, which in turn provide shoreline protection. Existing wildlife, in particular unique and rare marine and terrestrial species provide both recreational and educational opportunities for visitors, a services that are fundamental for the wellbeing of people and intellectual advancement of society. In addition, SARI provides a habitat for "nesting colonies of least terns and little blue herons, and also functions as an important migratory bird stopover" (NPS 2018b). Finally, the park provides a service as an area where numerous recreational and education activities take place, both inland and in the water, namely fishing, diving (Figure 2.2.3.1), snorkeling along reefs, boating, swimming, archeological site seeing, hiking, and scenic sites viewing (NPS 2018b).

In addition, over the years, Salt River Bay provides safe harbor/anchorage for vessels during tropical storm and hurricane events. In this context, SARI plays an important role in local disaster preparedness and management. As the frequency and strength of tropical storms are likely to increase in the future (USVI HRRT 2018), the use of Salt River Bay as a hurricane hole remains an important and direct link between the boating community and the park (NPS 2018b).

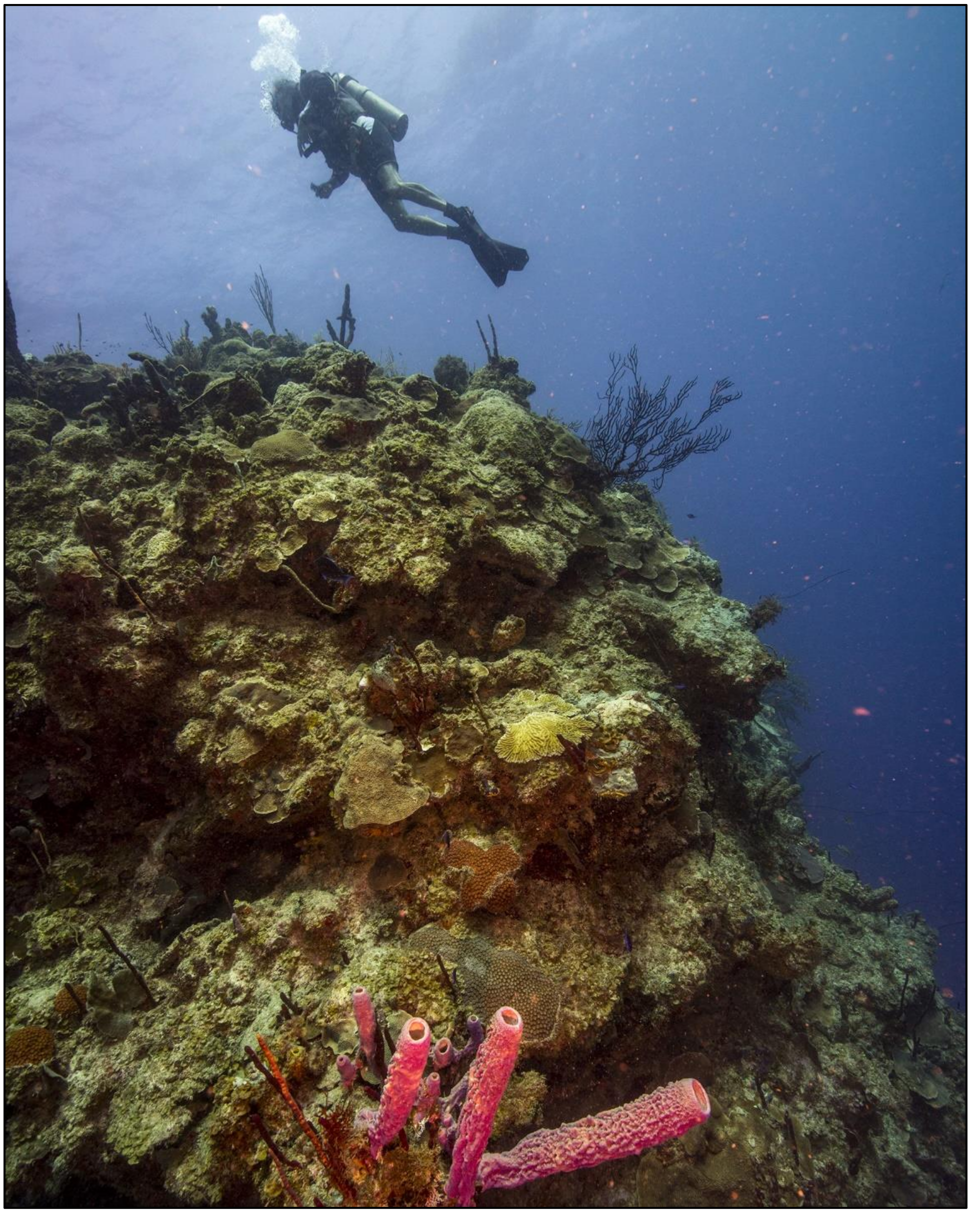

Figure 2.2.3.1. Diving in the Salt River Bay National Historical Park and Ecological Preserve (SARI) in St. Croix, U.S. Virgin Islands. (Photo property of NPS, https://www.nps.gov/articles/images/sari-reef-nocover.jpg)

Boat traffic and grounding

There are two ways to get to the park, either by vessel or by land. Boats visiting the park or passing near its boundaries can negatively impact natural habitats in many ways, such as oil or other discharges, spills, pumping of bilge water, release or sloughing of toxic material contained in bottom paint. (NPS 2018b). Another way of potentially harming coral reefs and seagrass beds are by groundings, anchoring, inappropriate use of anchors, or by propeller or hull damage. During the past three decades, there have been vessel groundings around St. Croix due to poor navigation and loss of engine power, but also related to illegal smuggling. In 2015, there were more than 30 abandoned and derelict boats in the bay left from hurricanes, some of these leaking fuels/hydrocarbons and solid waste trash throughout the bay (NPS 2018b). Following Hurricane Maria (2017), the number of derelict vessels in Salt River Bay rose by at least 30, many of which were targeted for removal (National Parks Traveler 2017). There are additional safety concerns related to anchoring at offshore moorings. An anchoring permit provided by the St. Croix Diving Association is required for all vessels, and anchoring is only allowed in a designated area at East and West Wall (NPS 2019a). Diver-down flag must be displayed while divers are in the water. No anchoring on coral areas is permitted. While boating in the park's waters, boaters must observe the Virgin Islands territorial rules and regulations regarding the taking of game and fish. No collecting of natural or cultural resources including coral or artifacts (NPS 2019a).

Debris, plastics, and microplastics

Debris resulting from human use of the park may stress some park resources, in particular in the marine environment. Marine debris consists mostly of floating manmade debris, remnants of fishing nets, abandoned or lost fishing buoys, and abandoned fish traps. Fishing lines, nets, rope, and other type trash can wrap around animals and cause drowning, infection, or amputation. Ingestion of marine debris by aquatic fauna can cause deformities, serious sicknesses, and even death. In addition, debris flows into Salt River Bay as a result of stormwater runoff from roads and driveways. In-land and marine debris can settle on hard bottom areas and kill coral colonies (Waddell 2005).

One kind of debris that is rapidly increasing in tonnage in the ocean is plastics of all kinds. The total global production of plastics grew nearly 200 times in the last half century, from about 1.5 million tons in 1950 to 280 million tons in 2012 (Rochman et al. 2013). The degradation processes of plastic materials is very slow; therefore, plastics can become a major environmental hazard to the marine environment. Except for the tiny fraction that has been incinerated, all plastics ever manufactured are still on the planet (Jambeck et al. 2015). Plastic entanglement and ingestion by marine mammals, fish, birds, and reptiles that result in injury and even death are frequently reported (Derraik 2002; Lozano and Mouat 2009).

Small plastic pieces less than five millimeters long, known as microplastics, are a type of debris of most emerging concern in marine environments. These are small enough to be ingested by a vast group of marine organisms. Furthermore, microplastics can adsorb and transport a variety of toxins because they have relatively large surface areas which are hydrophobic. In a study done in 2013, Whitmire and his co-investigators studied the occurrence and distribution of microplastics in the southeastern coastal region of the United States. They analyzed sand samples collected from various coastal sites from eighteen units within NPS Southeastern Region. Microplastics were isolated using density separation and counts of microplastic particles were compared among sites. In addition, they developed a predictive model to understand the drift of plastics via ocean currents.

One of the sampling sites in this study was located at the Western shoreline of the Buck Island Reef National Monument (BUIS), located less than 10 miles (15 kilometers) east of SARI. A total of 10 sand samples were collected from the site between July and October 2013. The analysis of the samples yielded an average of 102 microplastic pieces in 1 kg (2.2 lbs.) of sand. The percentage of microplastic items as pieces was 39.2% and that as fibers was 60.8%. The average length of the microplastic fibers was 2.65 cm (1.04 inches). The yield of microplastics was relatively low on BUIS, and considering that there is very little development in the area immediately surrounding the site and no large river nearby to transport wastewater to it, the microplastics found must have been transported via ocean currents or come from plastic debris being disintegrated near the site (Whitmire et al. 2016). Given the proximity of SARI to BUIS, the findings of the study suggest that microplastics are likely also to be present at SARI. In fact, it is likely that the anticipated yield of microplastics is greater at SARI given the considerable development in the area adjacent to Christiansted Harbor combined with the westward flow of ocean currents.

Poaching and Looting

SARI's law enforcement duties include ensuring the protection of the park's resources, both natural and cultural, as well as providing for visitor safety. Park rangers are tasked with enforcement of all park rules and regulations, which includes the "no-take" policy, beach closings for sensitive species' nesting seasons, no wake zones, the "pack-it-in/pack-it-out" policy, anchoring and mooring area, among other. In addition, park rangers are to work to prevent poaching of natural resources or looting of historical sites, and address any such cases inland in in the sea. Due to staffing limitations and funding constraints, law enforcement presence is not provided on a full-time basis. Also, mixed jurisdiction and private in-holdings can hamper proper patrolling of natural and cultural resource sites within SARI, or obstruct the enforcement of existing regulations (C. Pollock 2021, personal communication). Consequently, poaching episodes occur within the various parks in the Virgin Islands. Conversations with park rangers during the scoping visits for the development of this report yielded information attesting that there have been looting and poaching incidents reported. Information on poaching or looting episodes, in particular prior to the passage of hurricanes Maria and Irma, is not available in written format. No statistics could be found on the extent of poaching or looting in the park. Data on enforcement are needed. Invertebrates, such as conch and lobster have suffered from poaching. It is common knowledge that poaching of turtles and of turtle bird eggs occurs at certain level. Corals are also attractive to poachers. Looting of archeological sites, which has been documented within the park boundary, is an important threat to the park's resources (NPS 2015).

Land Use Changes

There have been several studies on the land cover of the Salt River watershed and SARI (USVI 2001; Kendall et al. 2005; Moser et al. 2011). However, it is difficult to assess the changes in land use by comparing the various data sets used in these studies, since the classifications used vary in each. Section 4.3.1 of this report analyzes the change in land use/cover of the Salt River watershed, including SARI, over the period 2002–2012. Maps presented in Chapter 4 were developed through the automated classification of high resolution National Agriculture Imagery Program (NAIP) imagery, available Lidar digital elevation data, and assorted ancillary information (NOAA 2002, 2007, 2012). During the period 2002 to 2012, the analysis of the data used to produce these two maps show that estuarine forested wetlands experienced the highest increase in spatial coverage with an addition of 6.7 ha over this period, followed by deciduous forests that had a coverage increase of 2.1 ha (see Section 4.3.1). Overall, during the ten year period, wetlands and deciduous forest coverage increased by 9.3 ha. Using the Brown and Vivas methodology (2005), the value of the land development intensity (LDI) index was calculated to be 1.52 for the period 2002–2012 (Donoso 2020). See Section 4.3.1 for more details.

Hurricanes and Tropical Storms

Because of a warming global atmosphere, and increasingly prolonged warming phases of sea-surface waters, there is a possibility of higher frequency of strong tropical storm events in the western Atlantic and Caribbean basins (Bengtsson et al. 2007). A recent study indicates that while there is a trend of increasing frequency of tropical storm activity in the Atlantic basin since the 1980s, long-term projections are not possible, because of the Atlantic Multidecadal Variability or Oscillation (AMV or AMO) (Murakami et al. 2020). In fact, including track records since the early 1900s an increase in overall number of tropical storms is not supported, but rather fewer tropical storms were registered for the Atlantic Basin, with the number of category 4 and 5 storms slightly increasing or not significantly changing (Bengtsson et al. 2007; Yoshida et al. 2017). Reliable long-term projections of frequency and strength of hurricane trends is not possible at this point in time (Murakami et al. 2020).

The potential of fewer but stronger storms will increase the probability of destructive storm surges and wave activity, which in combination with heavy precipitation could further erode the beaches of Salt River. Hurricane frequency by category shows that between 1900 and 2018, 36 tropical storms came within 50 nmi (nautical miles) of SARI, 16 of these storms did not reach hurricane strength and 6, 7, 4, and 3, storms reached hurricane categories 1, 2, 4, and 5, respectively, while they were located within 50 nmi of Salt River (Landsea and Franklin 2013) (Table 2.2.3.1, Figure 2.2.3.2).

Table 2.2.3.1. Tropical storm and hurricane frequency by decade. Storm categories were are determined by maximum strength gained within 50 nmi of SARI. TS = Tropical Storm, H1 = Hurricane Category 1, H2 = Category 2, H3 = Category 3, H4 = Category 4, H5 = Category 5. Data source: Best Track Data (HURDAT2) provided by NOAA https://www.nhc.noaa.gov/data/ (Landsea and Franklin 2013).

	Storm Category						
Decade	**TS**	**H1**	**H2**	**H3**	**H4**	**H5**	**Total**
1900–1909	2	1	–	–	–	–	3
1910–1919	2	–	3	–	–	–	5
1920–1929	1	–	1	–	–	1	3
1930–1939	1	2	–	–	1	–	4
1940–1949	1	–	–	–	–	–	1
1950–1959	1	–	1	–	–	–	2
1960–1969	–	–	–	–	–	–	0
1970–1979	2	–	–	–	–	–	2
1980–1989	2	–	–	–	1	–	3
1990–1999	–	2	2	–	1	–	5
2000–2009	2	1	–	–	1	–	4
2010–2018	2	–	–	–	–	2	4
Total	16	6	7	0	4	3	36

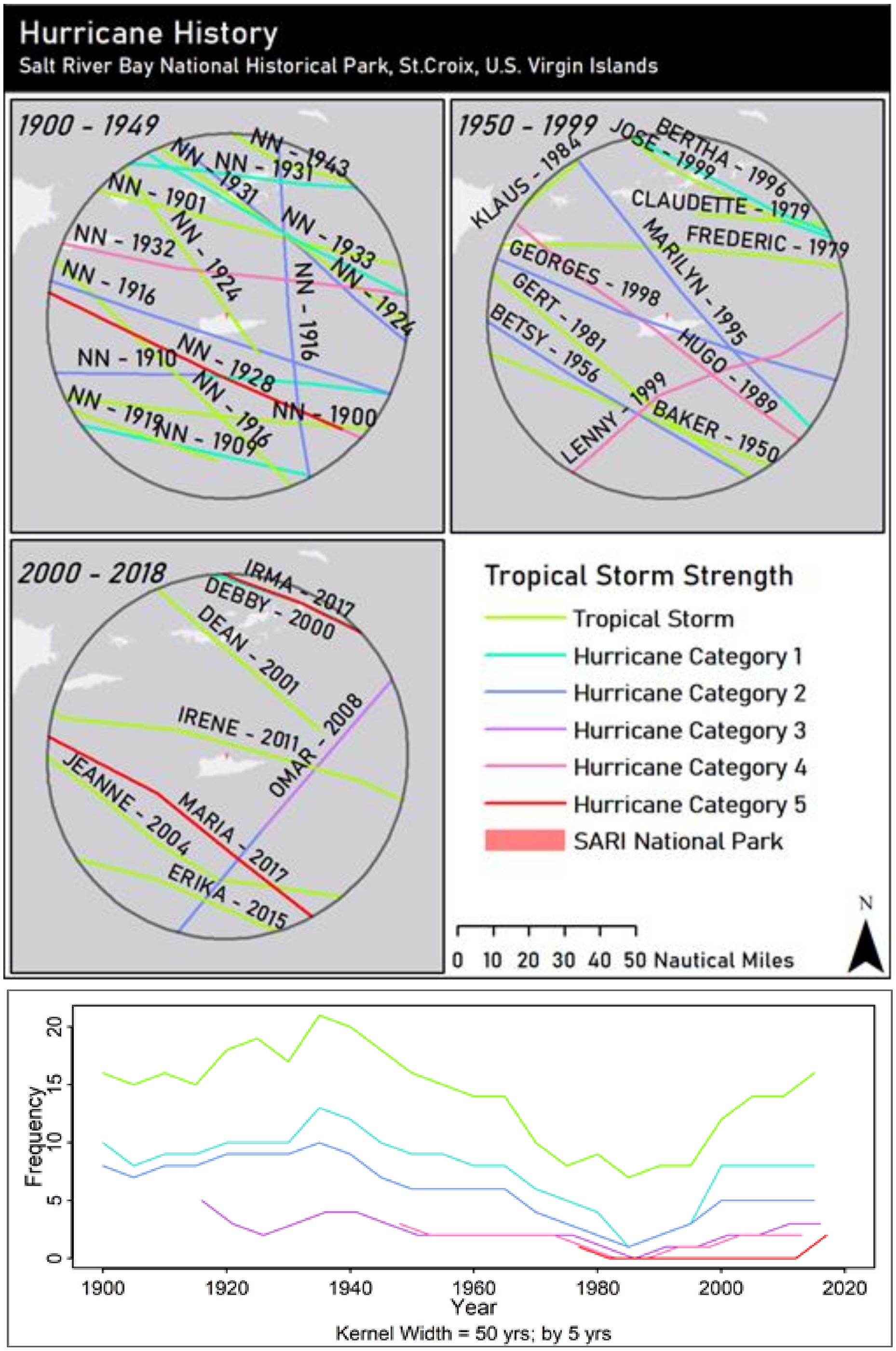

Figure 2.2.3.2. Top: Tropical storm and hurricane history within 50 nm of SARI. Tropical storm track labels indicate storm name and year. NN = No Name was given or is known for the storm. Bottom: Tropical storm frequency by category estimated for a 50-year moving window, predicted at 5-year intervals. Graphs generated with the Zoo package in R (Zeileis and Grothendieck 2005). Data source: Best Track Data (HURDAT2) provided by NOAA https://www.nhc.noaa.gov/data/ (Landsea and Franklin 2013).

2.3. Resource Stewardship

2.3.1. Management Directive and Planning Guidance

In 1994 a diverse group of SARI stakeholders developed the vision, purpose, significance, and management objectives for Salt River Bay. Input was received from Virgin Island government officials, Salt River Preserve Commission members, concerned citizens, environmental group representatives, and National Park Service officials. The result of these workshops was ultimately the creation of the SARI Foundation document, which provides guidance for both planning and management decisions.

In November 20, 2009, a Cooperative Management Agreement was signed between the Department of the Interior National Park Service and Government of the Virgin Islands for the management of the Salt River Bay National Park and Ecological Reserves (NPS 2015). The purpose of this Agreement was to set forth the roles and responsibilities of NPS and GVI in managing the park and to document the formation of a cooperative management partnership for the park. This Agreement also set forth provisions generally defining Management of the park and for the development of a planning process to implement the General Management Plan (GMP). Within the context of the Omnibus Insular Areas Act of 1992, 102 Public Law 247, NPS was charged the task to develop the GMP to describe the appropriate protection, management, uses of the Park in way that achieves the purpose of the referred Act. The GMP was to be developed with the involvement of stakeholders.

In January 2015, a Foundation Document was prepared as a collaborative effort between park and regional staff which was approved by the Southeast Regional Director on January 30th, 2015. The development of the Foundation Document permitted park managers, staff, and the public to identify and clearly provide the essential information that is necessary for the park management to consider when determining future planning efforts, outlining key planning issues, and protecting resources and values that are integral to park purpose and identity (NPS 2015).

The purpose of SARI is stated within the Foundation document as preserving, protecting, studying, and interpreting significant historical and cultural resources sites ... comprising the Salt River watershed (NPS 2015). Fundamental resources and values identified for SARI include archeological and historic resources, Amerindian village site and ballcourt, Columbus Landing, Fort Salé, Salt River Bay watershed complex, scenic views and vistas, and many recreational opportunities (NPS 2015).

2.3.2. Status of Supporting Science

To adequately manage the national parks, the National Park Service must have adequate knowledge of the condition of natural resources. Therefore, park managers require scientifically sound information that will allow them to acquire a broad-based understanding of the status and trends of park resources as a basis for making decisions and working with other agencies and the public for the long-term protection of park ecosystems. To acquire the needed information, the South Florida and Caribbean Inventory and Monitoring Network (SFCN) worked in putting together a long term monitoring program. At the individual park level, the program aims to monitor a set of key resources defined as the park’s vital signs. “Vital signs,” as defined by the NPS, are a subset of physical, chemical, and biological elements and processes of park ecosystems that are selected to represent the overall health or condition of park resources or elements that have important human values (Patterson et al. 2008). Table 2.3.2.1 shows the SFCN Vital Signs selected for monitoring SARI.

To facilitate the identification and prioritization of vital signs, SFCN divided the ecosystems in the South Florida and Caribbean parks into seven ecological zones and developed conceptual models for each as well as a region-wide overview and a marine benthic communities sub-model. The biological communities in these ecological zones are assumed to be affected by similar physical drivers and the same general set of stressors. The conceptual model for the Salt River Bay National Historical Park and Ecological Preserve can be found at https://irma.nps.gov/DataStore/DownloadFile/469987

For the present assessment, available data and reports varied significantly by focal resource. Datasets available from monitoring and inventory efforts used to assess condition and to develop reference conditions are described within each indicator summary in Chapter 4. Data and documents were obtained from numerous sources, including SFCN personnel, SARI staff, academic researchers with prior or ongoing research programs within the Monument, and publicly available datasets.

Table 2.3.2.1. SFCN Vital signs selected for monitoring in SARI (Patterson et al. 2008).[1]

Category	Vital Sign	Type 1	Type 2	Type 3	No Monitoring Planned
Air Quality	Air Quality-Deposition	–	–	x	–
	Air Quality-Mercury	–	–	x	–
Geology and Soils	Coastal Geomorphology	x	–	–	–
Water	Surface Water Hydrology	x	–	–	–
	Estuarine salinity patterns	–	–	x	–
	Water Chemistry	–	x	–	–
	Nutrient Dynamics	–	–	x	–
	Periphyton (Freshwater)	–	–	–	x
	Phytoplankton (Marine)	–	–	x	–
Biological Integrity	Invasive/Exotic Animals	–	x	–	–
	Invasive/Exotic Plants	–	x	–	–
	Marine Benthic Communities	x	–	–	–
	Mangrove-Marsh Ecotone	x	–	–	–
	Wetland Ecotones and Community Structure	–	–	–	x
	Forest Ecotones and Community Structure	x	–	–	–
	Marine Exploited Invertebrates	x	–	–	–
	Aquatic invertebrates in wet prairies & marshes	–	–	–	x

[1] Type 1 represents Vital Signs for which the network will develop protocols and implement monitoring; Type 2 represents Vital Signs that are monitored by PAIS, another NPS program, or by another federal or state agency using other funding; Type 3 represents Vital Signs for which monitoring cannot be currently implemented because of limited staff and funding but will likely be done in the future.

Table 2.3.2.1 (continued). SFCN Vital signs selected for monitoring in SARI (Patterson et al. 2008).[1]

Category	Vital Sign	Type 1	Type 2	Type 3	No Monitoring Planned
Biological Integrity (continued)	Marine Fish Communities	x	–	–	–
	Focal Fish Species	–	x	–	–
	Freshwater Fish and large macro-invertebrates	–	–	–	x
	Amphibians	–	–	x	–
	Colonial Nesting Birds	–	x	–	–
	Marine Invertebrates-Rare, Threatened, and Endangered	x	–	–	–
	Sea Turtles	–	–	x	–
	Protected Marine Mammals	–	–	x	–
Human Use	Visitor Use	–	x	–	–
Landscapes (Ecosystems Pattern and Processes)	Fire Return Interval	–	–	–	x
	Vegetation Communities Extent & Distribution	x	–	–	–
	Benthic Communities Extent & Distribution	x	–	–	–
	Land Use Change	x	–	–	–

[1] Type 1 represents Vital Signs for which the network will develop protocols and implement monitoring; Type 2 represents Vital Signs that are monitored by PAIS, another NPS program, or by another federal or state agency using other funding; Type 3 represents Vital Signs for which monitoring cannot be currently implemented because of limited staff and funding but will likely be done in the future.

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Chapter 3. Study Scoping and Design

The NRCA is a collaborative project between Florida International University, the University of the Virgin Islands (UVI), and the National Park Service (NPS). Stakeholders on this project include the Salt River Bay National Historical Park and Ecological Preserve (SARI) management and staff, as well as NPS Interior Region 2 – South Atlantic Gulf managers, the NPS South Florida/Caribbean Network (SFCN) scientists, and other NPS staff linked to the Virgin Islands sites.

This chapter describes the study scoping process, introduces the hierarchical indicator framework used in the assessment, and summarizes the general approach and types of methods used to evaluate and report condition findings reported in chapters 4 and 5.

3.1. Preliminary Scoping

3.1.1. Initial planning and scoping

During the initial stage of Phase I of the study, several in-person meetings and conference calls took place between the FIU Principal Investigator (Anna Wachnicka) and NPS staff. A preliminary scoping meeting took place on December 12, 2016, where the FIU project team met with staff from the NPS South Florida/Caribbean Network (SFCN) and the acting coordinator of the Regional NRCA and RSS Programs. The objective of the meeting was to identify (a) projects conducted by SFCN in the USVI parks; (b) reports, papers and data available at the SFCN office that could be used for the present project; (c) potential data gaps; and (d) important drivers of ecological change in the selected sites based on the research done in the parks.

The meeting started with a discussion of the vital signs being monitored by SFCN and partners within the NPS units located in the U.S. Virgin Islands. A preliminary subset of physical, chemical, and biological elements and processes of the park ecosystems were identified as important for the present NRCA, but it was agreed that the final list would be determined during the on-site scoping meetings planned for February, 2017. As a result of the discussion, a number of reports and papers were highlighted, as well as data sets available at the SFCN headquarters and in other NPS data centers. Information available from partner agencies and institutions was also identified. The names of potential contacts were provided to the FIU team. A preliminary list of identified documents and datasets and their online location was to be prepared by NPS.

Following the preliminary scoping meeting, the FIU project team met with the acting coordinator of the Regional NRCA and RSS Programs to plan future actions, in particular as it referred to the on-site park visits and scoping meetings. In the course of the meeting, it was reiterated that the purpose of the NRCA was to evaluate and report on current conditions for important park natural resources, and to identify critical data and knowledge gaps and potential factors that are influencing park resource conditions. As with other NRCAs, constraints were set on this assessment, namely: (a) the NRCA was to be performed utilizing available data sets and information; (b) the identification of data needs and gaps should be guided by the framework categories selected for the project; (c) as possible and appropriate, description and evaluation of conditions in each unit would be completed using GIS coverages and map products; and (d) study design and reporting products would follow national NRCA guidelines and standards (FIU 2017).

3.1.2. Onsite scoping and meetings with SARI NPS staff

The Salt River Bay National Historical Park and Ecological Preserve (SARI) was the second of the three NPS units visited (Appendix C). The FIU team traveled to St. Croix on February 5, 2017. On Tuesday, Feb. 7, a joint team of NPS staff and FIU staff carried out the Salt River Bay National Historical Park and Ecological Preserve (SARI) site visit. During the site visits, the team focused on identifying the major natural resources in the parks and the issues that were impacting these, both positively and negatively. Several points that were highlighted in the discussion included: the health of the mangroves in SARI, the importance of SARI as a historic location within the Caribbean, and concerns about the coral reefs and the fish that populate the waters of SARI. Brief notes of the visit to the SARI site are provided in the Phase I Project Report (FIU 2017).

During these meetings, the participants accomplished a series of tasks, namely:

- Revisit the most important issues examined during the site visits. Follow-up and/or clarify matters that required further discussion
- Discuss the methodology to be used in the assessment and revise dates set for the implementation of the phases of the project
- Confer with a preliminary scope of the content of the individual NRCAs for the units
- Jointly concur to a preliminary list of focal resources to be assessed in full or in a limited manner, based on the available information and data sets for each park, as per the knowledge of the meeting participants
- Agree on the responsibility of the different actors, in addition to the FIU team (NPS on-site staff, NPS in mainland staff, South Florida/Caribbean Network – SFCN, others) and their expected information and data input and datelines
- Complete the draft scoping tables reflecting the results of the deliberations of the participants
- Identify existing information and data sets in-situ that would be provided to the FIU team before the conclusion of their visit or sent to them on a later time.

3.2. Study Design

3.2.1. Indicator Framework, Focal Study Resources and Indicators

The framework used in the study of SARI is adapted from that presented in the H. John Heinz III Center for Science's "State of Our Nation's Ecosystems 2008" (Heinz Center 2008). The framework defines a way to organize the various resources that are considered important to the park in a hierarchal manner. The framework considers regional and landscape context, as well as historic condition influences, and constitutes a mechanism to summarize current natural resources conditions, risk factors, and critical data gaps.

The proposed framework encompasses two major categories, namely the Supporting Environment and Biological Integrity. In turn, Supporting Environment is subdivided into Coastal Dynamics and

Chemical/Physical, whereas Biological Integrity is subdivided into Terrestrial Plants, Marine Plants, Terrestrial Vertebrates/Invertebrates, Marine Vertebrates, and Marine Invertebrates.

The primary features in the selected framework are focal resource components, indicators, measures, stressors, and reference conditions. Resource "Components" in this process are defined as natural resources (e.g., lizards), natural processes or patterns (e.g., shoreline dynamics), or specific features or values (e.g., water quality) that are considered important to current managers. Each focal resource or component can be characterized by one or more "indicators". The term "indicator" is used in our assessment to refer to "a specific, well-defined, and measurable variable that reflects some key characteristic of a component that can be tracked through time" (Heinz Center 2008) to signal what is happening to the specific resource. Each indicator has one or more "measures" that best define the current condition of a resource being assessed in the NRCA. "Measures" are defined as those values or characterizations that evaluate and quantify the state of ecological health or integrity of a resource. In addition to measures, current condition of resources may be influenced by certain "stressors," which are also considered during assessment. A "stressor" is defined as any agent that imposes adverse changes upon a component. These typically refer to anthropogenic factors that adversely affect natural ecosystems, but may also include natural processes or disturbances such as hurricanes, floods, or predation (adapted from Amberg et al. 2014).

A "reference condition" is a benchmark to which current values of a given measure can be compared to determine the condition of that resource component. A reference condition may be a historical condition (e.g., species composition of seagrass in the 1980s), an established ecological threshold (e.g., predefined standards for water quality), or a targeted management goal/objective (e.g., abundance of reptiles) (adapted from Amberg et al. 2014 and Stoddard et al. 2006).

During the scoping process in SARI, key resources were identified by NPS staff. These are represented as "components" in the NRCA framework. The list of components was not a comprehensive list of all the resources in the SARI. Rather, a selection of components was made which included resources and processes that were of greatest concern or highest management priority. One or more indicators and respective measures for each, as well as known or potential stressors, were identified in collaboration with NPS staff.

Table 3.2.1.1 provides the framework for the SARI NRCA, including the list of focal resources considered, along with the associated condition indicators used to assess each focal resource. Full assessments were conducted for all focal resources except for queen conch. Authors responsible for each section are listed next to their respective focal resource.

Table 3.2.1.1. SARI NRCA framework table.

Framework Category	Focal Resource	Assessment Level	Section Author	Indicators and Measures
Supporting environment	Shoreline dynamics	Full assessment	P. Olivas	• Shoreline change (3 measures)
	Water quality (inside and outside Salt River Bay)	Full assessment	T. Smith	• Fecal indicator bacteria (1 measure) • Dissolved oxygen (1 measure) • Total suspended solids – TSS (1 measure) • Turbidity (1 measure) • Dissolved Nutrients (3 measures) • Chlorophyll (1 measure) • Terrestrial Sediments (1 measure)
	Watershed condition	Full assessment	M. Donoso	• Landover / Land use Change (1 measure)
Biological integrity – terrestrial plants	Mangrove	Full assessment	D. Ogurcak	• Vegetation community extent (2 measures)
	Semi-deciduous dry forest	Full assessment	D. Ogurcak	• Vegetation community extent (2 measures)
	Coastal grassland	Full assessment	D. Ogurcak	• Vegetation community extent (2 measures)
Biological integrity – Marine Plants	Macroalgae	Full assessment	T. Frankovich	• Macroalgae community extent (1 measure)
	Seagrass	Full assessment	E. Whitman T. Frankovich	• Seagrass community extent (1 measure)
Biological integrity – marine vertebrates and invertebrates	Corals	Full assessment	T. Smith	• Stony coral cover (1 measure) • Stony coral health (1 measure) • Seawater temperature (1 measure)
	Queen conch	Limited assessment	R. Ennis	• Community extent (1 measure)
	Reef fish	Full assessment	A. Duran	• Community and population status (3 measures)

3.2.2. Reporting Areas

SARI includes areas of both submerged and dry lands. The reporting area was treated as one unit and, depending of the resource being analyzed, encompassed the entire acreage within SARI's maritime or terrestrial boundaries unless otherwise noted in a specific focal resource section.

3.2.3. General Approach and Methods

This assessment includes the collection and review of available literature, datasets, as well as other types of existing information (maps, photographs, etc.) for each of the relevant resource identified in the framework. New data was not collected for this study. Existing data was analyzed to present summaries of the resource condition(s) and to compare with the reference condition(s). New spatial representations and maps were created as needed. Once all relevant information for each component was considered, a qualitative statement of the overall current condition was provided and compared to the reference condition wherever possible.

Data Gathering

Data, literature and overall information mining began with the collection of information during the scoping process. Information gathered includes NPS reports and monitoring plans, reports from various state and federal agencies, published and unpublished research documents, databases, tabular data and charts, GIS data, photographs, maps, which were either provided by NPS staff or obtained through personal communication with researchers and online bibliographic literature searches and inquiries.

Data analysis and assessment

Data analysis and development of the assessment was particular to each focal component identified in the framework and was based on the amount of existing information and recommendations provided by NPS staff and other experts. The methodology applied for each resource is defined in the corresponding section within Chapter 4 of this report.

Researchers and experts

Researchers and subject matter experts from FIU, NPS, and partner entities of these two organizations were consulted while developing the NRCA for SARI. Consultations were in the form of individual and group visits, correspondence via email or phone, virtual meetings, and reviews of resource sections. A list of the team of researchers and experts contributing to the assessment of each focal resource can be found in the respective chapter 4.

Summary Indicator Symbols

The “Indicator” and “Measurement” assessments for each component will be presented in a standard format throughout the document. This standard format is consistent with State of the Park reporting (NPS 2012). Condition/trend/level of confidence tables will be used for each resource to provide a representation of the condition assessment in a concise visual manner. The level of confidence will be depicted as high, medium or low, and will infer how confident the assessment is based on the information used to evaluate the condition. A detailed account will be provided in the various sections of chapter 4 of this report under the heading “Condition and Trend” for each resource.

Table 3.2.3.1 shows the “Condition/trend/level of confidence” scorecard to be used to describe the overall condition, trend, and level of confidence of the analysis assigned to each indicator for a focal resource. The color of the circles indicates the condition based upon the chosen indicators/measures and the reference conditions. Red circles imply that a resource is of significant concern; yellow circles denote that a resource is of moderate concern; and green circles signify that an indicator and/or measure are/is currently in good condition. A circle without any color, (which is almost always associated with the low confidence symbol-dashed line), signifies that there is insufficient information to make a statement about condition of the indicator, consequently, condition is unknown. The arrows within the circles represent the trend of the indicator/measure condition. Arrows pointing upward refer to an indicator which is improving; horizontal left-right pointing arrows express that the indicator’s condition is currently unchanging; and arrows pointing downward indicate that the indicator's condition is deteriorating. Circles with no arrows denote that the trend of the indicator's condition is currently unknown. Table 3.2.3.2 provides example indicator symbols and descriptions of how to interpret them in the assessment summary tables.

Table 3.2.3.1. Indicator symbols used to indicate condition, trend, and confidence in the assessment.

Condition Status		Trend in Condition		Confidence in Assessment	
Condition Icon	**Condition Icon Definition**	**Trend Icon**	**Trend Icon Definition**	**Confidence Icon**	**Confidence Icon Definition**
	Resource is in Good Condition		Condition is Improving		High
	Resource warrants Moderate Concern		Condition is Unchanging		Medium
	Resource warrants Significant Concern		Condition is Deteriorating		Low

Table 3.2.3.2. Example indicator symbols and descriptions of how to interpret them in the assessment summary tables.

Symbol Example	Verbal Description
	Resource is in good condition; its condition is improving; high confidence in the assessment.
	Condition of resource warrants moderate concern; condition is unchanging; medium confidence in the assessment.
	Condition of resource warrants significant concern; trend in condition is unknown or not applicable; low confidence in the assessment.
	Current condition is unknown or indeterminate due to inadequate data, lack of reference value(s) for comparative purposes, and/or insufficient expert knowledge to reach a more specific condition determination; trend in condition is unknown or not applicable; low confidence in the assessment.

Overall condition tables are presented for each focal resource in Chapter 5. To arrive at an overall status and trend for each focal resource, we followed the rules for combining multiple status and trends as outlined in the NPS-NRCA Guidance Update date January 20, 2014. Specifically, a combined condition score for a focal resource was determined by assigning any red symbol a value of 0, any yellow symbol a value of 50, and any green symbol a value of 100, summing the values of all indicators for each focal resource and dividing by the number of indicators/measures. Deviation from this method to arrive at the overall status was done on a case-by-case basis at the discretion of the resource assessment author and is noted in chapter 5 when applicable.

The overall trend for a focal resource was determined by adding the number of up arrows and subtracting the total number of down arrows. Calculated trend values greater than 2 were considered an increasing trend while values less than −2 were considered a negative trend. All values in between were considered no trend. In the case when there was less than three indicators for a particular focal resource and both trends for indicators/measures were the same, the overall trend took on the same value.

However, when only two indicators/measures were present for a focal resource and the status or trend was not in agreement between the two, the author of each focal resource assessment made a judgement as to whether one indicator should be more highly weighted. The condition and trend of the more highly weighted measure was used to represent the overall status of a focal resource. The rationale for this is described on a case by case basis when applicable in chapter 5.

Overall confidence level corresponded to the level most often indicated for a resource if indicators were equally weighted. In the case when indicators were not equally weighted, the confidence level of the higher weighted indicator was used for the overall indicator. The focal resource assessment author has noted which indicator was weighted more highly and has provided their reasoning in the text of chapter 5.

<u>Preparation and Review of Component Draft Assessments</u>
The preparation of draft assessments for each component was carried out by FIU and UVI analysts and researchers. Though the project team, analysts and researchers, rely heavily on peer-reviewed literature and existing data in conducting the assessment, the expertise of NPS resource staff also played a role in providing insights into the direction for analysis and assessment of each component.

Subsequent to the initial scoping engagements and general undertakings described above, the process of developing draft documents for each component began with a project team brainstorming session, followed by knowledge-sharing and planning meeting. In addition, personal and e-mail conversation among the members of the project team and with an individual or multiple individuals considered local experts on the resource components under examination took place throughout the draft assessment development process. These conversations were a way for the project team members to verify the most relevant data and literature sources that should be used and also to formulate ideas about current condition with respect to the NPS staff opinions. Throughout the draft assessment development process, the project team maintained communication, to the extent possible, with NPS staff, in particular with the acting coordinator of Regional NRCA and RSS Programs. Upon completion, draft assessments were forwarded to NPS component experts for initial review and comments.

Final Component Assessments

Final resource component assessments were made by incorporating comments provided by NPS staff, resource experts, and reviewers during the review of draft chapters. As a result of this process, and based on the recommendations and insights provided to the authors, the final component assessments were written. These final resource component assessments represent the most relevant and timely information and data available for each component and the insight and knowledge of park resource staff, researchers, external resources experts, and assessment writers.

Format of the focal resource assessment sections presented in chapter 4

All focal resource component assessments are presented in a standard format. The format and structure of these assessments is described below.

Description

This section describes the relevance of the resource component to the individual park and explains its characteristics. This section also refers to any existing interrelations that exist between the featured component and other resources components referenced in the assessment. Emphasis is to be given to issues that make the component a unique feature of the park, a key process or resource in the park ecology, or a resource that is of high management priority in the park.

Data and Methods

This section refers to the datasets used in the analysis as well as any type of information utilized in the assessment. The methods used for processing or evaluating the data are also discussed herein where applicable. The indicators and corresponding measures are presented in this section as well, describing to the best of our knowledge how each indicator was measured or qualitatively assessed the natural resource topic.

Reference Conditions/Values

This section describes the reference condition that were used to evaluate each resource component as it is delineated in the framework. Also, discussions of available data and documents that describe the reference conditions are located in this section. This section provides an explanation as to why specific reference conditions are appropriate or logical to use in this assessment.

Condition and Trend

This section provides and discusses key findings regarding the existing condition of the resource component and trends (when available). The information is presented primarily with text but is often accompanied by detailed maps or plates that display different analyses, as well as graphs, charts, and/or tables that summarize relevant data or show interesting relationships. All relevant data and information for a component is presented and interpreted in this section.

Threats and Stressors

This section presents the major threats and stressors that may affect the resource and influence on the current condition of a resource component based on a combination of available data and literature, and discussions with experts and NPS staff.

Data Needs/Gaps

In this section, critical data needs or gaps for the resource component are reported. It also refers to how these data needs/gaps, if addressed, would provide further insight in determining the current condition or trend of a given component in future assessments. The section is expected to help NPS staff seeking to prioritize monitoring or data gathering efforts.

Overall Condition

This section renders a qualitative summary statement of the current condition that was determined for the resource component. This determination is established based on the analysis and review of available literature, data, and any insights from NPS staff and experts, or other subject matter experts. The Overall Condition section summarizes the key findings and highlights the key elements used in determining and justifying the level of concern, if any, that authors attribute to the condition of the resource component. In addition, this section includes the condition assessment table.

Sources of Expertise

Individuals who provided data or references, or were consulted for the focal study resources will be listed in this section. A short paragraph presenting their title and affiliation with offices or programs is also included.

Literature Cited

This is a list of formal citations for literature or datasets used in the analysis and assessment of condition for the resource component. When possible, links to websites are also included. Citations used in appendices and plates referenced in each section (component) of Chapter 4 are listed in that section's "Literature Cited" section.

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Chapter 4. Natural Resource Conditions

4.1. Coastal Dynamics

4.1.1. Shoreline Dynamics

This section reviews the condition of the shoreline at Salt River Bay National Historical Park and Ecological Reserve (SARI). The condition assessment considers a 65-year period of aerial images provided by the National Park Service and U.S. Geological Survey for the following image dates: 1954 (USGS 1954), 1971–77 (NCCOS 2022a), 1992 (NCCOS 2022b), 2000 (NCCOS 2022c), 2007 (NPS 2007) and from ESRI basemap (2019) to generate data to assess the status of the shoreline. The shoreline is typically evaluated using metrics that detect changes in area, length, elevation, and type (e.g., sand/gravel, rocks, or vegetation). The condition metrics selected for this resource include area, length, and type (e.g., sand/gravel, rocks, or vegetation). Please note that some condition metrics could not be evaluated due to lack of detailed data related to elevation.

Description

Based on ocean current influence, the shoreline of SARI can be separated into three sections: the northern, central, and southern areas of the park (Figure 4.1.1.1). The northern shoreline is mostly influenced by longshore currents with rocky shores and some beaches of small sediments (sand, coral cobbles or gravel beaches) and coastal vegetation (Kendall et al. 2005). The central area flanks Salt River Bay, and it is mostly influenced by wind-driven currents (Kendall et al. 2005). The central west shorelines are mostly vegetated with some sections of sandy/gravel shores, while the central east shoreline presents a higher extent of sandy/gravel shores. Lastly, the southern shores of the park are within Sugar and Triton Bays and are mostly influenced by flooding and tidal dynamics and shorelines are colonized by mangrove species (Figure 4.1.1.1, see Section 4.4.1, Kendall et al. 2005).

Two main geological formations, Miocene Kingshill and Cretaceous Judith's Fancy formations underlie most of Salt River watershed. The southern area of SARI including the mouth of the river is underline by the Miocene Kingshill formation, while the northern area of the park consists of Cretaceous Judith's Fancy formation (Kendall et al. 2005). These formations play an important role in shoreline and sediment dynamics and the ecology of the park. Sediments within the park consist of mostly carbonate sediments along the sides of Sugar Bay and main body of the Salt River Bay, and fine terrigenous sediment, such as silt and clay found in near terrestrial sources in the southernmost areas of the bays (Kendall et al. 2005). Carbonate sediments consists primarily of calcareous algae and other benthic organisms, while the terrigenous sediment are primarily a result of upland erosion and river transport (Gerhard and Petta 1974, Kendall et al. 2005). Hubbard (1989) found that the reef is an effective barrier that separates sedimentation between bay, canyon and shelf. However, terrigenous sediment has been observed offshore after extreme events such as storms and hurricanes (Williams 1988), suggesting that a rise in storm activity and intensity could increase upland erosion and terrigenous sediment deposition into offshore environments. Although the role of wave action in coastal erosion might be low because of the dampening effect of the coral reef barrier, the presence of mangroves is likely to play an important role in the reduction of upland sediment deposition especially during extreme precipitation events. In the shelf and canyon, the main sediment comes from coral bioerosion transported as a longshore drift east-west by the trade winds (Hubbard 1989, Kendall et al. 2005).

Given that some areas within SARI have been heavily altered and are exposed to different environmental and anthropogenic pressures, it is important to qualify and quantify how the shoreline has changed over the past 65 years.

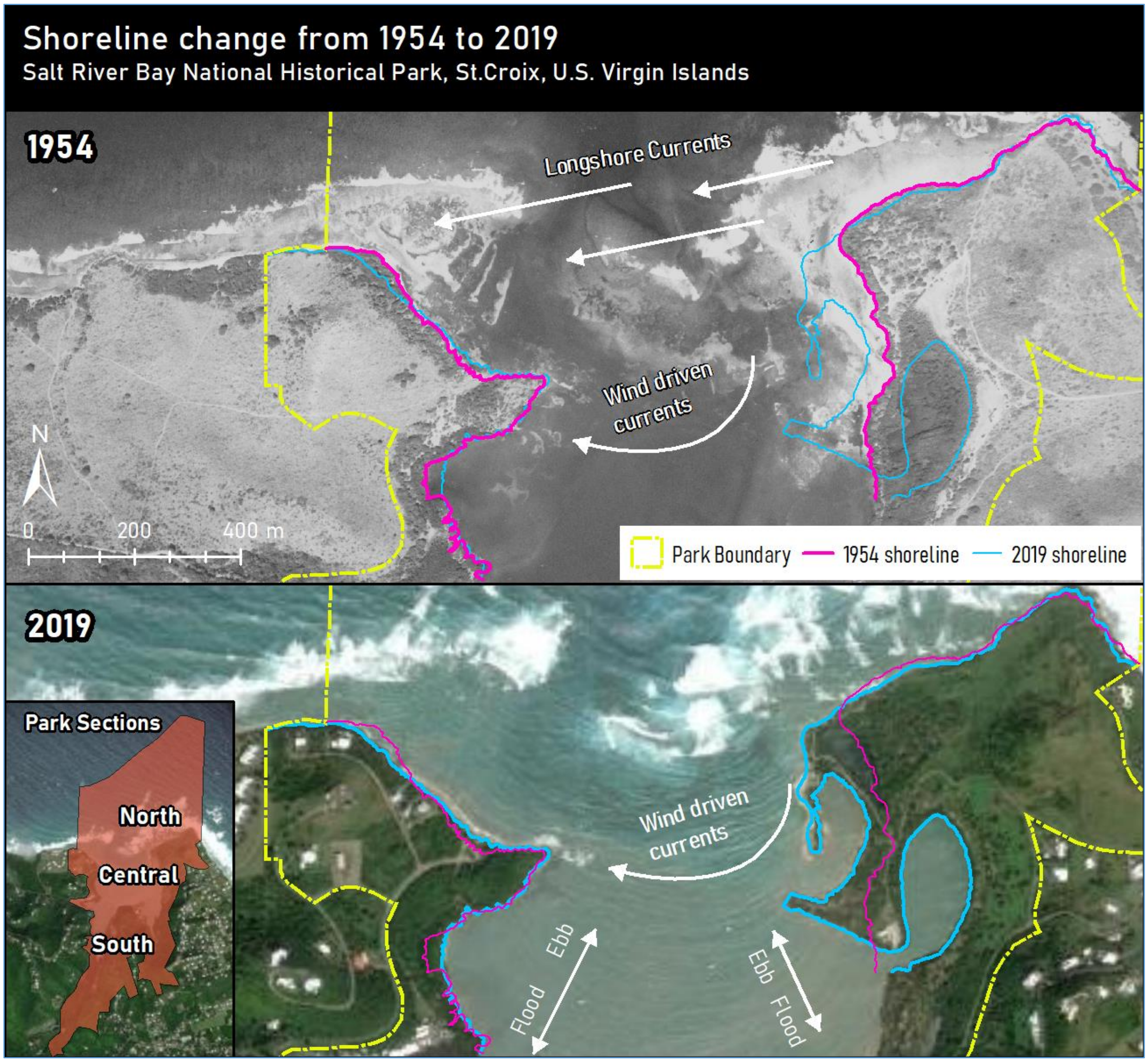


Figure 4.1.1.1. Shoreline change and ocean currents. Park boundary is indicated by hatched yellow line, and digitized and smoothed shorelines (inter border) for years 1954 and 2019 are in pink and blue respectively. Currents after Kendall et al. (2005). Shorelines for 1954 and 2019 were added to both images for comparison. Lower left insert depicts the geographical sections of SARI referenced in the text.

Data and Methods

To evaluate the long-term trends of the shoreline dynamics the following metrics were used: 1) difference in length of the shoreline (vegetation boundary) between 1954 and 2019, 2) shore area change between 1954 and 2019, and 3) shore (sand/rock/gravel) habitat change for multiple years between 1954 and 2019. The analysis focused on the shorelines in the northern section of the park excluding Sugar and Triton Bays. The vegetation boundary was used to calculate the shoreline length given that it is more clearly defined and less likely to present digitizing errors. The shoreline dynamic was defined as a function of the detectable high water mark (shoreline variability) and the migration of the vegetation boundary landward or seaward. The timeframe for the trend evaluation encompassed 1954 to 2019, with assessments done in 1954, 1971–77, 1992, 2000, 2007, and 2019.

The reference conditions for the park's shoreline dynamics were the shoreline and vegetation boundaries in 1954. The image from 1954 was black and white. The lack of color affected the ability to distinguish the high water mark in some areas during digitizing. The image was not georeferenced and presented considerable warping which together with the lack of landmark features was a challenge to correct. For the georeferencing of the 1954 image, the 2019 image (ESRI World Imagery) was used with a 3rd order polynomial transformation. The root mean square (RMS) error between 1954 and 2019 images was about 0.82 for the forward-inverse. The other images were already georeferenced, but presented some registration errors when compared to the 2019 image: 1971–77: 1.7 m; 1992 and 2000: 4.8 m, and 2007 3 m. The image from 1971–1977 is a mosaic with one section of the image from 1971 and the other from 1977. Although the park is only within one of the mosaic sections, it is not known whether it is on the 1971 or 1977 section. For the analysis this mosaic was referred to as year 1971. Image resolution was 1 m for 1954, 50 cm for 1971–1977, 1992, 2000, and 2019, and 30 cm for 2007.

Current condition as of 2019 was established from satellite data from September 18, 2019. Shoreline and vegetation boundaries were visually interpreted and digitized from the aerial photographs. Digitization of the shorelines was done at a scale of 1:1000 (Holdaway and Ford 2019), and vertexes were created every 5–10 m to capture the coastal variability. To reduce sharp vertices, a Polynomial Approximation with Exponential Kernel (PAEK) method was used to smooth the shorelines. This method is based on a smoothing tolerance parameter that controls the length of a "moving" path that is used for calculating the new vertices. For this study, the smoothing tolerance parameter was set to 5, allowing for smoothing of sharp vertices but preserving the detail of the shoreline. The shorter the length the more detail that will be preserved. All GIS data were processed in ArcMap 10.8 (ESRI Inc.).

Three classes were used to characterize the shoreline: 1. Sand/Gravel, 2. Rock, and 3. Vegetated. Sandy shores are defined as distinguishable fine or small particle sediment areas. Sand and gravel were kept together because they were indistinguishable across the aerial photography and satellite data, especially for the aerial from 1954. Rocky shores are areas where large boulders and rocks were clearly distinguishable from vegetated and sandy shores. Lastly, the vegetated shores are areas where vegetation, predominantly mangroves, covered the shore and no sand, gravel or rock were visible from the aerial image. For further information about the mangrove cover and species, see section

4.4.1. For digitizing the outer shoreline boundary high water mark, a conservative approach was used where the most visible high tide line was set as the lower boundary for the estimation of the shore area.

Data limitations and conditions that can influence the process of shoreline digitization and affect the detection probability of the high-water mark include data quality of the older aerial photography, the subjective process of digitization (line tracing), and time of the day and acquisition date of imagery. For example, the tide fluctuation in the Virgin Islands is about 30 cm (Kendall et al. 2005), therefore the difference between the high water mark at low and high tide can range from a few centimeters to a couple of meter depending on the slope at any particular location along the shore.

<u>Reference Conditions/Values</u>

Reference conditions for the shoreline of SARI were determined using digitized shoreline and vegetation boundary from aerial photographs from 1954 (1954 aerial image). The length of the shoreline in 1954 for the east and west were 1,342 m and 1,431 m, respectively (Table 4.1.1.1). Although the image presented some limitations (see Data and Methods section), it is an important record of the baseline conditions of the park before manmade modifications in the 1960's where an embayment was created by a hotel/marina development that dredged and connected an existing enclosed salt point to the Salt River Bay (Figures 4.1.1.1 and 4.1.1.2) (Kendall et al. 2005, Pinckney et al. 2014). These modifications resulted in a significant change of the shoreline, especially in the northeast and central areas of the park. Since the dredging ceased, the northeast section has been changing and sediment deposition from the northeast has been reshaping the contour of the east shoreline.

Table 4.1.1.1. Coastal shoreline length for 1954 and 2019. Length of the shoreline was determined by digitization of the landward vegetated boundary using aerial photography (see Data and Methods section).

Section	1954	2019	Change (m)
East	1342	2652	1310
West	1431	1302	129

<u>Condition and Trend</u>

The comparison of the baseline length of the shorelines (landward vegetated boundary or inter border) in the north and central zones of SARI (areas most affected by ocean currents) showed that the change in the length of the shoreline between 1954 and 2019 has been relatively small for the west section of the park. However, the east section of the park has experienced much larger changes (Table 4.1.1.1 and Figure 4.1.1.1). The west side experienced decrease in shoreline of just over 100 m, while the eastern side experienced a gain of about ~1300 m. The shoreline in the northwest section of the park has receded in some areas up to 25–30 m, while in the central west the shoreline has experienced gains of about ~30 m in a small area mostly surrounded by mangroves (Figures 4.1.1.1, 4.1.1.2, and 4.1.1.3). In the northeast corner of the park, the geomorphology of the shoreline is predominantly rocks and cliffs (Hubbard 1989), and as a result more resilient and resistant to weathering and erosion. Thus, the shoreline change in this area has been small, but still some shoreline recession has been observed (Figures 4.1.1.1 & 4.1.1.2).

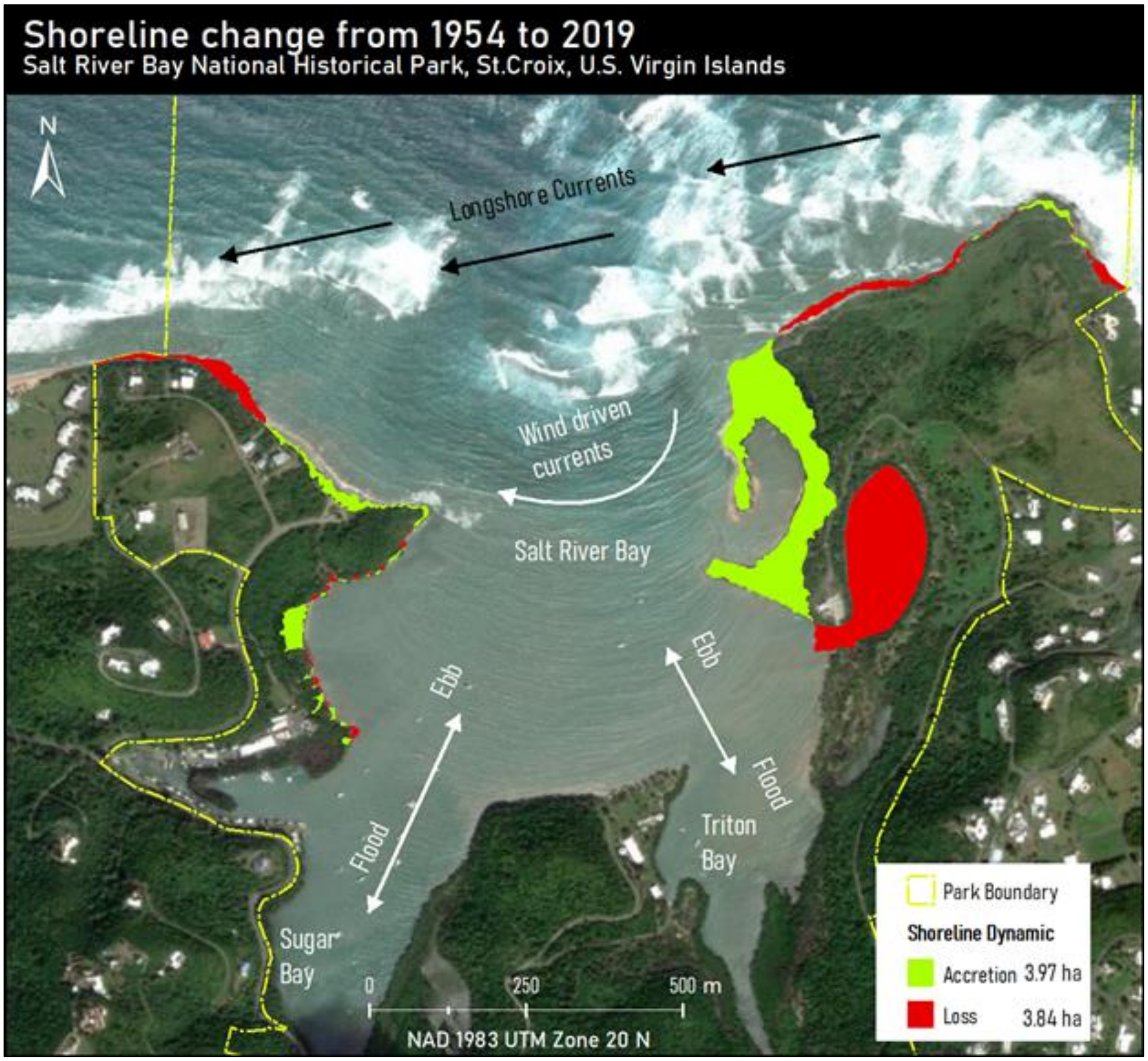


Figure 4.1.1.2. Shoreline accretion (green polygons) and loss (red polygons) for the northern and central shorelines of SARI for the period between 1954 and 2019. Park boundary in hatched yellow and currents modeled after Kendall et al. (2005).

