National Park Service
U.S. Department of the Interior

Natural Resource Stewardship and Science

Natural Resource Condition Assessment

Cumberland Island National Seashore

Natural Resource Report NPS/CUIS/NRR—2018/1773

ON THE COVER
An aerial view of the northeast portion of Cumberland Island (NPS photo).

Natural Resource Condition Assessment

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Natural Resource Report NPS/CUIS/NRR—2018/1773

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Executive Summary

The Natural Resource Condition Assessment (NRCA) Program aims to provide documentation about the current conditions of important park natural resources through a spatially explicit, multi-disciplinary synthesis of existing scientific data and knowledge. Findings from the NRCA will help Cumberland Island National Seashore (CUIS) managers to develop near-term management priorities, engage in watershed or landscape scale partnership and education efforts, conduct park planning, and report program performance (e.g., Department of the Interior's Strategic Plan "land health" goals, Government Performance and Results Act).

The objectives of this assessment are to evaluate and report on current conditions of key park resources, to evaluate critical data and knowledge gaps, and to highlight selected existing stressors and emerging threats to resources or processes. For the purpose of this NRCA, staff from the National Park Service (NPS) and Saint Mary's University of Minnesota – GeoSpatial Services (SMUMN GSS) identified key resources, referred to as "components" in the project. The selected components include natural resources and processes that are currently of the greatest concern to park management at CUIS. The final project framework contains 10 resource components, each featuring discussions of measures, stressors, and reference conditions.

This study involved reviewing existing literature and, where appropriate, analyzing data for each natural resource component in the framework to provide summaries of current condition and trends in selected resources. When possible, existing data for the established measures of each component were analyzed and compared to designated reference conditions. A weighted scoring system was applied to calculate the current condition of each component. Weighted Condition Scores, ranging from zero to one, were divided into three categories of condition: low concern, moderate concern, and significant concern. These scores help to determine the current overall condition of each resource. The discussions for each component, found in Chapter 4 of this report, represent a comprehensive summary of current available data and information for these resources, including unpublished park information and perspectives of park resource managers, and present a current condition designation when appropriate. Each component assessment was reviewed by CUIS resource managers, NPS Southeast Coast Network (SECN) staff, or NPS regional staff.

Existing literature, short- and long-term datasets, and input from NPS and other outside agency scientists support condition designations for components in this assessment. However, in some cases, data were unavailable or insufficient for several of the measures of the featured components. In other instances, data establishing reference condition were limited or unavailable for components, making comparisons with current information inappropriate or invalid. In these cases, it was not possible to assign condition for the components. Current condition was not able to be determined for four of the 10 components due to these data gaps.

For those components with sufficient available data, the overall condition varied. Three components were determined to be in good condition: salt marshes, mammals, and herpetofauna. However, the salt marshes had a condition score that was at the edge of the good condition range; any small decline in conditions could shift the component into the moderate concern range. Trends for all three of these

components were considered stable. One component (upland forest community) was of moderate concern with a stable trend. The remaining two components were of significant concern: air quality and barrier island geomorphology. A trend could not be determined for barrier island geomorphology, but air quality showed an improving trend. Detailed discussion of these designations is presented in Chapters 4 and 5 of this report.

Several park-wide threats and stressors influence the condition of priority resources in CUIS. Those of primary concern include feral wildlife, fire suppression, and impacts related to climate change. Understanding these threats, and how they relate to the condition of park resources, can help the NPS prioritize management objectives and better focus their efforts to maintain the health and integrity of the park ecosystem, as well as its historically significant landscape.

Acknowledgments

We acknowledge Cumberland Island National Seashore and Southeast Coast Inventory and Monitoring Network staff for the technical expertise provided during scoping, through multiple stages of review, and via phone and email; specifically, John Fry, Doug Hoffman, Brian Gregory, Mike Byrne, and Lisa Baron. Their logistical insight and critical review of interim documents was essential to this project’s completion. Additional information and reviews were provided by Emma Brown, Linda York, and Sharolyn Anderson. Dale McPherson, Regional NRCA and RSS Coordinator, and Jeff Albright, Natural Resource Condition Assessment Coordinator, provided program guidance. Thank you to all others who assisted the development of this document.

Acronyms and Abbreviations

ALR – Anthropogenic Light Ratio

AMBUR – Analyzing Moving Boundaries Using R

ARD – Air Resources Division

ARDs – Automatic Recording Devices

CAA – Clean Air Act

CBC – Christmas Bird Count

CFU – Colony Forming Units

CUIS – Cumberland Island National Seashore

dB – Decibels

DO – Dissolved Oxygen

DOD – Department of Defense

Dv – Deciviews

EIS – Environmental Impact Study

EPA – Environmental Protection Agency

Esri – Environmental Systems Research Institute

FAA – Federal Aviation Administration

FMP – Fire Management Plan

GA DNR – Georgia Department of Natural Resources

GA-EPPC – Georgia Exotic Pest Plant Council

GIS – Geographic Information System

HFSM – High Fringing Salt Marsh

I&M – Inventory and Monitoring

IMPROVE – Interagency Monitoring of Protected Visual Environments Program

IPCC – Intergovernmental Panel on Climate Change

JEA – Jacksonville Electric Authority

LCI – Little Cumberland Island

LWD – Laurel Wilt Disease

MDN – Mercury Deposition Network

NAAQS – National Ambient Air Quality Standards

NABat – North American Bat Monitoring Program

NADP-NTN – National Atmospheric Deposition Program–National Trends Network

NLCD – National Landcover Dataset

NMFS – National Marine Fisheries Service

NPS – National Park Service

NRCA – Natural Resource Condition Assessment

NST – Night Skies Team

NVC – National Vegetation Classification

NWI – National Wetlands Inventory

PM – Particulate Matter

Ppb – Parts Per Billion

SECN – Southeast Coast Network

SLR – Sea Level Rise

SMUMN GSS – Saint Mary's University of Minnesota, Geospatial Services

SpC – Specific Conductance

TDS – Total Dissolved Solids

TED – Turtle Excluder Device

USACE – U.S. Army Corps of Engineers

USCB – United States Census Bureau

USFWS – U.S. Fish and Wildlife Service

USGS – United States Geological Survey

VES – Visual Encounter Survey

VOCs – Volatile Organic Compounds

WCS – Weighted Condition Score

WNS – White Nose Syndrome

WRD – Water Resources Division

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 geographic information system (GIS) products;[4]
- Summarize key findings by park areas;[5] and
- Follow national NRCA guidelines and standards for study design and reporting products.

[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.

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 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)

[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.

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.

2. Introduction and Resource Setting

2.1. Introduction

2.1.1. Enabling Legislation

Cumberland Island National Seashore (CUIS) was established on 23 October 1972

> *… to provide for public outdoor recreation use and enjoyment of certain significant shoreline lands and waters of the United States, and to preserve related scenic, scientific, and historical values (PL 92-536)* (NPS 1984).

CUIS was designated as a National Seashore because of its outstanding natural, recreational, and historical values, including "a remarkable seashore area of beach dunes, forests and uplands, and marsh" (NPS 1984, p. 5). Ten years later, in September 1982, Congress designated approximately 3,577 ha (8,840 ac) of CUIS as wilderness area, and another 4,742 ha (11,718 ac) as "potential wilderness". As of 2014, the designated wilderness area had increased to 4,001 ha (9,886 ac) (NPS 2014a). The original establishing legislation specified that "no causeway" be built out to the island and that access shall continue by boat only (NPS 1984).

Looking onto an Atlantic Ocean beach at CUIS from the dunes (NPS photo).

2.1.2. Geographic Setting

Measuring 28.2 km (17.5 mi) long, Cumberland Island is the largest of Georgia's coastal barrier islands (NPS 2014a). It lies just north of the Florida/Georgia state line (Figure 1), between 1.6 and 4.8 km (1-3 mi) off the mainland. The CUIS park boundary includes 14,709 ha (36,347 ac), just over half of which is owned by the NPS (NPS 2015b). This includes approximately 283 km (176 mi) of shoreline (Curdts 2011). The Georgia Department of Natural Resources (GA DNR) has jurisdiction over the tidal beaches and marshes around the island (i.e., those below the mean high tide line) (NPS 1984). The U.S. Army Corps of Engineers (USACE) has jurisdiction of dredge spoil areas on the south end of Cumberland Island and on Drum Point Island, which is owned by the U.S. Navy. Various portions of the park, including all uplands on Little Cumberland Island on the northern edge, remain in private ownership and are not open to public visitation (NPS 1984, 2014a).

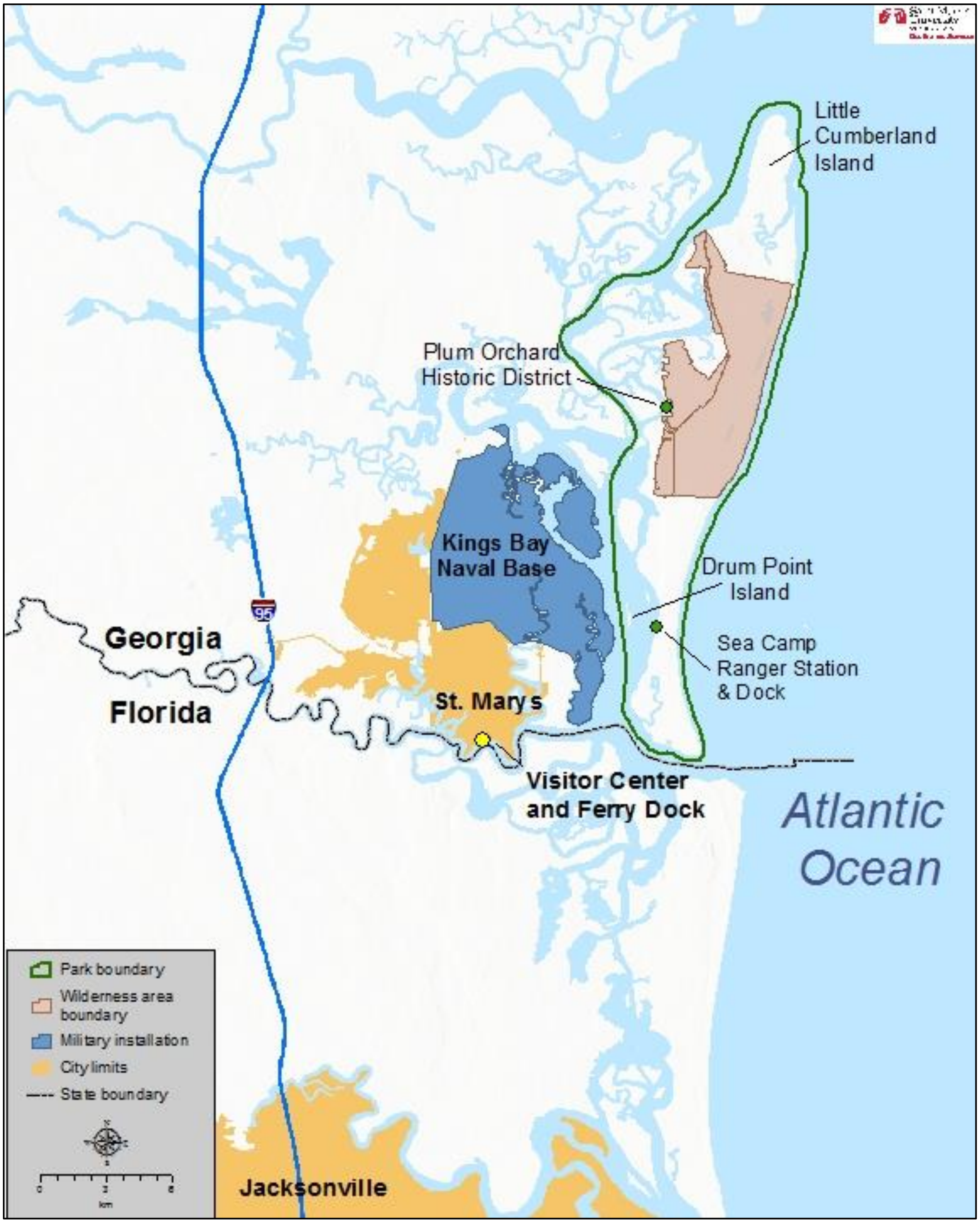


Figure 1. Location of CUIS along the Georgia Coast, and the location of designated wilderness area on the island.

Typical of the southeastern U.S., CUIS has a humid subtropical climate (Davey et al. 2007). However, temperatures are often moderated by ocean breezes, resulting in warmer conditions than on the mainland in the winter and fewer very hot days during the summer (NPS 1984). While tropical storms are common, CUIS historically has not experienced as many hurricane-strength storms as other portions of the southern Atlantic Coast. However, the park was impacted by Hurricane Matthew in October 2016 and by Hurricane Irma (after it had weakened to a tropical storm) in September 2017. The peak potential season for hurricanes normally runs from late June to mid-October (NPS 1984). Temperature and precipitation normals from the nearest long-term weather monitoring station (Fernandina Beach, FL) are shown in Table 1 and Table 2.

Table 1. 30-year temperature normals (1981-2010) from Fernandina Beach, FL, 6 km (3.7 mi) south of CUIS (NCDC 2015).

Average Temperature (°C)	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sept	Oct	Nov	Dec	Annual
Max	17.2	18.8	21.8	24.9	28.5	31.1	32.6	31.8	29.8	26.2	22.3	18.3	25.3
Min	6.9	8.4	11.4	14.8	19.3	22.7	23.7	23.8	22.7	18.3	13.1	8.7	16.2

Table 2. 30-year precipitation normals (1981-2010) from Fernandina Beach, FL, 6 km (3.7 mi) south of CUIS (NCDC 2015).

Average Precipitation (cm)	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sept	Oct	Nov	Dec	Annual
Total	8.7	8.1	10.0	7.2	5.9	13.4	14.0	14.8	17.6	11.7	5.3	7.5	124.0

2.1.3. Visitation Statistics

The park can only be accessed by passenger ferry or private boat, and visitation is limited to 300 people per day (NPS 2014a). Prior to 2007, the NPS recorded only those visitors that travelled out to the island. Since that time, visitors that stop only at the mainland visitor center have also been included in visitation statistics. On average, CUIS received close to 70,000 visitors per year between 2007 and 2016 (NPS 2017b). Visitation peaked at nearly 92,000 in 2010, with its lowest level during this period occurring in 2013 when there were just over 51,000 annual visitors. The park’s beaches and natural areas provide opportunities for hiking, camping, swimming, fishing, and nature exploration (Littlejohn 1999, NPS 2014a). A private concessioner offers guided tours of historic features on the island’s northern half, including Plum Orchard Mansion (Figure 2), Cumberland Island Wharf, and The Settlement (NPS 2014a).

Figure 2. The historic Plum Orchard Mansion on the west side of Cumberland Island (NPS photo).

2.2. Natural Resources

2.2.1. Ecological Units and Watersheds

CUIS lies within the Environmental Protection Agency's (EPA) Southern Coastal Plain Level III Ecoregion. The Southern Coastal Plain is a diverse ecoregion that includes coastal marshes, lagoons, barrier islands, and swampy lowlands along the Atlantic and Gulf coasts (EPA 2013). The region was historically covered by a variety of pine, hardwood, and mixed forests, but much of the area is now in less diverse second-growth forest, pasture for livestock, or human development (EPA 2013). The EPA divides Level III Ecoregions into smaller Level IV Ecoregions. The park falls in the Sea Islands/Coastal Marsh Level IV Ecoregion.

As an island, CUIS is not part of a larger "watershed", in the traditional sense; no water flows into the park from "upstream", and there are no terrestrial areas "downstream" of the park. Precipitation that falls on the island remains as surface water/shallow groundwater or flows into the surrounding ocean and sound directly or via several natural drainage systems. The Whitney/South Cut outflow and the Lake Retta outflow drain portions of the east side of the island, including the 121-hectare (300-ac) Sweetwater Lake Complex and other interdunal wetland complexes (Hillestad et al. 1975; John Fry, CUIS Chief of Resource Management, personal communication, 15 February 2018). The Malkintooh Creek outflow (in the vicinity of Brickhill Bluff) drains on the west side, along with Hawkins and Old House Creeks (Figure 3). An artificial drainage system exists in the Swamp Fields area south of Plum Orchard, where 2.1 km (1.3 mi) of canals and levees were excavated to drain historical agricultural lands into the White Branch outflow (Hillestad et al. 1975).

Figure 3. General locations of outflows (drainage points) within CUIS (based on Hillestad et al. 1975).

2.2.2. Resource Descriptions

CUIS can roughly be divided into coastal lowlands (beaches, salt marshes) and uplands (dunes, forests). The eastern coast of the island is dominated by flat, sandy beaches, while the western lowlands consist primarily of salt marshes and mud flats intersected by tidal creeks (NPS 1984, DeVivo et al. 2008). The salt marshes provide valuable breeding and nursery habitat for a variety of wildlife and important feeding grounds for game and fish species from neighboring estuarine areas (NPS 1984, Peek et al. 2016). Several different forest types occur in the park's uplands, with many dominated by live oaks (*Quercus virginiana*), some which reach 0.9-1.2 m (3-4 ft) in diameter (Figure 4). Many of the island's trees support Spanish moss (*Tillandsia usneoides*) and polypody ferns (*Phlebodium aureum*) on their trunks and branches (NPS 1984). Saw palmettos (*Serenoa repens*) can occur in dense thickets or scattered in the forest understory. Much of the upland forest is secondary growth that has returned following historical logging (NPS 1984). Extensive dune systems back the east coast beaches, with some dunes reaching 15 m (50 ft) in height (NPS 1984, DeVivo et al. 2008).

Figure 4. Live oaks at CUIS, with Spanish moss on the branches (SMUMN GSS photo).

The park supports nearly 800 confirmed vascular plant taxa, including subspecies and varieties, with approximately 50 additional taxa possibly present (NPS 2016f). These include four species considered rare and one considered threatened by the State of Georgia (Table 3) (GA DNR 2016, NPS 2016f).

Table 3. CUIS plant species designated as rare or threatened by the state of Georgia (GA DNR 2016, NPS 2016f).

Scientific Name	Common Name	State Status*
Pityopsis pinifolia	Taylor County goldaster	rare (S2)
Forestiera segregata	Florida swampprivet	rare (S2)
Carex dasycarpa	sandywoods sedge	rare (S3)
Sageretia minutiflora	smallflower mock buckthorn	threatened (S2)
Sapindus saponaria var. saponaria	wingleaf soapberry	rare (S1S2)

* S1 – critically imperiled; S2 – imperiled; S3 – vulnerable

Twenty-six mammal species have been confirmed within the park, and an additional five species (all marine mammals) are considered probably present (NPS 2016f). White-tailed deer (*Odocoileus virginianus*), gray squirrels (*Sciurus carolinensis*), and raccoons (*Procyon lotor*) are commonly observed, and bobcats (*Lynx rufus*) have become common since their reintroduction to the island in 1988-89 (NPS 1984, Diefenbach et al. 2013, NPS 2016f). Bottlenose dolphins (*Tursiops truncatus*) are often seen off shore (NPS 2016f).

Just over 330 bird species are considered present or probably present at the park, and CUIS is known for its abundance of shorebirds (DeVivo et al. 2008, NPS 2016f). Sixteen of the island's bird species are considered rare, threatened, or endangered by the State of Georgia (NPS 2016f). Two species are listed as federally threatened (wood stork [*Mycteria americana*] and piping plover [*Charadrius melodus*]) (USFWS 2017). Species considered "management priority" for the park include bald eagle (*Haliaeetus leucocephalus*), osprey (*Pandion haliaetus*), Wilson's plover (*Charadrius wilsonia*), and American oystercatcher (*Haematopus palliatus*) (NPS 2016f).

Wood storks (left) and osprey (right) are two bird species of conservation concern found at CUIS (NPS photo).

CUIS is also known for its nesting population of loggerhead sea turtles (*Caretta caretta*) (Figure 5), a federally threatened species (DeVivo et al. 2008). A total of 43 reptile species, including American alligators (*Alligator mississippiensis*) and four additional endangered or threatened sea turtle species, have been confirmed at the park (NPS 2016f). Nineteen amphibian species also occur at CUIS, with many known to breed within the park (NPS 2016f).

Figure 5. A loggerhead sea turtle on a CUIS beach (NPS photo).

The marshes, tidal creeks, beaches, and marine habitats in and around the park support a diversity of fish and aquatic invertebrates. Commercially important fish species such as black sea bass (*Centropristis striata*), southern flounder (*Paralichthys lethostigma*), red drum (*Sciaenops ocellatus*), and black drum (*Pogonias cromis*) are found in nearshore waters (Alber et al. 2005, Peek et al. 2016). Tidal creeks provide habitat for smaller fish, including killifish (*Fundulus* spp.) and juvenile spot (*Leiostomus xanthurus*) and silver perch (*Bairdiella chrysoura*). The marshes, creeks, and nearshore waters also contain grass shrimp (*Palaemonetes pugio*), blue crabs (*Callinectes sapidus*), stone crabs (*Menippe mercenaria*), mud crabs (*Panopeus* spp.), fiddler crabs (*Uca* spp.), and a variety of snail species (Class Gastropoda). The intertidal beaches offer habitat for sand crabs (*Emerita talpoida*), ghost crabs (*Ocypode quadrata*) (Figure 6), ghost shrimp (*Callianassa* spp.), coquina clams (*Donax* spp.), sand dollars (*Mellita* sp.), moon snails (*Polinices duplicatus*), and polychaete worms (Class Polychaeta) (Alber et al. 2005, Peek et al. 2016). A full list of fish and crustacean species documented at CUIS can be found in Appendix A.

Figure 6. A ghost crab and assorted sea shells on a CUIS beach (SMUMN GSS photo).

Shellfish beds are also found near CUIS, containing species such as the eastern oyster (*Crassostrea virginica*), ribbed mussels (*Geukensia demissa*), and hard clams (*Mercenaria mercenaria*) (Alber et al. 2005, Peek et al. 2016). These shellfish are considered "keystone" species because they provide physical structure for other aquatic organisms to attach to, and because the beds serve as habitat for many invertebrates and small fish (Bergquist et al. 2006, Peek et al. 2016). Shellfish are also filter feeders and can improve water quality, particularly by filtering excessive phytoplankton and dissolved solids from the water (Coen et al. 2007, Peek et al. 2016).

Wilderness Character

The Wilderness Act of 1964 defined wilderness as

> *…an area where the earth and its community of life are untrammeled by man, where man himself is a visitor who does not remain. An area of wilderness is further defined to mean in this Act an area of undeveloped Federal land retaining its primeval character and influence, without permanent improvements or human habitation, which is protected and managed so as to preserve its natural conditions and which (1) generally appears to have been affected primarily by the forces of nature, with the imprint of man's work substantially unnoticeable; (2) has outstanding opportunities for solitude or a primitive and unconfined type of recreation; (3) has at least five thousand acres of land or is of sufficient size as to make practicable its preservation and use in an unimpaired condition; and (4) may also contain ecological, geological, or other features of scientific, educational, scenic, or historical value.* (P.L. 88-577, 16 U.S.C. 1131-1136)

Wilderness character, as described by federal agencies includes the following qualities (NPS 2014b):

- Natural – substantially free from the effects of modern civilization;
- Untrammeled – essentially unhindered and free from the intentional actions of modern human control or manipulation;
- Solitude or a primitive and unconfined type of recreation – provides outstanding opportunities for people to experience wilderness;
- Undeveloped – retains its primeval character and influence, and is essentially without permanent improvement or modern human occupation.

To maintain these qualities, minimizing the intrusion of human activities is key. These intrusions can include sights and sounds from both inside and outside park boundaries. The developed areas closest to CUIS (St. Marys and Fernandina Beach) are near the island's southern end; human structures such as the Fernandina Beach paper mill are visible from southern portions of the park, they generally are hidden from view in the wilderness area further north. However, some taller structures at Kings Bay Naval Base (see Figure 1) or on Jekyll Island and in Brunswick to the north may be visible from the western edge of the wilderness.

Even if human structures and developments are not visible from CUIS during the day, they may produce light that interferes with the park's dark night skies. Although the NPS Night Skies Team (NST) has not assessed the night skies or anthropogenic light interference at CUIS, an estimated anthropogenic light ratio (ALR) is available from a nationwide GIS model. The ALR measures the average anthropogenic sky luminance as a ratio of natural conditions (Moore et al. 2013). A ratio of 1.0 indicates that anthropogenic light is 100% brighter than the natural light from the night sky. Current modeled ALR data for CUIS shows that much of the park has an ALR between 2.0 and 5.0 (Figure 7), or at least 200% brighter than the natural night sky.

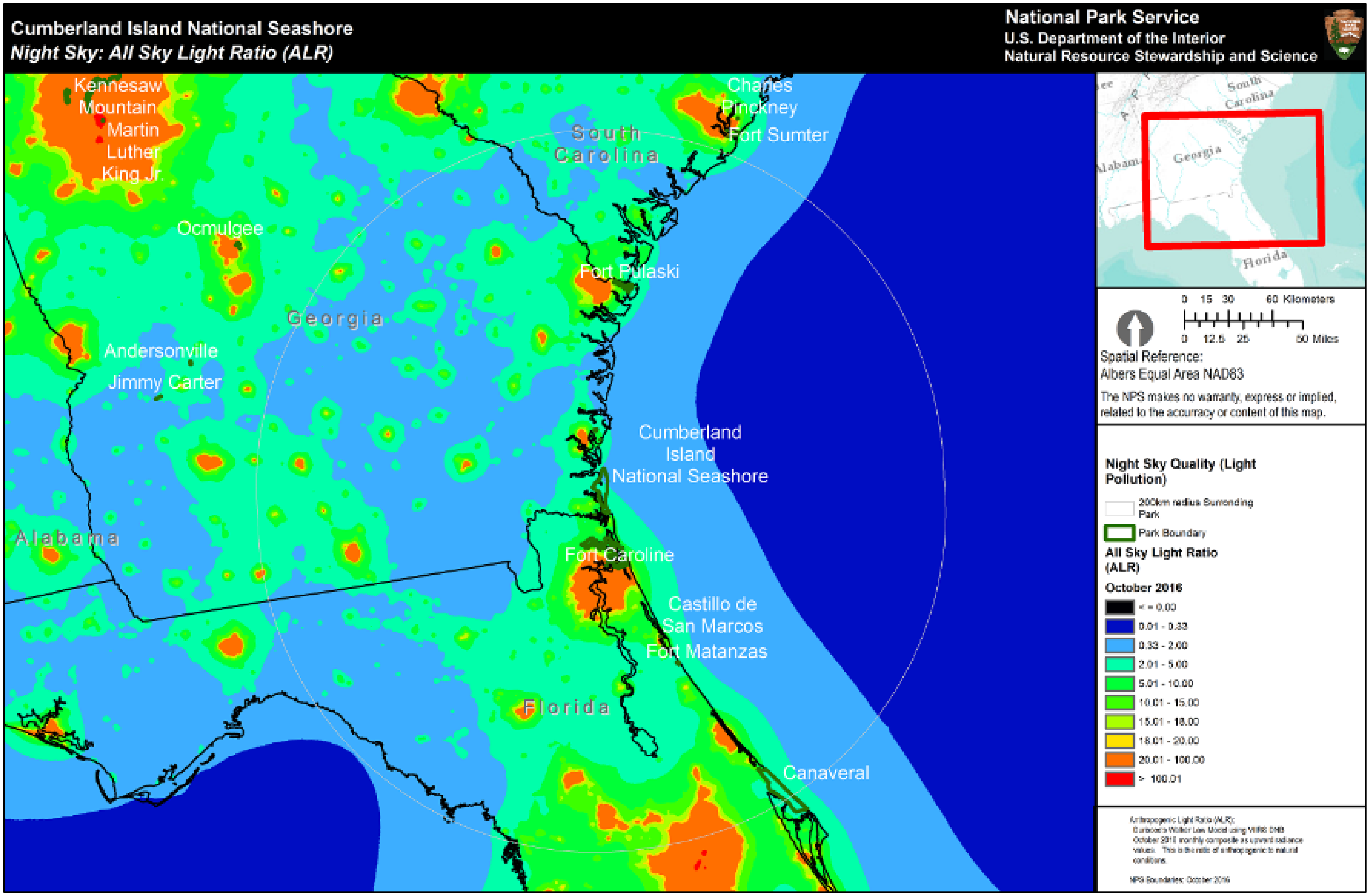


Figure 7. Map displaying modeled ALR data in and around CUIS (figure provided by NPS).

No soundscape monitoring has been conducted at CUIS, but the NPS developed a novel geospatial sound model that predicts natural and existing sound levels within 270 m (886 ft) resolution. The model is based on acoustic data collected at 244 sites and 109 spatial explanatory layers (e.g., location, landcover, hydrology, wind speed, and proximity to noise sources such as roads, railroads, and airports) (Mennitt et al. 2013). The model can also compare natural and existing sound level predictions to provide an estimated impact on the natural acoustic environment from anthropogenic sources. According to the NPS model, the natural sound levels at CUIS range from 36.7-39.2 dBA (A-weighted decibels) with a mean of 37.9 dBA (NPS 2017d). The current existing sound levels are estimated at 36.8-45.2 dBA, with a mean of 37.9 dBA. This suggests that there are still times and places at CUIS where anthropogenic sounds do not impact the soundscape, and when they do, the impact is relatively low (NPS 2017d). The majority of the park's wilderness area experiences little to no impact, on average (Figure 8).

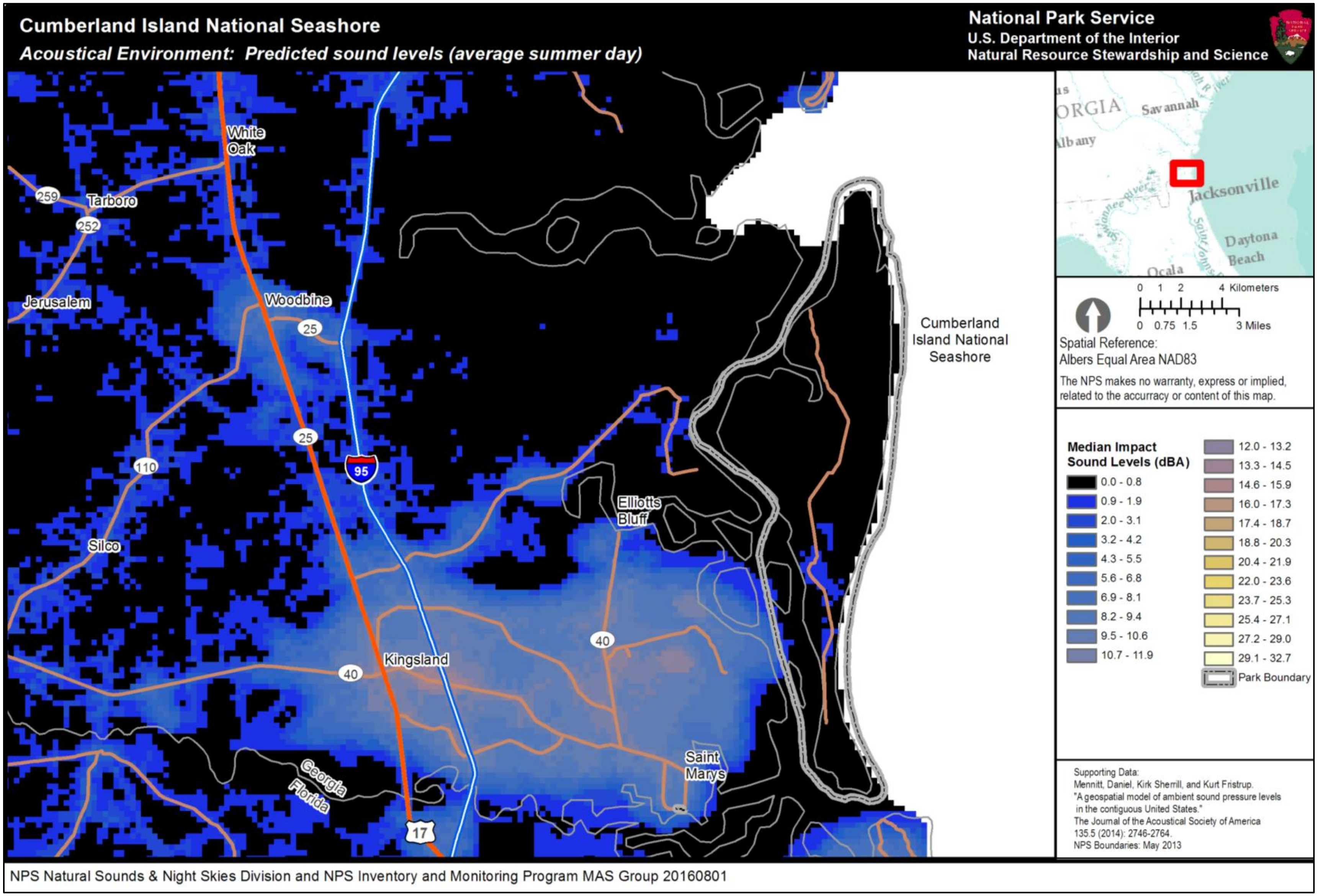


Figure 8. Map displaying modeled median impact sound levels (dBA) in and around CUIS (figure provided by NPS).

2.2.3. Resource Issues Overview

Cumberland Island has "a long history of human occupation and intensive use" (Hillestad et al. 1975, p. 3), from prehistoric indigenous people and colonial forts to cotton plantations and vacation estates. Modification of the land for agriculture (e.g., crop cultivation and grazing) and private estates occurred throughout the 19th century, including alteration of natural vegetation, soils, and hydrology (Hillestad et al. 1975, Dilsaver 2004). Livestock (cattle, horses, pigs) were introduced and often roamed the island freely to graze. Live oak stands were also logged for the ship building industry (Hillestad et al. 1975). The majority of resource issues that CUIS faces today are related to human impact, both historic and current.

Exotic Species

Exotic invasive species pose one of the greatest threats to biodiversity and ecosystem integrity worldwide, with the potential to impact ecological community composition, structure, and function (Mooney et al. 2005, Beard and App 2013). These species can compete with native plants and animals and disrupt ecosystem processes such as nutrient cycling and disturbance regimes (e.g., fire, flooding). According to NPS (2016f), 106 exotic plant species have been documented within CUIS. Many of these are ornamental or cultivated species that were intentionally brought to the island by previous inhabitants, although some are not a threat to invade natural ecosystems. Twenty-five of the exotic plant species confirmed at CUIS are considered invasive by the Georgia Exotic Pest Plant Council (GA-EPPC) (GA-EPPC 2016). During a 2003-2004 survey, only three of these invasive species were found in natural (i.e., undeveloped) areas of the park (Table 4) (Hunt and Langeland 2008). Since 2000, park management has been working with the Southeast Region Exotic Plant Management Team (EPMT) to control tungoil tree (*Vernicia fordii*), Chinese tallow (*Triadica sebifera*), tamarisk (*Tamarix gallica*), and tree-of-heaven (*Ailanthus altissima*) (NPS 2015d; Doug Hoffman, CUIS Biologist, written communication, April 2017).

Table 4. Exotic, invasive plant species documented within CUIS (NPS 2016f) with Georgia invasiveness ranks (GA-EPPC 2016). The final column highlights species found in natural areas during a 2003-2004 survey (Hunt and Langeland 2008).

Scientific Name	Common Name	Invasiveness Rank*	In Natural Areas (2003-04)
Ailanthus altissima	tree-of-heaven	1	–
Albizia julibrissin	silktree, mimosa	1	–
Alternanthera philoxeroides	alligatorweed	1	x
Arundo donax	giant reed	3	–
Cinnamomum camphora	camphortree	2	–
Colocasia esculenta	coco yam, wild taro	3	–
Cynodon dactylon	bermudagrass	2	–
Hedera helix	English ivy	1	–

* 1 = serious problem in natural areas, 2 = moderate problem in natural areas, 3 = minor problem in natural areas, or not yet known to be a problem in Georgia but is known to be a problem in adjacent states.

Table 4 (continued). Exotic, invasive plant species documented within CUIS (NPS 2016f) with Georgia invasiveness ranks (GA-EPPC 2016). The final column highlights species found in natural areas during a 2003-2004 survey (Hunt and Langeland 2008).

Scientific Name	Common Name	Invasiveness Rank*	In Natural Areas (2003-04)
Ligustrum japonicum	Japanese privet	2	–
Ligustrum lucidum	glossy privet	3	–
Lonicera japonica	Japanese honeysuckle	1	–
Lygodium japonicum	Japanese climbing fern	1	–
Melia azedarach	chinaberry	1	x
Myriophyllum aquaticum	parrot feather	2	–
Paspalum notatum	bahiagrass	2	–
Phyllostachys aurea	Golden bamboo	2	–
Poa annua	annual bluegrass	3	–
Sesbania punicea	rattlebox	2	–
Setaria faberi	Japanese bristlegrass	4	–
Sonchus asper	spiny sowthistle	4	–
Sonchus oleraceus	common sowthistle	4	–
Tamarix gallica	French tamarisk	2	x
Verbascum thapsus	common mullein	4	–
Vernicia fordii	tungoil tree	3	–
Wisteria sinensis	Chinese wisteria	1	–

* 1 = serious problem in natural areas, 2 = moderate problem in natural areas, 3 = minor problem in natural areas, or not yet known to be a problem in Georgia but is known to be a problem in adjacent states.

Five non-native mammal species are known to occur at the park: feral horses (*Equus caballus*), feral hogs (*Sus scrofa*), coyote (*Canis latrans*), nine-banded armadillo (*Dasypus novemcinctus*), and black rat (*Rattus rattus*) (NPS 2016f). The first two species – feral horses and hogs – have had a significant impact on the CUIS ecosystem (Dilsaver 2004, DeVivo et al. 2008). Livestock, including horses and hogs, were first brought to the Cumberland Island by Europeans in the late 1560s (Burkingstock et al. 1994, Dolan 2002). By 1785, an island resident wrote that the feral horse population had reached at least 200 (Burkingstock et al. 1994). During the late 1800s and early 1900s, residents introduced several domestic horse breeds to the island to "improve" the feral stock (Burkingstock et al. 1994, p. 2).

The earliest known census of the CUIS feral horse population occurred in 1981, when 144 horses were counted (Bjork 1996b). Intermittent surveys through 1990 utilized varying methodologies and showed an increasing population, reaching an estimated 240 horses in 1990 (Figure 9). During a 1995 census, the count number decreased to 203 horses (Bjork 1996b). Since 2003, horse counts have been conducted annually at CUIS using a consistent ground count methodology. This methodology provides an index to abundance rather than a total population count. During this time, census numbers have fluctuated from around 120 to nearly 150 (excluding 2013, when counts were likely low due to stormy weather) (NPS 2016c). Over the past decade (since 2007), the average census numbers have been around 130 horses. Over 75% of the observed population consists of adult horses, with 0-5 foals and 13-24 juveniles observed per year (NPS 2016c). This age distribution suggests that the horse population has stabilized and is no longer increasing (Hoffman 2015).

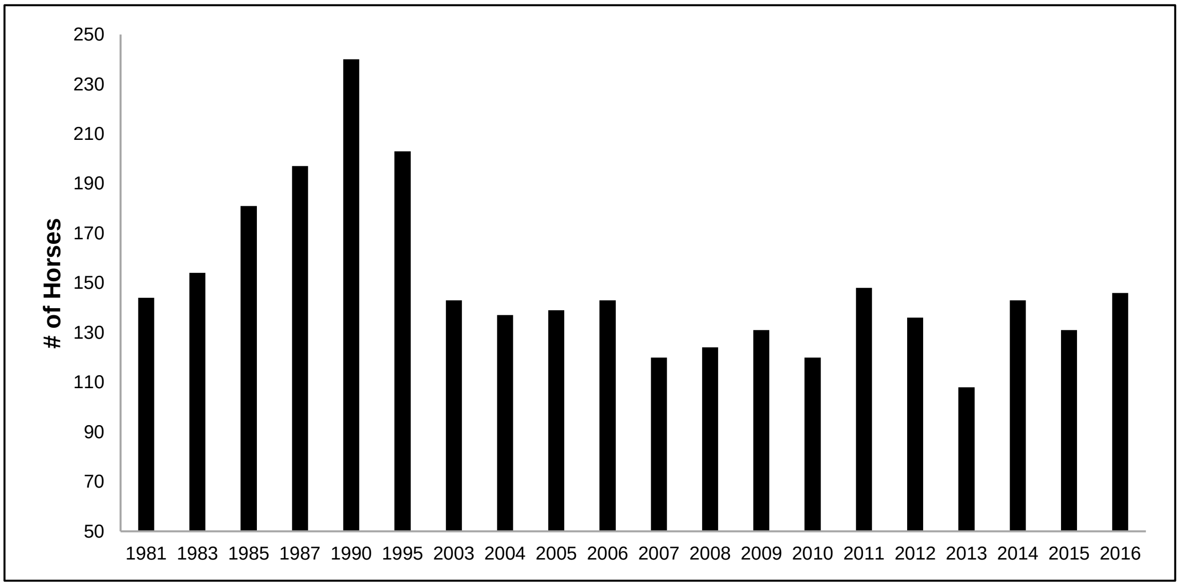


Figure 9. Feral horse survey/count results, 1981-2016 (Bjork 1996b, NPS 2016c). Note that survey methodology was not consistent prior to 2003, and counts were likely low in 2013 due to stormy weather.

While horses are found throughout the island, the salt marshes and interdune areas are two of the most utilized habitats (Figure 10) (Turner 1986, Dolan 2002). Studies of horse impacts at CUIS have found that grazing activity, including vegetation consumption and trampling, significantly reduces vegetative cover, growth, and reproduction in these habitats (Turner 1986, Dolan 2002). Grazing also appears to be altering plant species composition and is likely increasing the vulnerability of dunes and salt marshes to erosion and storm damage (Turner 1986, Dolan 2002). In addition to impacts on vegetation, feral horses compact wetland soils, altering soil properties (e.g., infiltration rates) and disturbing vital soil-dwelling organisms (Noon and Martin 2004). The wastes produced by horses contribute to nutrient enrichment or eutrophication of wetlands and waterbodies, and can contaminate waters with pathogens, including *E. coli* bacteria (Noon and Martin 2004). Together, these impacts make wetland habitats less favorable for native plants, fish, herpetofauna, and invertebrates.

Feral hogs have historically caused damage in nearly every habitat type at CUIS, although their rooting behavior can be particularly detrimental in wetlands and dunes. Rooting can disturb large patches of vegetation and soil, potentially destroying rare plant species and habitat for native wildlife (Dilsaver 2004, Kammermeyer et al. 2011). Feral hogs also consume the eggs of sea turtles, other reptiles, and ground-nesting birds, including several protected species at CUIS (Plauny 2000, Dilsaver 2004, Kammermeyer et al. 2011). Hogs compete for food resources with native wildlife, such as deer, raccoons, squirrels, and birds (Hillestad et al. 1975).

Figure 10. Feral horses grazing in salt marsh on the southern end of CUIS (SMUMN GSS photo).

Since park establishment, the NPS has been working to reduce the number of feral hogs on the island; from 1977-1979 alone, 1,300 hogs were removed (NPS 1984). Removal efforts increased again in the early 2000s, and park staff estimate that about 4,000 hogs have been removed by hunting and trapping since 2000 (Figure 11) (Hoffman, written communication, April 2017). As of early 2017, the population on the island is estimated to be around 120 hogs, and hog predation of sea turtle nests has been virtually eliminated (Figure 12) (Hoffman, personal communication, 7 March 2017).

Figure 11. Feral hogs caught in a trap at CUIS (NPS photo, courtesy of Doug Hoffman).

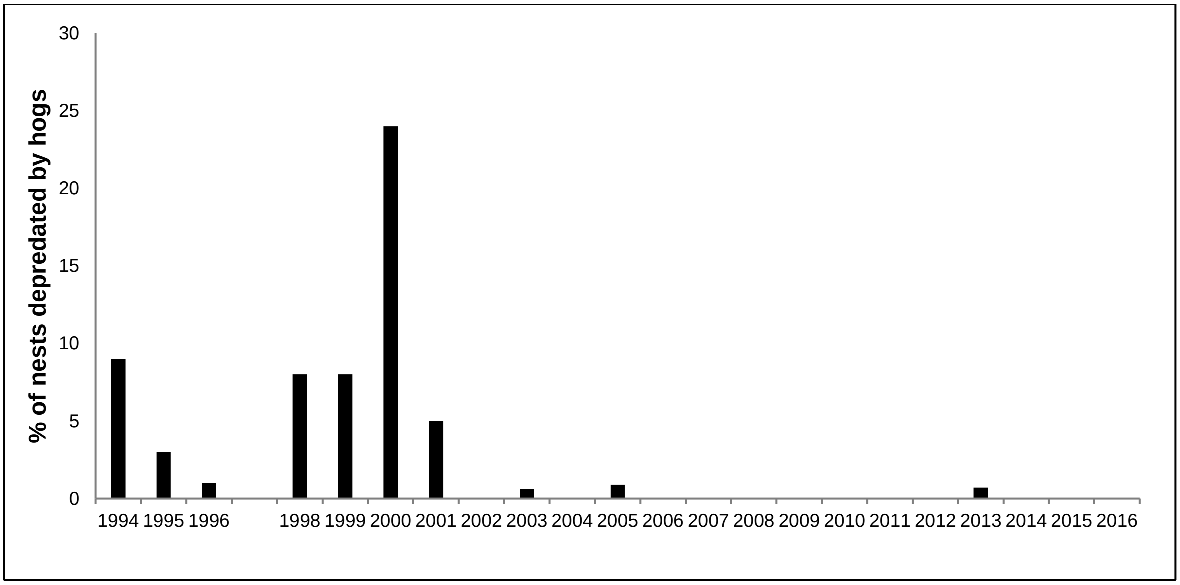


Figure 12. Percentage of sea turtle nests depredated by feral hogs on CUIS beaches, 1994-2016 (NPS 2016e).

Channel and Shoreline Modification

Alterations to maintain the waterways around CUIS, particularly channel access to the Atlantic Ocean south of the island, began over a century ago (Shabica et al. 1993). By the late 1800s, the channel entrance had migrated and inlet shoals had developed in and around the channel due to natural processes. These were considered hazards for boat navigation, and construction of two jetties at the entrance to the channel was initiated in 1881 in an effort to stabilize its location (Shabica et al. 1993). The northern jetty is located on the southern end of CUIS and extends straight into the Atlantic Ocean (Figure 13, Figure 14). The jetties, consisting of large rocks, were completed in 1905 and extended to their current length (5.8 km [3.6 mi] for the CUIS jetty) in 1927 (Shabica et al. 1993).

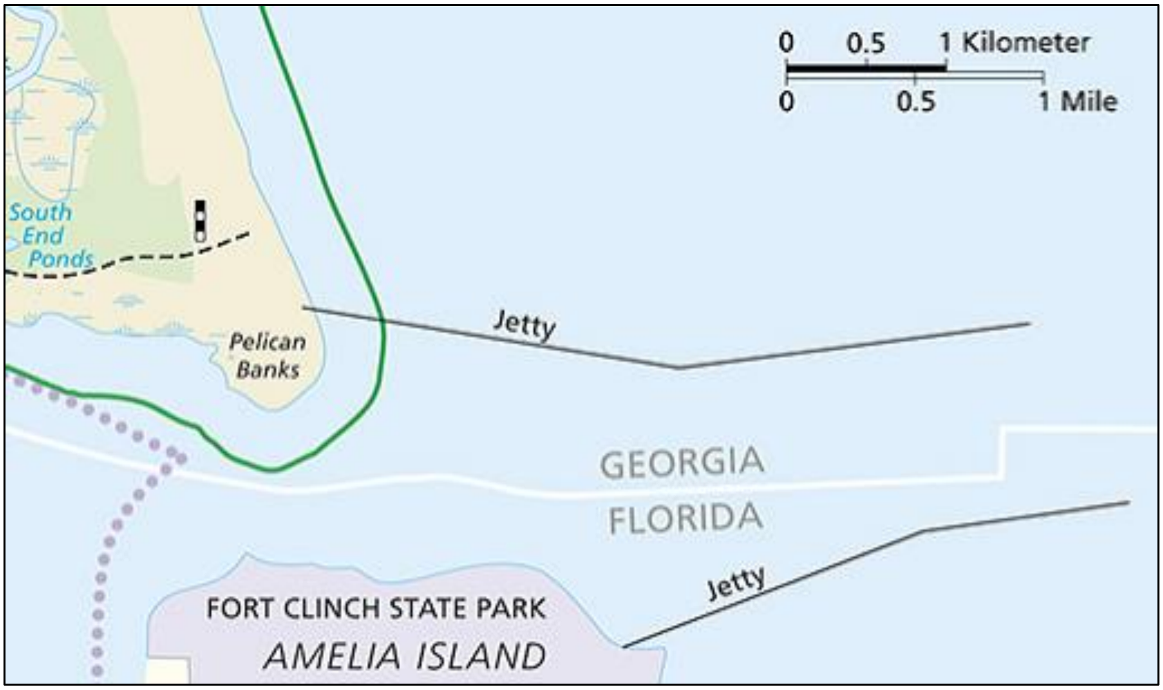


Figure 13. Map of the two jetties; the northern jetty extending from Cumberland Island is pictured below (NPS map).

Figure 14. The south end jetty stretching into the ocean (SMUMN GSS photo).

Since jetty construction, extensive channel dredging has occurred to maintain the waterway for navigation. The channel entrance was originally 5.8 m (19 ft) deep and 1,190 m (3,900 ft) wide (Shabica et al. 1993). Dredging and realignment efforts from 1940 through 1979 maintained the channel at a depth of 12.2 m (40 ft) and a width of 122 m (400 ft), and also extended the channel 13 km (8.1 mi) north from the inlet to the U.S. Navy's Kings Bay Submarine Base. Although most of the dredged material was deposited offshore, some was placed in spoil piles on the south end of

Cumberland Island, in a marshy area on the back-barrier side near Raccoon Keys (Shabica et al. 1993). During the late 1980s, the channel was deepened to accommodate the Trident-class submarines based at Kings Bay. Dredging deepened the channel to 15.5 m (51 ft) and increased channel width to 152 m (500 ft) for a stretch of 35.4 km (22 mi) from the submarine base into the ocean (Shabica et al. 1993). To accomplish this, approximately 26,800,000 m^3 (35,000,000 yd^3) of dredged material was removed from the channel.

Jetty construction and dredging have altered tidal sediment deposition around CUIS, resulting in a net increase in offshore deposition and a net loss in nearshore littoral deposition (Shabica et al. 1993). The reduction in nearshore deposition has contributed to shoreline erosion in some locations, particularly downdrift (south) of the jetties, such as Amelia Island. The relative permeability of the jetty allows sand to pass through and accumulate in extensive shoals on the inlet side of the Cumberland Island jetty (Shabica et al. 1993). Using historical maps, Griffin and Henry (1984) showed that the south end of Cumberland Island expanded seaward by 786 m (2,579 ft) between 1857 (prior to jetty construction) and 1982.

Dredging and other alterations may also increase tidal prism (i.e., the volume of water that flows in and out between high and low tides), which can cause back-barrier channels to encroach further upon adjacent shorelines (Jackson 2006). Jackson (2006, p. 18) stated that the widespread erosion along the CUIS back-barrier shoreline from 1983-2002 was “alarming and suggests factors other than sea-level rise are predominantly influencing change.” In developed areas, such as Plum Orchard and along the Dungeness dock area, efforts have been made to protect segments of the shore from erosion with sea walls, rip-rap, or other stabilizing structures. Unfortunately, erosion continues and may even intensify at the ends of these structures, causing an “end-around” effect where the shorelines bordering these structures are 5-10 m (16-32 ft) further inland than the stabilized shore (Figure 15) (Jackson 2006). Eventually, the stabilized or “armored” areas will begin eroding from the exposed sides.

Figure 15. “End-around” erosion south of a stabilized shoreline area at Plum Orchard (SMUMN GSS photo).

Increasing Development

The southern coast of Georgia has been experiencing an increase in residential and recreational development since the 1990s, with a brief pause during the economic slowdown of the late 2000s (Alber et al. 2005, DeVivo et al. 2008; Fry personal communication, 8 March 2017). Increased development has the potential to threaten many park resources, including air quality, water quality, and wilderness character (e.g., soundscape, night skies, viewshed) (Alber et al. 2005, NPS 2014a). Over the past two decades, multiple new and expanded developments have been proposed within 30 km (18.6 mi) of CUIS (examples are shown in Figure 16). The proposed development closest to CUIS is Point Peter (also called Cumberland Harbour), a 410-ha (1,014-ac) residential area with plans for 1,200 homes and marina space for up to 900 boats just 3.2 km (2 mi) from the southern portion of the park (Alber et al. 2005). This would likely increase recreational boat traffic on the south and west sides of CUIS. Development is also a potential threat within CUIS boundaries, on the approximately 373 ha (922 ac) that are still privately owned (Fry, personal communication, 8 March 2017).

More recently, the Camden County Board of Commissioners proposed developing a commercial space launch facility (i.e., a “spaceport”), which would be located just west of the northern portion of CUIS (Figure 16). If approved and constructed, the proposed spaceport would threaten visitor experience, wilderness character, and numerous natural resources. The NPS expressed their concerns regarding these potential impacts in written comments to the Federal Aviation Administration (FAA) shortly after the agency announced its intent to prepare an Environmental Impact Study (EIS) (Austin 2015).

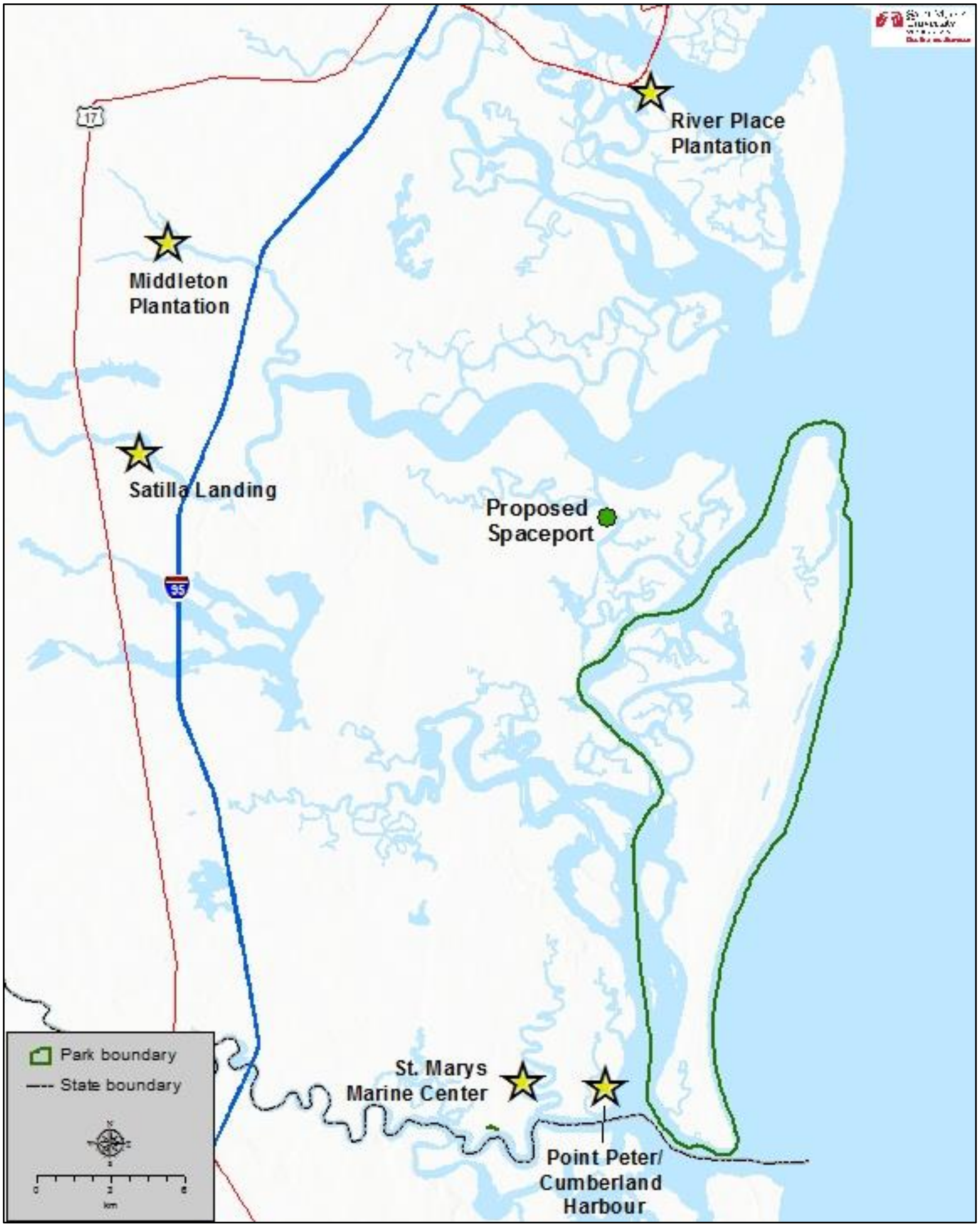


Figure 16. Some of the new or expanded developments proposed and/or completed in the vicinity of CUIS over the past two decades. The yellow stars represent residential developments.

Climate Change

Climate is a key driving factor in the ecological and physical processes influencing park ecosystems throughout the SECN (Davey et al. 2007). As a result of global climate change, temperatures are projected to increase across the southeastern U.S. over the next century (Carter et al. 2014). Warmer air temperatures will increase evaporation rates and plant transpiration (i.e., plant water use), meaning that even if annual precipitation remains constant or slightly increases, overall conditions could still become drier in the future (Carter et al. 2014). Higher air temperatures will lead to higher water temperatures, which will impact sensitive aquatic ecosystems (Peek et al. 2016). In the estuarine environment, for example, many species require a particular temperature change and may not survive if temperatures fluctuate too far or too frequently outside that range. Some organisms are adapted to seasonal temperature variation but rely on temperature cues to initiate behaviors such as migration or spawning (Peek et al. 2016). Climate changes may disrupt the timing of these vital processes. Warmer waters also hold less dissolved oxygen, which is necessary for most aquatic organisms, than cooler waters (Peek et al. 2016).

Warming temperatures will trigger sea level rise (SLR), due to both the thermal expansion of water and the melting of continental ice (IPCC 2013, Peek et al. 2016). Between 1993 and 2010, global SLR averaged 3.2 mm/year (0.13 in/year) (IPCC 2013). At nearby Fernandina Beach, FL, sea level rise has averaged 2.1 mm/year (0.08 in/yr) from 1897-2015, for an overall rise of approximately 20.7 cm (8.2 in) over 100 years (Figure 17) (NOAA 2016). The SLR rate is expected to increase over the remainder of this century, so that total SLR by 2100 will be between 0.28-0.98 m (0.9-3.2 ft) (IPCC 2013).