During the 1960s the east shoreline experienced a significant change because of a subdivision development, which consisted in a marina and dredging of a saltwater pond (Kendall et al. 2005, Pinckney et al. 2014). The dredged material was later used to create a peninsula, a beach, and other areas near the marina. This manmade embayment resulted in a considerable change of the shoreline including the increase in length. However, longshore sediment movement and deposition in the northern area of Crescent Beach has resulted in a "sandbar peninsula" that has been slowly growing southward (Figure 4.1.1.4). With time, it is likely that the "sandbar peninsula" will enclose Crescent Beach resulting in a new salt pond and restoring the shoreline to a contour that resembles the one before the development (Figures 4.1.1.1 & 4.1.1.2).

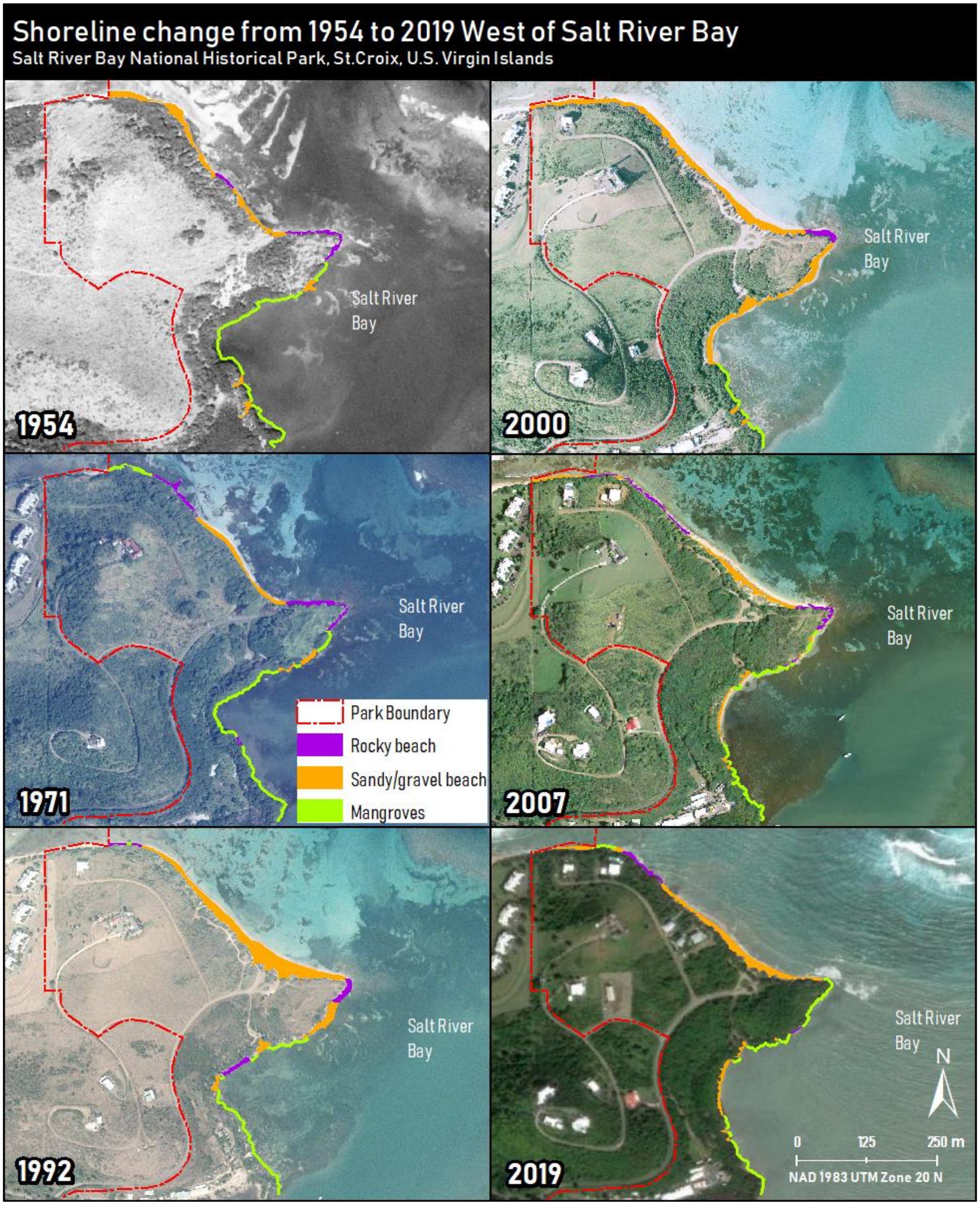


Figure 4.1.1.3. Morphology and extent of rocky (purple), vegetated (green) and sandy/gravel (orange) shores in the western section of SARI. Red hatched line shows the park boundary. Imagery available from NCCOS (for 1971, 1992, and 2000), NPS (for 2007), and ESRI (for 2019).

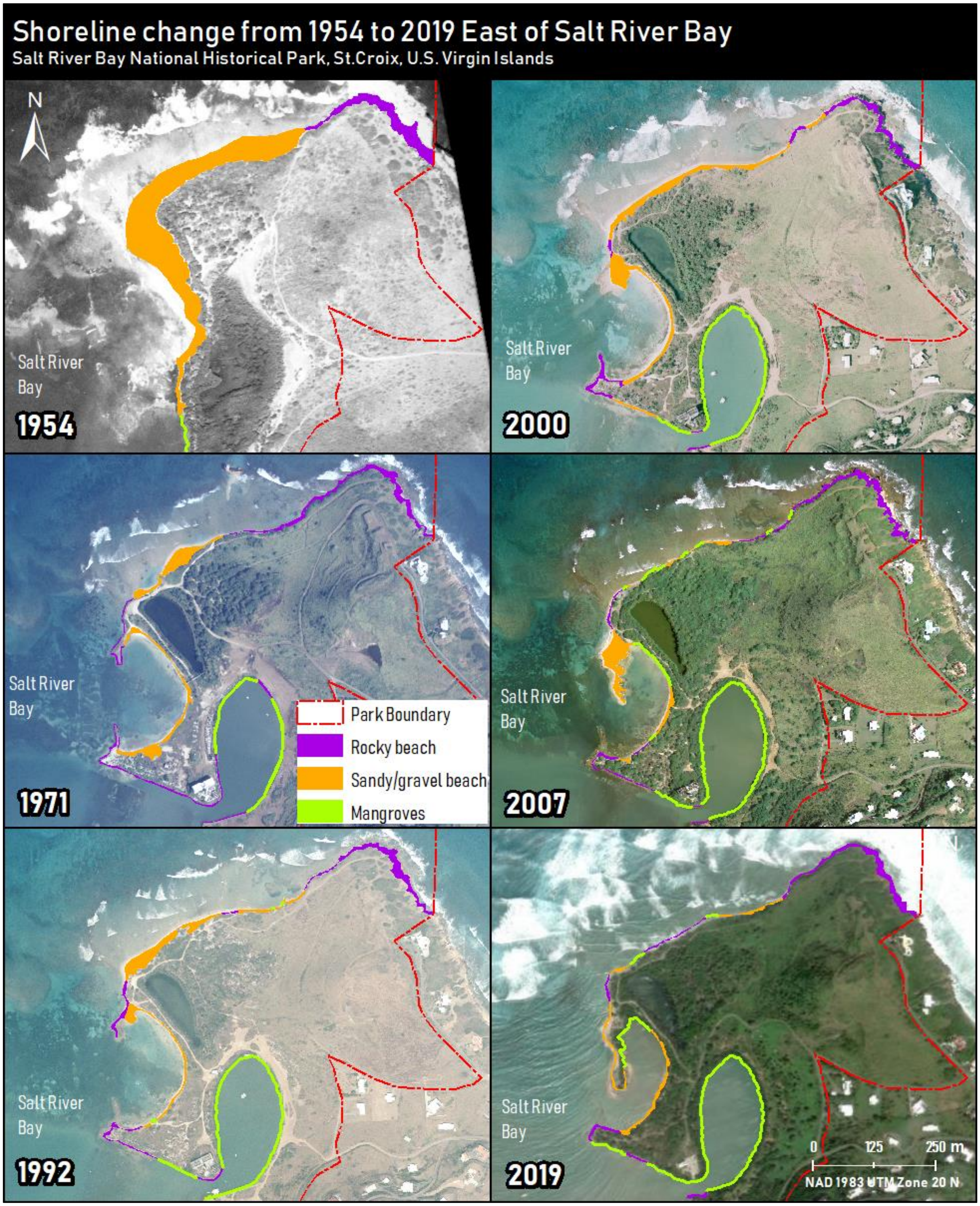


Figure 4.1.1.4. Morphology and extent of rocky (purple), vegetated (green) and sandy/gravel (orange) shores in the eastern section of SARI. Red hatched line shows the park boundary. Imagery available from NCCOS (for 1971, 1992, and 2000), NPS (for 2007), and ESRI (for 2019).

The long-term trends of shoreline accretion and erosion in SARI show that the shorelines along the northern areas of the park are receding the most with the shorelines in the northwest receding faster (Figure 4.1.1.2). However, overall, these changes are small for the 65-year period. Conversely, the east shore, and in particular the central east shore, has shown an important advancement between 1954 and 2019. This significant change has been a result of southward growth of the "sandbar peninsula" from longshore sediment deposition. For the period of assessment (65 years), the balance for the coast zone has been a net loss for the area measured. However, this net loss is mostly related the development from the 1960's. Thus, if the dredged area were to be excluded, accretion is happening at faster pace than erosion (Figure 4.1.1.2).

Additionally, although SARI shoreline has experienced some changes over the past 65 years, archeological records show that prehistoric materials from village settlements from over 2000 years ago in both east and west shores remain intact suggesting that these shorelines have been relatively stable (Z. Hillis-Starr 2018, personal communication).

On the west shores of SARI, there has been some change in the components of the shoreline. In terms of the rocky shore, the area has experienced a small decline, but the fluctuation in area since 1954 has been small (Figures 4.1.1.3 & 4.1.1.5 and Table 4.1.1.1). In the case of the sandy/gravel shore the west side of the park has seen almost no change between 1954 and 2019; however, the area of these shores has experience fluctuation within the years. For instance, there was an increase from 0.18 ha in 1971 to 0.81 ha in 2000 and then a reduction to 0.25 ha in 2007 (Figure 4.1.1.5 and Table 4.1.1.2). These fluctuations in the sandy/gravel area are most likely because of the impact of Hurricane Hugo on the vegetated coastline, which killed a large percentage of mangroves and significantly reduced their canopy density (Kendall et al. 2005). It is important to note that the change in shoreline area classification was not associated with the gain or loss of shoreline.

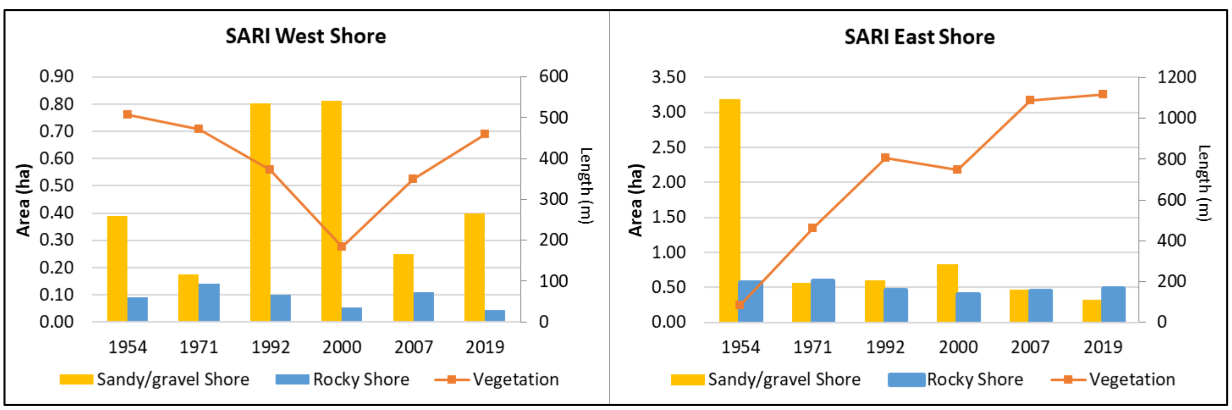


Figure 4.1.1.5. Shoreline dynamics for sandy/gravel (orange column), rocky (blue column), and vegetated (red line) shores for the west and east shores of SARI.

Table 4.1.1.2. Changes in rocky shore and sandy/gravel shore surface area and length of vegetated shore between 1954 and 2019 for the central and northern areas of SARI. Image type AP = aerial photography, SPOT = SPOT satellite Imagery.

				Rocky Shore (ha)			Sandy/gravel Shore (ha)			Vegetation Shore (m)		
Hurricane	**Image Type**	**Acquisition date**	**Year**	**Total area**	**Change[1]**	**Rate of change[2] (ha/yr)**	**Total area**	**Change[1]**	**Rate of change[2] (ha/yr)**	**Total length**	**Change[1]**	**Rate of change[2] (m/yr)**
Pre-Betsy	AP	1954	1954	0.66	–	–	3.57	–	–	590.3	–	–
Post-Betsy / Pre-Hugo	AP	1971	1971	0.74	0.08	0.005	0.74	−2.83	−0.167	935.0	344.7	20.278
Post-Hugo (1989)	AP	1992	1992	0.56	−0.18	−0.008	1.41	0.67	0.032	1175.5	240.5	11.452
	AP	2000	2000	0.46	−0.11	−0.013	1.65	0.24	0.030	933.0	−242.5	−30.314
	AP	2007	2007	0.56	0.11	0.015	0.72	−0.93	−0.133	1434.9	501.9	71.694
Post-Irma / Post-Maria	SPOT	09/18/19	2019	0.53	−0.03	−0.003	0.72	0.00	0.000	1576.3	141.4	11.785

[1] Change is defined as the difference between two assessment periods. 2 The rate of change was calculated by dividing the absolute change (area or length) by the number of years for a time period. Negative values mean loss while positive values mean gain.

In 2000, the area for sandy/gravel shore remained almost the same; however, the spatial distribution of these shores classes changed within the landscape. Between 2000 and 2007 the area covered by sandy/gravel decreased, while the extent of the vegetated shore increased. This result suggests that the process of recovery from Hurricane Hugo took more than 10 years. Between 2007 and 2019, the sandy/gravel shore area and vegetated shore length have shown a steady increase. The length of shoreline covered with vegetation has been low but relatively consistent with 508 m and 459 m in 1954 and 2019, respectively. The shortest length of shoreline covered by vegetation was 185 m in 2000, which coincided with the period of recovery from Hurricane Hugo (Figure 4.1.1.3).

Similar to the western rocky shores of SARI, the rocky shores in the east have experienced little fluctuation, with a loss of about 0.08 ha between 1954 and 2019 (Figures 4.1.1.4 & 4.1.1.5). In the case of the sandy/gravel shores, the fluctuation has been much larger than on the western side. As stated in previous sections, during the 1960s the east shoreline experienced a significant change caused by a subdivision development. This development dredge a large area to create a saltwater pond, a small peninsula, a beach, and a marina. This change of the morphology of the northeastern shore is most likely the cause for the significant reduction of sandy/gravel shore between 1954 and 1971 (Figure 4.1.1.5). Over time, longshore and wind-driven currents have transported this sediment southward to the northern area of Crescent Beach where it has settle forming a "sandbar peninsula". It is expected that with time the "sandbar peninsula" will restore the shoreline to a contour that resembles that of the 1954, before the development (Figures 4.1.1.1 and 4.1.1.4). Between 2007 and 2019, the "sandbar peninsula" has become larger, higher, and more stable and as a result, natural vegetation has colonized it. This has resulted in a steady reduction of sandy/gravel area and an increase of the vegetated shoreline. During this same time period the shoreline advanced noticeably (Figures 4.1.1.1 & 4.1.1.4). Between 2000 and 2019, the area of the sandy/gravel shore in the northeast has continued to decrease and has been replaced by rocky shores, suggesting the movement of sediment to the southwest towards the small peninsula.

In general, the eastern side of SARI has experienced a loss of about 2.87 ha of sandy/gravel shore area between 1954 and 2019, but has also seen a significant seaward advancement of the shoreline (Figures 4.1.1.1 & 4.1.1.4). In comparison with the western side of the park, the length of the east shoreline covered by vegetation has been increased significantly, from 82 m in 1954 to 1,118 m in 2019 (Figure 4.1.1.4). However, the large change in vegetated shoreline can be attributed to the increase in shoreline extent created by the dredging of the saltwater pond in the 1960s and subsequently the expansion of the peninsula, which created appropriate conditions for natural colonization. Additionally, there has been some native plant restoration and exotic removal, but not near the shoreline (Figure 4.1.1.4 and see 4.4.2. Coastal Grasslands).

Threats and Stressors

Since 1954, SARI has experienced important changes of the shoreline (Figure 4.1.1.1) especially after the dredging of the Crescent Bay. Some areas of the park have experienced noticeable losses, while other areas have gained surface area. In general, it seems that SARI is mostly losing shore area in the northwest and gaining it in the east. Because of the current dynamics and long-term trends, it is unlikely that this phenomenon is a result of seasonal sediment movement but more a result of the constant wave action generated by longshore and wind-driven currents (Figures 4.1.1.1 & 4.1.1.2). Therefore, changes in the intensity and frequency of these currents is likely to have a significant impact on the future shoreline dynamics of SARI. However, it is also important to consider the potential effect of seasonal fluctuations of wave action intensity, especially associated with extreme season events, such as storms and hurricanes.

Similar coastlines throughout the Virgin Islands and Caribbean, tropical storm strength and frequency are of major concern for the shoreline of SARI. Aerial photographs from 1992, show large areas of mangrove damage along the shoreline that are still recovering just a few years after Hurricane Hugo (1989) impacted St. Croix, taking over 12 hours to travel the length of the island from east to west; sustained winds were over 160 mph (Kendall et al. 2005). However, during the same period the sandy/gravel beach area increased. Thus, while the impact of storms on the shoreline in SARI can reduce mangrove cover, negatively affecting the stability of the shoreline, it can also increase beach surface area through sediment deposition, which could be of benefit for some marine species. Other large storms, such as Hurricanes Irma and Maria (2017) have also affected the park, but the effects on erosion of the shoreline appear to be small. This could be a result of the individual characteristics of the storm, in which path, proximity and in particular wind speed and direction can strongly affect the movement of water in the bays. For instance, Kendall (2005) documented a 1.0–1.5 m of storm surge associated with Hurricane Hugo in the bays.

Sea-level rise is another major concern for the area because it can increase coastal flooding, negatively affecting coastal ecosystems such as reefs and mangroves. Additionally, sea-level rise can also interact with or change the local currents and wave patterns, modifying the coastal sediment dynamics. At the moment of the preparation of this report no sea-level rise models were available for SARI. However, the current rate of sea level rise (SLR) as measured in nearby San Juan, PR (1963–2020) is 2.15 mm yr^{-1} (http://www.psmsl.org, Station ID 2118), which is slightly lower than the global rate (~3 mm yr^{-1} Nerem et al. 2018). Based on the long-term tends observed for the shoreline (Figure 4.1.1.1), it appears that some areas in the east part of SARI are keeping pace with the rise in sea level as a result of the mangrove recruitment (see Section 4.4.1). However, the shores in the northwest have experienced noticeable landward retraction and might experience accelerated erosion with increasing sea levels.

Data Needs and Gaps

To capture the inter-annual variability of shoreline dynamics in SARI, several methods could be implemented that would capture changes in accretion and erosion (loss of shoreline length and shore surface area) with higher certainty than what was available for the analyses in this section. These methods are listed below and can be implemented individually or in combination.

1. Automated classification of high-resolution satellite data in combination with digital image processing techniques to detect sand, water and vegetation. Automated image classification is inexpensive and would reduce the human introduced error in line digitization. However, tidal effect on the high water mark cannot be eliminated with this approach because image acquisition times for satellites are typically fixed.

2. *Shoreline assessment using differential GPS technology (high accuracy).* The use of differential GPS for assessment of coastal dynamics is an inexpensive tool that can be used as an effective method to quantify changes in the shoreline as a result of long-term trends, extreme events, or to identify seasonal dynamics. However, this method is limited by accessibility and the extent of the area of interest since it requires a person to physically walk the length of the shoreline with the GPS unit.

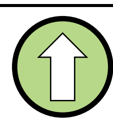
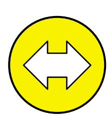
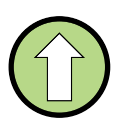
3. *Automated classification of airborne photography with the use of unmanned aerial systems (UAS technology)*. Unmanned aerial systems allow for better control of the image acquisition times and could eliminate the error associated with tidal dynamics and detection of the high water marks. Additionally, these images would provide much higher resolution data, which increases the precision of change estimates. However, unmanned aerial technology can be expensive as a result of maintenance of UAS, expensive photogrammetric software, data collection and data processing time. Furthermore, data management and compliance can also be impediments for the implementation of this method.

4. *Very high-resolution 3D shore profiles.* Using terrestrial LiDAR technology, 3D profiles of strategic locations could provide very high resolution and precision of surface extent change and volumetric estimates. Similar to UAS, terrestrial LiDAR can be expensive because it requires purchase and maintenance of equipment and proprietary software.

<u>Overall Condition</u>

The condition of shoreline dynamics in SARI was assessed using three indicators: shoreline length change, shoreline area change, and shore habitat change (Table 4.1.1.3). Shoreline length and habitat were found to be in good condition with a trend of improving. Shoreline area change was considered as warranting moderate concern with no trend in condition over the time period assessed.

Table 4.1.1.3. Graphical summary of status and trends for shoreline dynamics within the framework category coastal dynamics.

Component	Indicator	Condition Status /Trend	Rationale and Reference Conditions
Shoreline Dynamics	Shoreline length change		Since 1954 (reference condition) the shoreline extent has increased significantly as result of sediment deposition and dredging for the marina.
Shoreline Dynamics	Shoreline area change		Reduction of area since 1954 (reference condition) as a result of the dredging for the marina has been slightly outpaced by the increase in area accrued because of sediment deposition.
Shoreline Dynamics	Shore habitat change		Sandy/gravel shoreline area and extent of the vegetated shoreline have increased steadily since 1954 (reference condition), and rocky shorelines have experienced little change.

Source(s) of Expertise

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- Kristen Ewen, NPS, Technician BUIS
- Clayton G. Pollock, NPS, Biologist BUIS
- Zandy Hillis-Starr, NPS Resource Manager BUIS (retired)
- Ilsa B. Kuffner, PhD, USGS, St. Petersburg Coastal & Marine Science Center

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4.2. Chemical /Physical

4.2.1. Water Quality

This section reviews the condition of water quality in the Salt River Bay National Historical Park and Ecological Reserve (SARI). The condition assessment considers data provided by the USVI Department of Planning and Natural Resources Division of Environmental Protection (2000–2018), the USVI Territorial Coral Reef Monitoring Program (2004–2009), and individual research assessments between 2012 and 2018 (May and Woodley 2016, Bayless 2019, Pait et al. 2020). The condition of water quality for seawater is typically evaluated using metrics that detect changes away from conditions suitable for the maintenance and propagation of marine and aquatic life and for human contact recreation. The condition metrics selected for this resource assessment include fecal indicator bacteria, dissolved oxygen, total suspended solids, turbidity, dissolved and total nutrients, chlorophyll, sediments, contaminants, and pollution indicator assays. Temporal trends in condition metrics were evaluated for time-series measurements.

Description

Water quality in SARI is variable across space and time, reflecting seasonality and responses to episodic events, such as storms, as well as different physical, biological and anthropogenic processes. Water quality can have important impacts on organisms and human health. Conditions in SARI range from very clear oceanic waters offshore to highly turbid and occasionally contaminated inshore waters.

Water quality can be measured from numerous variables that are measurable on site, remotely, or from collected samples that are analyzed in a laboratory. These variables can indicate acceptable conditions for human health, such as fecal indicator bacteria that suggest the epidemiological risk for human contact-based development of gastrointestinal illness. These variables may also indicate suitability of water for maintenance and function of certain forms of marine life or deviation of conditions away from natural, unperturbed ecosystems. Of high relevance to SARI are water quality variables and associated values that support sensitive ecosystems, such as coral reefs, mangroves, and seagrass, and their associated flora and fauna. The USVI maintains standards of water quality and contaminants for territorial marine waters (USVI 2019).

Data and Methods

In the USVI, marine water bodies are classified into three categories of regulation based on their ability to affect wildlife and aquatic life and human health (USVI 2019). Classifications are the following: Class A. Waters are of exceptional recreational, environmental, or ecological significance; Class B. Designated for maintenance and propagation of desirable species of wildlife and aquatic life, contact recreation; Class C. waters are those waters which are located in industrial harbors and ports and have less stringent water quality standards for certain parameters than Class B waters (USVI 2019). All marine waters of SARI are in Class B and are subject to standards with the purpose of maintaining aquatic life and human health. Water quality included in this assessment were taken from publicly available databases and published and unpublished sources.

Common water quality metrics. Common water quality indicators included in this assessment with their standards for maintenance of aquatic life (where developed) are listed in Table 4.2.1.1. Temperature has high relevance to coral stress and is presented and discussed later in Section 4.6.1. Dissolved oxygen (DO) is important for maintenance of respiration in aquatic animals and can affect animal growth and movement (Prince and Goodyear 2006). Total suspended solids (TSS) can indicate both endogenous particles related to biological activities in the water column, such as plankton, and exogenous particles potentially related to pollution. There are no US Environmental Protection Agency (USEPA) nor local USI aquatic life standards for TSS (USEPA 2019; USVI 2019). Turbidity is a measure of water clarity, with values greater than 1 nephelometric turbidity unit (NTU) associated with waters of limited clarity that are less aesthetically pleasing and indicate impairment for coral reef environments of the USVI (USVI 2019). Less stringent standards of <3 NTU are listed for other Class B areas without coral reefs. This higher allowable standard may limit light transmission for some benthic photosynthetic organisms and constitute a source of stress (T.B. Smith unpublished observations; Smith et al. 2013).

Table 4.2.1.1. Common water quality indicators used in this assessment. When available, each unit is listed with its standard for the maintenance and activity of aquatic life, or deviation from natural conditions as determined by local regulations. For chlorophyll a, literature surveys served as a guideline for when values exceed oligotrophic conditions associated with coral reefs.

Variable	Unit	Standards or guidelines	Source
pH	None	<7, >8.3	USVI 2019
Temperature	°C	Dependent on taxa; <29°C for corals/<32°C elsewhere	see Section 4.4 for corals; USVI 2019
Dissolved Oxygen	mg L^{-1}	>4.8 mg L^{-1}; >5.5 mg L^{-1}	Prince and Goodyear (2006); USVI 2019
Total Suspended Solids	mg L^{-1}	None	–
Turbidity	Nephelometric turbidity units	<1 NTU reduction from oceanic clarity for coral reefs/<3 NTU maximum in general[1]	Smith et al. 2013 and USVI 2019
Ammonia	µg L^{-1}	None	–
Nitrate	µg L^{-1}	None	–
Phosphate	µg L^{-1}	None	–
Chlorophyll a	mg L^{-1}	<0.4 µg L^{-1}	Smith et al. 2013, Furnas et al. 2005
Fecal Indicators	Colony forming units per 100 mL seawater	<30 CFU (30-day geo. Mean), <110 CFP (<10% samples for 30 days)	USVI 2019

[1] NTU limits not applicable to Salt River Lagoon (Marina) and Salt River (Sugar Bay).

Nutrients and phototrophs. Dissolved inorganic nutrients are important and essential for aquatic life by supporting the growth of phytoplankton and benthic phototrophs, such as macroalgae. However, excessive nutrients can promote growth of unwanted types or abundance of phototrophs. For example, phytoplankton stimulated by nutrients can decrease light penetration to the benthos and some species are implicated in harmful algal blooms (Anderson et al. 2002). Or excessive nutrients can stimulate overabundance benthic plants at the expense of desired and natural foundational species, such as corals and seagrasses, particularly when herbivory is naturally or artificially low (McCook 1999). This includes competition with juvenile and adult stony corals for space.

Nutrient criteria have been developed for USVI Class B waters for Total Phosphorous and Total Nitrogen with limits set at 50 µg L^{-1} and 207 µg l^{-1}, respectively. Data were available for ammonia, nitrate, and phosphorous (orthophosphorous). There are no standards for these nutrients. Examples of reporting limits (minimum acceptable values for analysis) for these molecules in USVI waters are: ammonia (10 µg L^{-1}), nitrate (1.5 µg L^{-1}), phosphate/orthophosphate (7 µg L^{-1}) (Smith et al. 2013). Values that are close to these reporting limits are reasonably likely to indicate low concentrations (oligotrophic) conditions for that nutrient in reference to stimulation of phototrophs.

Chlorophyll concentrations and deviations from mean conditions can be important indicators of nutrient pollution in tropical waters. In general, dissolved nutrients in oligotrophic tropical seawater are rapidly taken up and used by pelagic and benthic phototrophs for growth, thus, free-water dissolved nutrient concentrations are very low (Furnas et al. 2005). For this reason, water column chlorophyll, the concentration of photosynthetic pigments indicating phytoplankton abundance, is often used as a proxy for dissolved nutrients (Furnas et al. 2005). Chlorophyll a values greater than about 0.4 µg L^{-1} are indicative of enrichment above oligotrophic oceanic conditions based on research conducted south of St. John (Smith et al. 2013) and are similar to values found on the Great Barrier Reef (Furnas et al. 2005).

Fecal indicator bacteria. Fecal indicator bacteria, such as enterococcus, can indicate human and animal waste contamination and are used to assess the suitability of marine water for contact-based activities. Values that exceed 35 colony forming units 100 ml^{-1} (CFU) are associated with marine waters considered at higher risk for development of human illness (at a rate of 36 per 1000 persons; USEPA 2012). The USVI standard indicates the 30-day geometric mean of enterococci should not exceed 30 CFU for 30 consecutive days or values of 110 CFU should not be found in more than 10% of 30 samples.

Data used in this assessment for the above water quality variables were taken from published sources and online databases. The US Environmental Protection Agency (USEPA) stores publicly available water quality data at https://www.waterqualitydata.us. This database (STORET) was queried on September 10, 2019 for all data within 0.8 kilometers radius from a centroid located at the southern end of the Salt River Canyon at coordinate 17.78214 N, 64.75634 W. This radius covered all long-term monitoring sites that fell within the boundaries of SARI. Single-event water quality sampling was conducted by different researchers at a few locations, but the lack of temporal replication limited the information from this sampling (Kendall et al. 2005), and these sites were not included. The query identified four unique long-term water quality stations, representing 652 sampling events conducted by the Division of Environmental Protection of the USVI Department of Planning and Natural Resources for the Ambient Water Quality Monitoring Program and the Beach Monitoring Program. (Figure 4.2.1.1, Table 4.2.1.2). Data were retrieved from STORET after the year 2000.

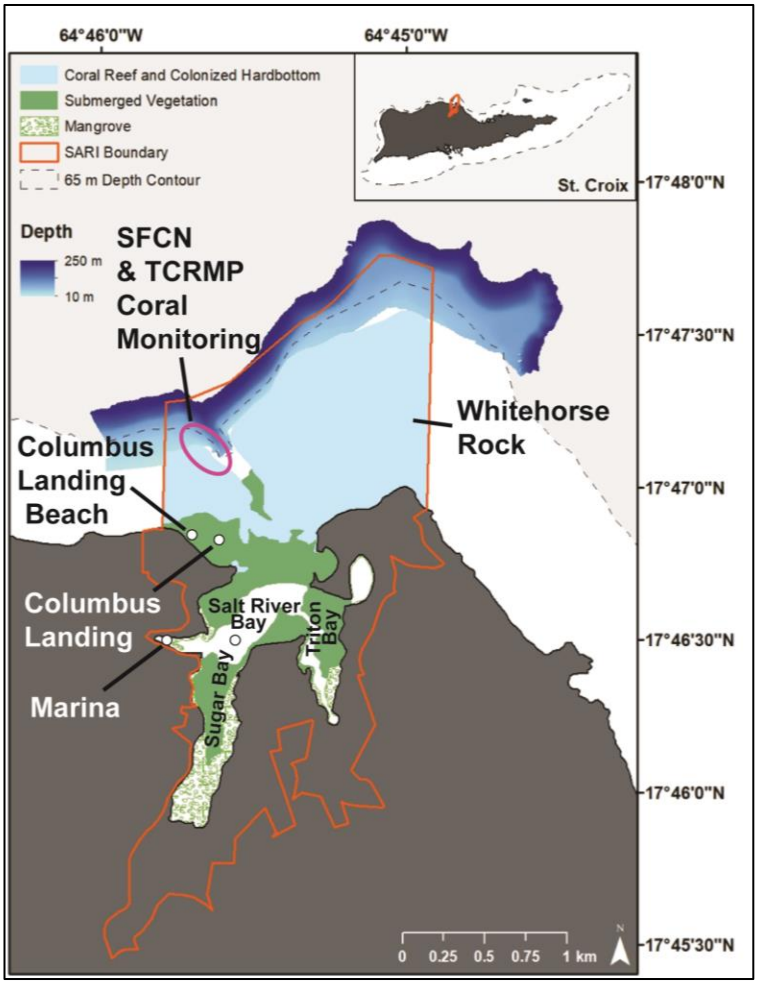


Figure 4.2.1.1. Map of the Salt River Historical Park and Ecological Preserve. Orange lines indicate preserve boundary. Monitoring areas for coral reefs included SFCN and TCRMP areas on the west Salt River wall. Monitoring areas for water quality are indicated with white circles: Columbus Landing Beach (STC-33A), Columbus Landing (STX-18), Salt River Marina (STC-33), and Salt River Bay (STC-33B) (Table 4.2.1.2). Bathymetry and habitat designations accessed from NOAA (8 August 2019, https://products.coastalscience.noaa.gov/collections/benthic/default.aspx).

T**able 4.2.1.2.** Water quality monitoring stations in SARI maintained by the USVI Department of Planning and Natural Resources Ambient Water Quality monitoring program (STX-18, STC-33 and STC-33B) and USVI Beaches Environmental Assessment and Coastal Health program (STC-33A). Metadata includes the coordinates, the start date and last included sampling of the program, as well as the number of sampling events (N).

Station	Description	Latitude	Longitude	Start	Last	N
STC-33	Salt River Marina	17.77517	−64.76183	6/30/00	6/01/05	14
STC-33A	Columbus Landing Beach	17.77992	−64.7586	6/30/00	5/15/19	53
STC-33B	Salt River Bay	17.77513	−64.758	3/18/08	5/15/19	36
STX-18	Columbus Landing	17.77997	−64.76008	8/9/04	8/27/18	563

Kendall et al. 2005 presented data prior to 2000 and a wider range of sampling sites, but for unknown reasons earlier data were not accessible from STORET in this query. Not all sites had representation of the full suite of water quality variables. Data were visually inspected for consistency in values over time that would indicate reliability and outliers that would indicate erroneous values. Site mean or median, standard deviation, and maximum value (or minimum for DO and pH) were calculated for represented variables, including DO, pH, salinity, TSS, turbidity, and fecal indicator bacteria.

Enterococcus values were represented in numerical data as "<10 CFU" for nine sampling periods. For graphical purposes these values were treated as a 0. Kjeldahl nitrogen, total nitrogen, and phosphorus concentrations were measured at Salt River Marina (STC-33), Columbus Landing Beach (STC-33A), and Salt River Bay (STC-33B). Kjeldahl nitrogen had lower detection limit of 500 μg L^{-1} prior to June 16, 2016 and 90 μg L^{-1} thereafter. Total nitrogen had a detection limit of 245 mg L^{-1}. Phosphorus had a limit of detection of 0.7 μg L^{-1}.

Reference Conditions/Values

There is a general paucity of information from which to establish reference conditions for Salt River Bay. Reference values for water quality prior to 1990 were taken inside the reef at the mouth of Salt River canyon and southward into the embayments. Since data collection started in 1981, there has been an apparent offshore to inshore gradient of water quality within SARI, with increasing turbidity and declining dissolved oxygen closer inside the embayments (Kendall et al. 2005). In many cases turbidity and dissolved oxygen at stations within embayments, such as the area of the Salt River Marina, had values below those acceptable for contact recreation and maintenance of marine life (Kendall et al. 2005). In the coral harboring areas around the Salt River canyon, it can be presumed from the historical presence of well-developed Holocene coral reef (Hubbard 1986) that water quality was good. Sedimentation and sediment transport in the areas immediately around and within the canyon was generally low, with the exception of during storm periods (Hubbard 1986). Sedimentation on the eastern canyon edge likely historically limited reef development to a low abundance coral community composed of more sediment stress-tolerant coral species (Hubbard 1986).

Current Condition and Trend

Coral reef sites outside of Salt River Bay

Similar to reference conditions, water quality in SARI shows strong spatial variability in an onshore to offshore gradient. Water quality along the margins of the Salt River canyon, where coral reefs are abundant, can range from fair to good. Water clarity can be very good (see Section 4.6.1), but also shows periods of higher turbidity after heavy rainfall events and with strong swell (T. Smith, unpub. obs.). Sediment flux of all sediment components and silt-clay sized particles (<0.63 μM) (primarily associated with coral damage, Weber et al. 2006) are up to an order of magnitude lower at the monitoring site of the USVI Territorial Coral Reef Monitoring Program (TCRMP) on the west canyon wall (Figure 4.2.1.1), relative to 20 other sites in the USVI measured with the same methodology (Salt River: $Flux_{Total} = 3.3 \pm 3.7$ SD, $Flux_{silt\text{-}clay}$ 0.6 ± 0.8, n = 27; Mean of 20 other sites in the USVI measured with the same methodology: $Flux_{Total} = 36.6 \pm 84.2$ SD, $Flux_{silt\text{-}clay} = 1.5 \pm 2.7$, n = 2314; T. Smith, unpub. data). Apparent sediment plumes emanating from the creeks (locally known as guts) draining into Salt River can reach the coastal areas near coral reefs (Kendall et al. 2005) but the emergent reef at the mouth of Salt River Bay may be a potent barrier to outward flow of terrigenous sediments to the reefs after major storms (Pait et al. 2020).

Water column nutrients are often low in tropical waters near coral reefs, such as the Great Barrier Reef of Australia, because free nutrients are rapidly scavenged by phytoplankton, often making chlorophyll concentrations more sensitive indicators of nutrient loading than nutrient concentrations (Furnas et al. 2005). This appears to be the same for the US Virgin Islands (Smith et al. 2013). In SARI nutrient concentrations may be low at the offshore coral reef locations, but this apparently has not been measured. Chlorophyll concentrations have also not been adequately measured in SARI. Observations of water color and clarity at offshore coral reef sites during monitoring do not suggest high chlorophyll levels, but levels are likely to be much higher inshore (T. Smith, unpub. obs.).

Indirect measures of water quality based on sediment contaminants, biological assays, and coral reproductive characteristics could indicate some potential issues with water quality in the coral reefs of SARI. Foraminifera assemblages from four coral reef sites spanning the outer areas of SARI showed index scores consistent with conditions suitable for coral reef growth (Bayless 2019). May and Woodley (2016) measured total phosphorus from sediment porewater samples on two coral reef sites on the east side of SARI (area near Whitehorse Rock, Figure 4.2.1.1). and found concentrations of 51.2 $\mu g\ L^{-1}$ and 79.9 $\mu g\ L^{-1}$, above the 50 $\mu g\ L^{-1}$ threshold for Class A, B, C waters. Ammonia-nitrogen, un-ionized ammonia, nitrite-nitrogen and inorganic phosphate were all in acceptable ranges. However, a later study by Bayless (2019) did find higher un-ionized ammonia at a site just east of SARI (SARI 1 about 300 m outside the eastern boundary). In addition, low δ^{15}N values and higher levels of zinc and lead in coral skeletons from SARI compared with Buck Island may be consistent with raw sewage and storm water run-off (Bayless 2019). An assay with the development of sea urchin embryos (*Lytechinus variegatus*) also showed evidence of arrested growth and abnormalities at two eastern coral reef sites in SARI (May and Woodley 2016; Bayless 2019). Furthermore, sediment porewater at an additional site just to the east of SARI and closer to residential communities and the town of Christiansted arrested development of 99.7% of embryos (SARI 1; Bayless 2019). Un-ionized ammonia in the porewater was high and may have contributed to the effects on embryos. Urchin fertilization assays at these same sites similarly did not demonstrate any issues. These indicators suggest contamination from unknown pollutants and it was posited that there could be upstream sources to the east associated with residential areas and the town of Christiansted (Bayless 2019). As a further indication of potential pollution, only 20% of colonies of the threatened elkhorn coral (*Acropora palmata*) had reproductive gamete development during the annual spawning period, which was the lowest of 34 other populations tested throughout the Caribbean (C. Woodley, unpublished data). Such reproductive inhibition can occur in corals exposed to toxins and endocrine disruptors, such as has been associated with oxybenzone in artificial human sunscreens (Downs et al. 2016).

Sites inside the Salt River Bay

Sites inshore from the Salt River canyon show more impacts from restricted circulation and land-based sources of pollution. A time series of water quality from 2000–2018 at four sites within SARI illustrates the spatial variability in water quality and deviations from conditions consistent with contact recreation and maintenance of marine life (Figure 4.2.1.2). The sites tended to be fully marine, as indicated by salinity values greater than 30 ppt. Values of pH were also consistent with tropical marine waters (near 8.0). Some deviations from water quality standards to more neutral pH conditions (≤ 7; 15 of 312 readings) and very basic pH conditions (≥ 8.3; 55 of 312), were likely due to sensor calibration issues, since it would be very unusual to have these values in marine waters, salinity was normal in these periods, and the deviations occurred across all sample sites on the same day (i.e., the same sensor had unusual values at multiple sites on the same day under the same calibration). The more oceanic Columbus Landing monitoring site tended to have low values of total suspended solids and turbidity. The sites Salt River Bay and the Salt River Marina had periods with very high total suspended solids and turbidity, and low dissolved oxygen. In particular, Salt River Bay and Salt River Marina had 13% and 57%, respectively, of dissolved oxygen values below the critical level of 4.8 mg L^{-1} for fishes (Prince and Goodyear 2006; Figure 4.2.1.2). However, these sites are exempted from the USVI Class B water quality standard of 5 mg L^{-1} (USVI 2019). All sites showed periods with enterococcus values higher than recommended for contact recreation (35 cfu 100 mL^{-1}). Periods exceeding recommended enterococcus values were highest (30%) at Salt River Marina, followed by Salt River Bay (15%), Columbus Landing (14%), and Columbus Landing Beach (9%).

Water column nutrients tended to be low when monitored in SARI (Figure 4.2.1.3). Kjedahl nitrogen had 76% of values below lower detection limits. There were occasional spikes of higher nutrients at the Salt River Marina station (3 of 20 samplings) and one high value at the Columbus Beach station. Total nitrogen was above lower detection limits across all sites, but all values were still less than 1mg L^{-1} and mean values (0.39 ± 0.11 SD) was near detection limits (0.245 mg L^{-1}). Phosphorus concentrations were also low across stations, with a mean = 0.033 mg $l^{-1} \pm 0.080$ SD. Values of 0.033 mg L^{-1} are at a purported threshold for soluble reactive phosphorus concentrations for Caribbean coral reefs above which macroalgal growth may be stimulated (Lapointe 1997).

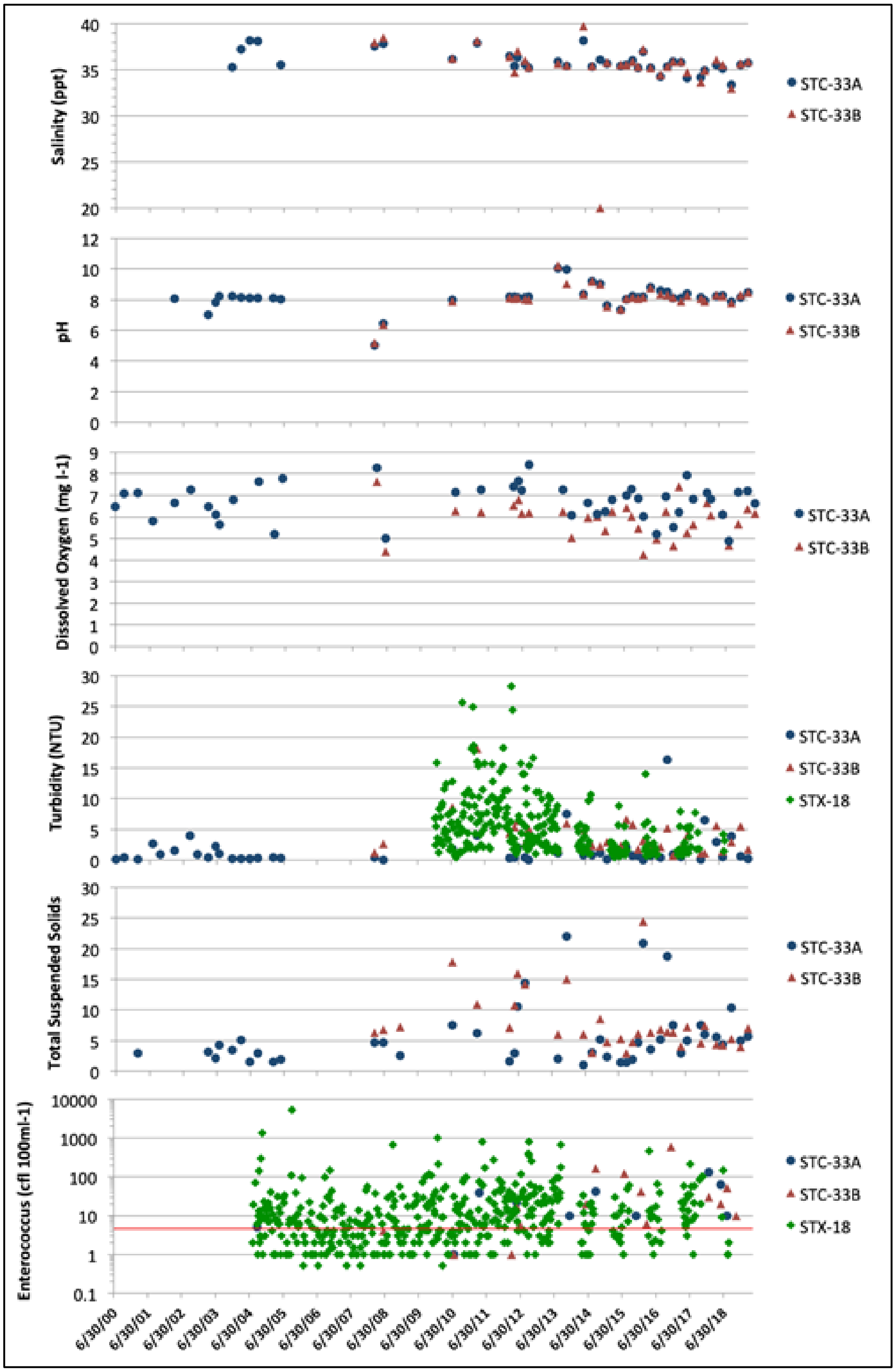


Figure 4.2.1.2. Water quality time series compendium from SARI sites listed in Table 4.2.1.2. For dissolved oxygen, the percentage of values that fall below 4.8 mg L^{-1}, the EPA value indicating impairment (USEPA 2000), are indicated by a red line on the dissolved oxygen figure panel. Fecal indicator values above 35 cfu 100 mL^{-1} suggest marine waters with elevated risk for contact-related human illness (USEPA 2012) and are indicated by a red line on the enterococcus figure panel. Site codes are Columbus Landing (STC-18), Salt River Marina (STC-33), Columbus Landing Beach (STC-33A), and Salt River Bay (STC-33B). Not all sites had representation of the full suite of water quality variables.

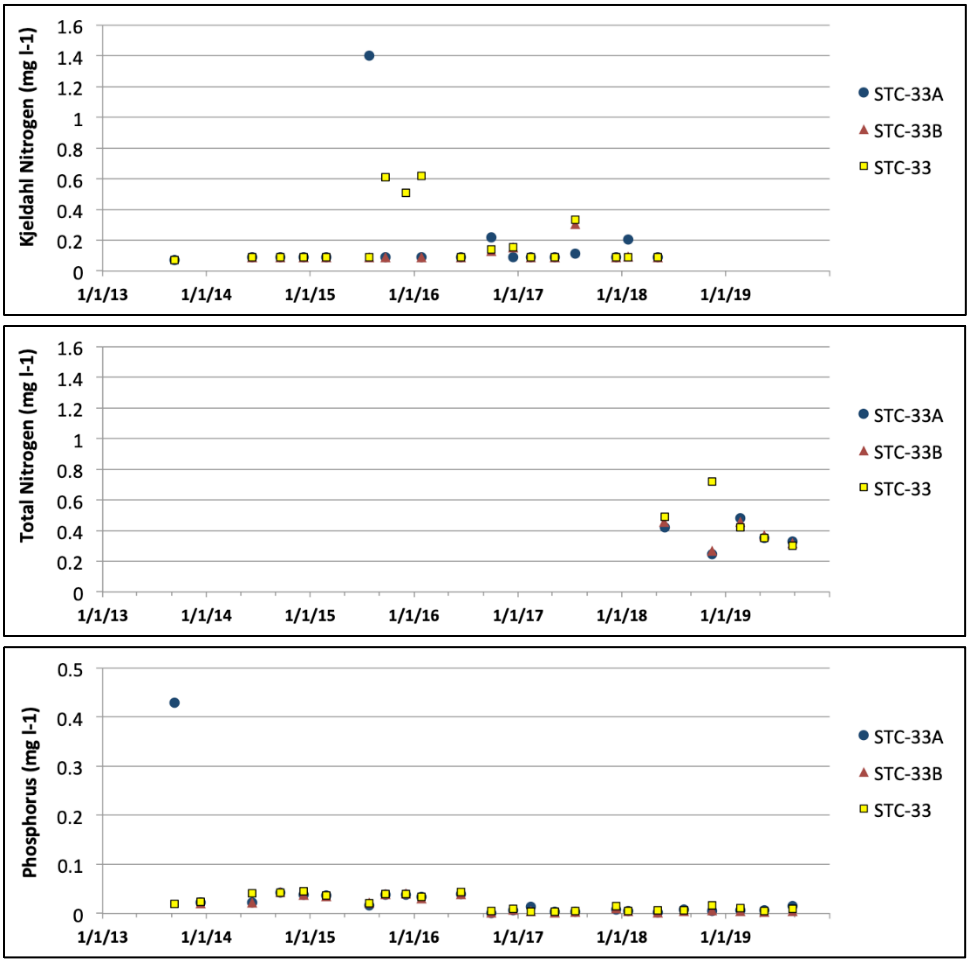


Figure 4.2.1.3. Water column nutrient concentrations from three long-term stations in SARI. Total Kjeldahl nitrogen (detection limit 0.09 mg L^{-1}) was recorded from Dec. 12, 2013 to Jan. 23, 2018. Total nitrogen (detection limit 0.24 mg L^{-1}) was recorded from May 7, 2018 onwards. Phosphorus (detection limit 0.0007) was recorded from 2013 to 2018. Site codes are Salt River Marina (STC-33), Columbus Landing Beach (STC-33A), and Salt River Bay (STC-33B). Data displayed in the graphs were retrieved from USEPA STORET https://www.waterqualitydata.us.

Indirect indicators of water quality showed possible issues with pollution inside Salt River Bay and the surrounding embayments. The sites Salt River Marina, Sugar Bay, and Triton Bay all had low foraminiferal index scores that could suggest high organic content in the sediment and high turbidity in the water column (Bayless 2019). The SARI sites Bioluminescent Pond (just north of Triton Bay), Salt River Marina, and Sugar Bay had sediment porewater with total phosphorus concentrations of 51 µg L^{-1}, 121 µg L^{-1}, and 100 µg L^{-1}, respectively, which are above guidelines (50 µg L^{-1}) for Class A, B, and C waters in the USVI (May and Woodley 2016). Ammonia-nitrogen, un-ionized ammonia, nitrite-nitrogen and inorganic phosphate were all in acceptable ranges. In a comprehensive survey of over 270 contaminants in SARI sediments inside Salt River Bay and surrounding embayments, Pait et al. (2020) found low to moderate concentrations relative to sediment quality guidelines. Sampling was conducted in September 2018 a year after impacts from Hurricane Maria, which may have homogenized surficial sediments within SARI (Pait et al. 2020). However, zinc and copper in the marina area were above or near concentrations at which impacts on organisms are expected. In addition, HRGS P450 assays, which indicate the presence of xenobiotic organic compounds, showed the presence of organic contaminants in the Marina area and Bioluminescent Pond that were not detected in the full contaminant scans (Pait et al. 2020). These two areas also had low benthic infaunal diversity, which can also indicate impacts for contaminants and/or low water quality. In addition, the anaerobic bacteria *Clostridium perfringens* associated with fecal wastes of humans and domestic and wild animals were detected in sediments across inner SARI, a possible indication of sewage pollution (Pait et al. 2020).

Threats and Stressors

Water quality at SARI is threatened by land-based sources of pollution. These pollutants may originate from upland agricultural activities, septic leaching fields from private residencies within the watershed (including terrestrial and live aboard boats), upstream sources associated with residences and the town of Christiansted, and contaminants from industry in the marina (Z. Hillis-Starr 2020, personal communication). This is particularly true for sites located landward of the barrier reef at the mouth of Salt River Bay. Water quality at coral reef sites is less threatened by land-based sources of pollution relative to the sites inside Salt River Bay. However, sediments associated with coral reefs show evidence of pollutants and this may be inhibiting coral reproduction. In addition, threats to the water quality for coral reefs in SARI comes from global factors related to greenhouse gas driven global warming and increasing carbon dioxide absorbed into seawater. Seawater temperature is detailed in Section 4.6.1. Ocean acidification is an emerging issue that will be a larger threat to stony corals and other calcifying organism in the future.

Data Needs and Gaps

The USVI Department of Planning and Natural Resources maintains an ambient water quality monitoring program that samples four sites within SARI and provides valuable information on water quality. However, the program is not comprehensive, as it does not sample coral reef areas around the canyon, has periodic lapses in sampling, and is episodic, potentially missing key acute events that impact water quality. An NPS led water quality sampling program with deployed sensors and discreet water samples would be a valuable addition to establish trends in water quality for SARI and could focus more preserving water quality for flora and fauna as opposed to human health. In particular, coral reef locations are very under sampled for water quality. A now decommissioned oceanographic station (Salt River ICON; https://data.noaa.gov/dataset/dataset/salt-river-bay-st-croix-u-s-virgin-islands) provided valuable continuous and real-time information on water temperature, salinity, light, winds, currents, etc. This site could be restored with logging sensors and frequent water sample collection to target other variables important to marine life and comparable with efforts of DPNR (e.g., total suspended solids and fecal indicators). This could also include deployed sensors for chlorophyll and turbidity and the use of remote sensing to detect water color. A calibration of satellite sensors was recently conducted for the northern USVI. This project could be a starting point for monitoring using remote sensing tools (Brandt et al. 2019). Furthermore, measurement of pH, alkalinity, and other variables related to the carbonate chemistry and aragonite/calcite saturation state would assist in understanding the emerging threat of ocean acidification to the calcifying organisms of SARI. Lastly, indications from recent studies on potential contaminants/pollutants and their impacts on sensitive biota, such as stony corals, merits follow up work to identify toxins and mitigate their impacts.

<u>Overall Condition</u>

The water quality of SARI is poor to moderate, depending strongly on the physical location within the park (Table 4.2.1.3, Table 4.2.1.4). Oceanic-influenced areas appear to have good water quality, consistent with contact recreation and maintenance of marine life. However, indicators point to potential toxins in sediments and impacts on sensitive stony corals. Farther inshore and inside embayments there are indications of water quality that is potentially harmful for contact recreation and marine life. Threats to current conditions offshore are largely derived from upland and upstream sources of pollution, and regional and global changes associated with human-induced climate change and ocean acidification. Threats to current condition inshore are related to increase land-development and human population pressures.

Table 4.2.1.3. Graphical summary of status and trends for Water Quality outside Salt River Bay.

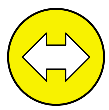
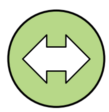
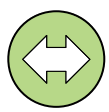
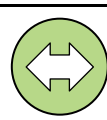
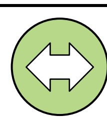
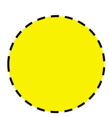
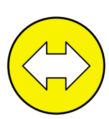
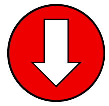
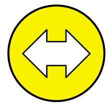
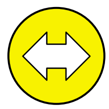
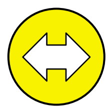
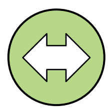
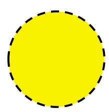
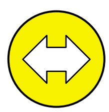
Component	**Indicator**	**Condition Status /Trend**	**Rationale and Reference Conditions**
Water Quality (outside Salt River Bay)	Fecal Indicator Bacteria		There are indications of nutrient pollution from sewage and impacts related to storm water discharge
	Dissolved Oxygen		Oceanic influenced areas have consistently good values. There do not appear to be strong trends in values over time.
	Total Suspended Solids		Total suspended solids are low in more oceanic areas. There do not appear to be strong trends in values over time.
	Turbidity		Oceanic influenced areas have consistently good values. There do not appear to be strong trends in values over time.
	Dissolved Nutrients		These are typically near detection limits in most areas. However, they may be a poor metric of nutrient loading.
	Chlorophyll		Chlorophyll concentrations have not been assessed directly but would provide a useful proxy for nutrient loading. Observations of water color and clarity at offshore coral reef sites do not suggest high chlorophyll levels, but levels are likely to be much higher inshore.
	Sediments		Terrestrial sediments have only been indirectly measured at one coral reef location and were low. There are indications of pollutants in some coral reef associated sediments

Table 4.2.1.4. Graphical summary of status and trends for Water Quality inside Salt River Bay.

Component	Indicator	Condition Status /Trend	Rationale and Reference Conditions
Water Quality (inside Salt River Bay)	Fecal Indicator Bacteria		There are indications of fecal contamination for some sites that periodically exceed values considered a risk for human contact. Continued development and poor enforcement of septic discharge may contribute to increasing incidences of fecal contamination.
	Dissolved Oxygen		Values in some constricted, low water exchange areas are consistently below values needed for support of marine life. There do not appear to be strong trends in values over time.
	Total Suspended Solids		Total suspended can be high inside embayments. There do not appear to be strong trends in values over time.
	Turbidity		Values in some constricted, low exchange areas are high and potentially harmful to photosynthetic organisms (e.g., foraminifera). There do not appear to be strong trends in values over time.
	Dissolved Nutrients		These are typically near detection limits in most areas. However, they may be a poor metric of nutrient loading.
	Chlorophyll		Chlorophyll concentrations have not been assessed directly but would provide a useful proxy for nutrient loading. Levels are likely high inside Salt River Bay.
	Sediments		There are indications of contaminants in some sediments and toxic effects on organisms. Effects of terrestrial sediments have not been evaluated inshore.