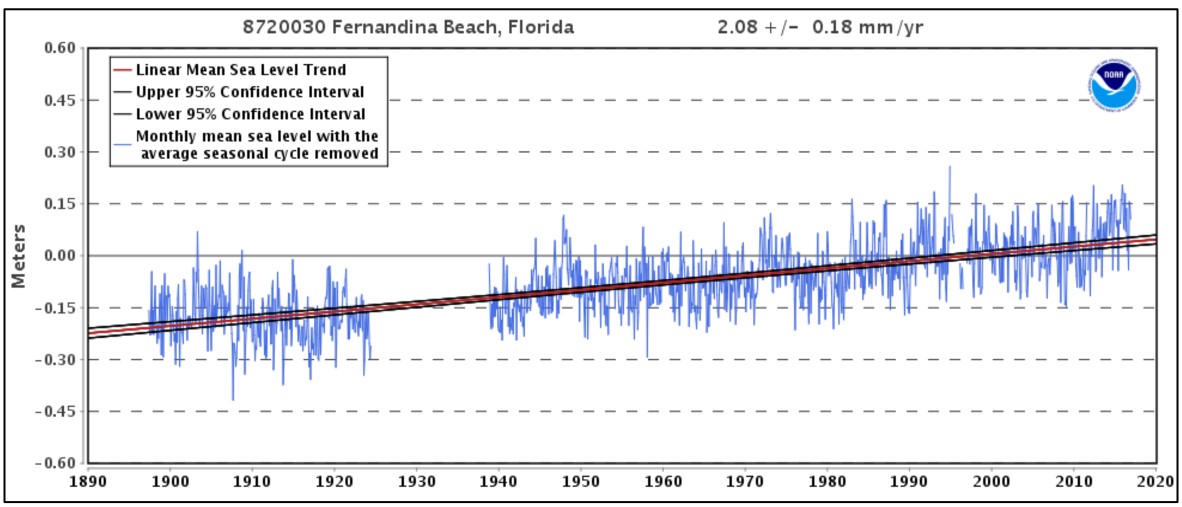


Figure 17. Mean sea level trend for Fernandina Beach, FL (NOAA 2016).

Sea level rise results in the loss of coastal lands, as rising waters inundate additional areas along the shore. Coastal "zones" will essentially shift inland, so that current intertidal areas become subtidal, and supratidal areas (rarely inundated) become intertidal (Peek et al. 2016). Organisms in high intertidal areas, which are adapted to and require frequent aerial exposure during low tide to function, will be particularly sensitive to SLR (Peek et al. 2016). Rising water levels are also likely to alter coastal dynamics, potentially exposing additional shoreline to accelerated erosion, including areas on the island with historically and/or culturally significant features (Jackson 2010, Calhoun and Riley 2016).

The increase in carbon dioxide (CO_2) levels that is contributing to climate change is also causing ocean acidification. Acidification refers to the decrease in ocean pH when CO_2 reacts with seawater to produce carbonic acid (NOAA 2016, Peek et al. 2016). The pH of ocean water is currently around 8.1, but is projected to decline 0.4 pH units by the end of this century (Feely et al. 2009, Peek et al. 2016). A decline in pH will impact many marine organisms, but particularly aquatic invertebrates that build shells or exoskeletons from calcium carbonate (e.g., shellfish, corals, some plankton) (Feely et al. 2009, Peek et al. 2016). Acidification reduces the amount of calcium carbonate dissolved in sea water and available for shell-building. If calcium carbonate saturation levels drop too low, the shells and exoskeletons of these organisms will begin dissolving and thinning (Feely et al. 2009).

2.3. Resource Stewardship

2.3.1. Management Directives and Planning Guidance

The original CUIS General Management Plan (NPS 1984, p. 16) established two primary objectives for park management of natural and recreational resources:

> *To protect and enhance the natural and recreational values of the park by encouraging environmentally compatible park activities and by providing an adequate mainland base to permit achievement of the park's purpose.*

and

> *To manage the seashore, to the extent possible, in ways that enhance the natural geological processes of the barrier island system and mitigate human impacts on these processes.*

To achieve the second objective, the NPS (1984) outlined the following practices

- Limit shoreline and dune stabilization to areas subject to damage or loss occasioned by human use and to allow natural movement of sand beaches and dunes.
- Perpetuate the marsh and freshwater pond environments and forested areas in ways that promote natural ecological succession and minimize the adverse impacts of man's activities.
- Manage wildlife in a manner that restores and enhances the natural ecosystem of the island environment. This is to be accomplished by the following practices.
- ▪ To the greatest extent possible, remove feral hogs from the seashore lands.
- ▪ Preserve or reintroduce rare and endangered species to the island
- ▪ Assure the preservation of dune areas that serve as nesting habitat for wildlife such as birds and loggerhead turtles.
- ▪ Ensure that hunting, fishing, and trapping activities are compatible with the wildlife management program.

The CUIS 2014 Foundation Document (NPS 2014a, p. 6) included the following "purpose statement" for the park:

> *Cumberland Island National Seashore maintains the primitive, undeveloped character of one of the largest and most ecologically diverse barrier islands on the Atlantic coast, while preserving scenic, scientific, and historical values and providing outstanding opportunities for outdoor recreation and solitude.*

The Foundation Document also identified several natural resources/features as Fundamental Resources and Values: intact barrier island system driven by coastal geological and biological processes; live oak maritime forests; pristine beach; and wilderness (primitive and undeveloped character, uncrowded setting) (NPS 2014a). Fundamental Resources and Values are defined as

> *those features, systems, processes, experiences, stories, scenes, sounds, smells, or other attributes determined to warrant primary consideration during planning and management processes because they are essential to achieving the purpose of the park and maintaining its significance.*

2.3.2. Status of Supporting Science

The SECN identified key resources network-wide and for each of its parks that can be used to determine the overall health of the parks. These key resources are called Vital Signs. In 2008, the SECN completed and released a Vital Signs Monitoring Plan (DeVivo et al. 2008); Table 5 shows the SECN Vital Signs selected for monitoring in CUIS.

Table 5. SECN Vital Signs selected for monitoring in CUIS (DeVivo et al. 2008).

Category	SECN Vital Sign	Category 1[a]	Category 2[b]	Category 3[c]
Air and Climate	Ozone	–	X	–
	Wet and Dry Deposition	–	X	–
	Visibility and Particulate Matter	–	X	–
	Air Contaminants	–	X	–
	Weather and Climate	–	X	–
Geology and Soils	Coastal Shoreline Change	X	–	–
	Salt Marsh Elevation	X	–	–
Water	Groundwater Dynamics	–	X	–
	Water Chemistry	X	–	–

[a] **Category 1** represents Vital Signs for which the network has developed protocols and implemented monitoring.

[b] **Category 2** represents Vital Signs that are monitored by the park, another NPS program, or by another federal or state agency using other funding

[c] **Category 3** represents priority Vital Signs for which monitoring has been deferred.

Table 5 (continued). SECN Vital Signs selected for monitoring in CUIS (DeVivo et al. 2008).

Category	SECN Vital Sign	Category 1[a]	Category 2[b]	Category 3[c]
Biological Integrity	Invasive/Exotic Plants	X	–	–
	Marine Invertebrates	–	–	X
	Fish Communities	–	–	X
	Amphibians	X	–	–
	Breeding Forest Birds	X	–	–
	Small Mammals	–	–	X
	Plant Communities	X	–	–
	Shorebirds (T&E species)	–	–	X
	T&E Species	–	X	–
Human Use	Fisheries Take	–	X	–
	Visitor Use	–	X	–
Landscapes (Ecosystem Patterns and Processes)	Fire and Fuel Dynamics	X	–	–
	Land Cover and Use	X	–	–

[a] **Category 1** represents Vital Signs for which the network has developed protocols and implemented monitoring.

[b] **Category 2** represents Vital Signs that are monitored by the park, another NPS program, or by another federal or state agency using other funding

[c] **Category 3** represents priority Vital Signs for which monitoring has been deferred.

3. Study Scoping and Design

This NRCA is a collaborative project between the NPS and SMUMN GSS. Project stakeholders include the CUIS resource management team, and SECN Inventory and Monitoring Program staff. Before embarking on the project, it was necessary to identify the specific roles of the NPS and SMUMN GSS. Preliminary scoping meetings were held, and a task agreement and a scope of work document were created cooperatively between the NPS and SMUMN GSS.

3.1. Preliminary Scoping

A preliminary scoping meeting was held from 7-9 March 2017. At this meeting, SMUMN GSS, SECN, and park staff confirmed that the purpose of the CUIS NRCA was to evaluate and report on current conditions, critical data and knowledge gaps, and selected existing and emerging resource condition influences of concern to CUIS managers. Certain constraints were placed on this NRCA, including the following:

- Condition assessments are conducted using existing data and information;
- Identification of data needs and gaps is driven by the project framework categories;
- The analysis of natural resource conditions includes a strong geospatial component;
- Resource focus and priorities are primarily driven by CUIS resource management;

This condition assessment provides a "snapshot-in-time" evaluation of the condition of a select set of park natural resources that were identified and agreed upon by the project team. Project findings will aid CUIS resource managers in the following objectives:

- Develop near-term management priorities (how to allocate limited staff and funding resources);
- Engage in watershed or landscape scale partnership and education efforts;
- Consider new park planning goals and take steps to further these;
- Report program performance (e.g., Department of Interior Strategic Plan "land health" goals, Government Performance and Results Act [GPRA]).

Specific project expectations and outcomes included the following:

- For key natural resource components, consolidate available data, reports, and spatial information from appropriate sources including CUIS resource staff, the NPS Integrated Resource Management Application (IRMA) website, Inventory and Monitoring Vital Signs, and available third-party sources. The NRCA report will provide a resource assessment and summary of pertinent data evaluated through this project;
- When appropriate, define a reference condition so that statements of current condition may be developed. The statements will describe the current state of a particular resource with respect to an agreed upon reference point;
- Clearly identify "management critical" data (i.e., those data relevant to the key resources). This will drive the data mining and gap definition process;

- Where applicable, develop GIS products that provide spatial representation of resource data, ecological processes, resource stressors, trends, or other valuable information that can be better interpreted visually;
- Utilize “gray literature” and reports from third party research to the extent practicable.

3.2. Study Design

3.2.1. Indicator Framework, Focal Study Resources and Indicators

Selection of Resources and Measures

As defined by SMUMN GSS in the NRCA process, a “framework” is developed for a park or preserve. This framework is a way of organizing, in a hierarchical fashion, bio-geophysical resource topics considered important in park management efforts. The primary features in the framework are key resource components, measures, stressors, and reference conditions.

“Components” in this process are defined as natural resources (e.g., birds), ecological processes or patterns (e.g., natural fire regime), or specific natural features or values (e.g., geological formations) that are considered important to current park management. Each key resource component has one or more “measures” that best define the current condition of a component 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 component. In addition to measures, current condition of components may be influenced by certain “stressors,” which are also considered during assessment. A “stressor” is defined as any physical, biological, or chemical agent that induces adverse changes within a component (EPA 2016b). These typically refer to anthropogenic factors that adversely affect natural ecosystems, but may also include natural processes or disturbances such as floods, fires, or predation.

During the CUIS NRCA scoping process, key resource components were identified by NPS staff and are represented as “components” in the NRCA framework. While this list of components is not a comprehensive list of all the resources in the park, it includes resources and processes that are unique to the park in some way, or are of greatest concern or highest management priority in CUIS. Several measures for each component, as well as known or potential stressors, were also identified in collaboration with NPS resource staff.

Selection of Reference Conditions

A “reference condition” is a benchmark to which current values of a given component’s measures can be compared to determine the condition of that component. A reference condition may be a historical condition (e.g., flood frequency prior to dam construction on a river), an established ecological threshold (e.g., EPA standards for air quality), or a targeted management goal/objective (e.g., a bison herd of at least 200 individuals) (adapted from Stoddard et al. 2006).

Reference conditions in this project were identified during the scoping process using input from NPS resource staff. In some cases, reference conditions represent a historical reference before human activity and disturbance was a major driver of ecological populations and processes, such as “pre-fire suppression." In other cases, peer-reviewed literature and ecological thresholds helped to define appropriate reference conditions.

Finalizing the Framework

An initial framework was adapted from the organizational framework outlined by the H. John Heinz III Center for Science's "State of Our Nation's Ecosystems 2008" (Heinz Center 2008). Key resources for the park were adapted from the SECN Vital Signs monitoring plan (DeVivo et al. 2008). This initial framework was presented to park resource staff to stimulate meaningful dialogue about key resources that should be assessed. Significant collaboration between SMUMN GSS analysts and NPS staff was needed to focus the scope of the NRCA project and finalize the framework of key resources to be assessed.

The NRCA framework was finalized and accepted by NPS staff at the end of March 2017. The framework contains a total of 10 components (Table 6) and was used to drive analysis in this NRCA. This framework outlines the components (resources), most appropriate measures, known or perceived stressors and threats to the resources, and the reference conditions for each component for comparison to current conditions.

Table 6. Cumberland Island National Seashore natural resource condition assessment framework.

Cumberland Island National Seashore NRCA Framework **Natural Resource Condition Assessment**			
Component	*Measures (Significance Level)*	*Stressors*	*Reference Condition*
Biotic Composition			
Ecological Communities			
Upland Forest Community	Upland forest acreage (3), upland forest plant species diversity (3), oak maritime forest acreage (3), oak maritime forest recruitment (3), longleaf pine acreage (3), longleaf pine recruitment (3), red bay presence/persistence (2)	Wildlife browsing of saplings, fire suppression, understory density, pests and pathogens, feral hog rooting, climate change	No realistic reference available; Frost et al. (2011) provides some insight, condition will be based on best professional judgement
Freshwater Wetlands	Total acreage (3), acreage by wetland type (3), plant species diversity by type (3), water quality (3), soil quality (2)	Feral horse and hog activity, fire suppression, climate change, salt water intrusion, dune encroachment, roads and trails	Same as above
Salt Marshes	Total acreage (3), percent of areas grazed vs. non-grazed (3)	Feral horse and hog impacts, erosion along shorelines and creeks/channels, boat wakes, rising tide levels, dredge spoil piles, roads and trails, sudden marsh dieback	Same as above
Interdune Communities	Acreage of communities (3), plant diversity (3)	Disturbance from feral horse and hog activity, dune migration and loss, prolonged drought, severe storm impacts	Same as above
Wildlife			
Mammals	Species richness (3), mesocarnivore species richness (3), deer population size (2)	Non-native mammals, disease (including white nose syndrome), interspecies competition, drought	Historic records for species richness (Bangs 1898, Hillestad et al. 1975); no reference for deer population size
Birds	Species richness (3), shorebird nesting numbers (3), shorebird fledging success (3), wading bird nesting numbers (2), wading bird fledging success (2)	Habitat loss and degradation, predation, human disturbance/ recreation, extreme weather events (e.g., storms, drought), fire suppression	Unknown; condition will be based on best professional judgement

Table 6 (continued). Cumberland Island National Seashore natural resource condition assessment framework.

Cumberland Island National Seashore NRCA Draft Framework Natural Resource Condition Assessment			
Component	*Measures (Significance Level)*	*Stressors*	*Reference Condition*
Biotic Composition (continued)			
Wildlife			
Herpetofauna	Amphibian species richness (3), amphibian species abundance (3), sea turtle species richness (1), sea turtle nesting numbers (2), sea turtle hatch success (3), gopher tortoise population size (1), gopher tortoise burrow count (1)	Habitat loss, drought, fire suppression, climate change, disease, predation Sea turtle-specific: fishery-related injuries/mortality, light pollution, strandings due to lack of offshore food resources	Tuberville et al. (2005) for amphibian species richness; CUIS monitoring (since mid-1980s) for sea turtle measures; current condition of gopher tortoise will serve as reference
Environmental Quality			
Water Quality	Nutrients (3), fecal coliform bacteria (3), salinity (3), dissolved oxygen (2), specific conductance (2), pH (2)	Feral horse and hog activity, atmospheric deposition, eutrophication, saltwater intrusion, roads and trails, abandoned artesian wells, fires	Range of values from USGS report (Frick et al. 2002)
Air Quality	Ozone (3), atmospheric deposition of sulfur/nitrogen (3), atmospheric deposition of mercury (3), visibility (3)	Power plants and industrial facilities (especially paper mills), Brunswick Superfund site, vehicle emissions, wildland fires	NPS ARD standards
Physical Characteristics			
Barrier Island Geomorphology	Back barrier shoreline change (3), ocean shoreline change (3), dunefield dynamics (3)	Erosion, natural ocean/inlet processes, storm events, hardened shoreline structures, feral animals, dredging (mostly historic), boat traffic, increased visitor use	Historic shoreline change rates and recent change rates from other Georgia barrier islands; no reference available for dunefield dynamics

3.2.2. General Approach and Methods

This study involved gathering and reviewing existing literature and data relevant to each of the key resource components included in the framework. No new data were collected for this study; however, where appropriate, existing data were further analyzed to provide summaries of resource condition or to create new spatial representations. After all data and literature relevant to the measures of each component were reviewed and considered, a qualitative statement of overall current condition was created and compared to the reference condition when possible.

Data Mining

The data mining process (i.e., acquiring as much relevant data about key resources as possible) began at the initial scoping meeting, at which time CUIS staff provided data and literature in multiple forms, including: NPS reports and monitoring plans, reports from various state and federal agencies, published and unpublished research documents, databases, tabular data, and charts. GIS data were provided by NPS staff. Additional data and literature were also acquired through online bibliographic literature searches and inquiries on various state and federal government websites. Data and literature acquired throughout the data mining process were inventoried and analyzed for thoroughness, relevancy, and quality regarding the resource components identified at the scoping meeting.

Data Development and Analysis

Data development and analysis was highly specific to each component in the framework and depended largely on the amount of information and data available for the component and recommendations from NPS reviewers and sources of expertise including NPS staff from CUIS and the SECN. Specific approaches to data development and analysis can be found within the respective component assessment sections located in Chapter 4 of this report.

Scoring Methods and Assigning Condition

Significance Level

A set of measures are useful in describing the condition of a particular component, but all measures may not be equally important. A “Significance Level” represents a numeric categorization (integer scale from 1-3) of the importance of each measure in assessing the component’s condition; each Significance Level is defined in Table 7. This categorization allows measures that are more important for determining condition of a component (higher significance level) to be more heavily weighted in calculating an overall condition. If a measure is given a Significance Level of 1, it is thought to be of low importance when determining the overall condition of the component. For this reason, measures with a Significance Level of 1 are not discussed in detail in the Current Condition and Trends section of a component’s chapter. Significance Levels were determined for each component measure in this assessment through discussions with park staff and/or outside resource experts.

Table 7. Scale for a measure's Significance Level in determining a components overall condition.

Significance Level (SL)	Description
1	Measure is of low importance in defining the condition of this component.
2	Measure is of moderate importance in defining the condition of this component.
3	Measure is of high importance in defining the condition of this component.

Condition Level

After each component assessment is completed (including any possible data analysis), SMUMN GSS analysts assign a Condition Level for each measure on a 0-3 integer scale (Table 8). This is based on all the available literature and data reviewed for the component, as well as communications with park and outside experts.

Table 8. Scale for Condition Level of individual measures.

Condition Level (CL)	Description
0	**GOOD CONDITION**. No net loss, degradation, negative change, or alteration.
1	Of **LOW** concern. Signs of limited and isolated degradation of the component.
2	Of **MODERATE** concern. Pronounced signs of widespread and uncontrolled degradation.
3	Of **SIGNIFICANT** concern. Nearing catastrophic, complete, and irreparable degradation of the component.

Weighted Condition Score

After the Significance Levels (SL) and Condition Levels (CL) are assigned, a Weighted Condition Score (WCS) is calculated via the following equation:

$$WCS = \frac{\sum_{i=1}^{\#\ of\ measures} SL_i * CL_i}{3 * \sum_{i=1}^{\#\ of\ measures} SL_i}$$

The resulting WCS value is placed into one of three possible categories: resource is in good condition (WCS = 0.0 – 0.33); condition warrants moderate concern (WCS = 0.34 - 0.66); and condition warrants significant concern (WCS = 0.67 to 1.00). Table 9 and Table 10 display and describe the symbology used to represent a component's condition in this assessment. The colored circles represent the categorized WCS; red circles signify a significant concern, yellow circles a moderate concern, and green circles are in good condition. White circles are used to represent situations in which SMUMN GSS analysts and park staff felt there was currently insufficient data to make a statement about the condition of a component. The border of the circles represents SMUMN GSS's confidence in the assessment of current condition; bold borders indicate high confidence, normal borders indicate medium confidence, and a dashed-border indicates low confidence. The arrows inside the circles indicate the trend of the condition of a resource component, based on data and literature from the past 5-10 years, as well as expert opinion. An upward pointing arrow indicates the condition of the component has been improving in recent times. An arrow that points to the left and right indicates a stable condition or trend and an arrow pointing down indicates a decline in the condition of a component in recent times. These are only used when it is appropriate to comment on the trend of condition of a component. An empty circle with no arrow is reserved for situations in which the trend of the component's condition is currently unknown.

Table 9. 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 10. Example indicator symbols and descriptions of how to interpret them in WCS tables.

Symbol Example	Description of Symbol
	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.

Preparation and Review of Component Draft Assessments

The preparation of draft assessments for each component was a highly cooperative process among SMUMN GSS analysts, and CUIS and SECN staff. Though SMUMN GSS analysts rely heavily on peer-reviewed literature and existing data in conducting the assessment, the expertise of NPS resource staff also plays a significant and invaluable role in providing insights into the appropriate direction for analysis and assessment of each component. This step is especially important when data or literature are limited for a resource component.

The process of developing draft documents for each component began with a detailed phone or conference call with an individual or multiple individuals considered local experts on the resource components under examination. These conversations were a way for analysts 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. Upon completion, draft assessments were forwarded to component experts for initial review and comments.

Development and Review of Final Component Assessments

Following review of the component draft assessments, analysts used the review feedback from resource experts to compile the final component assessments. As a result of this process, and based on the recommendations and insights provided by CUIS resource staff and other experts, the final component assessments represent the most relevant and current data available for each component and the sentiments of park resource staff and resource experts.

Format of Component Assessment Documents

All 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 park and the context within which it occurs in the park setting. For example, a component may represent a unique feature of the park, it may be a key process or resource in park ecology, or it may be a resource that is of high management priority in the park. Also emphasized are interrelationships that occur among the featured component and other resource components included in the NRCA.

Measures

Resource component measures were defined in the scoping process and refined through dialogue with resource experts. Those measures deemed most appropriate for assessing the current condition of a component are listed in this section, typically as bulleted items.

Reference Conditions/Values

This section explains the reference condition determined for each resource component as it is defined in the framework. Explanation is provided as to why specific reference conditions are appropriate or logical to use. Also included in this section is a discussion of any available data and literature that explain and elaborate on the designated reference conditions. If these conditions or values originated with the NPS experts or SMUMN GSS analysts, an explanation of how they were developed is provided.

Data and Methods

This section includes a discussion of the data sets used to evaluate the component and if or how these data sets were adjusted or processed as a lead-up to analysis. If adjustment or processing of data involved an extensive or highly technical process, these descriptions are included in an appendix for the reader or a GIS metadata file. Also discussed is how the data were evaluated and analyzed to determine current condition (and trend when appropriate).

Current Condition and Trend

This section presents and discusses in-depth key findings regarding the current 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 Stressor Factors

This section provides a summary of the threats and stressors that may impact the resource and influence to varying degrees the current condition of a resource component. Relevant stressors were described in the scoping process and are outlined in the NRCA framework. However, these are elaborated on in this section to create a summary of threats and stressors based on a combination of available data and literature, and discussions with resource experts and NPS natural resources staff.

Data Needs/Gaps

This section outlines critical data needs or gaps for the resource component. Specifically, what is discussed is 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. In some cases, the data needs/gaps are significant enough to make it inappropriate or impossible to determine condition of the resource component. In these cases, stating the data needs/gaps is useful to natural resources staff seeking to prioritize monitoring or data gathering efforts.

Overall Condition

This section provides a qualitative summary statement of the current condition that was determined for the resource component using the WCS method. Condition is determined after thoughtful review of available literature, data, and any insights from NPS staff and experts, which are presented in the Current Condition and Trend section. 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 analysts attribute to the condition of the resource component. Also included in this section are the graphics used to represent the component condition.

Sources of Expertise

This is a listing of the individuals (including their title and affiliation with offices or programs) who had a primary role in providing expertise, insight, and interpretation to determine current condition (and trend when appropriate) for each resource component. Sources are listed alphabetically by last name.

4. Natural Resource Conditions

This chapter presents the background, analysis, and condition summaries for the 10 key resource components in the project framework. The following sections discuss the key resources and their measures, stressors, and reference conditions. The summary for each component is arranged around the following sections:

1. Description
2. Measures
3. Reference Condition
4. Data and Methods
5. Current Condition and Trend (including threats and stressor factors, data needs/gaps, and overall condition)
6. Sources of Expertise
7. Literature Cited

The order of components follows the project framework (Table 6):

4.1 Upland Forest Community

4.2 Freshwater Wetlands

4.3 Salt Marsh

4.4 Interdune Communities

4.5 Mammals

4.6 Birds

4.7 Herpetofauna

4.8 Water Quality

4.9 Air Quality

4.10 Barrier Island Geomorphology

4.1. Upland Forest Community

4.1.1. Description

Upland forests account for nearly half of the natural vegetation at CUIS, and approximately 28% of the total island area (McManamay 2017). The composition of these forests differs depending on soil types, past land use, and fire history (Hillestad et al. 1975). The current upland forests in the park are largely secondary growth, since the majority of the original forests were cleared for agriculture, timber, or other historic human uses (1700s through mid-1900s) (Hillestad et al. 1975). However, these forests continue to provide valuable habitat for a variety of plant and animal species (NPS 2014a).

The various forest communities of CUIS are dominated by a mix of oak and pine species, with live oak present in nearly all communities and dominant in many (Table 11) (Hillestad et al. 1975). The distinctive appearance of these live oaks, with their spreading horizontal branches draped in Spanish moss, "has an aesthetic charm that many visitors associate with Cumberland Island" (NPS 2014a, p. 9). Some of the oldest and largest live oaks at CUIS occur in maritime forests, which are communities of broadleaf evergreen trees and shrubs occurring on barrier islands and adjacent mainland coasts from North Carolina to Florida (Bellis 1995). Oak maritime forests are adapted to salt spray exposure, high winds from oceanic storms, and limited freshwater availability (Bellis 1995). Maritime forests are one of the rarest and least studied coastal vegetation communities, and CUIS supports one of the largest remaining oak maritime forests in the U.S. (Bellis 1995, NPS 2014a).

Live oaks draped in Spanish moss, with an understory of saw palmetto (SMUMN GSS photo).

Table 11. Upland forest community vegetation types at CUIS and their common plant species, as described by a McManamay (2017).

Upland Forest Type	Common Plant Species
Live Oak - (Cabbage Palmetto) Forest Alliance / Southeastern Florida Maritime Hammock	live oak, sand live oak (*Quercus geminata*), saw palmetto, cabbage palmetto (*Sabal palmetto*), dwarf palmetto (*Sabal minor*), devilwood (*Osmanthus americanus*), rusty staggerbush (*Lyonia ferruginea*), Spanish moss

Upland Forest Type	Common Plant Species
Longleaf Pine / (Sand Laurel Oak, Turkey Oak) / Wax-myrtle / Southern Wiregrass Woodland	longleaf pine, black cherry (*Prunus serotina*), turkey oak (*Q. laevis*), myrtle oak (*Q. myrtifolia*), Darlington's oak (*Q. hemisphaerica*), deerberry (*Vaccinium stamineum*), wax myrtle (*Morella cerifera*)
Maritime Southern Yellow Pine Forest	slash pine (*Pinus elliottii*), longleaf pine, loblolly pine (*P. taeda*), Darlington's oak, live oak, myrtle oak, wax myrtle, deerberry, rusty staggerbush, muscadine grape (*Vitis rotundifolia*)
Slash Pine - (Longleaf Pine) Managed Forest	slash pine, thoroughwort (*Eupatorium* spp.), whip nutrush (*Scleria triglomerata*), greenbriars (*Smilax* spp.)

The park also supports a few remnant stands of old-growth longleaf pine (*Pinus palustris*) and associated fire-dependent plant species (Figure 18) (Frost et al. 2011). Longleaf pine communities were common on Cumberland Island, particularly in the northern portion, prior to European settlement. These largely open forests have diverse herbaceous layers and were maintained historically by frequent fires. Due to decades of fire suppression, CUIS's longleaf pine communities are becoming less open and are transitioning towards other pine-oak forest types (Frost et al. 2011).

Figure 18. Longleaf pine near Terrapin Point on the north end of CUIS (Frost et al. 2011).

4.1.2. Measures

- Upland forest acreage
- Upland forest plant species diversity
- Oak maritime forest acreage
- Oak maritime forest recruitment
- Longleaf pine acreage
- Longleaf pine recruitment
- Redbay (*Persea borbonia*) presence/persistence

4.1.3. Reference Condition/Values

The ideal reference condition for the upland forest community at CUIS would be the condition of the forest prior to European settlement. However, given the magnitude and duration of human use and alteration of the island's vegetation, returning the forest to pre-settlement condition is no longer practical. In addition, information from this time period is limited. Frost et al. (2011) may provide some insight into the extent of various forest communities prior to settlement, based on historical maps and documents. For this assessment, best professional judgement will be used to evaluate condition. Information presented in this report on current condition can be used as a baseline for assessing condition in the future.

4.1.4. Data and Methods

The earliest scientific study of Cumberland Island's vegetation communities was conducted by Hillestad et al. (1975) from November 1972 through September 1973. This broad study, which included detailed vegetation sampling, analysis, and mapping, sought to "inventory and describe the natural resources within the boundaries of the Cumberland Island National Seashore and to generally describe their functions and relationships" (Hillestad et al. 1975, p. 1). Vegetation on the island's interior, including upland forests, was sampled at 250 points along all accessible roads (30.5 m [100 ft] off the road) at 0.3-km (0.2-mi) intervals on alternating sides (Hillestad et al. 1975). Data were collected at each point for tree, woody understory, and herbaceous groundcover species. These data, along with aerial photo and collateral data (e.g., topography, soils) interpretation, were used to create a map showing the major vegetation communities on the island.

Bratton and Kramer (1989) studied the response of live oak regeneration to wildlife browsing (e.g., horses, hogs, and white-tailed deer) in CUIS's oak-pine forests. Areas with and without browsing were sampled. To eliminate browsing, researchers set up two types of exclosures (i.e., fenced plots to keep out selected wildlife): one that excluded just feral horses, and one that excluded horses, hogs, and deer. Exclosures were established in 1985, along with unfenced control plots, and live oak sprouts were measured along transects within each plot during the spring from 1986-1989 (Bratton and Kramer 1989).

Lieske et al. (1990) also studied oak regeneration, but included 10 sites spread across the island, from south to north (Figure 19). At each site, 10 trees were selected (live or laurel oak [*Q. laurifolia*]), five with significant evidence of regeneration and five with little or no regrowth. Two 2x4 m (6.6x13.1 ft) plots were established around each tree and the total number of tree sprouts and seedlings, along with the height of the tallest sprout, were recorded (Lieske et al. 1990). Each of the plots was also divided into four quadrats, and the heights of the closest eight oak sprouts within a 1-m (3.3 ft) arc of each quadrat were recorded. Sampling occurred during late February and early March of 1990 (Lieske et al. 1990).

Figure 19. General locations of sites sampled by Lieske et al. (1990) to study oak regeneration.

Zomlefer et al. (2008) completed an intensive floristic survey of CUIS, resulting in an annotated plant species list with information on general community type (i.e., habitat) and relative abundance. Surveys were conducted from April 2004 through September 2006 and included all island habitats.

An addendum to the species list was published later (Zomlefer and Kruse 2011), based on surveys from October 2006-October 2007.

In 2009, the NPS I&M program initiated a vegetation monitoring project at CUIS as part of the Vital Signs monitoring program (Byrne et al. 2012). Data were collected at 30 sampling locations across the island during September and October that year. Each sampling location consisted of a circular plot with a 15-m (49-ft) radius and subplots along six transects radiating from the center point (Byrne et al. 2012). Data collected include species composition, canopy cover, herbaceous cover, and canopy-species seedling frequency. Monitoring was repeated in March of 2012 (Heath and Byrne 2014) and again in 2016, although these most recent results were not yet available for inclusion in the NRCA. Over two-thirds of the sampling locations fell within upland forest communities, primarily live oak-palmetto vegetation communities.

Frost et al. (2011) developed maps of historical vegetation and original fire regimes on Cumberland Island based on 18th century English land grant surveys, the McKinnon (1802) survey of 1800-1802, and other sources from the Georgia State Archives. The resulting map and vegetation descriptions provide the "best approximation of original vegetation as it existed at the time of European settlement, under the original natural fire regimes" (Frost et al. 2011, p. 20).

The SECN completed an updated vegetation classification and mapping project for CUIS (McManamay 2017). Plant communities were classified into vegetation associations using the U.S. National Vegetation Classification (NVC) (FGDC 2008). NatureServe was consulted to provide a preliminary list of vegetation communities likely to occur at CUIS, and a draft dichotomous key to aid in the field identification of those vegetation community types. The mapping and classification involved a combination of field surveys and aerial imagery interpretation. A total of 81 field plots of various sizes were surveyed for classification purposes from 2006-2007, and aerial imagery was obtained in May 2011. Upon completion of a draft vegetation map in March 2012, field reconnaissance was conducted to finalize the image analysis phase. Field accuracy assessments were completed at 711 locations across the park during December 2012 and 2013 to ensure the FGDC-mandated map accuracy of 80% (McManamay 2017).

4.1.5. Current Condition and Trend

Upland Forest Acreage

Upland forests are one of the most dominant vegetation communities at CUIS. Prior to European settlement, Frost et al. (2011) estimates that 3,172 ha (7,838 ac) of the island was covered by upland forest (Table 12), with live oak, longleaf pine, and slash pine (*Pinus elliottii*) as the most dominant species. The pre-settlement extent of the various upland forest communities, as mapped by Frost et al. (2011), are shown in Figure 20.

Table 12. Extent of upland forest communities within the pre-settlement (around 1600) vegetation of CUIS (Frost et al. 2011).

Upland Forest Type	Area (ha/ac)	Percent of Total Area
Longleaf pine savannah and woodland	941 (2,325)	8.6
Longleaf pine-slash pine woodland	110 (272)	1.0
Mixed longleaf pine-slash pine-live oak woodland and savanna	922 (2,278)	8.5
Slash pine forest	102 (252)	0.9
Fire-maintained live oak-slash pine forest	976 (2,412)	9.0
Live oak-laurel oak	33 (82)	0.3
Fire sheltered live oak forest	88 (217)	0.8
Total	**3,172 (7,838)**	**29.1**

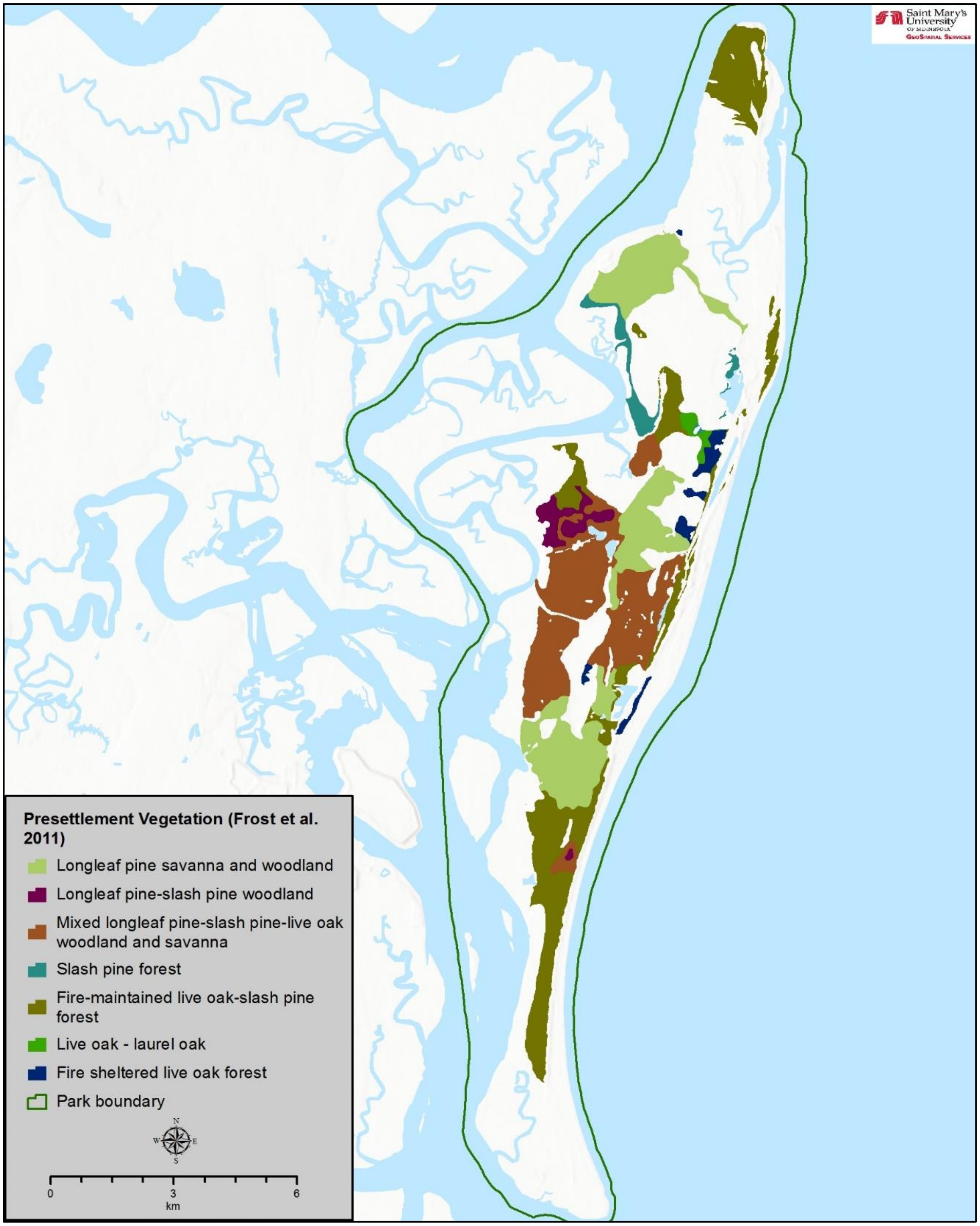


Figure 20. Estimated extent of upland forest communities at CUIS prior to European settlement, as mapped by Frost et al. (2011).

Around the time of park establishment (early 1970s), the total acreage of upland forest at CUIS was estimated at 4,015 ha (9,921 ac) (Hillestad et al. 1975). This accounted for nearly 39% of the total vegetated area mapped (Table 13). Oak-pine and oak-palmetto forests comprised the largest areas (Hillestad et al. 1975)

Table 13. Extent of upland forest community vegetation types at CUIS in 1974 (Hillestad et al. 1975).

Upland Forest Type	Cumberland Island Area (ha/ac)	Little Cumberland Isl. Area (ha/ac)	Total Area (ha/ac)	Percent of Total Area
Mixed oak-hardwood	472 (1,166)	0	472 (1,166)	4.5
Oak-pine	1,497 (3,699)	0	1,497 (3,699)	14.4
Oak-palmetto	1,221 (3,017)	285 (704)	1,506 (3,721)	14.5
Oak-scrub	238 (588)	37 (91)	275 (680)	2.6
Pine-oak scrub	265 (655)	0	265 (655)	2.6
Total	**3,693 (9,125)**	**322 (796)**	**4,015**	**38.7**

The most recent vegetation mapping project (McManamay 2017) identified 4,295 ha (10,613 ac) of upland forest communities (Table 14). The majority of the forests were dominated by live oak, other oaks, saw palmetto, and cabbage palmetto (*Sabal palmetto*). A comparison to Frost et al. (2011) suggests that live oak forests have greatly expanded since settlement, while pine-dominated communities have been lost. The extent of the various upland forest communities, based on 2011 aerial imagery, is shown in Figure 21.

Table 14. Extent of upland forest community vegetation types at CUIS based on 2011 aerial imagery, as reported in McManamay (2017).

Upland Forest Type	Area (ha/ac)	Percent of Total Veg Area
Live Oak - (Cabbage Palmetto) Forest Alliance/Southeastern Florida Maritime Hammock	3,746.6 (9,258)	39.0
Longleaf Pine /(Sand Laurel Oak, Turkey Oak)/Wax-myrtle/Southern Wiregrass Woodland	24.8 (61)	0.3
Maritime Southern Yellow Pine Forest	517.1 (1,278)	5.4
Slash Pine - (Longleaf Pine) Managed Forest	6.0 (15)	>0.1
Total	**4,294.5 (10,612)**	**44.7**

Figure 21. The vegetation of CUIS, as mapped by NPS I&M and NatureServe (McManamay 2017).

Upland Forest Plant Species Diversity

During the earliest known vegetation study at CUIS, Hillestad et al. (1975) listed 102 plant species from upland forest communities (Appendix B). However, the authors noted that this published list only included dominant woody understory species or "those with indicative value" (Hillestad et al. 1975, p. 214), and that tree species with importance values (a measure of species dominance) less than 10 were not listed. Therefore, it is likely that the actual number of plant species in upland forest communities was greater than the 102 that were reported.

Vascular plant surveys by Zomlefer et al. (2008) and Zomlefer and Kruse (2011) documented 161 species in CUIS upland forest communities (pine-oak forest and maritime hammock) (Appendix B). The pine-oak forest community appeared more diverse, supporting nearly twice as many species as the maritime hammock community (Zomlefer et al. 2008).

During SECN vegetation monitoring at CUIS in 2012, a total of 132 plant species were documented in upland forest community sampling plots (Heath and Byrne 2014). With the results of all three surveys combined, nearly 270 total plant species have been observed within the park's upland forest communities (Appendix B). Only nine of these species (~3%) are non-native, and just two are considered invasive: Japanese privet (*Ligustrum japonicum*) and English ivy (*Hedera helix*) (GA-EPPC 2016).

Oak Maritime Forest Acreage

Of the pre-settlement upland forest community types mapped by Frost et al. (2011), three would be considered oak maritime forest: Fire-maintained live oak-slash pine forest, live oak-laurel oak, and fire sheltered live oak forest. Together, these communities covered an estimated 1,097 ha (2,711 ac) of Cumberland Island prior to European settlement (Table 15). Fire-maintained live oak-slash pine, an early successional stage of oak maritime forest, was most prevalent.

Table 15. Extent of oak maritime forest at CUIS, according to various studies over time.

Source	Oak Forest Type	Area (ha/ac)
Frost et al. (2011) - presettlement	Fire-maintained live oak-slash pine forest	976 (2,412)
	Live oak-laurel oak	33 (82)
	Fire sheltered live oak forest	88 (217)
	Total	**1,097 (2,711)**
Hillestad et al. (1975)	Mixed oak-hardwood	472 (1,166)
	Oak-pine	1,497 (3,699)
	Oak-palmetto	1,506 (3,721)
	Total	**3,475 (8,587)**
McManamay 2015	Live Oak - (Cabbage Palmetto) Forest Alliance/ Southeastern Florida Maritime Hammock	**3,747 (9,259)**

The upland forest communities mapped by Hillestad et al. (1975) that are considered oak maritime forest are mixed oak-hardwood, oak-pine, and oak-palmetto. These three communities covered approximately 3,475 ha (8,587 ac) of CUIS around the time of park establishment (Table 15). Oak-palmetto, a later successional stage of oak maritime forest, and the early successional oak-pine forest covered nearly equal areas at this time (Hillestad et al. 1975).

More recently, McManamay (2017) mapped 3,747 ha (9,259 ac) of oak maritime forest between two community types: Live Oak - (Cabbage Palmetto) Forest Alliance and Southeastern Florida Maritime Hammock (Table 15). A large portion of the oak maritime forest lies within the wilderness boundary.

Oak Maritime Forest Recruitment

Studies of tree recruitment, such as the composition and density of the seedling and sapling layers, can provide insight into the future character of the forest (McWilliams et al. 2015). Shifts in the composition of seedlings/saplings may indicate an eventual change in the composition of the forest as a whole, which can impact forest dynamics and wildlife habitat (McWilliams et al. 2015). Based on personal observations, CUIS managers are somewhat concerned that oak species recruitment may be low in the park's maritime forest, which could threaten the long-term health of this forest type in the future (Fry, email communication, 14 July 2017).

During oak regrowth surveys in 1990, Lieske et al. (1990) visited 10 sites spread across CUIS from south to north (see Figure 19). The total number of sprouts/seedlings in quadrats surrounding 10 oak trees at each site is presented in Table 16 (data were incomplete for sites 4 and 5, so they are not included). The total number of sprouts/seedlings ranged from 176 at the Brickhill Bluff site to 305 at North Greyfield (Lieske et al. 1990). North Greyfield also contained the highest number of trees with taller regrowth (≥8 cm [3.1 in]) and the fewest trees with no evidence of regrowth. Sites towards the northern end of the island tended to have fewer trees with taller regrowth, and half of the trees showed no regrowth (Table 16) (Lieske et al. 1990).

Table 16. Total number of sprouts/seedlings and other regrowth information for quadrats around 10 selected oak trees at each site (Lieske et al. 1990). Data were incomplete for sites 4 and 5, so they are not included here.

Site	Selected Trees	Total # Sprouts/Seedlings*	# of Trees with Regrowth ≥ 8 cm tall	# of Trees with No Regrowth
1	9 laurel oak, 1 live oak	241	5	5
2	6 laurel oak, 4 live oak	305	8	2
3	6 laurel oak, 4 live oak	233	6	3
6	8 laurel oak, 2 live oak	297	2	3
7	9 laurel oak, 1 live oak	288	6	3
8	8 laurel oak, 2 live oak	176	4	5
9	6 laurel oak, 4 live oak	229	4	5
10	9 laurel oak, 1 live oak	256	4	5

Lieske et al. (1990) also measured the eight closest oak sprouts/seedlings in a 1-m (3.3-ft) arc from eight quadrats around each tree. Similar to the results for total sprouts/seedlings, the highest number of oak sprouts/seedlings was found at North Greyfield, followed by Kings Bottom (Table 17). The lowest sprout/seedling numbers were at Brickhill Bluff and east of Stafford Field. The tallest sprouts/seedlings, and therefore likely the oldest, were observed at the River Trail (66 cm [26 in]) and Kings Bottom (64 cm [25 in]) sites (Lieske et al. 1990). The number of seedlings ≥10 cm (4 in) was highest at the Yankee Paradise and Abraham Point sites towards the center and north end of the island, and lowest at sites towards the southern end of the island (between North Greyfield and Stafford Field) (Table 17).

Table 17. Number and height data for closest oak sprouts/seedlings (up to 8) within a 1-m (3.3 ft) arc of 10 selected oak trees at each site (Lieske et al. 1990).

Site	# of Oak Sprouts/Seedlings Measured (Out of 640 Possible)	Maximum Height (cm)	# Seedlings ≥ 10 cm
1	267	66	33
2	364	18	24
3	295	13	9
4	215	14	2
5	243	11	4
6	279	33	138
7	305	64	82
8	207	39	23
9	288	42	130
10	272	28	32

During SECN monitoring (Byrne et al. 2012, Heath and Byrne 2014), surveyors recorded the number of seedlings from tree/shrub species per plot. In 2009, a total of 345 live oaks were found in the seedling layers at 12 of the 30 plots sampled (Byrne et al. 2012). Only 145 seedlings of other oak species were documented. Of the 16 plots specifically within oak maritime forest that were sampled, four contained no oaks in the seedling layer (Table 18). An additional five plots had an oak seedling density of $<1/m^2$. Plots with few or no oaks in the seedling layer also tended to have low seedling densities (<1 seedling/m^2) for other species. The highest oak seedling density was just over $13/m^2$, and only three plots showed oak seedling densities $>3/m^2$ (Byrne et al. 2012).

Table 18. Density (individuals/m²) of oaks and other tree species in the seedling layer at vegetation monitoring sampling locations within oak maritime forest plots at CUIS, 2009 (Byrne et al. 2012). Values are rounded to the nearest whole number.

Location	Sand Live Oak (*Q. geminata*)	Laurel Oak (*Q. laurifolia*)	Myrtle Oak (*Q. myrtifolia*)	Live Oak (*Q. virginiana*)	Other Species
Plot 2	–	1.0	–	<1	–
Plot 4	1	–	<1	–	3
Plot 5	–	–	–	–	<1
Plot 6	–	–	–	4	<1
Plot 7	–	–	–	3	<1
Plot 8	–	1	–	12	2
Plot 9	–	–	–	–	<1
Plot 13	–	2	–	–	1
Plot 14	–	–	–	–	–
Plot 15	–	–	–	1	<1
Plot 18	–	5	–	–	1
Plot 19	–	<1	–	–	<1
Plot 21	–	<1	–	<1	1
Plot 28	–	–	–	–	1
Plot 29	–	–	–	<1	2
Plot 30	–	–	–	7	2

During 2012 monitoring, 847 live oak seedlings and 83 seedlings of other oak species were observed across 29 sampled plots (Heath and Byrne 2014). Of the 19 plots within oak maritime forest that were sampled, five contained no oak seedlings (Table 19). Three additional plots had an oak seedling density <1.0/m². As in 2009, many of the plots with few or no oak seedlings also had low seedling densities (≤1.0/m²) for other species. The highest oak seedling density was just below 35.0/m² (all live oak), with four plots showing oak seedling densities >3.0/m² (Heath and Byrne 2014).

Table 19. Density (individuals/m²) of oaks and other tree species in the seedling layer at vegetation monitoring sampling locations within oak maritime forest plots at CUIS, 2012 (Heath and Byrne 2014). Values are rounded to the nearest whole number.

Location	Chapman Oak (*Q. chapmanii*)	Sand Live Oak (*Q. geminata*)	Live Oak (*Q. virginiana*)	Unknown Oak sp.	Other Species
Plot 2	–	–	3	–	1
Plot 4	–	2	–	–	<1
Plot 5	–	–	–	–	14
Plot 6	–	–	–	–	1
Plot 7	–	–	1	2	1
Plot 8	–	–	4	–	1

Table 19 (continued). Density (individuals/m^2) of oaks and other tree species in the seedling layer at vegetation monitoring sampling locations within oak maritime forest plots at CUIS, 2012 (Heath and Byrne 2014). Values are rounded to the nearest whole number.

Location	Chapman Oak (*Q. chapmanii*)	Sand Live Oak (*Q. geminata*)	Live Oak (*Q. virginiana*)	Unknown Oak sp.	Other Species
Plot 9	–	–	1	–	1
Plot 13	–	–	2	–	<1
Plot 37	–	–	10	–	1
Plot 38	–	–	1	–	<1
Plot 41	–	–	1	–	<1
Plot 42	–	–	1	–	4
Plot 48	<1	–	–	–	1
Plot 50	–	–	–	–	<1
Plot 52	–	–	2	–	<1
Plot 54	–	–	–	–	<1
Plot 57	–	–	35	–	2
Plot 59	–	–	6	–	2
Plot 60	–	–	–	–	1

Longleaf Pine Acreage

Based on historic maps and documents, Frost et al. (2011) concluded that longleaf pine communities were among the most common types of upland forest on Cumberland Island prior to European settlement. Frost et al. (2011) estimated that two longleaf pine-dominated communities (longleaf pine savannah and woodland, and longleaf pine-slash pine woodland) covered 1,051 ha (2,597 ac) of the island (Table 20). These were primarily in the central and northern portions of the island (Figure 20).

Table 20. Extent of longleaf pine community types at CUIS, according to various studies over time.

Source	Longleaf Community Type	Area (ha/ac)
Frost et al. (2011) - presettlement	Longleaf pine savannah and woodland	941 (2,325)
	Longleaf pine-slash pine woodland	110 (272)
	Total	**1,051 (2,597)**
McNamany 2015	Longleaf Pine /(Sand Laurel Oak, Turkey Oak) /Wax-myrtle/Southern Wiregrass Woodland	24.8 (61)

The Hillestad et al. (1975) vegetation classification and mapping did not include any upland forest communities dominated by longleaf pine. Although longleaf pine was a substantial component of the oak-pine community described by Hillestad et al. (1975), at that time the oak-pine forest was generally dominated by live oak, laurel oak, and loblolly pine (*Pinus taeda*). Therefore, the acreage covered by longleaf pine communities at the time of park establishment is unknown. The more recent vegetation mapping effort (McManamay 2017) identified just 24.8 ha (61.3 ac) of longleaf pine-dominated community in the northwestern portion of CUIS (Table 20, Figure 21).

Longleaf Pine Recruitment

As with the oak maritime forest, CUIS park managers are concerned that longleaf pine recruitment on the island is low, which threatens the persistence of longleaf pine-dominated communities on the island. Little information is available regarding longleaf pine recruitment and regeneration at CUIS. The SECN vegetation monitoring program's seedling surveys did not record any longleaf pine in the seedling layer of vegetation plots sampled in 2009, and only three of the 29 plots sampled in 2012 contained longleaf seedlings (33 longleaf pine seedlings total) (Byrne et al. 2012, Heath and Byrne 2014). However, none of the plots with longleaf pine seedlings were classified as longleaf pine-dominated communities; two of the three were oak-dominated communities. Only two SECN monitoring plots sampled to date have fallen within longleaf pine-dominated vegetation communities, one in 2009 (Plot 17) and one in 2012 (Plot 39). Plot 17, in Maritime Slash Pine-Longleaf Pine Upland Flatwoods, had just one tree/shrub species in the seedling layer (yaupon [*Ilex vomitoria*], frequency of 0.58) (Byrne et al. 2012). Plot 39, in Longleaf Pine/(Sand Laurel Oak, Turkey Oak)/Wax-myrtle/Southern Wiregrass Woodland, also had just one tree/shrub species in the seedling layer (wax myrtle, frequency of 1.08) (Heath and Byrne 2014). The limited seedling presence in these plots may be due to high cover in mid- and upper vegetation layers. For example, shrub absolute cover in Plot 17 was just over 38% (Byrne et al. 2012) and average canopy cover in Plot 39 was nearly 84% (Heath and Byrne 2014).

Redbay Presence/Persistence

Redbay is a key native component of coastal maritime forests and provides habitat for a variety of wildlife and non-vascular plants (Heath and Byrne 2014). The fruits are eaten by numerous birds and the plant serves as a primary host for the larva of the Palamedes swallowtail (*Papilio palamedes*) (Figure 22) (Fraedrich et al. 2008). A decline in redbay along the southeastern Atlantic Coast was first noted in 2003 and was traced to laurel wilt disease (LWD), a lethal fungus (*Raffaelea lauricola*) spread by a non-native beetle (Shearman and Wang 2016). Redbay decline was observed at CUIS in the fall of 2006 and mortality has been high (Merten 2015).

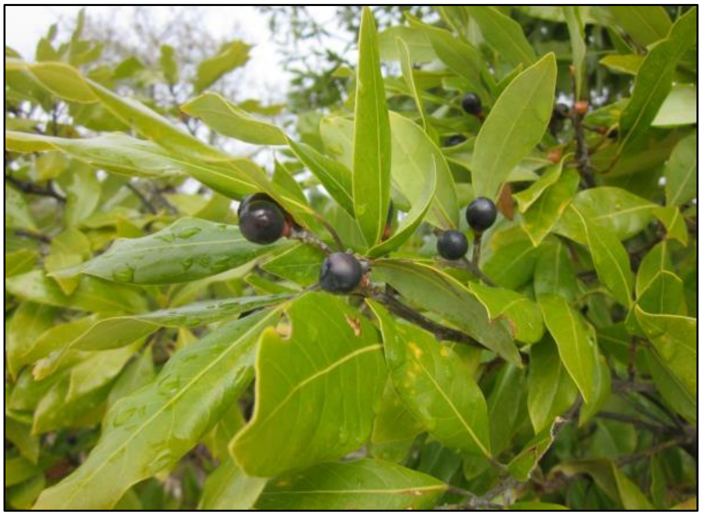


Figure 22. A seed-producing redbay tree in an interdune area on CUIS (Merten 2015).

Young longleaf pines at CUIS (SMUMN GSS photo).

Comprehensive, island-wide surveys for redbay have not occurred at CUIS. However, some information can be gleaned from SECN monitoring data. In 2009, redbay was documented in 11 of 16 oak maritime forest sampling plots (Table 21) (Byrne et al. 2012). The species was primarily found in the shrub layer (10 plots) and the ground layer (seven plots). Dead redbay was noted in the canopy layer of six plots, with live redbay (in addition to dead) in only one plot's canopy layer. Six plots had redbay in both the shrub and ground layers, and one plot in the north-central portion of CUIS had live redbay in all three layers (ground, shrub, and canopy) (Byrne et al. 2012).

Table 21. Presence of redbay, based on SECN monitoring, 2009 (Byrne et al. 2012).

Location	In Canopy	In Shrub Layer	In Ground Layer
Plot 2	–	x	–
Plot 4	–	x	x
Plot 5	–	–	–
Plot 6	–	–	–
Plot 7	–	–	–
Plot 8	x (dead)	x	–
Plot 9	x (dead)	x	–
Plot 13	–	–	–
Plot 14	–	–	–
Plot 15	–	x	–
Plot 18	–	x	x
Plot 19	–	–	x
Plot 21	x (dead)	x	x
Plot 28	x (dead)	x	x
Plot 29	x (dead)	x	x
Plot 30	x (live and dead)	x	x

In 2012, SECN monitoring documented redbay in 17 of 19 oak maritime forest plots (Table 22) (Heath and Byrne 2014). Again, the presence was primarily in the shrub layer (13 plots) and ground layer (nine plots). Dead redbay was noted in the canopy layer of five plots, with live redbay (along with dead) in only one plot's canopy layer. Six plots had redbay in both the shrub and ground layers and one plot had redbay in all three layers and just one plot towards the south end of CUIS had live redbay in all three layers (Heath and Byrne 2014). Of the plots sampled in both 2009 and 2012, four plots with no redbay present in 2009 (5, 6, 7, and 13) had redbay in the shrub and/or ground layer in 2012 (Table 21, Table 22). In 2012, an additional seven plots in vegetation communities other than the oak maritime forest (e.g., loblolly pine forest, oak scrubland) also contained redbay (Heath and Byrne 2014).

Table 22. Presence of redbay, based on SECN monitoring, 2012 (Heath and Byrne 2014).

Location	In Canopy	In Shrub Layer	In Ground Layer
Plot 2	–	–	x
Plot 4	–	x	x
Plot 5	–	x	–
Plot 6	–	x	–
Plot 7	–	x	x
Plot 8	x (dead)	x	x
Plot 9	x (dead)	x	x
Plot 13	–	x	–
Plot 37	–	–	x
Plot 38	–	x	x
Plot 41	–	–	–
Plot 42	–	x	x
Plot 48	–	x	–
Plot 50	–	–	–
Plot 52	x (dead)	x	–
Plot 54*	–	–	–
Plot 57*	–	–	–
Plot 59	x (live and dead)	x	x
Plot 60	x (dead)	x	–

* redbay was listed as present in these plots but the vegetation layer in which it was found was not specified.

Threats and Stressor Factors

Threats to upland forest communities identified by the CUIS NRCA project team include wildlife browsing of saplings, fire suppression, high understory density, pests and pathogens (especially LWD), feral hog rooting, and climate change. Wildlife browsing, particularly by white-tailed deer, is known to impact regeneration in eastern U.S. forests (Lorimer 1993, Russell et al. 2001). Bratton and Kramer (1989) found that deer browsing was a major source of suppression of regrowth in the park's live oak forests. Live oak sprouts within deer exclosures were significantly taller than sprouts in horse exclosures and control areas (Bratton and Kramer 1989). The results for the two areas within CUIS oak forests surveyed are shown in Figure 23 and Figure 24 below. Bratton and Kramer (1989) suggest that the island's deer population increased substantially during the 1900s, likely due to lack of predators and low human hunting pressure.

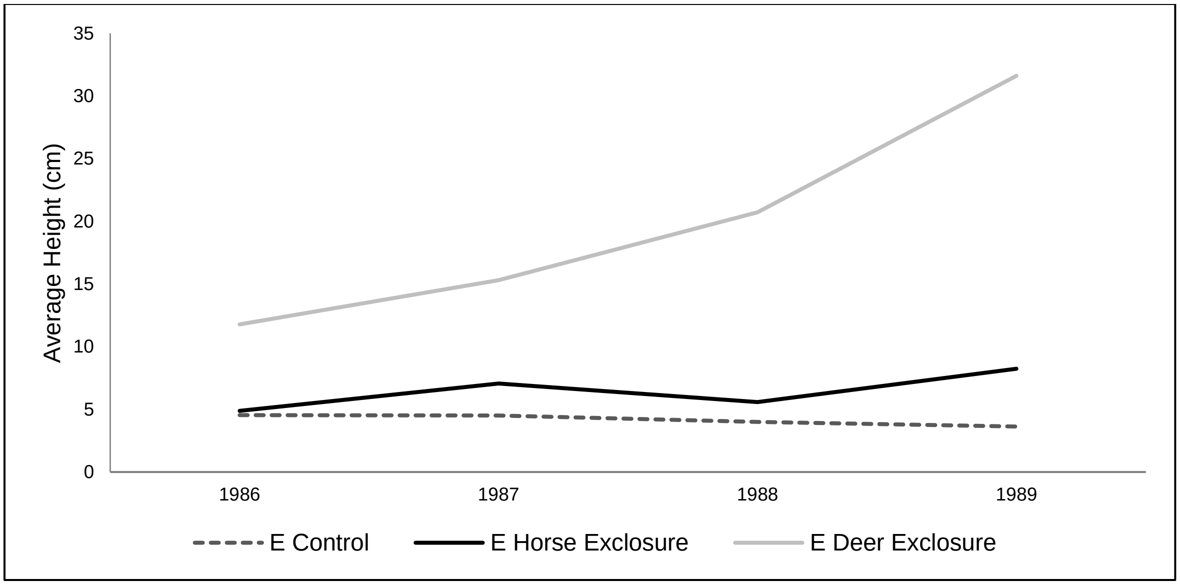


Figure 23. The average height of live oak sprouts in east exclosure and control plots (Bratton and Kramer 1989).

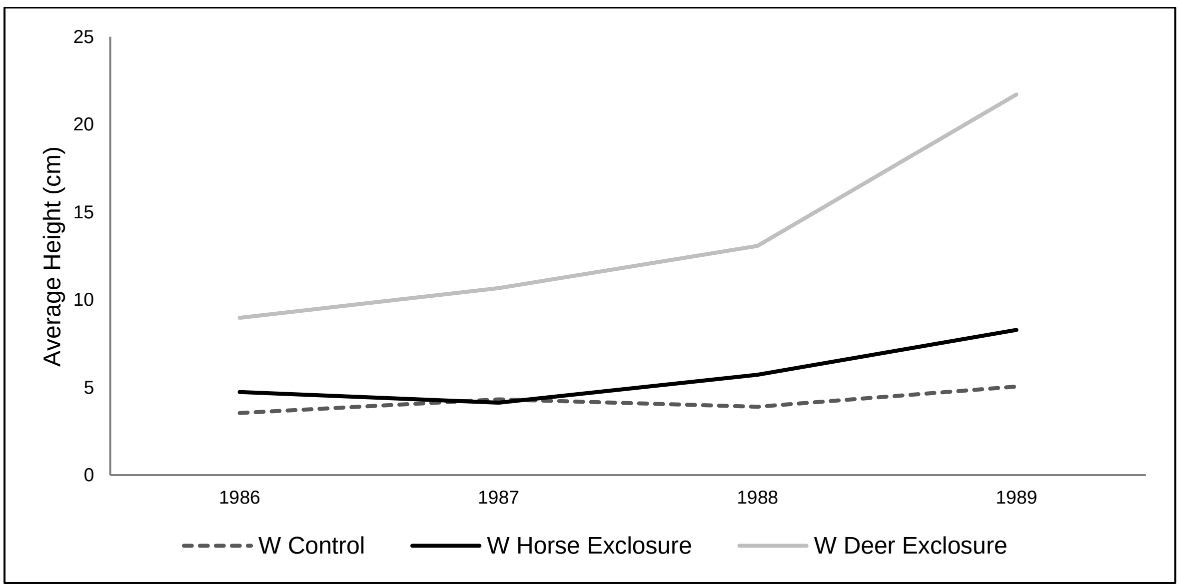


Figure 24. The average height of live oak sprouts in west exclosure and control plots (Bratton and Kramer 1989).

In addition to wildlife browsing, feral hog rooting may also impact forest regeneration (Hillestad et al. 1975, NPS 2014a). Rooting behavior can disturb large patches of vegetation and soil, which can kill tree seedlings and small saplings, as well as creating opportunities for invasion by non-native plants that would compete with native tree seedlings (Siemann et al. 2009, Kammermeyer et al. 2011). Hogs also feed on tree seeds (e.g., acorns), which can further reduce regeneration (Siemann et al. 2009, Kammermeyer et al. 2011).

Historically, fires were frequent (every 1-12 years) in the northern and central portions of Cumberland Island, particularly in longleaf pine communities where burning kept understory density low and promoted longleaf pine regeneration (Brockway et al. 2000, Frost et al. 2011). During a field visit to CUIS, Frost et al. (2011) found a small population of wiregrass (aka, Beyrich threeawn [*Aristida beyrichiana*]), a species that is indicative of high original fire frequency, southeast of Terrapin Point (Figure 25). However, over the past century, fires have become less frequent across the southeastern U.S., largely due to human suppression efforts (Frost 1993). The lack of fire has allowed an increase in the density of woody mid- and understory species, such as saw palmetto, that would normally be reduced by frequent growing season fires (Frost et al. 2011). This increased density may be inhibiting regeneration of canopy tree species (e.g., pines and oaks), as well as reducing the diversity and cover of herbaceous species in the ground layer (Brockway et al. 2000, McManamay 2017). Fire suppression also allows heavy fuel loads (e.g., woody debris, leaf litter) to accumulate in forests, which can result in large high-severity fires if ignition does occur (NPS Frost et al. 2011, 2014a). During field surveys for the most recent vegetation mapping and classification effort at CUIS, McManamay (2017), p. 103, 110 noted evidence of "extreme fire suppression" in the two upland pine-dominated communities. Until a 2015 revision, the CUIS Fire Management Plan (FMP) called for suppression of all fires on the island and did not allow for prescribed burning (NPS 2015a). Under the new FMP, naturally ignited fires are allowed to burn in much of the park, as long as they do not threaten human safety or property and are deemed beneficial to the park's natural resources. Prescribed burning will also be used to improve habitat, reduce dangerous fuel loads, and to maintain cultural landscapes (NPS 2015a).

Figure 25. The presence of wiregrass (in the lower right corner) at CUIS indicates a high original fire frequency in certain forest/woodland communities (1-10 years) (Frost et al. 2011).

As mentioned previously, redbay at CUIS has been severely impacted by LWD, a fungus transmitted by a non-native ambrosia beetle (*Xyleborus glabratus*) (Merten 2015, Shearman and Wang 2016). The initial decline, around 2006, was primarily on the south half of the island (Merten 2015). The first sign of LWD is often wilting of branch tips or entire branches; this wilting then spreads throughout the entire crown and trees typically die soon after (Figure 26) (Fraedrich et al. 2008). Larger trees are more likely to be affected than small trees, with Shearman and Wang (2016) finding a 5% increase in the likelihood of mortality with each 1-cm (0.4-in) increase in tree diameter. At CUIS, the largest redbay trees sampled (25.4-cm [10-in] diameter class) were the first to experience 100% crown decline (Figure 27) (Merten 2015). By October 2008, trees in the 12.7-cm (5-in) diameter size class or smaller were the only redbay not experiencing 100% crown decline. Percent decline was lowest in the 2.5-cm (1-in) size class. By December 2009, average crown decline across all size classes at CUIS was 90% (Merten 2015).

Figure 26. An aerial photo of CUIS upland forest in 2008 showing extensive redbay mortality in the canopy (reddish-brown foliage) (GA DNR Photo).

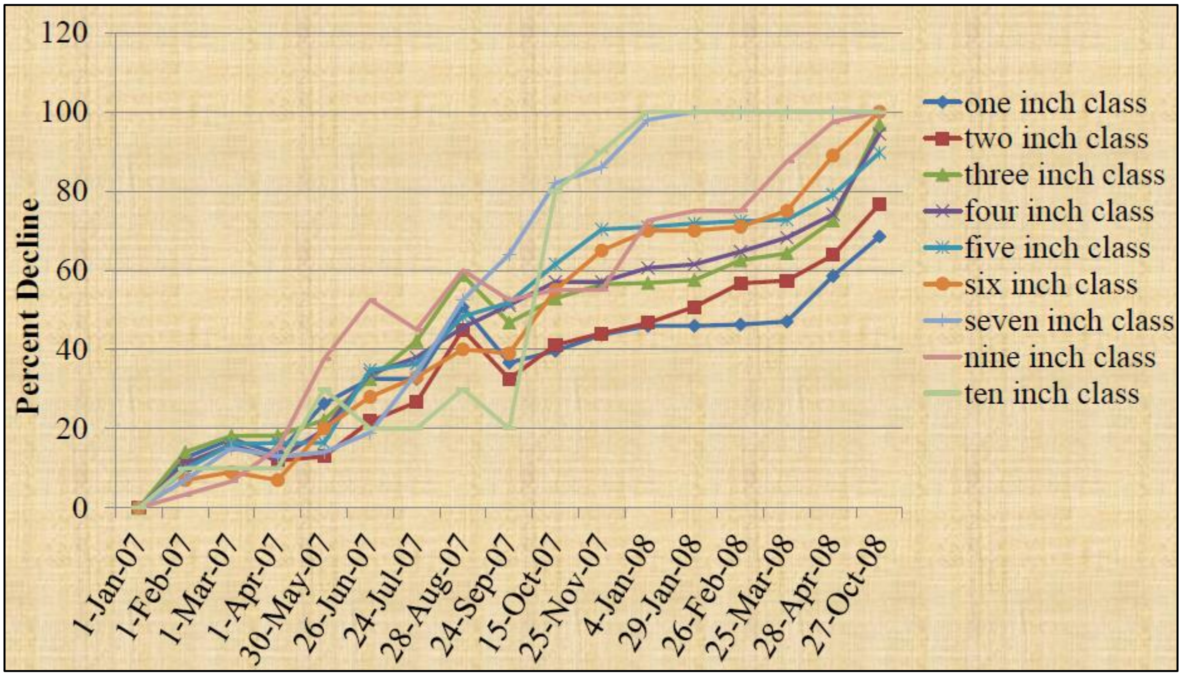


Figure 27. Percent of crown decline in redbay on CUIS by diameter size class (Merten 2015).

At CUIS and across the southeastern coast, some re-sprouting has occurred from stumps of redbay trees killed by LWD (Fraedrich et al. 2008, Merten 2015). Some of the sprouts grow into the forest mid-story but others succumb to herbivory, intense canopy shading (particularly by saw palmetto), or renewed LWD infection (Figure 28) (Merten 2015). In LWD-impacted plots across Georgia and South Carolina, Shearman and Wang (2016) found that redbay regained much of the basal area lost to LWD mortality approximately 10 years after the initial infection. However, the stand structure had changed, with the majority of redbay stems in the 1-5 cm (0.4-2.0 in) diameter size class. This suggests that redbay is not at immediate risk of extirpation, but it unclear whether the species will ever fully recover from the disease (Shearman and Wang 2016).

Figure 28. The photo on the left shows redbay re-sprouting from the trunk of a tree killed during the initial LWD infection at CUIS. On the right, this re-sprout has wilted, indicating the continued presence of LWD on the island (Merten 2015).

Climate is a key driving factor in the ecological and physical processes influencing vegetation in parks throughout the SECN (Davey et al. 2007). Climate also affects the spread of invasive plant species and pests, which also threaten CUIS's upland forests (Davey et al. 2007). As a result of global climate change, temperatures are projected to increase across the southeast over the next century (Carter et al. 2014). Warming temperatures will likely allow invasive plants and forest pests to expand their ranges and potentially their impact, as well as altering the habitat suitability of certain areas for some tree species (Fisichelli et al. 2014). Temperature changes may also alter weather patterns, resulting in more storms with forest-damaging high winds (Carter et al. 2014). As the impacts of climate change and related stressors compound over time, forests will experience more widespread changes in tree species composition, with cascading effects on other plants and wildlife (Fisichelli et al. 2014). In an effort to estimate the magnitude of potential change that forests on eastern national park lands may experience, (Fisichelli et al. 2014) assessed the percentage of tree species expected to show large decreases or large increases in habitat suitability under climate change scenarios. Across 121 national park properties in the eastern U.S., estimated potential forest change ranged from 22-77%. The estimated forest change for CUIS (i.e., percent of tree species expected to experience large increases or decreases in habitat suitability) was 36% (Fisichelli et al. 2014). Habitat suitability projections for several of CUIS's key upland tree species are shown in Table 23.

Table 23. Potential change in habitat suitability by 2100 for select CUIS upland tree species based on two future climate scenarios (the "least change" scenario represents strong cuts in greenhouse gas emissions and modest climatic changes, the "major change" scenario represents continued increasing emissions and rapid warming). Reproduced from Fisichelli (2015).

Scientific Name	Common Name	Least Change Scenario	Major Change Scenario
Pinus taeda	loblolly pine	small decrease	small decrease
Prunus serotina	black cherry	small decrease	small decrease
Quercus incana	bluejack oak	small increase	large increase
Quercus laevis	turkey oak	large increase	large increase
Quercus stellata	post oak	large increase	large increase
Quercus virginiana	live oak	small increase	large increase
Diospyros virginiana	common persimmon	small decrease	large increase
Magnolia virginiana	sweetbay	no change	small increase
Pinus palustris	longleaf pine	no change	small increase
Quercus falcata	southern red oak	no change	small increase
Quercus laurifolia	laurel oak	no change	small increase
Ulmus alata	winged elm	new potential habitat	new potential habitat

Forest damage at CUIS from Hurricane Irma, September 2017 (NPS photo).

Data Needs/Gaps

Information on recruitment in CUIS's oak maritime forests and longleaf pine communities is very limited. Further study of tree regeneration in these upland forests is needed to identify areas or species experiencing low recruitment and to determine contributing factors. Bratton and Kramer

(1989) recommended permanent sprout/seedling monitoring transects in the park's oak maritime forest. The establishment of eight NPS fire monitoring program plots in CUIS pine-dominated communities during 2014 (Burton and Fields 2016) may eventually provide some information on longleaf pine and the health of the park's fire-dependent upland forests.

Monitoring of the park's redbay will help to determine how the species is recovering from LWD and could help detect whether any other species in the Lauraceae family are being impacted by the disease (NPS 2014a). It is unclear what ecological impacts the reduction or loss of redbay will have on other components of CUIS's maritime forests (Heath and Byrne 2014, NPS 2014a).

Research into the ecological role of soil microbiota and mycorrhizae in soil formation and nutrient cycling could help managers better understand the upland forest ecosystem (Bellis 1995). The ecological role of lichens in maritime forests also has not been studied. Bellis (1995) hypothesized that lichens may offer tree branches and buds some protection from salt spray or could be involved in nutrient cycling.

Overall Condition

Upland Forest Acreage

The NRCA project team assigned this measure a *Significance Level* of 3. Based on a comparison of pre-settlement acreage estimates from Frost et al. (2011) and recent mapping efforts (McManamay 2017), upland forests have expanded on Cumberland Island over the past several centuries. As a result, this measure is currently of no concern (*Condition Level* = 0).

Upland Forest Plant Species Diversity

This measure was also assigned a *Significance Level* of 3. Across various surveys over time, nearly 270 total plant species have been observed within the park's upland forest communities. Only a fraction of these (~3%) are non-native, and just two are considered invasive. At this time, plant species diversity in CUIS upland forests is of low concern (*Condition Level* = 1).

Oak Maritime Forest Acreage

A *Significance Level* of 3 was assigned for this measure. As with overall upland forest acreage, oak maritime forest acreage has increased since pre-settlement times. The recent mapping effort (McManamay 2017) identified approximately three times as much oak maritime forest at CUIS as was estimated for the pre-settlement period by Frost et al. (2011). Therefore, this measure is assigned a *Condition Level* of 0, indicating no concern.

Oak Maritime Forest Recruitment

This recruitment measure was also assigned a *Significance Level* of 3. A focused survey of oak regrowth was conducted at CUIS during the early 1990s (Lieske et al. 1990), but more recent information is limited to observations from SECN vegetation monitoring. Of the 19 plots within oak maritime forest that were sampled in 2012, five contained no oaks in the seedling layer, and an additional three plots had an oak seedling frequency <1.0 (Heath and Byrne 2014). Low oak recruitment may be related to increased understory density and/or deer browsing. As a result, this measure is currently of moderate concern (*Condition Level* = 2).

Longleaf Pine Acreage

A *Significance Level* of 3 was assigned for this measure. According to Frost et al. (2011), longleaf pine communities were historically one of the island's more common upland forest types. Currently, very little of CUIS's vegetation is mapped as longleaf-pine dominated upland forest/woodland (Table 20) (McManamay 2017). This is likely due to fire suppression and a related increase in understory density (Frost et al. 2011, NPS 2014a). Park management is currently working to return fire to the landscape (NPS 2015a), which will likely benefit longleaf pine communities. At this time, however, this measure is assigned a *Condition Level* of 3, indicating significant concern.

Longleaf Pine Recruitment

The longleaf pine recruitment measure was also assigned a *Significance Level* of 3. Information regarding longleaf pine recruitment at CUIS is very limited. SECN vegetation monitoring has documented longleaf pine seedlings in the park, but none were observed in communities currently classified as upland longleaf pine forest/woodland (Heath and Byrne 2014). While park management is concerned that longleaf recruitment may be low due to decades of historical fire suppression, further information is needed before a *Condition Level* can be assigned.

Redbay presence/persistence

This final measure was assigned a *Significance Level* of 2. Heavy mortality of redbay at CUIS was first investigated in 2006 and determined to be a result of LWD (Merten 2015). Many larger individuals in the park's upland forests have died. However, stump re-sprouting has been observed, and SECN vegetation monitoring in 2012 documented redbay in 17 of 19 oak maritime plots (Heath and Byrne 2014, Merten 2015). At this time, redbay appears to be persisting in CUIS forests in a smaller, shrubbier form, but it is unclear how the species will respond if LWD attacks continue. Therefore, this measure is assigned a *Condition Level* of 2, indicating moderate concern.

Weighted Condition Score

The *Weighted Condition Score* for CUIS's upland forest community is 0.43, indicating moderate concern. While the overall acreage of upland forests and of oak maritime forests specifically are in good condition, oak recruitment and the overall health of upland longleaf pine communities are currently of concern (Table 24). The condition of the upland forest community as a whole appears to be stable at this time.

Table 24. Weighted Condition Score of Upland Forest Community in CUIS.

Upland Forest Community			
Measures	**Significance Level**	**Condition Level**	**WCS = 0.43**
Upland Forest Acreage	3	0	
Upland Forest Diversity	3	1	
Oak Maritime Forest Acreage	3	0	
Oak Mar. Forest Recruitment	3	2	
Longleaf Pine Acreage	3	3	
Longleaf Pine Recruitment	3	n/a	
Red Bay Presence/Persistence	2	2	

4.1.6. Sources of Expertise

Mike Byrne, SECN Terrestrial Ecologist

John Fry, CUIS Chief of Resource Management

Doug Hoffman, CUIS Wildlife Biologist

4.2. Freshwater Wetlands

4.2.1. Description

Freshwater wetlands are particularly valuable on a barrier island surrounded by salt marsh and ocean, as they provide critical habitat and resources for plants and wildlife, as well as performing vital ecosystem functions (Zedler and Kercher 2005, Dlugolecki 2012). CUIS has the largest and most diverse freshwater wetlands system of all Georgia's barrier islands (Hillestad et al. 1975, Frick et al. 2002). The island's wetlands vary in physical setting, moisture/water levels, and vegetation communities; there are seasonal wetlands dominated by herbaceous plants, forested swamps, and permanent man-made ponds (Hillestad et al. 1975, Frick et al. 2002). The extent of these wetlands and associated open water also varies, depending on climatic conditions (i.e., temperature and precipitation), groundwater levels, and disturbances (e.g., fire, hurricanes) (Frick et al. 2002).

Notable wetlands at CUIS include the Whitney Lake complex (Figure 29), the Sweetwater Lake complex, the Lake Retta complex, Swamp Fields, and Plum Orchard Pond. Whitney Lake is the deepest and most permanent open water wetland on the island, supporting impressive displays of flowering aquatic plants during the summer (Hillestad et al. 1975). The Sweetwater complex on the eastern side of CUIS is over 120 ha (300 ac) in size, at least half of it wooded, which provides exceptional breeding habitat for amphibians. Plum Orchard Pond is a small, highly eutrophic (nutrient-rich) man-made pond on the west side of the island, which offers roosting habitat for a variety of wading birds (Hillestad et al. 1975, Dlugolecki 2012). The major wetland vegetation types found at CUIS and the common plant species in each type are presented in Table 25.

Figure 29. A wetland in the Whitney Lake complex, near Roller Coaster Trail, in March 2017 (SMUMN GSS photo).

Table 25. Wetland vegetation community types (forest types are grouped) occurring on CUIS and their common plant species, as described by McManamay (2017).

Wetland Community Type	Common Plant Species
Swamp/Streamhead Forest	swamp bay (*Persea palustris*), sweetbay (*Magnolia virginiana*), swamp tupelo (*Nyssa biflora*), red maple (*Acer rubrum*), loblolly pine (*Pinus taeda*), wax myrtle (*Morella cerifera*), fetterbush lyonia (*Lyonia lucida*), resurrection fern (*Pleopeltis polypodioides*), greenbriers (*Smilax* sp.), muscadine grape (*Vitis rotundifolia*), slender woodoats (*Chasmanthium laxum*), lizard's tail (*Saururus cernuus*)
Wet Pine Flatwoods	pond pine (*Pinus serotina*), swamp bay, slash pine (*Pinus elliottii*), rusty staggerbush (*Lyonia ferruginea*), inkberry (*Ilex glabra*), wax myrtle, saw palmetto (*Serenoa repens*)
Atlantic Coast Interdune Swale	wax myrtle, cabbage palmetto (*Sabal palmetto*), greenbriers, muscadine grape, bluestem grasses (*Andropogon* sp.), manyflower marshpennywort (*Hydrocotyle umbellata*)
Grapevine - Peppervine - Trumpetvine Thicket	muscadine grape, greenbriers, crossvine (*Bignonia capreolata*), wax myrtle, saw palmetto, sawtooth blackberry (*Rubus argutus*)
Swamp-loosestrife Pond	swamp loosestrife (*Decodon verticillatus*), coastal plain willow (*Salix caroliniana*), peppervine (*Ampelopsis arborea*), herb-of-grace (*Bacopa monnieri*), bur marigold (*Bidens laevis*), lizard's tail, shortbristle horned beaksedge (*Rhynchospora corniculata*)
South Atlantic Coastal Pond	sand cordgrass (*Spartina bakeri*), panic grasses (*Panicum* sp.), swamp loosestrife, Virginia chainfern (*Woodwardia virginica*)
Sawgrass Head	Jamaica swamp sawgrass (*Cladium jamaicense*), Virginia chainfern
Southern Cattail Marsh	broadleaf cattail (*Typha latifolia*), sand cordgrass, swamp loosestrife, American spongeplant (*Limnobium spongia*)

4.2.2. Measures

- Total acreage
- Acreage by wetland type
- Plant species diversity by wetland type
- Water quality
- Soil quality

4.2.3. Reference Condition/Values

As with the upland forest community component, the ideal reference condition for freshwater wetlands would be the condition of the wetlands prior to European settlement. However, given the history of human use and alteration, this is not practical. For this assessment, best professional judgement will be used to evaluate condition. Information presented in this report on current condition can be used as a baseline for assessing condition in the future.

4.2.4. Data and Methods

Several of the sources utilized for the upland forest community component were also used for this component. These include Hillestad et al. (1975), Zomlefer et al. (2008), Zomlefer and Kruse (2011), Frost et al. (2011), SECN monitoring reports (Byrne et al. 2012, Heath and Byrne 2014), and McManamay (2017). Sources for wetland water quality information (Kozel 1991, Frick et al. 2002) will be discussed in Chapter 4.8 of this report

Data on the location and extent of wetlands on Cumberland Island was also obtained from the National Wetland Inventory (NWI), a database maintained by the U.S. Fish and Wildlife Service (USFWS). The data for CUIS are based on 2006 aerial imagery and can be downloaded through USFWS "Wetlands Mapper" website at http://www.fws.gov/wetlands/. Since this component focuses specifically on freshwater wetlands, only non-tidal palustrine wetlands identified by the NWI are included in any analysis; estuarine (i.e., tidal) and marine wetlands are excluded.

4.2.5. Current Condition and Trend

Total Acreage

The estimates/measurements of total wetland acreage within CUIS boundaries have varied over time (Table 26). Based on historic maps and other sources, Frost et al. (2011) estimated that the pre-settlement (~1600) extent of freshwater wetlands at CUIS was approximately 960 ha (2,372 ac). Hillestad et al. (1975) mapped 707 ha (1,747 ac) of freshwater wetland on the island around the time of park establishment. More recently, McManamy (2015) classified 776 ha (1,918 ac) of CUIS vegetation as freshwater wetland. The NWI, based on 2006 aerial imagery, mapped 1,112 ha (2,748 ac) as freshwater wetland (USFWS 2012). The variance between the NWI total and other findings may be due to differences in methodology. The NWI is a broad, national program that relies primarily on aerial imagery with limited field verification. Other efforts were focused specifically on CUIS, and Hillestad et al. (1975) and McManamay (2017) included more extensive field verification of mapping. As a result of this on-the-ground work, Hillestad et al. (1975) and McManamay (2017) are likely to be the most accurate assessments of wetland extent.

Table 26. Total acreage of freshwater wetlands at CUIS, according to various sources over time.

Source	Wetland Acreage (ha/ac)	Percent of Mapped Area
Frost et al. (2011) – pre-settlement	959.5 (2,371)	8.8
Hillestad et al. (1975)	707.2 (1,748)	6.8
NWI (USFWS 2012)	1,112.1 (2,748)	–
McManamay (2017)	776.1 (1,918)	8.1

Acreage by Wetland Type

Frost et al. (2011) classified the pre-settlement freshwater wetland vegetation of Cumberland Island into six categories (Table 27). Temporarily flooded freshwater, oligohaline and brackish ponds comprised the greatest area, at 417 ha (1,030 ac). These wetlands support various plant communities, including floating aquatic vegetation (e.g., water lilies; Figure 30) and emergent herbaceous vegetation (e.g., grasses, sedges, rushes) (Frost et al. 2011). The primarily wooded freshwater swamp wetland complex was also common, covering an estimated 251 ha (621 ac). One community type that occurred on Cumberland Island historically but is no longer present is pocosin (Frost et al. 2011). A pocosin is "an evergreen shrub bog on organic soils with a canopy of pond pine (often crooked and sparse) over a dense semi-evergreen shrub layer," (Frost et al. 2011, p. 51). Due to fire suppression, these areas are now overgrown with a dense woody understory, including redbay and black gum (*Nyssa sylvatica*). A few old pond pines (*Pinus serotina*) still mark the locations of former pocosins on the island (Figure 30). The pre-settlement distribution of freshwater wetlands, as mapped by Frost et al. (2011) is shown in Figure 31.

Table 27. Extent of freshwater wetland communities within the pre-settlement (around 1600) vegetation of CUIS (Frost et al. 2011).

Freshwater Wetland Community Type	Area (ha/ac)	Percent of Total Area
Pocosin	66.0 (163)	0.6
Freshwater Swamp Wetland Complex	251.3 (621)	2.3
Quercus virginiana/Lyonia ferruginea Savanna Mosaic	23.1 (57)	0.2
Deep Savannah-Great Swamp Field	60.3 (149)	0.6
Freshwater, Oligohaline and Brackish Ponds (Temporarily Flooded)	417.2 (1,031)	3.8
Freshwater, Oligohaline and Brackish Ponds (Seasonally Flooded)	141.6 (350)	1.3
Total	**959.5 (2,371)**	**8.8**

Figure 30. Floating aquatic vegetation in a freshwater pond within the Willow Pond Complex (left) and old pond pines in the former Table of Pines Pocosin, south of Table Point (right) (Frost et al. 2011).

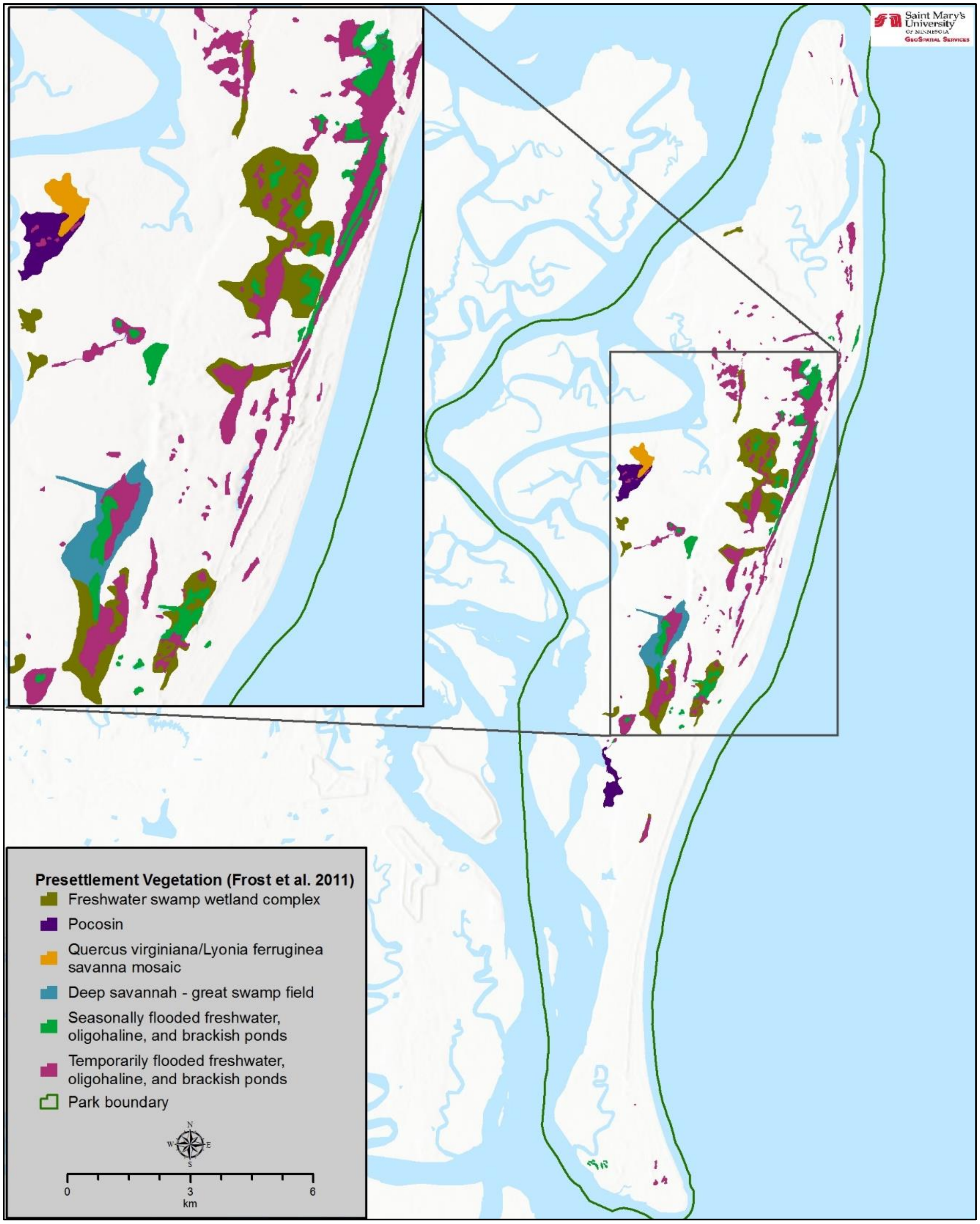


Figure 31. Estimated extent of freshwater wetland vegetation at CUIS prior to European settlement, as mapped by Frost et al. (2011).

Hillestad et al. (1975) grouped freshwater wetland communities into four types, based largely on vegetation. The most extensive wetland community at the time of park establishment was lowland mixed hardwood forest with 358 ha (885 ac) (Table 28). These forests included evergreen and deciduous trees, such as swamp bay, loblolly bay (*Gordonia lasianthus*), red maple (*Acer rubrum*), and oaks (*Quercus* sp.) (Hillestad et al. 1975). Grass-sedge communities were also common, covering 233 ha (575 ac). The pond-slough category, which included all open freshwater areas and some floating vegetation, covered the smallest area at just 17 ha (43 ac).

Table 28. Extent of freshwater wetland community vegetation types at CUIS in 1974 (Hillestad et al. 1975).

Freshwater Wetland Community Type	Cumberland Island Area (ha/ac)	Little Cumberland Isl. Area (ha/ac)	Total Area (ha/ac)	Percent of Total Area
Pond-Slough	17.3 (43)	0	17.3 (43)	0.2
Grass-Sedge	232.7 (575)	5.9	238.6	2.3
Shrub Marsh	93.5 (231)	0	93.5 (231)	0.9
Lowland Mixed Hardwood Forest	357.8 (884)	0	357.8 (884)	3.4
Total	**701.3 (1,733)**	**5.9 (15)**	**707.2 (1,748)**	**6.8**

McManamay (2017) divided freshwater wetlands into a larger number of categories than previous mapping efforts, identifying 13 wetland vegetation types (Table 29). The two most extensive wetland communities were wooded: wet longleaf pine-pond pine flatwoods at 224 ha (554 ac) and outer coastal plain sweetbay swamp forest at 121 ha (299 ac). The sparse canopy of the pine flatwoods, primarily found in the northern portion of CUIS, is dominated by pond pine (McManamay 2017). The most common herbaceous wetland type was south Atlantic coastal pond, with 111 ha (274 ac). These areas are comprised primarily of sand cordgrass, interspersed with other grasses and forbs. It is the dominant vegetation type in the Whitney Lake wetland complex (McManamay 2017). Two wetland vegetation communities covered less than 5 ha (12 ac): Water-hyacinth Aquatic Vegetation and Outer Coastal Plain Maidencane Pond. The extent of additional freshwater wetland community types are shown below in Table 29.

Table 29. Extent of freshwater wetland community vegetation types at CUIS as mapped by McManamay (2017).

Freshwater Wetland Community Type	Area (ha/ac)	Percent of Total Vegetated Area
Wet Longleaf Pine - Pond Pine Flatwoods	224.1 (554)	2.3
Outer Coastal Plain Sweetbay Swamp Forest	121.1 (299)	1.3
South Atlantic Coastal Pond	111.1 (275)	1.2
Southern Atlantic Coastal Plain Carolina Willow Dune Swale	83.4 (206)	0.9
Red Maple - Tupelo Maritime Swamp Forest	80.2 (198)	0.8
Swamp-loosestrife Pond	39.3 (97)	0.4
Rush Marsh / Sawgrass Head	37.8 (93)	0.4
Atlantic / East Gulf Coastal Plain Sweetbay - Tupelo Streamhead Forest	33.6 (83)	0.3

Table 29 (continued). Extent of freshwater wetland community vegetation types at CUIS as mapped by McManamay (2017).

Freshwater Wetland Community Type	Area (ha/ac)	Percent of Total Vegetated Area
Dotted Smartweed - Smooth Beggarticks Herbaceous Vegetation	16.3 (40)	0.2
Grapevine - Peppervine - Trumpetvine Thicket	13.0 (32)	0.1
Southern Cattail Marsh	12.5 (31)	0.1
Water-hyacinth Aquatic Vegetation	3.4 (8)	>0.1
Outer Coastal Plain Maidencane Pond	0.3 (1)	>0.1
Total	**776.1 (1,918)**	**8.1**

The NWI classifies wetlands by vegetation type (e.g., emergent, scrub-shrub, forested) and by water regime (e.g., permanently flooded, seasonally flooded, saturated). As with Hillestad et al. (1975) and McManamay (2017), the NWI mapping (USFWS 2012) shows that the most extensive freshwater wetlands at CUIS are forested (PFO) (Table 30). Forested wetlands totaled 822 ha (2,032 ac) or nearly 74% of the freshwater wetland area mapped. Emergent vegetation (herbaceous) covered 193 ha (478 ac) and scrub-shrub covered 67 ha (166 ac) (USFWS 2012). The distribution of these wetland types across the island is shown in Figure 32.

Table 30. Extent of freshwater wetlands at CUIS by vegetation type, as mapped by the NWI (USFWS 2012). P = palustrine, EM = emergent (herbaceous) vegetation, SS = scrub-shrub, FO = forest, AB = aquatic bed (plants that grow on or below the water's surface), UB = unconsolidated bottom (open water).

Wetland Community Type	# of Wetlands	Total Area (ha/ac)	% of Wetland Area
PEM	102	193.3 (478)	17.4
PEM/SS	3	8.2 (20)	0.7
PEM/FO	2	5.4 (13)	0.5
PSS	12	67.1 (166)	6.0
PFO	98	822.2 (2,032)	73.9
PAB	4	10.7 (26)	1.0
PUB	9	5.1 (13)	0.5
Total	**230**	**1,112.1 (2,748)**	**–**

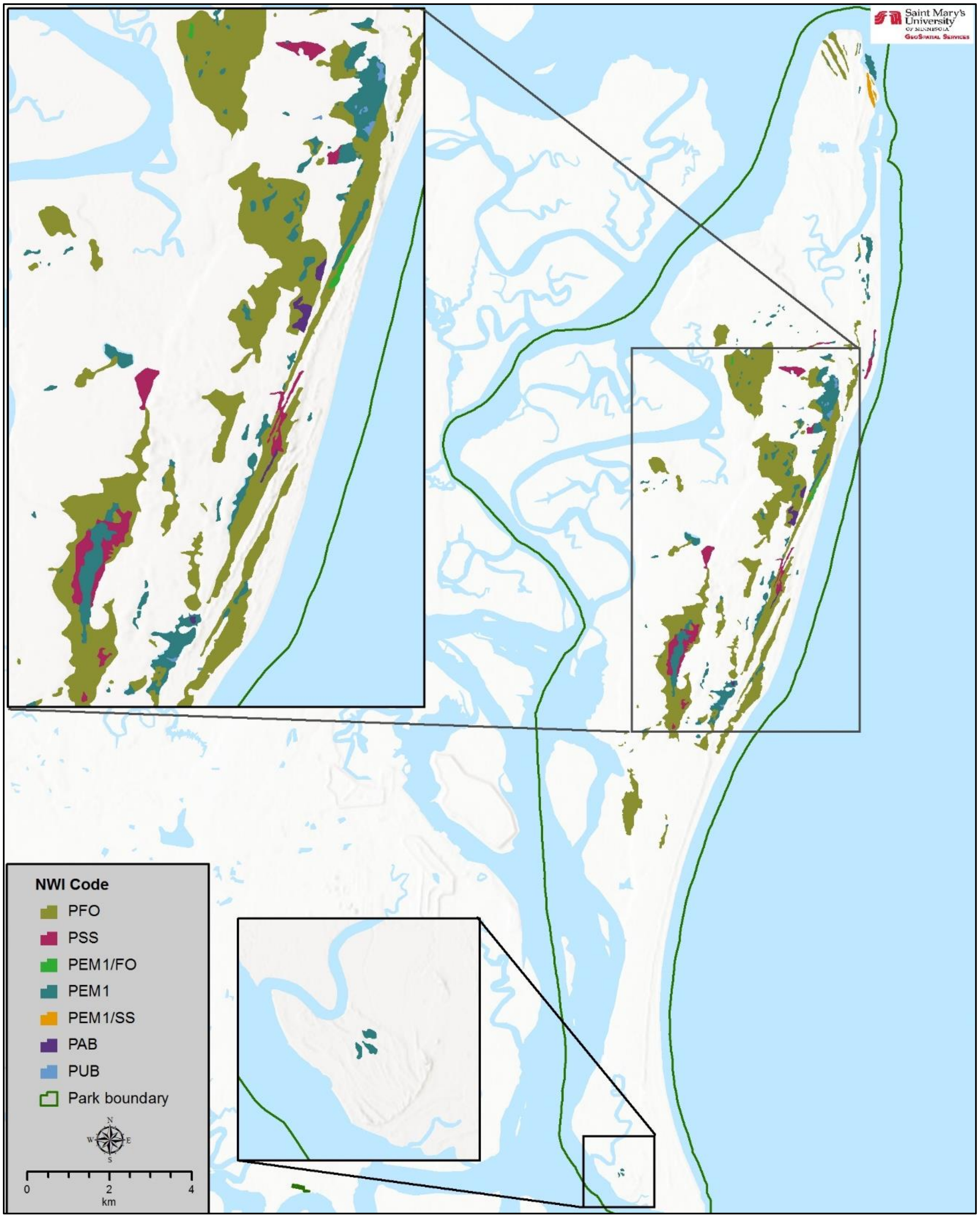


Figure 32. The location and extent of freshwater wetlands at CUIS, as mapped by the NWI (GIS data received from CUIS, originally obtained from USFWS 2012).

When analyzed by water regime, seasonally flooded (C) wetlands are most extensive, with 507 ha (1,253 ac) (Table 31) (USFWS 2012). In seasonally flooded wetlands, "surface water is present for extended periods especially early in the growing season, but is absent by the end of the season in most years," (Cowardin et al. 1979, p. 39). Seasonally saturated (B) wetlands were also common, covering 438 ha (1,082 ac). In saturated wetlands, the substrate (i.e., soil) is wet for extended periods but standing water is rarely present. The only other water regime to cover >100 ha (247 ac) was semipermanently flooded (F), with nearly 154 ha (378 ac) (USFWS 2012). In these wetlands, standing water is present throughout the growing season in most years (Cowardin et al. 1979). Permanently flooded (H) wetlands were uncommon, with just over 5 ha (12 ac) (Table 31). A table with full NWI codes (including modifiers) and acreages for all freshwater wetlands at CUIS can be found in Appendix C.

Table 31. Extent of freshwater wetlands by water regime, as mapped by the NWI (USFWS 2012). A = Temporary Flooded, B = Seasonally Saturated, C = Seasonally Flooded, F = Semipermanently Flooded, H = Permanently Flooded.

Water Regime	# of Wetlands	Total Area (ha/ac)	% of Wetland Area
A	1	8.3 (21)	0.7
B	29	437.8 (1,082)	36.7
C	145	507.2 (1,253)	42.5
F	46	153.7 (380)	12.9
H	9	5.1 (13)	0.4

Plant Species Diversity by Wetland Type

The diversity of plant species within various freshwater wetland types at CUIS has received little attention. Hillestad et al. (1975) noted the characteristic species of each of the four broad wetland types, but only provided more detailed species composition information for one community - lowland mixed hardwood forests – and uncommon tree/shrub species were excluded from the data tables. Information for other wetland types was likely lumped into interdune community data. A total of 28 plant species were reported from the lowland mixed hardwood forests (Hillestad et al. 1975).

Zomlefer et al. (2008) provided an annotated plant species list for CUIS, which included information on the general habitat where each species was found. Three of these general habitat types could be considered freshwater wetlands: pond-slough, marsh, and swamp. Pond-slough habitat typically supports open water, marsh consists of emergent herbaceous vegetation and shrubs, and swamps are wooded wetland areas (Zomlefer et al. 2008). Between this annotated species list and a later addendum (Zomlefer and Kruse 2011), 94 plant species were documented in pond-slough habitat, 18 species in marsh, and 37 species in swamp forests (Appendix D).

Heath and Byrne (2014) documented all the plant species documented at SECN vegetation monitoring sites during 2012 sampling. Three of these sites were located in freshwater wetlands, one each in South Atlantic Coastal Pond, Atlantic/East Gulf Coastal Plain Sweetbay - Tupelo Streamhead Forest, and Atlantic Coast Interdune Swale. During this single sampling season, surveyors documented 13 plant species at the coastal pond (herbaceous) site, 16 species at the streamhead forest, and 13 species at the interdune swale site (Heath and Byrne 2014).

Between these three studies, a total of 181 plant species have been documented in CUIS freshwater wetlands (Appendix D). Seven species are non-native but only one (alligatorweed [*Alternanthera philoxeroides*]) is considered invasive in Georgia (GA-EPPC 2016, NPS 2016f). Based on Zomlefer et al. (2008), just over half of all the plant species have been found in pond-slough habitats, suggesting that this may be the most diverse of the freshwater wetland communities surveyed to date.

Water Quality

Water quality influences the vegetation and aquatic organisms present within a wetland (UNEP 2008). Degraded water quality could reduce the biodiversity and productivity of wetlands, which can impact their ability to perform ecosystem services. Important parameters include temperature, pH, nutrients, salinity, clarity, and contaminants. The water quality of CUIS's freshwater wetlands is discussed in detail in Chapter 4.8 of this assessment.

Soil Quality

Soil quality also has a significant impact on the health and biodiversity of vegetation communities. Examples of soil quality parameters and the ecosystem components/processes they influence are given in Table 32. These factors are particularly important to freshwater wetlands. The soil quality of wetlands can vary both between and within sites depending on the frequency and duration of inundation (Winger 1986). During long-term flooding, for example, soils become anaerobic (lacking oxygen) which slows decomposition rates and nutrient cycling.

Table 32. Soil quality parameters and ecosystem components/processes they affect (Karlen et al. 1997).

Parameter	Component/Process Affected
organic matter content	nutrient cycling, water retention
infiltration	runoff/leaching potential, erosion potential, plant water use efficiency
pH	nutrient availability
microbial biomass	soil biological activity, nutrient cycling
bulk density	plant root penetration, soil biological activity
conductivity or salinity	water infiltration, plant growth
available nutrients	plant growth capacity

Most CUIS soils are acidic and highly permeable, with rapid recycling of nutrients (Hillestad et al. 1975). As organic matter decomposes, the nutrients released are quickly taken up by vegetation and do not remain in the soil long. Little is known about the soils of CUIS's freshwater wetlands specifically. During a study of vegetation response to fire, Davison and Bratton (1988) collected some soil data from several CUIS habitats in the northern portion of the island, including grass-dominated freshwater wetlands. These wetland soils were primarily low in nutrients with pH values between 3.6-4.4 (Davison and Bratton 1988). No more recent wetland soil quality information is available.

Threats and Stressor Factors

Threats to the park's freshwater wetlands include feral horse and hog activity, saltwater intrusion, dune encroachment, roads and trails, fire suppression, and climate change. Storm surge and sea level rise (SLR) can contribute to saltwater intrusion into freshwater wetlands, particularly wetlands close to the shoreline. Saltwater intrusion can occur over land during flooding or by encroaching into the shallow surficial water table (Hillestad et al. 1975, Frick et al. 2002). A sudden increase in salinity from saltwater intrusion can kill off aquatic vegetation not adapted to saline or brackish conditions, altering plant community composition (Hillestad et al. 1975, Zomlefer et al. 2008). If sea levels rise over the next century as predicted (IPCC 2013), overland saltwater intrusion is likely to become an increased threat to CUIS freshwater wetlands (Ataie-Ashtiani 2013).

As mentioned in Chapter 2 of this report, studies at CUIS have found that horse grazing activity, including trampling and plant consumption, significantly reduces vegetative cover, growth, and reproduction in these habitats (Turner 1986, Dolan 2002). Horses are regularly seen grazing in CUIS's freshwater wetlands, including Whitney Lake and Lake Retta (Figure 33) (Noon and Martin 2004, Dlugolecki 2012). In wetlands, feral horses can compact hydric soils, damage soil microorganisms, reduce water quality due to nutrient inputs from wastes, and spread non-native plant species (Noon and Martin 2004). Soil compaction reduces surface soil permeability, which impacts plant survival and soil formation processes. During a 2004 visit to CUIS, NPS Water Resource Division (WRD) staff concluded that, "maintenance of the feral horse herd causes unacceptable impacts to the park's wetland resources" (Noon and Martin 2004, p. 1).

Figure 33. Feral horses grazing in the Lake Retta wetland on CUIS in 2004 (NPS photo).

Prior to park establishment, horse and cattle grazing in dune areas de-stabilized the dunes and allowed sand to encroach west toward the interdunes and other wetlands (Hillestad et al. 1975). This was particularly notable on the northeast shore of Whitney Lake, and also threatened the Sweetwater Complex (Hillestad et al. 1975). Since cattle have been removed from the island and vegetation has recovered, these dunes have become more stable and the threat of encroachment has declined (Dlugolecki 2012). However, vegetation loss due to continued horse grazing, storm damage, or droughts could trigger renewed dune encroachment into freshwater wetlands (Hillestad et al. 1975, Pye 1983).

Feral hog rooting is also destructive to wetlands. This activity damages groundcover and prevents plant seedling recruitment (Heath and Byrne 2014). It also causes significant soil disturbance (Figure 34), which provides opportunities for invasive plant species establishment (Heath and Byrne 2014). During 2011 bird surveys focused on CUIS's freshwater wetlands, hogs were regularly seen rooting in the wetlands (Dlugolecki 2012).

Figure 34. Hog rooting activity in a wetland on the northern end of CUIS (NPS photo).

Many of the freshwater wetland communities at CUIS had evolved with frequent fires, which maintained the open character and diversity of these areas (Heath and Byrne 2014). During the late 1800s and early 1900s, island residents used prescribed fire to keep several wetlands and sloughs open for recreation (e.g., waterfowl hunting), including Whitney Lake and Willow Pond (Turner 1983, Dlugolecki 2012). Burning removes accumulated organic matter (vegetation, peat, etc.), which can fill in depressions capable of holding standing water, and sets back succession towards mesic, wooded vegetation types (Hillestad et al. 1975, Bellis 1995). Burning also appears to maintain plant species diversity in CUIS wetlands (Davison and Bratton 1988). However, fire has been largely suppressed on the island since the mid-1900s, and these wetlands have experienced woody species encroachment and drying due to filling with organic matter (Bellis 1995, Heath and Byrne 2014).

Roads and trails at CUIS have altered the island's hydrology (surface runoff patterns, wetland connectivity, etc.) which has impacted the park's freshwater wetlands (Hillestad et al. 1975, Alber et al. 2005). Some causeways on the island have few culverts to allow water flow, including Duck House Road, Willow Pond Trail, Roller Coaster Trail, South Cut Trail, and North Cut Road (Figure 35) (Alber et al. 2005). This can trap additional water in some wetlands and reduce the water supply to others. Altering the water regime of wetlands likely influences the vegetation and wildlife communities they support (Hillestad et al. 1975). Roads can also act as fire breaks, preventing the spread of fire into wetlands that would benefit from burning (Turner 1983).

As discussed in Chapter 2, temperatures are projected to increase across the southeastern U.S. over the next century as a result of global climate change (Carter et al. 2014). Warmer temperatures will increase evapotranspiration rates, meaning that even if annual precipitation remains constant or slightly increases, overall conditions could become drier in the future (Carter et al. 2014). The frequency and intensity of droughts is also projected to increase with higher temperatures (Karl et al. 2009), which will likely have a negative impact on wetland vegetation.

Data Needs/Gaps

Information regarding the island's freshwater wetland ecology and surface water resources in general is limited (Frick et al. 2002). Additional research into the vegetation communities, soils, hydrologic regime (e.g., water quantity and persistence), and ecosystem processes (e.g., nutrient cycling, disturbance) is needed to better understand these valuable resources. Specific examples of research interests related to freshwater wetlands on barrier islands include measurements of surface water drainage and groundwater transmissivity, microtopographic surveys to better map wetland habitat, water quality monitoring, and modeling of groundwater dynamics (Odum et al. 1986, Bellis 1995).

Overall Condition

Total Acreage

The NRCA project team assigned this measure a *Significance Level* of 3. The most recent estimate of total freshwater wetland acreage (776 ha [1,918 ac]) (McManamay 2017) was lower than the pre-settlement estimate of freshwater wetland acreage (960 ha [2,372 ac]) from Frost et al. (2011). However, the recent estimate was higher than the estimated wetland acreage at the time of park establishment (707 ha [1,747 ac]) (Hillestad et al. 1975). Since it appears likely that freshwater wetland acreage at CUIS has been reduced over time due to a combination of natural and anthropogenic factors, this measure is of moderate concern (*Condition Level* = 2).

Figure 35. Locations of roads and trails impacting hydrology and wetlands at CUIS (GIS data provided by NPS).



Lake Retta, an interdune freshwater wetland at CUIS, in spring 2015 (top, SMUMN GSS photo) and during a drought in 2004 (bottom, NPS photo).

Acreage by Wetland Type

This measure was also assigned a *Significance Level* of 3. Wooded wetlands have covered more acreage at CUIS than herbaceous and aquatic vegetation wetlands since the time of park establishment (Hillestad et al. 1975, USFWS 2012, McManamay 2017). This appears to represent a change from pre-settlement times, when temporarily flooded pond communities were most extensive (Frost et al. 2011). According to the most recent vegetation mapping (McManamay 2017), the most extensive freshwater wetland types are wet longleaf pine-pond pine flatwoods (224 ha [554 ac]) and outer coastal plain sweetbay swamp forest (121 ha [299 ac]). The most extensive herbaceous wetland type - south Atlantic coastal pond - covered 111 ha (274 ac). One unique type of freshwater wetland, the pocosin, occurred on CUIS historically but is no longer present, most likely due to fire suppression (Frost et al. 2011). Due to a seeming shift in extent of wetlands types from more open wetlands (e.g., pond-associated communities) to wooded wetlands, this measure is assigned a *Condition Level* of 2, indicating moderate concern.

Plant Species Diversity by Wetland Type

A *Significance Level* of 3 was assigned for plant species diversity. Several studies have included sampling and descriptions of wetland vegetation in some areas or communities within CUIS, and Zomlefer et al. (2008) described the general habitat of plant species found on the island. A total of 181 plant species have been documented in CUIS freshwater wetlands, with over half of these species found in pond-slough habitats. However, because of the lack of a comprehensive inventory of plant diversity in various CUIS freshwater wetland types and regular monitoring to detect any changes, a *Condition Level* is not assigned for this measure at this time.

Water Quality

The project team also assigned this measure a *Significance Level* of 3. Water quality impacts the vegetation and aquatic organisms present within freshwater wetlands. Data regarding CUIS water quality is limited (see Chapter 4.8) and, as a result, a *Condition Level* could not be assigned for this measure.

Soil Quality

Soil quality was assigned a *Significance Level* of 2. Like water quality, soil quality has a significant impact on the health and biodiversity of wetland vegetation communities. Limited information from a late-1980s study suggests that CUIS freshwater wetland soils are acidic (low pH) and generally nutrient poor, but no recent soil data are available to confirm or further assess CUIS wetland soil quality. Therefore, a *Condition Level* cannot be assigned for this measure.

Weighted Condition Score

A *Weighted Condition Score* was not calculated for CUIS's freshwater wetlands since *Condition Levels* could not be assigned for three of the five selected measures. Based on available literature and data, it is likely that freshwater wetlands are at least of moderate concern, but more information is needed to assess condition with any confidence. Therefore, the current condition and trend is considered unknown (Table 33).

Table 33. Weighted Condition Score of Freshwater Wetlands.

Freshwater Wetlands			
Measures	**Significance Level**	**Condition Level**	**WCS = N/A**
Total Acreage	3	2	
Acreage by Wetland Type	3	2	
Plant Species Diversity by Wetland Type	3	n/a	
Water Quality	3	n/a	
Soil Quality	2	n/a	

4.2.6. Sources of Expertise

John Fry, CUIS Chief of Resource Management

Brian Gregory, SECN Program Manager/Aquatic Ecologist

4.3. Salt Marshes

4.3.1. Description

The extensive salt marshes on the western side of Cumberland Island provide valuable feeding, breeding, and nursery habitat for birds, fish, and invertebrates (NPS 1984, Alber et al. 2005, Peek et al. 2016). Although these intertidal marsh areas lie within the CUIS boundary, not all of the marshes are owned by the NPS. Large portions are under the jurisdiction of the State of Georgia, with the GA DNR Coastal Resources Division having management authority (Alber et al. 2005), and some areas are owned by the U.S. military or private landowners. The vegetation in salt marshes is adapted to salinity, poorly-aerated soils, regular tidal inundation, and intense sunlight (Zomlefer et al. 2008). While salt marsh vegetation varies with elevation, soil type, salinity, and disturbance (Figure 36), the CUIS salt marshes can be broadly divided into "high" and "low" marshes. The low marshes, where inundation is more frequent and salinity fluctuates, are a near-monoculture of smooth cordgrass (*Spartina alterniflora*) (Hillestad et al. 1975, McManamay 2017). The high fringe marshes, with sandier soils and frequent exposure during low tides, support more diverse vegetation (Figure 36, Table 34). These high fringe marshes cover much smaller, narrower areas of CUIS than the low marshes, typically just below upland habitats (Peek et al. 2016, McManamay 2017).