Source(s) of Expertise

- Amanda L. Bayless, National Institute of Standards and Technology
- Benjamin Keularts, Division of Environmental Protection, USVI Department of Planning and Natural Resources
- Cheryl M. Woodley, National Oceanic and Atmospheric Administration
- Anthony S. Pait, National Oceanic and Atmospheric Administration
- Richard W. Berey, CoastWorks

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4.3. Surface Hydrology

There are no perennial streams in the Virgin Islands, and most natural surface-water drainages remain dry for long periods of time. A few streams are intermittent that is they flow year-round in some reaches. Jolley Hill Gut on St. Croix, once reported to be perennial, is the only intermittent stream in the Island (USGS 1999). On an annual basis, in most natural surface drainage channels water flows only during episodes of intense precipitation. Because of the impermeable underlying volcanic rocks, floodwaters accumulate and recede rapidly, generally in less than one day. In a year of average rainfall annual runoff ranges from about 2 to 8 percent of the rainfall, which is about 1.25 to 5 cm (0.5 to 2 in), depending on conditions in a particular basin. Mostly the topography, but also soil moisture, local evaporation rates, and vegetation cover control runoff flowing into the SARI premises. Normally, total runoff from individual storms exceeds 10 percent of the rainfall and can be as high as 30 percent when rainfall is intense and soil moisture demands are low (USGS 1999). On St. Croix, part of the runoff is stored in ponds for agricultural uses. However, when floodwaters reach the coastal areas, they overflow existing salt ponds and provide freshwater inflow to embayments that support mangrove stands and coral reefs (USGS 1999).

The Salt River is not a perennial river, but it flows during certain periods of the year, mostly during the occurrence of extreme rainfall events. During these periods, surface runoff from the Salt River watershed (Figure 4.3.0.1) converges into the natural surface drainage channels of the basin. The Salt River watershed drains an area of approximately 1,360 ha. There are no periodic measurements of the flow of Salt River. The only long-term discharge observations available were made by the USGS for the period starting September 1, 1991, and ending September 29, 1993 (USGS 2019, Figure 4.3.0.2). Average daily flow was derived from measurements taken at the USGS station number 50348000 Salt River at Cannan, St. Croix USVI. The location of the station is shown in Section 4.3.1. However, the registered data do not cover the entire two-year period; these are substantial time intervals lacking information. The average flow over the observation period was 49 ft^3/s (cubic feet per second). The gage height reading during this time oscillated from 14.90 m (48.89 ft) over the period August 06–17, 1993 to 14.96 m (49.07 ft) over the period October 03–05, 1992. USGS station n° 50348000 is presently inactive.

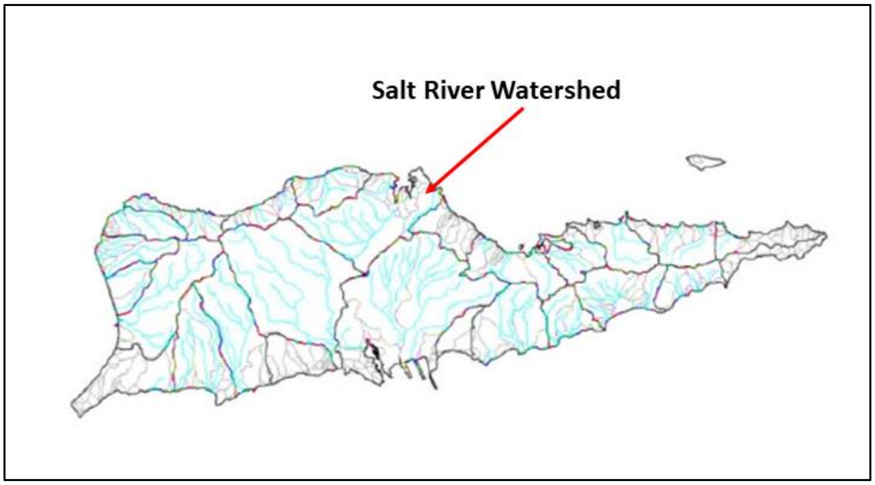


Figure 4.3.0.1. Map showing the watersheds of the island of St. Croix and depicting the location of the Salt River watershed (modified from WRI and NOAA 2005).

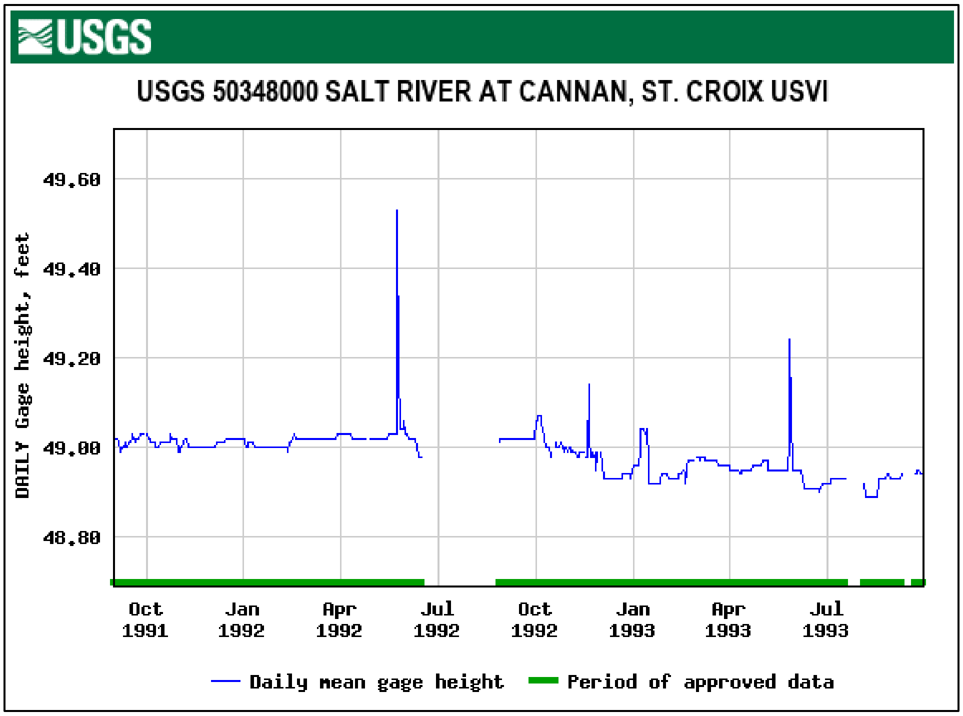


Figure 4.3.0.2. Measured flow at the location of the USGS station 50348000, Salt River at Cannan, St. Croix USVI (USGS 2019).

The Mon Bijou flood control project, completed in 2006, was constructed by the United States Army Corps of Engineers (USACE) in the upper reaches of the Salt River watershed to address flooding in residential areas (see Section 4.3.1). The Mon Bijou Levee System extends approximately 483 m (0.3 miles) along the right bank of Salt River, located at the north end of the Mon Bijou residential community (FEMA 2020). The project's function is to divert rainwater through a series of gabion structures before reconnecting with the Salt Run, decreasing water flow rate and transport of sediment through the gut, and potentially impacting the mangroves in Sugar Bay, as loss of terrestrial sediments decrease the magnitude of vertical sediment accretion in the mangroves (K. Whelan 2021, personal communication). The Mon Bijou System is continuous and uniformly constructed. It reduces flood hazards, but does not completely prevent these during extreme rainfall episodes, such as during the passage of hurricanes (FEMA 2020).

4.3.1. Watershed Condition

This section reviews the condition of the Salt River watershed. The condition assessment considers land use data for the years. 2002, 2007, and 2012. The state of the watershed is typically evaluated using metrics that detect changes originated by stressors such as unsustainable or poorly planned developments, population growth, erosion, among others. The condition metric selected for this resource is land use change. Changes in land use are viewed for the periods 2002–2007 and 2007–2012, and these are compared to a 2016–2018 composite ESRI basemap. The data to assess the status of the land use in the watershed was obtained from the National Oceanic and Atmospheric Administration (NOAA 2002, 2007, 2012). Temporal trends in condition metrics were evaluated using the land use change for the referred periods. Unfortunately, no detailed surveys exist after 2012, which would allow a more precise evaluation of the watershed at present.

Description

The Salt River watershed is located in the central northern part of St. Croix (Figure 4.3.0.1). The percentage of land of the Salt River watershed, equivalent to 1,360 ha (13.6 km^2—square kilometers), occupied by SARI is 14.87%. Within the Salt River watershed, SARI is located in the northeastern corner (Figure 4.3.1.1).

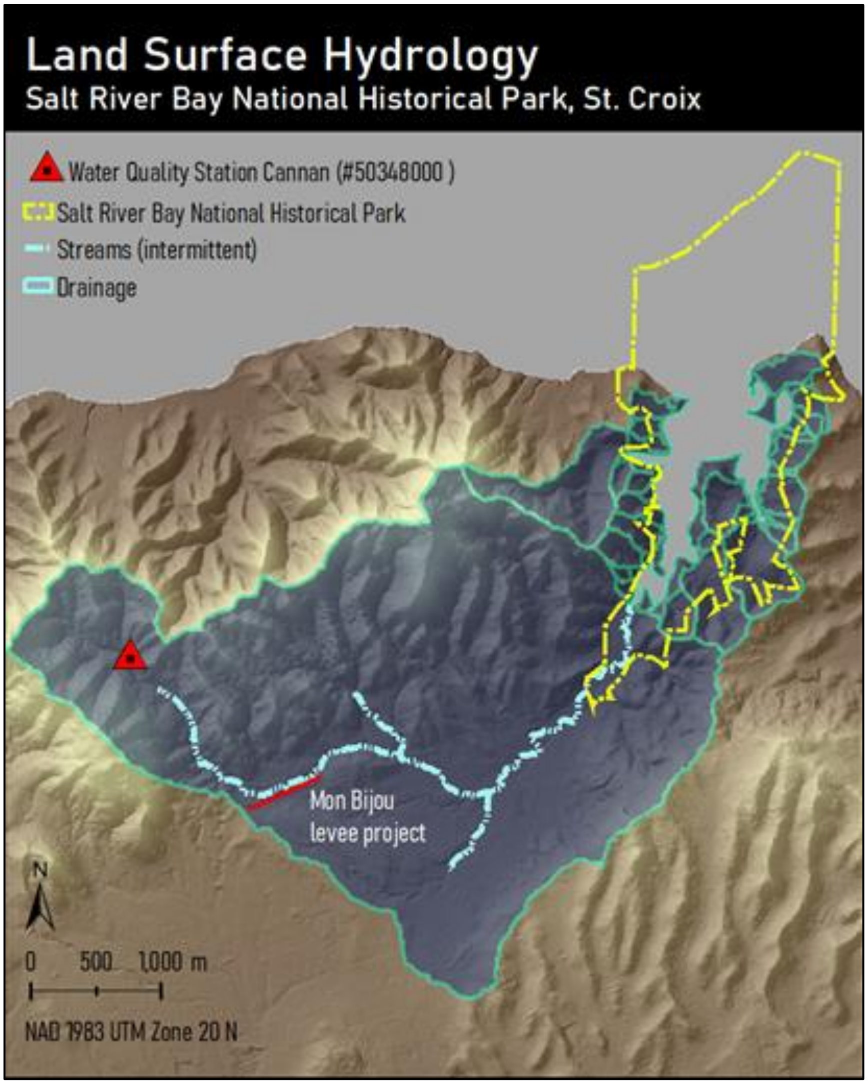


Figure 4.3.1.1. Salt River watershed (depicted with grey shading). Watershed, river, and smaller catchments were delineated in ArcGIS using Saint Croix DEM (St. Croix, U.S. Virgin Islands Coastal Digital Elevation Model – CKAN (data.gov)).

Human activities in watersheds influence the natural communities in them and affect ecological systems downstream. Similarly, changes in the landscape due to human interventions impact estuarine and other marine environments in coastal areas and adjacent zones. The more developed the watershed, the greater tends to be the intensity of the impact (Oliver et al. 2011). Land-based pollution constituted an important threat to delicate marine ecosystems such as corals (Emis et al. 2016). Pollution comes in the form of water-borne contaminants, sediments, debris, and other forms of exogenous materials. Pollutants may reach SARI via the discharge of the Salt River or surface runoff from neighboring areas surrounding the park (NPS 2015).

There are relatively few studies of the land cover/use of the Island of Saint Croix (University of the Virgin Islands 2001; Kendall et al. 2005; Moser et al. 2011). A first comprehensive data set organized in a common GIS geodatabase framework was put together by the Conservation Data

Center (CDC) of the University of the Virgin Islands. Vegetation maps created from 1995 aerial photographs were produced as part of the Rapid Ecological Assessment for St. Croix (University of the Virgin Islands 2001). Figure 2.2.1.2 presents land use for the Salt River watershed derived from the referred data set, while Table 2.2.1.2 provides the area of the watershed occupied by the various defined categories of land use.

In the early 2000s, the National Oceanic and Atmospheric Administration (NOAA) Biogeography Program, in consultation with the National Park Service (NPS) and the Government of the Virgin Islands Department of Planning and Natural Resources (VIDPNR), conducted an ecological characterization of SARI. This initiative consisted of three complementary components: a text report, digital habitat maps, and a collection of historical aerial photographs (Kendall et al. 2005). Literature review and year 2000 aerial photography were the base sources of information for the characterization of vegetation, land use, and soils in SARI resulting from this study. These results were also verified during field surveys carried out in 2004 (Kendall et al. 2005). Figure 2.2.1.1 in Section 2.2.1 of this report presents the Ecological Units within SARI based the data collected in 2000 and published 2005 (Kendall et al. 2005; NCCOS 2019). Using the same data, Table 2.2.1.1 presents the area of the park occupied by the various defined ecological units.

The two studies referenced in the previous paragraphs provided important information on the land cover/use of the Salt River watershed (University of the Virgin Islands 2001) and of SARI (Kendall et al. 2015). However, it is difficult to infer land use changes over time based on their maps and classification schemes because the scope and scale of their efforts differed and the intent of their respective characterizations was different. Therefore, for the purpose of analyzing changes in land use/cover over time, NOAA Office of Coastal Management (https://coast.noaa.gov/) data was used.

Land use/cover data have been used to determine the influence of anthropogenic modifications of the landscape on natural environments (Richmond et al. 2007; Oliver et al. 2011; Sabine et al. 2015; Donoso 2020). An indicator of anthropogenic activity calculated from land use/cover data used in various studies is the landscape development intensity (LDI) index (Brown and Vivas 2005; Oliver et al. 2011; Donoso 2020). The LDI has been used to define the relationship between modifications of the landscape due to human interventions and indicators of the health of natural ecosystems (Reis and Brown 2007; Oliver et al. 2011). In a study by Oliver et al. (2011), the LDI index was proven to be "more robust than other indicators of human activity, exhibiting negative correlations with stony coral colony density, taxa richness, colony size, and total coral cover".

Data and Methods

The land cover maps for SARI presented in this section were derived from the NOAA Coastal Change Analysis Program (C-CAP) national standardized land cover and change products for the coastal regions of the U.S. C-CAP products inventory coastal intertidal areas, wetlands, and adjacent uplands with the goal of monitoring changes in these habitats. The timeframe for this data is 2002, 2007, or 2012 (depending on the exact date of imagery used). These maps are developed through the automated classification of high resolution National Agriculture Imagery Program (NAIP) imagery, available Lidar digital elevation data, and assorted ancillary information (NOAA 2002, 2007, 2012).

Figure 4.3.1.2 depicts land cover/use in 2002, 2007, and 2012 for SARI. Similarly, Figure 4.3.1.3 shows the Salt River watershed land cover/use in 2002, 2007, and 2012. Figure 4.3.1.4 provides a more recent perspective of the land cover of the Salt River watershed. This image is derived from an ESRI basemap 2016–2018 composite. The majority of the watershed map was derived from 2016 aerials and only the urban section located in the southwest used aerial imagery from 2018.

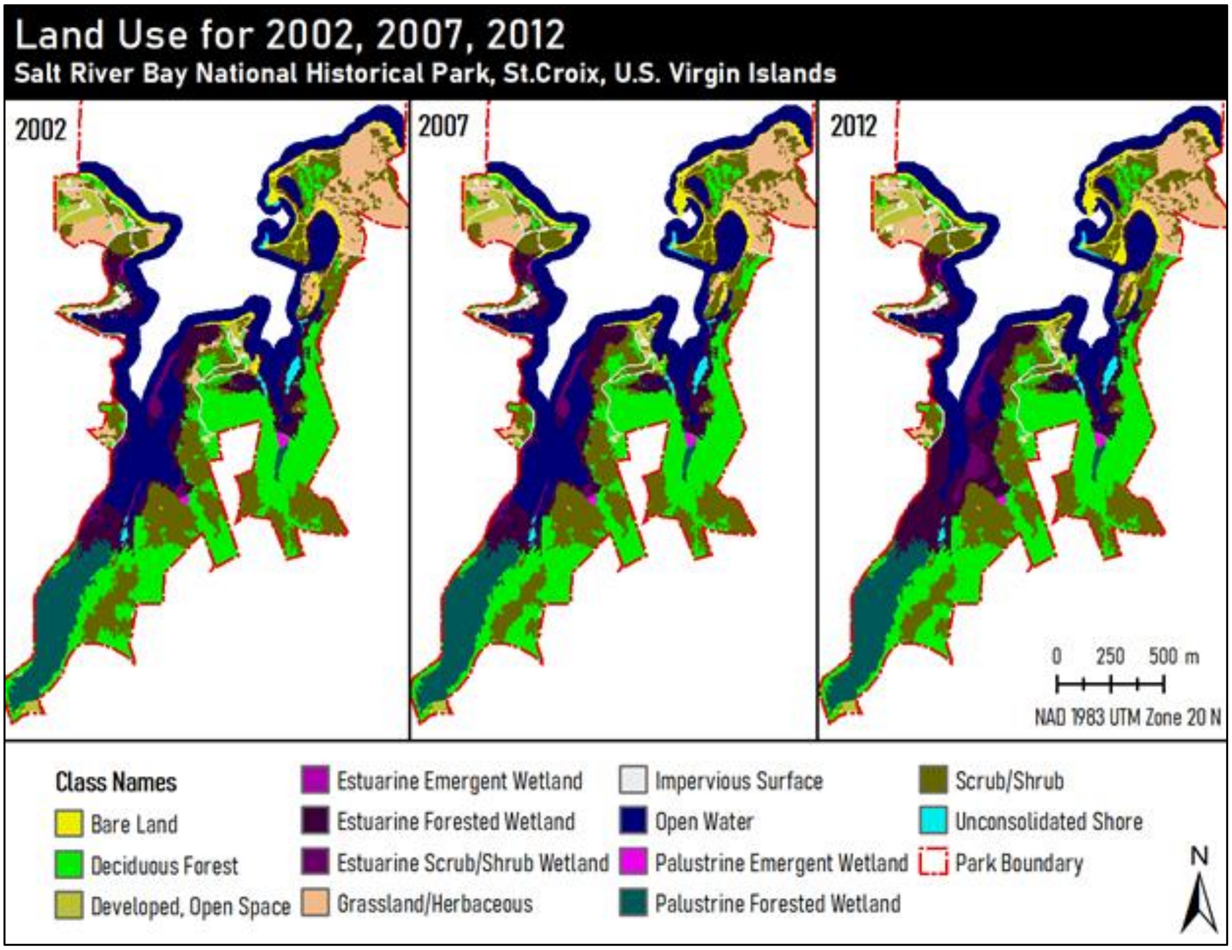


Figure 4.3.1.2. SARI land cover/use in 2002 (left panel), 2007 (middle panel), and 2012 (right panel). Land cover data from NOAA C-CAP 2002, 2007, and 2012.

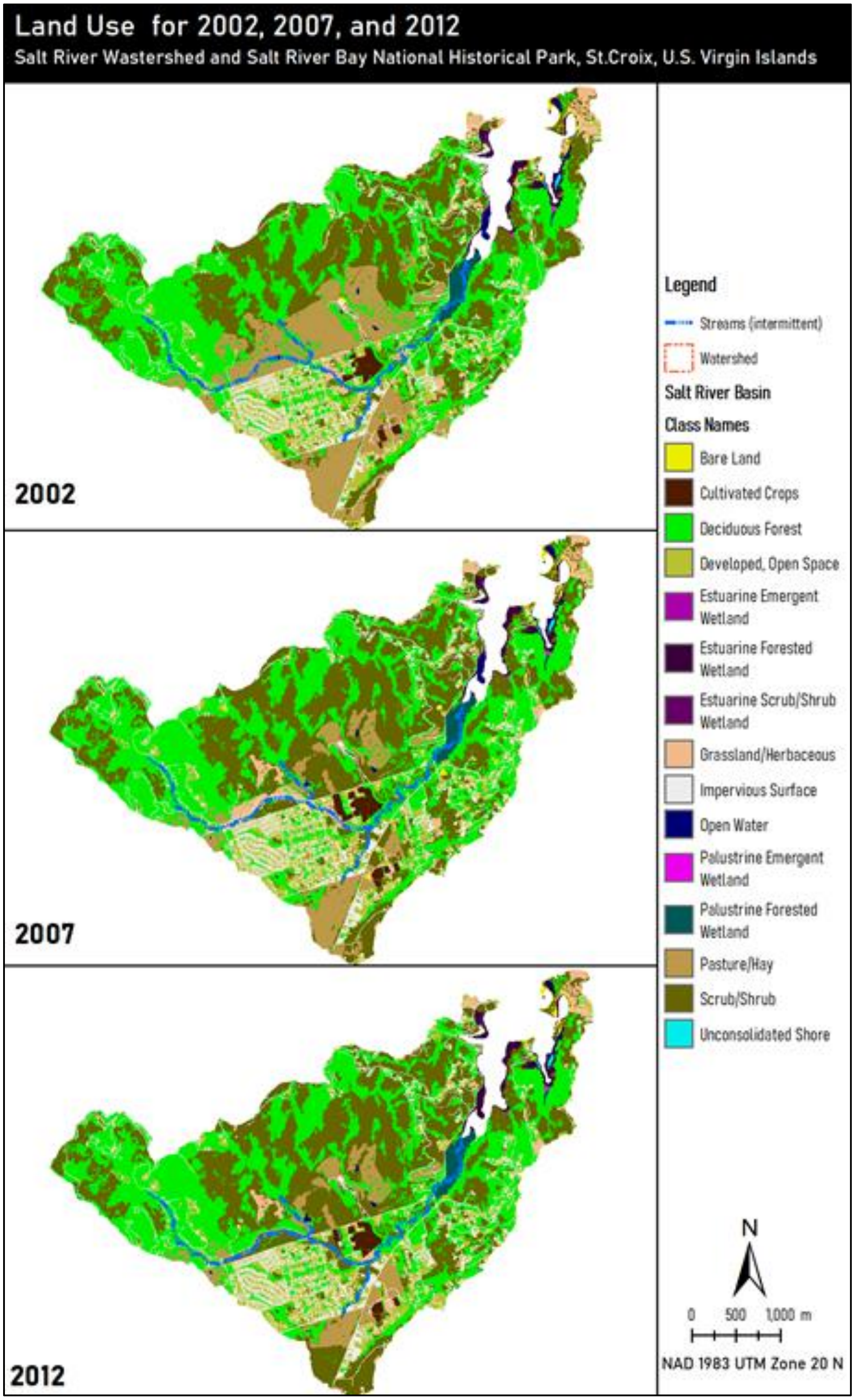


Figure 4.3.1.3. Salt River watershed land cover/use in 2002 (upper panel), 2007 (middle panel), and 2012 (lower panel). Land cover data from NOAA C-CAP. 2002, 2007, and 2012. Salt River watershed and smaller catchments were delineated in ArcGIS using the St. Croix DEM (St. Croix, U.S. Virgin Islands Coastal Digital Elevation Model – CKAN (data.gov)).

Figure 4.3.1.4. Salt River watershed on aerial imagery. Watershed border represented by the orange line. Salt River main surface drainage channels depicted by blue line. Boundary of SARI represented by the yellow hatched line. Basemap: 2016–2018 composite ESRI. Salt River watershed and smaller catchments were delineated in ArcGIS using the St. Croix DEM (St. Croix, U.S. Virgin Islands Coastal Digital Elevation Model – CKAN (data.gov)).

Changes in land use/cover in the Salt River watershed were defined for the periods 2002–2007 and 2007–2012, by comparing the land use/cover in the first year to the land use/cover of the last year of any given period. Changes in land use/cover for the Salt River watershed and SARI are captured in the maps provided in Figures 4.3.1.5 and 4.3.1.6 respectively.

Landscape development intensity (LDI) coefficients calculated by Oliver et al. (2011) using the Brown and Vivas (2005) methodology were assigned to each land use/cover type based on the classifications adopted by the NOAA Coastal Change Analysis Program. Table 4.3.1.1 presents the LDI coefficients assigned to each land cover/use class and the percentage of area corresponding to each class in the Salt River watershed. Similarly, Table 4.3.1.2 presents the LDI coefficients assigned to each land cover/use class and the percentage of area corresponding to each class in SARI. An area-weighted LDI index was calculated for each for the Salt River watershed and SARI as follows:

$$LDI_{TOTAL} = \left(\sum \%LU_i \times LDI_i\right) \div 100$$

where

LDI_{TOTAL} = area-weighted landscape development intensity (LDI) index for watershed/park

$\%LU_i$ = percent watershed (or park) land area in land use *i*, and

LDI_i = landscape development intensity (LDI) coefficient for land use *i*

Table 4.3.1.1. LDI coefficients assigned to each land cover/use class and the percentage of area corresponding to each class in the Salt River watershed (modified from Donoso 2020). Percent of the surface of the watershed for a given land use /cover class was derived from NOAA C-CAP 2002, 2007, 2012.

Land use /cover class	LDI coefficient	Percent of the surface of the watershed for a given land use /cover class (%)		
		2002	2007	2012
Bare Land	1.85	0.29	0.41	0.23
Cultivated Crops	4.42	0.79	1.08	0.93
Deciduous Forest	1.00	36.79	37.49	37.86
Developed, Open Space	1.85	7.82	7.83	8.18
Estuarine Emergent Wetland	1.00	0.01	0.01	0.01
Estuarine Forested Wetland	1.00	1.02	1.02	1.22
Estuarine Scrub/Shrub Wetland	1.00	0.06	0.06	0.02
Grassland/Herbaceous	2.06	2.32	2.92	1.80
Impervious Surface	8.28	8.88	9.46	9.61
Open Water	1.00	0.42	0.43	0.29
Palustrine Emergent Wetland	1.00	0.04	0.04	0.04
Palustrine Forested Wetland	1.00	0.99	0.99	0.99
Pasture/Hay	3.03	12.82	5.75	4.95
Scrub/Shrub	2.06	27.68	32.44	33.80
Unconsolidated Shore	1.00	0.06	0.06	0.06

Table 4.3.1.2. LDI coefficients assigned to each land cover/use class and the percentage of area corresponding to each class in SARI. Percent of the surface of the watershed for a given land use /cover class was derived from NOAA C-CAP 2002, 2007, and 2012.

Land use /cover class	LDI coefficient	Percent of the surface of the watershed for a given land use /cover class (%)		
		2002	2007	2012
Bare Land	1.85	2.32	2.41	2.59
Deciduous Forest	1.00	21.71	22.73	22.73
Developed, Open Space	1.85	2.05	2.04	2.26
Estuarine Emergent Wetland	1.00	0.06	0.06	0.06
Estuarine Forested Wetland	1.00	9.44	9.44	12.77
Estuarine Scrub/Shrub Wetland	1.00	1.38	1.38	1.61
Grassland/Herbaceous	2.06	9.22	7.71	7.58
Impervious Surface	8.28	2.22	2.26	2.43
Open Water	1.00	23.17	23.06	19.49
Palustrine Emergent Wetland	1.00	0.21	0.21	0.21
Palustrine Forested Wetland	1.00	6.87	6.87	6.87
Scrub/Shrub	2.06	20.70	21.22	20.80
Unconsolidated Shore	1.00	0.60	0.61	0.61

Reference Conditions/Values

The reference condition selected for the Salt River watershed was the condition of the basin in 2002, as derived from the NOAA Coastal Change Analysis Program (C-CAP) national standardized land cover product (NOAA 2002). The upper panel of Figure 4.3.1.3 shows the Salt River watershed land use in 2002. Within the boundaries of SARI, the land cover in 2002 can be observed in the left panel of Figure 4.3.1.2. The percent of the surface of the Salt River watershed for any given land use /cover class in 2002 is depicted in Table 4.3.1.1. Similarly, the percent of the surface of SARI for any given land use /cover class in 2002 is indicated in Table 4.3.1.2. The landscape development intensity (LDI) index calculated for the Salt River watershed for the year 2002 land use/cover of the Salt River watershed was used to represent the reference condition of human activity in the basin (Donoso 2020). Table 4.3.1.3 presents the LDI index of the Salt River watershed and SARI in 2002, 2007, and 2012.

Table 4.3.1.3. Landscape development intensity (LDI) index of the Salt River watershed and SARI in 2002, 2007, and 2012.

Geographical unit	Landscape development intensity (LDI) index		
	2002	2007	2012
Salt River watershed (Donoso 2020)	2.32	2.38	2.28
Salt River Bay National Historical Park and Ecological Preserve (SARI)	1.52	1.51	1.52

Current Condition and Trend

The examination of the land use/cover of the Salt River watershed for the period 2002–2012 (Figure 4.3.1.3) shows that there have been relatively small variations in the landscape over the 10 years. Changes in land use/cover for the Salt River watershed are shown in the maps provided in Figure 4.3.1.5. The analysis of these two sets of maps (Figures 4.3.1.3 and 4.3.1.5) indicates that changes occurred over the period 2002–2007 in the central and southern part of the basin. For the period 2007–2012, changes are localized in the south and southeast part of the basin, and along the upper half of the main Salt River drainage channel. Over the entire period (2002–2007), deciduous forest had the highest percentage of surface covered by an individual land use class or category, with a slight increase. In 2002, deciduous forests covered 37% (500 ha) of the surface of the Salt River watershed, 37.5% in 2007, and 39% (515 ha) in 2012. Scrub / shrub with a 27.7% (376.5 ha) coverage occupied the second largest area in percentage of the watershed surface in 2002. This type of vegetation has also expanded, occupying 32.4% (441.2 ha) of the surface of the watershed in 2007, and 34% (460 ha) in 2012. Pastures have declined in the watershed from covering a surface of 174 ha in 2002 (13% of the watershed's area) to occupying almost half this area, namely 67.3 ha (5%) in 2012. Grasslands increased over the period 2002–2207, from a surface of 31.6 ha, in 2002, to 39.8 ha in 2007, but subsequently declined to 24.4 ha in 2012. A similar trend was observed with cultivated crops; these increased their area coverage from 10.7 ha, in 2002, to 14.7 ha in 2007, but it declined to 12.6 ha in 2012. Likewise, bare land patches, which occupy a small percentage of the watershed (less than 40%), covered a total surface of 3.9 ha in 2002, 5.5 ha in 2007, and 3.1 ha in 2012.

Percent impervious surface is a relatively good indicator of surface water pollution in watersheds (Brown and Vivas 2005) and coastal areas. Clearing landscapes, creating impervious surfaces, and applying chemicals (pesticides, herbicides, and fertilizers) could simultaneously accelerate terrestrial runoff of sediments and associated chemical pollutants which cause decline in downstream ecosystems and in coastal zones habitats (Richmond et al. 2007; Oliver et al. 2011). As a measure of the intensity of human use of landscapes in the Salt River watershed, Oliver et al. (2011) calculations yielded the highest LDI coefficient value for impervious surfaces (8.28). Consequently, the increase of impervious surfaces cover could cause a decline in the wellbeing of humans and ecosystems in the watershed. The cover of impervious surfaces increased from 120.8 ha in 2002 to 128.7 ha in 2007, and 130.73 ha in 2012. This represents an increase of 10 ha in this category. By the end of 2012, impervious surfaces covered almost 10% of the area of the Salt River watershed. In terms of area covered by human developments and open spaces (classification "developed, open space), the total surface did not change for the period 2002–2007, remaining equal to 106.40 ha (7.8% of the surface of the basin); for the period 2007–2012, the area increased slightly, reaching 111.3 ha (8.2% of the basin). The total area covered by impervious, developed and open space surfaces was 227.2 ha, 235.3 ha, and 142 ha in 2002, 2007, and 2012, respectively, for a total gain of approximately 15 ha.

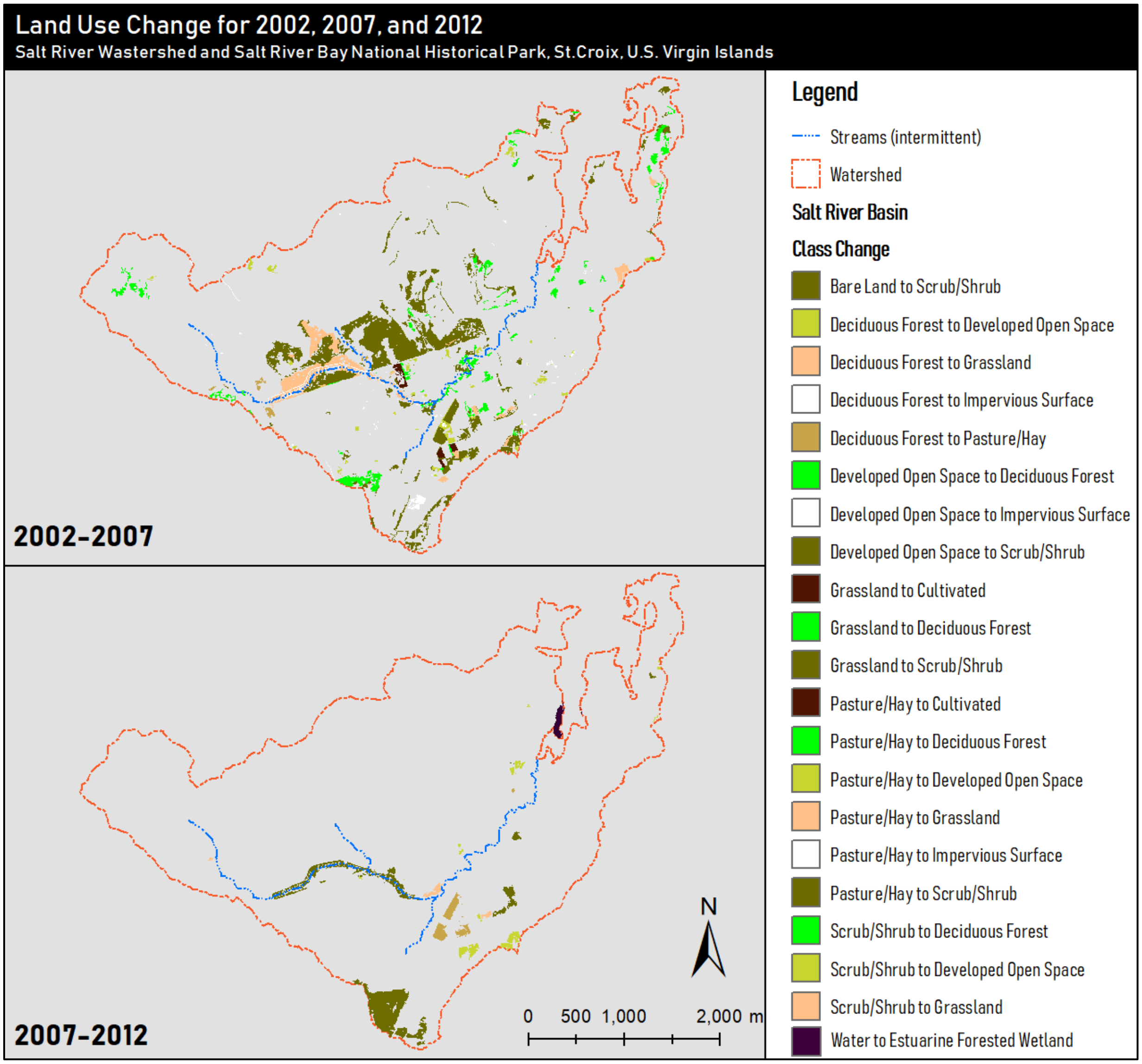


Figure 4.3.1.5. Changes in land use/cover for the Salt River watershed over the period 2001–2007 (upper panel) and 2007–2012 (lower panel). Land cover data from NOAA C-CAP. 2002, 2007, and 2012. Watershed boundary: red hatched line. Salt River watershed and smaller catchments were delineated in ArcGIS using the St. Croix DEM (St. Croix, U.S. Virgin Islands Coastal Digital Elevation Model – CKAN (data.gov)).

The total land cover of palustrine wetlands (emergent and forested) and estuarine emergent wetlands did not change over the period 2002 to 2012. Similarly, land covered by estuarine forested and scrub/shrub wetlands remained the same over the period 2002–2007. However, for the period 2007–2012, the land cover of estuarine forested wetlands increased from being 1313.9 ha to 16.6 ha, while the area of estuarine scrub/shrub wetlands declined from almost 1 ha to less than 0.4 ha. As of 2002, the total area of wetlands in the watershed was 28.9 ha and in 2012 it was 31.1 ha, an increase of 2.2 ha. Correspondingly, the total area of wetlands and forests in the Salt River watershed at the beginning of the study period was 529.4 ha (equivalent to 39% of the surface of the basin) in 2002, and in 2012, it was 546.1 ha (equivalent to 41.2% of the surface of the basin) which depicts an increase in cover of 16.7 ha (1.2% of the basin). Table 4.3.1.4 shows the change in land use trend for the various class of cover.

Table 4.3.1.4. Change in land use trend for various class of cover. Up arrow depicts cover area increase. Down arrow depicts cover area decrease. Horizontal double-sided arrow depicts no cover area change.

Land use / cover class	Arrow Direction
Bare Land	⇩
Cultivated Crops	⇧
Deciduous Forest	⇧
Developed, Open Space	⇧
Estuarine Emergent Wetland	⇔
Estuarine Forested Wetland	⇧
Estuarine Scrub / Shrub Wetland	⇩
Grassland / Herbaceous	⇩
Impervious Surface	⇧
Open Water	⇩
Palustrine Wetland (Emergent & Forested)	⇔
Pasture / Hay	⇩
Scrub / Shrub	⇧
Unconsolidated Shore	⇔

In summary, although there were changes in 17.5% of the land use/cover of the Salt River watershed for the period 2002–2012, these were not major changes in general. The most prominent one-category of land use changes correspond to the 83 ha increase of scrub/shrub coverage and the 107 ha decrease of pastures. Overall, there was a net gain of combined acreage of wetlands and deciduous forest of a little less than 17 ha. This increase slightly surpasses the increase in developed, open space and impervious surfaces, but is not significant enough to report an improvement on the condition of the watershed over the ten year period (2002–2012). All other changes increase or decrease of coverage were less than 15 ha, equivalent to a bit over 1% of the surface of the watershed. In terms of the impact of the observed land use/cover changes to the condition of the watershed from the perspective of the landscape development intensity (LDI), these are not significant. The value of the LDI index was calculated as 2.32, 2. 39, and 2.28, for the year 2002, 2007, and 2012, respectively (Donoso 2020). Oliver et al. (2011) calculated the LDI index to be 2.35 for 2007. The decrease of the LDI index over the last five years of the study period is not statistically significant to indicate a positive trend in the condition of the watershed due to a lesser effect of anthropogenic modifications to the landscape.

Comparing the land cover through the period 2002–2012 (refer to Figure 4.3.1.3) with the image shown in Figure 4.3.1.4, we observe that most of the Salt River watershed is covered by vegetation. Human settlements continue to be concentrated in the southern part of the watershed. Overall, the surface covered by vegetation and populated areas appear to have remained almost the same from 2012 to the date of the ESRI image composite (2016–2018). However, the lack of a more recent land use/cover survey impedes carrying out a detailed analysis such as the one carried out for the period 2002–2012. To determine the landscape development intensity of the watershed, it is necessary to have a detailed characterization of the spatial distribution of the various classes of land uses/covers present in the basin (Oliver et al. 2011; Donoso 2020).

Changes in land use/cover for SARI are shown in the maps provided in Figure 4.3.1.6. The observation of these two maps show that there are very few detectable changes in land cover SARI over the period 2002–2012. The more recent land cover image (Figure 4.3.1.4) shows that area within SARI continues to be covered by woody vegetation, which is consistent with the land cover shown in maps on Figure 4.3.1.2. This observation is consistent with previous studies (Moser et al. 2011) indicating that semi-deciduous forest sub-formation accounts for more than 80% of that habitat.

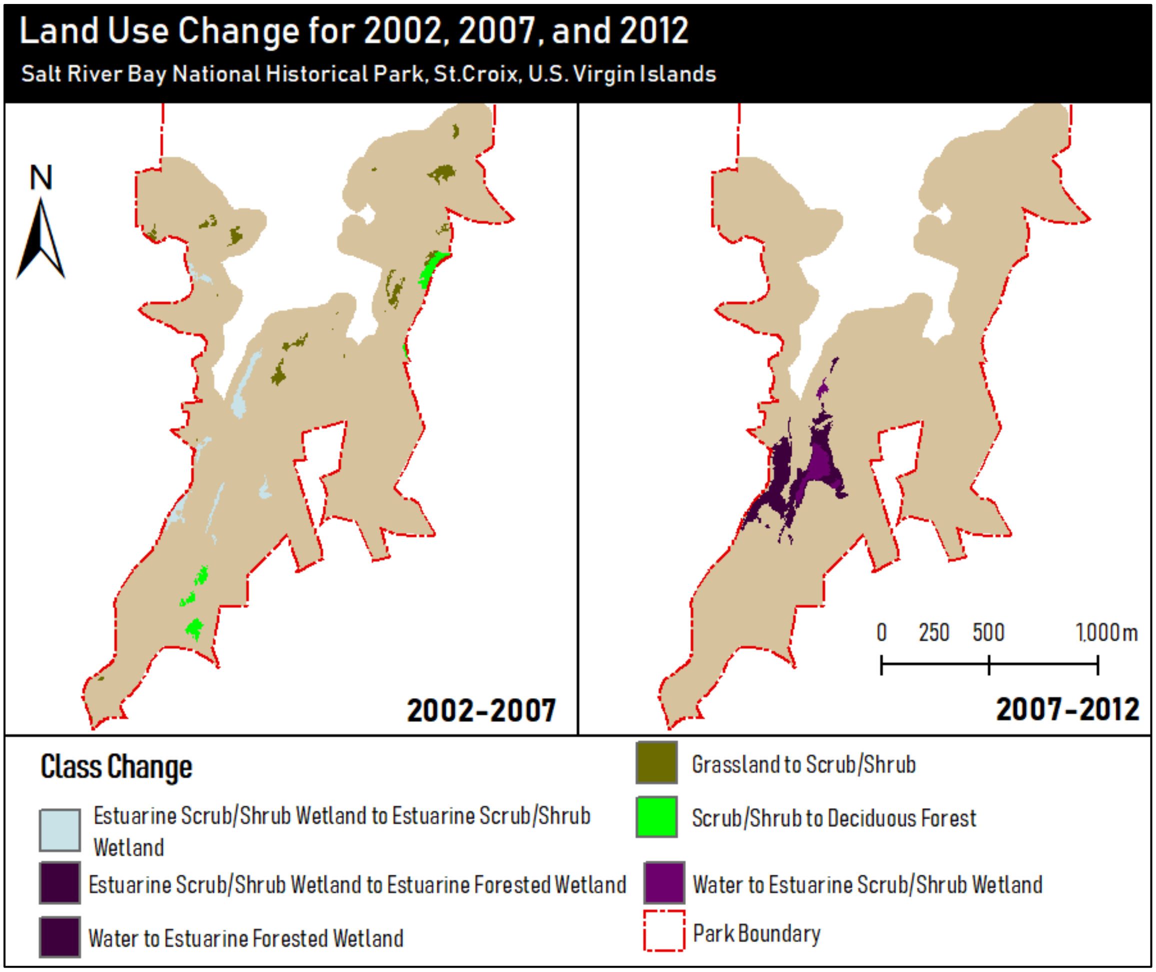


Figure 4.3.1.6. Changes in land use/cover for SARI over the period 2002–2007 (left panel) and 2007–2012 (right panel). Land cover data from NOAA C-CAP 2002, 2007, and 2012.

Threats and Stressors

The Salt River watershed contains a variety of habitats and ecosystems. In parallel, it also houses dwellings, commercial buildings and other developments. Human activities in the watersheds could affect natural communities downstream and the ecological systems within SARI. Changes in land use/cover due to human interventions could positively or negatively impact the watershed condition, as well as estuarine and other marine environments in coastal areas and adjacent zones (Donoso 2020).

Population growth could also be threat to the condition of the watershed. According the 2010 Census, the population at the time in St. Croix's population was 50,601 (US Census Bureau 2010). However, with the exodus that occurred after passage of Irma and Maria in 2017, the population in the Virgin Island had decreased. Furthermore, tourism was reduced due to the impact of the storms to hospitality facilities and services (USVI HRRTF 2018); consequently, the stress related to this industry was reduced. Similarly, today, with the advent of the COVID-19 health crisis, the influx of tourists, which was slowly picking up after the 2017 hurricane-related decline, has once more decreased. Therefore, it is difficult to measure the influence of population changes in the watershed, over the past years.

Infrastructure developments which redirect the flow in the vicinity of housing or commercial areas have caused the increase velocity and volume of the runoff into SARI. In addition, previous studies (Kendall et al. 2005) have reported that relict crop rows in the marsh south of Sugar Bay impacted the hydrology of the Salt River Gut and ultimately flow of water to Sugar Bay.

In general, erosion in the watershed is a major concern. However, sediment movement within the basin is complex with some areas losing sediment while other areas downstream may experience a gain, such as in areas within or nearby SARI. Upstream erosion from development and land cleaning within the watershed can have adverse effects both on the sedimentation rates and turbidity of the waters.

Pollution in the watershed comes from various sources. Trash and other large debris including tires, furniture, entire cars, buses and even boats can be found scattered in several places around the basin. Similarly, impromptu garbage disposal are observed when scouting the watershed. Pollutants in the water flow can be detected in the form of water-borne contaminants, sediments, debris, and other forms of exogenous materials. Water-borne contaminants can derive from spills or human waste discharges (NPS 2015). These can ultimately reach the park grounds as well as the estuarine and nearshore areas, affecting not just land ecosystem but also the marine environments. Other human threats to the Salt River environments includes more direct physical degradation and damage of natural habitats (Kendall et al. 2005).

Extreme weather events, such as hurricanes can pose a threat to the vegetation cover of the watershed. In the period from 1900 to 2018, 36 tropical storms came within a 50 mi distance from the Salt River watershed, of which 20 reached hurricane category (refer to Section 2.2.3 in Chapter 2 of this report). In particular in areas of the basin close to the coastline, tropical storm induced destructive storm surges can modify the landscape in coastal zones. During the passage of Hurricane Maria in September 2017, Christiansted Harbor on the north side of St. Croix observed a storm surge of two feet (Henson 2017). Hurricane force winds can also affect the land cover setting in the basin. Hurricane Maria affected all of the USVI, but St. Croix most severely, causing significant wind damage to roofs, structures, foliage, and aerial power and phone lines (USVI Hurricane Recovery and Resilience Task Force – USVI HRRTF 2018). Similarly, intensive precipitation can produce landslides, heavily impacting the land cover. Torrential rains during Hurricane Maria resulted in widespread flooding and mudslides throughout the St. Croix This island experienced at least an estimated 5–7 inches of rainfall (USVI HRRTF 2018).

Climate Change is considered a potential threat to the condition of the watersheds. Surge in air temperature, may increase the risk of wildfires in forested and shrub areas (USGCRP 2018). According the USVI Hurricane Recovery and Resilience Task Force, hurricanes are likely to become more damaging by the end of the century because they are likely to become more intense, although but not necessarily more frequent (USVI HRRTF 2018). Similarly, hurricane-related rainfall rates are expected to increase.

Data Needs and Gaps

Since the compilation the data set in this study (NOAA 2002, 2007, 2012), there have been no systematic long-term compilation of land use/cover parameters for the Salt River watershed. It would be useful to carry out an analysis of the watershed condition for a more recent date. For this purpose, the compilation of a data set depicting the same land use/cover classes as those in NOAA (2002, 2007, 20012) will be necessary. Similarly, since the SARI vegetation mapping conducted in 2009 (Moser et al. 2011), there has been no other major program to carry out a detailed mapping of the vegetation of the park.

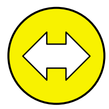
The availability of runoff information and flow measurement in the main Salt River drainage channel will be of interest, to better characterize the variations in the hydrological regime of the watershed. This information will be useful for managers at SARI to better understand the interconnection between the hydrology of the basin and the conditions of the resource in the park.

Finally, an interesting analysis that has not been carried out would consist in the sediment fingerprinting of the basin. This analysis will allow identification of areas of the basin that contribute the biggest volume of sediments. For this analysis, a soil sampling campaign needs to be designed and a systematic water sampling program should be developed to determine sediment loads in water discharge in various seasons (rainy and dry season). The benefits of this include helping USVI managers to identify areas undergoing high erosion within the watershed and would inform management actions to reduce the risk of landslides or reduce sediment loads into guts or those associated with runoff.

Overall Condition

Based on the assessment carried out, the condition of the resource warrants moderate concern (Table 4.3.1.5). The condition of the resource is considered to be unchanging. There is medium confidence in the assessment as it relates to the condition of the resource at present since there is no comparable data available to extend this analysis to a more recent date.

Table 4.3.1.5. Graphical summary of status and trends for watershed condition within the framework category of surface hydrology, including rationale and reference condition.

Component	Indicator	Condition Status /Trend	Rationale and Reference Conditions
Watershed Condition	Landover / Land use Change		The metric used to define the condition of the Salt River watershed was land use/cover change. The period used to define the trend was from 2002 to 2012. The land use in 2002 was considered the baseline condition. Based on the assessment undertaken, the condition of the watershed over the analysis period has not significantly changed in terms of land use/cover change. Notwithstanding the potential increase of the level of anthropogenic development intensity, warrants for moderate concern of the resource condition. The confidence of the assessment over the period of analysis is adequate, but not having comparable data for a more recent date, after 2012, makes the confidence of the assessment medium, as it relates to the present condition of the watershed.

Source(s) of Expertise

- Nate Herold, NOAA C-CAP Project Manager, NOAA Office of Coastal Management

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4.4. Terrestrial Plants

Salt River Bay NHP & EP consists of both marine and terrestrial components, including the last intact mangrove estuarine system in the USVI. The NPS and Government of Virgin Islands are co-managers overseeing the terrestrial portion of the park which includes NPS, GVI, and privately owned lands. This assessment covers the condition of three plant community types within the terrestrial portion of the park. These lands have been dramatically altered beginning approximately 2000 years ago with the arrival of pre-historic peoples from South America. In the last 300 years, the area has undergone substantial modification through agricultural practices (planting and harvest of sugar cane, cotton, and indigo), coastal dredge and fill operations (at least 5), harvest of useful hardwoods, livestock grazing, wildland fires, large-scale upland watershed and drainage mitigations, and hurricanes (Z. Hillis-Starr 2021, personal communication). The interaction of these factors has led to negative impacts to these terrestrial vegetation components within SARI.

In this section we focus specifically on changes to the extent of the park's semi-deciduous dry forest, coastal grassland, and mangrove communities within the greater terrestrial portion of the park (Figure 4.4.0.1). Together, these vegetation communities comprise approximately 110 ha, nearly three-quarters of the terrestrial area of SARI. Change in percent cover and change in species composition are the metrics used in these sections to access vegetation community extent (the selected indicator).

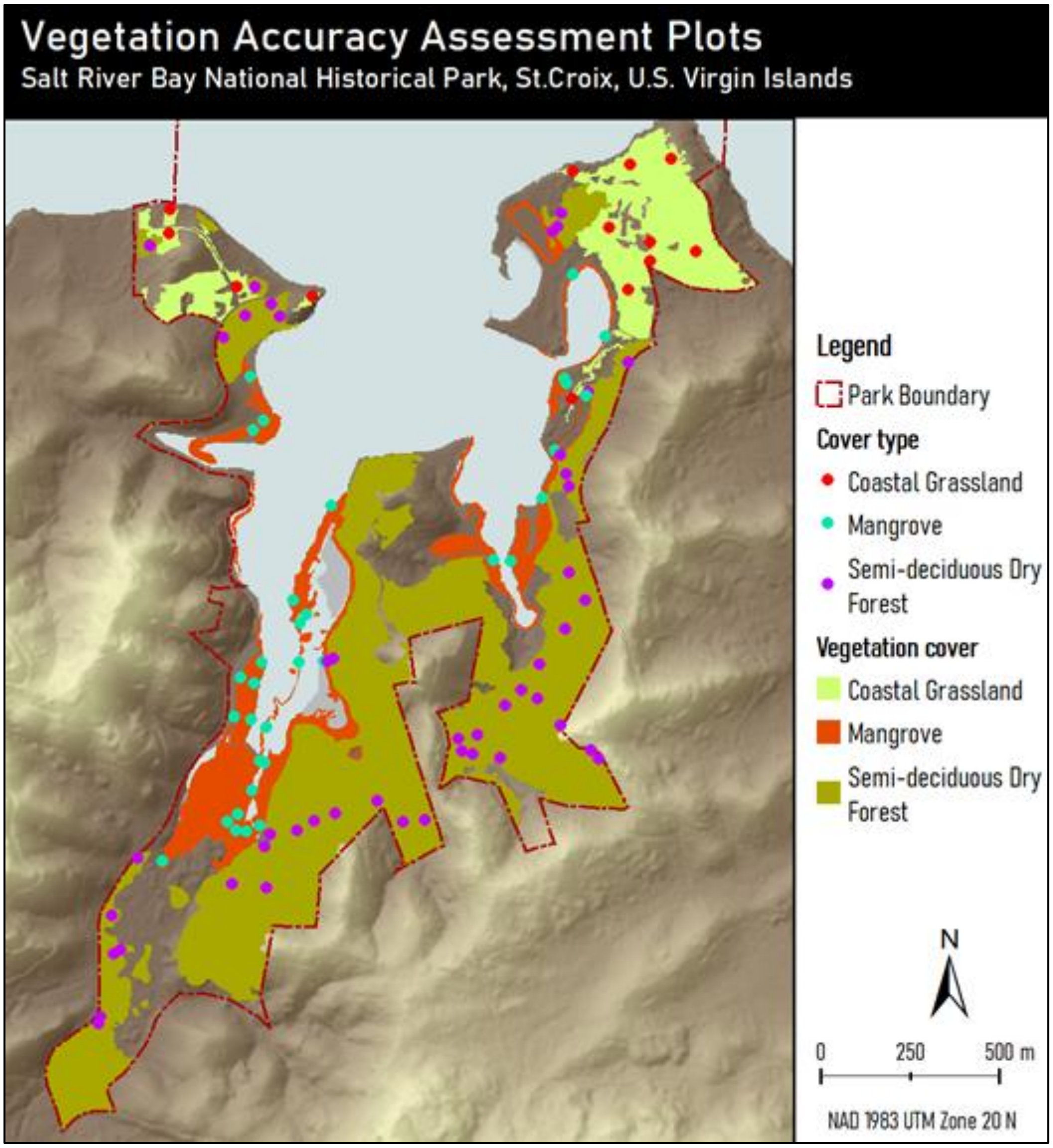


Figure 4.4.0.1. Areal extent of semi-deciduous dry forest, mangrove forest, and coastal grassland communities within the Salt River Bay National Historical Park and Ecological Reserve, classes aggregated from 2009 SARI Vegetation Mapping Project (Moser et al. 2011). Location of ~300 m^2 accuracy assessment plots included for reference.

4.4.1. Mangroves

This section reviews the condition of the mangroves in SARI. The condition assessment considers geospatial data for the years 1971, 1977, 1988, 1992, and 2006–2007 and plot-level data for years 2009, 2012, 2016, and 2018 provided by NOAA (1971–1992) and the South Florida Caribbean Network (2006–2018) to assess the status of the mangroves. The condition of mangrove forests is typically evaluated using metrics that detect changes in species composition, forest structure, fragmentation and habitat loss, diversity, percent cover of invasive species, mortality/damage, and changes in elevation. The condition metrics selected for this resource include change in species composition and change in percent cover. Temporal trends in condition metrics were evaluated.

Description

As of 2009, the mangrove community type in SARI occupied 18.7 ha of the terrestrial area at the confluence of land and sea, including forest, woodland, and shrubland classes, which are classes differentiated by tree height and density (Moser et al. 2011) (Figure 4.4.0.1). All three true mangrove species found throughout the Caribbean are present: black mangrove, *Avicennia germinans*, white mangrove, *Laguncularia racemosa*, and red mangrove, *Rhizophora mangle*. They are found as basin, riverine, and fringing mangrove ecotypes (Lugo and Snedaker 1974). Mangrove ecosystems provide numerous benefits to wildlife and humans, including storm protection (Blankespoor et al. 2017), nursery habitat (Nagelkerken 2009), carbon storage (Donato et al. 2011), and improved water quality (Adame et al. 2019). Mangroves in SARI have been documented as nesting habitat for breeding birds and serve as critical habitat for Nearctic-Neotropical migrant bird species (Wauer and Sladen 1992); Yntema et al. 2017). Adjacent mudflats provide foraging habitat for many waterbirds.

Prior to Hurricane Hugo (September 17–18, 1989), St. Croix had not been hit by a major hurricane for over half a century and as a result, mangrove forests contained trees greater than 6 m tall with closed canopies (Island Resources Foundation 1993). Hurricane Hugo caused extensive mortality throughout the forest, killing many old growth mangroves, and was especially severe in Sugar Bay (Island Resources Foundation 1993). As a result, live mangrove trees covered less than 50% of their pre-Hugo extent (Kendall et al. 2005). Both natural regeneration and reforestation efforts in the late 1990s/early 2000s led to a partial regrowth of the forest. Beginning in 1997, restoration efforts included the planting of red mangrove seedlings, primarily along the west side of Sugar Bay (SEA 2004). The mangrove seedlings had an estimated 80% survival rate, whereas test plots of planted black mangrove seedlings were largely unsuccessful (Kendall et al. 2005). Two separate mapping efforts have been conducted by NOAA (Kendall et al. 2005) and the South Florida Caribbean Network (SFCN) (Moser et al. 2011). Monitoring of forest condition using permanent plots has been conducted in recent years (2012–2018) within the basin forest of Sugar Bay. In 2012, the SFCN installed a Rod-Surface Elevation Table (R-SET) in mixed mangrove forest in Sugar Bay to monitor soil elevation in SARI mangroves (Whelan 2016). In 2017, SARI was impacted by two hurricanes, Hurricane Irma (September 6th and 7th) and Maria (September 18th and 19th), which resulted in tree mortality and damage within the forest.