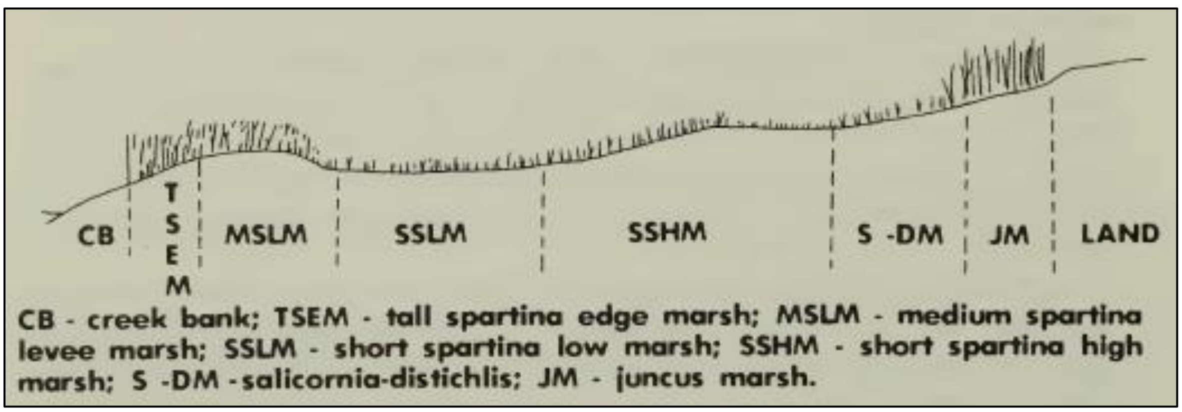


Figure 36. Profile of a salt marsh showing typical marsh types/habitats, with varying elevation and vegetation (Johnson et al. 1974b, adapted from Teal 1958). "Low marsh" includes the communities on the left through SSLM; "high marsh" includes SSHM and communities to its right.

Table 34. The two broad categories of salt marsh vegetation at CUIS and some common plants species in each (Dolan 2002, Peek et al. 2016).

Salt Marsh Vegetation Type	Common Species
High fringing marsh	needlegrass rush (*Juncus roemerianus*), saltgrass (*Distichlis spicata*), smooth cordgrass, chickenclaws (*Sarcocornia perennis*), turtleweed (*Batis maritima*), bushy seaoxeye (*Borrichia frutescens*)
Low marsh	smooth cordgrass, chickenclaws (*Sarcocornia perennis*)

Boardwalk overlooking low salt marsh at CUIS, just south of Dungeness (SMUMN GSS photo).

4.3.2. Measures

- Total acreage
- Percent of area grazed vs. non-grazed

4.3.3. Reference Condition/Values

As with previous vegetation community components, the ideal reference condition for salt marshes would be their condition prior to European settlement. Given the history of human alteration on and around the island, this is likely no longer practical. Therefore, best professional judgement will be used to evaluate condition for this assessment. Information presented in this NRCA on current condition can be used as a baseline for assessing condition in the future.

4.3.4. Data and Methods

Several of the data sources utilized for previous vegetation components were also used for this component, including Hillestad et al. (1975), Zomlefer et al. (2008), and McManamay (2017). These reports provide information on the extent (acreage) of salt marshes at various points in time.

Turner (1986) studied the impacts of feral horse grazing on CUIS salt marshes during 1983-84. Experimental plots (10x20 m [33x66 ft]), including horse exclosures as an ungrazed “control” for comparison, were established in a salt marsh on the south end of the island. The study also included an analysis of 1983 aerial photography to determine the total area of salt marsh accessible to feral horses (Turner 1986).

Dolan (2002) surveyed foredune-interdune and salt marsh vegetation at CUIS to study feral horse impacts in these communities. Salt marshes were first surveyed for species composition, percent cover, and height of smooth cordgrass between 23 July and 11 August 2000, during low tide (Dolan 2002). A total of 243 plots were sampled across salt marshes and a “grazing impact value” was calculated for plots where horse grazing was observed, based on the height of grazed vs. ungrazed cordgrass. In 2001, 40x40 m horse exclosures were installed in the study area (including two in the salt marsh) and vegetation was sampled inside and outside the exclosures from June-October (Dolan 2002). Dolan (2002) also used aerial photographs and GIS technology to calculate the areas of dune and salt marsh accessible to horses.

Peek et al. (2016) conducted a climate change vulnerability assessment for nine marine habitats at CUIS, including low salt marsh and high fringing salt marsh (HFSM). As part of the assessment, the extent of each marine habitat on and around the island was delineated. Field work to verify habitat delineations was conducted in 2014. Habitats were then assessed for their vulnerability to four climate-related stressors: temperature change, salinity change, sea level rise, and ocean acidification (Peek et al. 2016).

4.3.5. Current Condition and Trend

Total Acreage

Based on historic surveys and other sources, Frost et al. (2011) estimated that the pre-settlement extent of salt marshes on Cumberland Island was approximately 3,990 ha (9,860 ac) (Table 35). This accounted for over 36% of the vegetation mapped by Frost et al. (2011). The vast majority of this salt marsh vegetation was "low marsh" dominated by smooth cordgrass (Figure 37).

Table 35. Extent of salt marsh within the pre-settlement (around 1600) vegetation of CUIS (Frost et al. 2011).

Salt Marsh Community Type	Area (ha)	Percent of Total Area
Salt marsh islands and shoreline fringe hammocks	5	>0.1
Salt flat-salt marsh-brackish marsh mosaic	146	1.3
Salt marsh	3,839	35.2
Total	**3,990**	**36.6**

Hillestad et al. (1975) mapped 3,792 ha (9,370 ac) of salt marsh within CUIS around the time of park establishment (Table 36). Similar to Frost et al. (2011), this accounted for approximately 36% of the vegetation mapped. The majority of the salt marsh was again cordgrass-dominated low marsh (Hillestad et al. 1975).

The recent vegetation mapping effort for CUIS (McManamay 2017) identified 3,828 ha (9,460 ac) of salt marsh at CUIS, or nearly 40% of the area mapped (Table 37). Over 96% of the salt marsh area was low marsh. The remaining salt marsh vegetation types accounted for >1% of the total area mapped (McManamay 2017).

Peek et al. (2016) delineated just over 3,743 ha (9,250 ac) of salt marsh within CUIS boundaries (Figure 38). This accounted for 46% of all marine habitat mapped. The authors estimated that over 80% of this area was low marsh, dominated by smooth cordgrass, and less than 20% was HFSM (Peek et al. 2016).

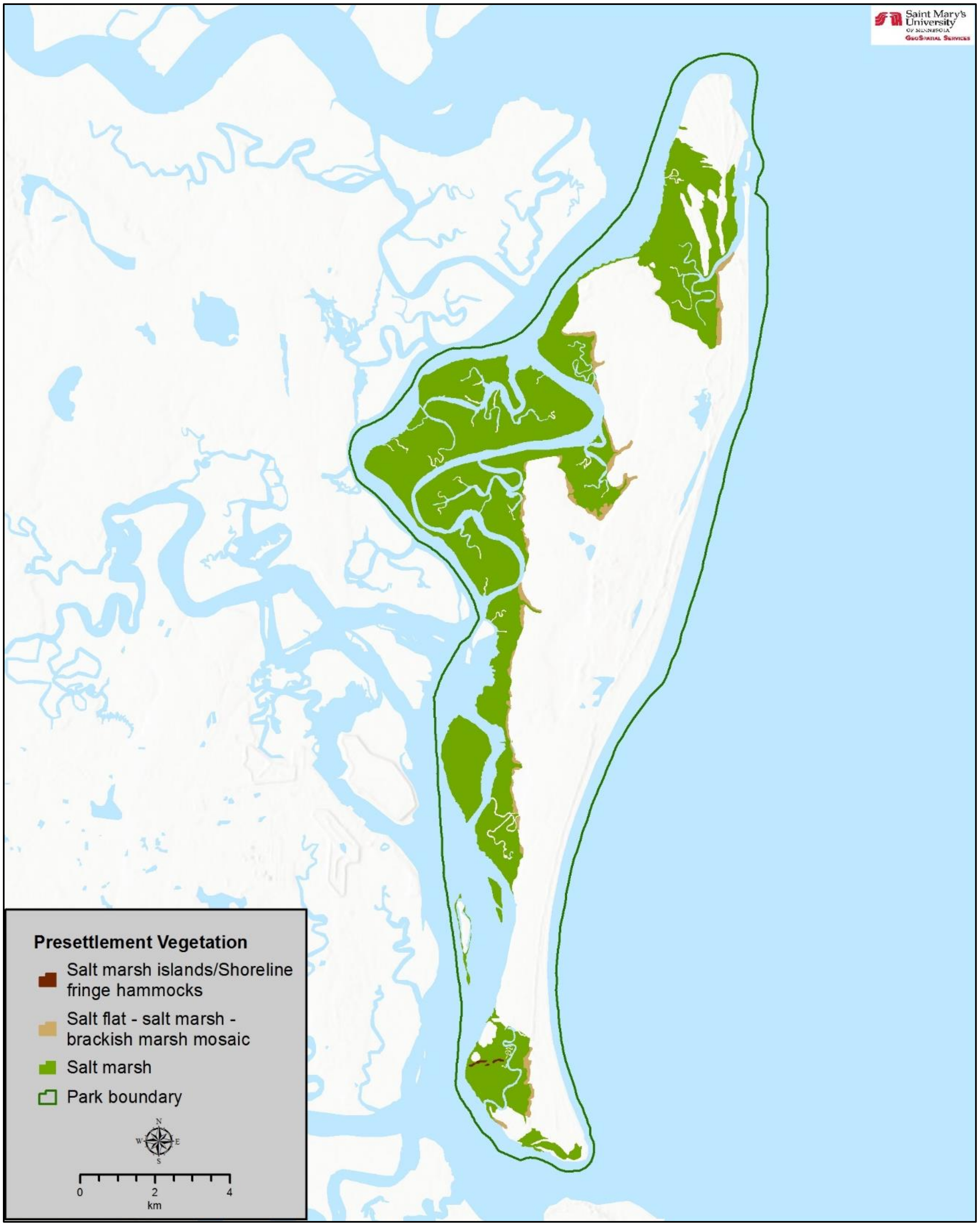


Figure 37. Estimated extent of salt marsh vegetation at CUIS prior to European settlement, as mapped by Frost et al. (2011).

Table 36. Extent of salt marsh vegetation types at CUIS in 1974 (Hillestad et al. 1975).

Salt Marsh Community Type	Cumberland Island Area (ha/ac)	Little Cumberland Isl. Area (ha/ac)	Total Area (ha/ac)	Percent of Total Area
Grass (*Spartina*)	3,214 (7,942)	350 (865)	3,564 (8,807)	34.3
Grass-forb-rush marsh	172 (425)	44 (109)	216 (534)	2.1
Shrub border	12 (30)	0	12 (30)	0.1
Total	**3,398 (8,397)**	**394 (973)**	**3,792 (9,370)**	**36.5**

Table 37. Extent of salt marsh vegetation types at CUIS based on 2011 aerial imagery (McManamay 2017).

Salt Marsh Community Type	Area (ha/ac)	Percent of Total Vegetated Area
Mid- and Southern Atlantic High Salt Marsh	74.3 (183.6)	0.8
Needlerush High Marsh	27.5 (68.0)	0.3
Sand Cordgrass - Seashore Mallow Herbaceous Vegetation	14.3 (35.3)	0.1
Seaside-tansy Tidal Shrub Flat	30.1 (74.4)	0.3
Coastal Salt Shrub Thicket	3.2 (7.9)	>0.1
Southern Atlantic Coast Salt Marsh / Salt Flat (Swampfire Type)	3,678.8 (9,090.5)	38.3
Total	**3,828.2 (9,459.7)**	**39.9**

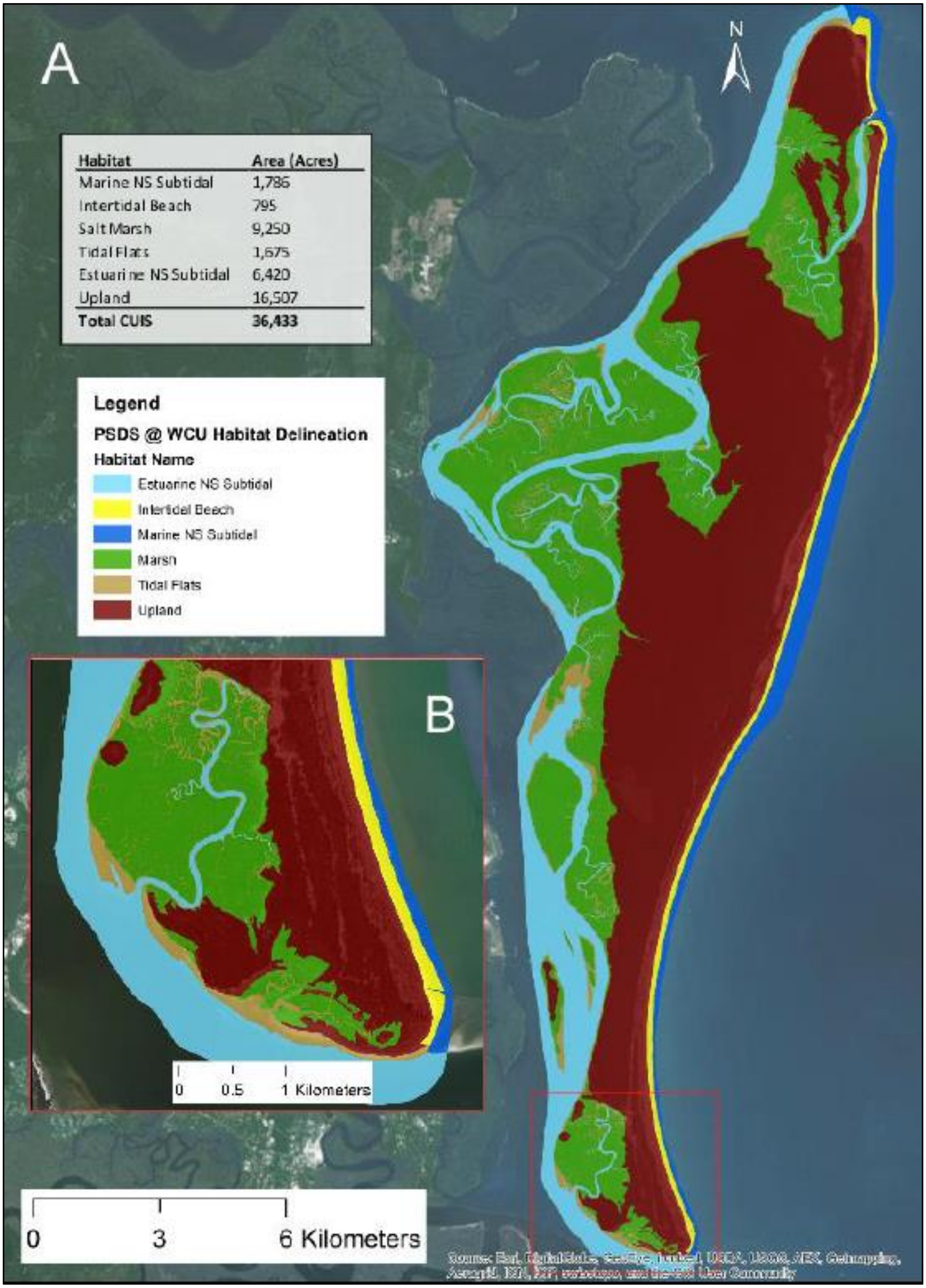


Figure 38. The extent of salt marsh (green) across CUIS (A) and a close-up example from the southern portion (B), as delineated by Peek et al. (2016). NS = Nearshore.

Low salt marsh (green in foreground) and high salt marsh (taller, in background) on the western side of Cumberland Island (Peek et al. 2016).

Percent of Area Grazed vs. Non-grazed

In an analysis of 1983 aerial photography of CUIS, Turner (1986) found that 411 ha (1,016 ac) of salt marsh were accessible to the feral horse population. According to personal observations by the author, around 50% of the accessible salt marsh area was severely overgrazed at that time (Turner 1986). A later GIS analysis by Dolan (2002) found that salt marsh accounted for just 3% of the total island area accessible to horses (6,333 ha), totaling approximately 190 ha (470 ac) (Figure 39). Over half of the accessible acreage is adjacent to the wilderness area; salt marsh areas are currently outside the wilderness area boundary but are considered “potential wilderness” (Dolan 2002; Fry, written communication, August 2017).

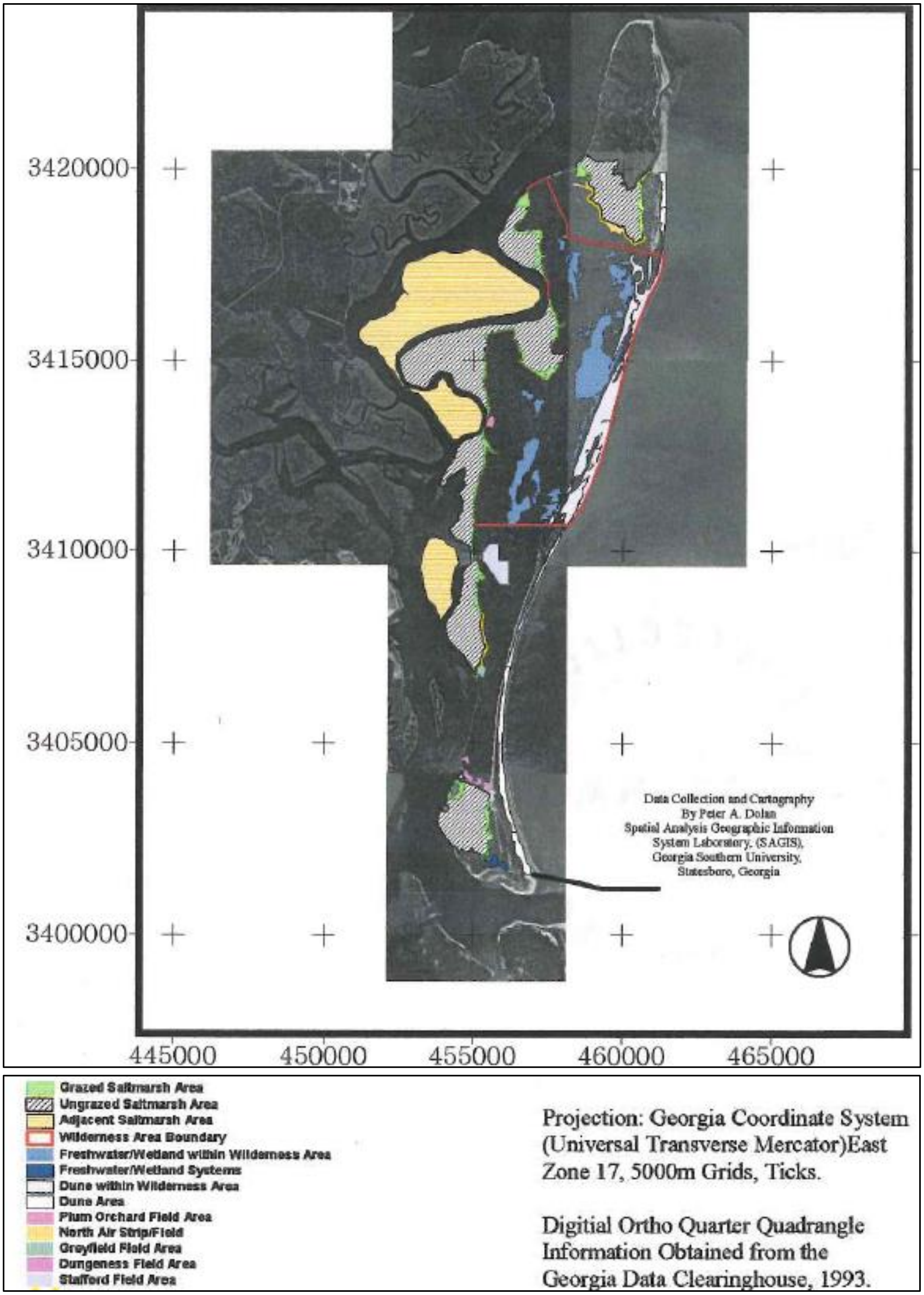


Figure 39. Horse grazing orthophoto map created by Dolan (2002). The green represents salt marshes grazed by horses at the time of the study. Additional close-up maps can be found in Appendix E.

Threats and Stressor Factors

Threats to the park’s salt marshes include feral horse and hog impacts, erosion along shorelines and creeks/channels, boat wakes, rising tide levels, dredge spoil piles, roads and trails, and sudden marsh dieback. Roads and trails have influenced salt marshes because they’ve altered the island’s hydrology (e.g., surface runoff patterns, sedimentation, etc.) and even cross some high salt marsh areas (NPS 1984, DeVivo et al. 2008). Dredge spoils on the south end of the island have filled in historic salt marsh habitat and also altered Beach Creek flow dynamics, which further influence the surrounding salt marshes (NPS 1984; Fry, personal communication, 7 March 2017).

Salt marshes are one of the habitats most frequently utilized by the island’s feral horses (Turner 1986, Dolan 2002). As mentioned previously, studies at CUIS have found that grazing activity, including trampling and plant consumption, significantly reduces vegetative cover, growth, and reproduction in these habitats (Figure 40) (Turner 1986, Dolan 2002). Turner (1986) found that the horse impact was greater in the high marsh than in the low marsh, with up to a 98% reduction in aboveground vegetation. Heavy grazing may also impact plant distribution, as species favored by horses (e.g., grasses) are replaced by plant species not eaten by horses (e.g., salt-tolerant forbs). Turner (1986) also hypothesized that heavily grazed salt marshes lacked sufficient vegetation to accrete sediment, making them more vulnerable to erosion and storm damage.

Figure 40. This photo shows the difference between a grazed area and an ungrazed area inside an exclosure within a CUIS salt marsh (NPS photo).

In contrast, Dolan (2002) found more intense horse grazing impacts in the low salt marshes. Grazed cordgrass was documented in 72% of low marsh plots and just 28% of high marsh plots. In the low marshes, grazed cordgrass was one-tenth of the height of cordgrass in control (ungrazed) plots, while in high marsh plots, grazed cordgrass was one-third of the height of ungrazed cordgrass (Dolan 2002). The percent cover of vegetation differed significantly between grazed and ungrazed plots in the low marsh, but the difference was not significant between high marsh plots. In addition, fewer flowering cordgrass stems were documented in grazed plots, suggesting that grazing significantly impacts grass reproduction. Dolan (2002) hypothesized that horses may favor low marsh over high marsh due to the presence of two different morphs or forms of cordgrass; the taller morph found in low marshes has a higher nutrient concentration (especially nitrogen) than the shorter morph found in high marshes. In addition to horses, feral hog rooting may also disturb CUIS's salt marshes (Kammermeyer et al. 2011, Sharp and Angelini 2016).

The western side of CUIS, where the majority of the salt marshes are located, has experienced significant shoreline erosion in recent decades (Figure 41) (Jackson 2006, Calhoun and Riley 2016). The erosion is occurring not only on the outer shoreline, but also along the banks of tidal creeks that meander through the salt marshes (Jackson 2006). Potential factors contributing to this erosion include SLR, storm surge, altered sediment dynamics due to channel modification, and boat wakes (Jackson 2006, Peek et al. 2016). Ongoing residential and recreational developments along the Georgia coast near CUIS may be increasing boat traffic near the park's western shore, further accelerating marsh erosion rates (Jackson 2006). Shoreline erosion will be discussed in more detail in Chapter 4.10 of this report.

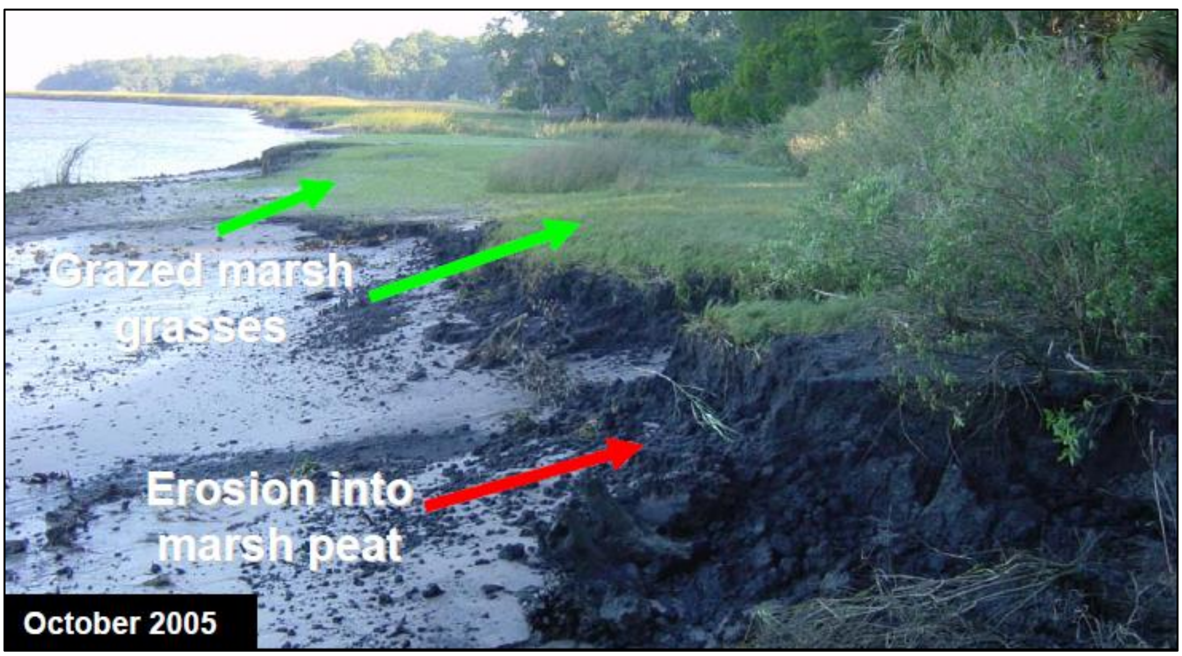


Figure 41. Evidence of shoreline marsh erosion and grazing on the western side of CUIS near Plum Orchard (Jackson 2006).

Sea level rise is occurring in the area around CUIS, averaging 2.1 mm/yr (0.08 in/yr) from 1897-2015 (see Figure 17 in Chapter 2) (NOAA 2016). As a result of global climate change, the rate of SLR is expected to increase during the remainder of the 21st century, with an overall rise between

0.28-0.98 m (0.9-3.2 ft) by 2100 (IPCC 2013). These rising waters will force salt marsh vegetation to “migrate” up the shore to find new suitable habitat (Peek et al. 2016). Cordgrass-dominated low marsh will likely move into current HFSM areas at CUIS, but Peek et al. (2016) suggest that HFSM may not be able to migrate into adjacent uplands due to the more significant elevation difference in many areas. This, combined with the smaller area currently occupied by HFSM, led Peek et al. (2016) to the conclusion that HFSM is the marine habitat most vulnerable to climate change at CUIS.

Around 2000, sudden dieback events were first reported in salt marshes along the Southeast and Gulf coasts (McFarlin 2012). There was no obvious cause for this abrupt vegetation loss, which started as yellowing and thinning of grasses, but quickly progressed to large bare patches where salt marshes had been. These diebacks, which impacted over 800 ha (1,977 ac) of marsh in Georgia, were associated with a severe drought from 2000-2002 (McFarlin 2012). At a dieback site in Louisiana, McKee et al. (2004) found that desiccated soils and standing dead grass contained elevated levels of iron aluminum. The authors suggested that oxidation in dried soils (due to drought) may have increased the concentration of available metals in the soils, which were then taken up by plants in toxic levels. Recovery from such dieback events is slow, and such events are likely to increase with climate change and anthropogenic influence (McFarlin 2012).

A potential site of sudden salt marsh marsh dieback towards the northern end of CUIS in 2008 (NPS photo).

Data Needs/Gaps

The percent of salt marsh area grazed by horses versus ungrazed has not been evaluated since Dolan (2002). This could be done by aerial imagery photointerpretation by an analyst familiar with the photosignatures of grazed vs. ungrazed salt marsh vegetation. Jackson (2006) noted that further investigations into salt marsh stressors (e.g., boat wakes, SLR, coastal storms, grazing) would help in assessing the full impacts and identifying strategies to mitigate those effects. Lastly, monitoring of HFSM area over time could determine if this community is being encroached upon by cordgrass low marsh as sea levels rise.

Overall Condition

Total Acreage

The NRCA project team assigned this measure a *Significance Level* of 3. Frost et al. (2011) estimated that, prior to European settlement, salt marsh covered approximately 3,990 ha (9,860 ac) at CUIS. More recent delineations of salt marsh extent at CUIS have ranged from 3,743 ha (9,250 ac) (Peek et al. 2016) to 3,828 ha (9,460 ac) (McManamay 2017). These acreages are all relatively similar, and some variance may be due to differences in study boundaries or mapping methodologies. As a result, this measure is currently of no concern (*Condition Level* = 0)

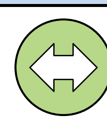
Percent of Area Grazed vs. Non-grazed

This measure was also assigned a *Significance Level* of 3. According to an analysis by Dolan (2002), 190 ha (470 ac) of salt marsh on CUIS was accessible to feral horses. This is a decrease from the 411 ha (1,016 ac) that Turner (1986) identified as accessible to horses in the early 1980s. However, this analysis has not been repeated within the past 15 years and it is unknown if the percent of grazed area has changed during this time. Yet given the known negative impacts of grazing on salt marshes and the concern expressed by park managers, this measure is assigned a *Condition Level* of 2.

Weighted Condition Score

The *Weighted Condition Score* for CUIS salt marsh is 0.33, indicating good condition. This score is on the edge of the moderate concern range and any small decline in condition could shift the community to moderate concern. The trend is currently unchanging and the confidence level is moderate, given the lack of recent information regarding grazed vs. ungrazed salt marsh area (Table 38).

Table 38. Weighted Condition Score of Salt Marsh.

Salt Marsh			
Measures	**Significance Level**	**Condition Level**	**WCS = 0.33**
Total Acreage	3	0	
Percent of Area Grazed vs. Non-grazed	3	2	

4.3.6. Sources of Expertise

Lisa Baron, SECN Coastal Ecologist

John Fry, CUIS Chief of Resource Management

4.4. Interdune Communities

4.4.1. Description

Interdune communities occur between the sparsely vegetated foredunes adjacent to the beach and the older, more vegetated rear or backdunes that border the island's maritime forests (Figure 42) (Zomlefer et al. 2008). At its widest, the interdune area extends approximately 340 m (1,115 ft) (Dolan 2002). These flats support a wide variety of plant species but are most often dominated by grasses, sedges, and rushes (Hillestad et al. 1975, Zomlefer et al. 2008). Ponds or sloughs often form in depressions within the interdunes, supporting wetland vegetation and providing habitat for a variety of wildlife, including aquatic invertebrates, amphibians, and waterbirds (Hillestad et al. 1975).

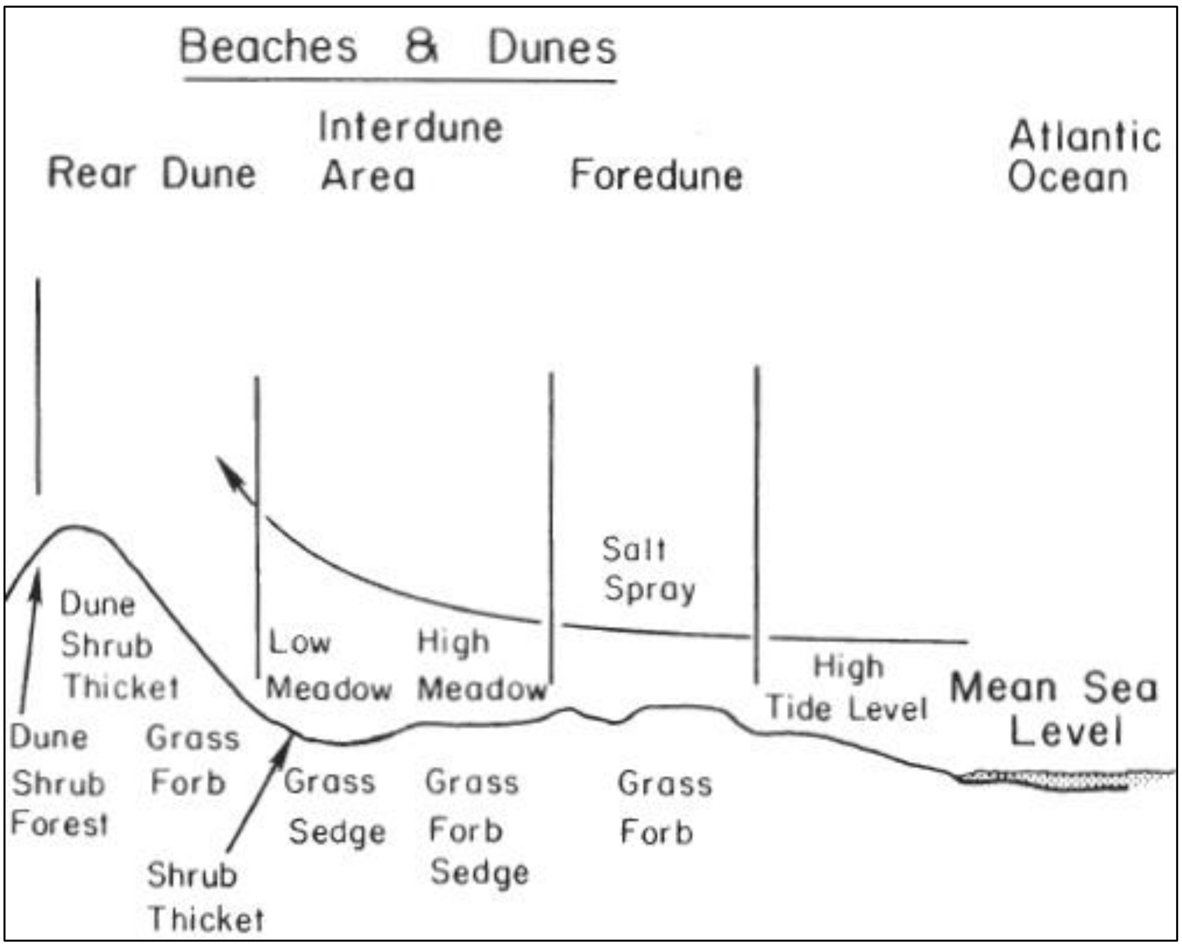


Figure 42. Location of the interdune communities on CUIS, relative to the beach and dunes (reproduced from Hillestad et al. 1975).

The structure and plant species composition of interdune communities varies throughout the island depending on microhabitat characteristics such as salinity, disturbance history (e.g., grazing, fire), and elevation (especially relative to the water table) (Hillestad et al. 1975, Dolan 2002, Frost et al. 2011). The height and stability of bordering dunes, which offer protection from wind, salt spray, and storm surges, also plays a role (Hillestad et al. 1975, Frost et al. 2011). Common plant species of interdune areas include seashore paspalum (*Paspalum vaginatum*), cordgrasses (*Spartina* sp.), flatsedges and nutsedges (*Cyperus* spp.), largeleaf pennywort (*Hydrocotyle bonariensis*), and wax-myrtle (*Morella cerifera*) (Hillestad et al. 1975, McManamay 2017). Table 39 presents two examples of vegetation associations that occur in interdune communities on CUIS and common plant species in each.

Table 39. Examples of vegetation associations that occur in CUIS interdune communities and common plant species in each (McManamay 2017).

Vegetation Community	Common Plant Species
Atlantic Coast Interdune Swale	wax-myrtle, cabbage palmetto, greenbriars, muscadine grape, dogfennel (*Eupatorium capillifolium*), umbrella pennyroyal (*Hydrocotyle umbellata*), turkey tangle fogfruit, bluestem grasses
Southern Hairgrass - Saltmeadow Cordgrass - Dune Fingergrass Herbaceous Vegetation	hairawn muhly (*Muhlenbergia* sp.), bluestem grasses, needleleaf rosette grass (*Dichanthelium aciculare*), dogfennel, turkey tangle fogfruit, greenbriars, wax-myrtle

The interdune community consists of the flat area between the foredunes (in the foreground) and the wooded backdunes (in the background) (SMUMN GSS photo).

4.4.2. Measures

- Acreage of communities
- Plant species diversity

4.4.3. Reference Condition/Values

As with previous vegetation components, the ideal reference would be pre-settlement conditions. However, information from this time is limited and restoring current vegetation to such conditions is not practical, given human use and alterations. Best professional judgement will be used to evaluate condition for this assessment, and information presented here related to current condition may be used as a baseline for assessing condition in the future.

4.4.4. Data and Methods

Several vegetation components that previously appeared in this NRCA (e.g., Chapter 4.1, 4.2) used and described many of the data sources that were used for the interdune communities component. These sources include Hillestad et al. (1975), Frost et al. (2011), Zomlefer et al. (2008), Zomlefer and Kruse (2011), SECN vegetation monitoring reports (Byrne et al. 2012, Heath and Byrne 2014), and McManamay (2017). Of the SECN monitoring locations summarized by Byrne et al. (2012) and Heath and Byrne (2014), only one fell within an interdune vegetation community.

Hillestad et al. (1975) sampled dune and interdune vegetation using 10 transects spread along the island's entire shoreline. Transects ran perpendicular from the shoreline and extended from the high tide line on the beach to the inland edge of the rear dunes. Vegetation was sampled in quadrats at 9-m (30-ft) intervals along each transect, resulting in a total of 151 sampling points (Hillestad et al. 1975).

Dolan (2002) surveyed foredune-interdune and salt marsh vegetation at CUIS to study feral horse impacts in these communities. Foredune-interdune vegetation was first surveyed from June-August 2000, with 10 random points selected per kilometer of available habitat along the entire island. Plant species composition and percent cover were sampled within 1x2 m (3.3x6.6 ft) quadrats in foredune and interdune subhabitats. In May of 2001, 40x40 m (131x131 ft) horse exclosures were installed in the study area (including three in the interdunes) and vegetation was sampled inside and outside the exclosures from June-October (Dolan 2002).

To provide some insight into the current extent and locations of interdune communities, an SMUMN GSS analyst used aerial imagery and collateral data to digitize interdune areas. A shaded relief map derived from 2010 Coastal Georgia LiDAR (Light Detection and Ranging) elevation data (available through NOAA [2017]) was used to locate low areas between higher areas (foredunes and backdunes) on the eastern side of CUIS. In Esri ArcMap, polygons were created (traced) around these low, interdune areas. Aerial imagery from 2015 showing vegetation was then utilized to further refine the interdune polygons. The "calculate geometry" tool was used to find the area of each interdune polygon. Because this process was done relatively quickly and no field verification or ground-truthing was performed, these results should be considered a rough estimate of interdune community extent. Some interdune areas were too small to be mapped using aerial imagery, and therefore this estimate is likely conservative (i.e., a minimum or underestimate).

4.4.5. Current Condition and Trend

<u>Acreage of Communities</u>

Determining and comparing the acreage or extent of interdune communities over time is challenging, as the definition/delineation of "interdune community" appears to have varied between studies and investigators. Hillestad et al. (1975) considered pine-mixed hardwood forests on broad flats an advanced successional stage of the interdune community, while Frost et al. (2011) included only herbaceous and shrub vegetation types within the interdune community. As a result of this variation, the acreage estimates from different vegetation mapping efforts are likely not directly comparable.

Frost et al. (2011) divided historic interdune communities into three types, based on successional status: early phase, intermediate phase, and late phase. The early phase included a mix of maritime grasslands and wax myrtle flats. The intermediate phase consisted of a more shrubby mix of cabbage palmetto and wax myrtle with an herbaceous understory. The late phase included the wettest vegetation types, such as freshwater marshes and slash pine pools. Frost et al. (2011) estimated that the pre-settlement vegetation of Cumberland Island included just over 344 ha (851 ac) of interdune communities, accounting for just 3.1% of the total island area. The late successional phase comprised the greatest area of the three phases, with 147 ha (363 ac) identified (Table 40).

Table 40. Extent of interdune communities within the pre-settlement (around 1600) vegetation of CUIS (Frost et al. 2011).

Interdune Community Type	Area (ha/ac)	Percent of Total Area
Interdune Flats, Primary Succession – Early Phase (grass-myrtle mosaic)	123.8 (306)	1.1
Interdune Flats, Primary Succession - Intermediate Phase	73.7 (182)	0.7
Interdune Flats, Primary Succession - Late Phase (organic accumulation and ponding)	146.9 (363)	1.3
Total	**344.4 (851)**	**3.1**

According to Hillestad et al. (1975), interdune vegetation communities covered a total of 603.5 ha (1,491 ac) within CUIS in 1974 (Table 41). This accounted for 5.8% of the island area. The pine-mixed hardwood community was most prevalent, covering nearly 262 ha (647 ac) (Hillestad et al. 1975). Hillestad et al. (1975, p. 86) described this hardwood community as "an advanced successional stage of the interdune shrub thicket community but is located in more protected areas."

Table 41. Extent of interdune community vegetation types at CUIS in 1974 (Hillestad et al. 1975).

Interdune Community Type	Cumberland Island Area (ha/ac)	Little Cumberland Isl. Area (ha/ac)	Total Area (ha/ac)	Percent of Total Area
Grass-sedge	146.4 (362)	9.2 (23)	155.6	1.5
Interdune Shrub Thicket	179.8 (444)	6.2 (15)	186.0	1.8
Pine-Mixed Hardwood	261.9 (647)	0	261.9 (647)	2.5
Total	**588.1 (1,453)**	**15.4 (38)**	**603.5 (1,491)**	**5.8**

Because the recent NPS I&M mapping effort (McManamay 2017) classified communities/associations strictly by vegetation composition and did not consider geographic location on the island, it is extremely difficult to estimate the total acreage of interdune communities. Vegetation associations that occur in interdune communities (e.g., rush marsh/sawgrass head) are also found in other locations on the island and it would require further analysis to determine which mapped polygons fall within interdune locations. Therefore, no acreage estimates for interdune communities from this study are available at this time.

A cursory digitizing of interdune areas by an SMUMN GSS analyst suggests that interdune communities cover at least 346 ha (855 ac) at CUIS (Figure 43). This is based on 2010 elevation data and 2015 aerial imagery, and should be considered a rough estimate.

Figure 43. The approximate extent of interdune communities on CUIS, based on 2010 LiDAR data (NOAA 2017a) and 2015 aerial imagery provided by Esri.

Plant Species Diversity

Since park establishment, various surveys have observed nearly 120 plant species within the interdune communities of CUIS (Appendix F). Twelve of these species are non-native, but only two are considered invasive by GA-EPPC (2016), common mullein (*Verbascum thapsus*) and Bermudagrass (*Cynodon dactylon*). Over 90% of the species documented are herbaceous, and nearly half are graminoids (e.g., grasses, sedges, rushes). There are some notable differences between the plant species documented by Zomlefer et al. (2008) and those documented by Hillestad et al. (1975) and Dolan (2002). Approximately a dozen native species reported by both Hillestad et al. (1975) and Dolan (2002) were not documented by Zomlefer et al. (2008) in the interdunes, and around 30 native species reported by Zomlefer et al. (2008) had not been found by previous surveys. However, it is likely that at least some of this variation is due to differences in the definition/delineation of "interdunes" and in survey effort, rather than change in plant species diversity over time.

Dolan (2002) noted that the interdunes were more diverse than the neighboring foredunes, with an average of 6.8 species per 2-m^2 (21.5-ft^2) plot in the interdunes compared to just 3.0 species per plot in foredunes. Dolan (2002) also found that interdune plant species richness varied across the island, with the richest plots occurring at the south end and within the wilderness area.

<u>Threats and Stressor Factors</u>

Threats to the interdune communities identified by NPS staff include disturbance from feral horse and hog activity, dune migration and loss, prolonged drought, and severe storm impacts (e.g., saltwater intrusion). The risks to interdune communities from saltwater intrusion during storm surge are similar to those discussed in the freshwater wetlands component. In addition to actual saltwater intrusion, interdune vegetation may be negatively impacted by an increase in salt spray (aerosol) during storms with high winds, given its proximity to the ocean (Michener et al. 1997, Kerr 2000).

Changes in the island's foredunes can impact the interdune communities, as the foredunes provide interdune areas with some protection from ocean forces (e.g., wind, waves, salt spray). The most dramatic changes to foredunes often occur during hurricanes or other strong storms; high winds and storm surge can reduce dune elevation, cause partial dune erosion, or completely flatten smaller dunes (Edmiston et al. 2008). Some of the sand from these dunes may wash into the areas behind the foredunes, possibly covering existing interdune vegetation and filling depressions that support wetlands. Even if dunes survive hurricanes, the salt spray and surge may destroy dune vegetation (Edmiston et al. 2008); this vegetation loss can destabilize the dunes, exposing them to accelerated migration/loss over time. When foredunes are damaged or destroyed by storms, the interdune communities behind them are more exposed to the elements (e.g., wind, salt spray), which is likely to negatively impact interdune vegetation. Dune loss and migration can occur on a smaller scale due to dune vegetation loss from grazing, human disturbance, or prolonged drought (Hillestad et al. 1975, Dolan 2002).

Interdune areas are one of the habitats most frequented by feral horses at CUIS (Turner 1986, Dolan 2002). In addition to the general negative impacts of horse grazing and trampling discussed in previous sections of this assessment, Dolan (2002) documented grazing impacts on plant community composition and cover in the interdunes. Evidence of grazing was observed in 50% of interdune sampling plots (compared to 30% of foredune plots) and 11 of 17 grass species present in the interdunes were grazed. The three most frequently grazed plants were Canadian horseweed (*Conyza canadensis*), sedges, and sea oats (*Uniola paniculata*). Plant species richness was slightly higher (four species) inside Dolan's (2002) horse exclosures and vegetative cover was also significantly higher (by about 35-40%) just 2 months after exclosures were completed (Figure 44). In addition, Dolan (2002) found evidence that horse grazing limits plant reproduction late in the growing season, as the number of species flowering was higher inside than outside exclosures.

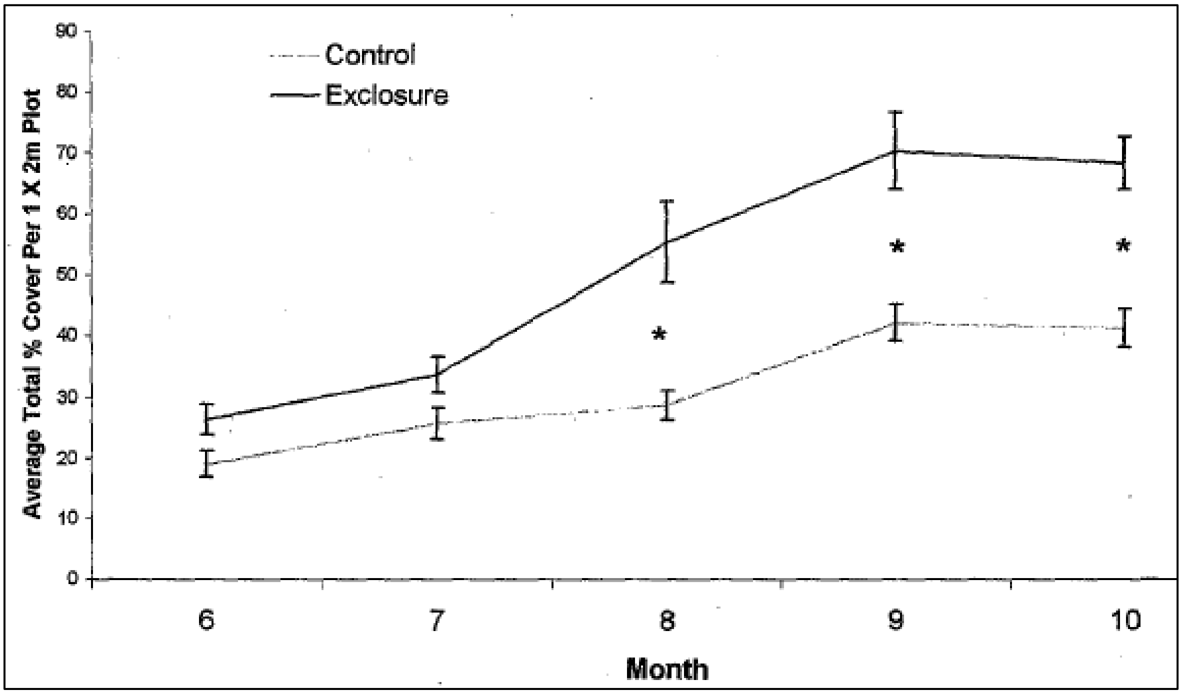


Figure 44. The average percent total vegetation cover (± standard error) for all interdune exclosures and control plots from June (6) through October (10) 2001 (Dolan 2002). * indicates statistical significance ($p<0.05$).

Data Needs/Gaps

Scientific study of interdune communities specifically has been limited. A focused survey of the area would help researchers and managers better understand the distribution and interaction of the various vegetation communities and plant species within the interdunes. Also, documenting environmental variables (e.g., climate, soils, surface water presence, physical disturbance) will assist in understanding how environmental factors influence interdune communities and potentially the value of the ecosystem services they provide (e.g., wildlife habitat, water storage, nutrient cycling).

Overall Condition

Acreage of Communities

The NRCA project team assigned this measure a *Significance Level* of 3. Because of some variation in what is included within "interdune communities" between studies/investigators and a lack of consistency in how to map them, the acreages found by various studies over time cannot be compared with any confidence. Therefore, a *Condition Level* is not assigned at this time.

Plant Species Diversity

Plant diversity was also assigned a *Significance Level* of 3. Various surveys over time have observed nearly 120 plant species within the interdune communities of CUIS, 90% of which are native. Differences between the plant species reported from the interdunes over time (Hillested et al. [1975] and Dolan [2002] vs. Zomlefer et al. [2008]) suggest that the definition/delineation of interdune communities has not been consistent, with some vegetation types being considered "interdune" by one study but not another. As a result of this uncertainty, it is not practical to compare plant diversity between studies or to confidently assess current condition. A *Condition Level* will not be assigned for this measure.

Weighted Condition Score

A *Weighted Condition Score* was not calculated for CUIS's interdune communities, given that a *Condition Level* could not be assigned with any confidence for either of the two selected measures. At this time, the condition and any trends within these communities is unknown (Table 42).

Table 42. Weighted Condition Score of Interdune Communities.

Interdune Communities			
Measures	**Significance Level**	**Condition Level**	**WCS = N/A**
Acreage of Communities	3	n/a	
Plant Species Diversity	3	n/a	

4.4.6. Sources of Expertise

John Fry, CUIS Chief of Resource Management

Doug Hoffman, CUIS Biologist

4.5. Mammals

4.5.1. Description

The mammalian community of CUIS has been well documented over the past 130 years, with several extensive surveys and checklists being completed on the island. NPS (2016a) confirms the presence of 26 mammal species in the park, and an additional five mammals that are likely to occur in the park. Two marine mammal species have been confirmed in the park (West Indian manatee [*Trichechus manatus latirostris*], and the bottlenose dolphin [*Tursiops truncates*]), with several other possibly occurring species (Appendix G).

The structure of the mammalian community at CUIS has undergone a relatively rapid change over the past 100 years. Several species that were once found on the island, such as the southeastern pocket gopher (*Geomys pinetis*; last documented in 1970 [Hillestad et al. 1975]), common gray fox (*Urocyon cinereoargenteus*; last reported in 1930 [Bent 1940]), and American black bear (*Ursus americanus*; likely extirpated between 1960-1970s [Hillestad et al. 1975]), have been extirpated from the island in the last century. Conversely, other mammal species have established new populations on CUIS in the past century without intentional human intervention. Such species include the nine-banded armadillo (first documented in 1973 [Hillestad et al. 1975]), Virginia opossum (*Didelphis virginiana*; likely reestablished on island in 1993) and the coyote (naturally expanded to island in 2004) (Webster 2010). The bobcat (Figure 45) was historically common on the island until the late 1900s when the species was wiped out by disease (Harper 1927). Reintroduction efforts in 1972-1973, and again in 1989-1990 reestablished a now thriving population on the island (Hillestad et al. 1975, Ragsdale 1993, Webster 2010).

Figure 45. A bobcat at CUIS (NPS photo).

Cumberland Island has been host to several non-native mammal species since the first European settlements on the island. The Franciscan Spaniards arrived at Cumberland Island in 1578 and brought livestock with them (Dilsaver 2004). It is believed that cattle, horses, and hogs were first brought to the island by about 1597 (Bullard 2003, Dilsaver 2004). Historic records indicate that the mast from live oaks on the island provided bountiful forage for hogs, and the high marshes provided forage for cattle and horses (Hillestad et al. 1975). Livestock were likely removed from the island following the conclusion of the American Civil War, and were later reintroduced when the Thomas Carnegie Family brought additional cattle, hogs, and horses to the island when they purchased much of the land (Hillestad et al. 1975).

When the NPS acquired CUIS land in 1972, extensive efforts were carried out to remove feral livestock from the park, and by 1980 nearly all feral cattle had been removed from the island (Seabrook 2004). Feral hogs have proven more difficult to eradicate, and are still present despite ongoing efforts of the park. Feral horses represent a unique issue for CUIS, as the species is not native to the island and represents a competitor to white-tailed deer for browsing forage. Horses have been known to trample native vegetation, wetlands, and bird nests, and has had dramatic impacts on the marsh habitats of the island due to over grazing (Turner 1986, Dolan 2002, Sabine et al. 2006). However, the species is also a substantial attraction for visitors of the island (Figure 46). The feral horse herd at CUIS is currently unmanaged and there is not a general horse management plan; however, the NPS does conduct an annual horse herd count when timing and volunteer scheduling allows. For a short time in the mid-1990s, a rider attached to a Congressional Bill made it illegal for the park to actively manage (i.e., remove) horses in the park (Dilsaver 2004). While the rider to the bill has expired, no horses in the park are removed, medically cared for, or provided with food/water.

Figure 46. Feral horses grazing on the grounds of the Dungeness ruins (NPS photo).

4.5.2. Measures

- Species richness
- Mesocarnivore species richness
- Deer population size

4.5.3. Reference Condition/Values

The ideal reference condition for mammals in CUIS would be a historic inventory of the park. The closest publication to this reference condition is Bangs (1898), which documented 11 mammal species in the park. Between the various studies that have taken place in CUIS over the past century, there exists a relatively solid record of when species became extirpated/established on the island. This component will compare species richness records from contemporary studies to historic records from Bangs (1898) and Hillestad et al. (1975).

A reference condition for deer population size does not currently exist, as deer populations in the park are not managed to a population estimate or specific number. Rather, the species is managed to sustain the population. The best professional judgement of NPS staff and experts will be used to assess the current condition of the deer population measure in this component.

4.5.4. Data and Methods

Bangs (1898) attempted to document all of the land mammals of Florida, and later included much of coastal Georgia in the surveys. As was typical of mammal inventories in the 1800s and early 1900s, Bangs (1898) physically collected most of the mammals, with only a few species being documented by sight alone. Bangs (1898) sampled in the St. Mary's, Georgia area from 9 March to 19 April 1896. Collections focused on Cumberland Island, as well as at Rose Bluff, which is on the Florida side of the St. Mary's River. Additional collections were completed at Cumberland Island from December 1896 to May 1897 by a cooperator.

From 1973-1975, Hillestad et al. (1975) inventoried and described the various terrestrial natural resource communities found on Cumberland Island. Using a combination of on-the-ground surveys, literature reviews, and museum specimens, Hillestad et al. (1975) documented all vertebrate fauna that were likely to occur on the island, and also defined, when possible, those species interactions with the island's many habitats.

Nelson et al. (1986) summarized the results of the managed white-tailed deer hunts at CUIS during the 1985-1986 season. Five controlled hunts took place on the island over the course of two weekends in November 1985, two weekends in December, and an additional weekend in January 1986. Hunting methods included archery, handguns, parent and child hunts where children could use high powered weapons, and primitive weapons (Nelson et al. 1986). Harvested deer were recorded by CUIS staff with notes being collected on the deer's age, sex, weight, kill location, date, and antler development. Effort was made to age several deer by either tooth eruption/wear or by jawbone analysis. Harvest data from the 1985-1986 hunt in CUIS were compared to harvest data from previous hunts on the island (1983-84, 1984-85) and to other barrier islands of Georgia.

In 1986, Ford (1987) established a spotlight survey technique for white-tailed deer on the island. Surveys took place in March and September of 1986, and consisted of a team of three NPS staff utilizing a pickup truck. The team would have one driver/data recorder, and two observers with spotlights in the bed of the pickup. Surveys began shortly after dark and followed a northern route and a southern route. Vehicles travelled slowly, normally around 5-8 km/h (3-5 mph), with observers sweeping their spotlights across the road looking for eye shine from deer or deer profiles within a 100 yard (300 ft) buffer from the road. When a deer was observed, the pickup truck would stop and observers would collect information on the deer. Characteristics recorded included: time, habitat, sex, age (i.e., fawn or adult), approximate location, and any other abnormal traits (e.g., piebald deer, injured, etc.). Bjork (1996a) repeated the methodology of Ford (1987) in order to assess the health and size of the white-tailed deer population in the park in 1994 and 1995.

Hayes et al. (1988) summarized and analyzed the results of the managed quota white-tailed deer hunts at CUIS in 1987 and a portion of 1988, and also included discussion from results obtained in CUIS during the 1985-86 and 1986-87 breeding seasons. The analysis included discussion of population structure, average weight, antler development, and overall hunter success.

Webster (2010) attempted to document mammalian species that were present in all 19 SECN parks. Webster (2010) had two major objectives:

1) To visit major North American museums in order to inspect/document mammal specimens that were either collected within or adjacent to park boundaries, and;
2) To visit each of the 19 SECN parks and conduct extensive field sampling in order to confirm/refute a species' presence and to document the overall mammalian faunal community in the park.

For the CUIS portion of the inventory, Webster (2010) utilized historic mammalian inventories (e.g., Bangs 1898) and checklists, and museum specimens to estimate what species were likely to inhabit the island. Museum collections from the American Museum of Natural History, Carnegie Museum of Natural History, Charleston Museum, Cornell University, Delaware Museum of Natural History, Field Museum of Natural History, Museum of Vertebrate Zoology, U.S. National Museum of Natural History, and to a limited extent, the Cumberland Island Museum (Webster 2010).