Data and Methods

The indicator used to assess the mangrove component is community extent and includes two measures: the change in percent cover and the change in species composition. Datasets used for the analysis include the following:

1. Geospatial data of mangrove cover as digitized from aerial photography from 1971, 1977, 1988, and 1992 by NOAA (Kendall et al. 2005),

2. Geospatial data of mangrove cover mapped by community type from SARI Vegetation Mapping Project (Moser et al. 2011),

3. List of dominant and frequently observed species noted in thirty-three (~300 m2) circular plots visited in 2009 by the South Florida Caribbean Network (SFCN) as part of the SARI Vegetation Mapping Project (Moser et al. 2011), and

4. Species composition of two 10 m radius permanent forest plots established by SFCN in 2012 (NPS 2017).

Change in percent cover of total mangrove cover was assessed between the 1970s and 2009 from geospatial datasets derived from the two aforementioned mapping projects (Kendall et al. 2005; Moser et al. 2011). All mangrove classes (which varied by species, percent cover, and height classes between mapping efforts) were combined for this analysis. For the NOAA maps, mangrove habitat was mapped from aerial imagery obtained from the National Geodetic Survey for 1992 (post-Hurricane Hugo), 1988 (pre-Hurricane Hugo) and the oldest available photos which covered SARI (a mosaic of aerials from 1971 and 1977), hereafter referred to as 1970s mangrove extent (Kendall et al. 2005). The scale of aerial imagery ranged from 1:30,000 for the 1971 image to 1:20,000 for the 1992 image. The imagery was orthorectified and mangrove extent was digitized at a minimum mapping unit (MMU) of 100 m^2. For the SARI Vegetation Mapping Project, color orthophotos from the U.S. Army Corps of Engineers (2006–2007) having a 1 ft ground sample distance were used to map the extent of mangrove habitat and plots were visited in the field for training and accuracy assessment in 2009 (Moser et al. 2011). The MMU of this map, hereafter referred to as 2009 mangrove extent, was 400 m^2, (although some polygons were digitized at smaller MMUs) and photointerpretation digitization was done at a scale of ranging from 1:200 to 1:7000. Using ArcGIS 10.5, changes (1970s to 2009) in total extent of park area covered by mangroves was calculated and displayed in three categories: no change, mangrove loss, and mangrove expansion (Table 4.4.1.1, Figure 4.4.1.1).

Table 4.4.1.1. Spatial extent (ha) and relative coverage (% of total in each year) of mangrove cover by each class in 1970s compared to that of 2009. 1970s data from Kendall et al. (2005) & 2009 data from Moser et al. (2011).

Year	Black mangrove	Red mangrove	White mangrove	Mixed mangrove	Total cover (ha)
1970s	11.69 (57.9 %)	7.90 (39.1 %)	0.13 (0.6 %)	0.47 (2.3 %)	20.20
2009	2.36 (12.6 %)	7.20 (38.6 %)	2.71 (14.5 %)	6.40 (34.3 %)	18.67

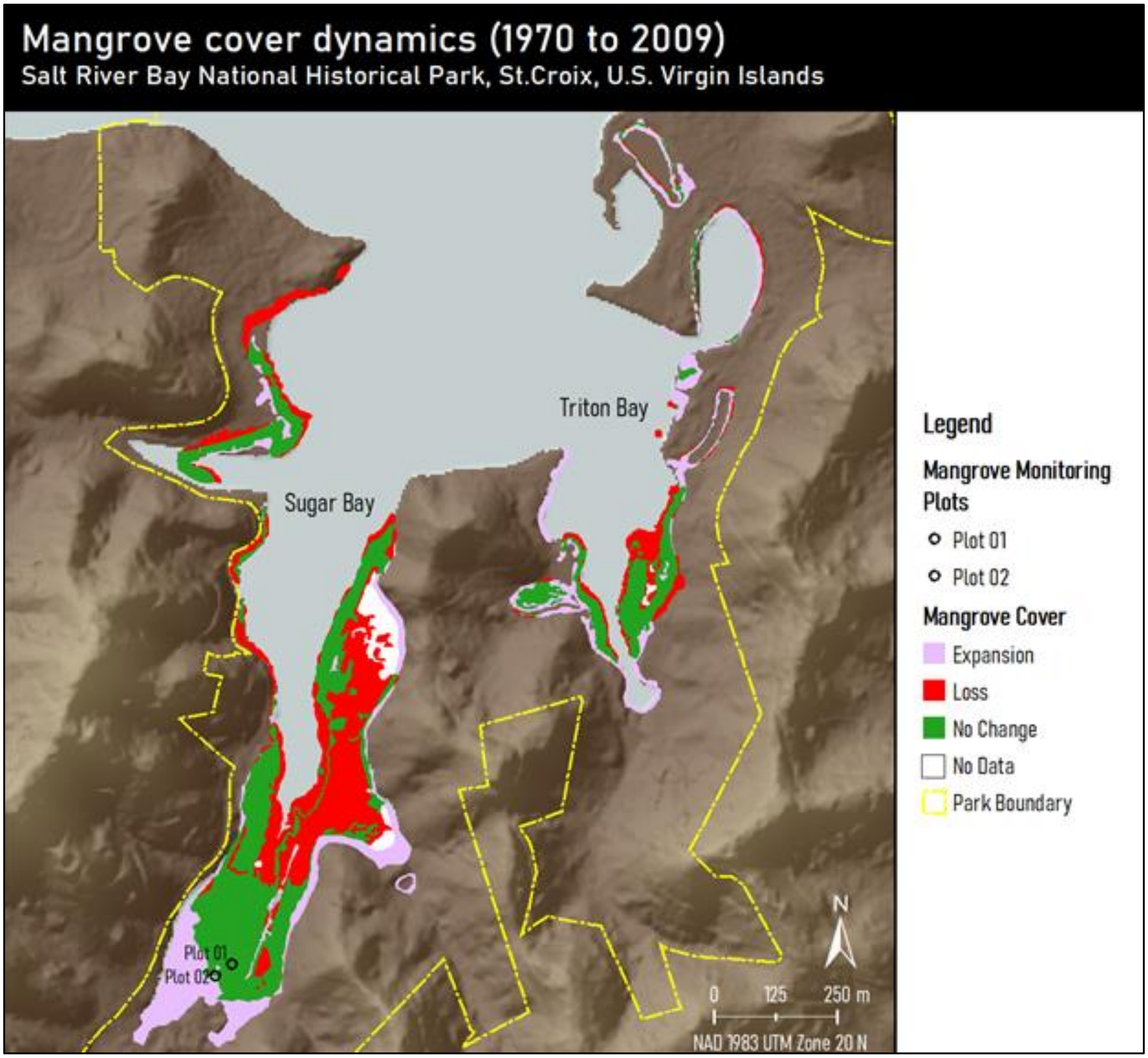


Figure 4.4.1.1. Areas of loss, expansion, and no change in overall mangrove cover in Salt River Bay National Historical Park and Ecological Reserve from the 1970s to 2009. Forest monitoring plots occurring in mangrove habitat are shown (NPS 2017). Cover data from Kendall et al. (2005) and Moser et al. (2011).

Changes in species composition from 1970s to 2009 were analyzed in 5 classes from those same datasets: red, black, white, and mixed species mangrove stands, as well as a mudflat/mortality class. Polygons of each class in 1970s were compared to what was mapped in those locations in 2009, resulting in a change matrix of area (ha) of the park occupied by each mangrove species (Table 4.4.1.2.). Compositional change in classes at each mapped time step (1970s, 1988, 1992, and 2009) are displayed as a 4-panel map (Figure 4.4.1.2). Differences in stand structure, including percent canopy cover and tree heights, were not considered for either of the above analyses as structural classes were not comparable between datasets. The estuarine coast of SARI was subdivided into four focal areas to summarize changes in area of mangrove forest over the approximate 40-year time period (Figure 4.4.1.3). These areas are as follows: 1) Sugar Bay, 2) Triton Bay (including the abandoned marina cut), 3) Northeast SARI (including East Cove and Dredged Basin), and 4) Northwest SARI (including Salt River Marina and Columbus Landing).

Table 4.4.1.2. Change in mangrove extent (area in hectares) for each composition class (black, red, white, or mixed mangrove, and non-mangrove) from 1970 to 2009. Data from Kendall et al. (2005) and Moser et al. (2011). Non-mangrove in 1970 is equivalent to areas of mangrove expansion in 2009, while 2009 Non-mangrove is equivalent to areas of loss.

Classes	2009 Black mangrove	2009 Red mangrove	2009 White mangrove	2009 Mixed mangrove	2009 Non-mangrove
1970 Black mangrove	**1.19** [1]	2.64	0.61	3.54	3.72
1970 Red mangrove	0.01	**2.43** [1]	0.13	0.82	4.51
1970 White mangrove	0	0	**0.03** [1]	0	0.11
1970 Mixed mangrove	0	0.04	0.03	**0.12** [1]	0.29
1970 Non-mangrove	1.16	2.10	1.92	1.93	**–** [1]

[1] Values on the diagonal indicate no change in composition class, also shown in bold.

Canopy cover and tree height were summarized from the data collected from thirty-three (~300 m^2) accuracy assessment plots visited in 2009 by SFCN (Figure 4.4.0.1). Two 10 m radius forest monitoring plots (see Figure 4.4.1.1) were visited in 2012, 2016, and 2018 by SFCN. Although the data are provisional and constitute a small sample size of the interior portion of mangrove extent within Sugar Bay, they provide a snapshot of changes to these forest stands within the last decade and include information on species frequency and forest structure, including diameter at breast height (DBH) and number of stems. Number of stems is reported rather than number of trees since a portion of mangrove individuals were multi-stemmed.

Reference Conditions/Values

The reference conditions for the mangrove community is the spatial extent and community composition as mapped by NOAA from 1970s mangrove extent (Kendall et al. 2005).

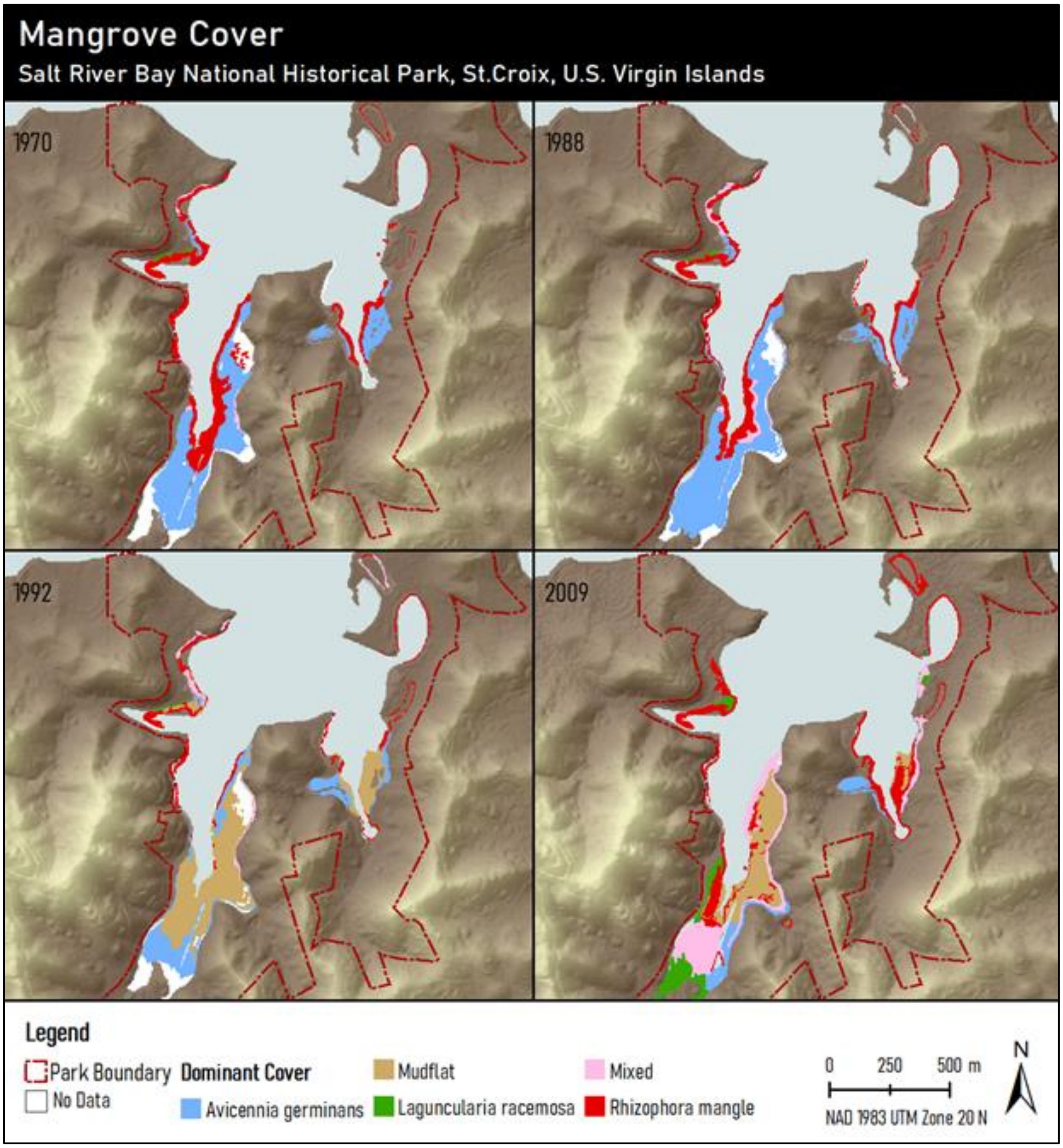


Figure 4.4.1.2. Mangrove cover as mapped by dominant species (*A. germinans, L. racemosa*, *R. mangle*) in 1970, 1988, 1992, and 2009. Mudflat corresponds both to areas of mangrove mortality after Hurricane Hugo and areas that were designated open tidal mudflat in 2009. Areas on the map described as having No Data are locations where spatial data was not available but appear on a 2000 NOAA land cover map as forested upland and, near the center of the panel, a saltwater pond. Mapped areas for 1970, 1988, and 1992 are from Kendall et al. (2005) and 2009 data is from Moser et al. (2011).

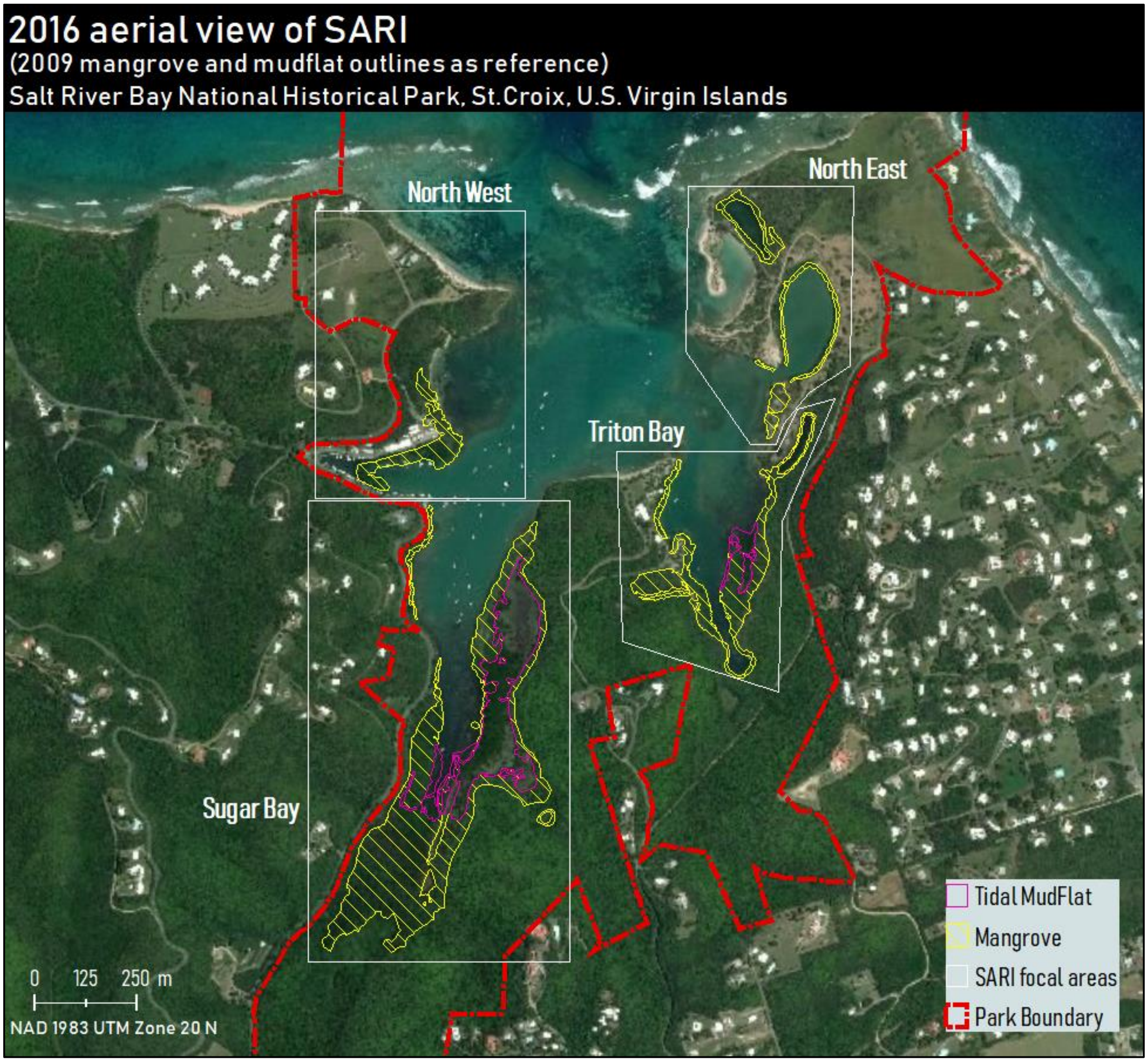


Figure 4.4.1.3. Extent of land cover mapped as mangrove or tidal mudflat in 2009 (Moser et al. 2011) overlying 2016 aerial photography. Designated focal areas within the estuarine environment of Salt River Bay: Northwest, Northeast, Sugar Bay, and Triton Bay.

Current Condition and Trend

Large changes in the mangrove cover and species composition have occurred within the mangrove ecosystem over a 40-year period despite small changes in overall mangrove extent (Table 4.4.1.1, Figures 4.4.1.1 and 4.4.1.2). Fifty seven percent of the land area that was in mangrove cover in the 1970s is still occupied by mangrove species today, corresponding to 11.6 ha that did not change from mangrove forest type but may have changed in composition of the dominant species (Figure 4.4.1.1).

Mangroves have expanded 7.1 ha into other habitats (1970s non-mangrove row, Table 4.4.1.2), including coastward as fringing mangroves and landward into lowland terrestrial forest and freshwater marsh. These gains have been marginally outpaced by mangrove loss of 8.6 ha over the same time period (2009 Non-mangrove column, Table 4.4.1.2). The greatest amount of this loss in mangrove cover occurred within Sugar Bay post-Hurricane Hugo (Figures 4.4.1.1 and 4.4.1.2), and while there has been recolonization in some areas that experienced mortality, other areas had not regained mangrove cover by 2009 and were now tidal mud flats communities (Figure 4.4.1.2). However, aerial imagery from 2016 shows colonization of mudflat on the east side of Sugar Bay approaching the coverage of mangrove as of 1988 in that area (Figure 4.4.1.3). Expansion of mangroves has primarily occurred at the landward margins of Sugar Bay, along the Salt River and Triton Bay, as well as surrounding salt ponds (Figure 4.4.1.1).

The relative coverage of each mangrove class in SARI has changed dramatically between the 1970s and 2009, with the exception of red mangrove, which maintained a relative cover of ~ 39 % (Table 4.4.1.1). In 2009, the mixed mangrove class was nearly as pervasive in the estuary as red mangrove, while the black mangrove class as of 2009 constitutes only 12.6% of the total mangrove area, having declined from being the most abundant class in the 1970s (57.9%). When mangrove expansion and loss are considered for each class, we see that most of the forest has changed in composition over the time period between 1970s and 2009, with very little mapped area remaining in the same class (Figures 4.4.1.1, Table 4.4.1.2). The extent of area that was black and red mangrove in the 1970s that remained in the same class as of 2009 is 10% (1.19 ha) and 31 % (2.43 ha) respectively (Table 4.1.1.2). A much larger percentage of originally black and red mangrove classes converted to open tidal mudflats as of 2009, 32% and 57% respectively (Table 4.1.1.2, 2009 Non-mangrove).

Changes in composition are noticeably tied to the passage of Hurricane Hugo and occurred primarily within Sugar Bay. A large amount of mortality to both the black and red mangrove forest is evidenced by changes in composition between 1988 and 1992 (Figures 4.4.1.2 and 4.4.1.4). Regeneration of the fringing red mangrove forest is mostly apparent on the west side of Sugar Bay, coincident with locations of restoration. Recolonization of the interior basin mangrove forest by black and white mangroves, with white mangrove solely colonizing the highest intertidal regions of the forest (Figure 4.4.1.2, 2009), is a post-disturbance pattern of species zonation observed in other locales (Piou et al. 2006).

Changes to the extent of mapped mangrove cover varies by focal area within SARI (Figure 4.4.1.3). While decreases in mangrove cover attributable to Hurricane Hugo (between the 1988 and 1992 mapped extent) are evident across most of SARI, the largest decreases occurred in Sugar Bay, followed by Triton Bay (Figure 4.4.1.4). Total mangrove cover in Sugar Bay has since rebounded but has not reached that mapped in the 1970s, and black mangrove has been replaced by white and mixed mangrove stands. Triton Bay follows the same general pattern yet has expanded beyond the 1970s extent. Mangroves in the Northeast of SARI have increased over the time period and do not seem to have been significantly impacted by Hurricane Hugo. Mangrove cover is more than 2 times that of the 1970s extent but was still the smallest of all the areas considered here. Changes in the amount of area in mangrove cover diverge in the Northwest section compared to that of the rest of SARI, mixed mangrove cover increased after Hurricane Hugo but has since been eliminated. The cause of the decline post Hurricane Hugo is related to coastal development in that area.

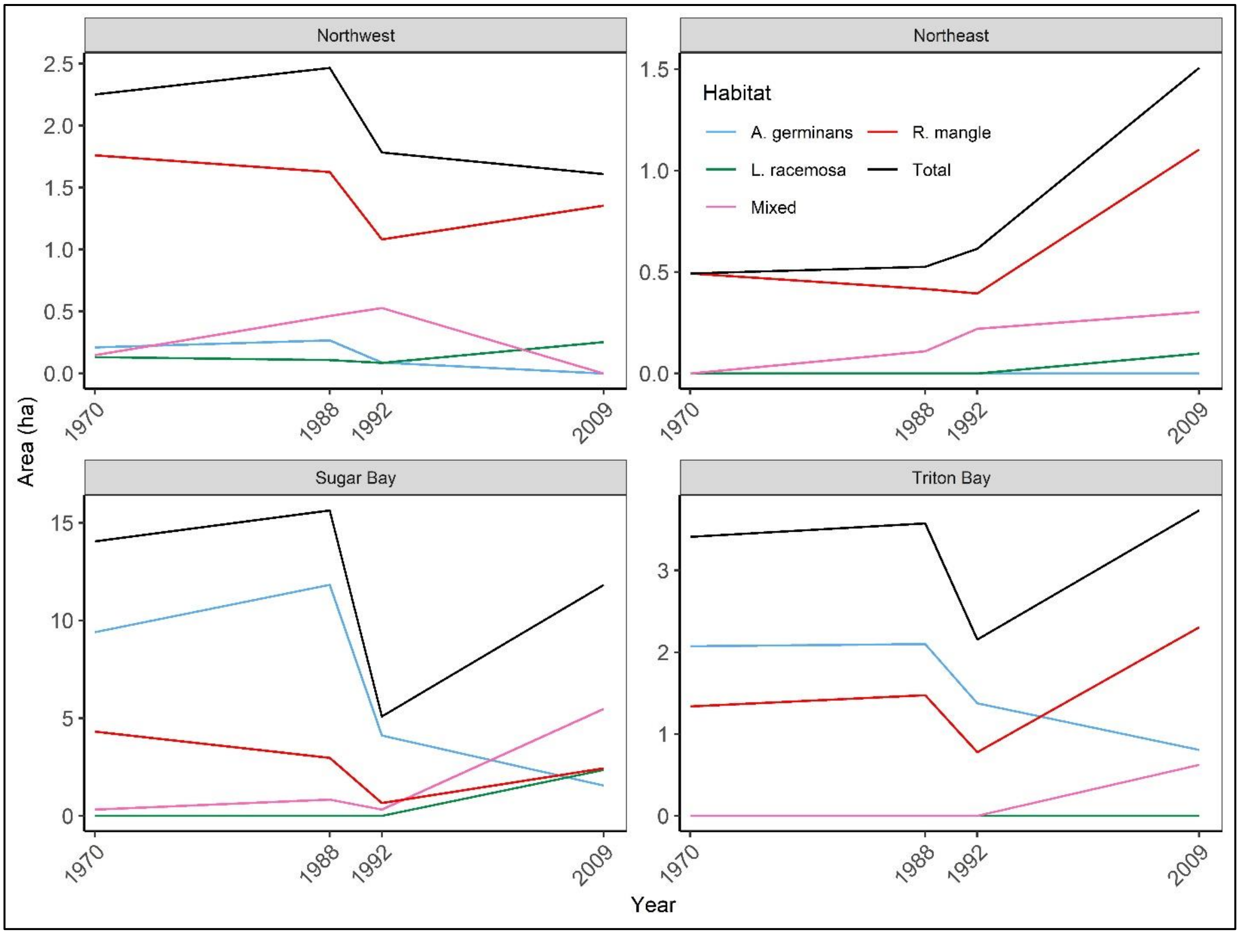


Figure 4.4.1.4. Area (ha) in each mangrove class (*A. germinans*, *L. racemosa*, *R. mangle*, mixed mangrove, and all classes combined, i.e., total) at each mapping date broken out by focal areas of SARI: Northwest, Northeast, Sugar Bay, and Triton Bay. 1970, 1988, and 1992 data from Kendall et al. 2005 and 2009 data from Moser et al. (2011).

During 2009 sampling for land cover map accuracy assessment, thirty-tree (~300 m^2 radius) plots were visited in the mangrove areas. The three aforementioned mangrove species were listed as the dominant/co-dominant species in all plots with the exception of two locations where the non-native invasive mangrove-associate, seaside mahoe, *Thespesia populnea*—identified by CABI (2019) as invasive, but considered to be native by Acevedo-Rodriguez (1996)—constituted at least 50% of the plots. Canopy heights ranged from 2 to 4 m in mangrove shrubland and from 5 to 10 m in mangrove forest. Percent canopy cover was generally greater than 60% in all plots and as much as 90%.

Mangrove Forest Monitoring Plots

Both mangrove forest monitoring Plots 1 and 2 (locations shown in Figure 4.4.1.1) are located in mixed mangrove forest composed of *A. germinans* and *L. racemosa* (Moser et al. 2011) and prior to the 2017 hurricane season, the plots had canopy covers of ~65% and ~ 75% respectively (data from SFCN forest monitoring plot database). Large changes in cover and composition have occurred at the location of the plots over the past 40 years. As of 1970, Plot 1 was located within a black mangrove dominated basin forest with a relatively open canopy (15–65% canopy closure estimated from map class annotation), and Plot 2 was located just on the landward edge of the areas mapped as mangrove (Kendall et al. 2005). By 1988, one year prior to Hurricane Hugo, canopy cover was described as closed (>65%) for both plots, and landward expansion of mangroves had moved the mapped area approximately 50 m interior (Figure 4.4.1.2). However, after Hurricane Hugo, the 1992 cover extent was sparse (1–15% canopy closure) for Plot 1, and Plot 2 was no longer classified as mangroves, suggesting both plots fell within areas of extensive damage from the hurricane. Recolonization by a combination of black and white mangroves has resulted in a forest that is recovering but has not yet reached pre-Hurricane Hugo forest structure.

Comparison of the number of stems in five DBH classes (< 5 cm, 5 to < 10 cm, 10 to < 15 cm, 15 to < 20 cm, 20+ cm) between plots (Plot 1 and 2) and between years (2012, 2016, and 2018) shows differences in stand structure in space and time (Figure 4.4.1.5, Table 4.4.1.3). A far greater number of smaller stems of both black and white mangrove were observed in Plot 1 as compared to Plot 2 in 2012, although both plots are clearly located in a relatively young forest. Both plots show a decrease in both species between 2012 and 2016, with a larger decrease in Plot 1 (Table 4.4.1.4). The reason for this decrease in small stems in Plot 1 is unknown. However, severe drought conditions were prevalent across the USVI and Puerto Rico during the summer of 2015 (NRCS 2015) and could be a cause of mangrove mortality.

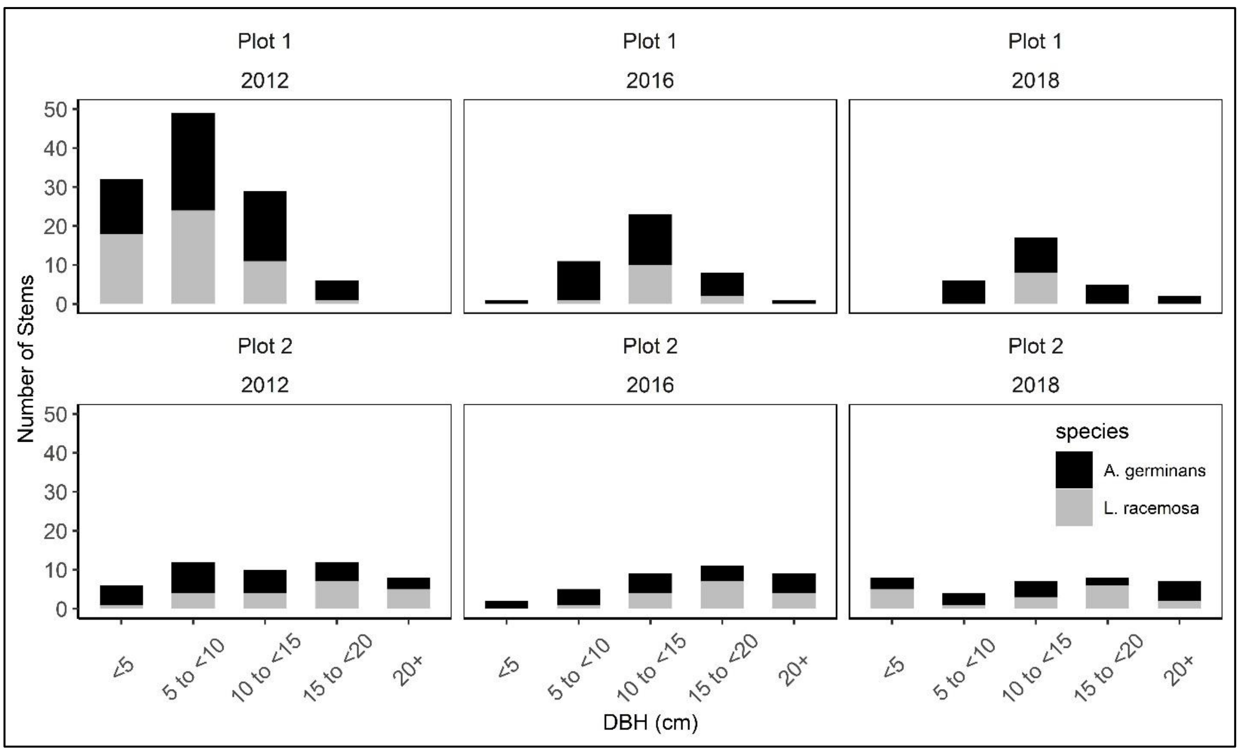


Figure 4.4.1.5. Total number of stems in each 5 cm DBH category (< 5 cm, 5 to < 10 cm, 10 to < 15 cm, 15 to < 20 cm, and 20+ cm) by species in the 10-m radius mangrove permanent plots in 2012, 2016, and 2018. Data from SFCN forest monitoring program (provided by K. Whelan in 2020).

Table 4.4.1.3. Total number of stems of black mangrove, *A. germinans*, and white mangrove, *L. racemosa*, in each plot in 2012, 2016, and 2018 in 5 size classes defined by DBH (cm). Data from SFCN forest monitoring program (K. Whelan 2020).

Plot	Species	Year	< 5	5 to < 10	10 to < 15	15 to < 20	≤ 20	Total Stems
1	*A. germinans*	2012	14	25	18	5	0	62
	A. germinans	2016	1	10	13	6	1	31
	A. germinans	2018	0	6	9	5	2	22
	L. racemosa	2012	18	24	11	1	0	54
	L. racemosa	2016	0	1	10	2	0	13
	L. racemosa	2018	0	0	8	0	0	8
2	*A. germinans*	2012	5	8	6	5	3	27
	A. germinans	2016	2	4	5	4	5	20
	A. germinans	2018	3	3	4	2	5	17
	L. racemosa	2012	1	4	4	7	5	21
	L. racemosa	2016	0	1	4	7	4	16
	L. racemosa	2018	5	1	3	6	2	17

Acrostichum danaefolium, leather fern, was present in Plot 2 and was establishing in the understory of Plot 1. Leather fern has been documented to restrict mangrove seedling recruitment (Roth 1992). During the 2016 census, Plot 2 had more trees in larger size classes. Large diameter black mangrove trees, including several old standing dead trees from Hurricane Hugo are still present (B. Shamblin 2019, personal communication). Between 2016 and 2018 surveys, Hurricanes Irma and Maria impacted St. Croix in close succession in September 2017. The mortality observed in 2018 was clearly attributable to hurricane impacts with many snapped trees or trees with canopy loss (Table 4.4.1.4). Recruitment of new individuals was restricted to Plot 2 with five new white mangrove saplings recorded in the plot in 2018. As of 2019, extensive mangrove recruitment has been observed, with numerous saplings establishing in the mangrove forests in SARI (K. Whelan 2020, personal communication).

Table 4.4.1.4. Mortality of black mangrove, *A. germinans*, and white mangrove, *L. racemosa*, in each plot in 2016 and 2018 as compared to the number of living trees in the prior survey. Data from SFCN forest monitoring program provided by K. Whelan 2020.

Plot	Species	# of Live Stems 2012	# of Live Stems 2016	# of Dead Stems 2016	Percent Mortality 2016	# of Live Stems 2018	# of Dead Stems 2018	Percent Mortality 2018
1	*A. germinans*	62	32	30	48.4	22	10	31.3
	L. racemosa	54	14	40	74.1	8	6	42.9
2	*A. germinans*	27	20	9	33.3	18	3	15.0
	L. racemosa	21	16	6	28.6	17	4	25.0

Threats and Stressors

Threats and stressors to the mangrove forests of SARI are numerous. Continued development and urbanization within the Salt River watershed will lead to reduced infiltration and increased run-off during precipitation events and will in turn facilitate increased sediments, contaminant loads, and excess nutrients being transported to mangrove communities. Legacies from farming activities, including a fish farm which operated in the 1950s, within the lower floodplain of the Salt River could be mobilized with increased run-off and flooding. Water quality in Sugar and Triton Bays, farthest from the bay mouth, is periodically impaired, having levels of turbidity and dissolved oxygen higher than allowable for Class B waters (Kendall et al. 2005). The Mon Bijou flood control project, completed in 2006, was constructed in the upper reaches of the Salt River watershed to address flooding in residential areas. It functions by diverting rainwater through a series of gabion structures before reconnecting with the Salt Run, decreasing water flow rate and transport of sediment through the gut, with potential impacts to Sugar Bay mangroves, especially any loss of terrestrial sediments that would help maintain vertical accretion rates. Both marine and terrestrial sources of waste and debris are an issue for mangrove communities worldwide. Leaking oil from boats that have been abandoned or sit derelict are a direct threat to this ecosystem. Improperly moored boats can damage prop roots of red mangroves (Kendall et al. 2005); this is especially true when SARI has been used as a "hurricane hole" with vessels tied up to mangroves during storms.

Tsunamis, while infrequent, have previously been generated from earthquake activity along faults in the Caribbean basin, leading to substantial saltwater flooding and suspended marine sediment deposition in coastal forests. Records of tsunamis impacting St. Croix include one generated by an earthquake in the Anegada Trough in 1867, causing waves which were recorded traveling 91 meters inland in nearby Christiansted (Lander et al. 2002). Impacts from hurricanes include similar flooding with saline water, but additionally bring strong winds, directly damaging trees and altering forest structure, with subsequent negative impacts to habitat quality as observed after Hurricane Hugo. Predicted increases in the incidence of Category 3–5 hurricanes in the Atlantic (Bender et al. 2010), could result in shorter return intervals between strong hurricanes impacting SARI mangroves, leading to permanent changes in forest structure. The current rate of sea level rise (SLR) as measured in nearby San Juan, PR (1963–2020) is 2.15 mm yr^{-1} (http://www.psmsl.org, Station ID 2118) , which is slightly lower than the global rate (~3 mm yr^{-1}; Nerem et al. 2018). However, projected increases in sea level of 30–130 cm by 2100 (Sweet et al. 2017) are a direct threat to mangrove ecosystems in locations where mangroves fail to keep pace with rising seas through sediment accumulation.

Data Needs and Gaps

We suggest continued monitoring of existing permanent plots, along with the establishment of permanent plots in other ecotypes and locations within mangrove forest to assess future changes in structure and composition and potential degradation. We also suggest monitoring to track any expansion of leather fern in the understory as it results in decreased seedling recruitment post-disturbance. Continued monitoring of the R-SET installed in Sugar Bay mangroves is necessary to assess changes in elevation relative to rising seas. Continued water quality monitoring within the estuary will provide a measure of contaminant load, excess nutrients, and edaphic conditions of the nearshore environment.

Overall Condition

While the mangroves of SARI are recovering from a state of reduced canopy cover following Hurricane Hugo (1989), this component has not yet regained the stature of the pre-hurricane forest. To assess the condition of this component, we used community extent as an indicator and considered two metrics: change in percent cover of mangroves and change in species composition. Based on the spatial analysis and assessment of plot-level data, we conclude that the extent of the mangrove forest has changed substantially over 40 years, and while the overall plant species assemblage has changed little, the relative distribution of the species has changed dramatically as a result of catastrophic damage from Hurricane Hugo in 1989 (Table 4.4.1.5). We consider the condition of the resource to be of moderate concern. Overall, mangrove community extent is increasing in SARI through expansion into the hinterland as well as recolonization of tidal mudflats. However, the mangrove component has not yet regained its pre-Hurricane Hugo extent, attesting to the several decades required for regeneration following hurricane disturbance. Given the scale and extent of aerial image interpretation at multiple time steps we have medium confidence in this part of the assessment. However, we would caution the use of this time series dataset for identifying changes in canopy openness or species composition at specific parcels within SARI, as differences in mapping methodologies and class definitions by NOAA and SFCN limit the utility to more coarse-scale assessments. In terms of species composition, it is clear that species' abundances have changed, but information on stand demographics is limited to that of two plots within basin forest. Additionally, it is unclear whether the percentage of the non-native seaside mahoe has changed over time. We have medium confidence in this part of the assessment. Continued restoration efforts focused on planting seedlings are not recommended at this time given that mangroves can naturally recolonize in locations that are hydrologically suitable and have an abundance of local seed source (Lewis 2005).

Table 4.4.1.5. Graphical summary of status and trends for Mangroves within the framework category Terrestrial Vegetation, including rationale and reference condition.

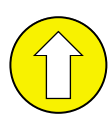
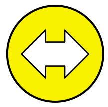
Component	**Indicator**	**Condition Status /Trend**	**Rationale and Reference Conditions**
Mangroves	Community Extent (Change in cover)		Change in cover is moving in a positive direction after extensive mortality resulting from Hurricane Hugo. Continued expansion of mangrove into mudflat and into the terrestrial freshwater marsh is expected, provided another strong hurricane doesn't impact the park and anthropogenic impacts are minimized. The multitude of threats to this community designate the resource as of moderate concern.

Table 4.4.1.5 (continued). Graphical summary of status and trends for Mangroves within the framework category Terrestrial Vegetation, including rationale and reference condition.

Component	Indicator	Condition Status /Trend	Rationale and Reference Conditions
Mangroves (continued)	Community Extent (Change in Species)		While changes in species composition have occurred over the time period, largely related to disturbance from Hurricane Hugo, the overall assemblage of species present has not changed. It is not clear if the mangrove associate, seaside mahoe, *T. populnea*, has increased over time, but presence of non-native invasive species should continue to be monitored. Similarly, increases in the cover of leather fern in the understory should be monitored since the species can suppress recruitment of mangrove seedlings.

<u>Source(s) of Expertise</u>

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4.4.2. Semi-deciduous Dry Forest

This section reviews the condition of semi-deciduous dry forest in SARI. The condition assessment considers data for the years 2008–2017, provided by the South Florida Caribbean Network to assess the status of the semi-deciduous dry forest. The condition of upland forests is typically evaluated using metrics that detect changes in species composition, forest structure, fragmentation and habitat loss, diversity, percent cover of invasive species, and mortality/damage. The condition metrics selected for this resource include change in species composition and change in percent cover. Temporal trends in condition metrics were not evaluated as the data was not available.

<u>Description</u>

Semi-deciduous dry forests in SARI occupy 75 ha, constituting 45% of the terrestrial area of the park (Figure 4.4.2.1) (Moser et al. 2011). This community includes the following Virgin Islands sub-formations according to Gibney et al. (2000): 1) semi-deciduous forest, 2) gallery semi-deciduous forest, and 3) semi-evergreen forest. These forests tend to be found on northwestern facing hillslopes, and in the case of gallery semi-deciduous forests, small riparian corridors (Gibney et al. 2000). Numerous non-native species are found across this forest type and in the most disturbed portions of the forests, several exotic species are the canopy dominants including tan tan, *Leucaena leucocephala*, genip, *Melicoccus bijugatus*, and raintree, *Samanea saman* (Moser et al. 2011). Areas of semi-deciduous forest dominated by raintree are restricted to the southwest corner of the park in the floodplain of the Salt River. Areas of semi-deciduous forest where tan tan is the dominant canopy species occur in the northeast portion of the park (Hemmer’s Peninsula). Fifty acres of 73-NPS owned parcel continues to be were targeted for invasive species control beginning in 2012 and on-going; work has been funded by the Florida and Caribbean Exotic Plant Management Team (FLCEPMT) (NPS 2014) (Figure 4.4.2.1). Additionally, NPS has been restoring native trees throughout the parcel (2012–2019), having planted more than 1200 trees of 20+ native species (Z. Hillis-Starr 2020, personal communication).

Data and Methods

The indicator used to assess the dry forest component is community extent and includes two measures: the change in exotic species cover and species composition. Datasets used for the analysis include the following:

1. a list of dominant and frequently observed species noted in 53 (~300 m2) circular plots visited in 2008/2009 as part of the SARI Vegetation Mapping Project (Moser et al. 2011),

2. treatment efficacy monitoring data from a single 15 m radius plot established in 2009 and revisited in 2011/2012 by the South Florida Caribbean Network (SFCN), and

3. Species composition of five 10 m radius forest plots established by SFCN between 2012 and 2017.

Canopy cover and tree height were summarized from the data collected from 53 accuracy assessment plots visited in 2008/2009 by SFCN (Figure 4.4.0.1). Species listed in these plots as being canopy dominant/co-dominant were used to gain a picture about most important species in this forest type. Additional species listed in the plots were used to assess the distribution of exotic invasive species in semi-deciduous forest.

A treatment efficacy plot, established to monitor effectiveness of invasive exotic species management actions, is located in semi-deciduous forest to the east of Triton Bay (Figure 4.4.2.1). Established in 2009 and expanded and resampled in 2011/2012, locations of all stems present with the 15 m radius plot were mapped, height and diameter at breast height (DBH) measured, and condition/health recorded.

From 2012 to 2017, five (~300 m^2) forest monitoring plots were established in semi-deciduous forest within all three Virgin Islands sub-formations: semi-deciduous forest, semi-evergreen forest, and gallery semi-deciduous forest (Figure 4.4.2.1). These plots were randomly selected within spatial blocks and criteria for establishment stipulated that the plot could not be dominated by exotic tree species (K. Whelan 2020, personal communication). All trees in each plot were identified to species, tagged and mapped, and DBH and height recorded, providing information on the distribution of size structure and species composition across several sub-formations in semi-deciduous forest within SARI. The forest monitoring plot established nearby in the basin moist forest by SFCN (Plot 4) was not considered in this analysis.

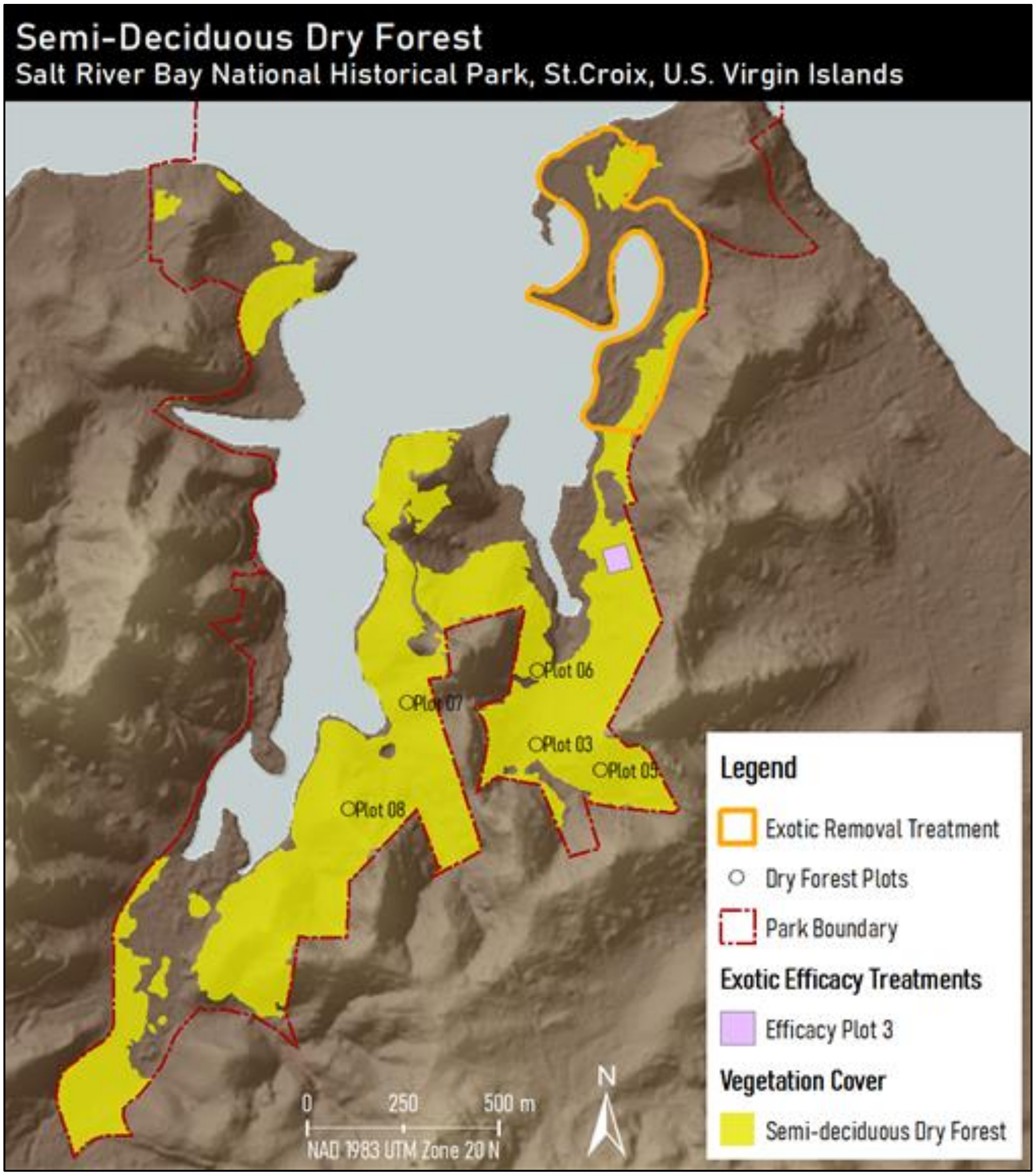

Figure 4.4.2.1. Extent of semi-deciduous dry forest community within Salt River National Historical Park and Ecological Reserve overlain by the locations of the five ~300 m^2 forest monitoring plots and single 15 m radius exotic treatment efficacy plot (Moser et al. 2011; NPS 2013, NPS 2017).

<u>Reference Conditions/Values</u>

Reference conditions for semi-deciduous dry forest date to 2008/2009 surveys conducted by SFCN as part of the accuracy assessment for land cover mapping (Figure 4.4.0.1).

<u>Current Condition and Trend</u>

During the 2008 and 2009 sampling for land cover map accuracy assessment, 53 (300 m^2) circular plots were visited in semi-deciduous forest (Figure 4.4.0.1). Seventy percent of these plots had a non-native invasive species serving as the canopy dominant or co-dominant. Tan tan, *L. leucocephala*, was the most prevalent non-native dominant (40% of plots), followed by genip, *M. bijugatus,* in 28%

of plots (Table 4.4.2.1). In plots where a native tree was listed as the dominant/co-dominant canopy species, pigeon berry, *Bourreria succulenta*, was most often observed (45% of plots), followed by white manjack, *Cordia dentata* (19% of plots). In all but two plots, at least one invasive species was present (in any strata), and in more than two-thirds of plots, multiple invasive species were present. The most widely recorded non-native species included the following: tan-tan (80% of plots), genip (38% of plots), the common understory species, wild lime, *Triphasia trifolia*, (34%), and guinea grass, *U. maxima*, which was found in 23% of plots. Average canopy height was 7.5 m and ranged as high as 16 m. Average canopy cover was 70%.

Table 4.4.2.1. List of dominant or co-dominant species recorded in 53 (300 m^2) (Moser et al. 2011).

Species Name	Common Name	Number of plots	Percentage of plots
Bourreria succulenta	Pigeon berry	24	45 %
Leucaena leucocephala [1]	Tan	21	40 %
Melicoccus bijugatus [1]	Genip	15	28 %
Cordia dentata	White manjack	10	19 %
Samanea saman [1]	Raintree	4	8 %
Capparis cynophallophora	Jamaican caper	3	6 %
Eugenia monticola	White stopper	3	6 %
Acacia tortuosa	Poponax	2	4 %
Coccoloba uvifera	Seagrape	2	4 %
Hippomane mancinella	Manchineel	2	4 %
Pithecellobium unguis-cati	Blackbead	2	4 %
Guapira fragrans	Wild mampoo	1	2 %
Pisonia subcordata	Water mampoo	1	2 %
Terminalia catappa	Indian almond	1	2 %
Thespesia populnea [1]	Seaside mahoe	1	2 %
Triphasia trifolia [1]	Sweet lime	1	2 %

[1] Non-native invasive species, also shown in bold.

A single 15 m radius plot to assess efficacy of treatment for invasive tree species was established in semi-deciduous forest in 2009 (NPS 2013) (Figure 4.4.2.1). All stems having a DBH equal to or greater than 5 cm were tagged, location mapped, and height and DBH recorded, for a total of 208 stems. Of these, only 6% were exotic invasive species (11 *L. leucocephala*, 1 *M. bijugatus*). The plot was revisited in 2011/2012 and all stems re-measured. All stems < 5 cm DBH were tagged and measured, increasing the total number of stems in the plot to 1012. Five percent of the expanded plot was also constituted by invasive exotic species, suggesting there is no difference in abundance of invasive species by size class (48 *L. leucocephala*, 6 *M. bijugatus*, 1 *Tecoma stans*). After the initial survey in 2009, treatment for non-native woody species was undertaken in the vicinity of the plot (Moser et al. 2011). Thirty-three of the stems within the plot were treated with herbicide (32 *L. leucocephala*, 1 *M. bijugatus*) and as of 2011/2012 visits, 100% of the tan tan were recorded as dead and the genip had died back. The plot is dominated by the native species *Guapira fragrans*, wild mampoo, and *B. succulenta*, pigeon berry. The results of this pilot study revealed that treatment of tan tan with herbicide using the cut stump method can be highly effective.

Forest Monitoring Plots

Across the five dry forest monitoring plots, a total of 35 species were recorded with species richness in each plot ranging from 13 to 20 species (Table 4.4.2.2). Stem density ranged from a low of 0.81 stems m^{-2} in Plot 8 to a high of 2.5 stems m^{-2} in Plot 7. A comparison of the number of stems in 5 DBH classes (< 2 cm, 2 to < 5 cm, 5 to < 10 cm, 10 to < 15 cm, 15+ cm) across the five plots is presented in Figure 4.4.2.2. Across all plots, seven species accounted for the majority of stems, each constituting at least 5% of the stems in any plot where present. These seven species are specified in Figure 4.4.2.2 while all other woody hardwood species are designated as "other" for ease of graphical interpretation. This young forest is dominated by stems falling in the two smallest DBH categories (< 5 cm) (Figure 4.4.2.2). The most prevalent species are *E. monticola* and *B. succulenta*, except in gallery semi-deciduous forest (Plot 3), which is instead dominated by two non-native invasive species: genip, *M. bijugatus* and sweet lime, *T. triphasia*. Yet Plot 3 also had the greatest species richness of all sampled plots. While exotic invasive species are present in all plots, with the exception of the aforementioned Plot 3 (51% non-native), abundance of invasive exotics was low (< 2% to 21%). However, given the criteria for plot selection by SFCN of avoiding locations dominated by non-native species, this finding is not unexpected. Tan tan and sweet lime were found in all five plots, whereas genip was restricted to Plots 3 and 6. The largest individuals in the plots (DBH > 15 cm) included the following species: *Acacia macrantha*, long-spine acacia, *B. succulenta*, *C. dentata*, *E. monticola*, *G. fragrans*, and *Swietenia mahagoni*, mahogany, a non-native timber species planted during reforestation.

Table 4.4.2.2. Compositional and structural attributes of 5 forest monitoring plots located in semi-deciduous dry forest (SDF) in SARI (NPS 2017).

Plot	Date Surveyed	Forest type within SDF	Canopy Cover (%)	Species Richness	Stem Count
3	1/27/2012	Gallery semi-deciduous	85.5	20	437
5	6/6/2013	Semi-deciduous	87.75	14	261
6	12/13/2013	Semi-deciduous	84	18	418
7	4/24/2014	Semi-evergreen	75.25	17	784
8	6/28/2017	Semi-deciduous	NA	13	255

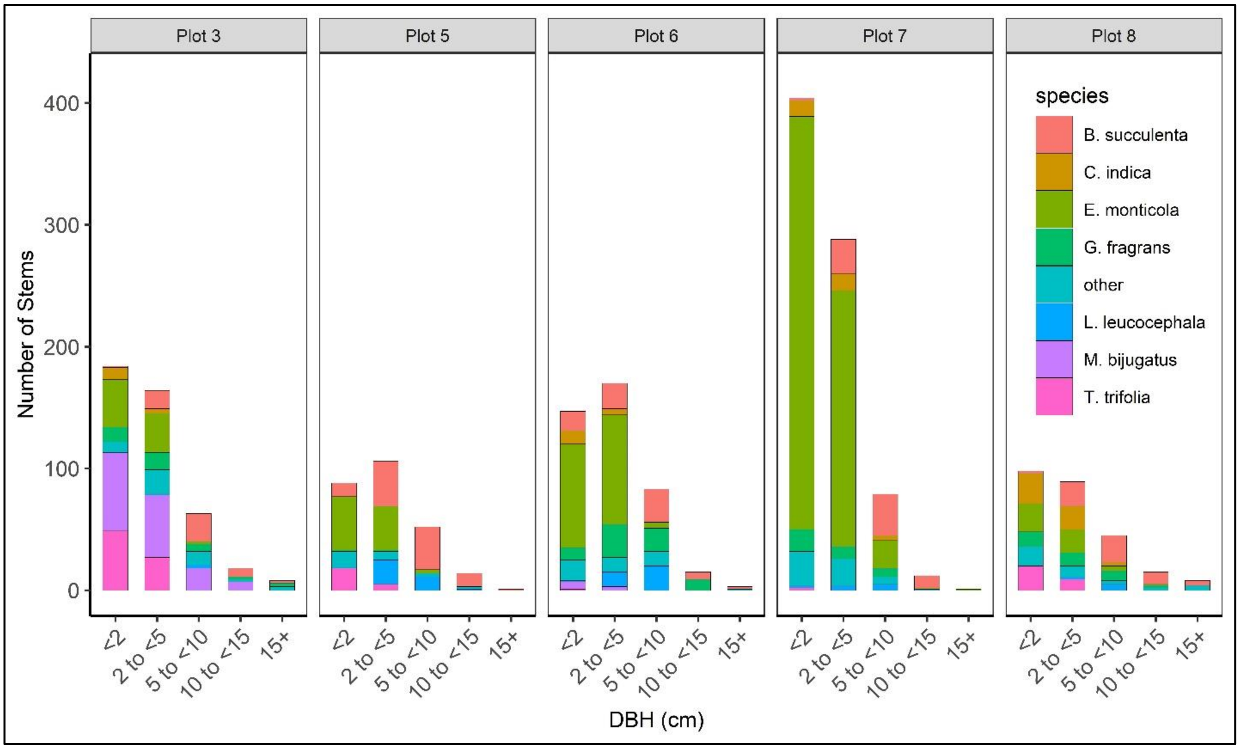

Figure 4.4.2.2. Total number of stems in each DBH category (< 2 cm, 2 to < 5 cm, 5 to < 10 cm, 15 to < 15 cm, and 15 + cm) by species in the 10-m radius semi-deciduous forest monitoring plots (2012–2017). Data provided by SFCN (K. Whelan).

<u>Threats and Stressors</u>

Threats and stressors to the semi-deciduous forests in SARI are numerous. Continued development and urbanization within and outside the boundary of the park lead to increased run-off and erosion, exacerbated by extreme weather events. Legacies from prior land-use have resulted in the component having a wide-spread distribution of invasive exotic plants that will likely increase without management action. Disturbance from hurricanes and anthropogenic land use changes will only exacerbate the pressure of invasive exotics. Grazing and rutting by non-native mammals, both inside and adjacent to SARI cause damage to vegetation and soil. The landward expansion of mangroves with rising seas in the past 40 years has likely come at the expense of low-lying semi-deciduous forest. This trend will most likely continue in the coming decades as accelerated rates of sea level rise (SLR) are predicted (Sweet et al. 2017).

<u>Data Needs and Gaps</u>

Continued monitoring of the component within existing permanent plots would permit an understanding of changes to composition and structure as this forest matures. Expansion of plot monitoring into areas of the forest that are also dominated by non-native species is suggested to understand impacts of invasive species to the dry forest component in SARI. Expansion and follow-up monitoring of treatment efficacy of invasive exotic species is necessary to understand long-term effectiveness of management interventions. A comprehensive survey of all the species occurring within the component is currently lacking. However, species lists were available from prior projects focused in specific areas of SARI and have been combined into a single list (Appendix A). Additional focused surveys within the dry forest may lead to detection of less common and locally threatened species like lignum vitae, *Guacium officinale*, stingingbush, *Malpighia infestissima*, and cow-itch, *Malpighia woodburyana*.

<u>Overall Condition</u>

Semi-deciduous dry forest in SARI is dominated by small trees, indicative of a secondary forest recovering from a long history of disturbance. Non-native invasive plant species are widely distributed throughout the forest and would be considered dominant or co-dominant across broad swaths of the park. However, areas of the dry forest in SARI dominated by native woody species occur as well, as evidenced by composition of the forest monitoring plots. To assess the condition of dry forest component, community extent was selected as the indicator, with two metrics: change in percent cover of invasive exotic species and change in species composition. Given that the plots producing the two datasets (accuracy assessment compared to the forest monitoring plots) did not overlap spatially and were sampled across a relatively short time span (2008/2009 compared to 2012–2017), no attempt to infer a trend in condition has been made. Instead, we consider distribution of non-native invasive species for the cover metric and species richness and dominance for the species composition metric. Species richness ranged from 13 to 20 species. Based on the wide distribution of invasive exotic species but considering the dominance of native tree species throughout much of the forest, we consider the condition of the resource to be of moderate concern (Table 4.4.2.3). However, criteria for establishment of the permanent plots included the stipulation that the location not be dominated by exotic species; the result of this decision likely biases the forest monitoring dataset toward increased coverage by native species. We have medium confidence in our assessment. While the results of efficacy of treatment for *L. leucaena* are very encouraging, the current extent of treatment for non-native species within the component appears to be restricted to a small area falling within the northeast section of the park. Current success of native species reforestation efforts (60% survival) in the northeast parcel is encouraging and continued monitoring of survival of planted individuals is recommended. Details regarding the reforestation efforts are provided in Section 4.4.3 *Coastal Grasslands.*

Table 4.4.2.3. Graphical summary of status and trends for semi-deciduous dry forest within the framework category Terrestrial Plants, including rationale and reference condition.

Component	Indicator	Condition Status /Trend	Rationale and Reference Conditions
Semi-deciduous Dry Forest	Community Extent (Change in cover)		Non-native invasive species are widespread throughout the component. It was not possible to assess any change in the indicator from the available data.
	Community Extent (Change in Species)		Composition of forests include both native and non-native invasive species and the dominance of either varies throughout the component. It was not possible to assess any change in the indicator from the available data.

Source(s) of Expertise

- Brooke Shamblin, NPS South Florida Caribbean Network (SFCN), Palmetto Bay, FL
- Kevin Whelan, PhD, NPS So Fl Caribbean I&M Network, SFCN, Palmetto, Bay, FL
- Florida and Caribbean Exotic Plant Management Team (FLCEPMT), Palmetto Bay, FL
- Zandy Hillis-Starr, NPS, Resource Manager BUIS (retired)

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4.4.3. Coastal Grasslands

This section reviews the condition of coastal grassland in SARI. The condition assessment considers data for the years 2009–2019, provided the South Florida Caribbean Network (2009), the NPS Florida/Caribbean Exotic Plant Management Team (2012 and 2014), and Environmental Quality Inc. (2012–2019) to assess the status of the coastal grassland. The condition of grasslands is typically evaluated using metrics that detect changes in species composition, fragmentation and habitat loss, diversity, and percent cover of invasive species. The condition metrics selected for this resource include change in species composition and change in percent cover of invasive exotic species. Temporal trends in condition metrics were evaluated.

Description

Coastal grassland encompasses 16.7 ha along the northeast and northwest points of Salt River Bay. It is a community type that is dominated by hardy grasses but also includes habitat with greater than

25% woody species, referred to specifically as mixed dry grassland (Gibney et al. 2000). Invasive exotic species, particularly *Urochloa maxima*, guinea grass, and *Cryptostegia madagascariensis,* rubber vine, are prevalent throughout the community type. In the northeastern portion of SARI, a failed coastal development (1960s–1970s) resulted in landscape-level disturbance that facilitated the incursion of invasive plant species after site abandonment. Management of coastal grassland in the northeastern portion of the park was initiated to address the extensive exotic plant problem. Rubber vine was mechanically removed by mowing, after which herbicide treatments were applied as follow up. Guinea grass was treated with foliar application, whereas woody vegetation was treated with a stump cut and herbicide method (NPS 2014). Restoration of mixed dry grassland and dry forest with the planting of native hardwood species began in 2012 (NPS 2013) and is ongoing on 50 acres of NPS owned 73-acre parcel at Hemmer's Peninsula (Z. Hillis-Starr 2020 personal communication).

<u>Data and Methods</u>

The indicator used to assess the coastal grassland component is community extent and includes two measures: the change in exotic species cover and species composition. Datasets used for the analysis include the following:

1. a list of dominant or co-dominant species noted in 13 (300 m^2) circular plots visited in 2009 as part of the SARI Vegetation Mapping Project (Moser et al. 2011),

2. locations of rubber vine clearing, number and species of native tree species planted in 2012, and list of 90 trees with condition information as of 2014 obtained from the NPS Florida/Caribbean Exotic Plant Management Team (FLCEPMT),

3. the area of treated individuals of invasive exotic species in 2012 and 2014 as reported by FLCEPMT, and

4. the number of planted trees as part of reforestation efforts in 50 acres on Hemmer's Peninsula in 2012, 2016, 2018, and 2019 and survival as recorded in 2019, as reported by Environmental Quality, Inc.

In this section, we qualitatively compared the canopy area and composition of the invasive exotic species management in coastal grassland to the distribution of species as recorded from accuracy assessment associated with land cover mapping (Moser et al. 2011) (Figure 4.4.0.1). Additionally, we present the total number of species planted in each year and survivorship data for the 2012, 2016, 2018, and 2019 planting events (Figures 4.4.3.1 and 4.4.3.2). For the year 2012, the number of trees planted for each species was arrived at by including all species recorded as being planted in both the FLCEMPT and EQI data sources. Both datasets appear to be incomplete since they are not quite in agreement on the number of individuals nor species planted. For any discrepancies in number of individuals planted by species we defer to the FLCEMPT database since it was created following the initial planting event. However, in cases where the number of surviving individuals as reported in 2019 by EQI was greater than the number reported as planted by FLCEMPT, we defer to the EQI dataset as those individuals would have been visited in the field as of 2019. Planted trees not recorded as of 2019 survey by EQI were assumed to not have survived.

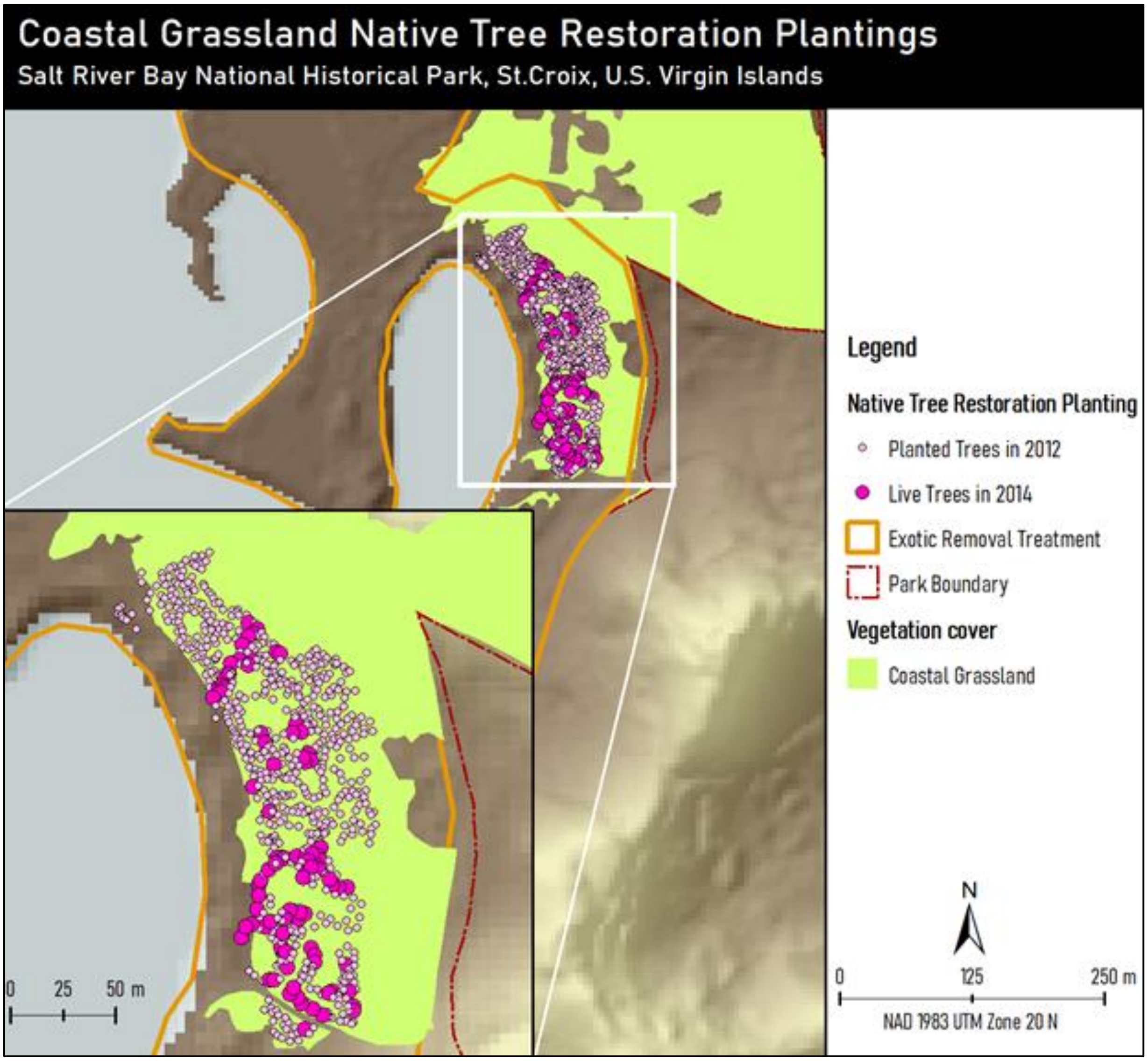


Figure 4.4.3.1. Extent of coastal grassland community in SARI compared to the area of exotic treatment and native tree reforestation. Distribution of native trees planted in 2012 (pink) with individuals revisited during the 2014 condition assessment highlighted in bright pink. Data from FLCEPMT.

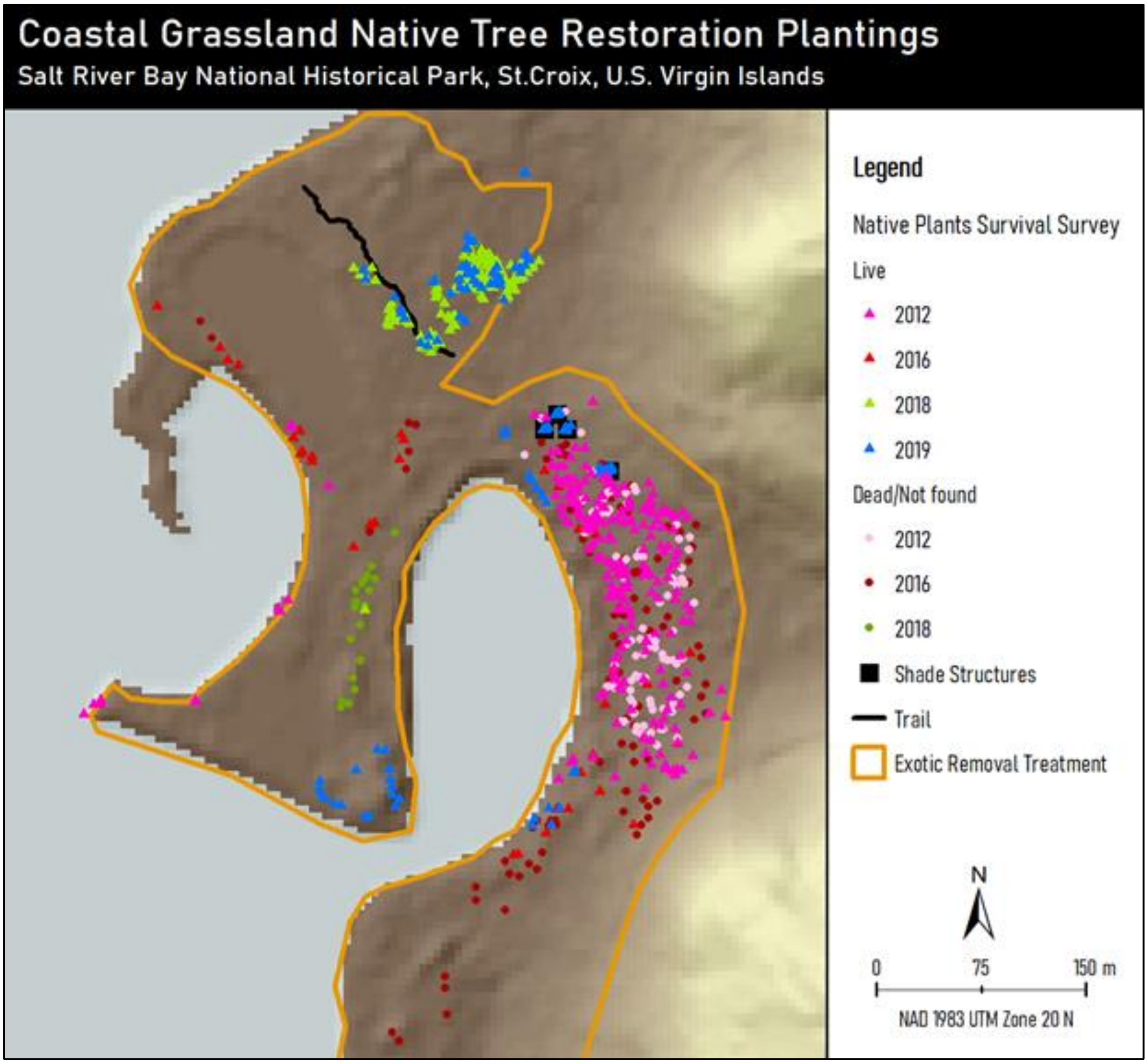


Figure 4.4.3.2. Location of planted native trees and shrubs as part of the reforestation effort on Hemmer's Peninsula, classified by year of planting (2012, 2016, 2018, or 2019) and survival (live or dead as of 2019). Locations of shade structures erected in 2019 and trail also indicated (EQI 2019).

<u>Reference Conditions/Values</u>

For the reference condition, we use the dominant and common species recorded from 13 (300 m^2) plots associated with land cover mapping for SARI conducted in 2009 (Moser et al. 2011).

<u>Current Condition and Trend</u>

Invasive exotic species were listed as the dominant or co-dominant species recorded in 2009 from 100% of the accuracy assessment plots falling within coastal grassland (Figure 4.4.0.1). Guinea grass, *U. maxima*, was the most abundant species recorded in 70% of the plots. Rubber vine, *C. madagascariensis*, Spanish bayonet, *Yucca aloifolia*, and tan tan, *Leucaena leucocephala*, were also present in large abundance in several plots. Native species listed as dominant or co-dominant included the tree species, poponax, *Acacia tortuosa*, which was present to some extent in 60% of plots, the shrub buttonsage, *Lantana involucrata*, and the climbing shrub, yellow nicker, *Caesalipina ciliata*, which was the dominant species in a single plot.

In 2012, 19 acres, just under half of the infested area at the Old Hotel site on the northeastern side of the bay, was treated for six invasive exotic species, including those mentioned above (NPS 2012). Treatment for invasive exotic species continued in 2014, both in coastal grassland and nearby dry tropical forest and shrublands on an additional 54 acres (Figure 4.4.3.1) (NPS 2014). Rubber vine, which was first mechanically mowed, was subsequently treated with herbicide, while guinea grass received foliar treatments, and woody vegetation was cut and herbicide applied (NPS 2014). Invasive species management within the area continues to the present, with guinea grass and tan tan treated as recently as 2019 (EQI 2019). No management for invasive species in grassland is currently planned or being conducted in the western grassland (Z. Hillis-Starr 2020, personal communication).

Native tree restoration of the parcel has been ongoing since 2012, with more than 1,000 trees of 30+ species having been planted from 2012–2019 (Table 4.4.3.1). During the first planting event (July 2012), approximately 800 trees, including over 20 species, were planted (Figure 4.4.3.1). Three species constituted more than 50% of the planted individuals: gri gri, *Bucida buceras*, black mampoo, *Guapira fragrans*, and pink manjack, *Tabebuia heterophylla*. In 2014, the condition of 90 of these individuals was categorized as being in poor, fair, average, good, or excellent condition, with most individuals recorded as being in poor to fair condition also having evidence of leaf die-back, over-topping or browse. Of the 90 trees evaluated, the majority (64%) were described as being in either excellent or good condition (Table 4.4.3.2). However, guinea grass had grown up extensively over the time period and most surviving trees and shrubs were buried in 1.5 high grasses (NPS 2020).

Table 4.4.3.1. Number of trees planted (PL) by species in each year (2012, 2016, 2018, and 2019) within the treatment area on Hemmer's Peninsula. Determination of tree survival (SU) made in 2019 for the 3 prior planting events. Total living = survived (SU) for each year and those planted in 2019. Data from FLCEMPT and EQI (2019).

	2012		2016		2018		2019	Total	
Species	**PL**	**SU**	**PL**	**SU**	**PL**	**SU**	**PL**	**PL**	**Living**
Acacia tortuosa	25	2	–	–	–	–	–	25	2
Agave eggersiana	7	4	–	–	–	–	4	11	8
Bourreria succulenta	48	0	1	0	3	3	–	52	3
Bucida buceras	187	96	–	–	–	–	–	187	96
Bursera simaruba	16	2	3	0	29	3	2	50	7
Capparis cynophallophora	–	–	–	–	5	5	–	5	5
Capparis flexuosa	11	11	–	–	–	–	–	11	11
Capparis indica	49	1	–	–	–	–	–	49	1
Citharexylum fruticosum	17	9	–	–	32	29	–	49	38
Coccoloba sp.	–	–	–	–	–	–	1	1	1
Coccoloba diversifolia	2	1	–	–	–	–	–	2	1

Table 4.4.3.1 (continued). Number of trees planted (PL) by species in each year (2012, 2016, 2018, and 2019) within the treatment area on Hemmer's Peninsula. Determination of tree survival (SU) made in 2019 for the 3 prior planting events. Total living = survived (SU) for each year and those planted in 2019. Data from FLCEMPT and EQI (2019).

	2012		2016		2018		2019	Total	
Species	**PL**	**SU**	**PL**	**SU**	**PL**	**SU**	**PL**	**PL**	**Living**
Coccoloba uvifera	26	25	22	19	–	–	17	65	61
Coccothrinax alta	1	0	–	–	–	–	–	1	0
Colubrina arborescens	1	1	–	–	15	10	–	16	11
Colubrina elliptica	3	3	–	–	–	–	–	3	3
Conocarpus erectus	18	3	–	–	–	–	15	33	18
Cordia dentata	22	20	–	–	–	–	–	22	20
Cordia ricksseckeri	10	2	–	–	–	–	–	10	2
Crescentia cujete	–	–	–	–	30	26	–	30	26
Crossopetalum rhacoma	1	0	–	–	–	–	–	1	0
Delonix regia	–	–	–	–	–	–	7	7	7
Erythroxylum brevipes	17	0	–	–	–	–	–	17	0
Eugenia spp.	41	0	–	–	–	–	–	41	0
Guaiacum officinale	13	2	6	1	5	2	–	24	5
Guapira fragrans	161	0	–	–	3	3	5	169	8
Jacquinea arborea	6	2	–	–	–	–	20	26	22
Lantana involucrate	8	0	–	–	–	–	2	10	2
Maytenus laevigata	–	–	22	1	–	–	–	22	1
Piscidia carthagenensis	–	–	4	0	–	–	–	4	0
Pisonia subcordata	17	2	50	8	–	–	7	74	17
Pithecellobium unguis-cati	12	10	–	–	–	–	–	12	10
Plumeria alba	1	0	–	–	–	–	–	1	0
Randia aculeata	1	0	–	–	–	–	–	1	0
Samyda dodecandra	–	–	–	–	–	–	4	4	4
Solanum conocarpum	–	–	1	0	16	14	15	32	29
Spondias mombin	1	1	–	–	–	–	–	1	1
Swietenia mahagoni	1	1	–	–	–	–	–	1	1
Tabebuia heterophylla	95	21	19	5	32	20	24	170	70
Total	818	219	128	34	170	115	123	1239	491

Table 4.4.3.2. Condition as of 2014 of 90 trees planted during July 2012 restoration. Data from FLCEMPT.

Species	Excellent	Good	Fair/Average	Poor
Bourreria succulenta	2	2	–	–
Bucida buceras	10	11	5	2
Capparis indica	–	–	2	1
Citharexylum fruticosum	2	1	2	–
Coccoloba diversifolia	–	–	1	–
Coccoloba uvifera	1	2	1	–
Cordia rickseckeri	2	1	1	–
Erythroxylum brevipes	–	–	–	1
Eugenia spp.	–	1	1	–
Guaiacum officinale	4	–	–	–
Guapira fragrans	–	8	4	6
Pisonia subcordata	–	2	1	–
Tabebuia heterophylla	2	7	3	1
Total	23	35	21	11

Trees evaluated included 13 species, with the three aforementioned species being the majority of stems evaluated (65%). Notably, the majority of gri and pink manjack were recorded as being in excellent or good condition. Black mampoo still had trees recorded as being in good condition in 2014, but by the 2019 comprehensive survey none of the trees planted in 2012 had survived (Table 4.4.3.1). Data on survival and mortality was not obtained and it is unknown the extent of overall tree mortality at the time of the survey. We caution using the 2014 condition results make conclusions about the 2012 planting at 2 years given that the survey was partial and information regarding representativeness of trees evaluated was not available.

For trees planted in 2012 and 2016, the 7-year and 3-year survival was approximately 27% for both cohorts, with over 6 times as many individuals planted in 2012. The planting events in 2018 and 2019 included a similar number of newly planted individuals. For trees planted in 2018, nearly 68% of individuals were recorded as living as of 2019. For the 2012 and 2016 cohorts, many species had few to no surviving individuals by 2019 (e.g., *Bourreria succulenta, Guapira fragrans, Lantana involucrate*), indicating either they were ill-suited for planting at the site or perhaps the climate conditions were too harsh for successful establishment. Severe drought conditions in the USVI during the summer of 2015 would have been a stressful event that may have contributed to mortality (NRCS 2015). Several species had more than 50% survivability during the project, including *B. buceras*, *Citharexylem fruticosum*, *Coccoloba uvifera*, *Cordia dentata*, and *Crescentia cujete*. Gri Gri has done particularly well, with some trees at tall as 9 m and natural recruitment of seedlings (Z. Hillis-Starr 2020, personal communication). Trees planted in the northern portion of the restoration area, which includes most of the individuals planted in 2018 and 2019, receive water from an irrigation system installed in 2019 (Figure 4.4.3.2) (EQI 2019). Four 40% shade cloths were also installed in 2019, south of the irrigated area, with native tree species planted under or nearby in hopes that decreasing the amount of direct sun will assist these individuals in getting established (EQI 2019).

Threats and Stressors

The threats to coastal grasslands in SARI are numerous. Continued development and urbanization within and outside the boundary of the park lead to increased run-off and erosion, exacerbated by extreme weather events and wildland fires during periods of extreme drought. Legacies from prior land-use, including the dumping of dredge materials from the bay and centuries of agricultural activity, have had numerous negative impacts on the component, most notably the dominance of invasive exotic plants in the grassland. Management of the component, including treatment with herbicides, mechanical removal of invasive species, and restoration with native trees is encouraging, but will require continued maintenance. Grazing and rutting by non-native mammals continue to disturb vegetation and soil, and help exotics re-establish. Illegal dumping of landscape cuttings onto NPS lands by private landowners also continues to spread non-native invasive plant species within the component (Z. Hillis-Starr 2020, personal communication). Soils within the area, especially just east of the Dredged Basin where the majority of restoration planting has taken place, are highly saline and poor as a result of dredge material being dumped in that location (Z. Hillis-Starr 2020, personal communication). The soil conditions in this area likely impact which planted species will do best in here. Disturbance from hurricanes and anthropogenic land use changes will only increase the pressure of invasive exotics species. Predicted increases in the frequency of Category 3–5 hurricanes in the Atlantic (Bender et al. 2010), could result in shorter return intervals between strong hurricanes impacting the SARI. Hurricanes and infrequent tsunamis can both result in flooding of the low-lying coastal zone, covering extensive areas with saline waters and marine sediments. Predicted rates of sea level rise by 2100 (Sweet et al. 2017) will put low elevation coastal grasslands at the risk of inundation by saline water. Near-shore grasslands, although primarily exotic grass species, are vulnerable to impacts from accidental coastal oil spills.

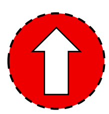
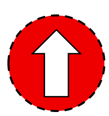
Data Needs and Gaps

Follow-up monitoring of treatment efficacy of invasive exotic species is necessary to understand long-term effectiveness of management interventions. Similarly, monitoring of the survival and growth of trees planted as part of the mixed dry grassland restoration 2012–2019 should be maintained. Information on which tree species were best suited for the location and the prevailing environmental conditions during and after planting would help inform any future planting events in this portion of SARI. Locating an undisturbed coastal grassland in the St. Croix could be useful for generating a species list of potential species present in non-invaded sites (B. Shamblin 2020, personal communication). A comprehensive survey of all the species occurring within the component is lacking although the combined surveys from distinct projects provide a decent species list (see Appendix A). This would provide data on the occurrence and distribution of any locally threatened and endangered species that may occur within the coastal grassland.

Overall Condition

The condition is likely improving as a result of management for non-native invasive species but the component is still considered to be of significant concern as a result of the widespread nature of the invasive plants across the resource (Table 4.4.3.3). To assess the condition of the coastal grassland, we used community extent as an indicator and considered two metrics: the change in exotic species cover and species composition. We qualitatively compared distribution of invasive plant species in ~300 m^2 accuracy assessment plots visited in 2009 to area of exotic management in 2012 and 2014. The treatment of invasive exotic species in the northeastern part of the park is encouraging and likely has resulted in a decrease in the cover of invasive plant species and perhaps an increase in the abundance of native species. However, we have low confidence in the assessment given the limited information available on the initial extent of invasive species, combined with sparse data on exact locations of treatment, and a lack of information regarding efficacy of treatment. Efforts to restore dry mixed grassland / dry forest in SARI are encouraging given the increased percentage of surviving species over the course of the restoration effort and the observations that natural seeding in of native species is occurring and shade provided by larger trees is helping to suppress regrowth of invasive non-native grasses (Z. Hillis-Starr 2020, personal communication). Continued management of invasive species in this component will be necessary as invasive plants from surrounding parcels colonize.

Table 4.4.3.3. Graphical summary of status and trends for coastal grassland within the framework category Terrestrial Plants, including rationale and reference condition.

Component	Indicator	Condition Status /Trend	Rationale and Reference Conditions
Coastal Grasslands	Community Extent (Change in cover)		The percent cover of non-native invasive species in this component has likely declined since management began in 2012. Although not quantified, the relative cover by native species has likely increased as a result. However, continued management will be necessary to maintain lower cover of invasive species.
	Community Extent (Change in composition)		Composition of the component was dominated by non-native invasive species prior to the start of management action and the introduction of native trees into the mixed dry grassland has increased the richness of native species. However, continued management and potentially future reforestation events will be needed to maintain the increase in species richness.

Source(s) of Expertise

- Brooke Shamblin, NPS South Florida Caribbean Network (SFCN), Palmetto Bay, FL
- Kevin Whelan, PhD, NPS So Fl Caribbean I&M Network, SFCN, Palmetto, Bay, FL
- Zandy Hillis-Starr, NPS Resource Manager BUIS (retired)

- Florida and Caribbean Exotic Plant Management Team (FLCEPMT), Palmetto Bay, FL

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4.5. Marine Plants

Marine plants include seagrass and macroalgae and are significant primary producers in coastal marine systems (Duarte and Cebrian 1996) including back reefs and lagoons of the Virgin Islands (Williams 1990). *Halophila stipulacea,* an invasive exotic seagrass, was first observed in Grenada in 2002 and has spread throughout the Caribbean including the Virgin Islands since then (Willette et al. 2014; Ruiz et al. 2017). Seagrasses and macroalgae populate Salt River Bay and the floor of Salt River Canyon. Total algae cover in the nearshore hard-bottom areas of SARI is high and may be considered an unhealthy condition, although algal composition is dominated by turf algae with less known deleterious effects on coral reef health. The condition metric addressed in this report is percent cover.

4.5.1 Macroalgae

This section reviews the condition of macroalgae focal resource in SARI. The condition assessment considers 4 years (Clark et al. 2015, NCRMP) of data to assess the status of macroalgae natural resources. The status of the macroalgae resource is evaluated using metrics that detect change in abundance. The condition metrics selected for this resource is percent cover.

Description

Macroalgae are an important component of both soft-bottom and hard-bottom marine communities providing habitat (Wilson et al. 1990) and food web support (Simenstad & Wissmar 1985). Recent attention of macroalgae on coral reefs have focused on space competition between algae and corals and increased relative abundance of algae on Caribbean reefs since the 1970s (Bruno et al. 2014). High macroalgae abundance is perceived as a negative attribute because macroalgae often replace seagrasses and corals which are more greatly valued in these ecosystems (Valiela et al. 1997; Bruno et al. 2014). Macroalgae abundance is determined by interplay of bottom-up and top-down factors, and when provided with ample light and nutrients, and low grazing pressure, inherent faster growth rates permit algae to outcompete corals and seagrasses (Valiela et al. 1997).

Caribbean reefs, including those within SARI and the rest of the Virgin Islands, are experiencing an aggressive spread of peyssonellid algal crusts (PAC) that are overgrowing corals and sponges (Eckrich and Engel 2013; Edmunds et al. 2019). Genetic analyses revealed the PAC in St. John to be *Ramicrustra textilis* (Wilson et al. 2020), a taxon recently described from Jamaica. It is still undetermined as to whether *Ramicrusta* is a recent invasive genus or if it is a natural component previously undetected but undergoing rapid spread (Edmunds et al. 2019).

Data and Methods

This report summarizes macroalgae observations and cover estimates produced during coral reef monitoring surveys conducted according to the National Coral Reef Monitoring Plan (NCRMP) of the NOAA Coral Reef Conservation Program (Clark et al. 2015). Benthic composition was surveyed by diver observation during 2015, 2017, and 2019 during the month of June at 12 to 15 hardbottom locations using line point-intercept (LPI) surveys. The transect sites differed between years. Macroalgae percent cover was estimated as the number of algal occurrence observations at 100 points spaced at 20 cm intervals along a 20 m linear transect. Each sample point was identified to predetermined major functional categories, 10–11 of which were algal categories. The occurrence and cover of the following algal categories were recorded: *Cyanophyta* spp., *Dictyota* spp., *Halimeda* spp., *Lobophora* spp., other calcareous macroalgae, other “fleshy” macroalgae, *Peyssonnellia* spp., *Ramicrusta* spp., *Crustose* spp., Rhodophyta (other), turf algae with sediment, turf algae free of sediment. For this report, only total algae percent cover and percent cover of *Ramicrusta* sp., a putative invasive exotic alga (Eckrich and Engel 2013) are discussed.

Reference Condition/Values

Previous published evaluations of the density of macroalgae resources within SARI either do not exist or were not located. As such, a historical reference condition for the abundance of macroalgae within SARI is not identified, but the percent cover estimates summarized below may serve as a future reference condition. Børgesen (1913–20) described the marine algal flora of the then Danish West Indies and those early works serve as an excellent reference condition for species composition within the region. Unfortunately, a detailed floristic survey of the algae at the species level was not produced for the NCRMP surveys presently analyzed, precluding a centennial floristic comparison.

Current Condition and Trend

Mean total algae percent cover observed during transect surveys in 2015, 2017, and 2019 (Figures 4.5.1.1–4.5.1.3) were similar at 82%, 74%, and 73%, respectively. Turf algae with sediment was the most frequently observed algal category during all years and was approximately 3X more abundant than any other algal category, comprising 40–52% of all observations during each year. *Ramicrusta* spp. was not observed until 2019 (Figure 4.5.1.4), when it was observed on 6 of 15 transects. Mean percent cover of *Ramicrusta* spp. from all transects surveyed in 2019 was 1.3%, and ranged from 0–5%.

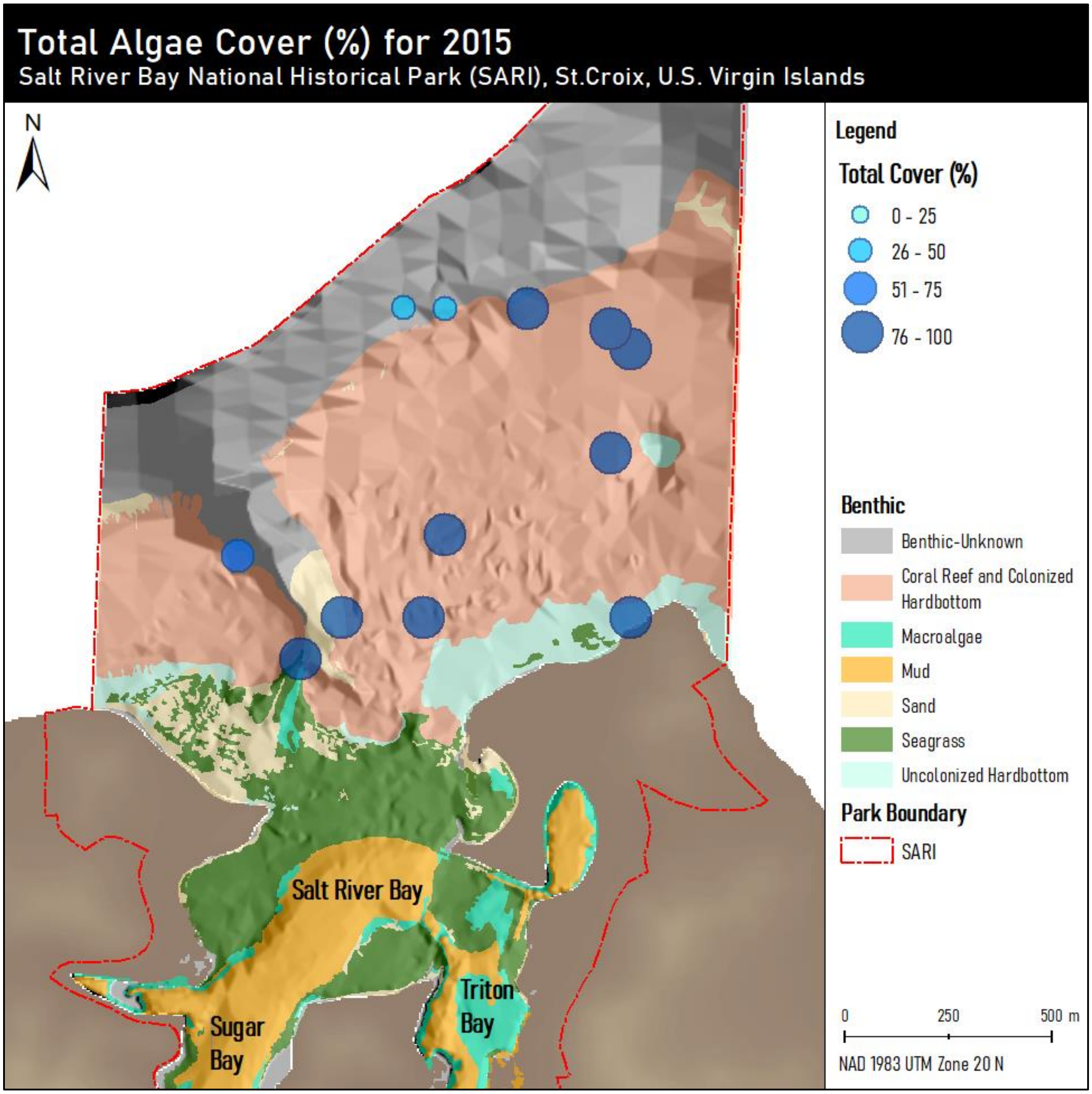


Figure 4.5.1.1. Total algae percent cover, Salt River Bay, 2015 (NCCOS-SEFSC 2021). Benthic habitat (Kendall et al. 2005).

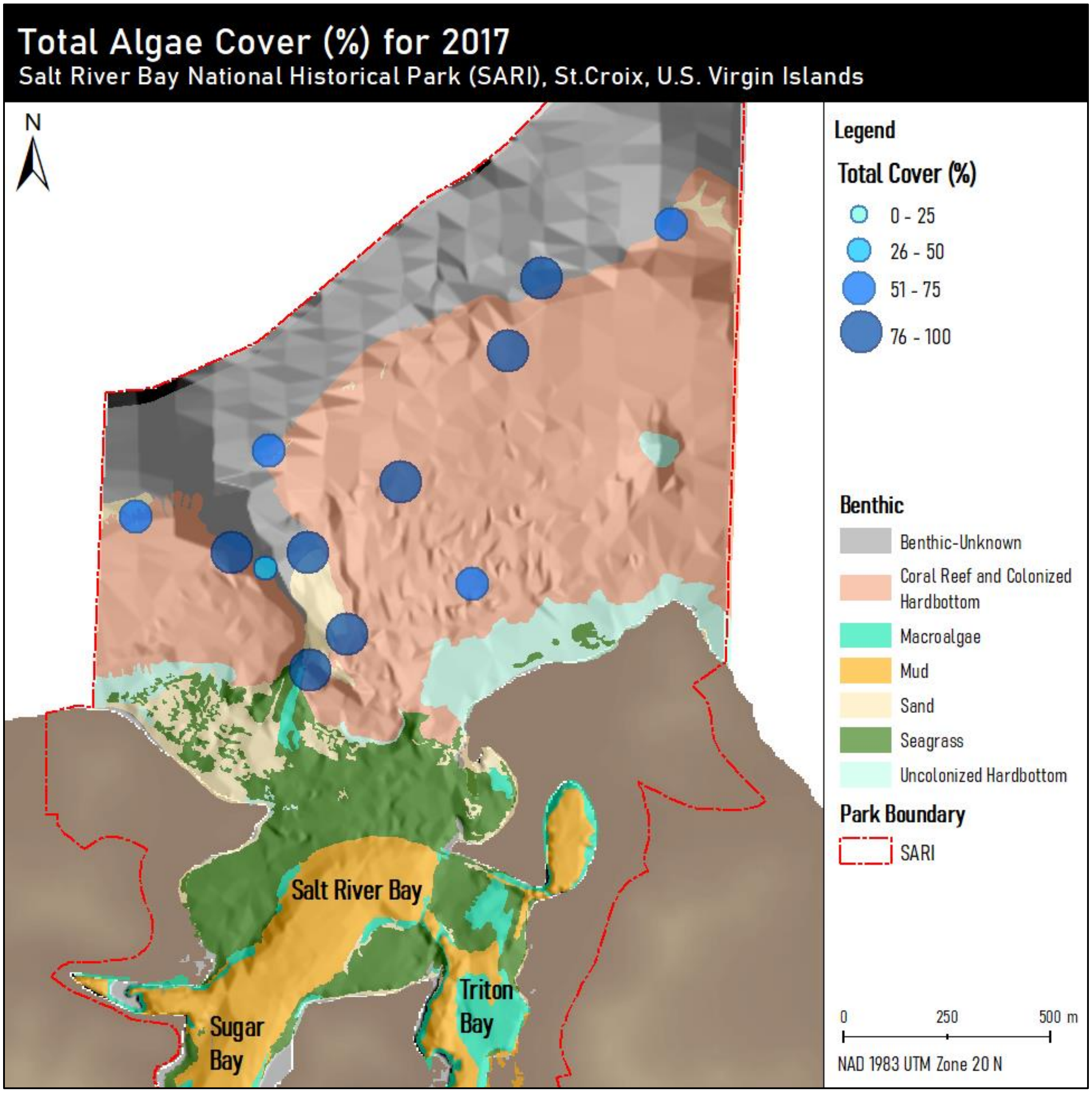


Figure 4.5.1.2. Total algae percent cover, Salt River Bay, 2017 (source – Lee Richter, NPS). Benthic habitat (Kendall et al. 2005).

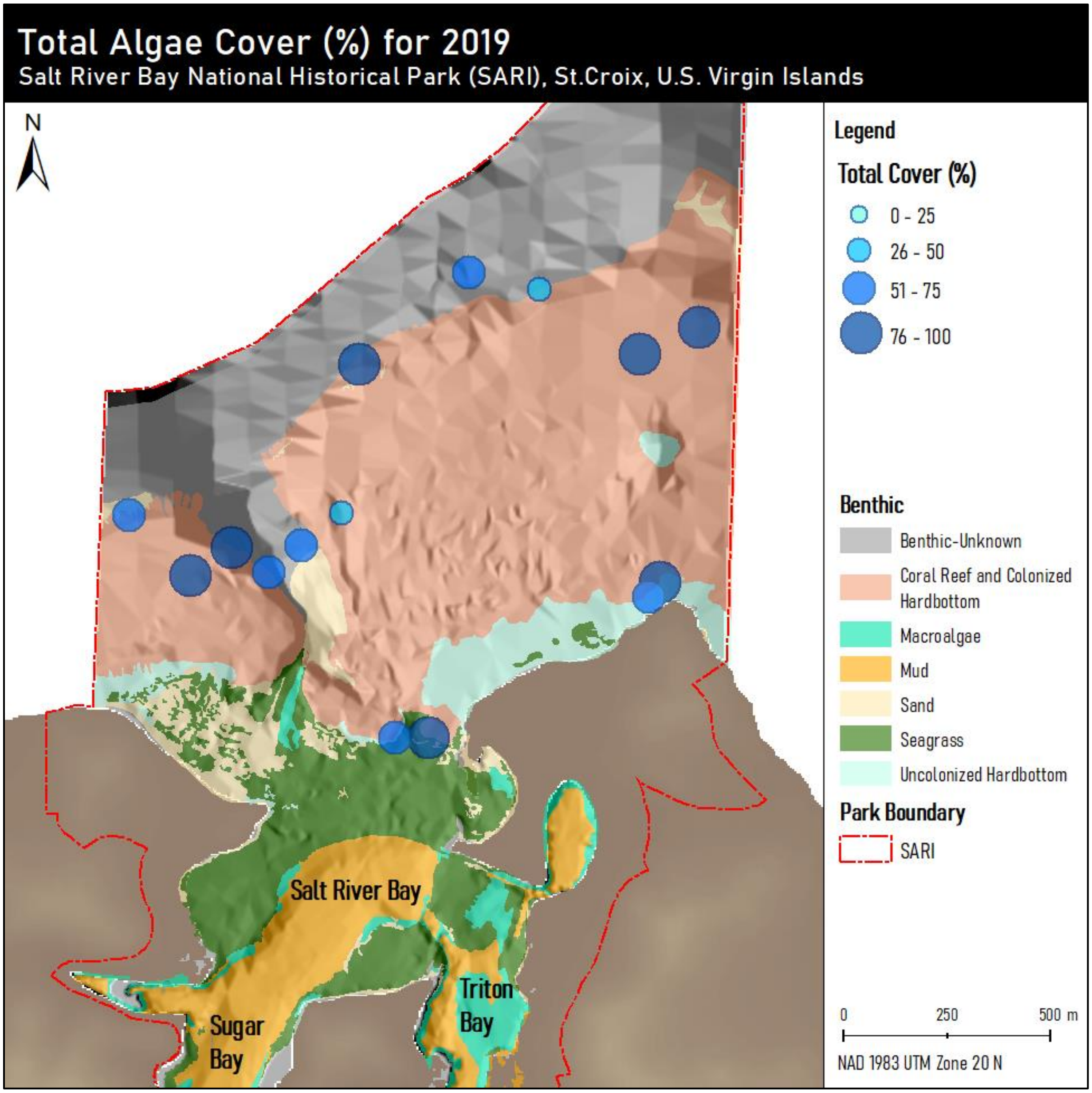


Figure 4.5.1.3. Total algae percent cover, Salt River Bay, 2019 (source – Lee Richter, NPS). Benthic habitat (Kendall et al. 2005).

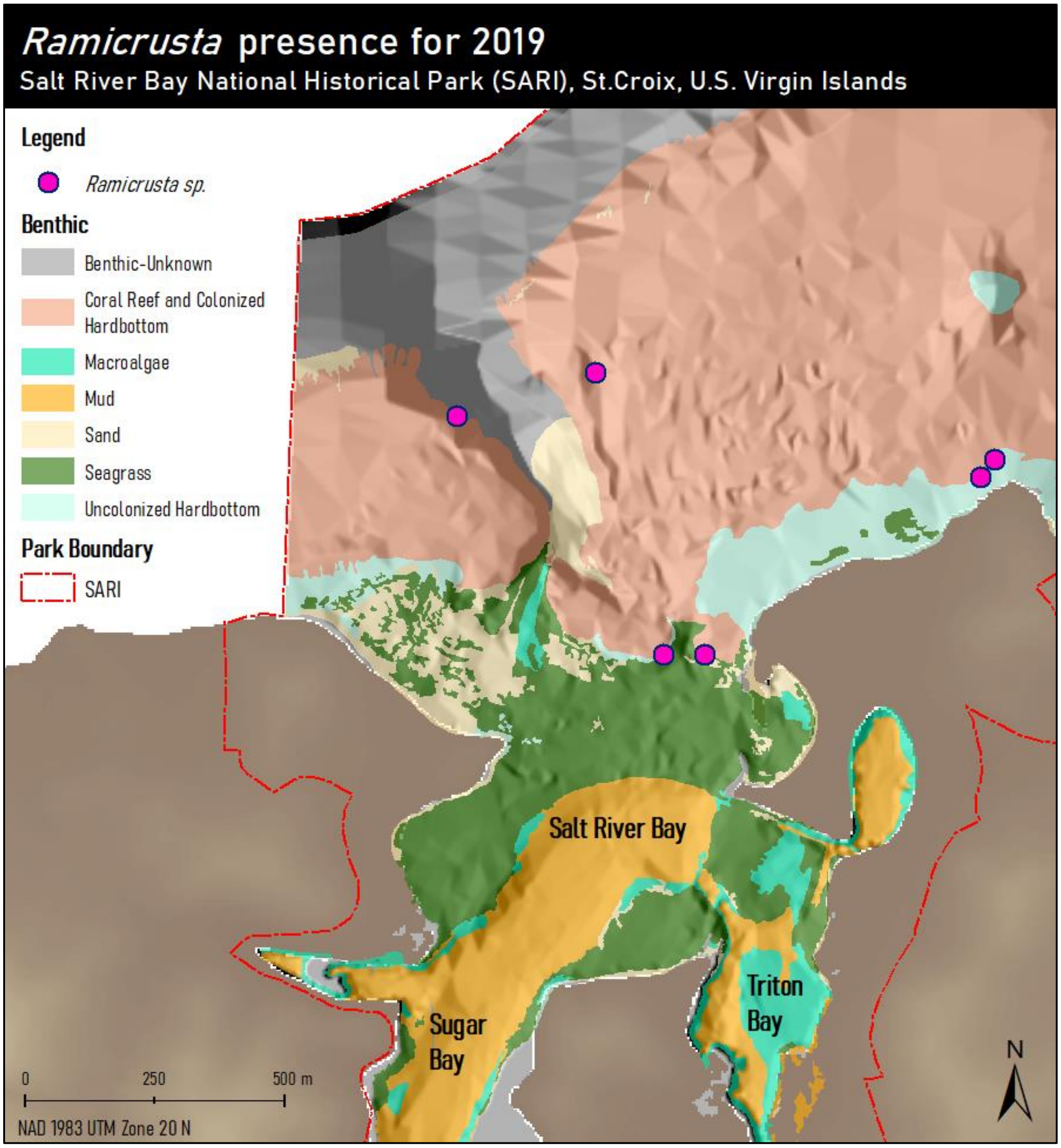


Figure 4.5.1.4. *Ramicrusta* presence, Salt River Bay, 2019 (source – Lee Richter, NPS). Benthic habitat (Kendall et al. 2005).

Threats and Stressors

Storm water runoff with increased nutrients and pollutants may impact nearshore algal hardbottom habitats. Algal abundance may increase and community composition may shift to algal species with higher potential growth rates. Herbivores exert grazing pressure on algal communities and changes to herbivore populations will also affect algal abundance and community composition. Changes to algal communities may occur via top-down control if predators of herbivores change in abundance. Such a change in predation pressure may result from changes in fishing practices. Climate change effects on water temperature and ocean acidity may produce unknown and unforeseen effects on algal communities.

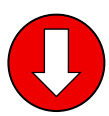
<u>Data Needs/Gaps</u>

Algal communities within Salt River Bay have not been surveyed. Algal surveys within SARI have been limited to hard-bottom and coral reef communities just outside the bay. Current algal community compositions are limited to mostly broad categories. Assessment of species richness and diversity has not been published since Fredrik Børgesen's surveys 100 years ago. Identification of the functionally important peyssonnelid algae are of specific concern so as to understand the origins of the recent rapid spread of peyssonelid crusts and overgrowth on corals. An increased understanding of nutrient and herbivore controls on reef algal communities is also needed to evaluate current algal cover and its relationship to reef health.

<u>Overall Condition</u>

According to current understanding, algae cover within the hard-bottom communities of SARI is high (mean total algae percent cover in 2019 = 73%) and algal dominance (considered >50% percent cover) is considered indicative of coral reef decline (Edmunds 2013; Bruno et al. 2014). The recent high levels of algae cover warrant concern but it must be considered that strict baselines for healthy levels of algae cover are not fully established as pristine conditions are difficult to verify (Table 4.5.1.1). Also, different functional groups of algae vary in their effect on reef health and little is known about any deleterious effects of turf algae which is the most abundant algal type observed throughout the hard-bottom surveys within SARI. The presence of the peyssonnelid algae *Ramicrusta* spp. for the first time in SARI in 2019 is of special concern. The threat posed by this algae is possibly more significant than SARI's high total algae coverage due to the rapid expansion of this species throughout the Caribbean and its known ability to overgrow corals.

Table 4.5.1.1. Graphical summary of status and trends for macroalgae within the framework category Marine Plants, including rationale and reference condition.

Component	Indicator	Condition Status /Trend	Rationale and Reference Conditions
Macroalgae	Community Extent (Percent Cover)		Though macroalgae cover estimates have only been documented since 2015, percent cover is high and indicative of poor ecosystem health. The current spread of *Ramicrusta* sp., an aggressive coral competitor is also a serious concern.

<u>Source(s) of Expertise</u>

- Lee J. Richter, Marine Biological Scientist, National Park Service, South Florida/Caribbean Network, 1300 Cruz Bay Creek, St. John, US Virgin Islands 00830

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4.5.2. Seagrass

This section reviews the condition of the seagrass focal resource in SARI. The condition assessment spans 30 years (1970–2000) (NCCOS 2004) of data to assess the status of seagrass. The status of the seagrass resource is evaluated using metrics that detect change in abundance. The condition metrics selected is areal extent of seagrass in 3 cover classes as interpreted by Kendall et al. (2005) from aerial images from 1970, 1988, 1992, and 2000 (NCCOS 2004).

<u>Description</u>

Seagrass ecosystems provide nursery habitat for commercially important fishes (Heck et al. 2003), store large amounts of carbon (Fourqurean et al. 2012), are the base of food webs (Fry et al. 1982, Zieman et al. 1984), and stabilize sediments (Fonseca 1989). Salt River Bay seagrass meadows offer nursery habitat for reef fish and protect the park's coral reefs by acting as a barrier from runoff and sediment loading. The coral reefs at the entrance to the bay protect the seagrass meadows from potentially damaging wave energy (NPS 2015).

Thalassia testudinum (turtle grass), *Syringodium filiforme* (manatee grass) and *Halodule wrightii* (shoal grass, to a lesser extent, and possibly seasonally) constitute these meadows in Salt River Bay while the more depth tolerant *Halophila decipiens* is found seasonally on the floor of Salt River Canyon (Josselyn et al. 1983; Kenworthy et al. 1989; Kendall et al. 2005). Though not yet observed in surveys in Salt River Bay, the invasive exotic seagrass *Halophila stipulacea* has spread throughout the Caribbean including to nearby eastern Puerto Rico, St. John, and the British Virgin Islands since its first observation in Grenada in 2002 (Willette et al. 2014; Ruiz et al. 2017).

A study done on pollination ecology in Salt River Bay and Robin Bay (southeast coast of St. Croix) found short shoots of *T. testudinum* at these two sites to be longer and possibly older than those in other parts of the Caribbean (Cox and Tomlinson 1988). They also observed varying intervals between flowering events suggesting no regular pattern of flowering events on individual short shoots from the study areas.

<u>Data and Methods</u>

This report builds upon a NOAA Technical Memorandum (NOS NCCOS14): *An Ecological Characterization of the Salt River Bay National Historical Park and Ecological Preserve, U.S. Virgin Islands* (Kendall et al. 2005). Kendall et. al (2005) compiled aerial photographs from the 1970s to 2000 to conduct a spatial inventory of ecosystems within SARI. In this report, we interpret the trends in seagrass percent cover.

Percent cover is often used to describe seagrass ecosystems as a two-dimensional habitat. This basic time-efficient metric provides us with estimates of available habitat for myriad fauna, food availability to large grazers such as green sea turtles, and the meadow's ability to provide other ecosystem services. Though percent cover estimates are quick to produce and are often indicative of habitat quality, low percent cover estimates may hide value of highly productive but highly grazed seagrass meadows (Moran and Bjorndal 2005). Therefore, grazing intensity (e.g., green turtle grazing plots) should be considered in areas where grazers are common, as well as the use of complementary measures of seagrass abundance (e.g., shoot density). Polygon areas were derived from orthorectified aerial photos for mapping and coverage estimations (Kendall et al. 2005). Weighted areas were calculated to account for patchiness using coefficients 0.3, 0.7, and 0.95 corresponding to coverages 10–50%, 50–90%, and 90–100% respectively.

<u>Reference Conditions/Values</u>

The reference condition for seagrass cover is the 1970s, from when there are publicly available aerial photographs. Kendall et al. (2005) noted higher turbidity in the 1970s images, relative to other years, which likely resulted in an underestimation of coverage.

<u>Current Condition and Trend</u>

Areal coverage

Kendall et. al (2005) identified an overall negative trend (~13% decline) in seagrass cover in Salt River Bay from the 1970s to 2000 (Figure 4.5.2.1), a notable drop in seagrass percent cover following Hurricane Hugo (1989), and no recovery trend as of 2000. The total and weighted areas of seagrass cover in 2000 were 84.9% and 87.1%, respectively, of the 1970s (Table 4.5.2.1). The negative trend in the 90–100% seagrass cover class (Fig. 4.5.2.2) may also suggest an increase in patchiness. These trends should be interpreted with caution for multiple reasons including multiple hurricanes since 2000 that likely significantly decreased seagrass cover, differences in the amounts and areas of visible seagrass cover in the imagery from the various time periods, and differences in technical aspects (e.g., minimum mapping units (MMU)) between imagery datasets (Kendall and Miller 2008). These considerations demonstrate the need for in-water surveys to groundtruth aerial imagery data. Also, the trend depicted from the analyses of 2000 and earlier imagery does not depict later and current trends since multiple major hurricanes since 2000 most likely significantly decreased seagrass cover. These considerations demonstrate the need for in-water surveys to groundtruth aerial imagery data and produce current cover estimates.

Aerial imagery is not sufficient for determining percent cover at the depths within the Salt River Canyon, however a study done while NOAA's Hydrolab was functional, estimated 37% coverage of the canyon floor between 14 to 32 meter depths by *H. decipiens* in May of 1985 with insufficient light and increased weather related disturbances discouraging growth in the fall and winter (Kenworthy et al. 1989; Williams 1988).

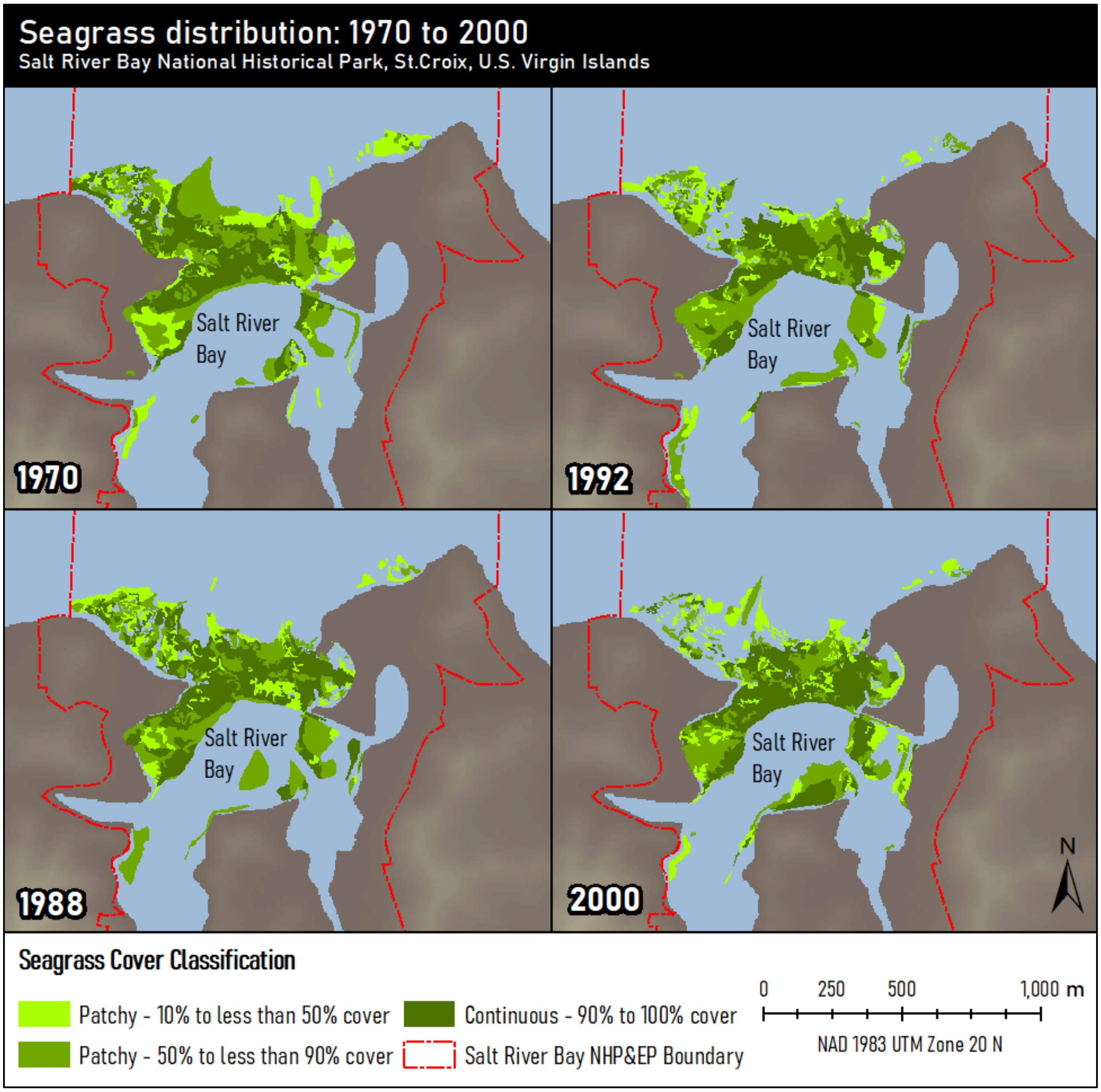


Figure 4.5.2.1. Seagrass cover classification, Salt River Bay, 1970–2000. Geospatial data from NCCOS 2004.