During the site visits, Webster (2010) utilized a variety of methodologies in order to document 90% of the expected species for a park (excluding bat species), and the methodologies varied depending on the sampling location in the park. In total, eight major habitats were selected for monitoring: the foredune complex, interdunal meadows, maritime thicket, maritime forest, salt grass meadow, mud flat, tidal creek, and salt marshes (Webster 2010). Each site was surveyed for evidence of mammals (e.g., scat, shed, tracks), and either Sherman livetraps or pitfall traps were deployed to capture and document mammals (Figure 47). According to Webster (2010, p. 5),

> *Pitfall traps consisted of unbaited #10 food cans buried such that their lips were flush at ground level. Sherman live traps were 7.6 cm X 7.6 cm X 23 cm (3 in X 3 in X 9 in) aluminum traps baited with rolled oats. One grid of pitfall traps and one grid of Sherman live traps typically*

were installed in each habitat. Each grid consisted of five rows of traps, each row containing five traps, spaced at 10-m (33-ft) intervals, thereby trapping an area of 0.5 acres (0.25 hectares).

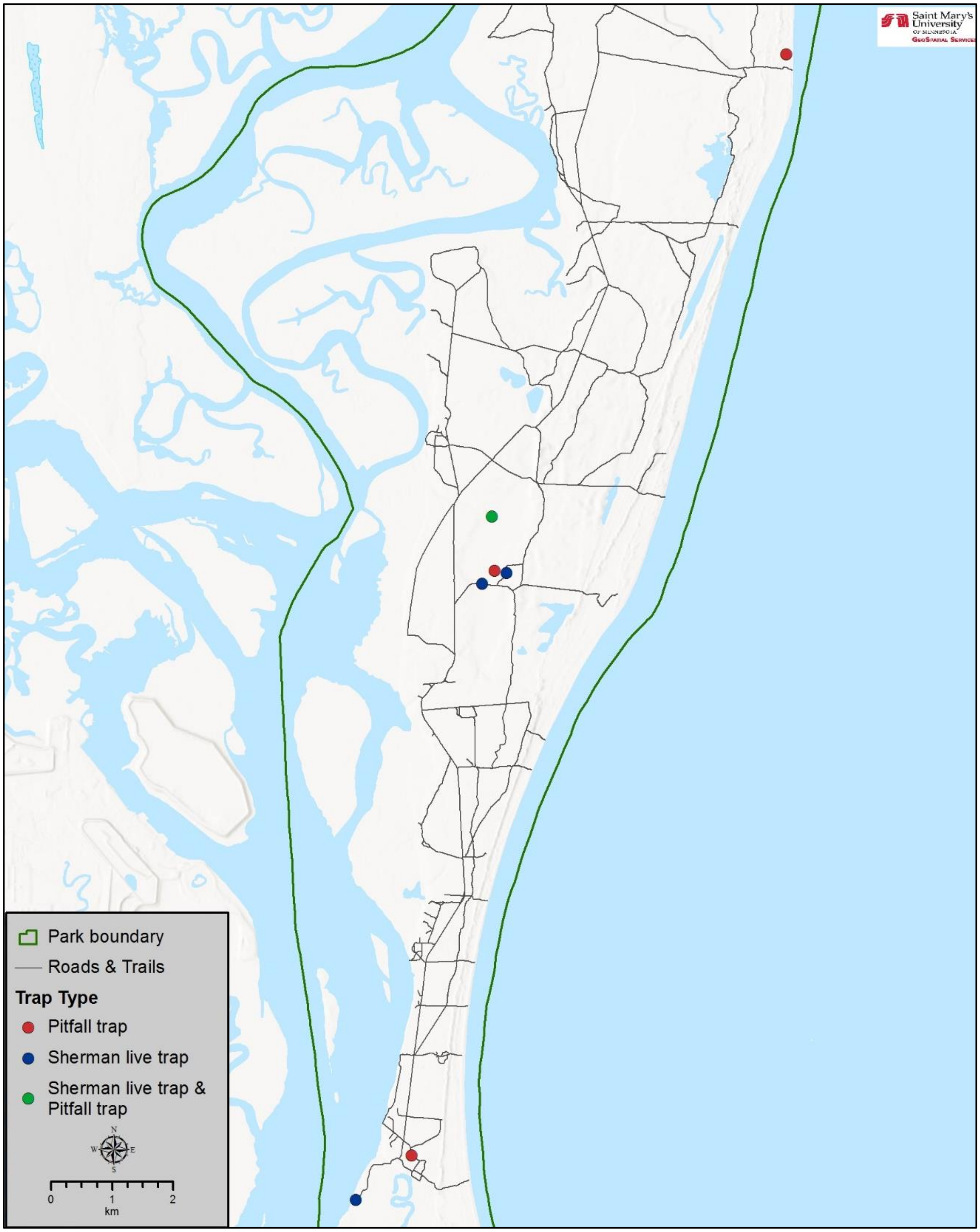


Figure 47. Mammal trap locations at CUIS used during the Webster (2010) mammalian inventory.

Castleberry and Morris (2017) summarized the results of bat inventories in the park during the 2011, 2015, and 2016 field seasons. Survey methodology varied by year and location in CUIS. Mist netting surveys were completed at North Beach Well Pond in 2011, and at Duck Pond in 2016, while mobile transects were completed in the park in 2015 and 2016. These surveys utilized an Anabat SD2 bat detector that was attached to the roof of a slowly moving (approximately <10 km/h) survey vehicle as it traversed transects shortly after sunset. Bat calls identified by the Anabat analysis software (Echoclass v. 3.1) were isolated to species. Additional bat surveys were completed in CUIS in 2016 as part of the North American Bat Monitoring Program (NABat). NABat monitors the long-term changes in bat populations and investigates the effectiveness of management strategies and actions aimed at bat species. NABat monitoring efforts utilize both stationary and mobile transects. Stationary surveys used Anabat detectors that were mounted to trees at preselected sites in the northern portion of CUIS, and collected bat echolocation sequences over four nights from 21-24 July 2016. Mobile NABat monitoring followed transects in the same manner as previous mobile surveys in the park (see above). The mobile survey followed park roads and covered nearly the entire length of the island, from North Cut Road to the Dungeness area. All observations that were able to be identified to the species level were reported in Castleberry and Morris (2017).

Since 2009, NPS staff has monitored the Dungeness area's white-tailed deer population using infrared-triggered trail cameras. The surveyed area covers roughly 5 km^2 (2 mi^2) of the park and was chosen due to the fact that this is an un-hunted herd far enough away from the public hunt zone to exhibit its own population dynamics (D. Hoffman, written communication, 2017). The methodology of the trail camera surveys followed that of Jacobson et al. (1997), and utilized four cameras stationed in four different locations across the park (Figure 48); cameras were set up in a way so that the sampled winter density was one camera per 100 acres. Camera surveys in CUIS took place during the winter (December-January) and lasted approximately 14 days (some longer, some shorter depending on camera conditions or other factors). Sites are pre-baited with feed for 4-6 days, have relatively cleared or low growing vegetation within a 3 m (10 ft) radius, and have cameras installed approximately 3.6-4.6 m (12-15 ft) from the center facing either north or south to improve image quality (Jacobson et al. 1997). The infrared cameras were set to record the date and time of the photo, and were on a 10-minute delay between photos. Photos were later observed on a computer, and the total number of doe photos, fawn photos, buck photos, and individual bucks were counted. These data were used to create population estimates including total deer, buck to doe ratios, and fawn to doe ratios. Complete methodology for ratio calculations can be found in Jacobson et al. (1997).

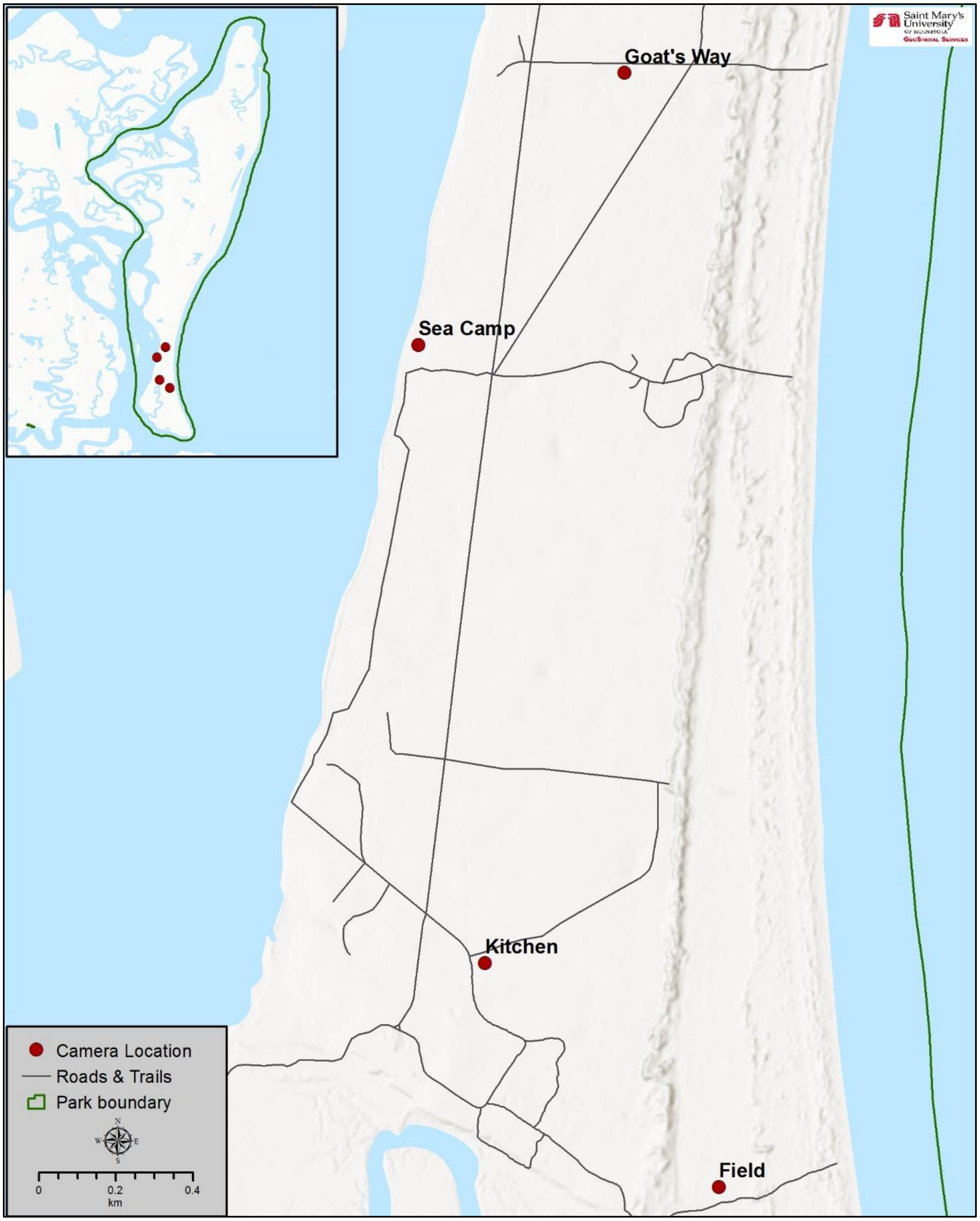


Figure 48. Locations of infrared-triggered trail cameras used for annual white-tailed deer winter surveys (NPS unpublished data).

A herd of white-tailed deer in the interdune area of CUIS. Note the two leucistic deer in the background (NPS Photo).

The NPS has established scent station transects at nine locations in the park in order to document the abundance of furbearer species (Figure 49). Eleven stations are set up along the nine transects and are monitored for four consecutive nights. Survey timing has ranged from mid-January to late-March since 2009. Scent is applied to a stick at each site in order to attract species, and tracks are identified by NPS staff when visiting each station. Stations are raked daily to provide a fresh soil bed to observe tracks from the next night's activity. Surveys were initiated in 2009, and have occurred annually during years with favorable track-making conditions.

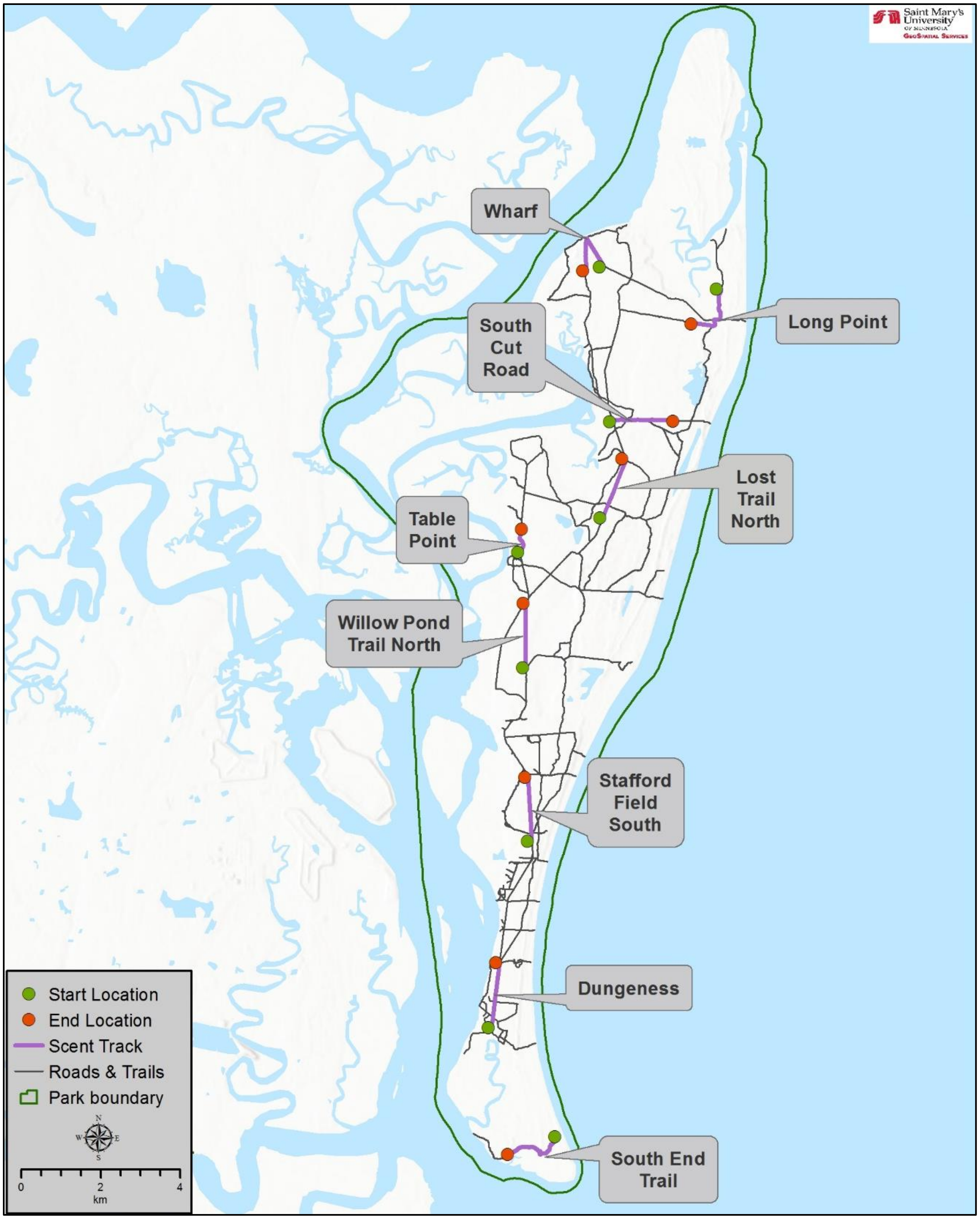


Figure 49. Scent station transects used to monitor furbearer species in CUIS from 2009-present (NPS unpublished data).

4.5.5. Current Condition and Trend

<u>Species Richness</u>

NPS (2016a)

The NPS Certified Mammals Species List (NPS 2016f) for CUIS identifies 30 terrestrial mammal species as either present, probably present, historically present, or unconfirmed inside the park. When excluding six species that occurred historically in the park, the total number of current confirmed terrestrial mammals in CUIS drops to 24 (Appendix G). Marine mammals included on the certified species list (NPS 2016a) include two confirmed species (bottlenose dolphin and West Indian manatee) and five species that were identified as probably present: dwarf sperm whale (*Kogia sima*), goose-beaked whale (*Ziphius cavirostris*), pygmy sperm whale (*Kogia breviceps*), rough-toothed dolphin (*Steno bredanensis*), and short-finned pilot whale (*Globicephala macrorhynchus*). This list, however, does not allow for a specific analysis of species composition over time, as no data are collected yearly, and the list only documents the presence (or historic presence) of identified species.

Bangs (1898)

Bangs (1898) represents the earliest comprehensive checklist of terrestrial mammals on Cumberland Island. In total, 11 mammal species were documented on the island during surveys in 1896 and 1897 (Appendix G). The Bangs (1898) checklist represents the last published document that confirmed the presence of the now-extirpated American black bear.

Hillestad et al. (1975)

During an island-wide ecological survey in 1973 and 1974, Hillestad et al. (1975) confirmed the presence of 21 terrestrial mammal species on Cumberland Island and Little Cumberland Island (Appendix G). An additional five species were identified as historically present on the island through the use of literature and museum specimen searches (Hillestad et al. 1975; Appendix G). The authors did not include the feral horses and feral cattle of the island on the observed mammal list. Two species (least shrew [*Cryptotis parva*] and eastern harvest mouse [*Reithrodontomys humulis*]) included on the confirmed mammal list by Hillestad et al. (1975) were included based on their presence in owl pellets; no living specimens were documented on the island. The nine-banded armadillo was documented for the first time on Cumberland Island in 1973, though it was not clear to the authors if the species was introduced deliberately or colonized the island naturally.

Hillestad et al. (1975) found no sign of several once abundant species, and indicated that the American black bear, Virginia opossum, common gray fox, and bobcat had been extirpated from the island in the time since the Bangs (1898) surveys. The only large native mammal present on Cumberland Island at the time of the Hillestad et al. (1975) survey was the white-tailed deer.

Webster (2010)

Seventeen terrestrial mammal species were recorded during the Webster (2010) inventory of CUIS (Appendix G). An additional pair of species, the black rat and feral cat, were identified as probably present in the park but were not directly observed during the inventory. Webster (2010) also suggested that approximately six species that historically occurred on the island had been extirpated from the island by the time of the survey. These species were: American black bear, American beaver (*Castor canadensis*), southeastern pocket gopher, eastern fox squirrel (*Sciurus niger*), African wild ass (*Equus africanus*), and feral cattle (*Bos taurus*). The eastern fox squirrel, African wild ass, and feral cattle were all introduced species that have since been removed from the island (Webster 2010). The authors noted that the transient nature of the black bear and the beaver may allow for their sporadic occurrence on the island, but their long-term persistence is not to be expected.

Two species documented by Webster (2010) represent relatively new species to the island: the nine-banded armadillo (first documented by Hillestad et al. [1975] in 1973) and the coyote (first documented in 2004). It is suspected that both species expanded naturally to the island, as the armadillo occurs regularly on the mainland and the coyote has been expanding its range eastward. Additionally, the Virginia opossum was documented during the survey efforts. This species was previously thought to be extirpated from the island (Hillestad et al. 1975). Webster (2010) notes that this species was reintroduced to the island in 1993 and has been thriving since its arrival.

A coyote observed in CUIS. The species naturally established on the island sometime in the early 2000s (NPS Photo).

Castleberry and Morris (2017)

Castleberry and Morris (2017) surveyed the bat population of CUIS in 2011, 2015, and 2016. Mist net captures in 2011 and 2016 identified six bat species, with five species observed in each year (Table 43). Two species observed during Castleberry and Morris (2017) are included on Georgia's Species of Concern List: tri-colored bat (*Perimyotis subflavus*) and northern yellow bat (*Lasiurus intermedius*).

Table 43. Bat species captured during mist netting efforts in CUIS in 2011 and 2016 (Castleberry and Morris 2017).

Species	2011	2016
big brown bat	X	–
northern yellow bat	X	X
evening bat	X	X
tri-colored bat	X	X
seminole bat	X	X
eastern red bat	–	X

Mobile transect surveys using the Anabat SD2 in CUIS resulted in the identification of two species in both 2015 and 2016 (Table 44); it was not possible to distinguish between acoustical observations of the eastern red bat (*Lasiurus borealis*) and the Seminole bat (*Lasiurus seminolus*) so the observations were grouped together. Additional unknown species were recorded and grouped without species classification; these species were unable to be identified to species due to either poor recordings or could not be isolated to a single species reliably.

Table 44. Species observed during mobile Anabat SD2 surveys in CUIS in 2015 and 2016 (Castleberry and Morris 2017).

Species	2015	2016
evening bat	X	X
tri-colored bat	X	X
eastern red bat/Seminole bat	X	X

NABat stationary surveys in CUIS in 2016 identified five bat species; similar to the Anabat surveys in 2015 and 2016, observations of the eastern red bat and Seminole bat could not be distinguished. Bats identified to species by the NABat mobile transects included: big brown bat (*Eptesicus fuscus*), silver-haired bat (*Lasionycteris noctivagans*), hoary bat (*Lasiurus cinereus*), evening bat (*Nycticeius humeralis*), and the tri-colored bat (Castleberry and Morris 2017). The most commonly detected specie(s) during the stationary surveys was the eastern red/Seminole bat.

Mobile NABat surveys in CUIS in 2016 identified three bat species: big brown bat, evening bat, and tri-colored bat. Acoustic observations of the eastern red bat and Seminole bat could not be identified to species. Additional unknown/unidentified bat species were also recorded during the mobile surveys.

NPS Scent Station Surveys (2009-2016)

Scent station surveys have identified eight species of mammals across the park since 2009, with tracks from an additional unidentified squirrel species also present. Tracks that could be identified to species included: bobcat, coyote, common raccoon, Virginia opossum, nine-banded armadillo, feral hog, feral horse, and white-tailed deer. Raccoons and opossums were the most ubiquitous of all mammal species, being observed at nearly every station in almost every year.

Marine Mammal Strandings

In conjunction with the Southeast U.S. Marine Mammal Stranding Network, the Georgia DNR has compiled a list of all marine mammal strandings in the state's coastal waters. Stranding records from the Cumberland Island area are up to date from 1996-2016. In the 21 years of stranding data, seven marine mammal species have washed up stranded on or near CUIS. These species are identified in Table 45.

Table 45. Marine mammal species that have been documented stranding on or very near CUIS since 1996 (GA DNR Unpublished Data).

Species	Years Reported
Atlantic spotted dolphin	2000, 2007, 2015
Atlantic white-sided dolphin	2011
bottlenose dolphin	1996-2016
dwarf sperm whale	2004, 2014
humpback whale	2000, 2011
pygmy sperm whale	1996-1998, 2001, 2003, 2006, 2008, 2010-2011, 2013, 2016
North Atlantic right whale	1996, 2005

Mesocarnivore Species Richness

The mesocarnivore species richness measure is similar to the previously discussed species richness measure in that it reports the total number of species observed by all mammalian surveys and inventories that have taken place in the park. The major difference between this measure and the previous one is that this measure excludes all non-mesocarnivore species. A mesocarnivore includes any species in the order Carnivora that is small or mid-sized (i.e., <15 kg [33 lbs]) (Roemer et al. 2009). Mesocarnivores typically have higher annual species richness numbers and more diverse behaviors and ecology than the larger carnivore species (Roemer et al. 2009). Because of the relatively small size of mesocarnivores, and the fact that they can survive and thrive in a variety of habitat types, mesocarnivores are typically more abundant in a given habitat type compared to other mammals. The size of the coyotes on the island is poorly documented, as they recently colonized the island in the early 2000s. For the purpose of this assessment, coyotes will be included in this measure's discussion as well. Coyotes typically have a body mass that exceeds that which would qualify as a mesocarnivoer. However, coyotes at CUIS have shown variable body mass measurements, as data taken from coyotes on the island from 2015-2017 showed mass estimates ranging from 12.7-14.5(28-32 lbs) (Hoffman, written communication, 2017), slightly below the 15 kg (33 lbs) threshold of mesocarnivores.

NPS (2016a)

NPS (2016a) identifies five mesocarnivore species that have been confirmed in the park (Table 46). An additional two species, the common gray fox and feral cat, were identified as historically present but extirpated from the park. Similar to the species richness measure discussed above, the NPS Certified Species List does not allow for a specific analysis of species composition over time, as no data are collected yearly.

Table 46. Mesocarnivore species documented on the NPS Certified Species List (NPS 2016a).

Common Name	Scientific Name
bobcat	*Lynx rufus*
common gray fox*	*Urocyon cinereoargenteus*
common raccoon	*Procyon lotor*
coyote	*Canis latrans*
feral cat	*Felis catus*
mink	*Mustela vison*
river otter	*Lontra canadensis*

* Historically present species

Bangs (1898)

Bangs (1898) documented a single mesocarnivore species, the common raccoon. There were likely other mesocarnivore species present on the island at the time of the Bangs (1898) surveys, but they failed to be detected by the various efforts.

Hillestad et al. (1975)

The Hillestad et al. (1975) inventory in CUIS identified four mesocarnivore species: bobcat, common raccoon, American mink (*Neovison vison*), and river otter (*Lontra canadensis*). At the time of the inventory, the bobcat population of the island was very small, just a few individuals. This was because a small reintroduction of the species had just taken place on the island in 1972 and 1973. The river otter and mink were described as common residents of the island's salt marsh creeks and freshwater habitats, with small fish and crustaceans making up the bulk of their diet. Hillestad et al. (1975) noted that the raccoon on the island was ubiquitous, and was the most common mammal of its size class. Additionally, Hillestad et al. (1975) called the raccoon one of the most ecologically important species on the island, primarily due to the species' predatory habits on beach nesting species, and their tolerance to disturbance.

Webster (2010)

Five mesocarnivore species were confirmed in CUIS during the Webster (2010) inventory, and an additional species (feral cat) was identified as probably present due to reports of the species in the park and nearby areas (Webster 2010). Species confirmed included the bobcat, common raccoon, coyote, American mink, and river otter. This inventory was the first such effort to document the presence of the coyote, which was first observed in the park in 2004. The bobcat population during Webster (2010) was described as thriving, and was hypothesized to continue to increase in size until the island's carrying capacity for the species is reached.

Webster (2010) briefly discussed the historic presence of the common gray fox in the park, and suggested that the original observation by Bent (1940) was perhaps a lone fox that had come over to the island from the mainland, or was an erroneous observation. No record for that species was found by any of the surveys summarized in this report.

Bobcat Reintroduction Efforts

Following the smaller reintroduction efforts in 1972 and 1973, the NPS reintroduced 32 bobcats to CUIS in 1988-89 (Diefenbach et al. 1993). The bobcats released were all adults, were captured in similar habitats to CUIS along coastal Georgia, and were fitted with radio collars to allow for population monitoring and to aid in recapture efforts. In 1988, 14 bobcats were released (11 females and three males), and by June of 1991 one female and one male had died, one female swam to the mainland, and one female's fate was unknown due to a failure in its radio transmitter (Diefenbach et al. 2006). Reintroductions in 1989 included 12 males and six females. One male died upon release, and by 1991 two additional males and a female had also died on the island.

Reproduction was first noted at CUIS in 1989, as 10 kittens from four litters were born on the island (Diefenbach et al. 2006). Population density was well-documented during the 1989 and 1990 breeding seasons as all adults were collared and their den locations were known. However, following the 1990 breeding season, density estimates could only be estimated as the population expanded and collars failed/adults died. Diefenbach (1992) completed a population viability analysis and determined that the CUIS bobcat population had a median time to extinction on the island of 65 years. This analysis also determined that bobcat abundance on the island would decline after reintroduction, and then average 12-13 bobcats. Shifts in abundance ranging from as low as five adults, to as many as 27 adults were predicted, depending on environmental conditions and reproductive success (Diefenbach et al. 2015).

The most recent population estimate for CUIS's bobcat population is from 2012, when Diefenbach et al. (2015) performed a genetic analysis on the population using bobcat scat collected on the island. Approximately 14 bobcats were identified using this technique, which was similar to the predicted population abundance (12-13 bobcats) of Diefenbach (1992). The 32 bobcats that were released likely represented a population size that was over the carrying capacity of Cumberland Island (Diefenbach 1992). With only a limited number of home ranges available for female bobcats to occupy, they likely had to compete with other females in their range, which may be responsible for the reduced reproductive rates (Diefenbach et al. 2006).

When the bobcat population was reintroduced to the island in 1988, previous scent station surveys had indicated that there were no other terrestrial predators on the island (Diefenbach et al. 1994). However, the establishment of a breeding coyote population on Cumberland Island in the early 2000s means that there are now two terrestrial predators in the park. Competition for food resources, as well as direct predation of coyotes on bobcats (Knick 1990), may result in a reduction in the bobcat population; however, the exact relationship between these two species on the island is yet to be understood.

Deer Population Size

CUIS hosts an annual managed white-tailed deer hunt each year that occurs only on lands in the park that are designated wilderness. As of 2017, three different hunt types occur across four different seasons (Table 47). Note that all of the hunts on the island through November, 2017, were cancelled due to the impacts of Hurricane Irma. During a hunt, hunters must register for each hunt on a first come first serve basis, and registration remains open until the hunt quota is filled. Hunting hours are from sunrise to sunset.

Table 47. 2017 white-tailed deer managed hunt types and seasons on CUIS (NPS 2017a).

Hunt Type	Dates[c]
Archery	16,17,18 October 2017
Adult/Child[a]	28, 29 October 2017
Primitive Weapons[b]	13,14,15 November 2017
Primitive Weapons[b]	4,5,6 December 2017

[a] Children aged 10-17 are accompanied by an adult on hunts. Children may use modern weapons and are the only hunters allowed to harvest.

[b] Archery equipment, muzzle loading firearms, and centerfire handguns

[c] 2017 hunts were cancelled due to the impacts of Hurricane Irma

The earliest mention of the white-tailed deer population on CUIS comes from Bangs (1898), when the population was described as a tremendous herd that was likely all the island could support. Hillestad et al. (1975) estimated the herd’s density as approximately one deer per 12 ha (30 ac) on the upland portion of CUIS. The estimated total population on Cumberland Island during the Hillestad et al. (1975) surveys was 500 deer, although the density and population size estimates come from the author’s educated estimate and not the result of a systematic population density/size effort. Additionally, the deer observed on the island were reported to have small body size compared to deer observed on the mainland. Other estimates of deer population size in CUIS in the past (unpublished reports from Georgia DNR in 1979, Nelson et al. 1986, Miller 1988) have suggested that the population in the park was at or above the island’s carrying capacity.

Nelson et al. (1986)

Nelson et al. (1986) summarized the results of controlled hunts in CUIS during the 1985-86 hunting season. Controlled hunts generally took place in the wilderness area of the park. While initially the report hoped to make significant comparisons to previous hunting seasons, the data from 1983-84 and 1984-85 proved to be incomplete. During the 1985-86 season, 91 deer were harvested in the park; hunters were limited to one buck and had a two deer limit for the season and had an overall success rate of 0.39 deer/hunter. Age structure analysis (obtained by aging harvested deer jaw bones) revealed a population that had high average age (>3.7 years) and low recruitment, traits indicative of a population that is either stable or declining (Nelson et al. 1986). Sex ratios of harvested deer were nearly identical (45 bucks:46 does), and eight of the 13 fawns killed during the 1985-86 season were male. Nelson et al. (1986) suggested that the apparent declines in body size, reproduction, and antler quality were perhaps indicative of a population that was at or above its carrying capacity, similar to what had been suggested by previous studies. Estimates of population size in 1979-80 indicated that the island supported an average of 2,400-4,000 deer, and Nelson et al. (1986) suggested that the population may likely have been at a similar size in 1985-86.

Hayes et al. (1988)

Data from the 1986-1987 controlled hunts in CUIS resulted in a hunter success ratio of 0.23 deer/hunter. The 1987-1988 controlled hunts in CUIS resulted in 134 deer being harvested by 346 hunters, indicating in a hunter success rate of 0.39 deer/hunter. Compared to 1986-87 hunts, the hunter success rate increased by 0.16 deer/hunter (Hayes et al. 1988). The sex ratio of the harvested deer was identical (1:1), although variations were present in age classes. Overall harvest data indicated that the deer herd in the park was improving in health and reproduction, and that general health was also notably improved. Deer weight remained low in 1987-1988.

Bjork (1996)

Spotlight surveys for white-tailed deer in CUIS in 1994 identified an average of 5.8 deer/night along the northern route, with total number of deer observed per night ranging from two to eight. The southern route had an average of 8.5 deer/night, and nightly observations ranged from eight to 11 (Table 48) (Bjork 1996a). Estimated density in 1994 was 8.8 deer/mi^2, and the observed fawn:doe ratio was 1:3.8 (Bjork 1996a). The total number of observed bucks in 1994 was eight (14%) (Table 48).

Table 48. Deer population parameters observed along the northern and southern survey routes in 1994 and 1995 (Bjork 1996a).

Year	North Route	South Route	Total Length of Island			
	Avg Deer/Night	Avg Deer/Night	Avg Deer/Night	Density (mi^2)	Fawn:Doe	% Bucks
1994	5.8	8.5	14.9	8.8	1:3.8	14%
1995	6.3	18.3	23	14.2	1:4.6	8.70%

In 1995, spotlight surveys resulted in more deer observations along both the northern and southern routes, but a reduced number of bucks compared to 1994 (Table 48). Fawn:doe ratio in 1995 (1:4.6) also indicated reduced reproductive success in 1995 when compared to 1994 (Table 48).

When comparing Bjork (1996a) results to results obtained by Ford (1987), which used a very similar methodology, estimated white-tailed deer abundance on the island appeared to be declining. In 1986, an average of 60.8 deer/night were observed during spring counts, and 67.2 deer/night were observed during fall counts (Ford 1987). Fawn mortality appeared to be increasing when comparing 1994-95 data to 1986, as fawn:doe ratios in 1986 ranged from approximately 1:2 in the spring, to 1:3.8 in the fall. Reasons for the decline in just under a decade were not fully understood, but Bjork (1996a) suggested that weather-related factors (e.g., heavy rainfall/standing water on island), or the reintroduction of bobcats to the island may potentially explain the size fluctuation.

NPS Trail Camera Dungeness Area Population Surveys (2009-2016)

Infrared-triggered trail cameras have been used to non-invasively monitor the deer population of the southern portion of CUIS since 2009. Estimates of population size for this part of the island vary depending on if calculations are based on one camera/100 ac or one camera/200 ac; currently, the NPS has deployed one camera/100 ac. The estimated total size of the southern CUIS deer population has ranged from 41 deer (2014) to 86 deer (2009), and has had an average estimated population size of 65 deer (32 deer per square mile). Using the one camera/200 ac estimate, annual estimated total population sizes have ranged from 53 (2014) to 123 (2009), and have averaged 88.5 deer, or 44 deer per square mile (Figure 50).

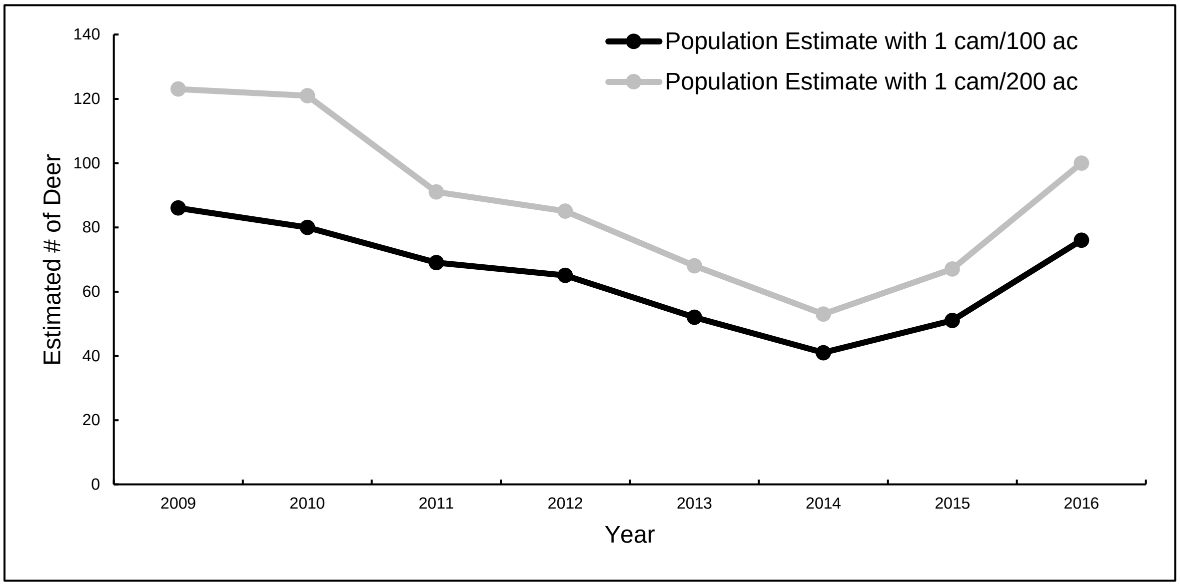


Figure 50. Estimated white-tailed deer population size in the southern, non-hunted portion of CUIS using survey data from infrared trail camera surveys (NPS unpublished data).

Recruitment in the southern portion of CUIS was variable but low between 2009 and 2016, with doe:fawn ratios ranging from 1:0.03 (0.03 fawns per doe) in 2013 to 1:0.6 in 2016. Six of the eight years surveyed had fewer than 0.5 fawns/doe (Table 49). Buck:doe ratios were also variable, with ratios near, or approaching 1:1. Peak buck:doe ratios were observed from 2009-2011, with all 3 years having ratios above 1:1 (Table 49). 2012 had the lowest observed buck:doe ratio, with just 0.4 bucks for every one doe.

Table 49. Estimated sex and recruitment ratios for white-tailed deer at CUIS between 2009 and 2016 (NPS unpublished data).

Year	Doe:Fawn	Bucks:Doe
2009	1:0.1	1.3:1
2010	1:0.55	1.2:1
2011	1:0.3	1.5:1
2012	1:0.2	0.4:1
2013	1:0.03	0.7:1
2014	1:0.05	1.2:1
2015	1:0.3	0.83:1
2016	1:0.6	0.65:1

NPS Controlled White-tailed Deer Hunt Results (2007-2017; NPS unpublished data).
The results of the annual controlled white-tailed deer hunting season have been summarized by NPS staff since 2007 (NPS unpublished data). Data summarized include the total number of deer harvested each year, as well as the overall percent hunter success (number of deer harvested/total number of hunters). The number of deer harvested in a season has ranged from 29 (2007) to 15 (2015), and the average number of deer harvested in the park was 23.4 (Table 50). The percent of hunters who are successful in a given hunting season has been variable, averaging 15.8% from 2007-2015. The highest percentage of successful hunters came in 2014, when hunter success was estimated at 20% (Table 50). Compared to hunting results from the mid-1980s (Nelson et al. 1986, Hayes et al. 1988), harvest numbers and hunter success have declined dramatically (see discussion above). The lower harvest estimates are likely due to the fact that the island's deer population has declined in size in response to the reintroduction of bobcats, and the establishment of coyotes on the island (Hoffman, written communication, 2017).

Table 50. Total deer harvest and hunter success estimates for annual controlled hunts in CUIS from 2007-2017 (NPS unpublished data).

Year	# of Deer Harvested	% Hunter Success
2007[a]	29	–
2008	26	13
2009	20	8.5
2010	27	17
2011[a]	22	–
2012	24	15.5
2013	22	17
2014	26	20
2015	15	11
2016[b]	–	–
2017[c]	–	–

[a] Hunter success estimates unavailable for this year

[b] Controlled hunt cancelled due to Hurricane Matthew

[c] Controlled hunt partially cancelled due to Hurricane Irma

Threats and Stressor Factors

NPS staff identified several threats and stressors to the mammal community of CUIS during project scoping. The major threat to the mammalian community as a whole is likely the non-native mammal populations present on the island (e.g., horses, hogs). Feral horses and hogs have been present on the island for hundreds of years, as both were documented during initial Spanish settlements and the American Civil War (Hillestad et al. 1975, Bullard 2003, Dilsaver 2004). Feral hog competition with native mammals represents a current threat; however, active management efforts have significantly reduced the size of the hog population on the island. Hogs may be harvested during the four managed public white-tailed deer hunts held each fall. Two public hunts are held specifically for hogs in

January of each year. Additionally, the NPS conducts hog management operations year-round in the form of hunting and trapping that are instrumental in maintaining the population at low levels. (Hoffman 2010, NPS 2017a).

The feral horse herd of the island has been more difficult to manage. The horses on the island represent a threat to many native communities (e.g., freshwater wetlands, salt marshes, shore-nesting birds), and are a direct competitor to the white-tailed deer population, as both species forage in many of the same areas. While the park has a good grasp of the horse population size on the island, management efforts are not currently underway, nor does a management plan exist for the species. Public and visitor sentiment, combined with political factors, have made the species nearly impossible to actively manage on the island (i.e., remove or reduce the herd size). Feral horses will continue to be a threat to the native mammals of the island as long as the population is left unmanaged.

With the newly established coyote population on the island also comes the threat of disease, as the species is known to be affected by canine parvovirus, canine distempter, canine infectious hepatitis virus, the plague bacterium (*Yersinia pestis*), and tularemia (*Francisella tularensis*). Coyotes may also carry rabies (Bekoff and Gese 2003) and a variety of ecto- (e.g., fleas, ticks, lice) and endo-parasites (e.g., flukes, tapeworms, heartworms), all of which may result in disease for the coyote (Bekoff and Gese 2003). Interspecies competition is also a threat, as coyotes on the island often displace raccoons (Hoffman, pers. communication, 2017). The recently established bobcat and coyote populations represent a relatively recent predation threat to the previously un-predated white-tailed deer population on the island.

The historic deer population of CUIS was overabundant, malnourished, and prone to outbreaks of disease, especially epizootic hemorrhagic disease. Herd health on the island generally improved as the density of the population decreased (Hayes et al. 1988; Hoffman, pers. communication, 2017). Years of drought or poor mast crops may still stress the health of the deer population on the island, but the threat of widespread outbreaks and die-offs is lower than it was 30 years ago. Continued monitoring of the population, and occasional analyses of herd health status will help managers better gauge the health and trend of the population.

Several bat species exist at CUIS throughout the year, including two species of concern in the State of Georgia (tri-colored bat, northern yellow bat). Recent threats to bat populations, particularly the sudden arrival of white-nose syndrome (WNS) and its rapid spread across the Eastern U.S., have been well publicized. Unfortunately, the disease is still poorly understood and continues to spread west across North America, reaching as far west as Washington State in 2016 (NPS 2016h). WNS has been confirmed in Georgia, but has not been documented at CUIS at this time. While the impacts of WNS are far-reaching, fluctuations in bat populations in the past decade have not been caused by WNS exclusively. Fluctuations in bat populations can be also be tied to climate change, changes in water quality, agricultural intensification, loss and fragmentation of forests, fatalities at wind turbines, disease, and pesticide use (Jones et al. 2009).

Data Needs/Gaps

Continuation of annual trail camera surveys will provide park managers with a data set capable of identifying long-term trends in the park's southern deer population. Similarly, scent station surveys should also be continued in order to document the presence of mammalian species across the island, and to estimate general population sizes for furbearing species. Trail camera surveys island-wide are an unfeasible task, as the number of cameras required to accurately survey the island-wide deer population would be too high. Instead, if managers desire an estimate of island wide population, a spotlight survey would be the most practical choice.

The marine mammal population of CUIS is poorly understood. Strandings occur frequently on the island, and are well documented, but the extent to which those mammals use CUIS's coastal waters is unknown. West-Indian manatees, a federally threatened species, frequent the island seasonally and forage around the coast. Bottlenose dolphins are also common in Cumberland Sound and on the Atlantic side of the island. Additionally, North Atlantic right whales (*Eubalaena glacialis*) use the coastal waters of Georgia for calving each winter and are likely present in the park's waters during these periods (Hoffman, written communication, 2017). An investigation into the marine mammal community of the park's waters would provide managers with valuable information regarding species composition.

The island's terrestrial mammal community has been well documented in the past 100 years, with all species expected to occur on the island having been observed. Semi-annual inventories, similar to what was done by Webster (2010) would continue to update the species richness and abundance estimates for the island and allow park managers to follow any potential trends or threats. Particular attention should be paid to the bobcat population on the island, as Diefenbach (1992) estimated a median time to extinction for the species on the island as 65 years.

With WNS present in Georgia, continuation of the annual bat surveys on the island will be important to document bat abundance, and to observe if the fungus makes an appearance on the island.

Overall Condition

Species Richness

The NRCA project team assigned the species richness measure a *Significance Level* of 3. The mammalian species composition of Cumberland Island has been well documented since the late 1800s (Bangs 1898), and the mammalian fauna of the park are about what should be expected for an island the size and location of Cumberland Island. The colonization of the island by new species has been recorded by various studies through the years, as approximate dates of colonization are available for the recently established bobcat (1988-89) and coyote (post-2003) populations. The work of Webster (2010) combined with the scent station data from the NPS appear to indicate that many of the mammals expected to occur on the island are still present. One area of uncertainty is the community structure of the park's marine mammals; only two species have been confirmed in the park, with several others expected to occur.

At this time, the species richness of CUIS's mammal population does not appear to be of concern. Because of this, a *Condition Level* of 0 was assigned to this measure.

Mesocarnivore Species Richness

The mesocarnivore species richness measure was assigned a *Significance Level* of 3 at project scoping. There is some debate over how many mesocarnivore species were on the island historically, as Bent (1940) documented a common gray fox. This remains the only documentation of that species on the island. The bobcat was formerly extirpated from the island in the early 1900s due to an outbreak of disease. This species has since been reestablished on the island through a successful reintroduction program in 1988-89, but continues to have a relatively small population size.

The coyote, a species usually larger than a typical mesocarnivore, recently migrated to the island and established a breeding population. The effect of this colonization is yet to be fully understood, and it may be possible that the species will outcompete the bobcats or displace the island's raccoons. Partially due to the uncertainty of this relationship, the mesocarnivore species richness measure was assigned a *Condition Level* of 1, indicating low concern.

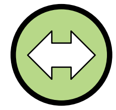
Deer Population Size

CUIS's deer population size measure was assigned a *Significance Level* of 2. According to CUIS Biologist Doug Hoffman (pers. communication, 2017), the CUIS deer herd is in excellent health due to low population density and minimal competition for resources. The ongoing feral hog removal program has significantly reduced the number of hogs on the island, which represent major resource competitors for the herd. Deer density on the island 20 years ago was likely much higher than it is currently, due in large part to the reintroduction of bobcats in 1988-1989 and the establishment of a coyote population in the last 15 years. The southern end of the island likely has a higher density than the northern portion, probably because the northern portion of the island experiences annual hunts that remove many deer from the island. Because the current population size is at a much healthier and sustainable level compared to the population 20-30 years ago, and due to the ongoing feral hog and coyote management efforts on the island, the deer population size measure was assigned a *Condition Level* of 0, indicating no concern at this time.

Weighted Condition Score

The *Weighted Condition Score* for mammals in CUIS was 0.13, indicating good current condition. A stable trend arrow was assigned since the mammalian community in the park is unlikely to experience significant changes in the future, as the species richness values for all species and the island's mesocarnivores are similar to what was historically present. The deer population size has improved in the past 30 years, but at present appears stable and is much healthier than the historically malnourished and disease-prone herd. A high confidence border was applied to this graphic due to the numerous historic studies at the park, and because the park's resource managers have a solid understanding of all measures at present (Table 51).

Table 51. Weighted Condition Score of Mammals in CUIS.

Mammals			
Measures	**Significance Level**	**Condition Level**	**WCS = 0.13**
Species Richness	3	0	
Mesocarnivore Species Richness	3	1	
Deer Population Size	2	0	

4.5.6. Sources of Expertise

John Fry, CUIS Chief of Resource Management

Doug Hoffman, CUIS Biologist

4.6. Birds

4.6.1. Description

Bird populations often serve as excellent indicators of an ecosystem's health (Morrison 1986, Hutto 1998, NABCI 2009). Birds are typically highly visible components of ecosystems, and bird communities often reflect the abundance and distribution of other organisms with which they co-exist (Blakesley et al. 2010). Resident birds provide insight into the current status of the habitats they frequent, while migratory birds serve as excellent ecological indictors because a disturbance adversely affecting any of the habitats used by these species (e.g., stopover, wintering, or breeding habitats) can cause declines in populations and a decrease in species' reproductive success (Hilty and Merenlender 2000, Zöckler 2005).

The unique ecosystems and physical formations of the Atlantic and Cumberland River/Sound coasts in CUIS provide bird species with ideal nesting, stopover, and overwintering habitat. Although heavily impacted by past human activity, the freshwater wetlands of CUIS are highly utilized by many bird species (Dlugolecki 2012). In fact, several avian species of conservation concern, such as the piping plover and wood stork, utilize a variety of habitats in CUIS throughout the year. Accordingly, the National Audubon Society also designated CUIS as an Important Bird Area (NAS 2017).

CUIS has confirmed the presence of more than 300 species of birds, many of which are migratory species (NPS 2016f). Additionally, CUIS has confirmed the presence of 17 bird species that are either federally listed as threatened or endangered, or state listed as rare, threatened, or endangered (Table 52).

Table 52. State and federally listed bird species that have been documented in CUIS (USFWS 2015, AAS 2017).

Species	GA Status[a]	Federal Status
American kestrel	R	–
American oystercatcher	R	–
Bachman's sparrow	R	–
bald eagle	R	–
black skimmer	R	–
golden-winged warbler	E	–
gull-billed tern	T	–
Henslow's sparrow	R	–
Kirtland's warbler	E	–
least tern	R	–
peregrine falcon	R	–
piping plover	T	T[b]
red knot	R	–

[a] R = Rare species; T = Threatened species; E = Endangered species

[b] Endangered in the Great Lakes region of the U.S.

Table 52 (continued). State and federally listed bird species that have been documented in CUIS (USFWS 2015, AAS 2017).

Species	GA Status[a]	Federal Status
red-cockaded woodpecker	E	E
swallow-tailed kite	R	–
Wilson's plover	T	–
wood stork	E	T

[a] R = Rare species; T = Threatened species; E = Endangered species

[b] Endangered in the Great Lakes region of the U.S.

CUIS is located along the Atlantic Flyway, one of the major migration flyways in North America (Figure 51), and many species, such as the red knot (*Calidris canutus*), pass through the park on their way from wintering grounds in the south to breeding grounds in the north.

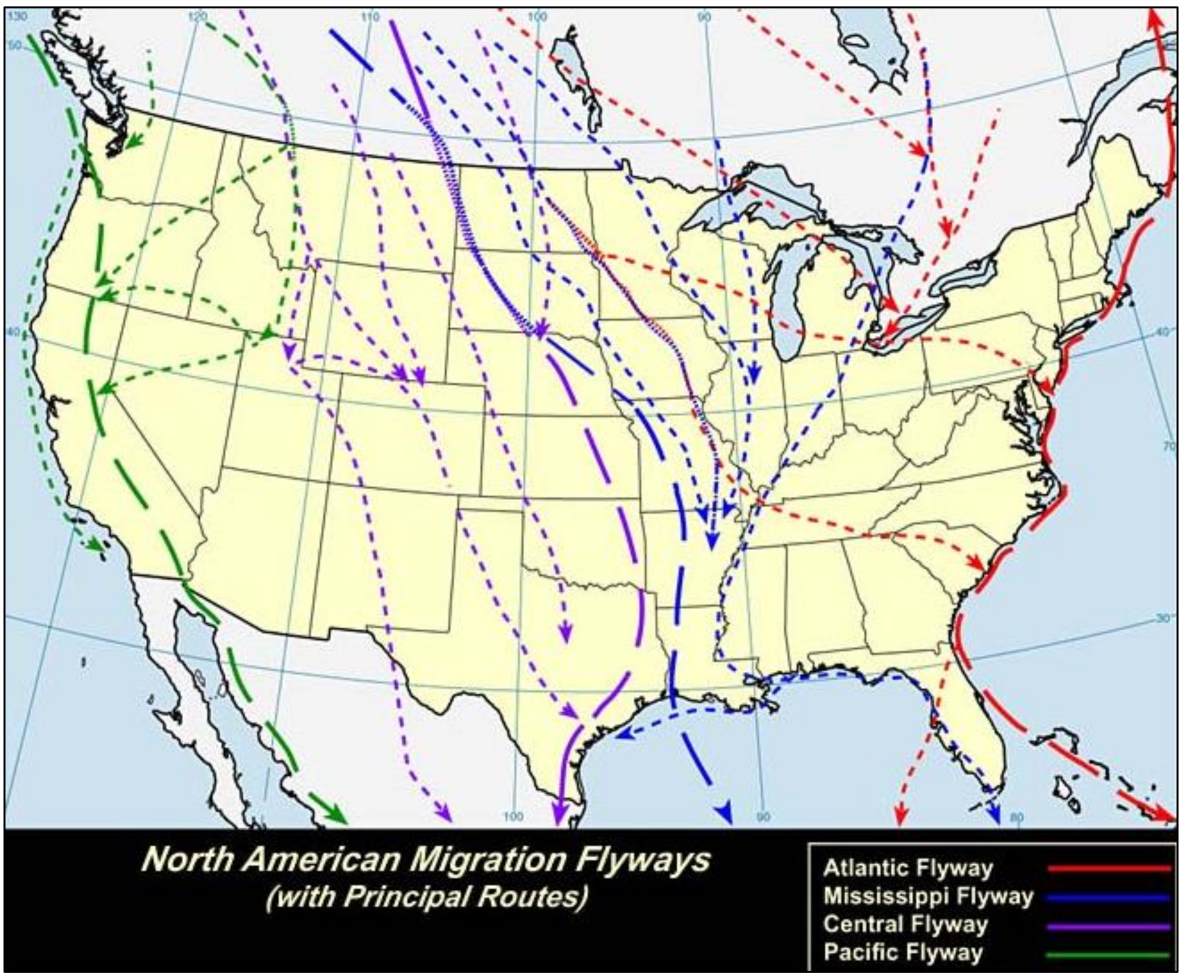


Figure 51. Major North American migratory flyways. CUIS is located along the Atlantic Flyway (NPS 2016g).

The park also acts as an important over-wintering area for several migratory species, such as the long-billed curlew (*Numenius americanus*) and the black-bellied plover (*Pluvialis squatarola*), as these species spend the winter months along CUIS's coastlines before returning to their breeding grounds in the spring.

Another species that is highly dependent on the park's coastlines for wintering habitat is the piping plover. The piping plover is a federally threatened species along the Atlantic coast, and research has shown that many of the birds that overwinter in the CUIS area are from the Great Lakes area where the species is considered endangered (Gibson et al. 2016). The Atlantic coast of the U.S. typically provides refuge for plovers as they escape cold winter conditions in the Midwest and Great Plains. However, cold snaps in the Southeast can have devastating effects on the wintering birds (Gibson et al. 2016). The piping plover has been highly studied in the park in the past decade, with various independent researchers conducting surveys throughout the year that monitor survivability and abundance (Pat and Doris Leary unpublished data, Gibson et al. 2016).

Long-distance migratory species are highly informative indicator species, as their overall health depends on several different ecosystems. Global Christmas Bird Count (CBC) and Breeding Bird Survey data indicate significant declines in migratory bird numbers in recent years (Peterjohn and Sauer 1999, Vickery and Herkert 2001, Niven et al. 2009). The red knot is one of the park's longest distance migrants, as it migrates south to the park from its circumpolar breeding habitats. Monitoring of long distance migratory species populations (such as the red knot) as they pass through or overwinter in CUIS may help managers to develop a better understanding of the overall health of not only the CUIS ecosystem, but also the other ecosystems that these bird species rely on.

4.6.2. Measures

- Species richness
- Shorebird nesting numbers
- Shorebird fledging success
- Wading bird nesting numbers
- Wading bird fledging success

4.6.3. Reference Condition/Values

A reference condition was not assigned to this component during project scoping. Historic bird lists exist from early visits to the Carnegie properties on the island (Pearson 1922, Sprunt 1936) that can be used as coarse snapshots of the bird population on the island near the beginning of the 20th Century. However, these species lists 1) were not the result of a rigorous bird survey and did not have specified methodologies, and 2) are useful only for the species richness measure of this component. These lists will be compared in the species richness measure; for all other measures in this assessment the best professional judgement of identified subject matter experts and NPS staff will be used to assess current condition. Future assessments of condition may be able to utilize this summary as a baseline for comparison.

4.6.4. Data and Methods

The NPS Certified Bird Species List (NPS 2016f) for CUIS was used for this assessment, as this list represents all of the confirmed bird species present in the park. The list was populated by the various bird inventories and surveys that occurred in the park's area, and in the case of parks with limited bird work, will likely resemble the overall species list of the primary bird inventory effort for the park.

In the summer of 1921, T. Gilbert Pearson (one of the founding members of the National Audubon Society) arrived at Cumberland Island as a guest of Andrew Carnegie II (son of Thomas Carnegie). While on the island, Pearson had access to all of the Carnegie's automobiles, boats, and guides, and was accompanied by Andrew on his journeys across the island. Pearson documented all of the bird species that could be accurately identified (Pearson 1922). SMUMN GSS adjusted the common and Latin names of some of these species to accurately reflect current taxonomic standards.

Similar to Pearson (1922), Sprunt (1936) reports the results of Alexander Sprunt Jr.'s visit to Cumberland Island as a guest of the Carnegie family. Sprunt (1936) documents all bird species observed and captured during two visits, the first from 13-21 April 1932, and the second from 7-15 April 1933.

Bratton et al. (1989) surveyed wood storks and least terns (*Sternula antillarum*) on CUIS during the summer of 1988. As this component does not have a measure that includes the least tern (i.e., least terns are not wading or shorebirds), only the wood stork surveys from Bratton et al. (1989) will be utilized. Wood stork surveys were completed by using a series of flights following a standardized route over the island. The first series of flights consisted of 18 2-hour surveys during July, August, and September 1988, with the primary objective of finding major concentrations of wood storks and their forage areas on the island. The second series of wood stork survey flights were completed between 20 September and 30 September 1988. This second round of surveys involved 13 flights, each lasting near 2 hours, and used standardized data sheets to record a suite of parameters for each wood stork or group of storks encountered (Bratton et al. 1989).

Every winter, the Georgia Department of Natural Resources (DNR) conducts mid-winter beach waterbird surveys (MWBS). The objective of these surveys is to document the wintertime distribution of waterbirds on the barrier islands of Georgia's Atlantic coast. At approximately the same time of day (generally coinciding with high tide) across Georgia's barrier islands, observers scan the beaches, mud and sand flats, and near shore waters for seabirds, wading birds, and shore birds (GA DNR Unpublished memo). Observers record the number of species and individuals observed during the survey, and also record any color bands seen on birds, with particular emphasis given to piping plovers, red knots, marbled godwits (*Limosa fedoa*), and American oystercatchers (*Haematopus palliatus*). Data are available for MWBS on Cumberland Island from 1998-1999, and 2003-2017 (NPS unpublished data).