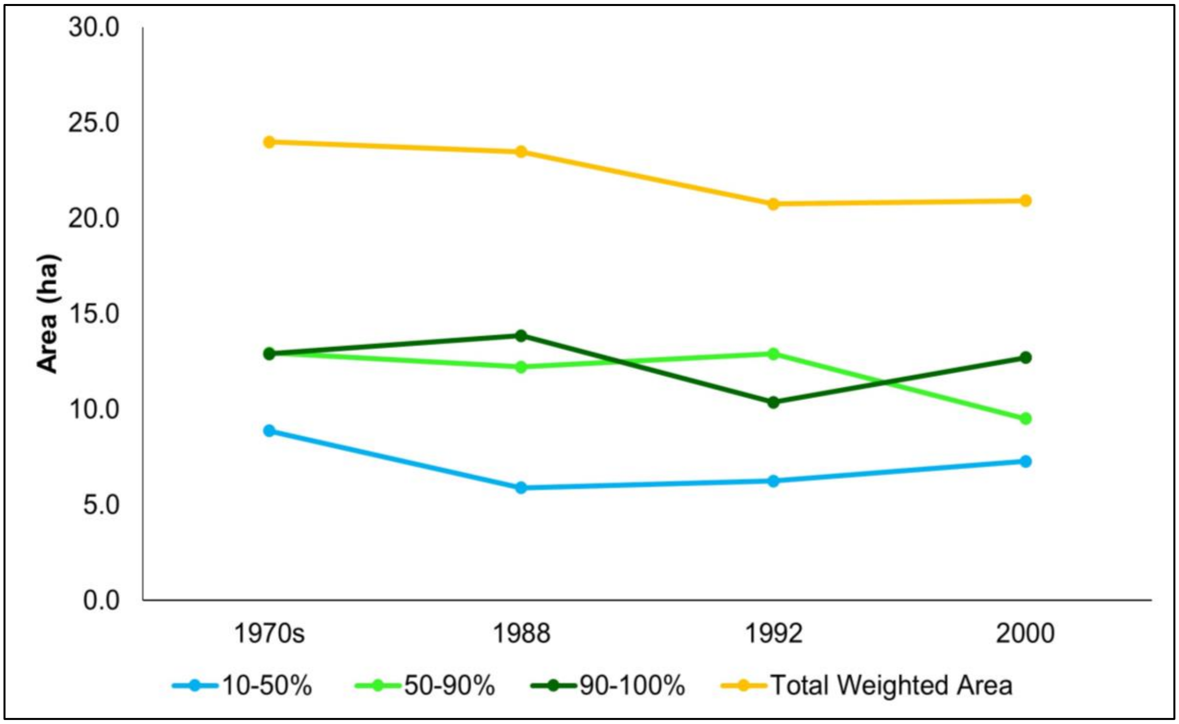


Figure 4.5.2.2. Trends in unweighted area (ha) for each seagrass percent cover class from aerial photographs and trend in total weighted area in Salt River Bay. Data from Kendall et al. 2005.

Table 4.5.2.1. Total polygon areas (ha) and weighted areas (WA) based on seagrass coverage classification for each time period in Salt River Bay; this is an expansion of table 9.1 in Kendall et al. (2005).

Seagrass Class	Coefficient	1970s		1988		1992		2000	
		Area	WA	Area	WA	Area	WA	Area	WA
10–49.9%	0.3	8.9	2.7	5.9	1.8	6.2	1.9	7.3	2.2
50–89.9%	0.7	13.0	9.1	12.2	8.5	12.9	9.0	9.5	6.7
90–100%	0.95	12.9	12.3	13.9	13.2	10.4	9.8	12.7	12.1
Total area	–	34.8	24.0	32.0	23.5	29.5	20.8	29.5	20.9
% of 70's extent	–	–	–	92.0	97.8	84.9	86.5	84.9	87.1

<u>Threats and Stressors</u>

The SARI Foundation Document (NPS 2015) identifies threats to the ecosystem primarily due to water quality degradation from increased runoff. The Salt River delivers runoff into Sugar Bay from the largest catchment area of SARI. The sources of runoff listed within the watershed include increased residential and commercial development, cleared and unvegetated land, roads, livestock and agriculture. From these sources, storm water runoff brings excess nutrients and pollutants to the bay and increases turbidity. The water catchment areas draining into Triton Bay and Mangrove Lagoon (“Bio Bay”) are much smaller and largely forested and therefore less impacted by stormwater runoff. Other sources of pollution identified include the dumping of solid waste directly into the watershed, poorly maintained and derelict boats that leak fuel and sewage into the bay, and improperly managed septic systems (NPS 2015).

Changes to temperature, salinity, nutrient inputs, depth, and turbidity can impact natural seagrass abundance and community composition in SARI. Degraded water quality is a greater risk after extreme weather events such as hurricanes or tsunamis which also inflict direct physical damage to seagrasses caused by boat groundings and increased anchorage when Salt River Bay is used as hurricane hole (NPS 2015). The health of seagrass meadows within SARI is also linked to the condition of other natural resources. Overgrazing, though not currently observed, is a potential threat that depends on the condition of grazer populations (e.g., sea turtles, urchins), and their predators (e.g., sharks, sea birds). Seagrass impacts can also be caused by degradation of the coral reefs that protect the meadows from high wave energy. During non-storm events, seagrasses may receive direct physical damage caused by dredging and boat propeller scarring. Invasive species such as *Halophila stipulacea* and disease are two biotic factors that also have the potential to cause ecosystem regime shifts.

Data Needs and Gaps

Detailed seagrass surveys including community composition, distribution, and abundance are needed in conjunction with water quality surveys. Future seagrass surveys for SARI are planned and approved. The combined water quality and seagrass data should be used to identify relationships between environmental conditions and seagrass health. This is especially critical now with the threat of invasion by the seagrass *Halophila stipulacea*. *H. stipulacea* has been observed in other seagrass meadows around St. Croix including Christiansted harbor which lies only a few miles east and upwind and current from SARI. This invasive seagrass can spread rapidly and has a wide depth tolerance range. To better understand the changes to ecosystem function caused by this invasive seagrass, a detailed baseline description of current seagrass conditions and trend monitoring is needed throughout the bay and in the canyon.

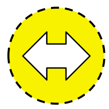
Additionally, the seagrass meadow is rather space-limited in Salt River Bay, bound by reef and canyon to the north, shoreline on the east and west, and freshwater runoff from the Salt River watershed to the south. Any significant changes in areal coverage will likely be due to changes in patchiness, shoot density, or community composition.

The data needs for an inventory of macroalgae and seagrasses has already been identified as a high priority in the SARI Foundation Document (NPS 2015). This report supports the previous assessment. More recent imagery, including that obtained subsequent to the 2017 hurricane season should be also be obtained and analyzed.

Overall Condition

The seagrass meadows in Salt River Bay warrant moderate concern, because, despite the current slight increasing trend in cover, the ecosystem is threatened by the invasion of *Halophila stipulacea* currently occurring nearby throughout the Caribbean (Table 4.5.2.2). Medium to low confidence in assessment is due to the lack of in-water surveys to ground truth aerial estimations, lack of trend data in seagrass coverage within Salt River Canyon, inconsistent turbidity levels across time and photographs used for analysis, and inconsistent methods for polygon creation.

Table 4.5.2.2. Graphical summary of status and trends for seagrass focal resource within the framework category Marine Plants, including rationale and reference condition.

Component	Indicator	Condition Status /Trend	Rationale and Reference Conditions
Seagrass	Community Extent (Areal Coverage)		Aerial photographs from the 1970s depict little change in cover through 2000, however, the spread of the invasive seagrass *Halophila stipulacea* in nearby areas is a serious concern. Low confidence in assessment is due to the lack of recent seagrass cover estimates, lack of in-water surveys to ground truth aerial estimations, and lack of trend data in seagrass coverage within Salt River Canyon.

<u>Source(s) of Expertise</u>

- Matt Kendall, Marine Biologist, NOAA/NOS/NCCOS/Marine Spatial Ecology Division, Silver Spring, MD 20910, USA

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4.6. Marine Invertebrates

4.6.1. Corals

This section reviews the condition of the stony corals and coral reefs in SARI. The condition assessment considers data provided by the South Florida and Caribbean Inventory and Monitoring Network of the US National Park Service (2012–2018), the USVI Territorial Coral Reef Monitoring Program (2001–2016), the National Coral Reef Monitoring Program (2015, 2017, 2019), as well as data sets from individual researchers working on corals during the Hydrolab Missions (1977–1989). The condition of stony corals is typically evaluated using metrics that detect changes in abundance/benthic cover, skeletal growth, coral health (bleaching, disease, partial mortality, reproduction), and temperature. The condition metrics selected for this resource assessment include benthic cover, coral health (bleaching, disease, partial mortality, reproduction), and temperature. Abundance and skeletal growth were not included in this assessment due to lack of data. Temporal trends in condition metrics were evaluated for time-series measurements.

<u>Description</u>

Salt River Bay National Historical Park and Ecological Preserve (SARI) has a diverse variety of coral reef habitats ranging from shallow partial barrier formations on either side of the mouth of the bay to colonized drop off wall formations with extensive plate forming corals to the spiral arms of black corals at greater depths. At least 41 species of stony corals have been documented (Kendall et al. 2005). The coral formations along the sheer walls provide a unique SCUBA diving experience and are important economically as one of the most popular dive sites in the U. S. Virgin Islands. SCUBA dive tours are conducted on either the east or the west walls where dive boats pick up moorings for dives that start at 12–18 m (40–60 ft). The unique topography of the Salt River canyon is a major draw for SCUBA diving tours and should be a priority for the NPS and Government of Virgin Islands Department of Planning and Natural Resources co-management team for conservation and educational opportunities.

Coral reefs range from emergent barrier reefs on either side of the entrance to the large south bay to steep sloped and vertical wall environments lining the mouth of the canyon. Each of these habitats tends to be either constructional coral reefs or hardbottoms (veneers on Pleistocene-Holocene antecedent topography), overlain by a constituent stony coral community. Shallow-water coral habitats range between diverse coral communities on hardbottom, particularly in the low angle areas surrounding the canyon, to higher cover *Orbicella* spp. coral reefs near the canyon inflection, to mostly dead and degraded *Acropora palmata* reefs surrounding emergent rocks (Kendall et al. 2005). These coral reefs likely form an important corridor for fishes between the mangrove and seagrass habitats with the lagoon and the outer coral reef environment. Monitoring programs have largely concentrated their studies of coral populations in the habitats in and surrounding the canyon and this is the focal area for this assessment (see Figure 4.2.1.1 in Section 4.2.1).

<u>Data and Methods</u>

Two monitoring programs have established long-term transects to study stony corals in SARI: the NPS South Florida/Caribbean Inventory & Monitoring Network (SFCN) and the USVI Territorial Coral Reef Monitoring Program (TCRMP). These programs maintain four monitoring sites at different depths (Figure 4.2.1.1 in Section 4.2.1). SFCN maintains one site on the eastern wall (SFCN SARI East, 9–31 m depth). The three sites on the western side of the canyon include: (1) upper shelf above wall (TCRMP Salt River West, 7–8 m depth; Figure 4.6.1.1), upper wall (SFCN SARI West, 9–31 m; Figure 4.6.1.2), and lower wall (TCMRP Salt River Deep; 28–31 m; Figure 4.6.1.3).

Figure 4.6.1.1. Representative photos of the USVI Territorial Coral Reef Monitoring Program Salt River West wall site (9 m depth). Low coral cover of weedy coral species predominate on the upper bench at this site, but higher coral cover orbicellid communities are present near the wall inflection and about 100m to the west. (photo credit: Rosmin S. Ennis, UVI)

Figure 4.6.1.2. Representative photo of the National Park Service South Florida Caribbean Inventory & Monitoring Network site on the upper area of the Salt River west wall (14 m depth). (photo credit: Rosmin S. Ennis, UVI)

Figure 4.6.1.3. Representative photos of the USVI Territorial Coral Reef Monitoring Program Salt River Deep wall site (30 m depth). Large carbonate buttresses are surrounded by sand grooves oriented downslope. The buttresses support large colonies of lettuce coral, predominately *Agaricia lamarcki*, as shown in bottom photo. (photo credit: Rosmin S. Ennis, UVI).

Monitoring protocols are described for SFCN here https://www.nps.gov/im/sfcn/index.htm and in Millet et al. (2017). Monitoring protocols are described for TCRMP here https://sites.google.com/site/usvitcrmp/. These two programs use fixed permanent transects to assess coral reef resources over time, a method often referred to as longitudinal monitoring. Additional monitoring resources include the National Coral Reef Monitoring Program (NCRMP), which incorporates randomized site surveys that fall within SARI. NCRMP also maintains environmental monitoring in SARI, which includes thermistors at depths of 1, 5, 10, and 25 m, bottle sampling for carbonate chemistry, and accretion panels.

TCRMP Monitoring

Monitoring by TCRMP was started on August 23, 2001 at Salt River West under a permanent diving mooring about 100 m east of the SFCN site at a depth of 8 m (Nemeth et al. 2002). An additional site, Salt River East, was also established on the eastern wall on October 11, 2001 in a depth of 13 m (N 17° 47.221, W 64° 45.445). In 2001 Salt River East had about 9% stony coral cover (Nemeth et al. 2002). However, due to limited resources this site was abandoned in 2002 in favor of maintaining only the Salt River West site and the Salt River East data is not included as part of the assessment. The long-term TCRMP Salt River West site is composed of 6 transects that radiate in three directions from the mooring base. The spokes of two transects each are arranged along the wall to east, to the west, and the remaining pair radiate shoreward. An additional deeper TCRMP site was established in 2009 at Salt River Deep below the 8 m monitoring site. Four of six initial transect locations were at 30 m depth and the remaining two were at 40 m depth. Because of low coral cover at the 40 m depth, the two transects here were moved to 30 m in 2010. All transects were between 28 and 31 m water depth and arranged in a line along the wall with about 3–5 m distance between transects.

SFCN Monitoring

Monitoring by SFCN at Salt River was started in 2012 with a site located at the inflection of the outermost part of the western wall in a depth of 15.9 m (min. transect depth = 9.7 m, max. = 31 m). The site is composed of 20 randomly positioned permanent transects.

At each of the three monitoring sites, temporary transects lines are stretched between permanent marking stakes. Divers swimming with a downward pointing digital video camera film the benthos. From the images, non-overlapping still frames are captured and processed for benthic cover (Kohler and Gill 2006). Stony coral summaries include the cover of the hydrocoral *Millepora* spp. since this genera contributes to reef structure. Along the TCRMP transects, each coral colony intercepted by the transect line is assessed for health indicators following a modified Atlantic and Gulf Rapid Reef Assessment protocol (Kramer et al. 2005; Smith et al. 2016a). Additionally, within a 10 x 2 m area along the SFCN transects, coral disease and species information is collected (Miller et al. 2017).

<u>Reference Conditions/Values</u>

Extensive coral reef research was conducted in the Salt River canyon as part of the NOAA National Undersea Research Program Hydrolab and Aquarius missions from 1977 to 1989. The Aquarius Habitat was removed after Hurricane Hugo in 1989 and there was a limited benthic research within this area until the initiation of the TCRMP in 2001. The Hydrolab mission characterized the canyon walls, reporting over 41 species of coral and 86 species of sponge on a shallow-water (< 25 m depth) coral reef area covering approximately 116.3 ha (Kendall et al. 2005). Rogers et al. (1984) established and surveyed 8 transects each at sites in 9, 18, 27, and 37 m depths on both the east and west wall (64 total transects) between 1981 and 1982. On the east wall living coral cover was 19% ±

7 SD, 24 ± 6, 21 ± 5, 5 ± 4 from the shallowest to deepest site. On the west wall coral cover was 13% ± 7 SD, 24 ± 8, 14 ± 9, 6 ± 6 from the shallowest to deepest site. Aronson et al. (1994) studied the impacts of Hurricane Hugo (Sep. 17, 1989) on coral communities at six sites from 8–33 m depth and found that relative coral cover declined from about 31%–73% across the depth, with the largest declines at depths between 21–33m. Coral cover on the upper wall was dominated by *Montastrea cavernosa*, *Agaricia agaricites*, and *Madracis decactis*, with *Agaricia lamarcki* dominating the lower wall (Rogers et al. 1983; Kendall et al. 2005).

Prior to the mid-1980s, macroalgae were likely to have been in low abundance in the Salt River Canyon since historically, herbivory was high on Caribbean reefs and macroalgae were rare (Dahl et al. 1974). A database chronicling earlier reef survey data also found generally low macroalgal abundance prior to the 1990s (Jackson et al. 2014). On St. Croix, cover of macroalgae changed dramatically after the die-off of the urchin *Diadema antillarum* (Carpenter 1988).

Current Condition and Trend

Mixed coral communities

Stony corals have declined in abundance at all depths on the west canyon wall relative to reference conditions in the late 1970s. Hurricane Hugo in 1989 and mass coral bleaching in 2005 were likely the largest drivers of change, followed by slow or no recovery due to localized factors. These factors include reductions in herbivory due to the die-off of black-spined sea urchins (Carpenter 1988) and fishing of herbivorous fishes, with concomitant increases in macroalgal abundance (Jackson et al. 2014). Reference values prior to Hurricane Hugo were above 20% coral cover, whereas no site monitored since 2005 exceeded 15% cover. It is possible there has been a failure of recovery since Hurricane Hugo in 1989, which caused a decline in absolute coral cover to 10–15%.

The upper shelf bench of the west wall (TCRMP Salt River West) had low (~8%) and stable coral cover between 2001 and 2016, a time period that includes the 2005 coral bleaching event (Figure 4.6.1.4). There has been a gradual decline of epilithic algae, which are diminutive filamentous and turf algae that can indicate higher levels of herbivore grazing. Epilithic algae were replaced by increasing cover of macroalgae, particularly the large brown algae (Phaeophyceae) *Sargassum* spp. and *Dictyota* spp.

Coral cover was diverse with 27 coral species recorded and no dominance by any one taxa (Figure 4.6.1.5). In contrast to findings by Rogers et al. (1984) the dominance of *Agaricia agaricites* from the same area in 1982 had declined from 17.8% in of the community to 5% of the community (Figure 4.6.1.5). The modern community on the rim above the canyon is co-dominated by *Porites astreoides* and *Siderastrea siderea.* The SFCN SARI sites had about 10% coral cover between 2012 and 2018, which remained stable (Figure 4.6.1.6). Macroalgae had a trend of declining abundance through time and was replaced by epilithic algae. The coral community assemblage was diverse, with 30 species recorded in 20, 10 m long video transects, and co-dominated by *A. lamarcki* and *Orbicella* spp. (Figure 4.6.1.7).

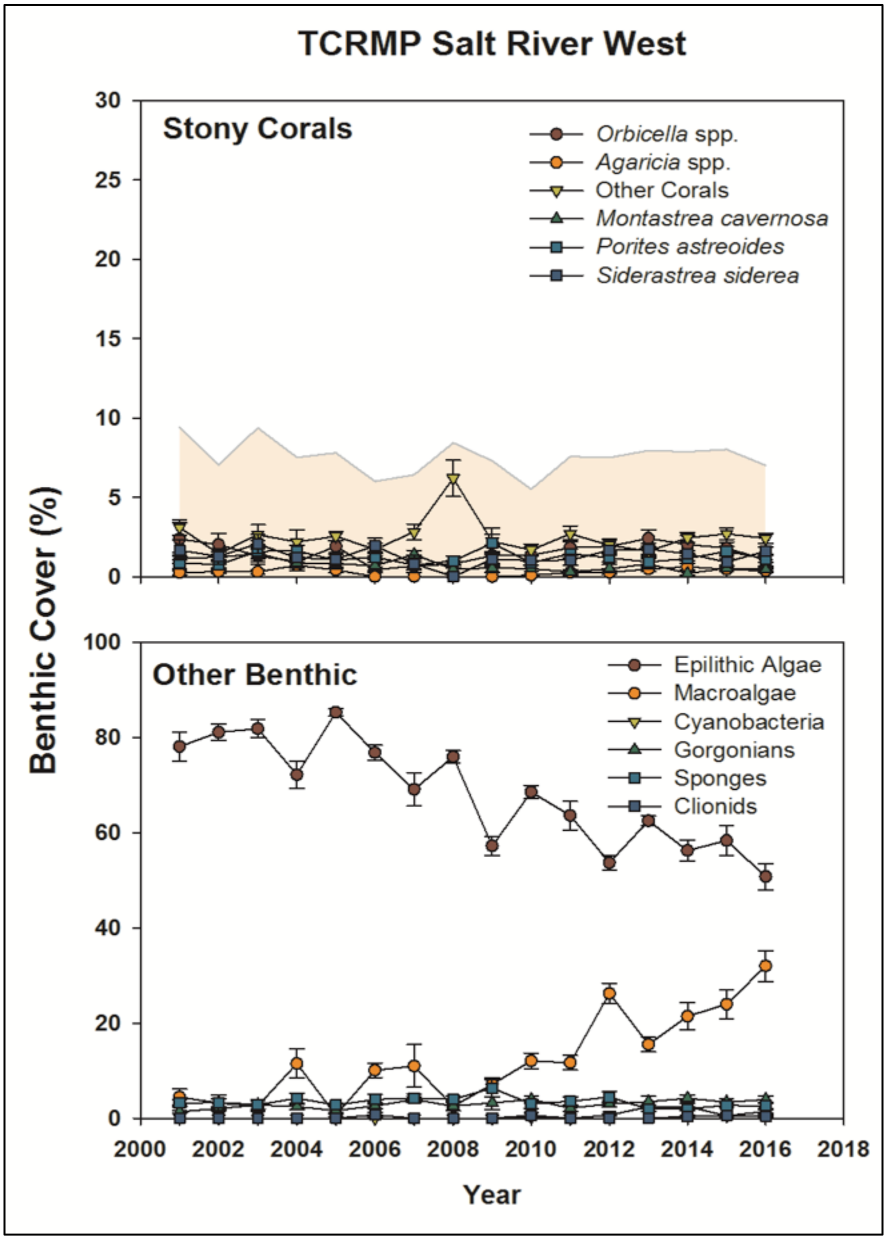


Figure 4.6.1.4. Cover of sessile epibenthic organisms (±SE) through time at the USVI Territorial Coral Reef Monitoring Program Salt River West site. (Top) Cover of stony corals, with total coral cover indicated by shaded area, then the most abundant individual species from the full data set indicated as separate markers and lines. (Bottom) Other benthic organisms. Data from TCRMP.

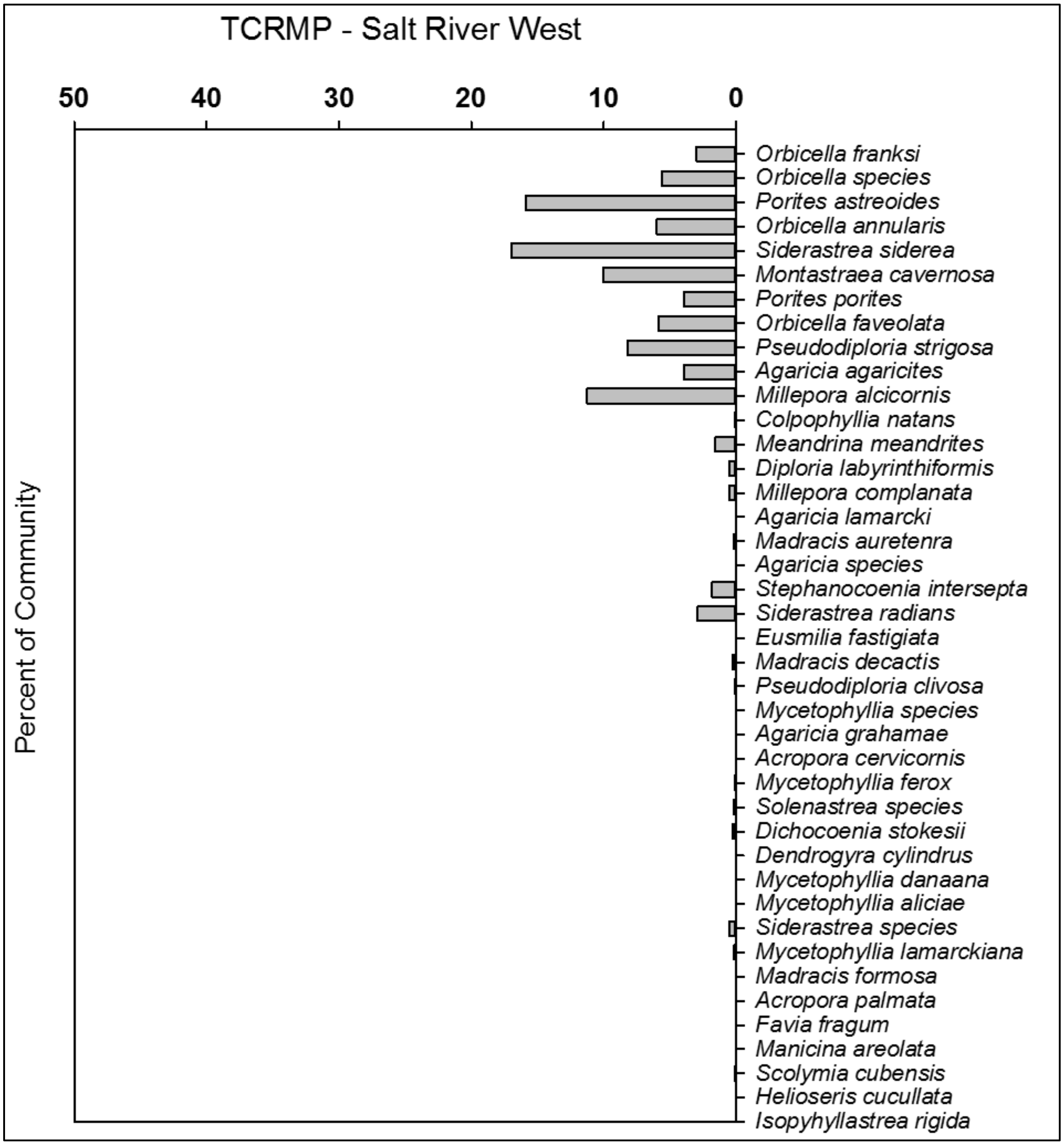


Figure 4.6.1.5. Relative abundance of coral species by benthic cover at USVI Territorial Coral Reef Monitoring Program Salt River West wall site. Coral species are ordered by the rank abundance (top to bottom) according to abundance across the TCRMP shallow water sites outside park areas (26 sites).

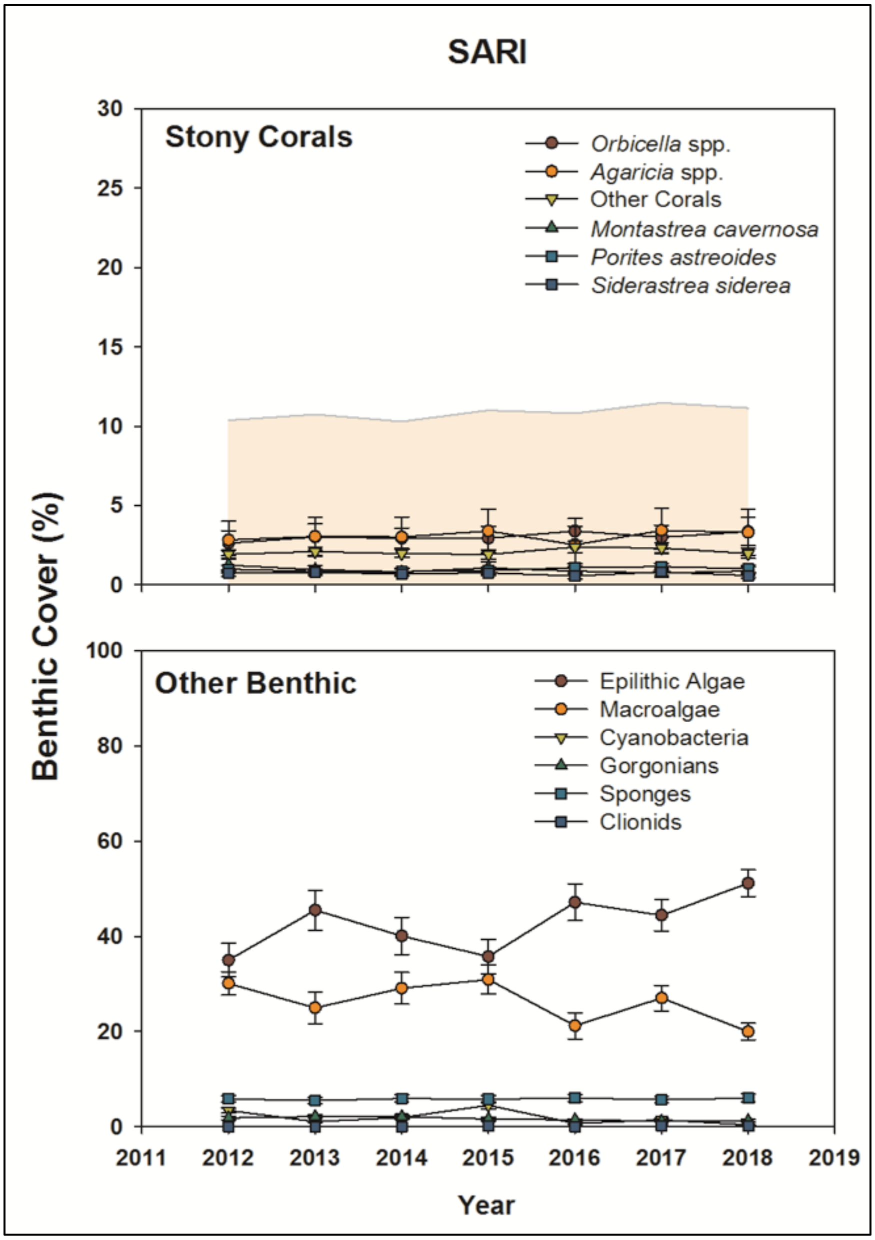


Figure 4.6.1.6. Cover of sessile epibenthic organisms (±SE) through time at the Salt River monitoring sites of the National Park Service South Florida Caribbean Inventory & Monitoring Network. (Top) Cover of stony corals. Total coral cover indicated by shaded area, then the most abundant individual species from the full data set indicated as separate markers and lines. (Bottom) Other benthic organisms. Data from SFCN.

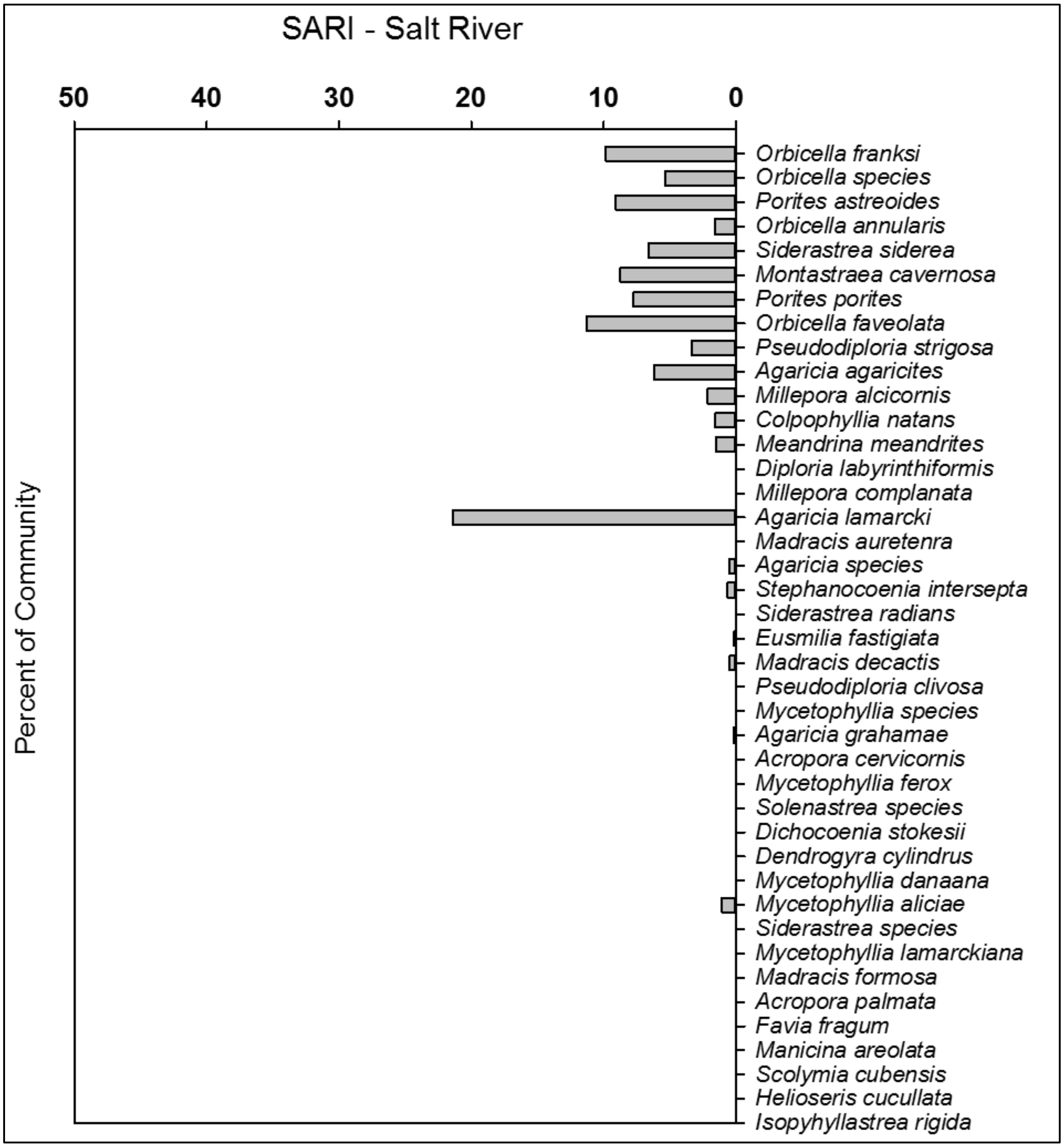


Figure 4.6.1.7. Relative abundance of coral species by benthic cover at the National Park Service South Florida Caribbean Inventory & Monitoring Network Salt River upper wall site. Coral species are ordered by the rank abundance (top to bottom) according to abundance across the TCRMP shallow water sites outside park areas (26 sites). Data from SFCN.

The lower wall (TCRMP Salt River Deep) had approximately 6% coral cover in 2010 after transect positions were permanently sited, but this increased to approximately 10% by 2013 (Figure 4.6.1.8) through rapid growth of *A. lamarcki* colonies. Macroalgae and epilithic algae have been largely stable at the lower wall (TCRMP Salt River Deep) station. The site is now about 80% composed of *Agaricia* spp. and is less diverse than the upper wall, with only 20 coral species present in transects compared with 27–30 on the upper wall (Figure 4.6.1.9).

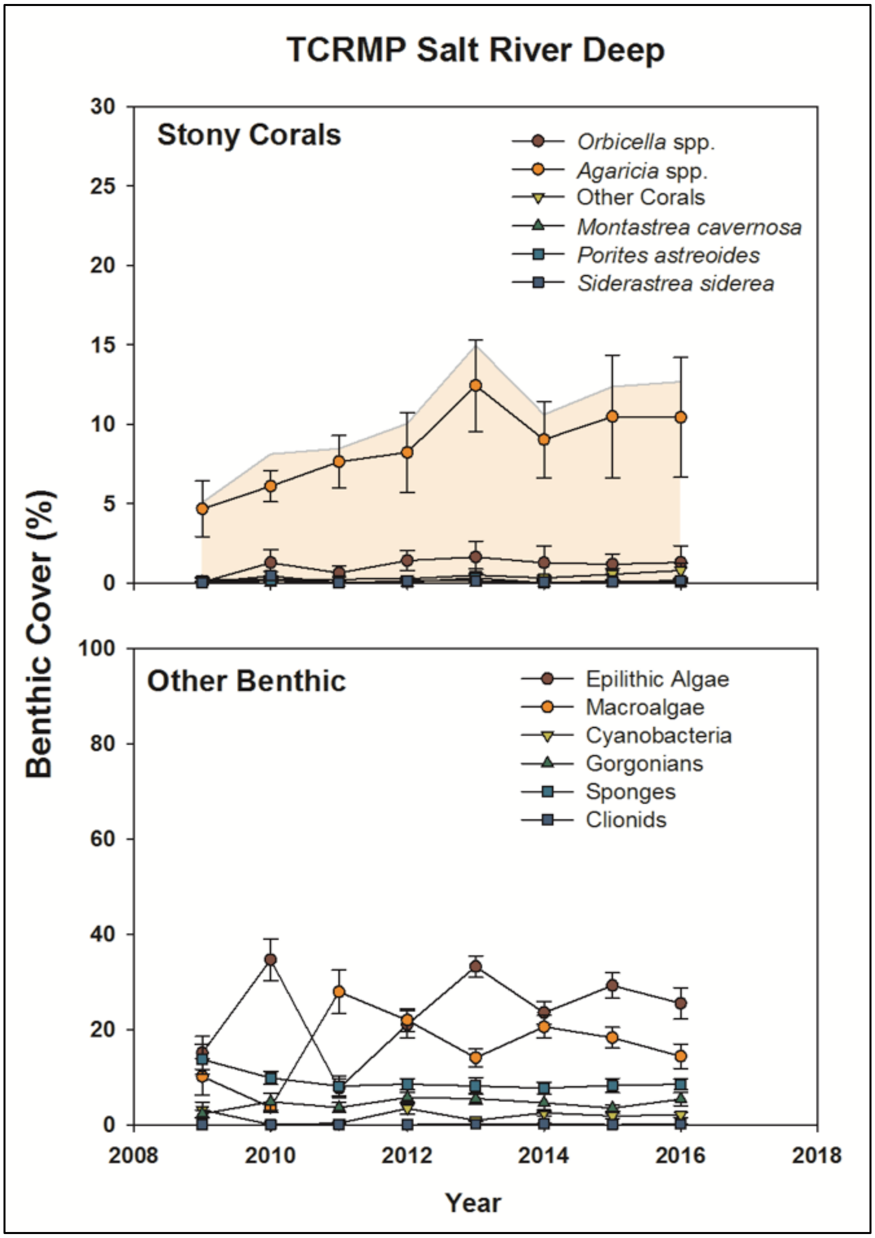


Figure 4.6.1.8. Cover of sessile epibenthic organisms (±SE) through time at the USVI Territorial Coral Reef Monitoring Program Salt River Deep wall site. (Top) Cover of stony corals. Total coral cover indicated by shaded area, then the most abundant individual species from the full data set indicated as separate markers and lines. (Bottom) Other benthic organisms. Data from TCRMP.

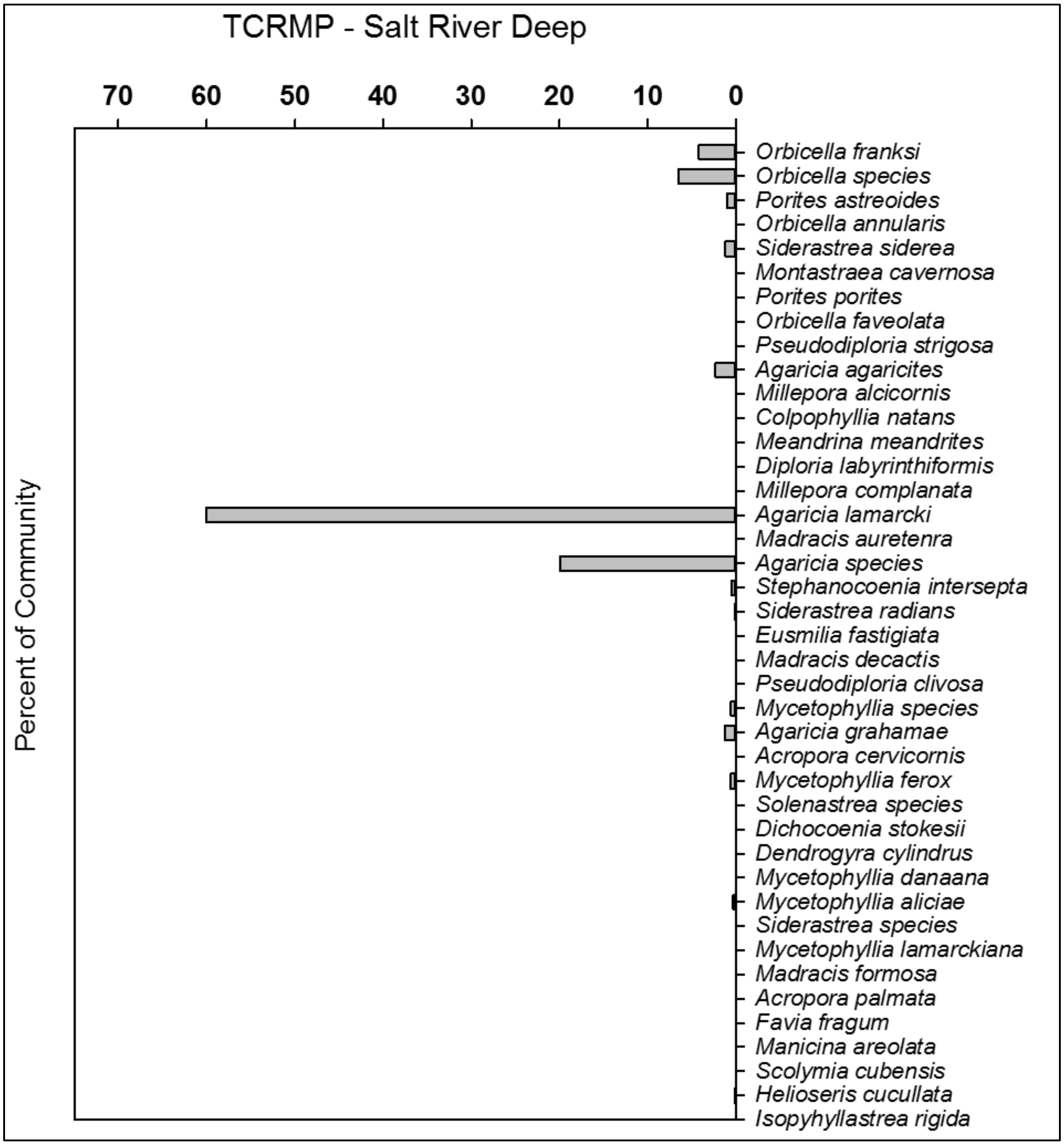


Figure 4.6.1.9. Relative abundance of coral species by benthic cover at the USVI Territorial Coral Reef Monitoring Program Salt River Deep wall site. Coral species are ordered by the rank abundance (top to bottom) according to abundance across the TCRMP shallow water sites outside park areas (26 sites).

Below the TCRMP Salt River Deep transects at depths greater than 30 m much less is known about stony coral populations. Observations by divers to 100 m depth on the wall indicate that most coral development ceases at about 46 m, with the community transitioning to sponges, whip corals, and black corals on carbonate outcrops surrounded by high amounts of sediment (Figure 4.6.1.10).

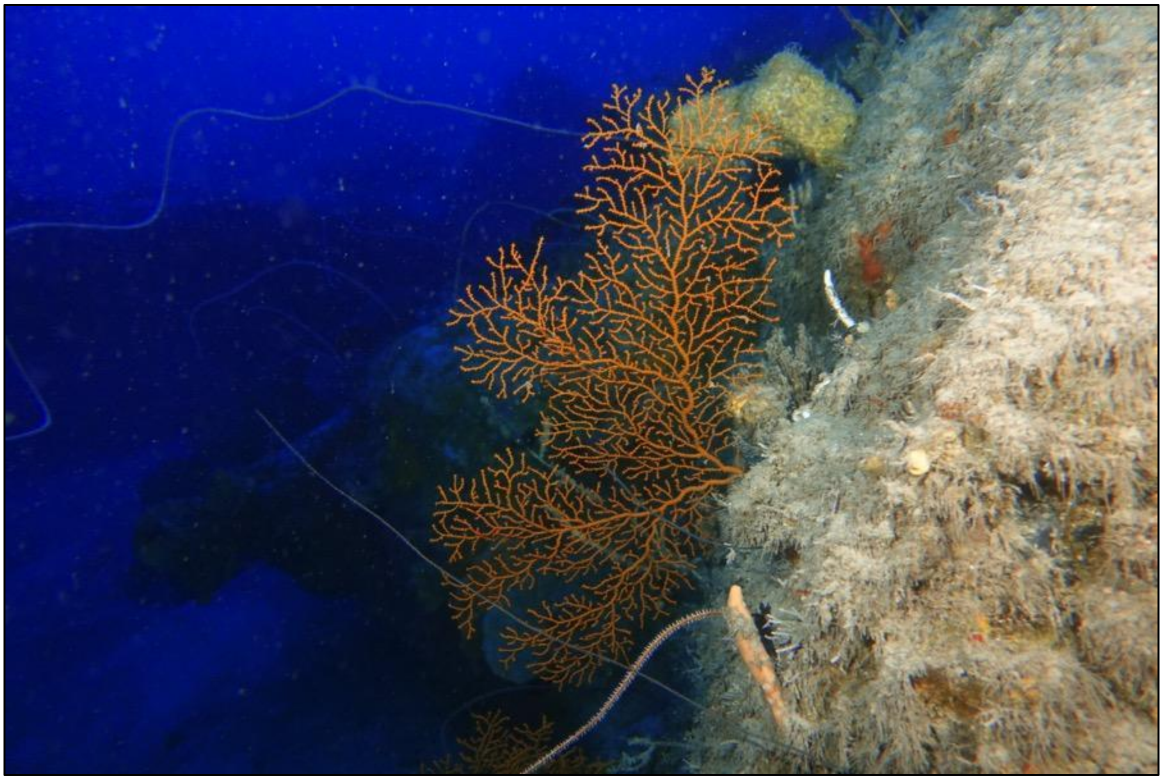


Figure 4.6.1.10. An outcrop in the Salt River canyon below monitoring sites at 90 m depth. An unidentified black coral (*Antipatharia*) is shown center, with further colonization by wire corals and sponges. (photo credit: Viktor Brandtneris)

Algae have shown contrasting trends in abundance at the different assessment sites in SARI. In general, epilithic algal community (EAC) cover, which can indicate reef substrata that is more heavily grazed, was higher at SARI than at other sites in the territory outside of park boundaries (mean $EAC_{Outside}$ = 33% compared with 40–80% inside), whereas macroalgal cover was lower (mean $macroalgae_{Outside}$ = 31% compared with 5–30% inside; Figure 4.6.11). However, at the TCRMP Salt River West upper shelf site, macroalgae has been increasing, and EAC cover decreasing in cover since about 2013 and macroalgal cover was near outside park means by 2016 (~35%, Figure 4.6.1.4). This increase did not occur at the upper wall site.

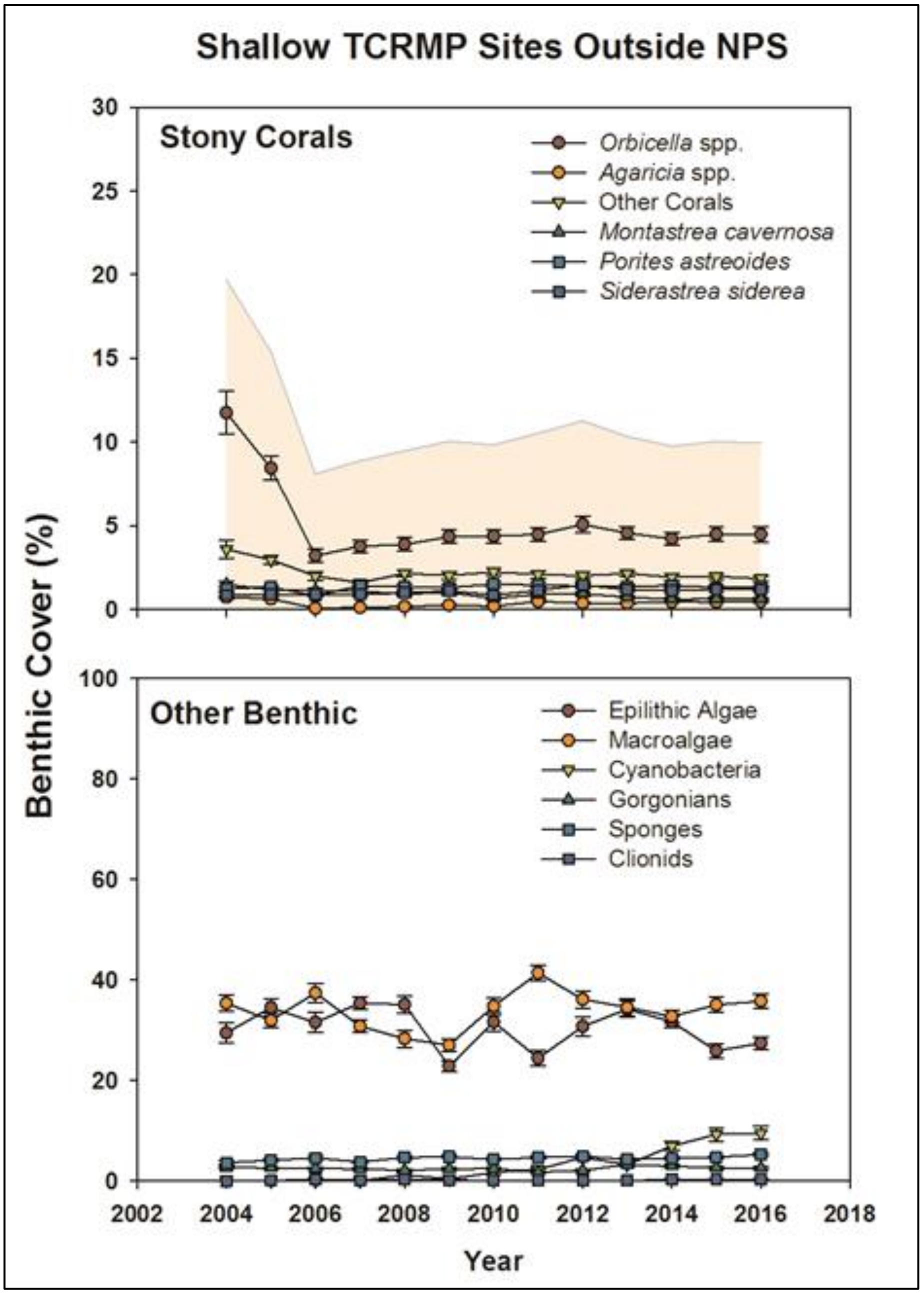


Figure 4.6.1.11. Cover of sessile epibenthic organisms (±SE) at coral reef monitoring sites outside of the USVI National Parks. (Top) Cover of stony corals. Total coral cover indicated by shaded area, then the most abundant individual species from the full data set indicated as separate markers and lines. (Bottom) Other benthic organisms. Data from TCRMP.

General distributions of corals in SARI were recorded in the National Coral Reef Monitoring Program at spatially randomized sampling sites without repeat sampling (Figure 4.6.12). Many sites recorded between 2015 and 2019 had 10–20% cover. Data also suggest that sites with higher coral cover may be more concentrated on the western wall area.

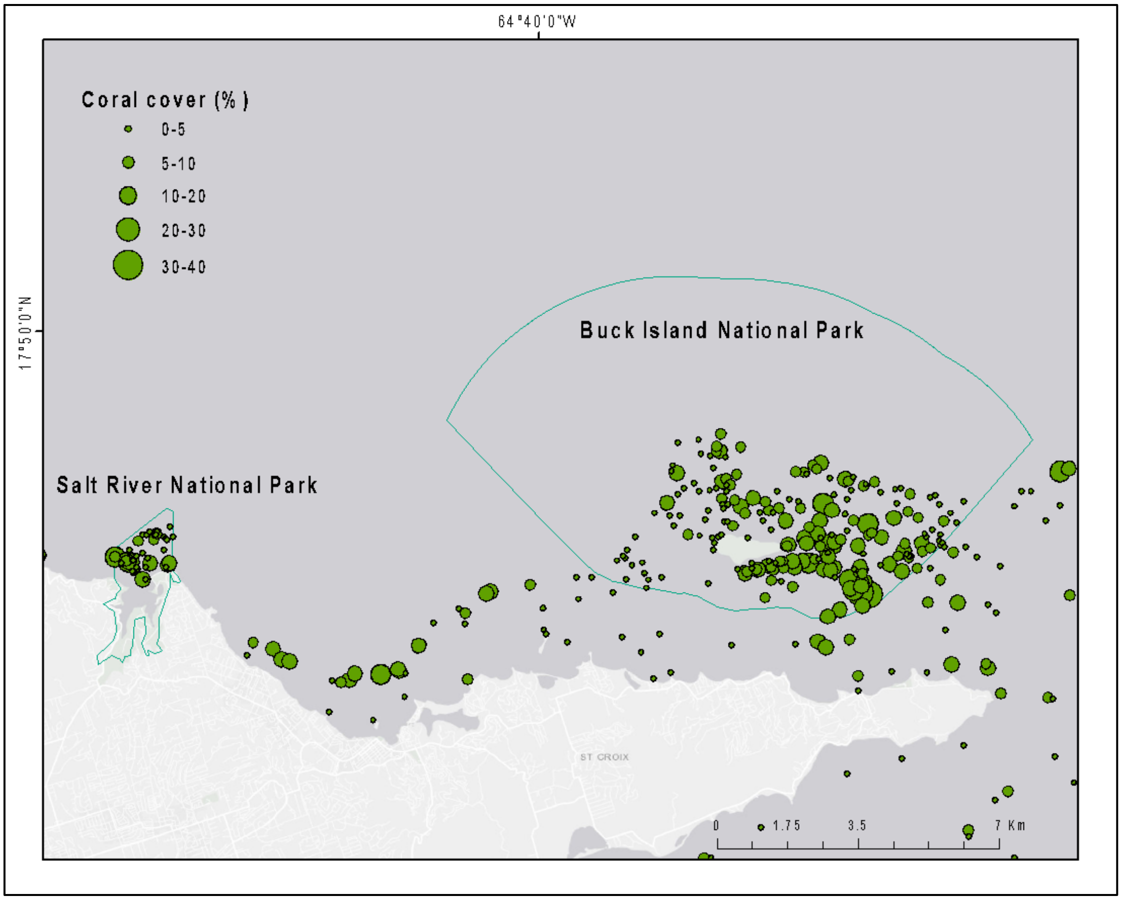


Figure 4.6.1.12. Stony coral cover recorded at randomly selected hardbottom sites for northeastern St. Croix and Salt River. Data from the National Coral Reef Monitoring Program covering years 2015, 2017, and 2019 (data and map courtesy of Sarah Groves, NOAA, Sep. 4, 2020).

Elkhorn and Staghorn Corals

Corals in the genus *Acropora* are listed as threatened on the U.S. Endangered Species List (NOAA 2014) and colonies are present in SARI but have not been systematically inventoried. Elkhorn coral (*Acropora palmata*) populations were evaluated around Whitehorse Rock on the northeastern side of SARI (Figure 4.6.1.13) as part of the Acroporid Monitoring and Mapping Program on March 17, 2013 (Smith et al. 2014). Corals were assessed within three 7-m radius circular plots established to encompass a target of about 120 colonies at monitoring sites. Corals were only surveyed once due to loss of funding for the program. Within the plots, 62 colonies of *A. palmata* were recorded that had an average maximum dimension of 38l.9 cm. Half of the colonies showed partial mortality from unknown sources. The abundance and status of additional elkhorn corals, staghorn corals, and a hybrid of the two, fused staghorn corals (*A. prolifera*), within SARI are not well documented. As part of a larger survey effort around St. Croix to predict the population of acroporids in 2012, four randomized site surveys were conducted in SARI and only one site had elkhorn coral and no sites had staghorn coral (Smith et al. 2014). The elkhorn coral colony was found around Whitehorse Rock located east of SARI bay.

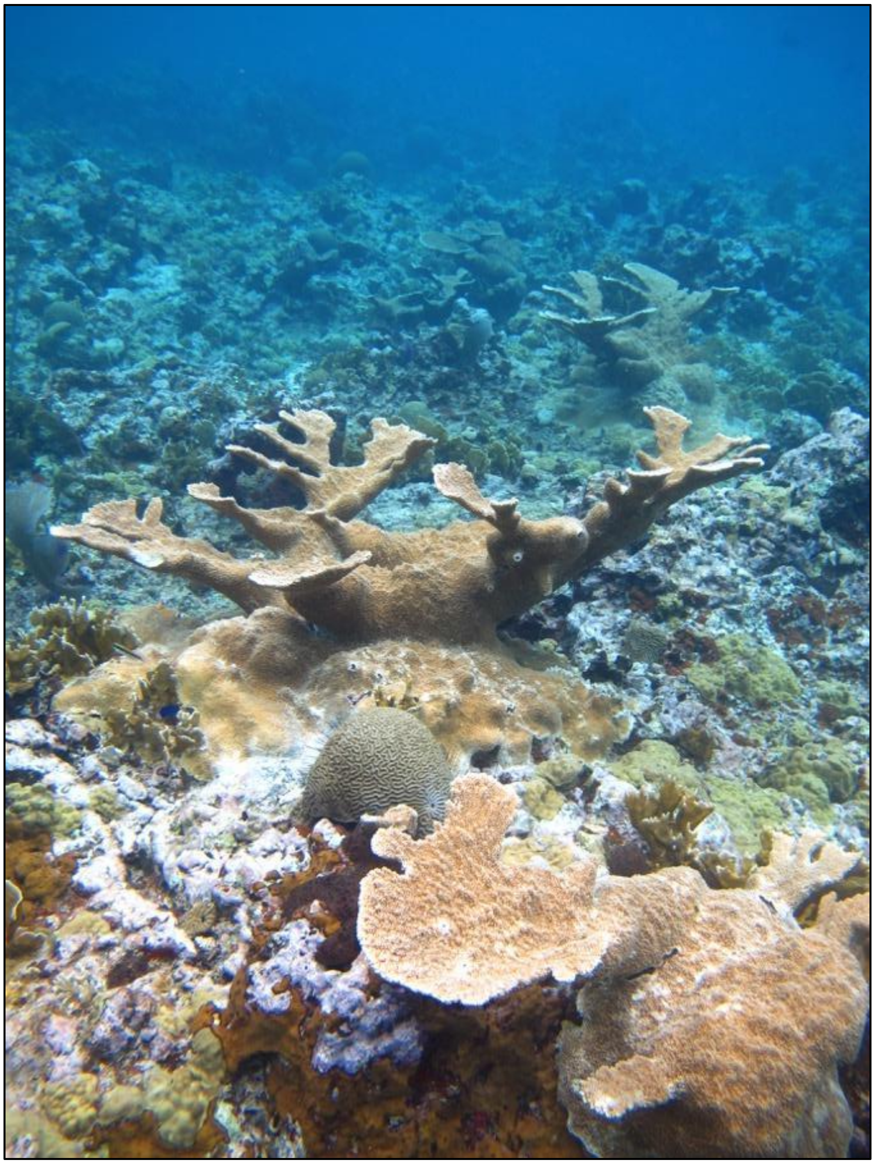


Figure 4.6.1.13. A stand of *Acropora palmata* that forms part of the discontinued Whitehorse Reef monitoring location of the USVI Acropora Monitoring and Mapping Program. Depth: 3m, Date: March 17, 2013 (Photo Credit: Tyler Smith).

Threats and Stressors

The coral reefs in SARI are threatened by climate change, disease, recreational and commercial fishing, land-based sources of pollution, recreational diving, and storms. The oceans surrounding SARI are warming at a rate of about 0.006°C per year and this is leading to repeated temperature anomalies surpassing coral bleaching thresholds (Figure 4.6.1.14). Warming oceans linked to climate change (Donner et al. 2007) contributed to the 2005 coral bleaching event in the NE Caribbean Sea (Eakin et al. 2010). This event caused a 50–60% decline in living shallow-water coral cover in the US Virgin Islands (Miller et al. 2009; Smith et al. 2013) and about a 28% decline in corals deeper than 30 m depth (Smith et al. 2016b). As a metric of coral heat stress, 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.