From 10-24 May 1999, Plauny (2000) surveyed CUIS's beaches for nesting American oystercatchers, Wilson's plovers, and least terns. The study area was divided into three segments: 1) the North Segment, stretching from Long Point to Duck House Trail; 2) the Middle Segment, running between Duck House Trail and Dungeness Beach; and 3) the South Segment, which ran between Dungeness Beach and the mouth of Beach Creek to the west of the jetty (Figure 52). Colonies were defined as areas where three or more nests or birds exhibiting nesting/defensive behavior were observed. Identified nests and colony sites were marked slightly away from the actual site in order to aid in relocating the nest in future visits. Marked nests were re-visited every 3-4 days in order to check for the number of eggs, depredated eggs, or hatched chicks (Plauny 2000). When possible, observers would also document the reproductive success of the nesting shorebirds and the potential cause of any nest failure or depredated eggs. Incidents of potential human disturbance, often due to recreational use of the beach, were also recorded.

Sabine et al. (2006) investigated the reproductive success of American oystercatchers in CUIS during the 2003 and 2004 breeding seasons (March-August). Researchers conducted daily foot and vehicle surveys along the beaches of CUIS in an effort to document all American oystercatcher breeding pairs and nests. When a nest or breeding pair was located, the GPS coordinates of the nest/pair was documented and the nest was marked with a fluorescent marker placed nearby. Sabine et al. (2006) also utilized video monitoring equipment at each nest. When the nest was identified and marked, a black-and-white infrared camera and a time-lapse recorder were placed near the nest, but still far enough away to avoid disturbance. Observers used the video equipment to help determine the cause of nest failure. Sabine et al. (2006) recorded hatching and fledging success for each nest, and attempted to determine the cause of nest failure in instances of failed nesting attempts.

Landbirds were identified as a high-ranking Vital Sign by the SECN during the Vital Sign selection process (DeVivo et al. 2008). Consequently, the SECN began a landbird monitoring program in all network parks, with the specific objective of determining trends in landbird species occupancy, distribution, diversity, and community composition in network parks (Byrne et al. 2011). Monitoring began in CUIS in 2010, and followed the sampling protocol published by Byrne et al. (2014). Thirty sites were chosen (Figure 53) using a random, spatially balanced algorithm, and each site was surveyed using a variation of the variable-circular plot (VCP) technique from April to June. Using this technique, observers were stationed at the center point of a 0.5 ha (1.2 ac) macroplot and recorded all species observed and heard during a 12-minute window. Birds that flew over the macroplot during sampling were recorded as flyover species. When possible, observers documented the time frame that the bird was recorded (e.g., 0-3 minutes after start, 3-6 minutes, etc.), and the distance of the observation from the observer. Distance was recorded in one of four intervals: 0-25 m (0-82 ft), 25-50 m (82-164 ft), 50-100 m (164-328 ft), and >100 m (328 ft).

The 2012 sampling methodology varied from that which was used in 2010, and instead of field observers conducting surveys, automatic recording devices (ARDs) were utilized. These devices were programmed to record audio for 12 minutes, twice a day (07:30-07:42 and 08:00-08:12) every 5 days over a total sampling period of 72 days. The devices were deployed at 30 locations across the island, and the recordings were manually evaluated in a lab. Complete methodology descriptions are provided in Kurimo-Beechuk and Byrne (2016).

SECN landbird monitoring in CUIS was summarized for the 2010 (Byrne et al. 2011) and 2012 (Kurimo-Beechuk and Byrne 2016) seasons. Monitoring efforts following the 2012 season have yet to be published, but repeat monitoring visits to CUIS are planned at approximate 3-year intervals (Byrne et al. 2014).

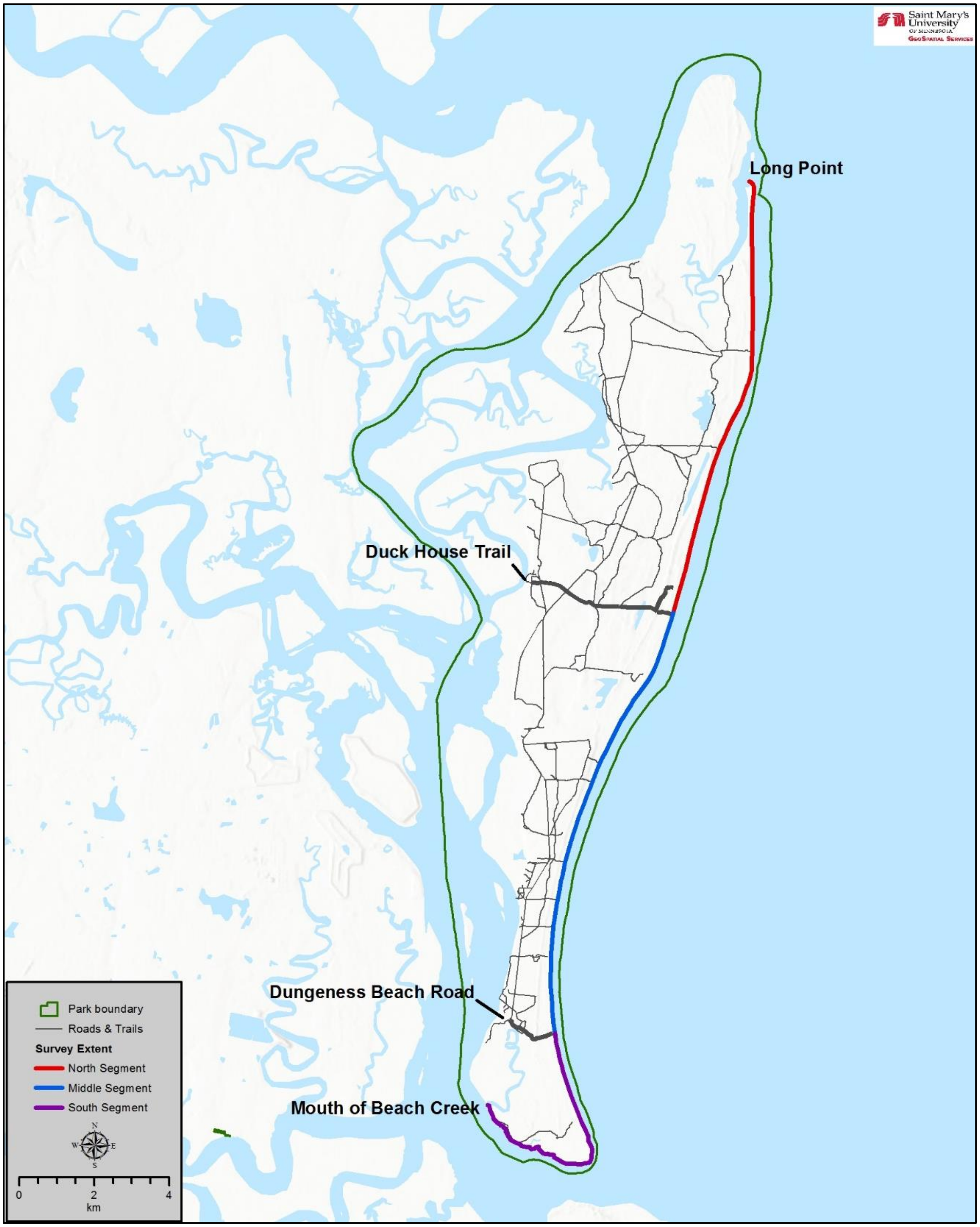


Figure 52. Sampling segments used by Plauny (2000) along CUIS's Atlantic coast. Segments focused on areas of the beach between the foredunes and the high tide line (Plauny 2000).

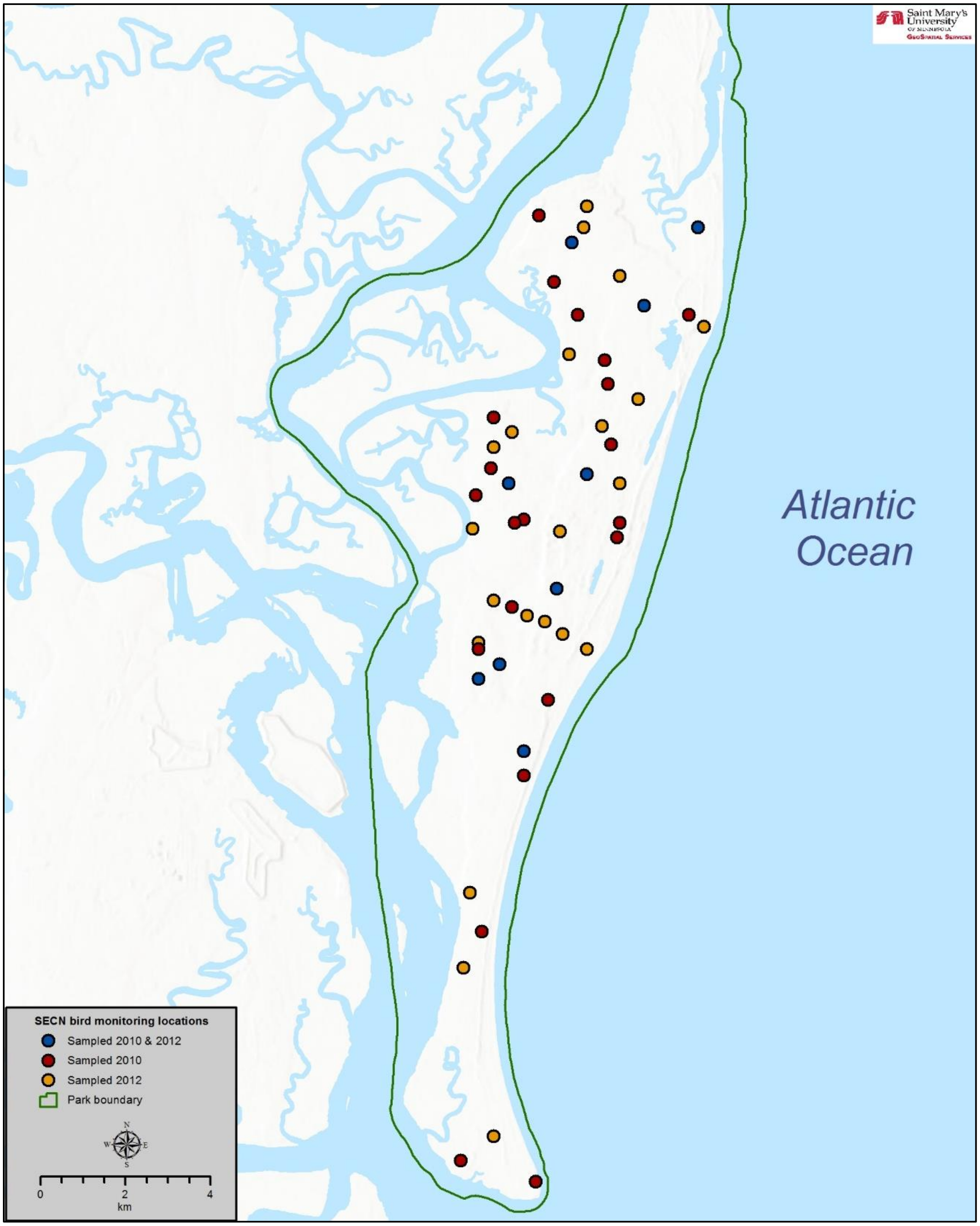


Figure 53. Landbird monitoring locations selected by the SECN for monitoring in the 2010 and 2012 sampling seasons. Sites that were utilized both years are displayed in blue (Byrne et al. 2011, Kurimo-Beechuk and Byrne 2016).

Dlugolecki (2012) investigated the bird use of freshwater wetlands in CUIS by conducting monthly surveys in the park from December 2010 to November 2011. Surveys were broken up into eight total points scattered across the freshwater wetlands of the island (Figure 54). The majority of the survey points were located near or along trails due to the dense vegetation surrounding the targeted wetlands. Sites were visited once a month, with observations typically occurring on the weekend near the 15th of each month. All species that were detected visually or that responded to broadcast calls were documented (Dlugolecki 2012).

During the breeding season, Dlugolecki (2012) followed the methodology and guidance of Conway (2009). Wetland observations began up to 2 hours before sunset and were conducted until dusk. The observer documented all species visually observed, and then conducted audio playback of the observed species' calls for 30 seconds followed by 30 seconds of silence. Total duration of the broadcast surveys were approximately 18 minutes, including the silent period. Surveys were repeated three times approximately 15 days apart during the breeding season (15-17 April, 29 April–1 May, and 14-16 May 2011).

Strickland (2015) observed the predators and anti-predator behavior of Wilson's plovers in CUIS from March to August in 2014 and 2015. All observations focused on 7 km (4 mi) of beach along the southern portion of the island. The majority of the surveys were completed during the breeding season to identify nesting pairs. Observers travelled the beach by foot and vehicle in order to document breeding pairs and nesting sites. When a plover track was observed in the sand, observers would follow the tracks until the nest was located. The number of nesting pairs, nests, eggs, and the approximate clutch initiation date were all recorded in the field. Predation and anti-predator behavior responses were also monitored, but are not relevant to the measures identified in this component.

An annual CBC is centered on Cumberland Island near the Stafford Plantation (Figure 55) and has been completed annually since 1992. The Cumberland Island CBC is part of the International CBC, which started in 1900 and is coordinated by the Audubon Society. Multiple volunteers surveyed a 24-km (15-mi) diameter area on one day, typically between 14 December and 5 January, by foot, boat, or car. The center point of the 24-km (15-mi) diameter was 30.815108ºN, -81.467261ºW (Figure 55). Unlike surveys that occur during the breeding season (such as a breeding bird survey), the CBC surveys overwintering and resident birds that are not territorial and singing. The total number of species and individuals were recorded each year

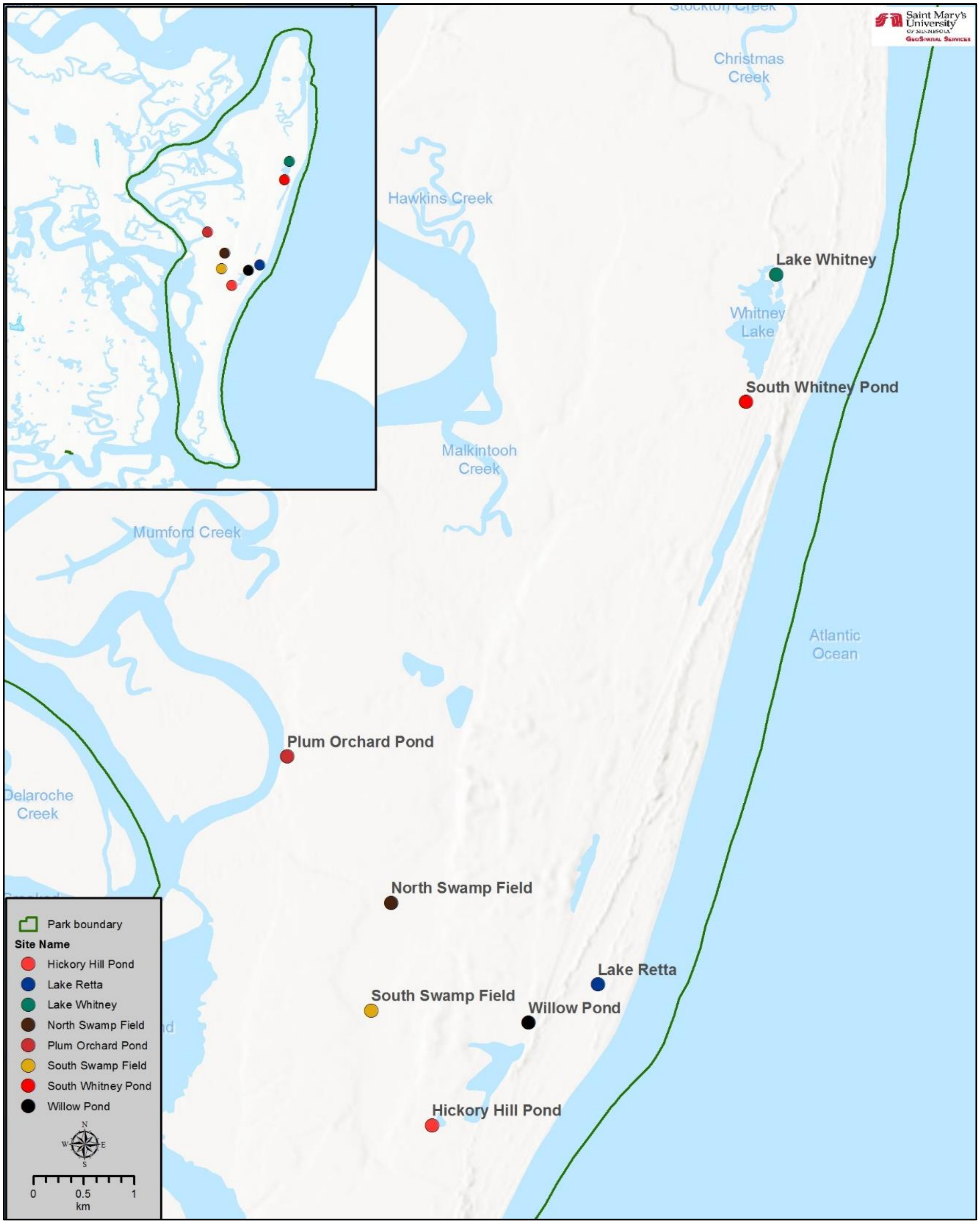


Figure 54. Wetland survey points monitored by Dlugolecki (2012) from December 2010 to November 2011.

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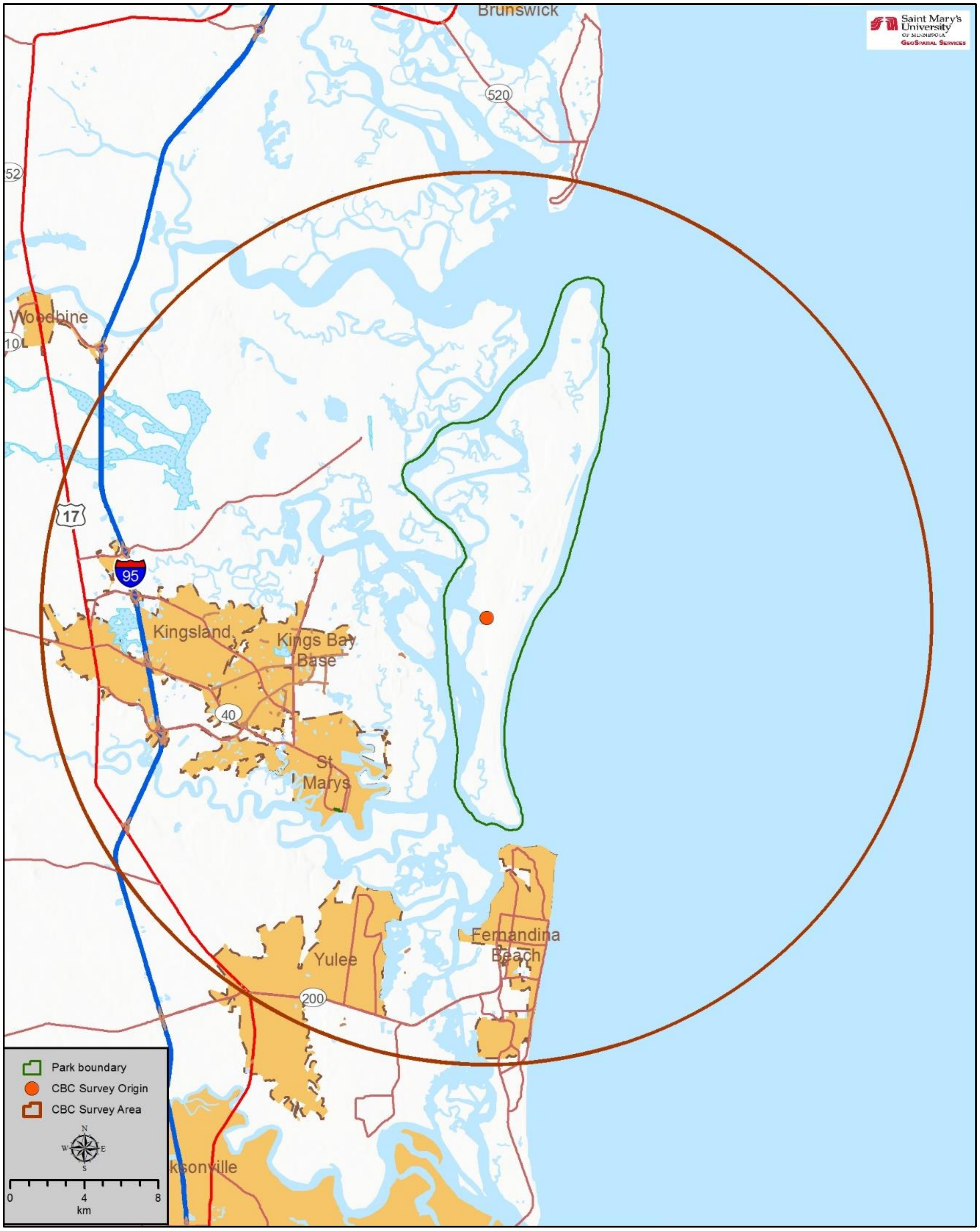


Figure 55. Christmas bird count survey area for the Cumberland Island CBC. The Cumberland Island CBC has been conducted annually since 1992.

Data from the Cumberland Island CBC were obtained from: http://netapp.audubon.org/CBCObservation/Historical/ResultsByCount.aspx and the following edits were made to the dataset: required SMUMN GSS to make some adjustments:

- All incomplete species identifications were omitted (e.g, *Buteo* spp., *Vireo* spp., American black duck/mottled duck);
- Observations of American green-winged teal and green-winged teal were treated as one category as both refer to *Anas crecca*.
- Aggregated observations of species artificially separated by color variations (e.g., white and blue form of *Ardea herodias*) were not treated as unique species observations.

4.6.5. Current Condition and Trend

Note, for the purpose of this assessment, the definition of shorebird and wading bird will largely follow the classification scheme followed by the MWBS. The term shorebird will include species groups such as stilts, avocets, oystercatchers, and sandpipers (e.g., turnstones, curlews, godwits, dowitchers, snipes, woodcocks, and plovers). Wading birds will include herons, egrets, storks, ibises, and spoonbills. Other commonly observed water bird species that are not specifically highlighted in the measures below include the seabirds (gulls, terns, and allies) and waterfowl (ducks, loons, grebes).

Species Richness

Each of the many bird surveys that have occurred in CUIS have focused on a unique guild of birds (e.g., shorebirds, wetland species, landbirds). Because of this, it is difficult to compare the results of the studies to each other. The differing methodologies, focal species, and timing make trends and patterns observed in each study difficult to compare and the results are best analyzed individually. This assessment presents the results of each study, but does not compare the species richness values between any studies.

NPS Certified Species List (NPS 2016f)

The NPS Certified Bird Species List contains 307 species that are confirmed in the park (Appendix H). This list also identifies species that may be present in the area but have not been confirmed within the park's boundaries. These species were identified as "Probably Present" by NPS (2016a), and included species such as the greater shearwater (*Puffinus gravis*), sooty tern (*Sterna fuscata*), and Wilson's storm petrel (*Oceanites oceanicus*). Other designations included in NPS (2016a) include "Unconfirmed", which is indicative of a species that has been attributed to the park, but little or no evidence to support its presence exists, and "Historic", which indicates species that historically occurred in the park but have since been extirpated from the area.

Unlike annual bird surveys, NPS (2016a) is not well suited for an analysis of annual species richness, as no data are collected yearly. The NPS Certified Species List documents the presence (or historic presence) of the identified species and serves as a useful point of comparison to determine which species have been documented in the park.

Pearson (1922) and Sprunt (1936)

During a visit to Cumberland Island in 1921, Pearson (1922) documented 97 bird species. This survey serves as the only documentation for several species on the island, including the barn swallow (*Hirundo rustica*), blackpoll warbler (*Setophaga fusca*), bobolink (*Dolichonyx oryzivorus*), least tern, magnolia warbler (*Setophaga magnolia*), and solitary sandpiper (*Tringa solitaria*) (Appendix H).

Sprunt (1936) visited Cumberland Island twice, once in 1932 and once in 1933, and documented 147 species (Appendix H). Similar to Pearson (1922), the historic work of Sprunt (1936) serves as the only record for several species on the island. Species documented by Sprunt (1936) that have yet to be documented by subsequent surveys include the bank swallow (*Riparia riparia*), Louisiana waterthrush (*Parkesia motacilla*), orchard oriole (*Icterus spurius*), red-cockaded woodpecker (*Picoides borealis*), and the tundra swan (*Cygnus columbianus*).

Between Pearson (1922) and Sprunt (1936), a total of 163 species were documented. These represent the earliest species records for the park prior to transfer of land ownership from the Carnegies to the NPS.

Georgia DNR Mid-winter Beach Waterbird Survey (1998-2017)

The Georgia MWBS has documented shorebirds on the beaches of CUIS since 1998. This study is focused specifically on shorebirds, and because of this it does not capture many avian species outside of this guild. Select species of seabirds and waterfowl are also included on many data summaries each year, and while these records are summarized in this report, it needs to be noted that not all years include records of these non-shorebird species. Estimates of richness and abundance for these non-shorebird species should be interpreted with caution. Additionally, several birds that were observed were not identified to the species level, primarily due to the similarity between species of shorebird species (e.g., “Peeps” group of sandpipers) and due to the distance that the species were observed (e.g., unidentified scoters or gulls). These generalized observation classifications may have resulted in a species richness estimate that was lower than what was actually observed.

From 1998-2017, 63 species of shorebirds, seabirds, and waterfowl were observed along CUIS’s beaches. Total species richness estimates ranged from 15 (2009) to 40 (2006), with an average total species richness of 31.4 species per year (Figure 56). The low species richness count from 2009 was likely due to the fact that the majority of non-shorebird species were not included in the data summary for that season (NPS unpublished data).

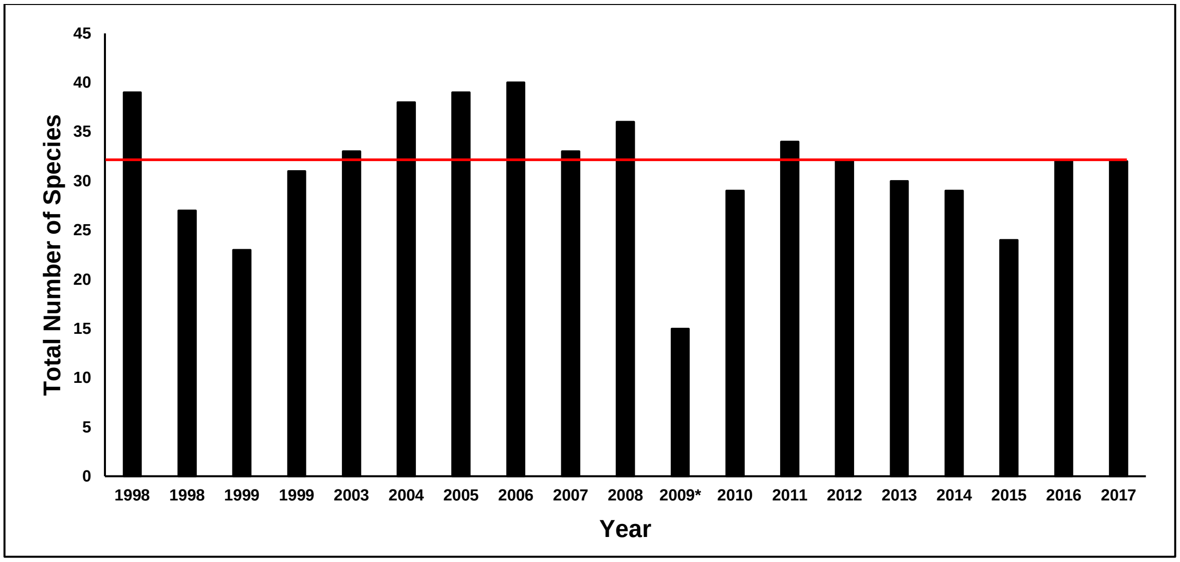


Figure 56. Yearly total species richness (shorebirds, seabirds, wading birds, and select waterfowl species) observed during MWBS efforts in CUIS (NPS unpublished data). * Indicates a year in which non-shorebird species were largely absent from the yearly data summary. The solid red line indicates the yearly species richness average of 31.4 species.

The focal avian guild of the MWBS, shorebirds, had the highest annual species richness average (14.4 species/year). Shorebirds were followed by seabirds (9.9 species/year), waterfowl (5.1 species/year), and wading birds (1.9 species/year) (Figure 57). As these species groups were not the target group of the MWBS, the species richness estimate for these groups was likely lower than what was actually present on the island during the surveys.

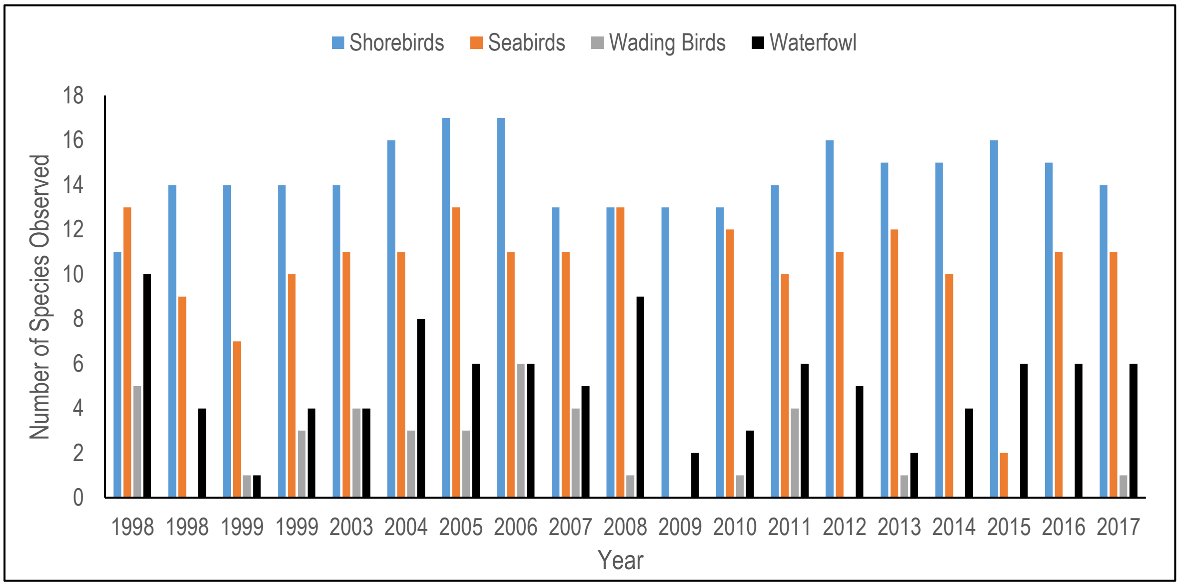


Figure 57. Yearly species richness numbers for the four species guilds reported at CUIS during the MWBS from 1998-2017 (NPS unpublished data). The low count in 2009 represents a year in which non-shorebird species were largely absent from the yearly data summary.

The annual MWBS has documented seven federal or state listed species in CUIS between 1998-2017: American oystercatcher, black skimmer (*Rynchops niger*), piping plover, red knot, Wilson's plover (*Charadrius wilsonia*), and wood stork. Three of these species were observed during every year of the MWBS in CUIS: American oystercatcher, piping plover, and the Wilson's plover. The wood stork and the piping plover were the only federally listed threatened species documented by the MWBS.

SECN Landbird Monitoring (2010, 2012)

Observer-based surveys in 2010 documented the presence of 50 landbirds across 30 survey plots in CUIS (Byrne et al. 2011) (Appendix H). Surveys were repeated at CUIS in 2012 using a different methodology (using ARDs instead of field observers) (Kurimo-Beechuk and Byrne 2016). In-lab data analyzers identified the vocalization of 55 unique species at the 30 sample sites in CUIS in 2012 (Appendix H).

Dlugolecki (2012)

Bird surveys of the wetlands of CUIS in 2010 and 2011 yielded 36 unique species (Dlugolecki 2012) (Appendix H). Surveys were tailored only to wetland species in CUIS, and survey points were scattered across eight freshwater wetlands in the park. The federally threatened wood stork was one of the most commonly observed species during the surveys of Dlugolecki (2012). Other common wetland species observed included great blue herons, great egrets (*Ardea alba*), snowy egrets (*Egretta thula*), and black-crowned night herons (*Nycticorax nycticorax*).

Cumberland Island Christmas Bird Count (1986, 1992-present)

The Cumberland Island CBC survey area encompasses CUIS (Figure 55). Counts such as the CBC (or other index counts, e.g., breeding bird surveys) are neither censuses nor density estimates (Link and Sauer 1998). The overall usefulness of index count data is often limited by possible biases of count locations and the number of observers, and it is often not advisable to estimate overall population sizes from these data alone (Link and Sauer 1998). These biases may influence how many individuals are observed in a given year, and may potentially explain the annual variation observed in species each year. Results of the Cumberland Island CBC should be interpreted with a degree of caution.

During the 25 years of CBC efforts for the entire Cumberland Island count circle (not just within CUIS boundaries), 211 bird species have been observed (Appendix H). The highest number of species observed in a given year was 157 (2006; 15 observers), while the lowest number of species observed was 83 (1986; 7 observers) (Figure 58). The average number of bird species observed during the Cumberland Island CBC was 137.1, and the average number of observers per year was 16.4.

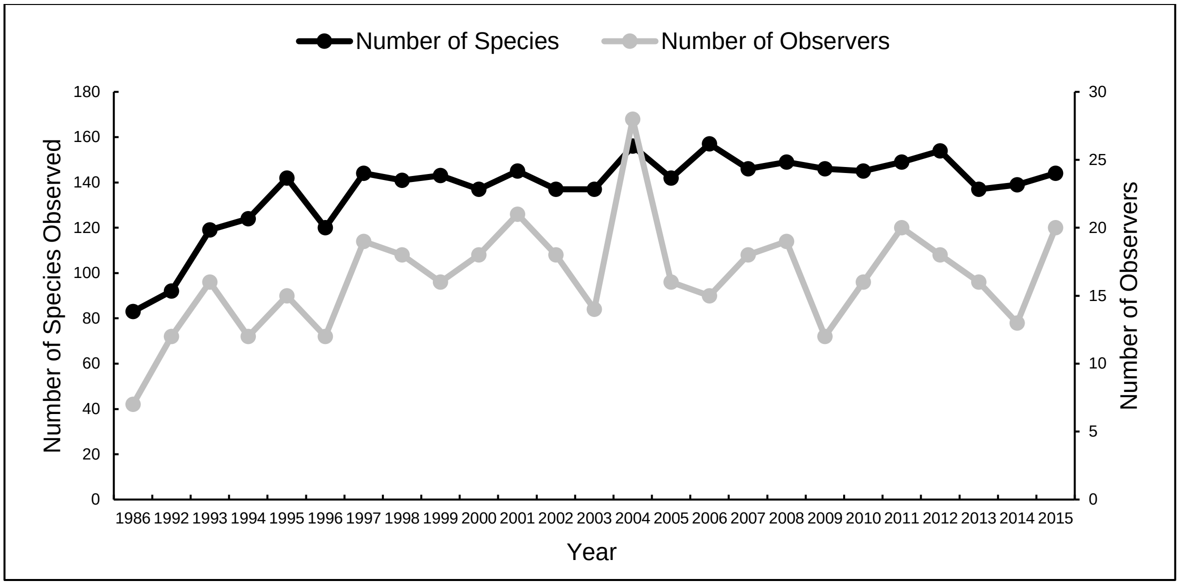


Figure 58. Number of bird species and observers during the Cumberland Island CBC between 1986 and 2015. Note that data include all count circle results and are not specific to CUIS. Data retrieved from http://netapp.audubon.org/CBCObservation/Historical/ResultsByCount.aspx.

Shorebird Nesting Numbers

Georgia DNR Mid-winter Beach Waterbird Survey (1998-2017)

The MWBS efforts document shorebird numbers, but the timing of the surveys do not typically overlap with the shorebird nesting season. The data from the MWBS are presented here for reference, but their utility in assessing the overall condition of the shorebird nesting numbers measure is marginal.

The species with the highest annual abundance during the MWBS in CUIS included the dunlin (*Calidris alpina*), semipalmated plover (*Charadrius semipalmatus*), black-bellied plover, western sandpiper (*Calidris mauri*), and the sanderling (*Calidris alba*) (Table 53). The MWBS typically highlights several priority species for observers so that they can document observation coordinates. These species include the piping plover, American oystercatcher, marbled godwit, Wilson's plover, and the red knot. The red knot had the highest abundance of these focal species from 1998-2017, with an average annual abundance of 337 individuals. The average abundance of the remaining focal species were similar, ranging between 9 ind/year (Wilson's plover) and 31 ind/year (American oystercatcher) (Table 54).

Table 53. The five most frequently observed shorebird species during 1998-2017 MWBS efforts in CUIS (NPS unpublished data).

Species	Avg # Ind/Year
dunlin	5,828
semipalmated plover	2,342
black-bellied plover	966
western sandpiper	669
sanderling	532

Table 54. Average annual abundance of focal shorebird species during the MWBS at CUIS from 1998-2017 (NPS unpublished data).

Species	Avg # Ind/Year
red knot	337
American oystercatcher	31
piping plover	27
marbled godwit	12
Wilson's plover	9

Plauny (2000)

Plauny (2000) recorded approximate nesting numbers and reproductive success estimates for American oystercatchers and Wilson's plovers at CUIS in 1999. Researchers documented nine nesting pairs of American oystercatchers at CUIS, with an additional nesting pair suspected, but not confirmed, near Christmas Creek (Plauny 2000). In total, these nine nesting pairs combined for 13 nests during the 1999 breeding season (Table 55). The South Segment (Figure 52) supported the highest number of nesting oystercatchers, with five pairs of birds nesting near the jetty at the southern border of the park. These five pairs combined for eight nesting attempts.

Table 55. Nesting numbers of American oystercatchers (AMOY) and Wilson's plovers (WIPL) in CUIS during the 1999 breeding season (Plauny 2000).

<table>
<tr><th>Segment</th><th>Colony</th><th>AMOY Nests</th><th>WIPL Nests</th></tr>
<tr><td rowspan="2">North</td><td>Long Point</td><td>2</td><td>24</td></tr>
<tr><td>North Cut</td><td>–</td><td>–</td></tr>
<tr><td>Middle</td><td>Sea Camp</td><td>3</td><td>4</td></tr>
<tr><td rowspan="2">South</td><td>North Jetty</td><td>4</td><td>8</td></tr>
<tr><td>Beach Creek</td><td>4</td><td>9</td></tr>
<tr><td>Total</td><td>–</td><td>13</td><td>45</td></tr>
</table>

Forty-five Wilson's plover nests were observed during the breeding season of 1999 (Plauny 2000). The majority of Wilson's plover nests were found in the North Segment, as a sizable nesting colony of 11 plover pairs was documented. An additional nesting colony of four pairs was also documented in the North Segment, and five other pairs were scattered individually along the shore of the Segment. These 20 plover pairs combined for 24 nesting attempts during the breeding season (Plauny 2000) (Table 55). Thirteen plover pairs were observed in the Middle Segment, but only four nests were observed in this Segment. Approximately 20 nesting pairs of Wilson's plovers were observed in the South Segment. Plauny (2000) documented 17 nests in the South Segment (Table 55).

Sabine et al. (2006)

Sabine et al. (2006) conducted American oystercatcher surveys in 2003 and 2004 along the length of the Atlantic beach in CUIS. Eleven pairs of American oystercatchers were documented in 2003, with the birds combining for 19 nesting attempts (11 primary nests, six renests, and two second renest attempts) (Table 56). In 2004, 10 pairs attempted to nest in CUIS, with 13 nest attempts reported (10 primary nests, three renests) (Table 56).

Table 56. American oystercatcher nesting numbers at CUIS during the 2003 and 2004 breeding seasons (Sabine et al. 2006).

Location	Year	# of Pairs	# of Clutches
North End	2003	5	6
	2004	5	7
South End	2003	6	13
	2004	5	6
Total	**–**	**–**	**32**

Strickland (2015)

During Wilson's plover breeding surveys at CUIS in 2014 and 2015, Strickland (2015) documented 136 nests; 63 nests were recorded in 2014, and 73 in 2015. Peak breeding abundance was noted between May and June 2015 when approximately 42 breeding pairs were sighted along the park's Atlantic shoreline. The average number of breeding pairs along CUIS's shoreline during Strickland (2015) was 6.07 pairs (±1.16 SE) per km, with an average range of observations between three and 12 pairs per km.

<u>Shorebird Fledging Success</u>

Many shorebird species are long-lived and have highly variable and generally low reproductive rates (Schulte and Simons 2015). For example, the American oystercatcher's reproductive rates are highly variable and often average well below one chick per nesting pair, with average annual nesting success rates on the Atlantic coast ranging from 0.2-0.75 chicks per pair (Davis et al. 2001, McGowan et al. 2005, Wilke et al. 2005, Traut et al. 2006, AOWG 2012). Shorebird chicks are usually precocial or semi-precocial and leave the nest rapidly after hatching. This mobility, combined with their cryptic behavior following hatching, makes determining fledging success even more difficult. Fledging success is often hard to estimate because of these factors, and accurate assessments of nesting/fledging success are often the result of careful monitoring by observers or recording equipment.

A fledged American oystercatcher chick walking the beach in CUIS (NPS Photo).

Plauny (2000)

Plauny (2000) reported fledging success for Wilson's plovers and American oystercatchers in CUIS by expressing the percentage of all nests that were observed to be successful (i.e., the percent of nests that fledged chicks). However, there were several instances where Plauny (2000) observed plover pairs and their chicks but could not identify a nest. Because of this, some estimates of nesting success may not be truly representative of the actual nesting success that occurred in the study areas.

The average nesting success of Wilson's plovers in CUIS in 1999 was 53.1% (±16.2%), and all documented Wilson's plover colonies on the island successfully hatched at least one nest (Table 57). The North Segment of the island had seven successful nests (29.2%) which produced 14 chicks, while the Middle Segment had four successful nests (100%) and 13 chicks, and the South Segment had three successful nests (33.3%) and eight chicks.

Average reproductive success for American oystercatchers in CUIS during the 1999 breeding season was 18.8% (±12.0%) (Table 57). Only two nests were successful in the park, with one nest in the Long Point Colony in the North Segment fledging one chick, and one nest in the North Jetty Colony fledging one chick. Plauny (2000) indicated that nine nesting pairs were observed during 1999, with potentially a tenth pair present. Using the estimate of nine nesting pairs, nesting success in 1999 was approximately 0.22 chicks per pair, which is within, but on the low end of, comparable nesting success ranges along the Atlantic coast (0.22-0.75 chicks/pair) (Davis et al. 2001, McGowan et al. 2005, Wilke et al. 2005, Traut et al. 2006, AOWG 2012).

Table 57. Observed reproductive success of Wilson's plovers (WIPL) and American oystercatchers (AMOY) at CUIS during the breeding season of 1999 (Plauny 2000).

Segment	Colony	Total Number of Nests		Number of Successful Nests		Apparent Nesting Success (%)*	
		AMOY	WIPL	AMOY	WIPL	AMOY	WIPL
North	Long Point	2	24	1	7	50	29.2
	North Cut	–	–	–	–	–	–
Middle	Sea Camp	3	4	0	4	0	100
South	North Jetty	4	8	1	4	25	50
	Beach Creek	4	9	0	3	0	33.3
Totals; mean±SE	**–**	**13**	**45**	**2**	**18**	**18.8±12.0**	**53.1±16.2**

* A nest was considered successful when $\geq$1 egg hatched successfully.

Sabine et al. (2006)

During the 2003 and 2004 breeding season, Sabine et al. (2006) documented American oystercatcher fledging success rates of 21% and 38%, respectively. Clutches on the northern end of the island fledged six chicks from four successful nests (67%) in 2003, and fledged six chicks from three successful nests (43%) in 2004 (Table 58). The southern end of the island saw no successful clutches in 2003, and three chicks from two successful nests (33%) in 2004 (Table 58).

Table 58. Total number of pairs/clutches and fledging success of American oystercatchers at CUIS, 2003 and 2004 (Sabine et al. 2006).

Location	Year	# of Pairs	# of Clutches	# of Clutches That Fledged Chicks (%)	# of Chicks Fledged
North End	2003	5	6	4 (67)	6
	2004	5	7	3 (43)	6
South End	2003	6	13	0 (0)	0
	2004	5	6	2 (33)	3
Total	**2003**	**11**	**19**	**4 (21)**	**6**
Total	**2004**	**10**	**13**	**5 (38)**	**9**
Total	**–**	**21**	**32**	**9 (28)**	**15**

In 2003 and 2004, the northern end of the island had an average of 1.20 chicks/pair, which is well above the average range that has been observed in the Atlantic coast (0.22-0.75 chicks/pair). The southern end of the island had much lower reproductive success, with no chicks being fledged in 2003, and an average of 0.60 chicks/pair in 2004. Collectively, the island averaged 0.55 chicks/pair in 2003, and 0.90 chicks/pair in 2004 (Sabine et al. 2006).

Strickland (2015)

Wilson's plover reproductive success (defined as hatching at least one egg) along the middle and southern portions of CUIS during the 2014 and 2015 breeding seasons (combined) was approximately 29%. Observers documented 136 nests during the two breeding seasons, with 39 nests hatching at least one egg. Unlike some previous studies, reproductive success during Strickland (2015) was not defined by fledging and estimates may not be accurately compared to other studies.

Wading Bird Nesting Numbers

Few studies have documented wading bird nesting numbers in the park. The studies that have observed wading birds did so either as part of late-summer occupancy/abundance estimates (Bratton 1988), or as part of wetland use surveys (Dlugolecki 2012). Wading birds likely nest on Cumberland Island, but the degree to which they are present yearly is poorly documented. Early surveys of the island by Pearson (1922) and Sprunt (1936) recorded great blue herons, great egrets, snowy egrets, tricolored herons (*Egretta tricolor*), little blue herons (*Egretta caerulea*), green herons (*Butorides virescens*), black-crowned night herons, and yellow-crowned night herons (*Nyctanassa violacea*) nesting on the island. Hillestad et al. (1975) noted that wading birds nested in emergent vegetation all around the island, but the location of the rookeries and colonies varied yearly. Wood storks and white ibises (*Eudocimus albus*) were documented breeding on the island through the late 1980s (Ruckdeschel and Shoop 1987), but these rookeries have since disappeared (Carol Ruckdeshel, personal communication as cited in Dlugolecki 2012).

The alteration of the island's wetlands may be responsible for reduced numbers in nesting wading birds. Wood storks typically use Cumberland Island for foraging and roosting habitat, but suitable nesting habitat may not be available year round as water levels fluctuate in the park's wetlands. Dlugolecki (2012) noted that the presence of water in the freshwater wetland complexes of the island (e.g., Lake Retta, Whitney Lake) increased the likelihood of wading bird presence for almost all wading species.

While the narrative of these studies suggest that many wading birds nested historically on the island, none of these sources provide any actual estimate of current nesting numbers in CUIS. Until such data exists, an assessment of current condition for wading bird nesting numbers cannot be completed.

Wading Bird Fledging Success

Similar to the wading bird nesting numbers measure, no data have been collected in CUIS regarding the fledging success of wading birds. A detailed study that documents the nesting numbers and fledging success of wading species in CUIS is needed before the current condition of these measures is determined.

Threats and Stressor Factors

Migratory bird species face deteriorating habitat conditions along their migratory routes and on wintering grounds. Most of the birds that breed in the U.S. winter in the Neotropics (MacArthur 1959); deforestation in these wintering grounds has occurred at an annual rate up to 3.5% (Lanly 1982). While forest and habitat degradation does occur in the U.S., it does not approach the level of degradation seen in the tropics (WRI 1989). Furthermore, Robbins et al. (1989) supported the suggestion that deforestation in the tropics has a more direct impact on Neotropical migrant populations than deforestation and habitat loss in the U.S.

Wetlands represent a habitat type that has been declining across much of the continental U.S., with Dahl (1990) estimating that 53% of the wetlands in the continental U.S. were lost between 1780 and 1980. The wetlands of CUIS have been heavily impacted by historic human use, and represent a habitat that is declining in the park. The extent of wetlands in the park has likely declined by nearly 200 ha (494 ac) since pre-settlement (~1600) based on estimates from Frost et al. (2011) (see Table 26).

Before the turn of the 20th century many of the wetlands had their stream flows and water channels altered, either through diking, tramping by livestock, or other mechanisms, in order to improve hunting and agricultural conditions (Dlugolecki 2012). The historic presence of cattle and other livestock on the island resulted in heavy grazing around the wetlands, and when these animals were removed, the lack of wetland management resulted in the encroachment of many woody species in these areas. Whitney Lake, the park's largest body of freshwater, experienced an 87.5% reduction in open water between 1973 and 2011 (Hillestad et al. 1973, Dlugolecki 2012), and many other wetlands in the park only have water present seasonally or after rainfall events.

Predation also represents a major threat to birds (particularly shorebirds) in CUIS. Many of the nesting shorebird studies completed in CUIS have focused on the nest fate of American oystercatchers and Wilson's plovers, and investigated the effects that predators and human disturbance had on productivity (Sabine et al. 2006, Sabine et al. 2008, Strickland 2015). Common predators of shorebirds along the Atlantic coast include raccoons, feral cats (*Felis catus*), coyotes (*Canis latrans*), foxes (*Vulpes* sp.), raptors, gulls (*Larus* sp.), feral hogs, and bobcat (Johnson et al. 1974a, Sabine et al. 2006, Sabine et al. 2008, Strickland 2015). While not a predator of the shorebirds in the park, feral horses have also been documented trampling shorebird nests (Sabine et al. 2006). The NPS has established various predator control measures since being designated as a national seashore, with feral hog trapping and hunting occurring in the park since 1974, sporadic raccoon removal efforts, and coyote trapping. While these predators affect many resources in the park, their removal/reduction in population size has likely improved shorebird nesting success in the park compared to pre-NPS management of the island.

Human disturbances to nesting shorebirds is also a concern, as shorebirds typically respond to human disturbance by fleeing the nest, at least temporarily, to escape the perceived threat. Sabine et al. (2008) documented human disturbances in the park causing direct and indirect failure of nesting American oystercatchers, and in one instance documented a child entering a nest, picking up and destroying the eggs, and ultimately destroying the nest. Based on 2003 and 2004 visitation, Sabine et al. (2008) found that American oystercatchers would safely tolerate human pedestrian traffic that was approximately 137 m (449 ft) from the nest. As most visitors to CUIS arrive by ferry, recreational use of CUIS's beaches and dune areas, especially near the ferry docking locations, is high. The southern end of the island, particularly along the Pelican Banks, experiences more private boat traffic and visitation from park guests who do not ferry to the island.

Being located along a major migration route, CUIS is frequently home to migratory "fallout" events. Fallout is when migratory birds descend to the ground in large numbers following a disturbance of some kind. While exhaustion is one of the most common causes of fallout, many factors can influence a species' migration pattern and cause fallout events. Examples of these factors include food availability (Niles et al. 1996), the presence of a large desert (Berthold 1993) or open body of water (Alerstam 1990), topographic features (Berthold 1993, Strickland 2015), or weather events (Alerstam 1990, Niles et al. 1996). In CUIS, weather-related fallout events are common, as hurricanes and strong thunderstorms may occur along the coast during migratory periods. Spring fallout events in Georgia may occur after strong, fast-moving cold-fronts move across the coast. The heavy rain and wind that accompany these cold fronts force migratory birds to the ground to avoid exhaustion. Migratory species that reach CUIS via a transoceanic flight (across the Caribbean Sea or Gulf of Mexico) typically avoid periods of unfavorable weather, and large-scale movements often coincide with favorable wind conditions (Richardson 1976, Williams et al. 1977, Williams 1985, Moore et al. 1995, Butler 2000). Birds migrating over landmasses tend to ground when wind and weather conditions deteriorate (Butler 2000).

While extreme weather events such as hurricanes, tropical storms, severe thunderstorms, and tidal surges represent a significant threat to CUIS's birds, drought is also a source of stress for birds. Drought is a major threat to most of the natural resources in CUIS. Not only do periods of drought remove many sources of standing water in CUIS (particularly the freshwater wetland areas), but these periods also affect availability of food for birds. Drought may reduce forage items such as insects and plant species (Smith 1982), and could lead to starvation for many birds in the park. Another impact of drought is that it may alter the nesting success of species, as Gaines et al. (2000) noted that wood storks have been observed relocating entire rookeries in response to drought conditions. Drought could also interrupt or alter the migratory patterns of species (Zeng 2003, Dai et al. 2004, Gordo 2007).

As has been discussed previously, the freshwater wetlands of CUIS are high priority areas for many avian species in the park. The wetlands of CUIS evolved with the presence of a semi-regular fire regime (Heath and Byrne 2014). Around the turn of the 19th century, the human residents of the island utilized many prescribed burns to keep freshwater wetland areas of the island open for recreation (primarily hunting). Areas that were managed with frequent prescribed burns included Whitney Lake and Willow Pond (Turner 1983, Dlugolecki 2012). Fires have largely been suppressed in CUIS since the mid 1900s. The lack of a consistent, reliable fire regime in the park means that many of the benefits of burning (e.g., removing accumulated organic matter that can fill depressions that usually hold water) are no longer occurring. Fires typically maintain and promote plant species diversity on the island (Davison and Bratton 1988), and without regular burning, wetlands on the island have experienced elevated levels of woody species encroachment and drying (Bellis 1995, Heath and Byrne 2014). The overall reduction in wetland extent, combined with a shifting plant species composition in these areas, will likely impact avian foraging and nesting habitat across the island.

<u>Data Needs/Gaps</u>

Continuation of the SECN landbird monitoring efforts are needed to better characterize the species richness of the park, and to analyze any potential trends in species presence or abundance over a longer period. Summarization of visits to the park since 2012 is also needed (currently underway). Many of the measures in this assessment represent data gaps due to the lack of shorebird/wading bird-specific research. There have been several studies focusing on priority shorebird species (e.g., American oystercatcher, Wilson's plover), but a broad characterization of the nesting population of shorebirds (and associated fledging success) is needed. Similarly, there is very little information on wading bird nesting numbers and fledging success. Until expanded research efforts are established in the park, an assessment of current condition for these measures is not possible.

<u>Overall Condition</u>

Species Richness

The NRCA project team assigned this measure a *Significance Level* of 3 during project scoping. CUIS is home to many different bird species (307 species) of many different guilds (e.g., shorebirds, wading birds, landbirds, seabirds). However, the collective bird guilds of the park have been understudied. The SECN has monitored landbirds in the park semi-annually since 2010, and several independent studies have completed species-specific shorebird studies. Yet the park is in need of broad surveys across the entire island in order to accurately document the species composition of the island. The wading bird community, raptor community, and seabird community are underrepresented in the current data for CUIS.

The two informal historic bird surveys that took place on Cumberland Island (Pearson 1922, Sprunt 1936) could be used as reference conditions for future surveys on the island, assuming they focus equally on all communities and guilds. Many species that are present on NPS (2016a) are likely sporadic visitors to the park (especially seabird species), and the results of the early general surveys may serve as more appropriate reference conditions than the overall species list for the park. Repeat surveys of the park would allow for potential species richness trend analyses, and would facilitate a more accurate comparison of annual richness to reference conditions. Until a more comprehensive bird survey for CUIS takes place that documents all bird species in the park, a *Condition Level* for this measure cannot be assigned.

Shorebird Nesting Numbers

CUIS managers assigned the shorebird nesting numbers measure a *Significance Level* of 3. With the exception of the MWBS, many of the shorebird-specific studies that have taken place in CUIS have focused on only two species: American oystercatcher and Wilson's plover. Plauny (2000) documented nine nesting pairs and 13 American oystercatcher nests in CUIS in 1999. Almost five years later, Sabine et al. (2006) documented 11 American oystercatcher pairs and 19 nests in 2003, and 10 pairs with 13 nesting attempts in 2004 (Table 56). Over the limited sample size, nesting numbers observed along CUIS's Atlantic beach were relatively similar. Methodologies varied between Plauny (2000) and Sabine et al. (2006), and direct comparisons or trends between the breeding seasons may not be statistically valid.

In 1999, 53 pairs of Wilson's plovers were observed along the coast of CUIS and were responsible for 45 nests. The highest concentration of nesting plovers was observed on the northern portion of the island, from Long Point to Duck House Trail (Table 55). Total Wilson's plover nest numbers in 2014 increased to 63, and to 73 in 2015, although the survey methodology and timing in 2014 and 2015 were not identical to that which was used in 1999 (Strickland 2015).

It is difficult to assess the overall current condition of shorebird nesting numbers using only sporadic nesting surveys for two species. Further, the most comprehensive abundance estimates for shorebirds in the park comes from the annual MWBS. As discussed previously, this survey is great for providing abundance estimates for shorebirds overwintering or stopping over in the park, but the timing of the survey does not align with the nesting season for many of the shorebird species in the park. A broader nesting survey for shorebirds is needed to document the overall nesting numbers of shorebirds in the park. Until these data exist, a *Condition Level* for this measure cannot be assigned.

Shorebird Fledging Success

A *Significance Level* of 3 was assigned to the shorebird fledging success measure. Similar to the shorebird nesting number measure, limited data exist for the shorebird fledging success measure, and the data that do exist are specific to the same two species as the previous measure. Shorebird reproductive and fledging success rates vary annually, and the establishment of an annual survey in the park is needed to accurately assess the current condition of this measure. The typically low reproductive success rates of shorebirds and the relationship between fledging success and population size is relatively poorly understood. As described in Sabine et al. (2006, p. 312), "It is unclear how current reproductive rates are affecting population trends, although high annual survival rates and long life spans may help to sustain populations with low and variable reproduction." Some species, such as the American oystercatcher, have been well studied across the Atlantic coast. Annual fledging success rates at CUIS could be compared to coast-wide reference conditions (Davis et al. 2001, McGowan et al. 2005, Wilke et al. 2005, Traut et al. 2006, AOWG 2012) to gauge the current health or trends of the species at the park. A *Condition Level* was not assigned to the shorebird fledging success measure.

Wading Bird Nesting Numbers

Wading bird nesting numbers was assigned a *Significance Level* of 2 during project scoping. Wading birds have been under-studied in CUIS, with no formal survey existing that has documented nesting numbers of the many species that inhabit the island. The actual extent to which wading birds nest on the island is relatively unknown, as the freshwater wetlands where they typically nest fluctuate in water level yearly. It is likely that wading bird nesting numbers on the island will be tied to the water levels of the island, with years when there is abundant open freshwater having the highest levels of productivity and nesting numbers. However, until a study takes place in the park documenting any of these data, such assumptions are only conjecture. A *Condition Level* for the wading bird nesting numbers measure was not assigned at this time due to the lack of data on the island.

Wading Bird Fledging Success

The NRCA project team assigned the wading bird fledging success measure a *Significance Level* of 2 during project scoping. A *Condition Level* cannot be assigned for this measure until data exist for the wading bird community in CUIS.

Weighted Condition Score

The *Weighted Condition Score* for the birds component in CUIS is currently undefined. While it is known that the park has a broad assemblage of birds, additional annual monitoring of the many groups of birds, specifically shorebirds and wading birds, is needed. There are several species of high conservation concern that utilize the park at various stages of the year, and annual monitoring would also help to identify potential trends in these species (Table 59).

Table 59. Weighted Condition Score of Birds in CUIS.

Birds			
Measures	**Significance Level**	**Condition Level**	**WCS = N/A**
Species Richness	3	n/a	
Shorebird Nesting Numbers	3	n/a	
Shorebird Fledging Success	3	n/a	
Wading Bird Nesting Numbers	2	n/a	
Wading Bird Fledging Success	2	n/a	

4.6.6. Sources of Expertise

Mike Byrne, SECN Terrestrial Ecologist

Doug Hoffman, CUIS Biologist

4.7. Herpetofauna

4.7.1. Description

Herpetofauna (i.e., reptiles and amphibians) are vital components of ecosystems in the southeast; because they are often both predators and prey, herpetofauna serve as "critical trophic links" in these ecosystems (Tuberville et al. 2005, p. 538). Amphibians in particular are considered indicators of environmental quality, given their sensitivity to environmental change and degradation (Tuberville et al. 2005, Smrekar et al. 2013). Consequently, amphibian communities have been identified as a priority for SECN monitoring efforts (Smrekar et al. 2013).

To date, 19 amphibian and 43 reptile species have been confirmed as present at CUIS (NPS 2016f). These include five federally-protected sea turtle species, four additional reptiles, and one amphibian considered high priority conservation species by the State of Georgia (gopher tortoise [*Gopherus polyphemus*], diamondback terrapin [*Malaclemys terrapin*], eastern diamondback rattlesnake [*Crotalus adamanteus*], island glass lizard [*Ophisaurus compressus*], southern dusky salamander [*Desmognathus auriculatus*]) (GA DNR 2015a).

As mentioned in Chapter 2, CUIS serves as a major nesting area for the loggerhead sea turtle. Females nest along the island's ocean coast from May through early September, typically laying multiple clutches during this time (Richardson 1987, NPS 2016e). CUIS beaches and dunes are generally gently sloping without steep faces, which allows for relatively easy access to favorable nesting sites for female loggerheads (Ruckdeschel and Shoop 1994). The loggerheads nesting at CUIS are part of the Northern Nesting Subpopulation (Mays and Shaver 1998). Although relatively small, this subpopulation is an important source of male hatchlings. Sex determination in sea turtles is temperature-dependent; as a result, the majority of loggerhead hatchlings from warmer Florida nesting beaches are female (Mays and Shaver 1998). Other sea turtle species documented nesting at CUIS in the past decade are the green sea turtle (*Chelonia mydas*) and leatherback turtle (*Dermochelys coriacea*) (NPS 2016e).

American alligators also occur at CUIS and are often found around Whitney Lake, in the Sweetwater Lake Complex, and at Plum Orchard Pond (Hillestad et al. 1975). Female alligators nest in many of the island's freshwater marshes. Even in 1973, when the alligator was protected as a federal endangered species, the population at CUIS was estimated at approximately 100 individuals (Hillestad et al. 1975). The island is also home to three venomous snakes: the eastern diamondback rattlesnake, timber rattlesnake (*Crotalus horridus*), and the cottonmouth (*Agkistrodon piscivorus*) (NPS 2016f).

American alligator in Plum Orchard Pond (left) and eastern diamondback rattlesnake (NPS photos).

The gopher tortoise population, although considered introduced to CUIS, has become a species of management interest due to its decline in the majority of its range on the mainland (Hillestad et al. 1975, Moore 2016). Historically, gopher tortoises were found throughout longleaf pine communities along the southeast coastal plain; the loss and fragmentation of this habitat has likely contributed to a decline in tortoise populations of up to 80% during the 20th century (Jones and Dorr 2004, Moore 2016). The CUIS population is important for the species as a whole, because it occurs in a protected area and faces few anthropogenic stressors. The burrows of this long-lived reptile provide habitat for numerous other species, including many species of conservation concern, from invertebrates and amphibians to snakes and lizards. As a result, the gopher tortoise is considered a keystone species of southeastern sandy upland ecosystems (Moore 2016).

4.7.2. Measures

- Amphibian species richness
- Amphibian abundance
- Sea turtle species richness*
- Sea turtle nesting numbers
- Sea turtle hatch success
- Gopher tortoise population size*
- Gopher tortoise burrow count*

* These measures were assigned *Significance Levels* of 1. Measures with a *Significance Level* of 1 are not discussed in the current condition section of the text, but are briefly summarized in the Overall Condition section.

4.7.3. Reference Condition/Values

The reference conditions for this component will vary between measures. Tuberville et al. (2005), an intensive herpetofaunal survey, will serve as the reference condition for amphibian species richness. However, information regarding amphibian abundance is limited and a reference condition cannot be identified. The reference condition for sea turtle nesting numbers and hatch success will be based on the data collected at CUIS since monitoring was expanded to the entire oceanfront beach (mid-1980s). Given the limited information on the island's gopher tortoise population, the current condition (as outlined by Moore 2016) will serve as a reference or baseline for future assessments.

4.7.4. Data and Methods

The earliest known survey to document herpetofauna specifically at CUIS was Hillestad et al. (1975). While the systematic survey focused primarily on documenting the species that occurred on the island, some information on habitat use and distribution was also reported.

Sea turtle research began on the loggerhead nesting beaches of Little Cumberland Island (LCI) in 1964 (Hillestad et al. 1975). During the 1960s and early 1970s, the program included tagging female turtles on the beach, relocating eggs from nests to an artificial hatchery, and releasing hatchlings back into the ocean. The tagging project was expanded to the northern portion of Cumberland Island in 1972 and monitoring of nesting activity on the northernmost 8 km (5 mi) of Cumberland Island began in 1974 (Camhi and Ehrenfeld 1986, NPS 2016e). At that time, monitoring involved patrolling the nesting beach nightly from approximately mid-May to mid-August to document the number of "crawls" (tracks made by female turtles emerging on the beach), the number of nests, and hatch success (McMillen 1980).

CUIS sea turtle interns conducting patrol and nest relocation activities (NPS photos)

The dates of the first and last crawls were also recorded and evidence of nest predation was noted. Hatch success was measured by excavating nests after hatchling emergence to count the numbers of egg shells (successful hatching) and unhatched or broken eggs. In the mid-1980s, patrols were expanded to the entire island, although monitoring efforts were still primarily focused on the northern portion (Camhi and Ehrenfeld 1986, NPS 2016e). The entire island has been monitored since the mid-1990s, with the exact same length of beach monitored (28.4 km [17.6 mi]) since 2003 (Mays and Shaver 1998, NPS 2016e). Over time, the monitoring season has expanded to run from late April to mid- or late October in most years. Sources that have reported on these loggerhead monitoring efforts at CUIS include Ruckdeschel (1977), Stoneburner (1979), McMillen (1980) Camhi and Ehrenfeld

(1986), (Richardson (1987), 1992)), and Ruckdeschel and Shoop (1994). Mays and Shaver (1998) and Dodd and Mackinnon (2002) include CUIS monitoring results in reports that cover additional islands and/or National Seashores. The CUIS monitoring results are also summarized in a spreadsheet maintained by the NPS (2016e) and reported to Seaturtle.org for addition to their online database.

In addition to this annual loggerhead monitoring, Ruckdeschel and Shoop (1989) examined stranded sea turtles that came ashore at CUIS in 1986-87 for signs of trauma. Only one source reported on the occurrence of other sea turtle species at CUIS. Rabon et al. (2003) documented leatherback nests on the Atlantic coast between North Carolina and Georgia.

Tuberville et al. (2005) conducted herpetofaunal surveys of the 16 SECN parks, including CUIS, from 2001 to 2003. Field surveys included a variety of sampling techniques such as terrestrial drift fences, coverboards, aquatic traps, aquatic dip netting, automated recording of anuran (i.e., frog and toad) calls, road-cruising, and opportunistic visual searches. These results were supplemented with searches of museum records, literature accounts, and personal collections/reports (Tuberville et al. 2005).

In 2009, the SECN initiated an amphibian community monitoring program at CUIS, with the objective to identify trends in species occupancy, distribution, diversity, and community composition (Byrne et al. 2010). The monitoring protocol incorporates three survey techniques: a time- and area-constrained visual encounter survey (VES), dip-netting in sampling locations with aquatic communities, and ARDs programmed to capture anuran calls (Smrekar et al. 2013). Thirty locations were sampled in 2009 and 31 locations were visited when monitoring was repeated in 2012 (Figure 59). The ARDs were co-located at the center of the 0.5-ha macroplot where VESs were conducted. ARDs are programmed to record for 30 seconds every 10 minutes from dusk to dawn every 3-4 days during a 77-day deployment (Byrne et al. 2010, Smrekar et al. 2013). ARDs were deployed for 10 days in 2009 and for approximately 16 weeks in 2012. VESs occurred during September and early October in 2009 (24 days) and during early March in 2012 (17 days) (Byrne et al. 2010, Smrekar et al. 2013).

Moore (2016) conducted a study of the CUIS gopher tortoise population in order to analyze population dynamics (population size, age distribution, sex ratio) and burrow characteristics (size, orientation, temperature, tortoise and other species use), as well as to explore the effects of a spring 2016 controlled burn on tortoise habitat. Tortoise burrows were located by walking transects during June of 2015 and 2016. Basic population data was collected from multiple locations (Figure 60) in both seasons and live trapping of tortoises occurred in the Stafford Field and Woods area during June-July 2016. Radio transmitters were placed on ten captured tortoises for tracking purposes (Moore 2016).

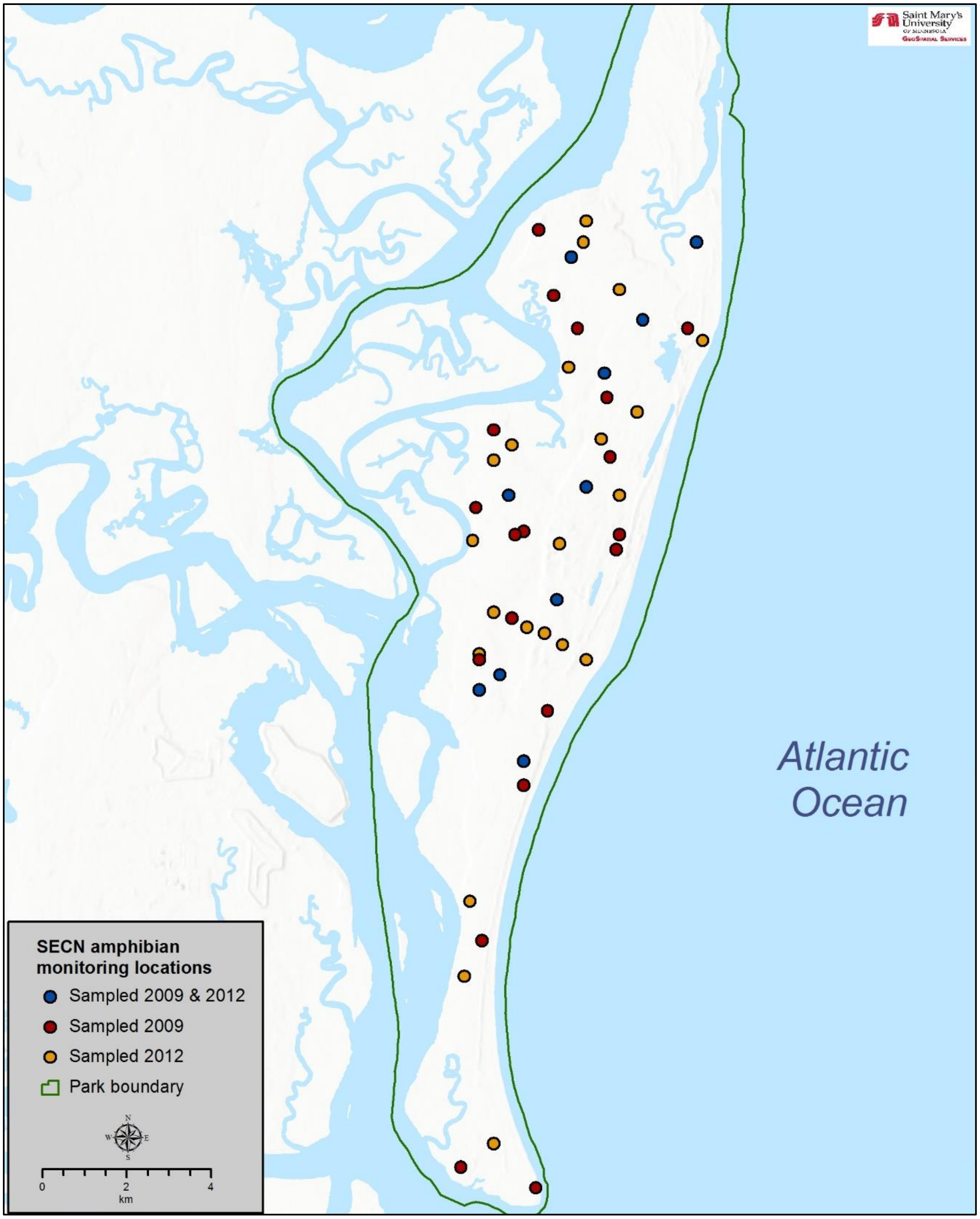


Figure 59. Amphibian monitoring locations selected by the SECN for sampling during the 2009 and 2012 seasons (Byrne et al. 2010, Smrekar et al. 2013).

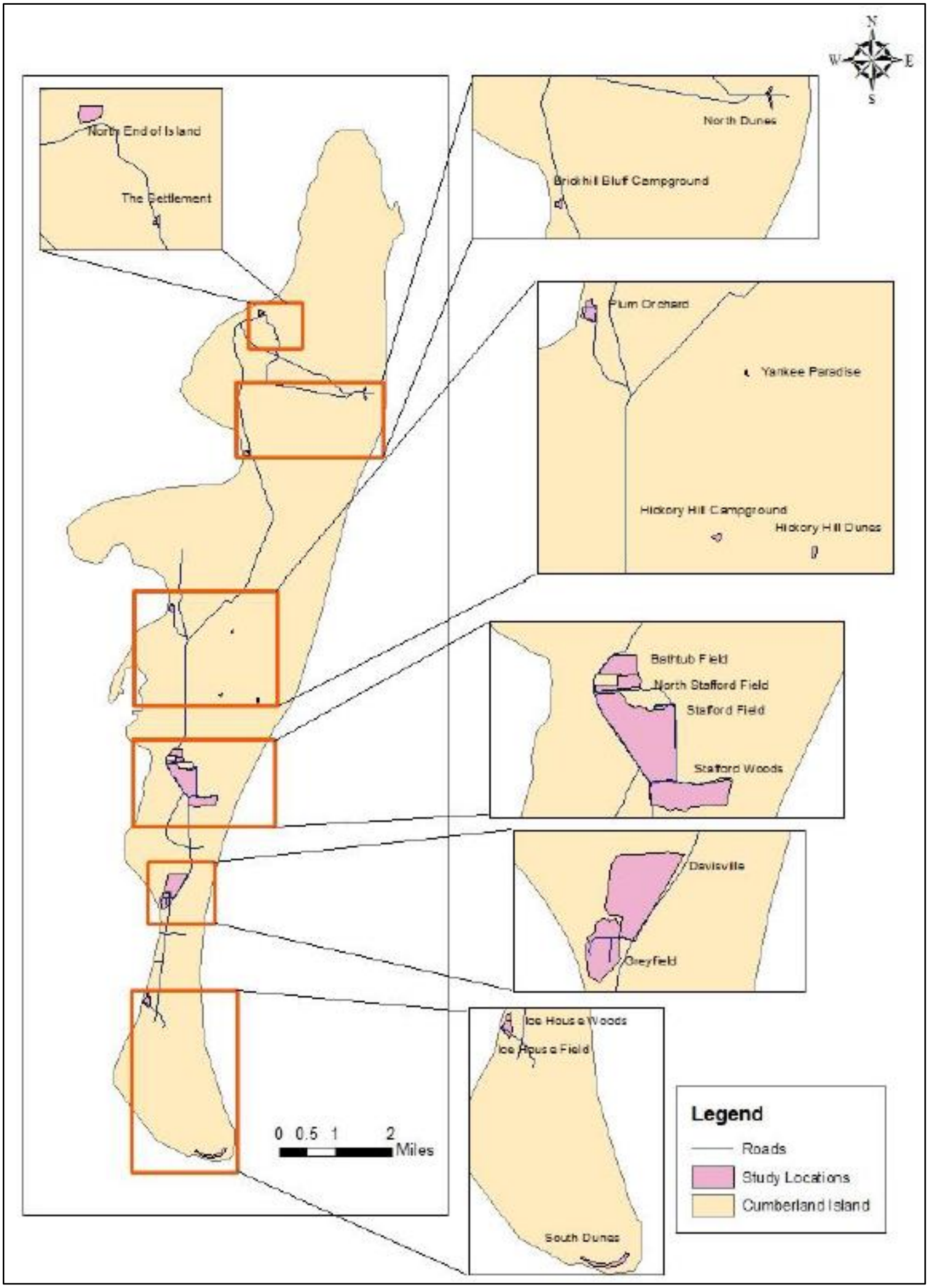


Figure 60. Locations surveyed for gopher tortoises and burrows by Moore (2016). Live trapping and radio tracking efforts were concentrated around Stafford Field and Woods in the central portion of the island.

4.7.5. Current Condition and Trend

Amphibian Species Richness

The saltwaters surrounding CUIS have apparently inhibited the colonization of the island by amphibians (Hillestad et al. 1975). According to NPSpecies (NPS 2016f), 19 amphibian species have been confirmed as present at CUIS: 14 from the order Anura (frogs and toads) and five from the order Caudata (salamanders and newts). All of the confirmed species are native. Table 60 shows the species documented by various survey efforts from Hillestad et al. (1975) to the most recent SECN monitoring (Smrekar et al. 2013). Between the two seasons of SECN monitoring, which is the most recent information available, 13 amphibian species were documented (Byrne et al. 2010, Smrekar et al. 2013). Five of the species not documented by SECN monitoring are salamanders or newts, which are rare or unlikely to be detected by the monitoring techniques used (Mike Byrne, SECN Terrestrial Ecologist, written communication, 20 September 2017).

Table 60. Amphibian species present at CUIS according to various surveys over time. M = existing museum specimen, L = published literature record.

Scientific Name	Common Name	Hillestad et al. (1975)	Tuberville et al. (2005)	Byrne et al. (2010)	Smrekar (2013)
Acris gryllus	southern cricket frog	x	L	x	x
Ambystoma talpoideum	mole salamander	M	L	–	–
Amphiuma means	two-toed amphiuma	–	M, L	–	–
Anaxyrus quercicus	oak toad	M	L	–	–
Anaxyrus terrestris	southern toad	x	x	x	x
Desmognathus auriculatus	southern dusky salamander	M	M, L	–	–
Eurycea quadridigitata	dwarf salamander	M	L	–	–
Gastrophryne carolinensis	eastern narrow-mouthed toad	x	x	–	x
Hyla chrysoscelis	Cope's gray treefrog	–	L	x	
Hyla cinerea	green treefrog	M	M, L	x	x
Hyla femoralis	pine woods treefrog	M	M, L	x	x
Hyla gratiosa	barking treefrog	x	L	–	–
Hyla squirella	squirrel treefrog	x	M, L	x	x
Hyla versicolor	gray treefrog	M	L	–	–
Lithobates grylio	pig frog	M	M, L	–	x
Lithobates sphenocephalus	southern leopard frog	M	x	x	x
Notophthalmus viridescens	eastern newt	M	L	–	–
Pseudacris crucifer	spring peeper	–	x	x	x
Pseudacris nigrita	southern chorus frog	M	L	x	
Pseudacris ocularis	little grass frog	M	x	x	x
Scaphiopus holbrookii	eastern spadefoot	x	M, L	x	–

Amphibians documented at CUIS (left to right): eastern spadefoot, Cope's gray treefrog, and green treefrog (NPS photos by J.D. Wilson).

<u>Amphibian Species Abundance</u>

Long-term changes in amphibian abundance may indicate local or regional environmental degradation that could impact other natural resources (Shoop and Ruckdeschel 1997). Until recently, very little attention has been given to documenting amphibian abundance at CUIS. Hillestad et al. (1975), p. 160 reported that anurans were abundant within the island's Sweetwater Complex, with breeding choruses of treefrogs and toads occasionally "so intense during spring nights that normal conversation cannot be conducted."

Because estimating abundance for many species often requires multiple intensive surveys and results can often be confusing, the SECN amphibian monitoring protocol was not designed to estimate species abundance (Byrne, written communication, 20 September 2017). Rather, the 2009 and 2012 SECN monitoring reported the frequency of occurrence (i.e., percent of sampling locations present) for amphibians detected during sampling. This provides some insight into abundance, in terms of whether a species is commonly encountered or not (Byrne et al. 2010). In 2009, across VESs and ARDs combined, the most frequently documented amphibians were the squirrel treefrog (*Hyla squirella*) and the green treefrog (*Hyla cinerea*) (Table 61). The squirrel treefrog again showed the highest frequency in 2012, along with the southern toad (*Anaxyrus terrestris*) (Smrekar et al. 2013).

Table 61. Amphibian species frequency of occurrence (% of sampling locations), as documented during 2009 and 2012 SECN monitoring.

Species	Byrne et al. (2010) Frequency (VES +ARD)	Smrekar et al. (2013) Frequency (VES +ARD)
southern cricket frog	27.0	6.5
southern toad	17.0	90.3
eastern narrow-mouthed toad	–	6.5
Cope's gray treefrog	7.0	–
green treefrog	77.0	51.6
pine woods treefrog	30.0	41.9
squirrel treefrog	80.0	90.3
pig frog	–	6.5
southern leopard frog	10.0	6.5
spring peeper	13.0	22.6
southern chorus frog	7.0	–
little grass frog	27.0	6.5
eastern spadefoot	7.0	–

Sea Turtle Nesting Numbers

As mentioned previously, loggerhead turtles are the dominant nesting species at CUIS; over the past decade, leatherback and green turtle nests have occasionally been found on the CUIS beach (NPS 2016e). In 2017, Kemp's ridley (*Lepidochelys kempii*) nests were documented at Cumberland Island for the first time since monitoring began (Seaturtle.org 2017). Because female loggerheads typically nest every 2-3 years rather than every year, the number of nests on any beach can vary widely between years (Hillestad et al. 1975, Richardson 1987). Since monitoring was extended to the entire CUIS ocean beach, annual loggerhead nest numbers have ranged from around 50 to nearly 900 (NPS 2016e). Although numbers have been variable between years, an overall increasing trend is apparent over the past decade (Figure 61, Table 62). CUIS has consistently supported some of the highest loggerhead nest numbers among all Georgia barrier islands (Dodd and Mackinnon 2002).

Since 2010, green turtle nests have been observed at CUIS in all but one year, with nest numbers ranging from 1-14 (Table 56) (NPS 2016e). Two leatherback nests were reported in 2001 (Rabon et al. 2003) and nests have been documented in 5 additional years since 2003. The highest number of leatherback nests observed was five in 2011 (NPS 2016e).

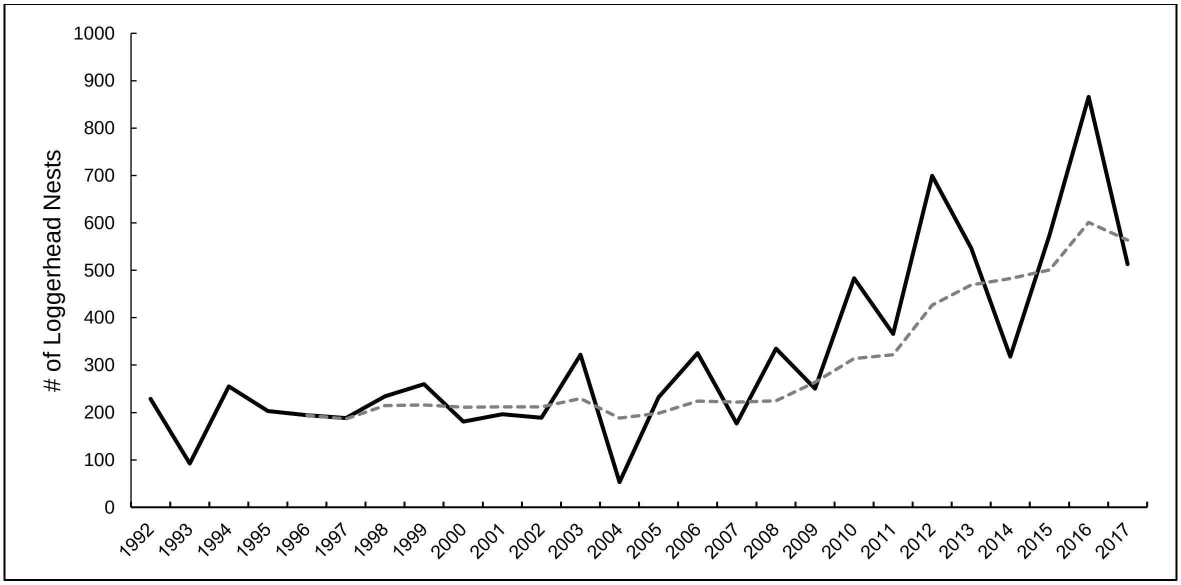


Figure 61. Loggerhead nest numbers for CUIS, 1992-2017 (NPS 2016e, Seaturtle.org 2017). The gray dashed line shows the 5-year moving average. The total beach length monitored during this time ranged from 26.0-28.7 km (16.2-17.8 mi) but has been consistent at 28.4 km (17.6 mi) since 2003.

Table 62. Annual sea turtle nest numbers by species since 2002 (NPS 2016e). Full records back to 1974 are included in Appendix I.

Year	Loggerhead Nests	Green Nests	Leatherback Nests	Unknown sp. Nests
2002	189	0	0	0
2003	322	0	1	0
2004	53	0	0	0
2005	232	0	0	0
2006	325	0	0	0
2007	177	0	0	0
2008	335	0	0	1
2009	250	0	2	0
2010	483	3	0	0
2011	366	1	5	0
2012	699	0	1	0
2013	547	14	0	0
2014	318	1	0	0
2015	575	3	1	4
2016	866	1	0	0
2017*	513	11	1	1

* Information in this row is from Seaturtle.org (2017); it is preliminary and subject to change.

Female loggerhead completing a nest (USFWS photo).

Sea Turtle Hatch Success

Sea turtle reproductive success can be measured in several ways. Hatch success simply measures the proportion of eggs that hatch and does not take into account the fate of turtles after hatching (i.e., hatchlings that die in the nest are considered successfully hatched) (Mays and Shaver 1998). Emergence success measures the proportion of hatchlings that successfully emerge from each nest and would not include hatchlings that died while still in the nest. According to Richardson (1987), hatch success under ideal conditions (i.e., no predation or inundation) typically ranges from 70-85%.

The earliest report of hatch success for CUIS came from Camhi and Ehrenfeld (1986) for the 1985 nesting season. Hatch success in that year, calculated for only the northernmost 8 km (5 mi) of the island was 79.2% (Camhi and Ehrenfeld 1986). Emergence success was 74.3%. According to Camhi and Ehrenfeld (1986), this hatching success rate was significantly higher than natural success rates reported for other southeastern nesting beaches at the time. Hatch success was higher for nests laid earlier in the season (May-June) than later (July-August) (Camhi and Ehrenfeld 1986).

Mays and Shaver (1998) documented loggerhead reproductive success from CUIS and two other national seashores from 1992-1997. Emergence success was reported for reproductive success, not hatch success. However, given that emergence success is typically lower than hatch success, emergence success could be considered a minimum or low-end estimate of hatch success. During this 6-year period, emergence success at CUIS ranged from 45.1-67.5% (Table 63). For comparison, emergence success at the two other national seashores (Cape Hatteras and Cape Lookout) in North Carolina ranged from 50.3-85.0% (Mays and Shaver 1998).

Since 2009, hatch and emergence success for Cumberland Island and for the state of Georgia as a whole have been documented and are available through seaturtle.org. Over the past 9 years, hatch success at CUIS has ranged from a high of 78.3% (2010) to a low of 55.0% (2017), with success exceeding 70% in six of the 9 years (Table 64). In comparison, loggerhead hatch success across the Georgia coast as a whole ranged from 53.1% (2017) to 68.3% (2010) (Seaturtle.org 2017). Emergence success at CUIS ranged from 52.4% (2017) to 76.4% (2010), and was higher than 1992-

1997 emergence rates (Table 63) in nearly all years. Statewide emergence rates from 2009-2017 ranged from 50.4% (2017) to 62.9% (2014). CUIS hatch and emergence success were consistently higher than success for the Georgia coast as a whole (Figure 62). Factors contributing to low hatch and emergence success at CUIS in 2017 include unexpected high tide events early in the season and the loss of approximately 130 nests to Hurricane Irma in September (Seaturtle.org 2017; Hoffman, written communication, October 2017).

Table 63. Reproductive success ([# hatched eggs - # dead hatchlings] / total # eggs x 100) for nesting sea turtles at CUIS and two additional National Seashores (Mays and Shaver 1998).

Year	CUIS	Cape Hatteras, NC	Cape Lookout, NC
1992	45.1	58.0	73.0
1993	52.2	50.3	74.0
1994	63.3	56.2	85.0
1995	56.1	63.9	51.0
1996	67.5	51.9	75.5
1997	64.7	60.5	73.0

Table 64. Loggerhead hatch and emergence success at Cumberland Island and throughout Georgia, 2009-2017 (Seaturtle.org 2017).

Year	Mean Hatch Success		Mean Emergence Success	
	CUIS	Georgia	CUIS	Georgia
2009	73.2	61.3	71.0	56.2
2010	78.3	68.3	76.4	62.7
2011	68.6	61.1	66.2	55.9
2012	73.5	66.9	71.7	61.9
2013	75.8	66.1	73.6	62.4
2014	77.5	67.8	75.4	62.9
2015	68.8	64.2	65.1	59.4
2016	72.8	61.9	70.7	58.5
2017	55.0	53.1	52.4	50.4

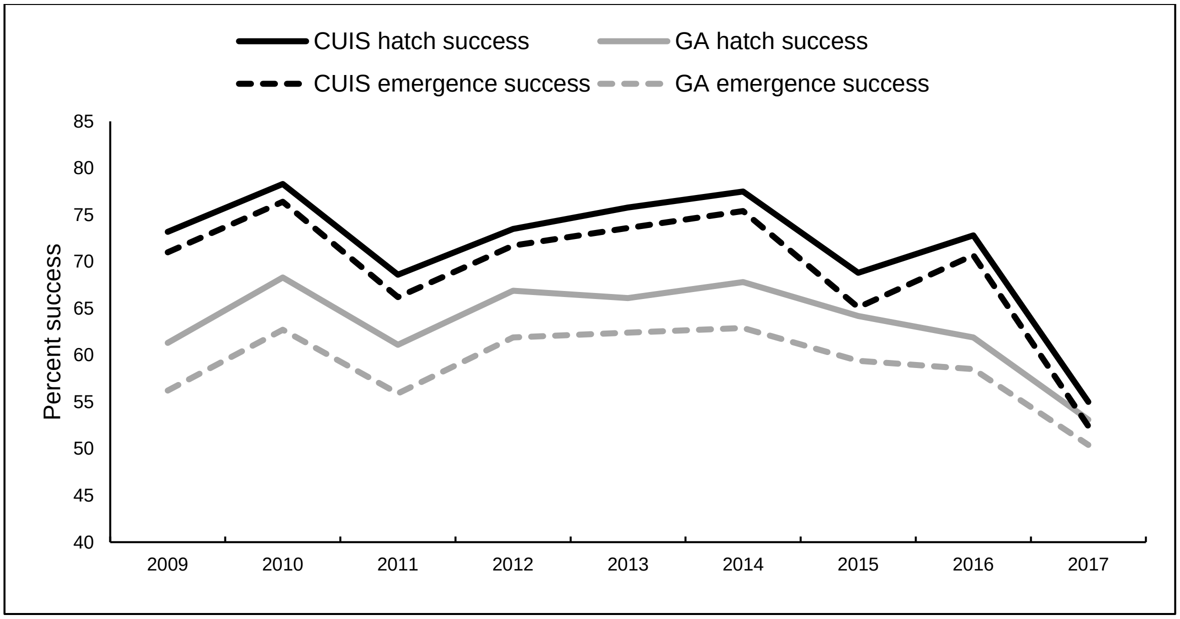


Figure 62. Loggerhead hatch and emergence success at CUIS (black lines) compared to success along the Georgia coast as a whole (gray lines) (Seaturtle.org 2017).

Loggerhead hatchlings emerging at night (USFWS photo). Red lights are often used when observing hatchlings at night to avoid misorientation.

Threats and Stressor Factors

Threats to CUIS's herpetofauna include habitat loss, drought, fire suppression, climate change, disease, and predation. Factors that negatively impact water quality and/or freshwater wetlands (as discussed in Chapter 4.2 and 4.8) can also threaten herpetofauna, particularly amphibians and alligators. Additional threats to sea turtles specifically are boat strikes, fishery-related injuries/mortality, light pollution, and illness/death due to lack of offshore food resources. A "stranding" is a sea turtle found dead, injured, sick, or otherwise abnormal in appearance and out of the water or in very shallow water, usually along the shoreline (NOAA 2015).

Habitat loss and alteration are considered a primary threat to herpetofauna populations. In North America, losses in area of freshwater wetlands have been substantial (Dahl 2000). A reduction in these important aquatic habitats, along with an increase in landscape fragmentation, have been implicated in declining trends in aquatic biodiversity, particularly aquatic reptile and amphibian taxa (Bates et al. 2008). At CUIS, fire suppression has contributed to habitat loss for both aquatic and terrestrial herpetofauna (GA DNR 2015a, Moore 2016). As discussed in Chapter 4.2, frequent fire historically maintained the open character of CUIS wetlands and removed accumulated organic matter that would otherwise fill in depressions capable of holding standing water (Hillestad et al. 1975, Bellis 1995). Due to fire suppression on the island since the mid-1900s, many of the wetlands that provide valuable herpetofauna habitat are experiencing woody species encroachment and drying due to filling with organic matter (Bellis 1995, Heath and Byrne 2014). Lack of fire has also contributed to woody species encroachment into open longleaf pine habitats, such as those preferred by the gopher tortoise (Moore 2016).

With many of CUIS's herpetofaunal species dependent on aquatic habitat at some stage in their life cycles, drought is a major threat to these populations. Climate change has been implicated in widespread drought events, which are interspersed with deluges (Bates et al. 2008). This results in huge amounts of runoff, erosion, and occasional flooding that have damaged riparian areas and other important aquatic habitats, as well as degrading water quality (Bates et al. 2008). An overall increase in global temperatures associated with climate change, which contributes to extended periods of drought, will have a combined effect on biota by causing temperature and water stress (Bates et al. 2008).

Warming temperatures associated with climate change may also impact reptile species with temperature-dependent sex determination (GA DNR 2015a). The temperature of the nest environment determines the sex of alligator, sea turtle, and some terrestrial turtle hatchlings (including gopher tortoise and diamondback terrapin) (Mrosovsky et al. 1984, GA DNR 2015a). Warmer ambient temperatures may unnaturally skew the sex ratio in these species. Atlantic coast Loggerhead hatchling sex ratios already vary throughout the nesting season, ranging from 10% female during the cooler beginning and end of the season and 80% female during the warmer mid-season (Mrosovsky et al. 1984). Shifting temperatures may also influence other aspects of sea turtle reproduction such as the timing of nesting (warmer water triggers earlier nesting and shorter intervals between clutches), the overall length of nesting season, and incubation periods (warmer sand temperatures shorten incubation) (Richardson 1987, Peek et al. 2016).

Ranavirus is a genus in the family Iridoviridae which can infect multiple species of amphibians and some reptiles (USGS 2016c). Ranaviruses have been associated with die-offs of more than 20 species of amphibians and turtles in over 25 states across the U.S., including gopher tortoises in Florida (GA DNR 2015a, USGS 2016c). Mortality due to ranaviruses occurs mostly in larval amphibians, true frogs, and chorus frogs. Infected individuals may exhibit subtle or severe hemorrhages in ventral skin, often appearing as an irregular rash; onset of illness is sudden and frequently affects most individuals within a wetland (up to or exceeding 90%) (USGS 2016c). Observed outbreaks have often been within wetland ecosystems and cause mass die-off of frogs and salamanders, with the highest mortality rates occurring in juveniles (USGS 2016c).

Chytrid fungus, specifically *Batrachochytrium dendrobatidis*, is a pathogen of amphibians that could potentially affect amphibian populations at CUIS. The pathogen has been identified as the cause of severe population declines on several continents, including North America (Piotrowski et al. 2004). Amphibians infected by *B. dendrobatidis* develop chytridiomycosis, an infectious non-hyphal zoosporic fungus that causes roughening and reddening of the skin, convulsions, ulcers and hemorrhages, and sporadic death. Not all amphibians infected with *B. dendrobatidis* develop chytridiomycosis or die; environmental factors, such as pH of the environment, drought, and temperature at time of infection, may affect mortality rates. Some research indicates that the fungus growth is inhibited by high temperatures (28°C [82°F]) and exposure of infected individuals to high temperatures may kill the fungus (Woodhams et al. 2003). If this is the case, the warm summer temperatures at CUIS may somewhat alleviate the threat of chytrid to the island's amphibians. Neither ranaviruses nor chytrid fungus infection have been detected at CUIS to date (Byrne and Moore 2011), but they may greatly impact amphibian populations if the diseases reach the park.

Predation is a threat to many of CUIS's herpetofauna species, but particularly for turtle and tortoise species (Hillestad et al. 1975, Ruckdeschel and Shoop 1994, Moore 2016). Although adult turtles and tortoises have few terrestrial predators, juveniles and nests are especially vulnerable to predation. In some Georgia gopher tortoise populations, 80-90% of nests may be depredated and less than 10% of hatchlings survive their first year (Landers et al. 1980, Moore 2016). Predators of gopher tortoises and diamondback terrapins at CUIS include raccoons, armadillos, coyotes, and predatory birds (D'Amato 2015, Moore 2016). Most of these predators, as well as feral hogs and other small mammals, also prey upon adult amphibians.

Nest predation has had a major negative impact on CUIS loggerheads in the past and continues to pose a threat to sea turtles today. This predation includes digging into nests to consume eggs and preying upon hatchlings as they emerge (Camhi and Ehrenfeld 1986). The primary predators at CUIS are raccoons, feral hogs, coyotes, ghost crabs, and armadillos (NPS Ruckdeschel and Shoop 1994, 2016d). Raccoons have long been known as a nest predator on barrier islands, although their activity tends to be localized to areas where forest cover occurs near the dunes and beaches (Hillestad et al. 1975, Camhi and Ehrenfeld 1986). Once a nest is raided by a raccoon, all the eggs are lost, either to the raccoon, ghost crabs, or other mammalian predators attracted to the disturbed area (McMillen 1980, Camhi and Ehrenfeld 1986). On many barrier islands, including CUIS, raccoons have been removed from nesting beaches through trapping and hunting, and beach patrols will often place large screens over nests to deter digging by predators (Dodd and Mackinnon 2002). Feral hogs and coyotes are also hunted and/or trapped on CUIS to reduce nest predation. Despite their small size, ghost crabs can prey upon or disturb a relatively large number of sea turtle hatchlings and nests. McMillen (1980) found that ghost crabs impacted 29.2% of nests and took 7.5% of eggs and hatchlings in one season, while Camhi and Ehrenfeld (1986) attributed 5% of unhatched egg loss to ghost crabs. These crabs have been observed following the path of ocean-bound hatchlings back to a nest and into the nest chamber, where they kill emerging hatchlings and break remaining eggs (McMillen 1980). Although not predators, feral horses may also disturb sea turtle nests, hatchlings, and nesting females (Ruckdeschel and Shoop 1994).

Since 1992, the NPS has tracked the number of sea turtle nests depredated by four mammalian predators (NPS 2016d). Over this entire period, raccoons and feral hogs have caused the greatest amount of depredation, although their impact has been drastically reduced since 2002 (Figure 63). Coyotes, which arrived on the island around 2004, have been significant nest predators since 2011 and armadillo depredation has also increased in recent years (NPS 2016d). Park staff will continue efforts to minimize nest predation at CUIS through targeted trapping and hunting of these mammalian predators (Hoffman, personal communication, March 2017).

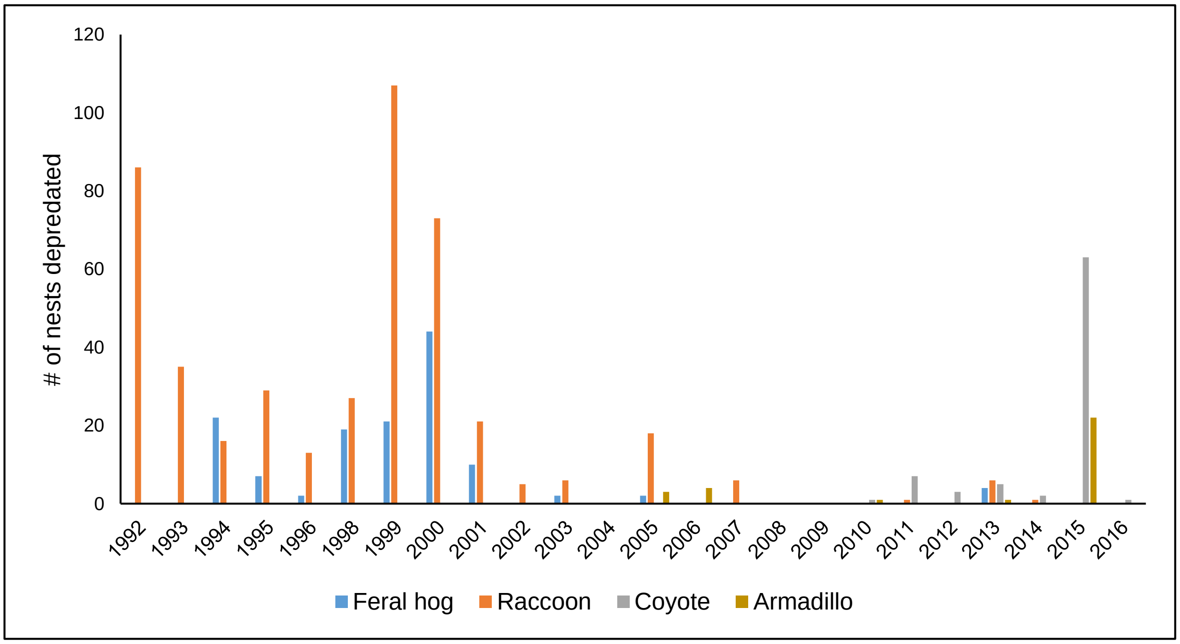


Figure 63. Sea turtle nest predation by four common mammalian predators at CUIS (NPS 2016d). 1997 is excluded due to incomplete data. Note that coyotes were not present on the island until around 2004.

Sea turtle nesting habitat is also affected by beach and/or dune erosion. If dune-stabilizing vegetation is lost due to grazing, drought, or storm impacts, the dunes may experience deflation, causing them to migrate further away from the beach (Hillestad et al. 1975). Female sea turtles often use the location of dunes as a cue to locate suitable nesting habitat. Beach and dune erosion can also cause the formation of vertical escarpments (Figure 64), which prevent sea turtles from accessing nesting habitat (Ruckdeschel 1977, Mays and Shaver 1998). This occurred at the north end of CUIS in the mid-1970s, forcing emerging loggerheads to crawl parallel to the dunes along the beach until they could find a lower access point to their nesting habitat (Ruckdeschel 1977).

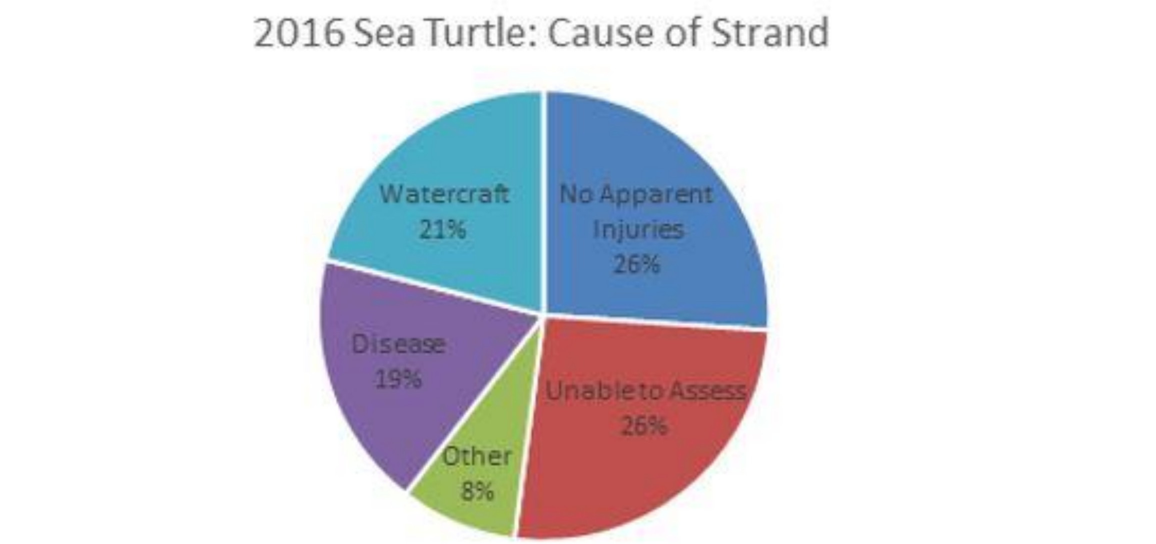
Figure 64. Dune erosion at CUIS caused by a spring storm in 2012 created an escarpment that would be insurmountable to nesting sea turtles (Peek et al. 2016).

Sea turtles may suffer injury or mortality as a result of boat strikes or interactions with commercial fishing gear (Camhi and Ehrenfeld 1986, GA DNR 2015a). Turtles are vulnerable to becoming entangled and drowning in large fishing nets, particularly those used by shrimp trawlers. This is particularly the case for loggerheads, whose feeding grounds overlap with productive shrimping areas (Camhi and Ehrenfeld 1986, Ruckdeschel and Shoop 1989). In the mid-1980s, an estimated 12,000 sea turtles were dying in fishing nets each year, and the Georgia Marine Turtle Stranding Network noted an increase in stranded turtles at CUIS in July, coinciding with the opening of shrimping operations in Georgia's nearshore waters (Camhi and Ehrenfeld 1986). Fortunately, the National Marine Fisheries Service (NMFS), in cooperation with the shrimp trawling industry, developed an effective turtle-excluder device (TED) that has been required on shrimp trawlers since the early 1990s (Figure 65) (Camhi and Ehrenfeld 1986, Alber et al. 2005). The proper use of TEDs can reduce incidental sea turtle catch by 97% (Camhi and Ehrenfeld 1986, Eayrs 2007). Since 1989, the Georgia DNR has noted a significant decline in loggerhead turtle strandings in the state (GA DNR 2017), which may be related to the use of TEDs. A decline in the state's shrimp trawling industry may also have contributed; total "trips" made by shrimp trawlers annually were around 7,000-8,000 during the 1990s but dropped to approximately 2,000 annual trips from 2006-2012 (GA DNR 2013). The decline in trawling may be linked to decreased catch, higher fuel costs, and competition (e.g., from farm-raised shrimp) (Hall 2013; Hoffman, written communication, October 2017).

Figure 65. A sea turtle escaping a fishing net through a TED (NMFS photo). Shrimp and other targeted species are small enough to pass through the metal bars, but turtles are stopped and can swim out through a loose mesh opening.

Each year, the Georgia DNR tracks the number of sea turtle strandings along the state's coast and attempts to determine the cause through necropsies. Data for each turtle is entered into NOAA's Sea Turtle Stranding and Salvage Network (STSSN) and the stranding database in Seaturtle.org. In 2016, 119 strandings were documented in Georgia, and over half of them (56%) were loggerheads (GA DNR 2017). Watercraft were implicated in 21% of strandings and disease for 19% (Figure 66). In 26% of strandings with "no apparent injuries", the cause of stranding is assumed by the GA DNR to be fishery-related mortality (see note in Figure 66). Other potential causes of stranding include poor condition due to lack of offshore food resources. Based on the appearance of live sea turtles that have washed ashore and the gastrointestinal (GI) tract conditions of dead stranded turtles at CUIS in recent years, there are some signs that turtles in the area are experiencing nutritional stress (Hoffman, personal communication, March 2017).

Sea turtle hatchlings emerge at night to avoid diurnal predators and lethally high sand temperatures (Camhi and Ehrenfeld 1986). Scientists believe that hatchlings find their way to the ocean by following the reflection of ambient light off the water (NMFS and USFWS 2008). Bright artificial lights in the vicinity of a nesting beach may misorient hatchlings and draw them away from the water, increasing the risk of predation, exposure, and desiccation (Camhi and Ehrenfeld 1986, Mays and Shaver 1998). At CUIS, hatchlings that emerge on moonless nights may be drawn to lights from the mainland or from Fernandina Beach to the south. Camhi and Ehrenfeld (1986) observed apparent misorientation in the tracks of hatchlings from three nests at CUIS in 1985.



* Animals categorized as "No apparent injuries" were found with no abnormalities, abundant fat and muscle mass, and food in the GI tract. They appear to have been completely healthy prior to death. These turtles are generally assumed to be fishery-related mortalities.

Figure 66. The causes for 119 sea turtle strandings along the Georgia coast in 2016 (GA DNR 2017).

Inundation or erosion due to storm surge or excessive rainfall may cause turtle nest failure (Camhi and Ehrenfeld 1986, Mays and Shaver 1998). During development, turtle eggs require a constant supply of oxygen; high moisture content in the surrounding sand may inhibit gas exchange, causing developing eggs to "suffocate" (Camhi and Ehrenfeld 1986, McGehee 1990). In September 1976, a single heavy rainfall event caused the partial failure of a loggerhead hatchery (i.e., relocated nests) on LCI (Kraemer and Bell 1980). Tropical Storm Fay impacted over 50% of CUIS sea turtle nests in August 2008 with long-term tidal inundation. More recently, in 2012, early season tropical storms Beryl and Debby produced such an unseasonably high amount of rainfall on CUIS that nearly all sea turtle nests deposited on beach flats were complete failures due to prolonged saturation (Hoffman, written communication, October 2017). Unusually high tides during hurricanes and tropical storms may simply wash away nests laid close to the beach. In addition, eroding or shifting sand dunes may bury nests so deep that hatchlings are unable to reach the surface (Mays and Shaver 1998). Due to the seasonality of storms in the southeast, late season nests (laid after 1 July) are more vulnerable to inundation than early season nests (Camhi and Ehrenfeld 1986).

Data Needs/Gaps

While loggerhead sea turtles have received long-term research attention at CUIS, information regarding the park's other herpetofauna are somewhat limited. The SECN amphibian monitoring program initiated in 2009 (Byrne et al. 2010, Smrekar et al. 2013) will continue to gather data regarding amphibian species richness, composition, and distribution and will help managers detect any trends in these populations. However, the SECN monitoring protocol is not intended to monitor salamanders. If these species become of interest, additional sampling techniques or other research projects may need to be added.

Study of the CUIS gopher tortoise population just began in 2015 (Moore 2016). Continued monitoring of the population and burrow counts will contribute to a better understanding of the species at the park. Given that the history of how gopher tortoises came to Cumberland Island is unknown, genetic testing and comparison to mainland populations should be conducted to provide insight into the population's origin (Moore 2016). The development of a management strategy and plan for the park's healthy, reproducing population would benefit the species as a whole, given the decline of tortoises in the rest of their range (Moore 2016).

Given the sensitivity of herpetofauna, particularly amphibians, to environmental change, research into the impacts of climate change on reptiles and amphibians would be beneficial (GA DNR 2015a). This includes the monitoring of climate change impacts on sea turtles and their nesting habitat.

<u>Overall Condition</u>

Amphibian Species Richness

The NRCA project team assigned this measure a *Significance Level* of 3. A total of 19 amphibian species have been confirmed as present at CUIS (NPS 2016f). Thirteen of these species have been documented by SECN monitoring in the past decade (Byrne et al. 2010, Smrekar et al. 2013). The majority of known species not found by SECN monitoring can be difficult to detect, particularly with ARDs. This measure is currently of low concern (*Condition Level* = 1).

Amphibian Abundance

The abundance measure was also assigned a *Significance Level* of 3. Information regarding amphibian abundance is limited to the results of SECN monitoring (Byrne et al. 2010, Smrekar et al. 2013). While the findings demonstrate that certain anurans are abundant at CUIS, there is not enough information to accurately assess the condition of the amphibian community as a whole. Therefore, a *Condition Level* has not been assigned.

Sea Turtle Species Richness

This measure was assigned a *Significance Level* of 1. Measures with a *Significance Level* of 1 are not discussed in the current condition section of the text, rather they are briefly summarized in the Overall Condition section. According to the NPSpecies list (NPS 2016f) and Tuberville et al. (2005), five species of sea turtle have been confirmed at CUIS. All five species are listed as federally threatened or endangered (Table 65). Three of these species nested on the park's beaches in 2015 and an additional species nested in 2017 (NPS 2016e, Seaturtle.org 2017). Only the loggerhead is considered common at CUIS; the remaining species are rare or occasional occurrences (NPS 2016f). At the present time, this measure is of no concern (*Condition Level* = 0).

Table 65. Sea turtle species documented within CUIS and their conservation status. T = threatened, E = endangered.

Scientific Name	Common Name	Fed Status	Tuberville et al. (2005)	Nesting (NPS 2016a, Seaturtle.org 2017)
Caretta caretta	loggerhead	T	x	x
Chelonia mydas	green sea turtle	T	x	x

Table 65 (continued). Sea turtle species documented within CUIS and their conservation status. T = threatened, E = endangered.

Scientific Name	Common Name	Fed Status	Tuberville et al. (2005)	Nesting (NPS 2016a, Seaturtle.org 2017)
Dermochelys coriacea	leatherback turtle	E	x	x
Eretmochelys imbricata	hawksbill	E	x	–
Lepidochelys kempii	Kemp's ridley	E	x	x

Sea Turtle Nesting Numbers

A *Significance Level* of 2 was assigned for this measure. While loggerhead nesting numbers at CUIS vary between years, an overall increasing trend is apparent over the past decade (Figure 61). Green and leatherback sea turtles also appear to be nesting more consistently at CUIS now than during the earlier decades of monitoring (NPS 2016e) and a Kemp's ridley turtle nest was documented on Cumberland Island for the first time since monitoring began (Seaturtle.org 2017). As a result, this measure is currently of low concern (*Condition Level* = 1).

Sea Turtle Hatch Success

Hatch success was assigned a *Significance Level* of 3. Since 2009, sea turtle hatch success on Cumberland Island has exceeded 70% in 6 of the 9 years and only fell below 65% in one year (2017) (Seaturtle.org 2017). Cumberland Island success rates consistently exceed statewide averages and are generally near or within Richardson's (1997) estimated range of hatch success under ideal conditions (70-85%). Lower success in 2017 due to high tides and storms should not cause long-term impacts, as long as it is an isolated event and rates rebound in coming years. Currently, this measure is of low concern (*Condition Level* = 1).

Gopher Tortoise Population Size

This measure was assigned a *Significance Level* of 1. It will be briefly summarized here rather than discussed in the current condition section. The only study of CUIS's gopher tortoise population documented 114 individuals, with over half of the individuals found at Stafford Field (Table 66) (Moore 2016). At three survey areas, only one tortoise was found. Tortoise density by study area ranged from 0.15-1.69 individuals/ha. In the two Stafford areas, where a controlled burn was conducted in early spring of 2016, the number of tortoises increased between 2015 and 2016. While it is unclear if the controlled burn contributed to the increase in tortoise numbers, it certainly seems that burning was not detrimental to the population. Moore (2016), p. 28 concluded that

> *The population of tortoises on Cumberland Island appears to be robust, reproductively active, and contain all age classes. These data suggest that the population of gopher tortoises on Cumberland Island is healthy when compared to other populations of tortoises elsewhere.*

Therefore, this measure is assigned a *Condition Level* of 0, indicating good condition.

Table 66. The number and density (individuals/ha) of gopher tortoises documented at CUIS by study area (Moore 2016). Two entries are included for Stafford Field and Woods, as they were surveyed in both years of the study.

Location	Area (ha)	# of Tortoises	Density
Bathtub Field	6.9	1	0.15
Davisville	25.3	10	0.30
Greyfield	10.0	3	0.30
South Dunes	5.3	1	0.19
Ice House Dock Field	4.3	1	0.23
Stafford Woods 2015	22.2	5	0.23
Stafford Woods 2016	22.2	18	0.81
Stafford Field 2015	47.4	75	1.58
Stafford Field 2016	47.4	80	1.69


Gopher tortoises from CUIS's Stafford Field population (Photos by John Enz, Jacksonville University).

Gopher Tortoise Burrow Count

The burrow count measure was also assigned a *Significance Level* of 1. Gopher tortoises use their burrows for nesting, thermoregulation, and protection from predators (Moore 2016). Tortoises may dig multiple burrows and move between them frequently (Moore 2016). Some tortoises have been found using three burrows at once. At CUIS, two of the tortoises captured by Moore (2016) were observed moving to different burrows.

Moore (2016) documented the total number of gopher tortoise burrows along with the number of occupied burrows (tortoise present), active burrows (showed signs of use but unoccupied), and inactive burrows (not in use – spiderwebs, debris, or other obstructions present). Outside of the Stafford area, only 30 burrows were found, with all but one being occupied or active (Table 67). In 2015, 280 total burrows were documented in Stafford Field and Woods. Eighty of the burrows were occupied and an additional 67 were considered active (Moore 2016). In 2016, the total number of burrows in the two Stafford areas increased to 296, with 98 occupied and 72 active. As with the previous measure, tortoise burrow count is currently considered in good condition (*Condition Level* = 0).

An active gopher tortoise burrow at Stafford Field (SMUMN GSS photo).

Table 67. The number of total, occupied, active, and inactive burrows by CUIS study area (Moore 2016).