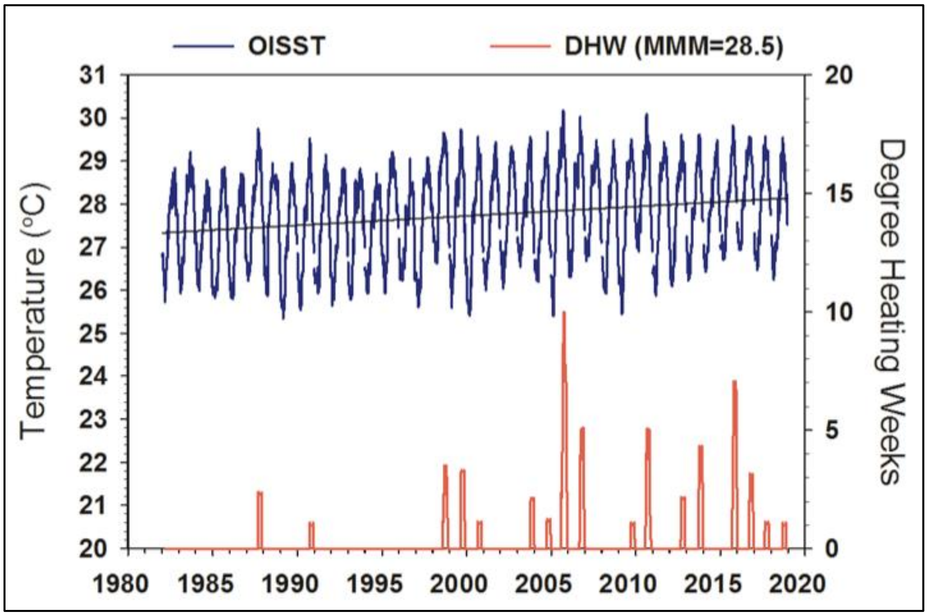


Figure 4.6.1.14. Optimum Interpolation Sea Surface Temperature (OISST; blue line, left vertical axis) and degree heating weeks (red line, right vertical axis) for the USVI. The black line is a linear fit of the OISST data and shows about 0.007°C increase in temperature per year (y = 0.000669/year*x − 25.545). OISST values averaged from coordinates 17.5N/65.5W, 17.5N/64.5W, 18.5N/65.5W 18.5N/64.5W from https://www.ncdc.noaa.gov/oisst; (accessed 6 June 2019).

At Salt River, heat stress likely surpassed 9 Degree Heating Weeks (DHW) (Figure 4.6.1.15). The regional estimate for the USVI based on SST was 10.2 DHW (50km product; NOAA 2019). A lower-impact shallow-water thermal stress event also occurred in 2010, with stress values of about 7 DHW, a level that suggests widespread bleaching but limited mortality. Corals at TCRMP Salt River West in 2005 showed extensive impacts, with about 88% of colonies bleached on Oct. 13, 2005 (Figure 4.6.1.16). Yet, mortality was limited based on coral cover estimates (Figure 4.6.1.4), which contrasted with most shallow-water sites outside of Salt River Bay (Figure 4.6.1.11). This may reflect a diverse coral assemblage dominated by corals resistant to thermal mortality, such as *S. siderea*, *P. astreoides*, and *M. cavernosa* (Smith et al. 2013). The impacts on the SARI wall environment were not directly assessed; however, there likely was bleaching and coral mortality. Evidence includes extensive bleaching of lettuce corals on the Cane Bay wall, 6 km west of Salt River (Smith et al. 2016b), to a depth of 50 m, gradual recovery of coral cover from the TCRMP Salt River Deep wall site, which was established after bleaching in 2009, and high prevalence (60–70%) of partial mortality on colonies recorded in TCRMP coral health transects between 2009 and 2019 (Ennis et al. 2019). Deeper coral reefs may also bleach out of synchronization with shallow reefs due to lower bleaching thresholds and different thermal environments (Smith et al. 2016b). Temperatures are cooler in the deeper parts of the wall (Figure 4.6.1.17).

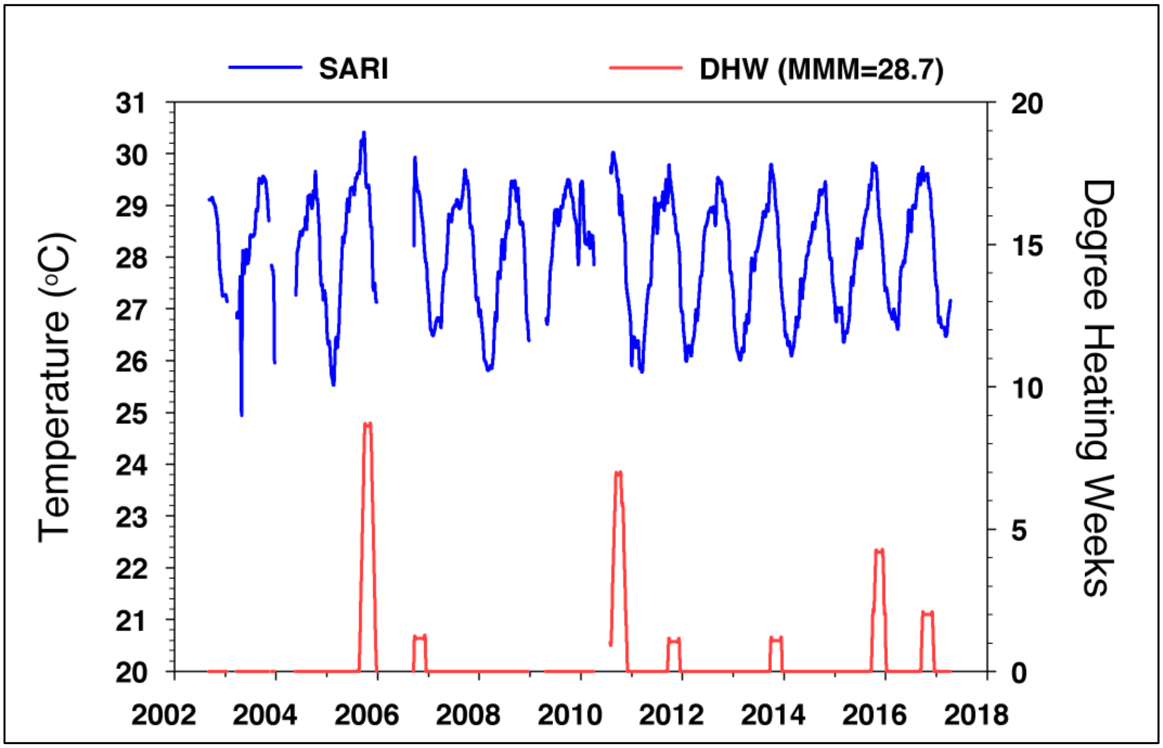


Figure 4.6.1.15. Water temperature (blue line, left vertical axis) and degree heating weeks (red line, right vertical axis) at Salt River. The monthly maximum mean for Salt River was empirically estimated from in situ bleaching responses as 28.7. Temperatures are a composite of the Salt River ICON station 1 m and 5 m sensors (https://www.coral.noaa.gov/crews-icon/icon.html), the National Coral Reef Monitoring Program 1 m sensor (NOAA 2018), and the SFCN 14m west wall sensor.

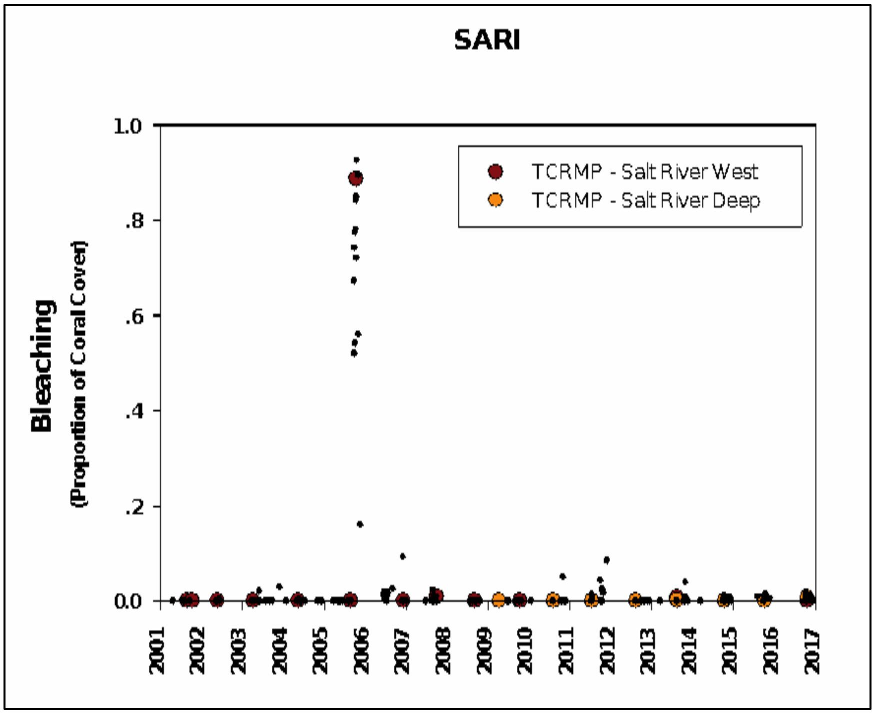


Figure 4.6.1.16. Proportion of coral cover bleached at the Territorial Coral Reef Monitoring Program Salt River West (from 2002) and Salt River Deep (from 2009) monitoring sites. Black dots are estimates from 23 other shallow water sites of the Territorial Coral Reef Monitoring Program outside park boundaries shown for reference. The 2005 event is clearly visible. Estimates from captured digital video. Data from TCRMP.

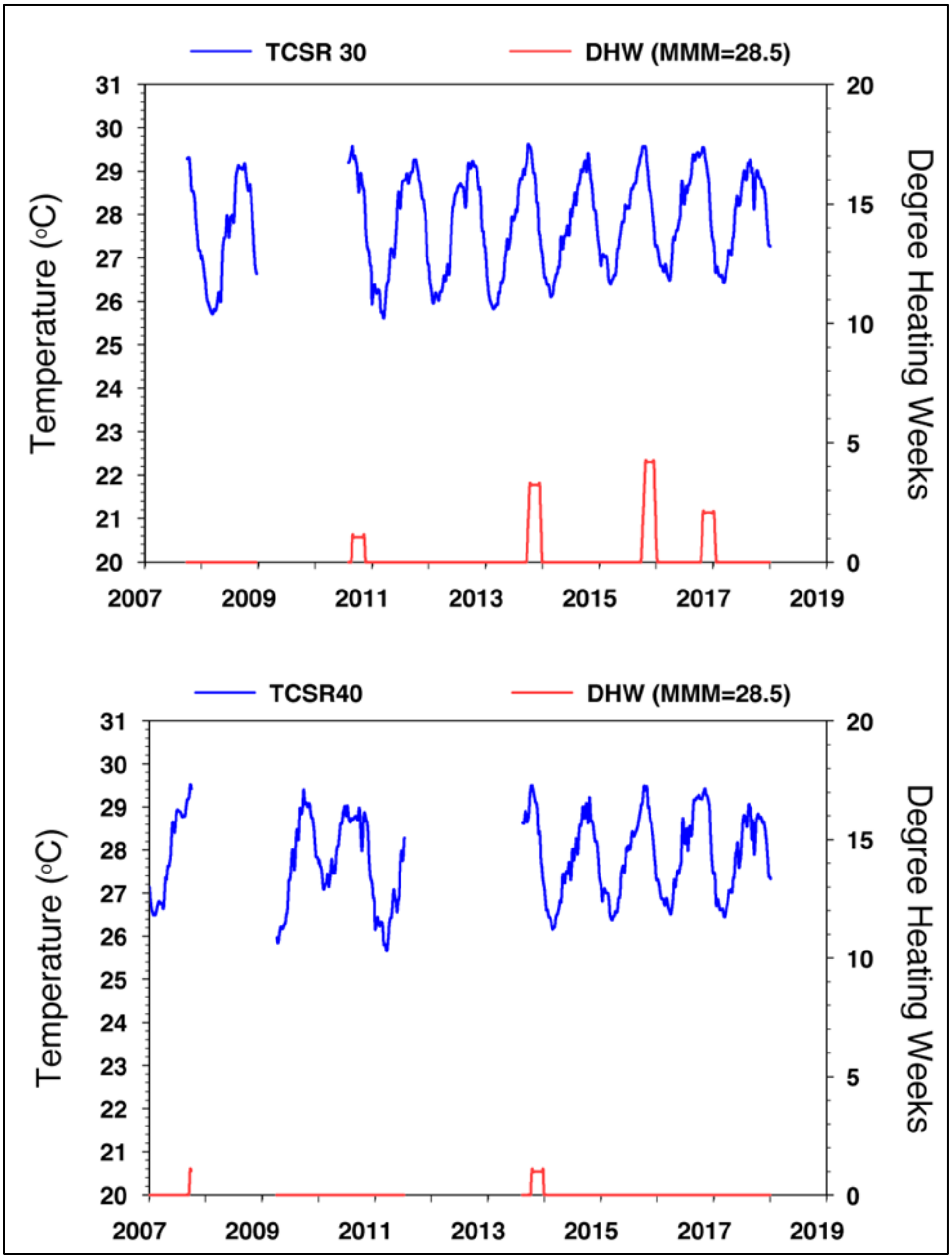


Figure 4.6.1.17. Water temperature (blue line, left vertical axis) and nominal degree heating weeks (red line, right vertical axis) at Salt River 30 m and 40 m stations of the USVI Territorial Coral Reef Monitoring Program.

The monthly maximum mean for Salt River has not been empirically estimated due to lack of observations of in situ bleaching responses at depths deeper 10 m. Thus, the DHW estimates in Figure 4.6.1.17 are underestimates of bleaching stress, as temperatures decrease with depth, leading to less at apparent stress, but coral bleaching thresholds also decline with depth, leading to higher sensitivity of the coral fauna (Smith et al. 2016b).

Fishing is also likely having an impact on the fish assemblage and their ecological roles in Salt River. The area is listed as a no-take marine reserve per V.I.C., Title 12, Chapter 1, §96. (Salt River Bay National Historical Park and Ecological Preserve, St. Croix; Designated July 19, 1995. "It is unlawful to (1) collect, take, or possess any fish…"; DPNR-DFW 2018). However, the Virgin Islands Government never fully implemented the no-take marine reserve so only territorial recreational and commercial fishing regulations are in place for SARI. The area is small, only encompassing about 100 hectares of coral reef habitat (Kendall et al. 2005), meaning fishes, such as species of parrotfish, grouper, and snapper, that have daily, seasonal, or ontogenetic movements into and outside of park boundaries remain vulnerable to fishing. Commercially important fish species are heavily exploited on St. Croix (Kadison et al. 2017), with three of the largest parrotfish species ecologically extinct.

Estimating the amount of water borne pollution (e.g., oil from boat discharge) and land-based sources of pollution (e.g., terrestrial sediments) affecting Salt River is challenging. Sediment trap studies at the TCRMP Salt River West site indicated trap accumulation rates for total sediments that were on the lower end for typical shallow water, nearshore habitats throughout the USVI (from data referenced in Smith et al. 2007; Smith et al. 2008; Henderson et al. submitted). The sediment trap accumulation rate for Salt River between May 2006 and October 2007 was 3.3 mg cm^{-2} day^{-1} (N =27) whereas the USVI mean for 10 other nearshore sites was 38.4 mg cm^{-2} day^{-1} (N = 777). This would not indicate stressful levels of sedimentation (Rogers 1990; Henderson et al. submitted). However, other forms of run-off from land, such as organic pollutants and sewage, from residential, live-aboard boating, marina, and industrial activities (e.g., the Gold Coast Yachts construction facility located within the mangrove on west side of bay) could be having an impact on water quality (Kendall et al. 2005; Bayless 2019). According to biological assays and sediment pore water analyses, sites within the Salt River Bay show indications of nutrient pollution and potential impact from some heavy metals, and this may impact coral reefs outside the bay (May and Woodley 2016; Bayless 2019; See also Section 4.2.1). In addition, upstream sources of pollution from sewage discharge and port activities near Christiansted may impact coral populations (Bayless 2019). There is evidence that corals around SARI are impacted by pollution as *A. palmata* populations show some of the lowest reproductive output of 34 populations assessed (C. Woodley et al. *in preparation*).

Lastly, recreational diving in the canyon is a daily occurrence. There are likely small, cumulative impacts from diving tourists (coral breakage, coral abrasion, pollution from personal care products) that are difficult and expensive to measure but likely impact coral health.

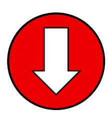
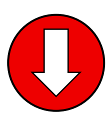
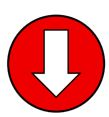
Data Needs and Gaps

Annual estimates of long-term trends in stony coral populations at three depths on the west wall by the SFCN and TCRMP are likely adequate to capture the dynamics of most coral populations, with the exception of acroporid corals and corals that occur deeper than 30 m. Trends in the population of acroporid corals are no longer part of an active monitoring program. The NPS should consider investing in an acroporid monitoring program or request that NOAA find funds to restart the Acroporid Monitoring and Mapping Program that ceased in 2013. In addition, an inventory of coral resources based on a spatially-stratified random design, such as NCRMP, would help with population estimates of key species. However, a few more years of data from the program may be necessary to develop a detailed picture of coral distributions, including the threatened acroporid corals (elkhorn and staghorn) within SARI. Monitoring for pollution and detection of sources of pollution should be a priority since there are indications of pollutants and their impacts on corals of SARI. Furthermore, the NPS may also consider a rapid response plan to capture coral mortality events, such as bleaching and disease outbreaks. Furthermore, Stony Coral Tissue Loss Disease (SCTLD, Precht et al. 2016) was reported from St. Thomas in January 2019, St. John in February 2020, and St. Croix in May 2020, and is rapidly spreading (https://www.vicoraldisease.org/sctld-disease-tracking). According to NPS researchers (N. Holloway, unpub. data) SCTLD was confirmed at Salt River on the West Wall in late January 2021. As of April 2021, the disease was severe on both the East Wall and the West Wall with most highly and intermediate susceptible species either completely dead or infected. SCTLD will have profound negative impacts on coral abundance and diversity at SARI. The NPS should be prepared to monitor changes to the coral reef fauna and consider instituting any interventions, should they become available.

Overall Condition

Based on the historical condition of coral reefs at SARI prior to and through the 1980s, the condition of corals is presently moderate to poor and trending downward (Table 4.6.1.1). Corals have declined in abundance relative to historical levels, with shifts in species composition that may reflect a high preponderance of stress tolerant species (e.g., *Porites astreoides*). At the same time abundance of macroalgae, a competitor for space with corals, may be increasing on the rim above the wall. Recovery following hurricanes may be limited and coral bleaching events may be causing coral mortality, although the TCRMP Salt River West site did show some resistance to the 2005 bleaching event.

Table 4.6.1.1. Graphical summary of status and trends for coral focal resource within the framework category Marine Invertebrates, including rationale and reference condition.

Component	Indicator	Condition Status /Trend	Rationale and Reference Conditions
Corals	Stony coral coverage		Coral reefs of SARI have declined in abundance compared to historical levels and have shown damage from hurricanes and thermal stress from surface to 30 m depth, and lack of sufficient recovery.
	Stony coral health		The incidence of coral bleaching events and coral disease epizootics has increased and is likely to continue increasing in the near future (e.g., introduction of Stony Coral Rapid Tissue Loss Disease)
	Seawater temperature		In 2005, heat stress likely surpassed 9 Degree Heating Weeks (DHW) while in 2010, another shallow water stress event occurred with DHW of 7. Extensive impact to coral but little mortality.

Source(s) of Expertise

- Nathaniel Holloway, NPS BUIS
- Tyler B. Smith, UVI, UVI Territorial Coral Reef Monitoring Program
- William Jeff Miller, NPS South Florida/Caribbean Inventory & Monitoring Network (SFCN)
- Michael Feeley, NPS South Florida/Caribbean Inventory & Monitoring Network (SFCN)
- Judd Patterson, NPS South Florida/Caribbean Inventory & Monitoring Network (SFCN)
- Doug Wilson, Caribbean Wind LLC (OISST data sets)
- Cheryl M. Woodley, National Oceanic and Atmospheric Administration
- Anthony Pait, National Oceanic and Atmospheric Administration
- Amanda L. Bayless, National Institute of Standards and Technology

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4.6.2. Queen Conch

Description

Queen conch (*Lobatus gigas*) are most commonly found in seagrass, macroalgal plains, and sand which account for about 28% of the benthic habitat in the SARI (Figure 4.6.2.1). The species has been an important fishery in the U. S. Virgin Islands (USVI); however, populations in the territory substantially declined in the 1970s and 1980s (Doerr and Hill 2013). Historically, SARI provided a unique setting to study queen conch populations because of the presence of the NOAA National Undersea Research Program, original habitat Hydrolab and then the Aquarius, which ran from 1977–1989. These underwater habitat facilities lay at 18 m (60 ft) at the mouth of the submarine canyon providing location from which to dive and study relatively protected semi-enclosed deep-water habitat of the nearshore submarine canyon where Queen Conch were regularly located.

Data and Methods

All data used for historical assessments of conch populations in the SARI were obtained through literature review. Hurley et al. (1980) collected conch population information during saturation dives along transects on the Salt River wall. Conch were observed throughout the day and at varying depths. Coulston et al. (1987) tagged conch during saturation dives along the Salt River wall over the course of several months. After conch were tagged, divers searched the wall monthly over an area of approximately 28,770 m^2 and ranged from 15–30 m depth to record information about recaptured tagged conch. Additionally, hatchery-reared juvenile conch were placed in shallow water cages with varying degrees of protection within view of the Hydrolab underwater habitat and observed for predation. Surviving juveniles were tagged and recorded during recapture surveys when they appeared.

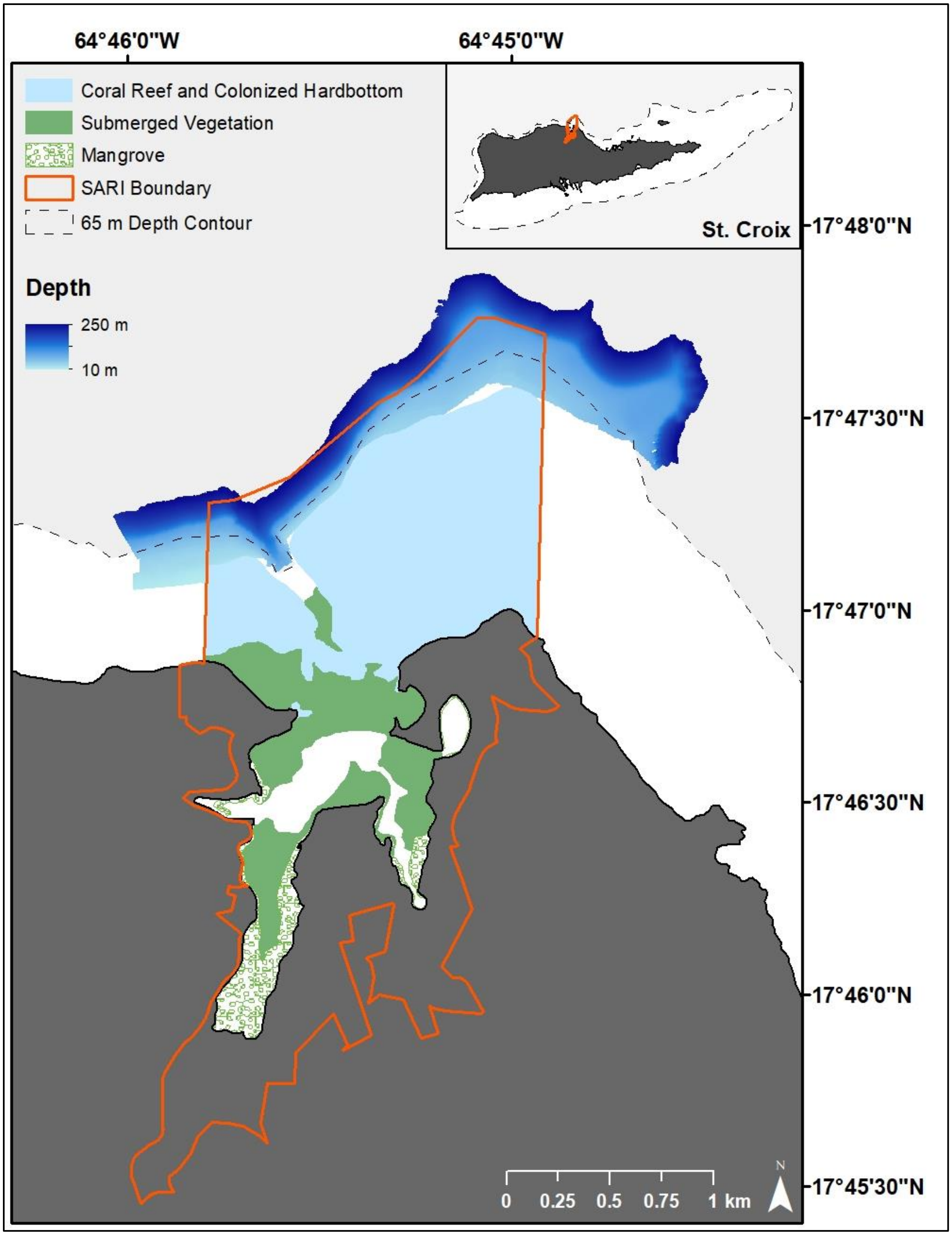

Figure 4.6.2.1. The Salt River Bay National Historic Park and Ecological Preserve (SARI) location and boundary (inset). Major benthic cover categories and shelf edge bathymetry within the SARI are displayed (source) (NOAA-NOS 2001).

The current condition of conch populations within the SARI was estimated from datasets provided by NOAA from the National Coral Reef Monitoring Program (NCRMP) and Doerr and Hill (2019 pers.

comm.). Since 2013, queen conch in the SARI have continued to be monitored biennially as part of the National Coral Reef Monitoring Program (NCRMP). Queen conch abundance is recorded on transects at randomly stratified locations throughout St. Croix. During NCRMP conch surveys, individuals are counted along 15 x 2 m belt transects stratified by depth, hardbottom habitat type, and management regime (i.e., inside or outside the SARI). Surveys range in depth from the surface to 30 m and were conducted throughout the insular shelf of St. Croix. Doerr and Hill conducted radial surveys in July 2017 at the inner entrance to the Salt River canyon.

Reference Conditions/Values

Early study of queen conch in the USVI focused primarily on biological aspects of the species (see Randall 1964; Berg 1975). However, queen conch has long been an important fishery throughout the USVI, and decreased catch and populations had been reported as early as the 1970s most likely due to overexploitation (Wood and Olsen 1983; Doerr and Hill 2013). Therefore, commercial fishing regulations were signed into law in 1972 (Virgin Islands Code), which were later amended to include minimum shell length and lip thickness, commercial and recreational take limits, and seasonal closures during spawning specifically for queen conch. Subsequent research shifted to focus on management actions to stabilize the fishery and the potential for fishery replenishment through juvenile outplanting (Wood and Olsen 1983; Coulston et al. 1987).

Hurley et al. (1980) provided one of the earliest studies of queen conch in the Salt River Canyon. Densities ranged from 50–1050 conch ha^{-1}, with conch more frequently observed between 27–30 m (Hurley et al. 1980). Additionally, conch size was found to be large in the Salt River Canyon. Several years later, Coulston et al. (1987) compared deeper water (15–30 m) conch populations to those in shallower areas more vulnerable to overexploitation and examined the viability of outplanting hatchery-reared juveniles to replenish stocks. They found the deep-water habitat provided by the canyon to be densely populated by reproductively active adults that were much older and larger than found elsewhere. However, juvenile conchs were rare and it was unclear if they were victims of predation or were perpetually buried in sand. Alternatively, the deeper habitat provided by the canyon is likely not ideal for juvenile queen conch or they are active at night when surveys were not conducted. Additionally, survival of hatchery-reared juveniles was very low unless given additional protection from predators. Therefore, they concluded that outplanting was not feasible without further investigation and that priority should be given to the identification and protection of densely populated deep-water habitat similar to the canyon walls in SARI (Coulston et al. 1987).

Current Condition and Trend

During the most recent NCRMP sampling (2017), no conch were recorded within the boundaries of the SARI down to a depth of 30 m (Table 4.6.2.1); however, the NCRMP is limited in its assessment of queen conch as surveys are only performed on hardbottom habitat and most likely underestimates conch populations. In addition, previous work has noted deep populations of mature queen conch in the canyon to 30 m depths and likely there are additional populations of conch below 30 m and the depths of NCRMP monitoring. Additionally, about 90% of all locations sampled around St. Croix by NCRMP, inside and outside protected areas, had no conch present. Doerr and Hill recorded queen conch densities of 116.7 conch/ha during their radial surveys in 2017. However, sample size was very limited due to both lack of conch detection and poor water visibility.

Table 4.6.2.1. Conch densities calculated from the National Coral Reef Monitoring Program sampling in 2017. Densities were calculated for the following management regimes in St. Croix: open (open area – territorial fishing regulations), BUIS (Buck Island National Park – no take zone), EEMP (St. Croix East End Marine Park – no take on inshore areas), and SARI (Salt River Bay National Historic Park and Ecological Preserve – no take zone).

Management Regime	Conch Density (#/ha) ± SEM
Open (n = 79)	97.0 ± 49.9
BUIS (n = 59)	56.5 ± 20.0
EEMP (n = 25)	40.0 ± 22.1
SARI (n = 12)	0.0 ± 0.0

Threats and Stressors

The largest threat to queen conch populations in the SARI is overfishing. SARI was designated a territorial marine reserve, but regulations were never implemented. Conch fishing within park boundaries is only regulated by VI fishing regulations for size and catch numbers; commercial harvest is conducted in the park. Additionally, destruction or loss of seagrass habitat in which queen conch spend the majority of their life could lead to population declines. There was a slight decrease in seagrass cover after Hurricane Hugo in 1989 and this has not recovered (Kendall et al. 2005). There are no apparent restrictions on anchoring inside the park boundaries, which could contribute to damage of seagrasses (Rogers and Beets 2001). Furthermore, development in the watershed could lead to excess sediment runoff and burial of seagrasses and macroalgal plains necessary to support queen conch populations (Kendall et al. 2005). Lastly, the invasion of the Indo-Pacific seagrass *Halophila stipulacea* in the USVI (Willette et al. 2014) could have unknown consequences on queen conch. This invasive seagrass can displace native seagrass (Willette and Ambrose 2012) and invasive populations may change the ecology of queen conch (Becking et al. 2014). Queen conch do not avoid meadows of *H. stipulacea* (Becking et al. 2014), but it is not clear if they derive the same nutritional benefit from consuming *H. stipulacea* and its epiphytes.

Data Needs and Gaps

Historical data regarding conch populations within the SARI appears to be relatively extensive and benefited greatly from the research conducted from NOAA NURP Hydrolab and Aquarius underwater habitats. However, current data is severely lacking. Although the NCRMP surveys within the SARI every two years, monitoring focuses on hardbottom habitats, and conch populations are likely severely underestimated as evidenced by the lack of conch recorded within the SARI during the most recent survey year. In general, SARI is in need of a baseline ecological assessment for many organisms (Kendall et al. 2005). Therefore, future monitoring should include surveys in seagrass and other conch habitats to provide a more accurate current baseline of the population.

Overall Condition

Current conch population studies within the SARI are severely lacking. The estimate recorded by the NCRMP dataset (0 conch surveyed) cannot provide an accurate assessment of the population as all surveys were conducted on habitat rarely frequented by conch. Therefore, the current condition of conch populations within the SARI is relatively unknown and confidence in the assessment is low (Table 4.6.2.2). Observations during other research do not suggest numbers as high as 1000 conch ha^{-1} as was seen in the 1970s (T. Smith, unpub. obs.), but this needs to be verified.

Table 4.6.2.2. Graphical summary of status and trends for queen conch within the framework category of marine invertebrates including rationale and reference condition.

Component	Indicator	Condition Status /Trend	Rationale and Reference Conditions
Queen Conch	Abundance		No trend can be established because of lack of suitable data on soft bottom habitats; however, observation would suggest the population has declined since the 1970s.

Source(s) of Expertise:

- Jennifer Doerr. Research Fishery Biologist, Galveston Laboratory of the Southeast Fisheries Science Center, National Oceanic and Atmospheric Administration

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4.7. Marine Vertebrates

4.7.1. Reef Fish

Description

In the USVI, reef fish comprise a critical economic, ecological, and cultural resource (Bryan et al. 2013). Salt River National Historical Park and Ecological Preserve (SARI) has offered fish and other marine resources protection since its creation in 1992. Since 1995, the collection of marine resources of any kind is unlawful within the park (DPNR 2018). SARI encompasses mangroves, seagrass beds, and different types of coral reef habitats (e.g., deep reef walls and shallow aggregated patch reefs) that provide a wide variety of nursery and feeding habitats for multiple fish species (National Park Service 2015).

Most studies on reef fish in the park have been conducted within Salt River Canyon along the canyon walls during saturation missions from NOAA's National Undersea Research Program Hydrolab and Aquarius missions in 1977 and 1985. Kendall et al. (2005) compiled a list of nearly 200 reef fishes observed in SARI's coral reefs and pointed out the need to study the rest of the park's shallow reefs. From 2012 to 2019, several surveys conducted by National Park Service (NPS), National Oceanic Atmospheric Administration, and the University of Virgin Islands (UVI), here referred to as National Coral Reef Monitoring Program (NPS-NCRMP-UVI), have produced enough data for this first attempt to characterize the status of reef fish in SARI. Our analysis focuses on three parameters, density, biomass, and richness. Years covered by the datasets considered in this analysis include the following: 2012 (data provided by Jeremiah Blondeau, NOAA) and 2015, 2017, and 2019 (NCCOS 2018).

Data and Methods

Surveys used in this report were conducted on hardbottom habitats, including aggregated reef (AGRF), bedrock (BDRK), hardbottom (HARD), patch reef (PTRF), pavement (PVMT), and scattered coral/rock (SCR) between 2012–2019 using two different methodologies (Table 4.7.1.1). Data sets are available from the NOAA National Centers for Environmental Information at https://data.noaa.gov/datasetsearch/. Surveys in 2012 and 2015 were carried out along 25 m x 4 m belt transects (100 m^2). During each survey, the number of individuals by species and length were recorded from which we obtain density (Ind. 100 m^2) and richness (the number of species). Fish surveys conducted in 2017 and 2019 followed Reef Visual Census (RVC; Bohnsack and Bannerot 1986; Bryan et al. 2013) within a 15 m diameter imaginary cylinder (~177 m^2). The method differs from the belt transect in several aspects, including stationary counts, rather than counts along the transect, and fish parameter collection (first round species list and later number of individuals and length). Fish density for 2017 and 2019 is expressed as the number of individuals per sampling unit. Data (individual fish length) from both methods were used to estimate individual weight using weight (W) length (L) relationships ($W=aL^b$, "a" and "b" are species-specific morphometric coefficients) (Bohnsack and Harper 1988; Stevens et al. 2019). In a few cases (less than 1% of individuals), equations from similar species (e.g., *Hypoplectrus* sp.) were used. Biomass (g 100 m^2) was calculated using individual weights by sampling area for belt transect. Biomass for 2017 and

2019 surveys is expressed as g per sampling unit. Given the methodological differences between the two data sets, all graphical and statistical analyses are separated from 2012–2015 and 2017–2019.

Table 4.7.1.1. Number of surveys conducted in SARI by year and method from 2012 to 2019.

Year	Method	Number of Surveys
2012	Belt transect	17
2015	Belt transect	12
2017	RVC	13
2019	RVC	15

Density and biomass were also analyzed by trophic level: (H = herbivore, I = invertivore, Pl = planktivore, P = piscivore). Herbivore included all species of scarids (family Scaridae), acanthurids (family Acanthuridae), and other species such as the Bermuda chub (*Kyphosus sectratix*). Invertivores comprised many reef fishes within families Haemulidae, Lutjanidae, and Pomacanthidae, whereas fewer planktivorous species included the blue chromis (*Chromis cyanea*) and creole wrasse (*Clepticus parrae*). Piscivores contained large and medium-sized predator species such as barracuda (*Sphyraena barracuda*), multiple species of serranids (family Serranidae), and jack (family Carangidae).

For statistical reasons, large and mobile shark observations (family Carcharinidae and Ginglymostomatidae) were removed from the analysis. Similarly, herrings (*Jenkinsia* spp.) that form large fish schools were not considered because it skews density data distributions. We used one-way ANOVA to compare data sets collected with the same methodology (2012–2015 and 2017–2019). Dispersion in all graphs and text descriptions is expressed as standard error.

<u>Reference Conditions/Values</u>

SARI is a relatively small park (~ 1015 acres) encompassing several essential terrestrial, estuarine and marine habitats such as mangrove forest and seagrass beds used by reef fish at multiple life stages. Mangrove and seagrass have suffered a 50% and 13% decline, respectively (Kendall et al. 2005). The loss of these critical habitats may have had impacted the reef fish community before reef fish monitoring programs began. The earliest reference for fish communities in the park dates back to 1994 (Kaufman and Ebersole 1984; Adams and Tobias 1994). The study focused on the fish community associated with red mangrove prop-roots. They observed 40 species and 19 families in fish traps, 48 species and 20 families in transects, and a total of 57 species (primarily juveniles) across sampling methods between 1990 and 1993. Snappers (family Lutjanidae) and grunts (family Haemulidae) were reported as the most abundant in the mangrove lagoon. In 2011, Dorfman and Battista performed a gap analysis of ecosystem data and found that SARI does not have regular fish surveys. The UVI-CMES reef fish census provided limited available data. Salt River Canyon, including the fauna along the walls, was extensively studied during the commission of NOAA's Hydrolab facility from 1977 to 1989. Kaufman and Ebersole (1984) reported approximately 108 species for Salt River Canyon, which constitutes the oldest data for reef fish in the park. Fish surveys conducted in 1998–2000 (10 m deep) reported 117 fish species and estimated densities of surgeonfish (14.5 ± 8.5 Ind. $100m^{-2}$), parrotfish (10.0 ± 6.0 Ind. $100m^{-2}$), and other species in SARI (Nemeth et al. 2003).

<u>Current Condition and Trend</u>

Compared to 2012 (352.9 ± 59.4 Ind. $100m^{-2}$) the average total density of fish decreased by half by 2015 with 143.3 ± 14.4 Ind. $100m^{-2}$ (Figure 4.7.1.1, F = 8.46, p = 0.007). No changes in total fish density were observed from 2017–2019 surveys (Figure 4.7.1.1, F = 0.005, p = 0.946). Similarly, total fish biomass in 2012 (9151.7 ± 1905.0 g. 100 m^{-2}) was more than double the 2015 total fish biomass, 3626.8 ± 726.3 g. 100 m^{-2} (Figure 4.7.1.1, F = 5.49, p = 0.027). No changes were observed in 2017–2019 surveys when total fish biomass averaged 9694.3 ± 1998.8 g. $100m^{-2}$ (Figure 4.7.1.1, F = 1.90, p = 0.180).

In 2012, approximately 21 species were observed per survey, six species more than in 2015 (Figure 4.7.1.1, ANOVA, F = 9.58, p = 0.005). The average number of species recorded per survey 2017–2019 was 31, with no differences between years (Figure 4.7.1.1, F = 1.91, p = 0.178). Notice that the number of species in 2012 is half of 2017–2019, but the latter was recorded using RVC. We cannot compare this value to our references (Nemeth et al. 2003, 117 species) because the authors used a 30-minute roving diver survey following the AGRRA methodology.

Density of herbivorous fish also decreased from 2012 (33.3 ± 4.9 Ind. $100m^{-2}$) to 2015 with 15.3 ± 3.3 Ind. $100m^{-2}$ (Figure 4.7.1.2, F = 7.59, p = 0.010). Density of 2015 is qualitatively lower than pooled herbivorous fish density (~ 25 Ind. $100m^{-2}$) reported by Nemeth et al. 2003 (density of acanthurids 14.5 ± 8.5 Ind. $100m^{-2}$, density of scarids 10.0 ± 6.0 Ind. $100m^{-2}$, density of *Microspathodon chrysurus* 0.4 ± 1.0 Ind. $100m^{-2}$). While these data sets are not directly comparable because of 2012–2015 (25 x 4 m) belt transect surveys that included all species and Nemeth et al. (2003) followed AGRAA (30 x 2 m) belt transect with selected species, we would expect higher density of herbivorous fish in 2012. We also observed over 50% decline in herbivorous fish density between 2017 and 2019 (Figure 4.7.1.2, F = 10.68, p = 0.003). Fish density of other trophic groups, invertivore, planktivore, and piscivore, indicated no changes between 2012–2015 or 2017–2019 (Figure 4.7.1.2 C–H).

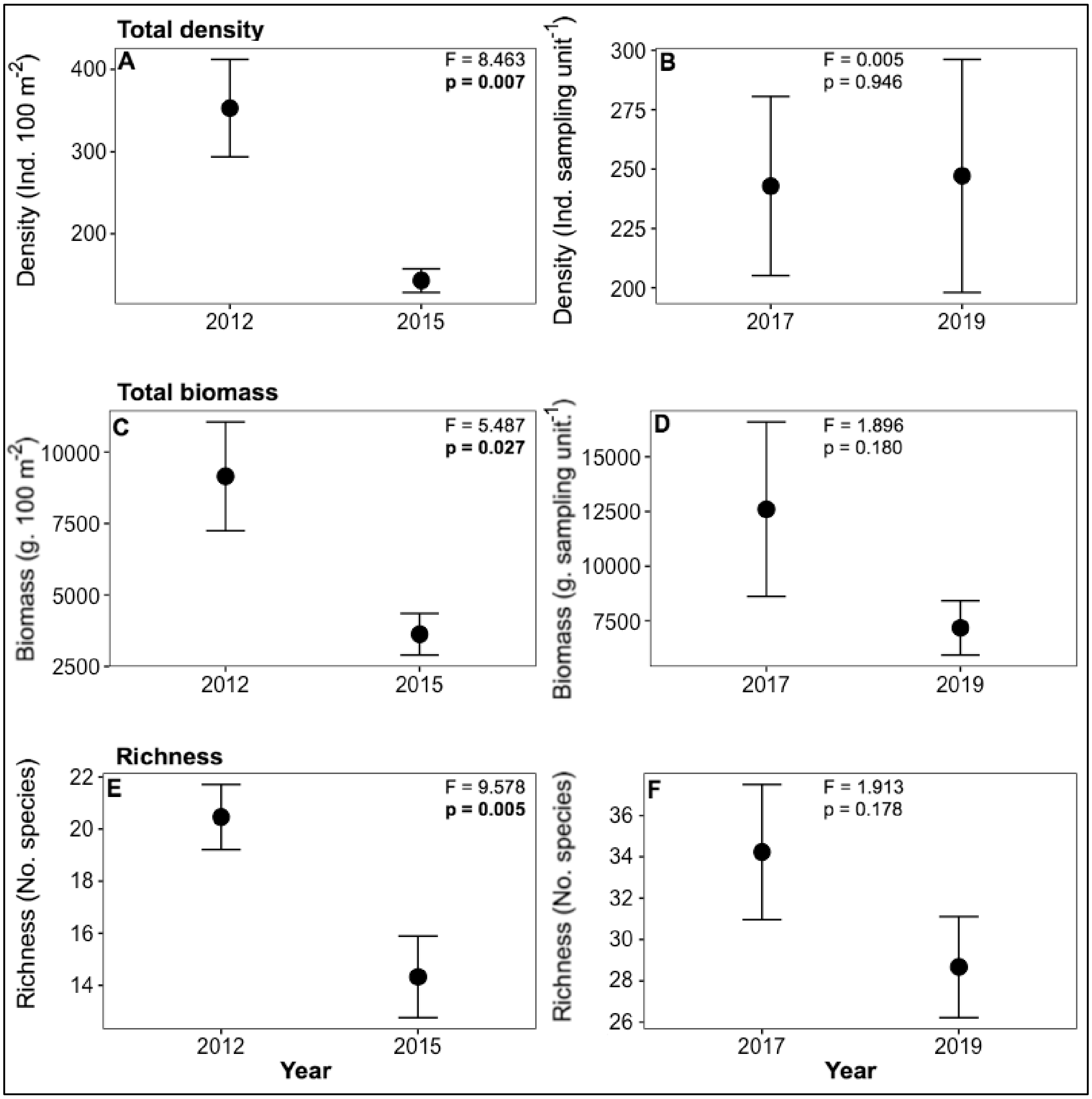


Figure 4.7.1.1. Total fish density, total fish biomass, and reef fish richness in Salt River National Historical Park and Ecological Preserve (SARI) from 2012 to 2019. Surveys from 2012–2015 were conducted using belt transects, while surveys in 2017 and 2019 used Reef Visual Survey (stationary point count conducted within 15 m diameter). Mean ± SE. Bold letters indicate statistical significance. Data source: NPS-NCRMP-UVI program.

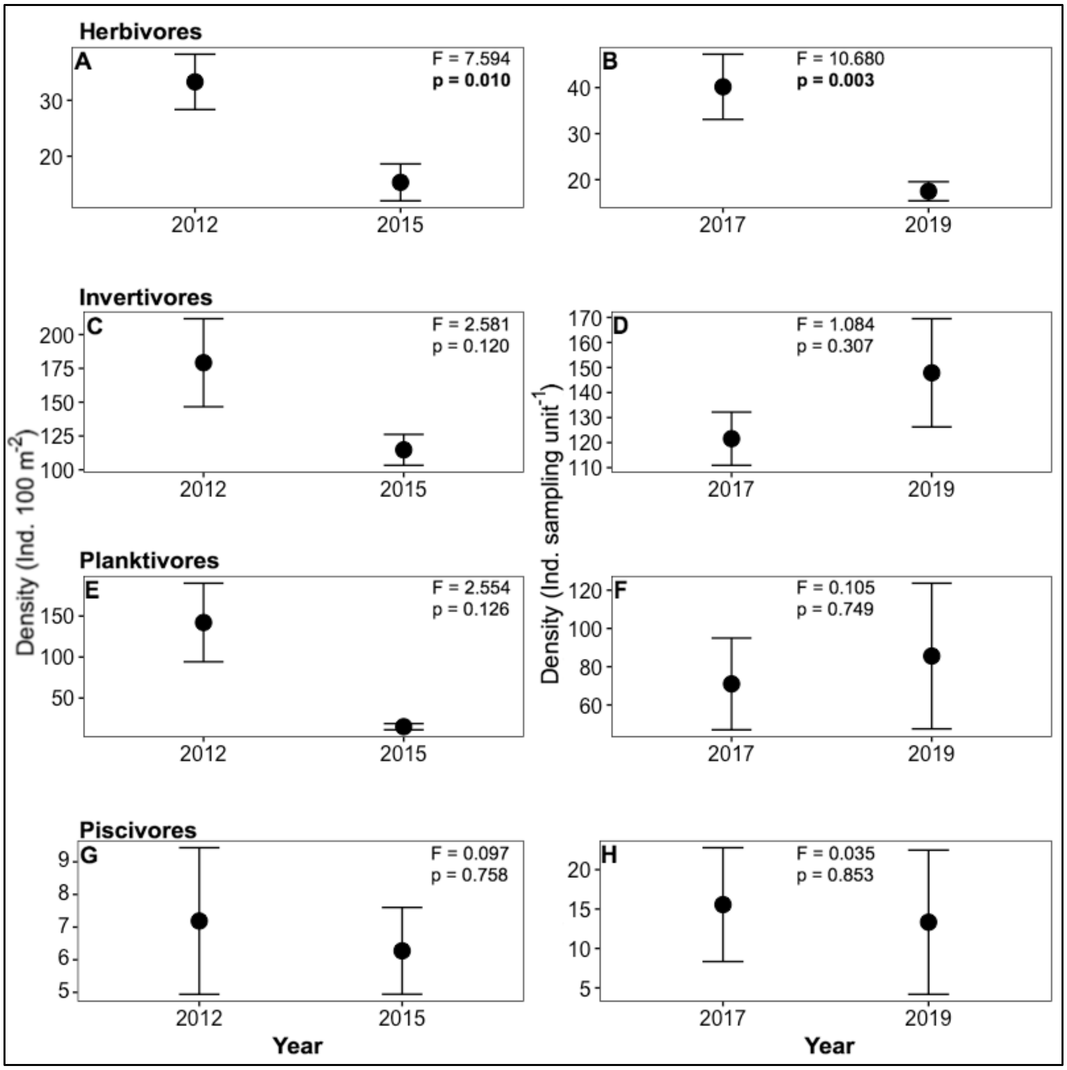


Figure 4.7.1.2. Fish density by trophic groups in Salt River National Historical Park and Ecological Preserve (SARI) from 2012 to 2019. Surveys from 2012–2015 were conducted using belt transect, while surveys in 2017 and 2019 used Reef Visual Survey (stationary point count conducted within 15 m diameter). Mean ± SE. Bold letters indicate statistical significance. Data source: NPS-NCRMP-UVI program.

Biomass of herbivorous fish also fell more than one third from 2012 (4111.4 ± 868.9 g. 100m^{-2}) to 2015 with 1153.7 ± 402.5 g. 100m^{-2} (Figure 4.7.1.3A, F = 7.33, p = 0.012). Biomass of both parrotfish and surgeonfish declined more than half between 2012 and 2015 (parrotfish, F = 5.78, p = 0.024, surgeonfish, F = 4.98, p = 0.034). Two herbivore species contributed to this decrease, redband parrotfish (*Sparisoma aurofrenatum*) and ocean surgeonfish (*Acanthurus bahianus)*. Biomass of ocean surgeonfish decreased from 332.0 ± 52.7 g. 100m^{-2} in 2012 to 159.4 ± 30.3 g. 100m^{-2} in 2015 (Figure 4.7.1.4, F = 6.73, p = 0.015). Biomass of redband parrotfish went from 179.2 ± 31.1 g. 100 m^{-2} in 2012 to 74.9 ± 30.7 g. 100m^{-2} in 2015 (Figure 4.7.1.5, F = 4.98, p = 0.036). Herbivorous biomass averaged 5576.3 ± 2097.9 g. 100m^{-2} in 2017 and 2791.8 ± 705.6 g. 100m^{-2} in 2019, but no statistical differences were observed (Figure 4.7.1.3 B). Notice the high variation (SE) in 2012, which could have masked these results' ecological significance. Biomass of other trophic groups showed no changes in either survey period. To illustrate the spatial distribution of reef fish in SARI, we created two maps with the most recent monitoring data collected in 2017 and 2019. There are not clear spatial patterns of total fish density (Figure 4.7.1.6) and total fish biomass (Figure 4.7.1.7), but further analysis is needed to investigate spatial distribution.

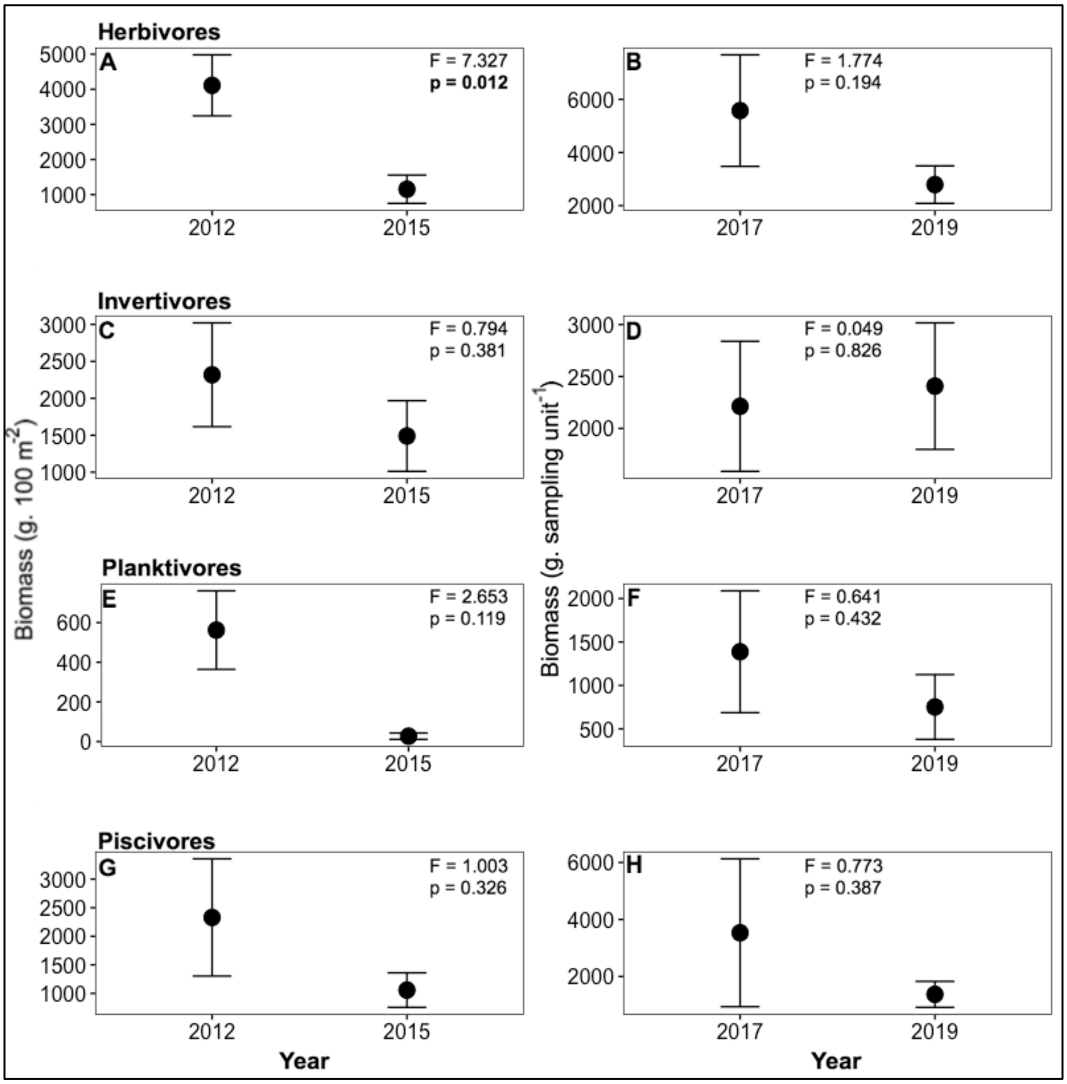


Figure 4.7.1.3. Fish biomass by trophic groups in Salt River National Historical Park and Ecological Preserve (SARI) from 2012 to 2019. Surveys from 2012–2015 were conducted using belt transect, while surveys in 2017 and 2019 used Reef Visual Survey (stationary point count conducted within 15 m diameter). Mean ± SE. Bold letters indicate statistical significance. Data source: NPS-NCRMP-UVI program.

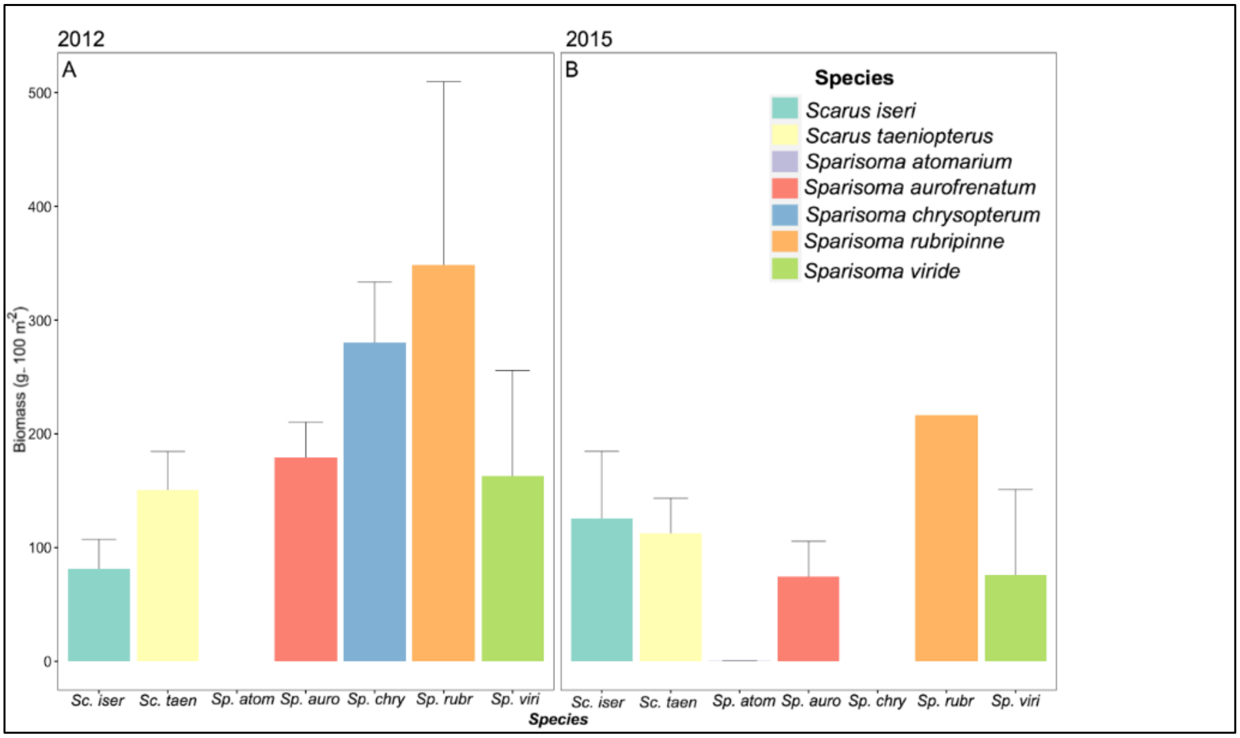


Figure 4.7.1.4. Fish biomass of scarids (family Scaridae) in Salt River National Historical Park and Ecological Preserve (SARI) from 2012 to 2019. Surveys from 2012–2015 were conducted using belt transect, while surveys in 2017 and 2019 used Reef Visual Survey (stationary point count conducted within 15 m diameter). Data source: NPS-NCRMP-UVI program.

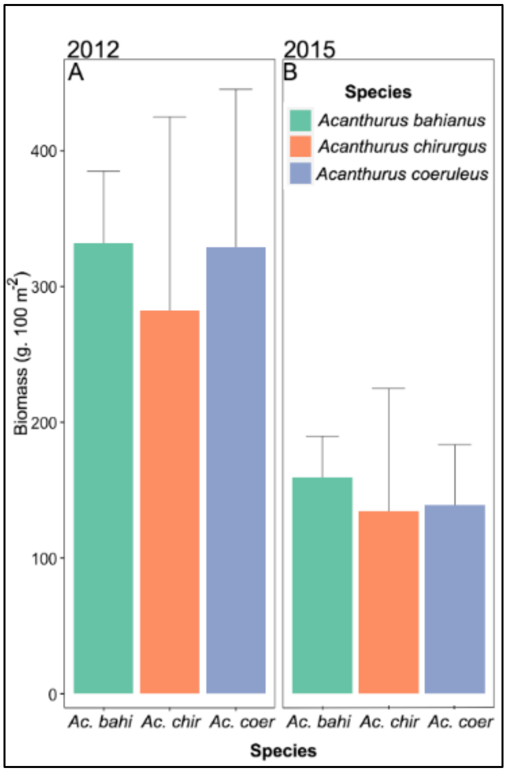


Figure 4.7.1.5. Fish biomass of acanthurids (family Acanthuridae) in Salt River National Historical Park and Ecological Preserve (SARI) from 2012 to 2019. Surveys from 2012–2015 were conducted using belt transect, while surveys in 2017 and 2019 used Reef Visual Survey (stationary point count conducted within 15 m diameter). Data source: NPS-NCRMP-UVI program.

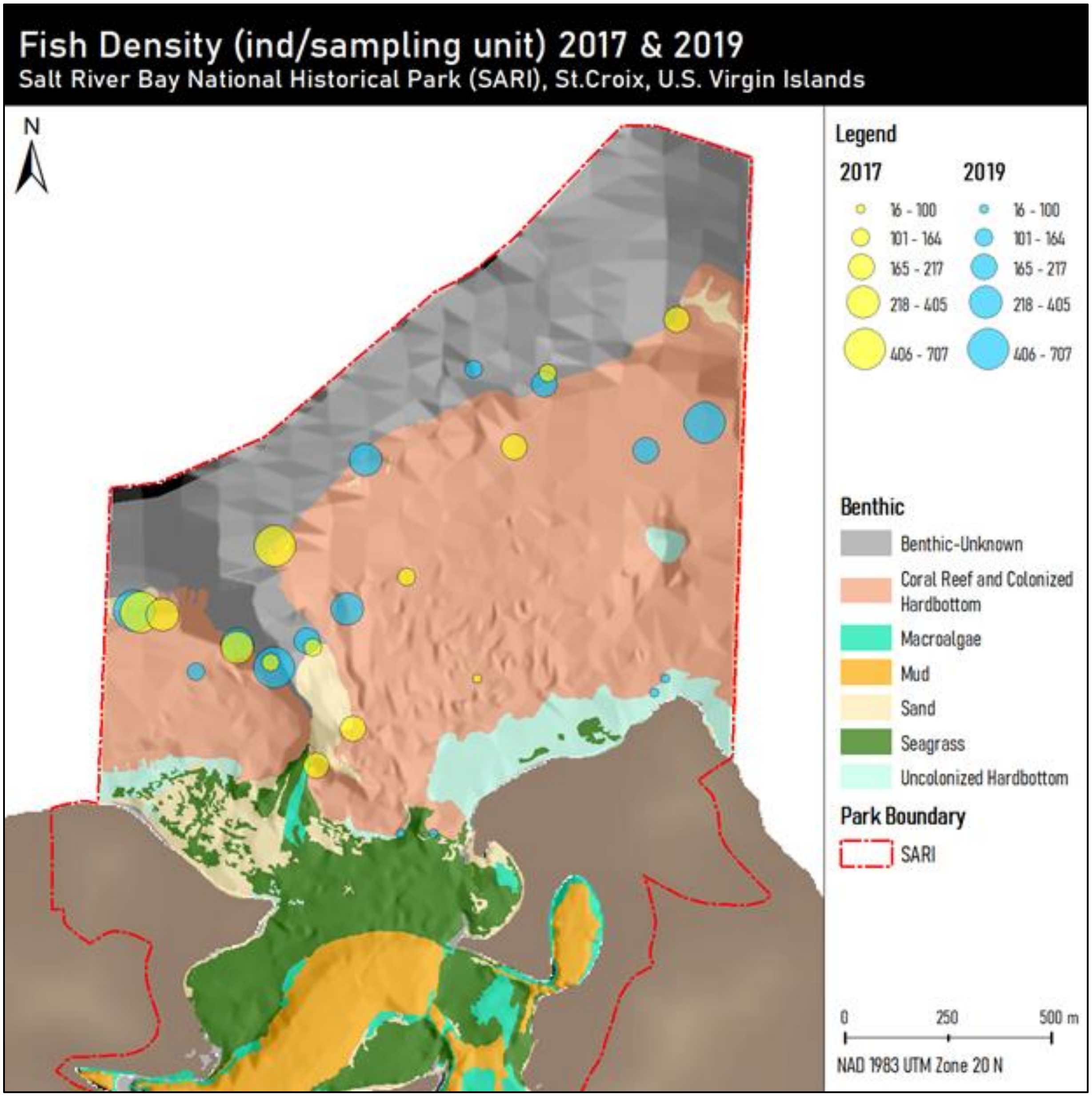


Figure 4.7.1.6. Total fish density (Ind. sampling unit^{-1}) estimated from 2017 (yellow circles) and 2019 (blue circles) surveys conducted in Salt River National Historical Park and Ecological Preserve (SARI). Data source: NPS-NCRMP-UVI program. Habitat cover obtained from Kendall et al. (2005).

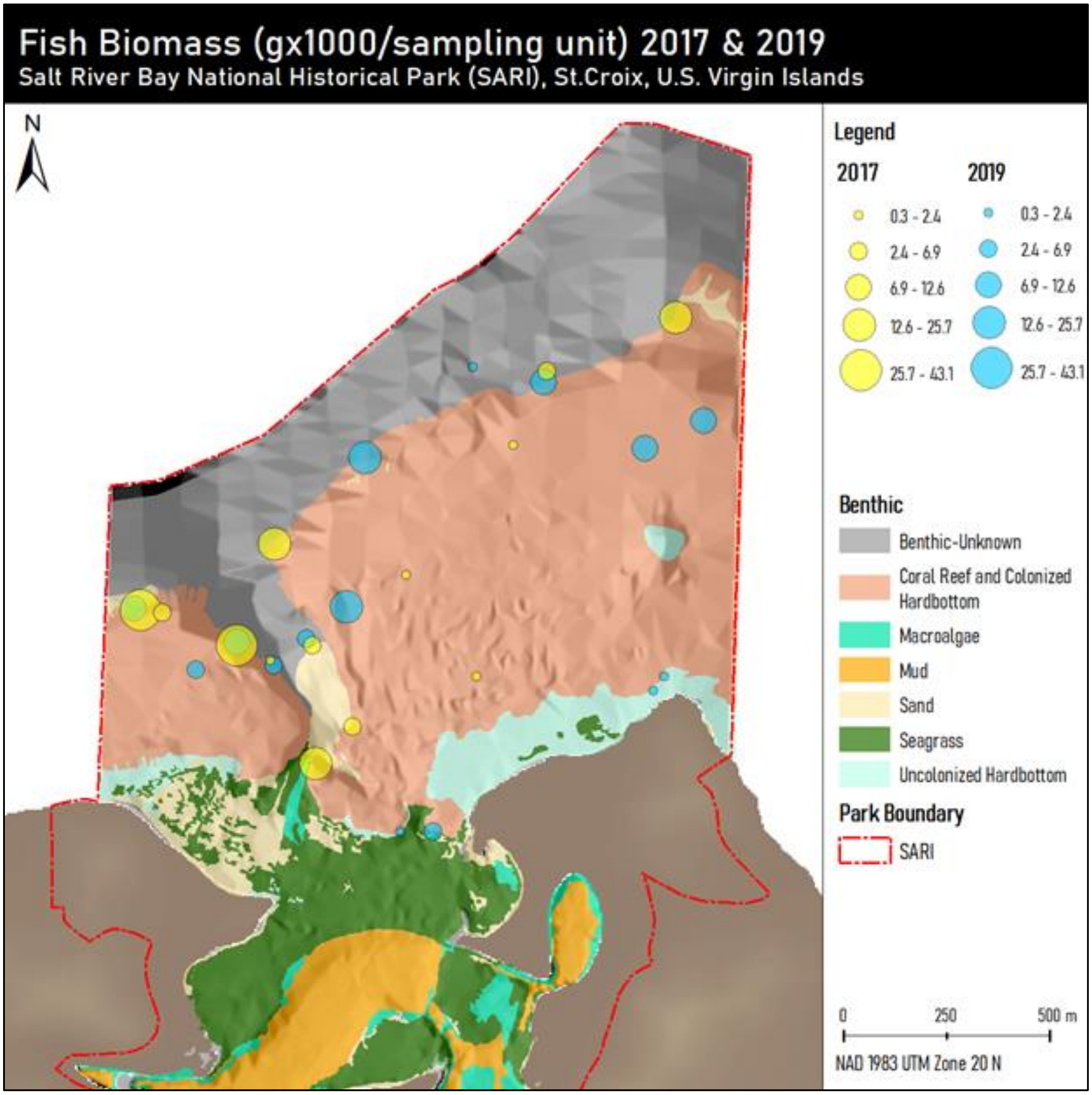


Figure 4.7.1.7. Total fish biomass (g. sampling unit^{-1}) estimated from 2017 (yellow circles) and 2019 (blue circles) surveys conducted in Salt River National Historical Park and Ecological Preserve (SARI). Data source: NPS-NCRMP-UVI program. Habitat cover obtained from Kendall et al. (2005).

Threats and Stressors

Reef fish within SARI face a myriad of anthropogenic threats and stressors, including illegal/over-harvest, habitat and water quality degradation, and non-native species introduction (lionfish). All fish community metrics, density, biomass, and richness showed a significant decline between 2012–2015, likely as the result of the stressors mentioned above. The absence of large reef species such as groupers, large parrotfishes, and snappers indicates continual illegal fishing pressure in the park. Our conclusions are still speculative, given the lack of continuous fish monitoring data. Lionfish also pose a threat, but only three individuals have been recorded since 2012. However, anecdotal information indicates the species is present.

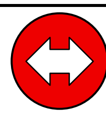
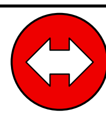
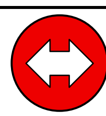
Data Needs and Gaps

SARI is a relatively small park with high habitat diversity, including large nursery habitats for fish. Surveys exist, but the park needs more frequent standardized monitoring. NPS staff and researchers could focus efforts on reef fish to better understand changes in diversity and biomass to manage the resource better. Additionally, a monitoring program could encompass sites in different habitats to understand the biological and abiotic connection better. As in other parks recently monitored using RVC, there is an urgent need to carry out cross-validation studies that allow data comparison before and after 2015. Such a study is currently underway with funding from NOAA NMFS (M. Feeley 2021, personal communication). A first approach could be standardizing fish density and biomass given the survey surface area (belt transect 100 m^2 vs. RVC 15 m diameter), considering that RVC produces more accurate metric estimates (Colvocoresses and Acosta 2007).

Overall Condition

Both Salt River Canyon and Salt River Bay provide critical habitat to reef fish at multiple life stages. Reef fish communities throughout SARI have been impacted by the effects of various anthropogenic stressors with no signs of recovery currently. Reef fish communities warrant significant concern because of their susceptibility to changes to their habitat and fishing pressure (Table 4.7.1.2). No trend in condition was detected over the time of monitoring data available.

Table 4.7.1.2. Graphical summary of status and trends for reef fish within the framework category of marine vertebrates including rationale and reference condition.

Component	Indicator	Condition Status /Trend	Rationale and Reference Conditions
Reef fish	Density		Reef fish density warrants significant concern because of the negative trend between 2012–2015. No changes in 2017–2019
	Biomass		Reef fish biomass warrants significant concern because of the negative trend between 2012–2015. No changes in 2017–2019
	Richness		Reef fish richness warrants significant concern because of the negative trend between 2012–2015. No changes in 2017–2019

Source(s) of Expertise

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- Christy Pattengill-Semmens, Ph.D, Director of Science, Reef Environmental Education Foundation (www.reef.org)

- Judd Patterson, Acting National Data Manager, National Park Service South Florida / Caribbean Network, judd_patterson@nps.gov
- Matt Kendall, Researcher, National Oceanic Atmospheric Administration, Marine Spatial Ecology Division, matt.kendall@noaa.gov

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Chapter 5. Discussion

5.1 Reporting Category Condition Summaries

Resource condition summaries for each focal resource assessed in chapter 4, along with the indicators used in each, are presented in Tables 5.1.1 to 5.1.11. These include focal resources pertaining to the supporting environment of SARI, specifically shoreline dynamics, water quality, with inside and outside Salt River Bay considered separately, and watershed condition (Tables 5.1.1 to 5.1.4), as well as focal resources falling within the framework category of biological integrity, including mangroves, semi-deciduous dry forest, coastal grassland, macroalgae, seagrass, corals, conch, and reef fish (Tables 5.1.5 to 5.1.12). We present an overall summary of all focal resources in Table 5.1.13. The overall summary table provides an overview of the condition, trend, and confidence in the assessment of all focal resources in a single table. Unless otherwise stated, we follow the methods for combining status and trends for individual indicators as outlined in the NPS-NRCA Guidance Update from January 20, 2014.

Table 5.1.1. Indicator summary for Shoreline Dynamics focal resource.

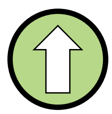
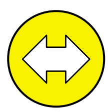
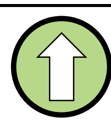
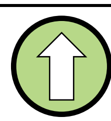
Indicators of Condition	**Measures or Criteria**	**Condition Status /Trend**	**Rationale**
Shoreline Change	Shoreline length change		Since 1954, the shoreline extent has increased significantly as result of sediment deposition and dredging for the marina.
Shoreline Change	Shoreline area change		Reduction of area since 1954 as a result of the dredging for the marina has been slightly outpaced by the increase in area accrued because of sediment deposition.
Shoreline Change	Shore habitat change		Sandy/gravel shoreline area and extent of the vegetated shoreline have increased steadily since 1954, and rocky shorelines have experienced little change.
Shoreline Dynamics Overall	–		–

Table 5.1.2. Indicator summary for Water Quality (Outside Salt River Bay) focal resource.

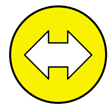
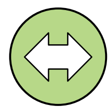
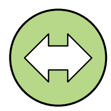
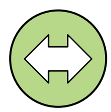
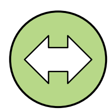
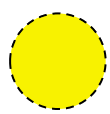
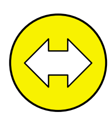
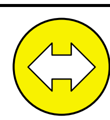
Indicators of Condition	**Measures or Criteria**	**Condition Status /Trend**	**Rationale**
Fecal Indicator Bacteria	USVI 2019 Amended Water Quality Standards Rules and Regulations		There are indications of nutrient pollution from sewage and impacts related to storm water discharge
Dissolved Oxygen	USVI 2019 Amended Water Quality Standards Rules and Regulations		Oceanic influenced areas have consistently good values. There do not appear to be strong trends in values over time.
Total Suspended Solids	NA		Total suspended solids are low in more oceanic areas. There do not appear to be strong trends in values over time.
Turbidity	USVI 2019 Amended Water Quality Standards Rules and Regulations		Oceanic influenced areas have consistently good values. There do not appear to be strong trends in values over time.
Dissolved Nutrients	NA		These are typically near detection limits in most areas. However, they may be a poor metric of nutrient loading.
Chlorophyll	Enrichment above oligotrophic oceanic conditions		Chlorophyll concentrations have not been assessed directly but would provide a useful proxy for nutrient loading. Observations of water color and clarity at offshore coral reef sites do not suggest high chlorophyll levels, but levels are likely to be much higher inshore.
Terrestrial Sediments	Annual number of events associated with high rainfall		Terrestrial sediments have only been indirectly measured at one coral reef location and were low. There are indications of pollutants in some coral reef associated sediments
Water Quality Overall	–		–

Table 5.1.3. Indicator summary for Water Quality (Inside Salt River Bay) focal resource.

Indicators of Condition	Measures or Criteria	Condition Status /Trend	Rationale
Fecal Indicator Bacteria	USVI 2019 Amended Water Quality Standards Rules and Regulations		There are indications of fecal contamination for some sites that periodically exceed values considered a risk for human contact. Continued development and poor enforcement of septic discharge may contribute to increasing incidences of fecal contamination.
Dissolved Oxygen	USVI 2019 Amended Water Quality Standards Rules and Regulations		Values in some constricted, low water exchange areas are consistently below values needed for support of marine life. There do not appear to be strong trends in values over time.
Total Suspended Solids	NA		Total suspended can be high inside embayments. There do not appear to be strong trends in values over time.
Turbidity	USVI 2019 Amended Water Quality Standards Rules and Regulations		Values in some constricted, low exchange areas are high and potentially harmful to photosynthetic organisms (e.g., foraminifera). There do not appear to be strong trends in values over time.
Dissolved Nutrients	NA		These are typically near detection limits in most areas. However, they may be a poor metric of nutrient loading.
Chlorophyll	Enrichment above oligotrophic oceanic conditions		Chlorophyll concentrations have not been assessed directly but would provide a useful proxy for nutrient loading. Levels are likely high inside Salt River Bay.
Terrestrial Sediments	Annual number of events associated with high rainfall		There are indications of contaminants in some sediments and toxic effects on organisms. Effects of terrestrial sediments have not been evaluated inshore.
Water Quality Overall	–		–

Table 5.1.4. Indicator summary for Watershed Condition focal resource.

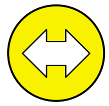
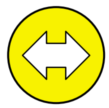
Indicators of Condition	**Measures or Criteria**	**Condition Status /Trend**	**Rationale**
Landover / Land use Change	Landover / Land use change		The condition of the watershed over the analysis period has not significantly changed in terms of land use/cover change. Notwithstanding the potential increase of the level of anthropogenic development intensity, warrants for moderate concern of the resource condition.
Watershed Condition Overall	–		–

Table 5.1.5. Indicator summary for Mangrove focal resource.

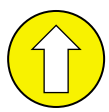
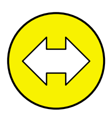
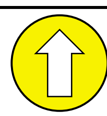
Indicators of Condition	**Measures or Criteria**	**Condition Status /Trend**	**Rationale**
Vegetation Community Extent	Percent Cover		Change in cover is moving in a positive direction after extensive mortality resulting from Hurricane Hugo. Continued expansion of mangrove into mudflat and into the terrestrial freshwater marsh is expected, provided another strong hurricane doesn't impact the park and anthropogenic impacts are minimized. The multitude of threats to this community designate the resource as of moderate concern.
Vegetation Community Extent	Species composition		While changes in species composition have occurred over the time period largely related to disturbance from Hurricane Hugo, the overall assemblage of species present has not changed. It is not clear if the mangrove associate, seaside mahoe, *T. populnea*, has increased over time, but presence of non-native invasive species should continue to be monitored. Similarly, increases in the cover of leather fern in the understory should be monitored since the species can suppress recruitment of mangrove seedlings.
Mangrove Forest Overall	–		–

Table 5.1.6. Indicator summary for Semi-Deciduous Dry Forest focal resource.

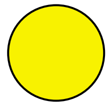
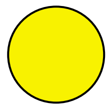
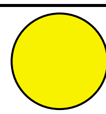
Indicators of Condition	Measures or Criteria	Condition Status /Trend	Rationale
Vegetation Community Extent	Percent Cover		Non-native invasive species are widespread throughout the component. It was not possible to assess any change in the indicator from the available data.
Vegetation Community Extent	Species Composition		Composition of forests include both native and non-native invasive species and the dominance of either varies throughout the component. It was not possible to assess any change in the indicator from the available data.
Tropical Dry Forest Overall	**–**		**–**

Table 5.1.7. Indicator summary for Coastal grassland focal resource.

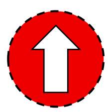
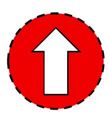
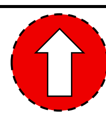
Indicators of Condition	Measures or Criteria	Condition Status /Trend	Rationale
Vegetation Community Extent	Percent Cover		The percent cover of non-native invasive species in this component has likely declined since management began in 2012. Although not quantified, the relative cover by native species has likely increased as a result. However, continued management will be necessary to maintain reduced cover of invasive species.
Vegetation Community Extent	Species Composition		Composition of the component was dominated by non-native invasive species prior to the start of management action and the introduction of native trees into the mixed dry grassland has increased the richness of native species. However, continued management and potentially future reforestation events will be needed to maintain the increase in species richness.
Coastal Grassland Overall	**–**		**–**

Table 5.1.8. Indicator summary for Macroalgae focal resource.

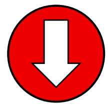
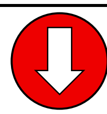
Indicators of Condition	**Measures or Criteria**	**Condition Status /Trend**	**Rationale**
Macroalgae Community Extent	Percent Cover		Though macroalgae cover estimates have only been documented since 2015, percent cover is high and indicative of poor ecosystem health. The current spread of *Ramicrusta* sp., an aggressive coral competitor is also a serious concern.
Macroalgae Overall	–		–

Table 5.1.9. Indicator summary for Seagrass focal resource.

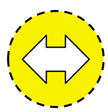
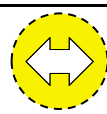
Indicators of Condition	**Measures or Criteria**	**Condition Status /Trend**	**Rationale**
Seagrass Community Extent	Percent Cover		Aerial photographs from the 1970s depict little change in cover through 2005, however, the spread of the invasive seagrass *Halophila stipulacea* in nearby areas is a serious concern. Low confidence in assessment is due to the lack of recent seagrass cover estimates, lack of in-water surveys to ground truth aerial estimations, different methods for creating mapping polygons in 2005, and lack of trend data in seagrass coverage within Salt River Canyon.
Seagrass Overall	–		–

Table 5.1.10. Indicator summary for Coral focal resource.

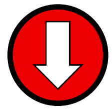
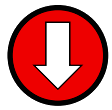
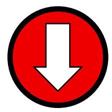
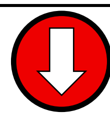
Indicators of Condition	**Measures or Criteria**	**Condition Status /Trend**	**Rationale**
Stony coral coverage	Percent of benthic cover		Coral reefs of SARI have declined in abundance compared to historical levels and have shown damage from hurricanes and thermal stress from surface to 30 m depth, and lack of sufficient recovery.
Stony coral health	Percent coral bleaching and incidence of disease		The incidence of coral bleaching events and coral disease epizootics has increased and is likely to continue increasing in the near future (e.g., introduction of Stony Coral Rapid Tissue Loss Disease)
Seawater temperature	Number of degree heating weeks above bleaching threshold		In 2005, heat stress likely surpassed 9 Degree Heating Weeks (DHW) while in 2010, another shallow water stress event occurred with DHW of 7. Extensive impact to coral but little mortality.
Corals Overall	–		–

Table 5.1.11. Indicator summary for Queen Conch focal resource.

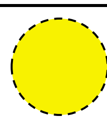
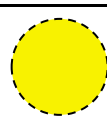
Indicators of Condition	**Measures or Criteria**	**Condition Status /Trend**	**Rationale**
Abundance	Density		No trend can be established because of lack of suitable data on soft bottom habitats. However, observation would suggest the population has declined since the 1970s.
Queen Conch Overall	–		–

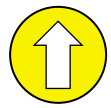
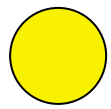
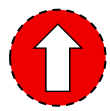
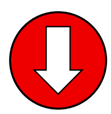
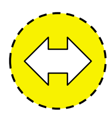
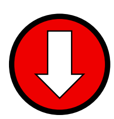
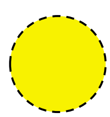
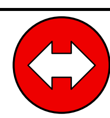
Table 5.1.12. Indicator summary for Reef Fish focal resource.

Indicators of Condition	**Measures or Criteria**	**Condition Status /Trend**	**Rationale**
Community and population status	Density		Reef fish density warrants significant concern because of the negative trend between 2012–2015. No changes in 2017–2019
Community and population status	Biomass		Reef fish biomass warrants significant concern because of the negative trend between 2012–2015. No changes in 2017–2019
Community and population status	Richness		Reef fish richness warrants significant concern because of the negative trend between 2012–2015. No changes in 2017–2019
Reef Fish Overall	–		–

Table 5.1.13. Overall resource-level summary table.

Resource Category	**Focal Resource**	**Condition Status /Trend**	**Rationale**
Supporting Environment	Shoreline Dynamics		Although the unit has experienced a significant reduction in sandy/gravel shoreline area in the northeast section since 1954, the park has also seen an expansion in land area as result of sediment accretion near Crescent Beach and an increase in the shoreline extent covered by vegetation since 1954.
	Water Quality (Outside Salt River Bay)		There are some indications of fecal pollution and contaminants that could impact corals, but water clarity is good, with low nutrients and high dissolved oxygen.
	Water Quality (Inside Salt River Bay)		There are periodic high levels of fecal indicator bacteria and this may be getting worse with development of the watershed and live-aboard boating activities. There are indications of contaminants in sediments and low oxygen in areas of constricted circulation.
	Watershed Condition		No significant land use/cover change was detected over the period 2002–2012. Increase of the level of anthropogenic development intensity, warrants for moderate concern of the resource condition beyond 2012. No data was available to assess present condition.

Table 5.1.13 (continued). Overall resource-level summary table.

Resource Category	Focal Resource	Condition Status /Trend	Rationale
Biological Integrity	Mangroves		The percent area in mangrove habitat has changed considerably over the years, decreasing primarily from Hurricane Hugo and recovering to the current day. Restoration efforts aided in recovery.
	Semi-deciduous Dry Forest		Distribution of invasive species is widespread, but areas dominated by native woody plants are found throughout the component. No trend could be assessed with the data available.
	Coastal Grassland		The component has large coverage of several highly invasive species, but efforts at removal of exotics and reforestation have led to improvement. Low confidence in the assessment is due to a lack of pre-post invasive treatment monitoring and only qualitative assessment was possible.
	Macroalgae		Though macroalgae cover estimates have only been documented since 2015, percent cover is high and indicative of poor ecosystem health. The current spread of *Ramicrusta* sp., an aggressive coral competitor is also a serious concern
	Seagrass		Aerial photographs from the 1970s depict little change in cover through 2005. However, the spread of the invasive seagrass *Halophila stipulacea* in nearby areas is a serious concern. Low confidence in assessment is due to the lack of recent seagrass cover estimates, lack of in-water surveys to ground truth aerial estimations, different methods for creating mapping polygons in 2005, and lack of trend data in seagrass coverage within Salt River Canyon.
	Corals		Coral cover and abundance is declining, thermal stress events are more common, disease is more common and novel diseases are appearing. Impacts are felt in shallow and deep coral populations.
	Conch		There appears to have been a decline of the population from historical levels, but this is difficult to evaluate because of lack of surveys optimized for conch demographics.
	Reef fish		Reef fish density warrants significant concern because of the negative trend between 2012–2015. No changes in 2017–2019.

A comparison of the eleven focal resources assessed in this report shows that the majority of resources, six of 11 (55%), were considered to be of moderate concern, four resources were judged to be of significant concern, and only one resource was considered to be in good condition. Trends in condition were nearly equally divided between improving, deteriorating, stable, and undetermined.

The focal resources assessed in this report are a mix of marine and terrestrial resources. Terrestrial resources included condition assessments for three vegetation communities (mangroves, dry tropical forest, and coastal grassland) and two supporting watershed condition and shoreline dynamics. With the exception of coastal grasslands (which warrant significant concern), the other terrestrial focal resources were considered to be of moderate concern or good condition in the case of shoreline dynamics, with improving trends or stable conditions. The marine focal resources of SARI were either of significant concern (reef fish, corals, and macroalgae) or moderate concern (conch, seagrass, and water quality) with deteriorating or stable trends. Taken as whole, the assessment suggests that the focal resources of SARI are experiencing degraded conditions compared to reference conditions for these resources and appear to be under a wide range of threats. Deteriorating conditions for corals and macroalgae combined with a lack of recovery of the reef fish communities are especially concerning. The current conditions for these resources appear to have resulted from the result of the interaction of disturbance events and anthropogenic impacts, including extent of hurricane damage, increasing sea surface temperatures, contaminants, introduction of invasive species and continued fishing pressure.

A status warranting significant concern with a deteriorating trend for macroalgal cover in our assessment serves as an indicator of worsening conditions for the coral resources, and perhaps seagrass, but that it unclear, suggesting long-term change in these ecosystems. While algae have both negative and positive effects within the marine environment, increased presence of algae on coral reefs is alarming. Algae compete with coral for space and inhibit recolonization of damaged reefs by corals. While there is no baseline data for macroalgae within SARI to comment on a trend compared to a reference condition, current condition for algae cover within the hard-bottom communities of SARI is high (73%) and algal dominance (considered >50% percent cover) is considered indicative of coral reef decline. Additionally, the presence of the invasive encrusting red algae *Ramicrusta* spp. in SARI is concerning and the threat it poses is great given the rapid expansion of this species throughout the Caribbean and its known ability to overgrow corals.

Shoreline dynamics were assessed as being in both good condition and as having an improving trend as a result of increasing shoreline length and extent of the shoreline currently in vegetated cover. As a supporting resource, the differing character of the shoreline, vegetated vs. sandy or rocky, will undoubtedly benefit particular biological resources at the expense of others. We consider the overall land accretion that is happening at a steady pace, especially in the northeast area of the park, to be an improving trend as it supports terrestrial resources, providing land for mangrove colonization on mudflats and shorebird use on sandy/gravel shorelines.

The overall condition status and trend for several resources was calculated in a manner that weighted particular indicators of condition more highly than others for a particular resource. For mangroves, the indicator percent area in mangrove cover was weighted higher than species composition for the trend in condition as it was considered of greater importance than species assemblage. The greatest decreases from the reference condition occurred as a result of Hurricane Hugo (1989) within Sugar Bay and have been followed by increasing coverage in the subsequent decades albeit with an altered species assemblage that typically follows hurricane disturbance and recolonization (Piou et al. 2006).

Recent hurricane disturbance (2017) combined with drought conditions in 2015, and potentially altered water flow within the watershed attributable to the Mon Bijou flood control project have likely led to additional impacts on this focal resource.

Water quality as a supporting environmental resource is an important driver of change in the condition of biological integrity. All indicators of condition status for water quality outside Salt River Bay were not considered equally weighted. Instead, fecal indicator bacteria and terrestrial sediments were more highly weighted and as a result, the overall status of moderate concern is reflective of their influence. This decision to weight some indicators of water quality more heavily than others was due to the potential linkage of those specific indicators to coral degradation. While water quality inside Salt River Bay was similarly considered of moderate concern and stable trend, it should be noted that fecal indicator bacteria was considered to be of significant concern and with a trend of deteriorating conditions along with evidence of contaminants in sediments, including heavy metals. This has important implications for the biological integrity of focal resources like seagrass and corals that are impacted by water quality conditions. Increased run-off within the Salt River Watershed, discharging to Sugar Bay, is identified within the SARI Foundation document (NPS 2015) as a primary threat to marine resources.

5.2 Reporting Category Information Gaps

Confidence of assessment was high only for the coral focal resource and for shoreline dynamics. Three focal resources had low confidence in the assessment, including: coastal grassland, conch, and seagrass. The remainder had moderate levels of confidence in the condition assessment. Given that a minority of focal resources had high confidence in their assessments, assessments of condition are constrained by a lack of recent data, insufficient temporal or spatial coverage of datasets, or differences between survey methods for datasets compared in this assessment. Important information gaps with some suggestions for future data acquisition are listed for each focal resource in Table 5.2.1.

Table 5.2.1. Summary of important information gaps for each focal resource.

Resource Category	Focal Resource	Important Information Gaps
Supporting Environment	Shoreline Dynamics	Methods that can be used to capture the inter-annual variability of coastal dynamics, specifically those related to accretion and loss of shoreline length and shore surface area are suggested. Methods, which can be used individually or in combination, include the following: automated classification of high-resolution satellite imagery, shoreline assessment using differential GPS technology (high accuracy), automated classification of airborne photography with the use of unmanned aerial systems (UAS technology), and very high-resolution 3D shore profiles.

Table 5.2.1 (continued). Summary of important information gaps for each focal resource.

Resource Category	Focal Resource	Important Information Gaps
Supporting Environment (continued)	Water Quality	A comprehensive water quality sampling program that includes sensitive coral reef ecosystems lining the canyon would provide much more information on status and trends of water quality. Such a program, led by NPS, could include deployed sensors for continuous measurements, discreet sampling for contaminants, and establishment of satellite based remote sensing stations to measure water optical properties (turbidity, chlorophyll, colored dissolved organic matter) and benthic cover.
	Watershed Condition	Runoff information and flow measurement in the main Salt River drainage channel is needed to characterize the variations in the hydrological regime of the watershed. Watershed land use/cover data, depicting the same classes as in NOAA 2002–2007, 2007–2012, are required to define changes over the past ten years.
Biological Integrity	Mangroves	Plot-level information on mangrove structure is not available outside of Sugar Bay and expansion of the current plot network into other mangrove ecotypes in the unit is recommended. Continued regular monitoring of permanent plots and the R-SET in Sugar Bay is suggested.
	Semi-Deciduous Dry Forest	Data on exotic treatment efficacy is restricted to a single plot in the northeast part of the unit. Permanent plots should be expanded to include areas of semi-deciduous dry forest with predominant exotic species coverage as current plots were located in primarily native-dominated forests.
	Coastal Grassland	Collection of environmental variables related to soil and water status could be useful to monitoring species best suited for out-planting in reforestation efforts. Use of treatment efficacy plots and monitoring of extent of re-establishing invasive plant species is strongly suggested.
	Macroalgae	Algal surveys at the species level of determination are recommended for characterization of the resource or for increased detection of changes in community composition. Algal surveys are currently limited to hard-bottom habitats throughout the unit and are lacking in Salt River Bay. Experiments are recommended to elucidate the roles of herbivory and nutrient availability controls on macroalgal abundance and composition.
	Seagrass	Field surveys of seagrass abundance and community composition are recommended; there are no recent surveys since 2005.
	Corals	Monitoring of iconic elkhorn coral populations is currently lacking. The potential of evidence-based coral restoration to rehabilitate coral habitats and threatened species needs to be assessed.
	Conch	There needs to be a consistent and standardized program for monitoring populations of conch if their status and trends are to be adequately understood.
	Reef fish	A continuation of the current monitoring program is necessary to identify trends in reef fish communities, as is employing cross-validation methods among disparate datasets (underway).

Additional research and data collection are needed to answer questions related to how non-native invasive species are impacting focal resources within marine environments, while increased monitoring is needed to understand success of invasive plant management actions on terrestrial vegetation focal resources. Within marine ecosystems, the non-native invasive seagrass *Halophila stipulacea* and the encrusting red algae *Ramicrusta* spp. are concerns for seagrass and coral reefs respectively. *Halophila stipulacea* has the potential to settle in areas where seagrasses have previously not competed with macroalgae. *Ramicrusta* spp. is rapidly increasing at sites in the USVI with the potential to devastate stony corals. Data are needed to understand interactions between colonization of these invasive species and other disturbances, and their potential impacts on the native species. For reef fish, the recent arrival (three observations reported in SARI since 2012) of the invasive Indo-Pacific lionfish (*Pterois volitans*) is another potential threat, as lionfish consume a large amount of prey species and subsequently reduce recruitment of coral-reef fish (Albins and Hixon 2008).

For coastal grasslands, extensive herbicide and mechanical treatments of highly invasive plant species combined with reforestation efforts are improving the condition of these focal resources. However, given a lack of data on treatment efficacy, it is difficult to quantify the magnitude of improvement. Species specific monitoring of invasive species that combined on the ground plot sampling and use of hyperspectral imagery could be used to quantify changes in areal coverage of invasive plant species with the resource.

An integrated approach to monitoring and data collection of the assessed marine focal resources of SARI is suggested as a way to capture changes in these resources and better understand causes impacting the nearshore marine system. A monitoring approach could consist of metrics (like water quality, coral health and abundance, seagrass cover, and the presence of non-native invasive species) collected relative to one another in time and space. The designs for such a sampling scheme are various but should build on existing datasets and infrastructure. Research on the use of the marine and terrestrial resources by visitors are suggested to estimate benefits from ecosystem services provided, as well as amount of anthropogenic pressure on the resource. Information on both legal and illegal fishing would allow for estimates of fishing pressure, which is crucial to understand the temporal and current status of reef fish communities. Rapid responses and management intervention are needed to combat coral diseases like stony coral tissue loss and newly emergent invasive species threats.

For terrestrial resources, expansion of a permanent plot network throughout the terrestrial vegetation focal resources will be necessary to understand long-term changes to species assemblages and abundances related to expansion of invasive species, future hurricane disturbance, and increasing temperatures and changing rainfall patterns expected in a warming climate. Additionally, hydrological monitoring of the Salt River Watershed, including establishment of a weather station, is suggested as it could provide data on the temporal frequency of the flow of water, nutrients, sediments, and contaminants from the terrestrial to the near-shore marine environment. For shoreline dynamics, methods useful in evaluating temporal changes in beach sediments include conducting GPS shoreline surveys at regular set intervals throughout the year and the use of high-resolution satellite data for automated classification schema of sand, water and vegetation. More advanced and precise method options include airborne photography with the use of unmanned aerial systems (UAS technology), and terrestrial LiDAR scans to capture beach profiles.

5.3 Literature Cited

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Appendix A.

Plant species in SARI are presented in Table A-1.

Table A-1. Plant species documented in SARI organized alphabetically by family. The data in this Appendix was compiled from the following sources: Island Resources Foundation, 1993; Kendall et al., 2005; Moser et al., 2011; NPS 2017.

Family	Scientific Name	Common Name
Acanthaceae	*Oplonia microphylla*	thicketwort
Acanthaceae	*Oplonia spinosa*	pricklybush
Agavaceae	*Yucca aloifolia*	Spanish bayonet
Aizoaceae	*Sesuvium portulacastrum*	sea purslane
Amaranthaceae	*Achyranthes indica*	man-better-man
Amaranthaceae	*Blutaparon vermiculare*	salt weed
Anacardiaceae	*Comocladia dodonaea*	Christmas bush
Anacardiaceae	*Mangifera indica*	mango
Anacardiaceae	*Schinus terebinthifolius*	Christmasberry
Annonaceae	*Annona muricata*	soursop
Annonaceae	*Annona squamosa*	sugar apple
Apocynaceae	*Calotripis procera*	giant milkweed
Arecaceae	*Cocos nucifera*	coconut palm
Arecaceae	*Cocothrinax alta*	type palm
Arecaceae	*Roystonea regia*	royal palm
Asclepiadaceae	*Cryptostegia grandiflora*	Indian rubber, rubber vine
Asclepiadaceae	*Cryptostegia madagascariensis*	rubber vine
Asparagaceae	*Agave americana*	century plant
Asparagaceae	*Agave eggersiana* [1]	egger's agave
Asteraceae	*Ageratum conyzoides*	goat weed
Asteraceae	*Bidens pilosa*	beggarticks
Asteraceae	*Borrichia aborescens*	sea oxeye
Asteraceae	*Pluchea symphitifolia*	sweet scent
Bataceae	*Batis maritima*	saltwort
Bignoniaceae	*Crescentia cujete*	calabash
Bignoniaceae	*Macfadyena unguis-cati*	cat claw
Bignoniaceae	*Tabebuia heterophylla*	pink cedar
Bignoniaceae	*Tecoma stans*	ginger thomas
Boraginaceae	*Bourreria succulenta*	pigeon berry
Boraginaceae	*Cordia alba*	white manjack
Boraginaceae	*Cordia collococca*	red manjack

[1] Indicates locally threatened and endangered species according to the VI Division of Fish and Wildlife, 1991.

Table A-1 (continued). Plant species documented in SARI organized alphabetically by family. The data in this Appendix was compiled from the following sources: Island Resources Foundation, 1993; Kendall et al., 2005; Moser et al., 2011; NPS 2017.

Family	Scientific Name	Common Name
Boraginaceae	*Cordia dentata*	flute boom
Boraginaceae	*Cordia rickseckeri*	San Bartolome
Boraginaceae	*Tournefortia volubilis*	twining soldierbush
Brassicaceae	*Cakile lanceolata*	searocket
Brassicaceae	*Carica papaya*	papaya
Burseraceae	*Bursera simaruba*	gumbo limbo
Cactaceae	*Mammilaria nivosa* [1]	woolly nipple
Cactaceae	*Melocactus intortus*	turk's Cap
Cactaceae	*Opuntia* spp.	–
Cactaceae	*Pilosocereus royenii*	pipe organ cactus
Capparaceae	*Capparis cynophallophora*	Jamaican caper
Capparaceae	*Capparis flexuosa*	limber caper
Capparaceae	*Capparis frondosa*	rat-bean
Capparaceae	*Capparis indica*	caper
Capparaceae	*Cleome viscosa*	tickweed, spider flower
Capparaceae	*Morisonia americana*	rat-apple
Celastraceae	*Maytenus laevigata*	wild cinnamon
Celastraceae	*Schaefferia frutescens*	Florida boxwood
Combretaceae	*Bucida buceras*	gregre
Combretaceae	*Conocarpus erectus*	buttonwood
Combretaceae	*Laguncularia racemosa*	white mangrove
Combretaceae	*Terminalia catappa*	Indian almond
Commelinaceae	*Commelina diffusa*	blue day-flower
Commelinaceae	*Commelina erecta*	French grass
Commelinaceae	*Tradescantia spathacea*	oyster Plant
Convulvulaceae	*Ipomoea pes-capre*	beach morning glory
Convulvulaceae	*Merremia quinquefolia*	merremia
Crassulaceae	*Kalanchoe pinnata*	leaf of life
Curcurbitaceae	*Momordica charantia*	maiden apple
Cyperaceae	*Cyperus involucratus*	umbrella plant
Cyperaceae	*Fimbristylis spathacea*	hurricane grass
Erythroxylaceae	*Erythroxylum brevipes*	brizzlet
Euphorbiaceae	*Croton astroites*	wild marrow
Euphorbiaceae	*Croton rigidus*	yellow maran
Euphorbiaceae	*Euphorbia tirucalli*	milk bush

[1] Indicates locally threatened and endangered species according to the VI Division of Fish and Wildlife, 1991.

Table A-1 (continued). Plant species documented in SARI organized alphabetically by family. The data in this Appendix was compiled from the following sources: Island Resources Foundation, 1993; Kendall et al., 2005; Moser et al., 2011; NPS 2017.

Family	**Scientific Name**	**Common Name**
Euphorbiaceae	*Gymnanthes lucida*	crab wood
Euphorbiaceae	*Hippomane mancinella*	manchineel
Euphorbiaceae	*Jatropha gossypifolia*	physic nut, bellyache bush
Fabaceae	*Acacia macracantha*	long-spine acacia
Fabaceae	*Acacia tortuosa*	poponax
Fabaceae	*Albizia lebbek*	woman's tongue
Fabaceae	*Caesalpinia bonduc*	gray nicker
Fabaceae	*Caesalpinia ciliata*	yellow nicker
Fabaceae	*Canavalia maritima*	baybean
Fabaceae	*Clitoria ternatea*	butterfly pea
Fabaceae	*Crotalaria retusa*	rattle box
Fabaceae	*Dalbergia ecastaphyllum*	coinvine
Fabaceae	*Delonix regia*	flamboyant tree
Fabaceae	*Desmanthus virgatus*	wild tantan
Fabaceae	*Leucaena leucocephala*	tantan
Fabaceae	*Piscidia carthagenensis*	Jamaican dogwood
Fabaceae	*Piscidia piscipula*	fish poison
Fabaceae	*Pithecellobium unguis-cati*	blackbead
Fabaceae	*Samanea saman*	raintree
Fabaceae	*Senna siamea*	yellow cassia
Fabaceae	*Sesbania sericea*	papagayo
Fabaceae	*Tamarindus indica*	tamarind
Lauraceae	*Cassytha filiformis*	love mine
Liliaceae	*Sansevieria hyacinthoides*	iguanatail
Malpighiaceae	*Malpighia infestissima* [1]	stingbush
Malpighiaceae	*Malpighia woodburyana* [1]	cow-itch, cowage cherry
Malvaceae	*Gossypium barbadense*	creole cotton
Malvaceae	*Hibiscus* sp.	hibiscus
Malvaceae	*Malvastrum corchorifolium*	false mallow
Malvaceae	*Malvastrum coromandelianum*	threelobe false mallow
Malvaceae	*Sida acuta*	broom weed
Malvaceae	*Thespesia populnea*	seaside mahoe
Meliaceae	*Swietenia mahagoni*	mahogany
Moraceae	*Ficus citrifolia*	shortleaf fig
Myrtaceae	*Eugenia biflora*	blackrodwood

[1] Indicates locally threatened and endangered species according to the VI Division of Fish and Wildlife, 1991.

Table A-1 (continued). Plant species documented in SARI organized alphabetically by family. The data in this Appendix was compiled from the following sources: Island Resources Foundation, 1993; Kendall et al., 2005; Moser et al., 2011; NPS 2017.

Family	Scientific Name	Common Name
Myrtaceae	*Eugenia cordata*	lathberry
Myrtaceae	*Eugenia monticola*	white stopper, birdcherry
Myrtaceae	*Eugenia procera*	rockmyrtle
Myrtaceae	*Eugenia rhombea*	spiceberry
Nyctaginaceae	*Boerhavia coccinea*	boerhavia, hog weed
Nyctaginaceae	*Guapira fragrans*	wild mampoo
Nyctaginaceae	*Pisonia aculeata*	prickly mampoo
Nyctaginaceae	*Pisonia subcordata*	water mampoo
Onagraceae	*Ludwigia* sp.	–
Orchidaceae	*Epidendrum ciliare* [1]	spider orchid
Orchidaceae	*Psychilis bifidum* [1]	–
Petiveriaceae	*Petiveria alliacea*	garlic weed
Petiveriaceae	*Rivina humilis*	cat's blood
Poaceae	*Bambusa vulgaris*	bamboo
Poaceae	*Bothriochloa pertusa*	pitted beardgrasss
Poaceae	*Cenchrus echinatus*	sandburr
Poaceae	*Cynodon dactylon*	bermudagrass
Poaceae	*Distichlis spicata*	beach grass
Poaceae	*Panicum maxium*	guinea grass
Poaceae	*Spartina patens*	salt grass
Poaceae	*Sporobolus indicus*	smut grass
Poaceae	*Sporobolus virginicus*	seashore dropseed
Poaceae	*Urochloa maxima*	guinea grass
Polygonaceae	*Antigonon leptopus*	coral vine
Polygonaceae	*Coccoloba swartzii*	swartz's pigeonplum
Polygonaceae	*Coccoloba uvifera*	seagrape
Polygonaceae	*Coccoloba venosa*	cherry grape
Portulacaceae	*Portulaca oleracea*	little hogweed, purslane
Pteridaceae	*Acrostichum daneaifolium*	swamp fern
Pteridaceae	*Doryopteris* spp.	–
Rhamnaceae	*Colubrina arborescens*	greenheart
Rhamnaceae	*Gouania lupuloides*	urban whiteroot
Rhamnaceae	*Krugiodendron ferreum*	urban ironwood
Rhizophoraceae	*Rhizophora mangle*	red mangrove
Rubiaceae	*Morinda citrifolia*	painkller

[1] Indicates locally threatened and endangered species according to the VI Division of Fish and Wildlife, 1991.

Table A-1 (continued). Plant species documented in SARI organized alphabetically by family. The data in this Appendix was compiled from the following sources: Island Resources Foundation, 1993; Kendall et al., 2005; Moser et al., 2011; NPS 2017.

Family	**Scientific Name**	**Common Name**
Rubiaceae	*Psychotria nervosa*	wild coffee
Rubiaceae	*Randia aculeata*	box-briar
Rutaceae	*Triphasia trifolia*	sweet lime, limeberry
Rutaceae	*Zanthoxylum flavum*	satin wood
Rutaceae	*Zanthoxylum martinicense*	white prickle
Rutaceae	*Zanthoxylum monophyllum*	yellow prickle
Sapindaceae	*Melicoccus bijugatus*	genip
Sapindaceae	*Serjania lucida*	basket wood
Sapotaceae	*Chrysophyllum pauciflorum*	caimito de perro
Sapotaceae	*Manilkara jaimiqui* subsp. *emarginata*	wild sapodilla
Sapotaceae	*Sideroxylon foetidissima*	false mastic
Sapotaceae	*Sideroxylon obovatum*	–
Sapotaceae	*Sideroxylon salicifolium*	willow bustic
Surianaceae	*Suriana maritima*	bay cedar
Theophrastaceae	*Jacquinea arborea*	braceletwood
Theophrastaceae	*Jacquinia armillaris*	–
Tiliaceae	*Corchorus hirsutus*	jackswitch
Typhaceae	*Typha domingensis*	southern cattail
Urticaceae	*Laportea* sp.	stinging nettle
Verbenaceae	*Avicennia germinans*	black mangrove
Verbenaceae	*Citharexylum fruticosum*	fiddlewood
Verbenaceae	*Lantana camara*	yellow sage
Verbenaceae	*Lantana involucrata*	sage
Verbenaceae	*Stachytarpheta jamaicensis*	blue porterweed
Vitaceae	*Cissus sicyoides*	pinekoop
Vitaceae	*Cissus trifoliata*	sorrel Vine
Vitaceae	*Cissus verticillata*	pudding Vine
Zygophyllaceae	*Guaicium officinale* [1]	lignumvitae
Zygophyllaceae	*Guaicium sanctum*	holywood lignumvitae

[1] Indicates locally threatened and endangered species according to the VI Division of Fish and Wildlife, 1991.

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Appendix B.

Bird species in SARI are presented in Table B-1.

Table B-1. Bird species organized alphabetically by Order documented in SARI Data was compiled from the following sources: McNair et al. 2005, McNair 2008, eBird 2017, Yntema et al. 2017.

Order	Scientific Name	Common Name	Abundance
Accipitriformes	*Buteo jamaicensis*	Red-tailed Hawk	Uncommon
Accipitriformes	*Pandion haliaetus*	Osprey	Common
Anseriformes	*Anas bahamensis*	White-cheecked Pintail	–
Apodiformes	*Anthracothorax dominicus* [1]	Antillean Mango	Rare
Apodiformes	*Eulampis holosericeus*	Green-throated Carib	Common
Apodiformes	*Orthorhyncus cristatus*	Antillean Crested Hummingbird	Common
Charadriiformes	*Actitis macularius*	Spotted Sandpiper	Common
Charadriiformes	*Arenaria interpres*	Ruddy Turnstone	Common
Charadriiformes	*Calidris alba*	Sanderling	Rare
Charadriiformes	*Calidris himantopus*	Stilt Sandpiper	Uncommon
Charadriiformes	*Calidris minutilla*	Least Sandpiper	Common
Charadriiformes	*Calidris pusilla*	Semipalmated Sandpiper	Common
Charadriiformes	*Charadrius semipalmatus*	Semipalmated Plover	Common
Charadriiformes	*Charadrius vociferus*	Killdeer	Uncommon
Charadriiformes	*Charadrius wilsonia*	Wilson's Plover	Common
Charadriiformes	*Gallinago delicata*	Wilson's Snipe	Uncommon
Charadriiformes	*Haematopus palliatus*	American Oystercatcher	Uncommon
Charadriiformes	*Himantopus mexicanus*	Black-necked Stilt	Abundant
Charadriiformes	*Leucophaeus atricilla*	Laughing Gull	Uncommon
Charadriiformes	*Limnodromus scolopaceus*	Short-billed Dowitcher	–
Charadriiformes	*Pluvialis squatarola*	Black-bellied Plover	Common
Charadriiformes	*Sterna hirundo*	Common Tern	Rare
Charadriiformes	*Sternula antillarum*	Least Tern	Common
Charadriiformes	*Thalasseus maximus*	Royal Tern	Common
Charadriiformes	*Tringa flavipes*	Lesser Yellowlegs	Common
Charadriiformes	*Tringa melanoleuca*	Greater Yellowlegs	–
Columbiformes	*Columbina passerina*	Common Ground-Dove	Common
Columbiformes	*Geotrygon mystacea* [1]	Bridled Quail-Dove	Uncommon
Columbiformes	*Patagioenas leucocephala*	White-crowned Pigeon	Common
Columbiformes	*Patagioenas squamosa*	Scaly-naped Pigeon	Common
Columbiformes	*Zenaida asiatica*	White-winged Dove	Common

[1] Indicates species presence in SARI unconfirmed.

Table B-1 (continued). Bird species organized alphabetically by Order documented in SARI Data was compiled from the following sources: McNair et al. 2005, McNair 2008, eBird 2017, Yntema et al. 2017.

Order	Scientific Name	Common Name	Abundance
Columbiformes	*Zenaida aurita*	Zenaida Dove	Common
Coraciiformes	*Megaceryle alcyon*	Belted Kingfisher	Common
Cuculiformes	*Coccyzus americanus*	Yellow-billed Cuckoo	Uncommon
Cuculiformes	*Coccyzus minor*	Mangrove Cuckoo	Common
Falconiformes	*Falco peregrinus*	Peregrine Falcon	Uncommon
Falconiformes	*Falco sparverius*	American Kestrel	Common
Passeriformes	*Coereba flaveola*	Bananaquit	Common
Passeriformes	*Elaenia martinica*	Caribbean Elaenia	Uncommon
Passeriformes	*Geothlypis trichas*	Common Yellowthroat	Uncommon
Passeriformes	*Helmitheros vermivorum*	Worm-eating Warbler	Rare
Passeriformes	*Hirundo rustica*	Barn Swallow	Uncommon
Passeriformes	*Icterus galbula*	Baltimore Oriole	–
Passeriformes	*Loxigilla noctis*	Lesser Antillean Bullfinch	Uncommon
Passeriformes	*Margarops fuscatus*	Pearly-eyed Thrasher	Common
Passeriformes	*Mimus polyglottos*	Northern Mockingbird	Common
Passeriformes	*Mniotilta varia*	Black-and-white Warbler	Rare
Passeriformes	*Parkesia noveboracensis*	Northern Waterthrush	Common
Passeriformes	*Passer domesticus*	House Sparrow	Uncommon
Passeriformes	*Protonotaria citrea*	Prothonotary Warbler	Rare
Passeriformes	*Seiurus aurocapilla*	Ovenbird	Uncommon
Passeriformes	*Setophaga americana*	Northern Parula	Uncommon
Passeriformes	*Setophaga caerulescens*	Black-throated Blue Warbler	Rare
Passeriformes	*Setophaga citrina*	Hooded Warbler	Uncommon
Passeriformes	*Setophaga coronata*	Yellow-rumped Warbler	–
Passeriformes	*Setophaga discolor*	Prairie Warbler	Uncommon
Passeriformes	*Setophaga dominica*	Yellow-throated Warbler	Rare
Passeriformes	*Setophaga magnolia*	Magnolia Warbler	Rare
Passeriformes	*Setophaga pensylvanica*	Chestnut-sided Warbler	Rare
Passeriformes	*Setophaga petechia*	Yellow Warbler	Abundant
Passeriformes	*Setophaga ruticilla*	American Redstart	Uncommon
Passeriformes	*Setophaga striata*	Blackpoll Warbler	–
Passeriformes	*Setophaga tigrina*	Cape May Warbler	Uncommon
Passeriformes	*Setophaga virens*	Black-throated Green Warbler	Rare
Passeriformes	*Tiaris bicolor*	Black-faced Grassquit	Common
Passeriformes	*Tyrannus dominicensis*	Gray Kingbird	Common

[1] Indicates species presence in SARI unconfirmed.

Table B-1 (continued). Bird species organized alphabetically by Order documented in SARI Data was compiled from the following sources: McNair et al. 2005, McNair 2008, eBird 2017, Yntema et al. 2017.

Order	Scientific Name	Common Name	Abundance
Passeriformes	*Vermivora cyanoptera*	Blue-winged Warbler	Rare
Passeriformes	*Vireo altiloquus*	Black-whiskered Vireo	Uncommon
Passeriformes	*Vireo flavifrons*	Yellow-throated Vireo	–
Pelecaniformes	*Ardea alba*	Great Egret	Common
Pelecaniformes	*Ardea herodias*	Great Blue Heron	Uncommon
Pelecaniformes	*Bubulcus ibis*	Cattle Egret	Uncommon
Pelecaniformes	*Butorides virescens*	Green Heron	Uncommon
Pelecaniformes	*Egretta caerulea*	Little Blue Heron	Common
Pelecaniformes	*Egretta thula*	Snowy Egret	Common
Pelecaniformes	*Egretta tricolor*	Tricolored Heron	Common
Pelecaniformes	*Nyctanassa violacea*	Yellow-crowned Night-Heron	Common
Pelecaniformes	*Nycticorax nycticorax*	Black-crowned Night-Heron	Common
Pelecaniformes	*Pelecanus occidentalis*	Brown Pelican	Abundant
Piciformes	*Sphyrapicus varius*	Yellow-bellied sapsucker	Rare
Podicepiformes	*Tachybaptus dominicus*	Least Grebe	–
Suliformes	*Fregata magnificens*	Magnificent Frigatebird	Common

[1] Indicates species presence in SARI unconfirmed.

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Appendix C.

On site visit to BUSI and SARI (Feb. 6–10, 2017)

AGENDA

**NATURAL RESOURCE CONDITION ASSESSMENT SCOPING MEETING
ST. CROIX, USVI (BUIS/SARI PARKS)**

- **Monday Feb 6** – Buck Island Reef National Monument (BUIS) site visit with focus on natural resource issues:
    - Team meets at **8:30 AM** in front of Fort Christiansvaern at Christiansted National Historic Site
- **Tuesday Feb 7** – Salt River Bay National Historical Park and Ecological Preserve (SARI) site visit with focus on natural resource issues
    - Team meets at **8:30 AM** at Headquarters (2100 Church Street #100 Christiansted, St. Croix, USVI)
- **Wednesday-Friday Feb 8–10** – BUIS/SARI NRCA scoping and supplemental data transfers
    - Meeting at the Headquarters (2100 Church Street #100, GCW (first floor), Christiansted, St. Croix, USVI)

Participants (in person):
Nathaniel Hanna (NPS), Clayton Pollock (NPS), Tessa Code (NPS Technician), Zandy Hillis-Starr (NPS Chief of Resource Management BUIS), Dale McPherson (NPS Natural Resource Program Manager), Elizabeth Whitman (PhD Candidate FIU), Daniel Gann (Research Associate FIU), Anna Wachnicka (Research Assistant Professor FIU)

Participants (joining by phone):
Maria C. Donoso (Research Associate Professor FIU), Danielle E. Ogurcak (Postdoctoral Associate FIU), Mike Feeley (NPS SFCN)

Table C-1. Agenda.

Date	Time	Topics for Feb 8–9 Meeting & Feb 9–10 Activities
February 8th Meeting (Park HQ)	**8:30–9:00**	• Room set-up
	9:00–9:15	• Arrival/Call-in/Introductions
	9:15–9:45	• Introduction to NRCAs (Dale) • Project Schedule & Meeting Expectations (Anna)
	9:45–12:00	• Setting expectations for the BUIS and SARI NRCA reports • Reviewing park resources, threats/stressors, issues, and gaps that will be used for populating the Heinz framework tables; Completing scoping tables for the parks • Collecting contact information for experts identified in framework table
	12:00–1:00	• Lunch Break
	1:00–4:30	• Continuation of the scoping meeting; completing scoping tables for the parks (Discussion between FIU team & Park representatives)
February 9 Meeting and Activities (Park HQ)	**8:20**	• Room set-up
	8:30–12:00	• Continuation of the scoping meeting; completing scoping tables for the parks (Discussion between FIU team & Park representatives)
	12:00–1:00	• Lunch Break
	1:00–4:30	• Discussion on data management/ArcGIS files storage and management, including handling of any sensitive data • Supplemental data transfers • Consolidating info on literature sources (reports/papers) available for writing the reports
February 10 Activity and Closing Meeting (Park HQ)	**8:30–12:00**	• Supplemental data transfers
	12:00–1:00	• Lunch Break
	1:00–4:00	• Supplemental data transfers • Final remarks/comments/Q & A

NPS 141/181609, June 2022

National Park Service
U.S. Department of the Interior

Natural Resource Stewardship and Science
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