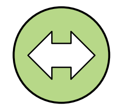
Station	Total Burrows	Occupied Burrows	Active Burrows	Inactive Burrows
Stafford Field 2015	252	75	50	127
Stafford Field 2016	259	80	60	119
Stafford Woods 2015	28	5	17	6
Stafford Woods 2016	37	18	12	7
Bathtub Field	2	1	1	0
Greyfield	6	3	3	0
Davisville	14	10	4	0
South Dunes	1	1	0	0
Ice House Dock Field	7	1	5	1

Weighted Condition Score

The *Weighted Condition Score* for CUIS herpetofauna is 0.24, indicating good condition (Table 68).

Given the limited information regarding amphibian abundance and the gopher tortoise population, a moderate confidence border has been applied.

Table 68. Weighted Condition Score for Herpetofauna in CUIS.

Herpetofauna			
Measures	**Significance Level**	**Condition Level**	**WCS = 0.24**
Amphibian Species Richness	3	1	
Amphibian Abundance	3	n/a	
Sea Turtle Species Richness	1	0	
Sea Turtle Nesting Numbers	2	1	
Sea Turtle Hatching Success	3	1	
Gopher Tortoise Population Size	1	0	
Gopher Tortoise Burrow Count	1	0	

4.7.6. Sources of Expertise

Mike Byrne, SECN Terrestrial Ecologist

Doug Hoffman, CUIS Biologist

4.8. Water Quality (Freshwater)

4.8.1. Description

Water quality and quantity influence nearly all aspects of wetland and aquatic ecosystems, from vegetation and soils to wildlife, particularly sensitive species such as fish, aquatic invertebrates, and amphibians (UNEP 2008). Impaired water quality can alter plant and animal species composition, health, and reproduction (UNEP 2008, USGS 2016d). As an island surrounded by saltwater, freshwater sources are especially valuable at CUIS. However, the surface water quality of barrier islands in the Southeast is typically variable, ranging from freshwater ponds and sloughs completely isolated from the ocean to brackish and saline water bodies with intermittent connections to seawater (Bellis 1995, Frick et al. 2002). These variations are influenced by: 1) the water body's proximity to the ocean (e.g., degree of tidal or storm surge influence), 2) its interactions with the groundwater table and other surface waters, and 3) long-term and recent precipitation patterns (Frick et al. 2002).

Freshwater systems typically have a salinity of 0.5 ppt or less (EPA 2006). At CUIS, freshwater resources include ponded wetlands, streams above tidal influence, and a small number of seeps. Some of the surface waters are fed by groundwater from a shallow water table, while others are recharged primarily by precipitation (Frick et al. 2002). The water quality of these surface waters varies seasonally, primarily in response to the frequency and amount of rainfall (Kozel 1991, Frick et al. 2002). For water bodies closer to the beach, drastic changes in water quality can occur following storm events if saltwater inundation occurs due to storm and tidal surges (Frick et al. 2002).

Old Swamp Field, a freshwater wetland at CUIS (NPS photo).

4.8.2. Measures

- Nutrients
- Fecal coliform bacteria
- Salinity
- Dissolved oxygen
- pH
- Specific conductance

Nutrients

Nutrients, such as nitrogen and phosphorus, are crucial in supporting healthy aquatic environments. However, elevated concentrations of these nutrients can negatively impact water quality and threaten the ability of plants and aquatic organisms to thrive (USGS 2016a). Nitrogen occurs naturally in the atmosphere and in soils and is deposited into surface waters through precipitation and runoff; nitrogen deposition is increased by human inputs such as sewage, fertilizers, and livestock waste (USGS 2017b). Nitrate (NO_3) can cause a host of water quality related problems when present in high concentrations including, but not limited to, excessive plant and algae growth, eutrophication, and depleted dissolved oxygen available to aquatic organisms (USGS 2017b). Nitrate in drinking water can be harmful to humans, particularly young children, and livestock (USGS 2017b). Phosphorus is commonly found in agricultural fertilizers, manure, organic wastes in sewage, and sometimes industrial effluent (USGS 2016b). In excess, phosphorus in water systems can increase the rate of eutrophication, encourage overgrowth of aquatic plants, deplete dissolved oxygen, and threaten fish and macroinvertebrate populations (USGS 2016b).

Fecal Coliform Bacteria

Bacteria are a common natural component of surface waterways and are mostly harmless to humans. However, certain bacteria, specifically those found in the intestinal tracts and feces of warm-blooded animals, can cause illness in humans (USGS 2011). Fecal coliform bacteria are a subgroup of coliform bacteria that, when used in monitoring water quality, can indicate if fecal contamination has occurred in a specific waterway. It is often tested by counting bacterial colonies that grow on filters placed in an incubator for 22-24 hours. High concentrations of certain fecal coliform, such as *E. coli*, can cause serious illness in humans (USGS 2011).

Salinity

Salinity is the measure of dissolved salts in water, usually reported in parts per thousand (ppt) (EPA 2006). The level of salinity also controls the types of organisms (plants and animals) that can survive in the body of water. Some species, such as smooth cordgrass, can withstand higher levels of salinity, while other species only tolerate lower salinity levels (EPA 2006). Chemical methods for measuring salinity can be time-consuming and inconvenient, so salinity is often calculated from measurements of conductivity or total dissolved solids (TDS), as higher salinity levels lead to higher TDS and conductivity (EPA 2006).

Dissolved Oxygen

Dissolved oxygen (DO) is critical for organisms that live in water. In order to survive, fish and zooplankton filter out or "breathe" DO from the water (USGS 2016d). Oxygen enters water from the air, when atmospheric oxygen mixes with water at turbulent, shallow riffles in a waterway, or when released by algae and other plants as a byproduct of photosynthesis. As the amount of DO drops, it becomes more difficult for aquatic organisms to survive (USGS 2016d). According to the EPA (2016d), waters with DO levels below 1 mg/l are typically hypoxic and devoid of life. The concentration of DO in a water body is closely related to water temperature; cold water holds more DO than warm water (USGS 2016d). Thus, DO concentrations are subject to seasonal fluctuations as low temperatures in the winter and spring allow water to hold more oxygen, and warmer temperatures in the summer and fall allow water to hold less oxygen (USGS 2016d).

pH

pH is a measure of the level of acidity or alkalinity of water and is measured on a scale from 0 to 14, with 7 being neutral (USGS 2016d). Water with a pH of less than 7.0 indicates acidity, whereas water with a pH greater than 7.0 indicates alkalinity. Alkalinity and acidity are determined by the relative amount of free hydrogen (H^+) and hydroxyl (OH^-) ions in a liquid; more H^+ ions make a liquid acidic while more OH^- make it alkaline (USGS 2016d). Aquatic organisms have a preferred pH range that is ideal for growth and survival. Chemicals in water can change the pH and harm animals and plants living in the water; thus, monitoring pH can be useful for detecting natural and human-caused changes in water chemistry (USGS 2016d).

Specific Conductance

Specific conductance (SpC) is a measure of the ability of water to conduct electrical current, which depends largely on the amount of dissolved ions in the water (Allan and Castillo 2007). Water with low amounts of dissolved ions (such as purified or distilled water) will have a low SpC, while water with high amounts of dissolved solids (such as sea water) will have a higher SpC (Allan and Castillo 2007). SpC is an important water quality parameter to monitor because high levels can indicate that water is unsuitable for drinking or aquatic life (USGS 2016d). The SECN uses SpC observations to calculate salinity values during water quality monitoring (Rinehart et al. 2013).

4.8.3. Reference Condition/Values

Because the fresh waters of CUIS have high natural variability, the NRCA project team did not think that EPA or Georgia state water quality standards would be appropriate reference conditions. Rather, the range of values from Frick et al. (2002) will serve as a reference or baseline for future assessments.

4.8.4. Data and Methods

The majority of water quality monitoring efforts at CUIS have focused on estuarine waters (Gregory et al. 2010, Wright et al. 2012, 2013), with little attention given to the quality of freshwater. From April 1988 through May 1990, Kozel (1991) monitored the surface water quality and fish fauna of three interdunal ponds on the south end of CUIS (the "South End Ponds"). The study was initiated as part of a larger effort to evaluate the impacts of channel dredging on park resources and included the monthly monitoring of 14 water chemistry parameters. Two of the ponds had intermittent connections to Cumberland Sound, causing brackish to saline conditions, while one pond was isolated from the Sound (Kozel 1991). Three sampling points were selected in each pond. Since this NRCA component focuses on freshwater, only the results from the isolated pond were evaluated.

The U.S. Geological Survey (USGS) and NPS cooperated to conduct a water quality study at CUIS from April 1999 to March 2000 (Frick et al. 2002). Both ground and surface water sources were sampled, including six freshwater wetlands that supported standing water for the majority of the year. One of the sampling locations, South End Pond 3, was the same freshwater pond sampled by Kozel (1991). Parameters measured included temperature, pH, DO, SpC, nutrients, and major ions (Frick et al. 2002). Each wetland was visited one to four times during the survey period; one wetland, the Lake Retta complex, was sampled at two different locations (Figure 67). These surface waters are not used for drinking water, but Frick et al. (2002) noted exceedances of EPA primary and secondary standards for drinking water, to have some point of comparison. The researchers noted that during the sampling period, precipitation was nearly 33 cm (12.9 in) below the 30-year average, which likely influenced their findings (Frick et al. 2002).

Figure 67. USGS and NPS cooperative water quality study sampling sites (Frick et al. 2002).

4.8.5. Current Condition and Trend

<u>Nutrients</u>

Kozel (1991) sampled phosphate (PO_4) in the isolated South End pond from April 1988 through May 1990. Several elevated levels were detected, which suggest eutrophic conditions (i.e., excessive nutrients). Phosphates peaked during the summer, as water levels dropped and dissolved constituents became more concentrated (Figure 68) (Kozel 1991). Levels spiked again in late fall and winter, likely due to nutrient release from aquatic plant and phytoplankton decomposition. According to Kozel (1991), high nitrate and nitrite levels were also measured in the pond.

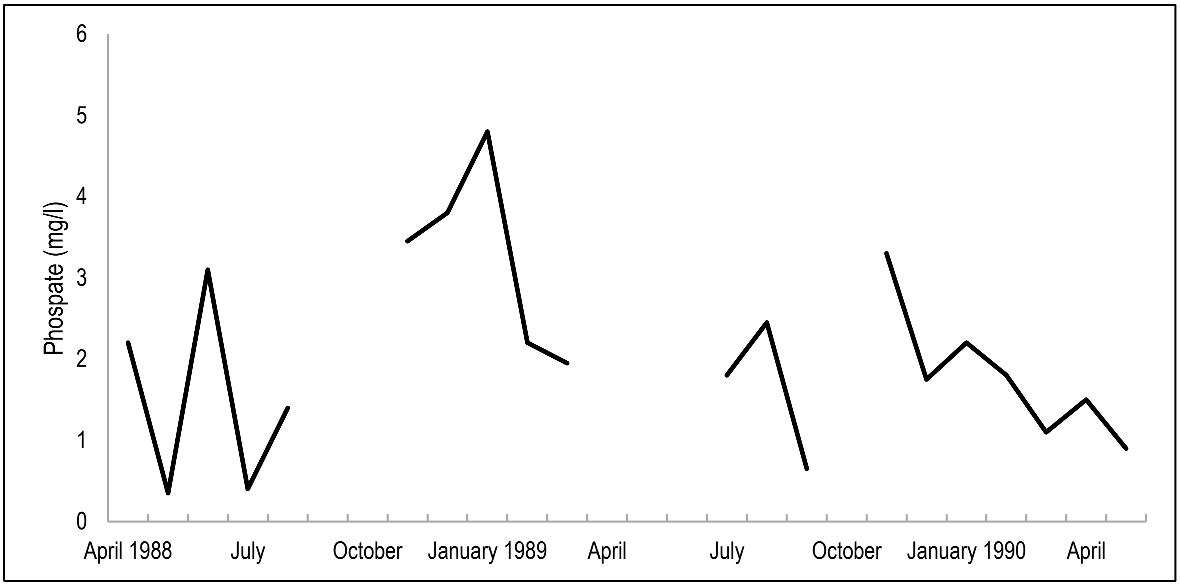


Figure 68. Total phosphate measurements from the isolated South End pond, April 1988-May 1990 (recreated from Kozel 1991).

Frick et al. (2002) sampled both nitrogen (as nitrate and ammonia) and phosphorous (as orthophosphorus) levels in CUIS freshwater wetlands during April, October, and December of 1999. Nitrogen levels were generally low across sampling locations, with many observations <0.20 mg/l (Table 69). The one slightly elevated measurement (2.0 mg/l) from Lake Retta in December 1999 was likely due to the concentrating effects of low water levels (Frick et al. 2002).

Phosphorous levels were also generally low in CUIS wetlands, with the majority of observations <0.20 mg/l (Table 70). However, one extremely elevated measurement (32.6 mg/l) was recorded at South End Pond 3 in October 1999 (Frick et al. 2002). This was just weeks after Hurricane Floyd triggered storm surges and coastal flooding along the East Coast, including at CUIS. Frick et al. (2002) theorized that storm surge from the hurricane may have washed horse manure or another source of phosphates into the pond.

Table 69. Nitrogen levels, as nitrate and ammonia, in CUIS water bodies (Frick et al. 2002). ns = not sampled.

Sites	Nitrate, Dissolved (mg/l as N)			Ammonia (mg/l as N)
	April 1999	October 1999	December 1999	April 1999
North Cut Pond 2A	<0.02	<0.02	0.20	0.013
Whitney Lake	0.04	<0.02	<0.20	0.042
Lake Retta	0.10	<0.02	2.0	0.019
Lake Retta complex at foot bridge on Willow Pond Trail	ns	<0.02	ns	ns
Lake Retta complex 128 m (420 ft) south of foot bridge	0.04	ns	ns	0.013
South End Pond 3	<0.02	<0.02	ns	0.825

Table 70. Phosphorus levels, as orthophosphorus, in CUIS water bodies (Frick et al. 2002). ns = not sampled.

Site	Orthophosphorus, Dissolved (mg/l as P)		
	April 1999	October 1999	December 1999
North Cut Pond 2A	<0.02	0.013	<0.20
Whitney Lake	0.111	0.104	0.065
Lake Retta	0.082	0.391	<0.20
Lake Retta complex at foot bridge on Willow Pond Trail	ns	0.31	ns
Lake Retta complex 128 m (420 ft) south of foot bridge	0.059	ns	ns
South End Pond 3	<0.02	32.6	ns

Fecal Coliform Bacteria

Fecal coliform bacteria levels have not been measured in any CUIS freshwater sources (Frick et al. 2002). However, coliform bacteria contamination is a serious concern for water bodies on the island, given their frequent use by feral horses and hogs (Noon and Martin 2004). The State of Georgia's bacterial water quality standard for recreational waters is, "Culturable *E. coli* not to exceed a geometric mean of 126 CFU (colony forming units) per 100 mL," and "no greater than a ten percent excursion frequency of an *E. coli* statistical threshold value (STV) of 410 CFU per 100 mL in the same 30-day interval" (GA DNR 2015b).

Salinity

Kozel (1991) reported salinity levels for the isolated South End pond. Salinity was <1 ppt (parts per thousand) in the majority of samples, but did rise to nearly 3 ppt during the summer of 1989 when water levels dropped. Salt spray from nearby Cumberland Sound was likely a contributing factor (Kozel 1991).

Salinities for five of the six freshwater wetlands sampled by Frick et al. (2002) remained below 2 ppt for the entire study period. The only wetland with higher salinity was South End Pond 3, where measurements ranged from 26-45 ppt, most likely due to saltwater encroachment into the shallow aquifer. Concentrations of the major constituents (ions) that contribute to salinity in South End Pond 3 are presented in Table 71.

Table 71. Salinity (ppt) in CUIS water bodies (Frick et al. 2002).

Site	Salinity Range (ppt)
North Cut Pond 2A	<2
Whitney Lake	<2
Lake Retta	<2
Lake Retta complex at foot bridge on Willow Pond Trail	<2
Lake Retta complex 128 m (420 ft) south of foot bridge	<2
South End Pond 3	26-45

Frost et al. (2011) reported salinity levels for several CUIS wetlands during 2010. Salinity in a sawgrass wetland just north of South Cut Road and near the eastern wooded edge of the Whitney Lake complex was 0.45% (4.5 ppt). The measurement from Willow Pond south of the trail contained 0.5% salinity (5 ppt) (Frost et al. 2011).

<u>Dissolved Oxygen</u>

Frick et al. (2002) is the only available study that documented DO levels in freshwater sources at CUIS. Concentrations ranged from <0.5 mg/l at South End Pond 3 in April 1999 to 9.8 mg/l, also in South End Pond 3 in March 2000 (Table 72). As mentioned previously, waters with DO levels below 1 mg/l are typically hypoxic and devoid of life (EPA 2016d). Other water bodies showed a narrower range but were still highly variable across seasons. For example, in Whitney Lake, the island's largest and most permanent freshwater body, DO observations varied from 2.8-6.8 mg/l (Frick et al. 2002). The lowest levels of DO in each water body occurred in the fall with the highest observations in the winter or spring. This pattern may be related to water temperatures, as cooler water holds more oxygen than warmer water (USGS 2016d). In smaller water bodies that shrink and dry up over the summer months, the biological process driving decomposition likely used up more of the available DO, which also contributed to the observed lower levels (Brian Gregory, SECN Program Manager/Aquatic Ecologist, written communication, July 2017).

Table 72. Dissolved oxygen levels (mg/l) in CUIS water bodies (Frick et al. 2002). ns = not sampled.

	Dissolved Oxygen (mg/l)			
Site	**April 1999**	**October 1999**	**December 1999**	**March 2000**
North Cut Pond 2A	6.8	1.7	5.2	–*
Whitney Lake	3.1	2.8	6.8	5.7
Willow Pond	ns	ns	ns	2.0
Lake Retta	7.5	1.5	2.0	4.2
Lake Retta complex at foot bridge on Willow Pond Trail	ns	1.0	ns	ns
Lake Retta complex 128 m (420 ft) south of foot bridge	3.4	ns	ns	ns
South End Pond 3	<0.5	ns	8.2	9.3

* North Cut Pond 2A was dry in March 2000.

pH

Kozel (1991) measured pH in the isolated South End Pond several times between April 1988 and May 1990. pH ranged from a low just above 4.0 during early fall of 1989 to around 9.0 in late summer of 1988 and again in spring of 1989 (Kozel 1991). The lowest value was observed after a rainfall while the higher values were noted during periods with greater photosynthesis by plants. Kozel (1991) noted that pH was lower and more variable in the isolated pond than in the nearby saline/estuarine ponds.

Observations of pH from CUIS freshwater wetlands by Frick et al. (2002) ranged from 4.3 (North Cut Pond 2A) to 8.0 (South End Pond 3) (Table 73). Readings from North Cut Pond 2A were consistently lower than at other locations, while South End Pond 3 tended to have higher observations than other locations. pH levels were relatively consistent at Whitney Lake (5.3-5.8) and Lake Retta (6.9-7.3) (Frick et al. 2002). Factors contributing to the lower pH levels of North Cut Pond and Whitney Lake could include low pH rainwater and the decomposition of aquatic vegetation (Frick et al. 2002).

Specific Conductance

Measurements of SpC by Frick et al. (2002) were highly variable across CUIS water bodies. Observations from North Cut Pond 2A were consistently below 150 µS/cm while readings from South End Pond 3 were all above 33,000 µS/cm (Table 74). The remaining water bodies fell between these two extremes, with ranges between 170 and 310 µS/cm for Whitney Lake and Willow Pond, and a range of 628 to 1,040 µS/cm for Lake Retta (Frick et al. 2002).

Table 73. pH observations from CUIS water bodies (Frick et al. 2002). ns = not sampled.

Site	pH			
	April 1999	October 1999	December 1999	March 2000
North Cut Pond 2A	4.3	4.3	4.5	–*
Whitney Lake	5.3	5.4	5.5	5.8
Willow Pond	ns	ns	ns	5.9
Lake Retta	7.3	6.9	7.3	6.9
Lake Retta complex at foot bridge on Willow Pond Trail	ns	6.4	ns	ns
Lake Retta complex 128 m (420 ft) south of foot bridge	7.0	ns	ns	ns
South End Pond 3	8.0	6.3	7.6	8.0

* North Cut Pond 2A was dry in March 2000.

Table 74. SpC observations (µS/cm) from CUIS water bodies (Frick et al. 2002). ns = not sampled.

Site	Specific Conductance (µS/cm)			
	April 1999	October 1999	December 1999	March 2000
North Cut Pond 2A	99	110	141	–*
Whitney Lake	179	208	215	240
Willow Pond	ns	ns	ns	308
Lake Retta	640	628	1,040	986
Lake Retta complex at foot bridge on Willow Pond Trail	ns	647	ns	ns
Lake Retta complex 128 m (420 ft) south of foot bridge	370	ns	ns	ns
South End Pond 3	56,000	33,300	43,100	37,600

* North Cut Pond 2A was dry in March 2000.

Threats and Stressor Factors

Threats to CUIS's freshwater water quality include feral horse and hog activity, atmospheric deposition, eutrophication, saltwater intrusion, roads and trails, abandoned artesian wells, and fires (wildfires and controlled burns). Roads and trails at CUIS have altered the island's hydrology (surface runoff patterns, wetland connectivity, etc.) which, in turn, impacts water quality (Hillestad et al. 1975, Alber et al. 2005, DeVivo et al. 2008). Controlled burns are unlikely to impact water quality, as they are generally well-planned to minimize environmental impacts, but a sudden and unexpected rainfall or wind shift could deposit ash, embers, or other debris into freshwater bodies. The unplanned disturbance to vegetation and soils from wildfire can impact physical water quality parameters (temperature, DO, sediment levels) as well as biological and chemical characteristics (Baker 1990, Smith et al. 2011). For example, nutrients and minerals in the ash from burned vegetation can be blown or washed into nearby waterbodies (Baker 1990, Smith et al. 2011).

The feral wildlife populations at CUIS are among the greatest threats to the island's freshwater quality. Feral horses frequently graze in freshwater wetlands and visit ponds to drink (Noon and Martin 2004). While grazing in wetlands, horses can stir up sediment which becomes suspended in the water, increasing turbidity (Noon and Martin 2004). Increased suspended sediment can alter water temperatures and other parameters, and may inhibit the ability of aquatic plants and organisms to respire or breathe. Horse waste contributes large amounts of nutrients to the wetlands and freshwater bodies of CUIS. On average, a horse produces 16-20 kg (36-45 lbs) of waste (liquid and solid) per day (EPA 2001, Noon and Martin 2004). With a population of around 150 horses currently at CUIS, this amounts to 876,000-1.1 million kg (1.9-2.5 million lbs) of horse waste annually. This results in approximately 6,800 kg (15,000 lbs) of nitrogen and 2,600 kg (5,700 lbs) of phosphorous deposited per year on the island (Noon and Martin 2004). When the organic matter in the solid wastes decomposes in water, oxygen is consumed, lowering the DO levels and potentially starving other aquatic life of oxygen (Noon and Martin 2004). Also, horse waste may contain pathogenic microorganisms, including virulent strains of *E. coli* bacteria (EPA 2001, Noon and Martin 2004). Feral hogs and their rooting behavior pose similar threats to water quality (Kaller and Kelso 2006, Kammermeyer et al. 2011), but perhaps on a smaller scale given their smaller body size and the current relatively small population on the island.

The addition of nutrients such as nitrogen and phosphorous to surface water bodies often causes eutrophication (USGS 2016b, 2017b). Eutrophication triggers excess algal growth in water bodies. As the algae die and decompose, oxygen becomes depleted in the water and may drop to levels where aquatic organisms can no longer survive (USGS 2017a). Some algae may also produce toxins or promote bacterial growth that can harm aquatic life (Paerl et al. 2001, Noon and Martin 2004). At CUIS, water bodies that have been described as eutrophic are Lake Retta (Hillestad et al. 1975), South End Pond 3 (Kozel 1991), Plum Orchard Pond (Hillestad et al. 1975), and Whitney Lake (Frick et al. 2002). Some of these waters may be naturally eutrophic (Gregory, written communication, July 2017). The high productivity of aquatic plants in a eutrophic system can cause a water body or wetland to fill over time, eventually reducing the amount of standing water it can hold (Kozel 1991, Callisto et al. 2014).

Research has shown that sulfate enrichment can also contribute to eutrophication, as chemical reactions during sulfate reduction mobilize (i.e., make available) phosphate and ammonium ions in the water (Lamers et al. 1998). Lamers et al. (1998) found that increased sulfate availability in freshwater wetlands led to an increase in phosphate release from the soil. Sulfate enrichment is a concern at CUIS due to the presence of several abandoned artesian wells that tap into the deep Floridian aquifer, which has a relatively high concentration of hydrogen sulfide (Frick et al. 2002, Alber et al. 2005). The water from the artesian wells, which pre-date the establishment of CUIS, typically produce a sulfide smell. These abandoned and unmaintained wells began flowing uncontrollably after the Gilman Paper Company in St. Marys closed and ceased its groundwater withdrawals in October 2002 (Alber et al. 2005). Four of the wells were capped or plugged in 2011, but two of the remaining wells are directly affecting two of the island's freshwater ponds: Plum Orchard Duck Pond and a pond in the Willow Pond Complex (Fry, written communication, July

2017). Although the impact of this artesian flow is unknown, park management is concerned about the potential influence on water quality and freshwater wetlands (Alber et al. 2005).

Saltwater intrusion is also a significant threat to the park's freshwaters, particularly those near the shoreline. Saltwater can reach freshwater bodies on the island either over land through high tides and storm surges or by encroaching into the shallow surficial water table (Hillestad et al. 1975, Frick et al. 2002). Salt spray from the ocean (i.e., airborne salt particles) can also reach wetlands and contribute to salinity (Frick et al. 2002). Overland saltwater intrusion was believed to have contributed to elevated concentrations of chlorine, sulfate, and TDS in South End Pond 3 during Frick et al.'s (2002) sampling period. Frick et al. (2002) found that the major ion composition of South End Pond 3 was similar to seawater, and at one point the TDS measurement of 39,700 mg/l was higher than the average TDS of seawater (32,800 mg/l). A sudden increase in salinity from saltwater intrusion can kill off aquatic vegetation not adapted to a saline or brackish environment (Hillestad et al. 1975). If sea levels continue to rise as predicted (IPCC 2013), overland saltwater intrusion is likely to become an even greater threat to CUIS freshwaters (Ataie-Ashtiani 2013).

Atmospheric wet deposition (e.g., rain, snow, fog) in the region around CUIS is likely acidic. The mean annual pH of wet deposition measured at Sapelo Island, GA (approximately 47 km [29.2 mi] north of CUIS) from 2012-2014 was around 5.0 (Figure 69) (NADP 2016c). pH means were similar at a station 62 km (38.5 mi) west of the park at Okefenokee National Wildlife Refuge (NTN Site GA09) (NADP 2016b).

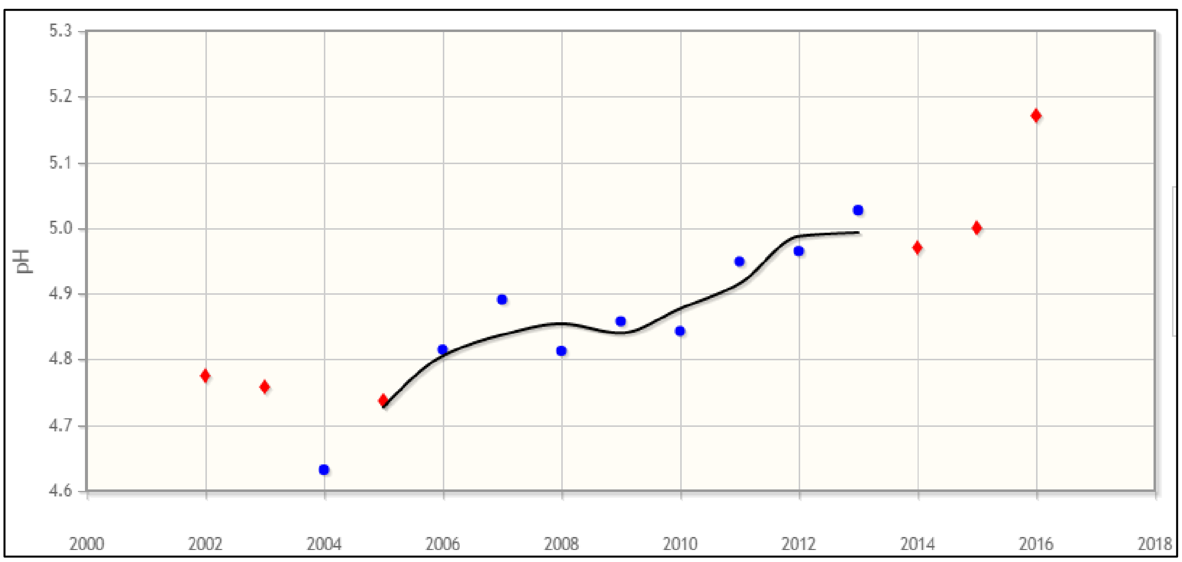


Figure 69. Annual mean pH of wet atmospheric deposition at NTN Site GA33 on Sapelo Island (approximately 47 km [29.2 mi] north of CUIS) (NADP 2016c). Red diamonds represent years when NADP's data completeness criteria (valid samples and precipitation amounts for 75% of time period) were not met.

If the primary source of water for a wetland or pond at CUIS is rainfall, the low pH of precipitation/deposition may be retained in the park's surface waters. It is unclear if the acidic nature of regional wet deposition is due to natural or anthropogenic causes. Human-related contributors to acidic deposition include motor vehicles, electric power generation (e.g., coal-burning facilities), and industrial/chemical plants (NADP 2014).

Data Needs/Gaps

As noted previously, little attention has been given to documenting the quality of surface freshwater sources within CUIS. Additional data are needed for all the measures selected by the NRCA team so that changes can be detected in these areas that are vulnerable to feral wildlife impacts, climate change, and saltwater intrusion (NPS 2014a). Water quantity has a significant influence on water quality at CUIS. When water volumes and flows are higher, dissolved constituents are more diluted and are less likely to have detrimental effects on aquatic life (Whitehead et al. 2009, Mosley 2015). A study of water quantities and the hydrology of freshwater sources on the island (e.g., stream and spring flow) would contribute significantly to a better understanding of park water quality. In addition, gathering data from other barrier island freshwater systems similar to CUIS would offer points for comparison and would help to better define reference conditions.

Overall Condition

The NRCA project team assigned *Significance Levels* of 3 to the nutrients, fecal coliform bacteria, and salinity measures. The DO, pH, and SpC measures were assigned *Significance Levels* of 2. Because of the lack of data for surface water quality at CUIS, *Condition Levels* could not be assigned for any of these measures.

Weighted Condition Score

A *Weighted Condition Score* was not calculated for CUIS water quality since *Condition Levels* could not be assigned to any of the selected measures. The current condition and any trends in the park's freshwater water quality are unknown (Table 75).

Table 75. Weighted Condition Score for Water Quality in CUIS.

Water Quality			
Measures	**Significance Level**	**Condition Level**	**WCS = N/A**
Nutrients	3	n/a	
Fecal Coliform Bacteria	3	n/a	
Salinity	3	n/a	
Dissolved Oxygen	2	n/a	
pH	2	n/a	
Specific Conductance	2	n/a	

4.8.6. Sources of Expertise

Brian Gregory, SECN Program Manager/Aquatic Ecologist

4.9. Air Quality

4.9.1. Description

Air pollution can significantly affect natural resources, their associated ecological processes, cultural resources, and the health of park visitors. In the Clean Air Act (CAA), Congress set a national goal "to preserve, protect, and enhance the air quality in national parks, national wilderness areas, national monuments, national seashores, and other areas of special national or regional natural, recreational, scenic or historic value" (42 U.S.C. §7470(2)). This goal applies to all units of the NPS. The act includes special provisions for 48 park units, called "Class I" areas under the CAA; all other NPS areas are designated as Class II, including CUIS. For Class II airsheds, the increment ceilings for additional air pollution above baseline levels are slightly greater than for Class I areas which can allow for more development (NPS 2004). Additional authority to consider and protect air quality in Class II parks is provided by Title 54 (54 USC 100101(a) et seq.), commonly known as the NPS Organic Act.

Parks designated as Class I and II airsheds typically use the EPA's National Ambient Air Quality Standards (NAAQS) for criteria air pollutants as the ceiling standards for allowable levels of air pollution. EPA standards are designed to protect human health and the health of natural resources (EPA 2016e). To comply with CAA and NPS Organic Act mandates, the NPS established a monitoring program that measures air quality trends in many park units for key air quality indicators, including atmospheric deposition, ozone, and visibility (NPS 2008). In addition, the SECN has identified ozone, wet and dry deposition, and visibility and particulate matter as a Vital Signs for all network parks, including CUIS (DeVivo et al. 2008).

4.9.2. Measures

- Nitrogen deposition
- Sulfur deposition
- Mercury deposition
- Ozone
- Visibility

<u>Atmospheric Deposition of Sulfur and Nitrogen</u>

Sulfur and nitrogen are emitted into the atmosphere primarily through the burning of fossil fuels, industrial processes, and agricultural activities (EPA 2012a). While in the atmosphere, these emissions form compounds that may be transported long distances, eventually settling out of the atmosphere in the form of pollutants such as particulate matter (e.g., sulfates, nitrates, ammonium) or gases (e.g., nitrogen dioxide, sulfur dioxide, nitric acid, ammonia) (NPS 2008, EPA 2012a). Atmospheric deposition can be in wet (i.e., pollutants dissolved in atmospheric moisture and deposited in rain, snow, low clouds, or fog) or dry (i.e., particles or gases that settle on dry surfaces as with windblown dusts) form (EPA 2012a). Deposition of sulfur and nitrogen can have significant effects on ecosystems including acidification of water and soils, excess fertilization or increased eutrophication, changes in the chemical and physical characteristics of water and soils, and accumulation of toxins in soils, water and vegetation (NPS 2008, reviewed in Sullivan et al. 2011a, 2011c). The acidic nature of nitrogen and sulfur deposition can also contribute to the deterioration of stone in historic structures (Charola 1998).

Atmospheric Deposition of Mercury

Sources of atmospheric mercury (Hg) include anthropogenic sources such as fuel combustion and evaporation (especially coal-fired power plants), waste disposal, mining, industrial sources, along with natural sources such as volcanoes and evaporation from enriched soils, wetlands, and oceans (EPA 2008). Atmospheric deposition of mercury from coal-burning power plants has been identified as a major source of mercury to remote ecosystems (Landers et al. 2008). Mercury is a potential problem for ecosystems in regions with heavy current or historic coal use.

Mercury deposited into rivers, lakes, and oceans can accumulate in various aquatic species, resulting in exposure to wildlife and humans that consume them (EPA 2008). Mercury exposure can cause liver, kidney, and brain (neurological and developmental) damage (EPA 2008). High mercury concentrations in birds, mammals, and fish can result in reduced foraging efficiency, survival, and reproductive success (Mast et al. 2010, Eagles-Smith et al. 2014).

Ozone

Ozone (O_3) occurs naturally in the earth's upper atmosphere where it protects the earth's surface against ultraviolet radiation (EPA 2012a). However, it also occurs at the ground level (i.e., ground-level ozone) where it is created by a chemical reaction between nitrogen oxides (NO_x) and volatile organic compounds (VOCs) in the presence of heat and sunlight (NPS 2008). Ozone precursors are emitted from both anthropogenic and natural source types, including power plants, industry, motor vehicles, oil and gas development, forest fires, and other sources (EPA Beitler 2006, 2008).

Ozone is one of the most widespread pollutants affecting vegetation in the U.S. (NPS 2008). Considered phytotoxic, ozone can cause significant foliar injury and growth defects for sensitive plants in natural ecosystems. Specific defects include reduced photosynthesis, premature leaf loss, and reduced biomass; prolonged exposure can increase vulnerability to insects and diseases or other environmental stresses (NPS 2008). Plant species occurring in CUIS that are known to be sensitive to ozone include loblolly pine, sweetgum (*Liquidambar styraciflua*), Virginia creeper (*Parthenocissus quinquefolia*), and smooth cordgrass (Kohut 2004).

At high concentrations, ozone can aggravate respiratory and cardiovascular diseases in humans through reduced lung function, increased acute respiratory problems, and elevated susceptibility to respiratory infections (EPA 2016c). Visitors and staff engaging in aerobic activities in the park (e.g., hiking, biking, maintenance/physical labor), as well as children, the elderly, and people with heart and lung diseases are especially sensitive to elevated ozone levels.

Visibility (Particulate Matter)

Air pollution, especially particulate matter (PM), influences a visitor's ability to view scenic vistas and landscapes at parks (NPS 2007). PM is a complex mixture of extremely small particles and liquid droplets that become suspended in the atmosphere. It largely consists of acids (such as nitrates and sulfates), organic chemicals, metals, and soil or dust particles (EPA 2016f). In coastal areas, salt spray also contributes PM and can impact visibility (Lewis and Schwartz 2004). There are two particle size classes of concern: $PM_{2.5}$ – fine particles found in smoke and haze, which are 2.5 micrometers or less in diameter; and PM_{10} – coarse particles found in wind-blown dust, which have diameters between 2.5 and 10 micrometers (EPA 2012a). Fine particles are a major cause of reduced visibility (haze) in many national parks and wilderness areas (EPA 2012a). $PM_{2.5}$ can either be directly emitted from sources (e.g., forest fires) or they can form when gas emissions from power plants, industry, and/or vehicles react in the air (EPA 2016f). Particulate matter can either absorb or scatter light, causing the clarity, color, and distance seen by humans (i.e., visibility) to decrease, especially during humid conditions when additional moisture is present in the air. $PM_{2.5}$ is also a concern for human health as these particles can easily pass through the throat and nose and enter the lungs (EPA 2016f). Exposure to these particles can cause airway irritation, coughing, and difficulty breathing (EPA 2016f).

4.9.3. Reference Condition/Values

The NPS Air Resources Division (ARD) developed an approach for rating air quality conditions in national parks, based on the current NAAQS, ecosystem thresholds, and visibility improvement goals (NPS 2015c). This approach is discussed by indicator in the following paragraphs and the ratings are summarized in Table 76 and Table 77.

Table 76. National Park Service Air Resources Division air quality index values for wet deposition of nitrogen or sulfur, ozone, particulate matter, and visibility (NPS 2015c).

Condition Level	Human Health Risk from O_3 (ppb)	Vegetation Health Risk from O_3 (ppm-hrs)	Wet Deposition of N or S (kg/ha-yr)	Visibility (dv*)
Significant Concern	≥71	>13	>3	>8
Moderate Concern	55–70	7-13	1–3	2–8
Good Condition	≤55	<7	<1	<2

*a unit of visibility proportional to the logarithm of the atmospheric extinction; one deciview (dv) represents the minimal perceptible change in visibility to the human eye.

Table 77. National Park Service Air Resources Division air quality assessment matrix for mercury status (NPS 2015c). Green = Good Condition, yellow = Moderate Concern, and red = Significant Concern.

Predicted Methylmercury Concentration Rating	Mercury Wet Deposition Rating				
	Very Low (<3 µg/m2/yr)	Low (≥3–<6 µg/m2/yr)	Moderate (≥6–<9 µg/m2/yr)	High (≥9–<12 µg/m2/yr)	Very High (≥ 12 µg/m2/yr)
Very Low (< 0.038 ng/L)					
Low (≥0.038–< 0.053 ng/L)					
Moderate (≥0.053–<0.075 ng/L)					
High (≥0.075–<0.12 ng/L)					
Very High (≥0.12 ng/L)					

Ozone

The primary NAAQS for ground-level ozone is set by the EPA, and is based on human health effects. The 2008 NAAQS for ozone was a 4th-highest daily maximum 8-hour ozone concentration of 75 parts per billion (ppb) (NPS 2015c). On 1 October 2015, the EPA strengthened the national ozone standard by setting the new level at 70 ppb (EPA 2015). The NPS ARD recommends a benchmark for *Good Condition* ozone status in line with the updated Air Quality Index (AQI) breakpoints (NPS 2015c).

Current condition for human health risk from ozone is based on the estimated 5-year 4th-highest daily maximum 8-hour ozone average concentration in ppb (NPS 2015c). Ozone concentrations ≥71 ppb are assigned a *Significant Concern*, from 55–70 ppb are assigned *Moderate Concern*, and <55ppb are assigned a Good Condition (NPS 2015c).

In addition to being a concern to human health, long-term exposures to ozone can cause injury to ozone-sensitive plants (EPA 2014). The W126 metric relates plant response to ozone exposure and is a better predictor of vegetation response than the metric used for the primary (human-health based) standard (EPA 2014). The W126 metric measures cumulative ozone exposure over the growing season in "parts per million-hours" (ppm-hrs) and is used for assessing the vegetation health risk from ozone levels (EPA 2014).

The W126 condition thresholds are based on information in the EPA's Policy Assessment for the Review of the Ozone NAAQS (EPA 2014). Research has found that for a W126 value of:

- ≤7 ppm-hrs, tree seedling biomass loss is ≤2% per year in sensitive species; and
- ≥13 ppm-hrs, tree seedling biomass loss is 4–10% per year in sensitive species.

The NPS ARD recommends a W126 of <7 ppm-hrs to protect most sensitive trees and vegetation. Levels below this guideline are considered *Good Condition*, 7-13 ppm-hrs is *Moderate Condition*, and >13 ppm-hrs is considered to be of *Significant Concern* (NPS 2015c).

Atmospheric Deposition of Sulfur and Nitrogen

Assessment of current condition of nitrogen and sulfur atmospheric deposition is based on wet (rain and snow) deposition. Wet deposition is used as a surrogate for total deposition (wet plus dry), because wet deposition is the only nationally available monitored source of nitrogen and sulfur deposition data (NPS 2015c). Values for nitrogen (from ammonium and nitrate) and sulfur (from sulfate) wet deposition are expressed as amount of nitrogen or sulfur in kilograms deposited over a 1 ha (2.5 ac) area in 1 year (kg/ha/yr). The NPS ARD selected a wet deposition threshold of 1.0 kg/ha/yr as the level below which natural ecosystems are likely protected from harm. This is based on studies linking early stages of aquatic health decline correlated with 1.0 kg/ha/yr wet deposition of nitrogen both in the Rocky Mountains (Baron et al. 2011) and in the Pacific Northwest (Sheibley et al. 2014). Parks with ≤1 kg/ha/yr of atmospheric wet deposition of nitrogen or sulfur compounds are assigned *Good Condition*, those with 1-3 kg/ha/yr are assigned *Moderate Concern*, and parks with depositions ≥3 kg/ha/yr are assigned *Significant Concern* (NPS 2015c).

Mercury Deposition

The condition of mercury was assessed using estimated 3-year average mercury wet deposition (micrograms per m^2 per year [$\mu g/m^2/yr$]) and the predicted surface water methylmercury concentrations (nanograms per liter [ng/L]) at NPS I&M parks (NPS 2015c). It is important to consider both mercury deposition inputs and ecosystem susceptibility to mercury methylation when assessing mercury condition because atmospheric inputs of elemental or inorganic mercury must be methylated before it is biologically available and able to accumulate in food webs (NPS 2015c). Thus, mercury condition cannot be assessed according to mercury wet deposition alone. Other factors, like environmental conditions conducive to mercury methylation (e.g., dissolved organic carbon, wetlands, pH), must also be considered (NPS 2015c). Mercury wet deposition and predicted methylmercury concentration are considered concurrently in the mercury status assessment matrix shown in Table 68 to determine park-specific mercury/toxics status (NPS 2015c).

Visibility

Visibility conditions are assessed in terms of a Haze Index, a measure of visibility (termed deciviews [dv]) that is derived from calculated light extinction and represents the minimal perceptible change in visibility to the human eye (NPS 2013b). Conditions measured near 0 dv are clear and provide excellent visibility, and as dv measurements increase, visibility conditions become hazier (NPS 2013b). The NPS ARD assesses visibility condition status based on the deviation of the estimated current visibility on mid-range days from estimated natural visibility on mid-range days (i.e., those estimated for a given area in the absence of human- caused visibility impairment, EPA-454/B003-005) (NPS 2015c). The NPS ARD chose reference condition ranges to reflect the variation in visibility conditions across the monitoring network. Visibility on mid-range days is defined as the mean of the visibility observations falling within the 40^{th} and 60^{th} percentiles (NPS 2015c). A visibility condition estimate of <2 dv above estimated natural conditions indicates a *Good Condition*, estimates ranging from 2-8 dv above natural conditions indicate *Moderate Concern*, and estimates >8 dv above natural conditions indicate *Significant Concern* (NPS 2015c).

Visibility trends are computed from the Haze Index values on the 20% haziest days and the 20% clearest days, consistent with visibility goals in the CAA and Regional Haze Rule, which include improving visibility on the haziest days and allowing no deterioration on the clearest days (NPS 2015c). Although this legislation provides special protection for NPS areas designated as Class I, the NPS applies these standard visibility metrics to all units of the NPS. If the Haze Index trend on the 20% clearest days is deteriorating, the overall visibility trend is reported as deteriorating. Otherwise, the Haze Index trend on the 20% haziest days is reported as the overall visibility trend (NPS 2015c).

4.9.4. Data and Methods

Monitoring in the Park

Air quality monitoring in the park has been extremely limited, with only one study measuring total air mercury concentration on a single occasion in 2011 (Sutton 2012). This was part of a research project to evaluate Spanish moss as a bioindicator of mercury contamination. Sutton (2012) compared mercury concentrations in Spanish moss already present at several Georgia locations, including CUIS, to concentrations from moss transplanted to the locations. Sutton (2012) also measured total air mercury concentrations at each transplant location.

NPS Data Resources

Although data on most air quality parameters are not actively collected within park boundaries, data collected at several regional monitoring stations for various parameters can be used to estimate air quality conditions in CUIS. NPS ARD provides estimates of ozone, wet deposition (nitrogen, sulfur, and mercury), and visibility that are based on interpolations of data from all air quality monitoring stations operated by NPS, EPA, various states, and other entities, averaged over the most recent 5 years (2011–2015). Estimates and conditions data for CUIS were obtained from the NPS Air Quality by park data products page (http://www.nature.nps.gov/air/data/products/parks/index.cfm).

On-site or nearby data are needed for a statistically valid trends analysis (within 10 km [6.2 mi] for ozone and within 16 km [10 mi] for deposition) (NPS 2015c). There are no on-site or near-enough representative monitors for such an assessment of ozone, $PM_{2.5}$, and nitrogen, sulfur and mercury deposition trends at this time. For visibility trend analysis, monitoring data from an Interagency Monitoring of Protected Visual Environments Program (IMPROVE) station is required. An IMPROVE monitoring site considered representative of a Class II park has to be between within +/- 30.48 m (100 ft) or 10% of maximum and minimum elevation of the park and at a distance of no more than 150 km (93 mi) (NPS 2015c). The IMPROVE visibility monitor at Okefenokee National Wildlife Refuge in southeastern Georgia (Monitor ID: OKEF1) is approximately 62 km (38.5 mi) west of CUIS and is considered representative for the park (Figure 70).

Other Air Quality Data Resources

The EPA Air Trends Database provides annual average summary data for ozone and $PM_{2.5}$ concentrations near CUIS (EPA 2016a). The nearest ozone and $PM_{2.5}$ monitor is located at Risley Middle School in Brunswick, GA (Site ID: 13-127-0006) and is operated by the Georgia Air Protection Branch Ambient Monitoring Program (Figure 70). This station, which has collected ozone data since 1995 and $PM_{2.5}$, data since 1999, is located approximately 22 km (13.7 mi) north of CUIS. Although this station is not close enough to CUIS to provide data for a statistically valid trend analysis, it does offer some insight into air quality conditions in the region.

The National Atmospheric Deposition Program–National Trends Network (NADP-NTN) database provides annual average summary data for nitrogen and sulfur concentration and deposition across the U.S. (NADP 2016c). The NADP-NTN monitoring site closest to CUIS is located at Sapelo Island, on the Georgia coast (site ID: GA33), approximately 47 km (29.2 mi) north of CUIS (Figure 70). This site has collected deposition data for the region since 2002 and is currently active in monitoring (NADP 2016c). Data summaries for this monitor are available on the NADP-NTN website (NADP 2016c). This station also is not close enough to CUIS for a statistically valid trend analysis, but provides insight regarding regional conditions.

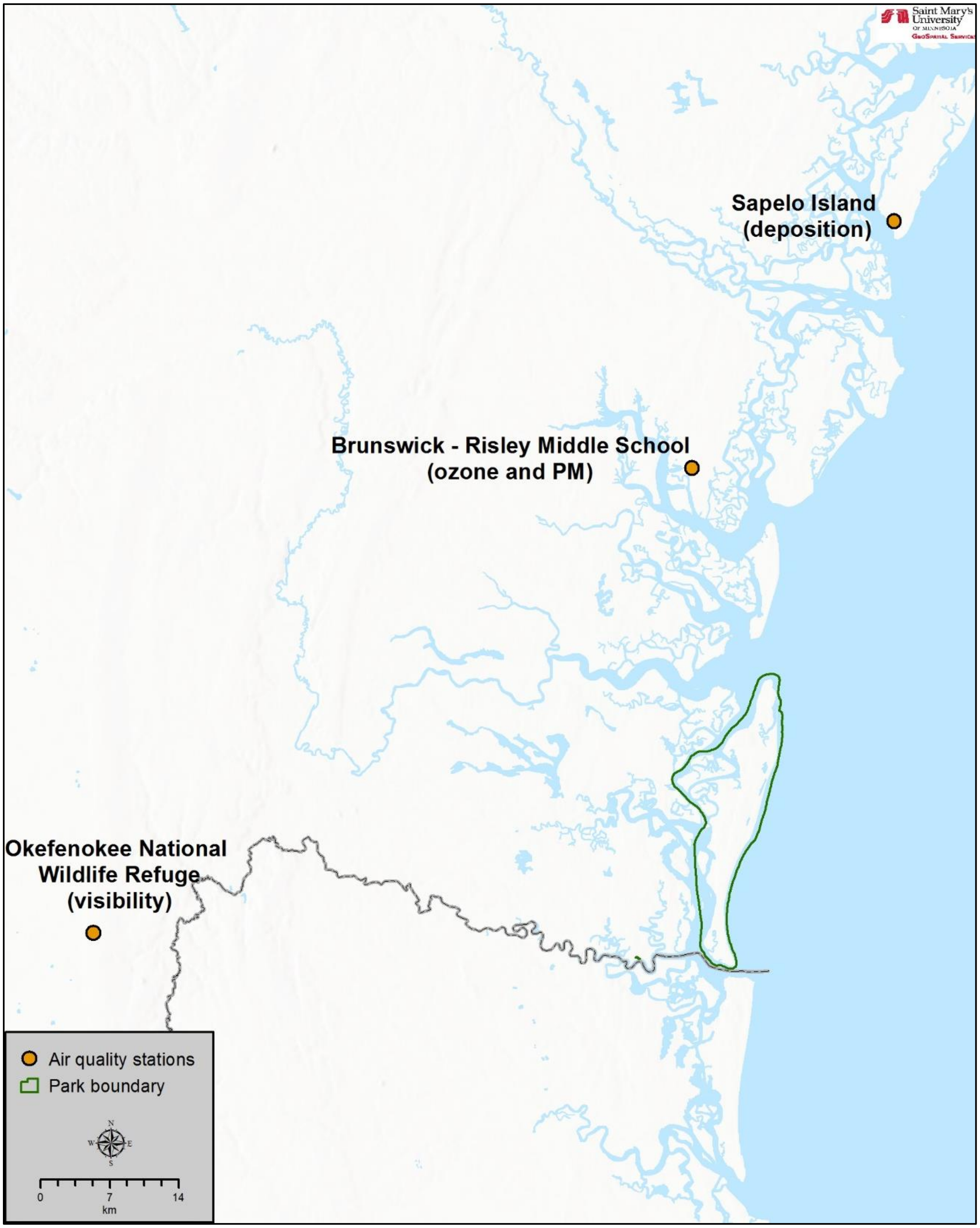


Figure 70. Air quality monitoring locations in relation to CUIS.

The NADP Mercury Deposition Network (MDN) provides weekly summary data for mercury deposition and concentration (NADP 2016a). Wet mercury deposition trends are evaluated using pollutant concentrations in precipitation (micro equivalents/liter) so that yearly variations in precipitation amounts do not influence trend analyses. Trends are computed for parks with a representative NADP-MDN wet deposition monitor that is within 16 km (10 mi) of park boundaries (NPS 2015c). The monitor closest to CUIS is on Sapelo Island, which is more than 16 km to the north (NADP 2016a). Predicted methylmercury concentrations in surface water were obtained from a model that predicts surface water methylmercury concentrations for hydrologic units throughout the U.S. based on relevant water quality characteristics (pH, sulfate, and total organic carbon) and wetland abundance (USGS 2015).

Special Air Quality Studies

Sullivan et al. (2011a, 2011c) identified ecosystems and resources in national parks that were at risk to acidification and excess nitrogen enrichment. These reports provided a relative risk assessment of acidification and nutrient enrichment impacts from atmospheric nitrogen and sulfur deposition for parks in 32 I&M networks. Ecosystem sensitivity ratings to acidification from atmospheric deposition were based on percent sensitive vegetation types, number of high-elevation lakes, length of low-order streams, length of high-elevation streams, average slope, and acid-sensitive areas within the park (Sullivan et al. 2011a). Ecosystem sensitivity ratings to nutrient enrichment effects were based on percent sensitive vegetation types and number of high-elevation lakes within the park (Sullivan et al. 2011c).

Kohut (2004) employed a biologically-based method to evaluate the risk of foliar injury from ozone at parks within the SECN. The assessment allowed resource managers at each park to better understand the risk of ozone injury to vegetation within their park and permits them to make a better informed decision regarding the need to monitor the impacts of ozone on plants.

Pardo et al. (2011) synthesized current research relating atmospheric nitrogen deposition to effects on terrestrial and aquatic ecosystems in the U.S. and identified empirical critical loads for atmospheric nitrogen deposition.

4.9.5. Current Condition and Trend

Nitrogen Deposition

Five-year interpolated averages of nitrogen (from nitrate and ammonium) wet deposition are used to estimate condition for deposition. The most recent 5-year (2011–2015) estimate for nitrogen deposition at CUIS is 2.7 kg/ha/yr (NPS 2016b). Based on the NPS ratings for air quality conditions (see Table 77), this falls in the *Moderate Concern* range. A comparison to previous 5-year estimates shows that nitrogen deposition rates have been relatively stable over recent years (Figure 71).

In addition to assessing wet deposition levels, critical loads can also be a useful tool in determining the extent of deposition impacts (i.e., nutrient enrichment) to park resources (Pardo et al. 2011). A critical load is defined as the level of deposition below which harmful effects to the ecosystem are not expected (Pardo et al. 2011). For the Eastern Temperate Forest, the ecoregion where CUIS is located, Pardo et al. (2011) suggested critical loads for total nitrogen deposition (wet plus dry) of 4-8 kg/ha/yr to protect lichens, 8 kg/ha/yr to protect hardwood forests, and <17.5 kg/ha/yr to protect herbaceous species. The lowest critical load level (4.0 kg/ha/yr) is identified as an appropriate management goal because it will protect the full range of vegetation in the park (Pardo et al. 2011).

The 2011-2015 estimated deposition at CUIS of 2.7 kg/ha/yr was below the minimum ecosystem critical load for the ecoregion, suggesting that sensitive vegetation elements may not be at risk for harmful effects. However, Sullivan et al. (2011d) identified CUIS as being at very high risk of nutrient enrichment from nitrogen deposition, due to moderate pollutant exposure and high levels of ecosystem sensitivity.

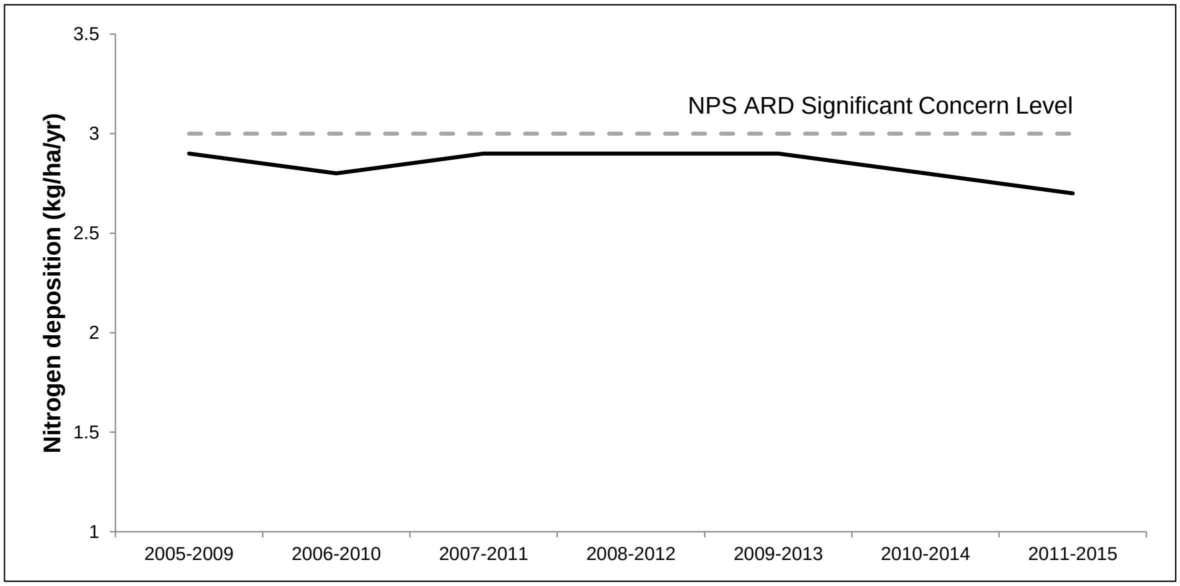


Figure 71. Estimated 5-year averages of nitrogen wet deposition (kg/ha/yr) at CUIS (NPS 2016b).

Concentrations (mg/L) of nitrogen compounds in wet deposition can also be used to evaluate overall trends in deposition. Since atmospheric wet deposition can vary greatly depending on the amount of precipitation that falls in any given year, it can be useful to examine concentrations of pollutants, which factor out the variation introduced by precipitation. Figure 72 suggests that nitrate concentrations in the Georgia coastal region have fluctuated since 2002 but appear to be decreasing over time (NADP 2016c). Ammonium concentrations in the region have also fluctuated, but with no clear increasing or decreasing trend (Figure 73) (NADP 2016c).

In contrast to the nutrient enrichment assessment discussed previously, Sullivan et al. (2011b) ranked CUIS as being at moderate risk of acidification from acidic (nitrogen and sulfur) deposition, due to high pollutant exposure but low levels of ecosystem sensitivity.

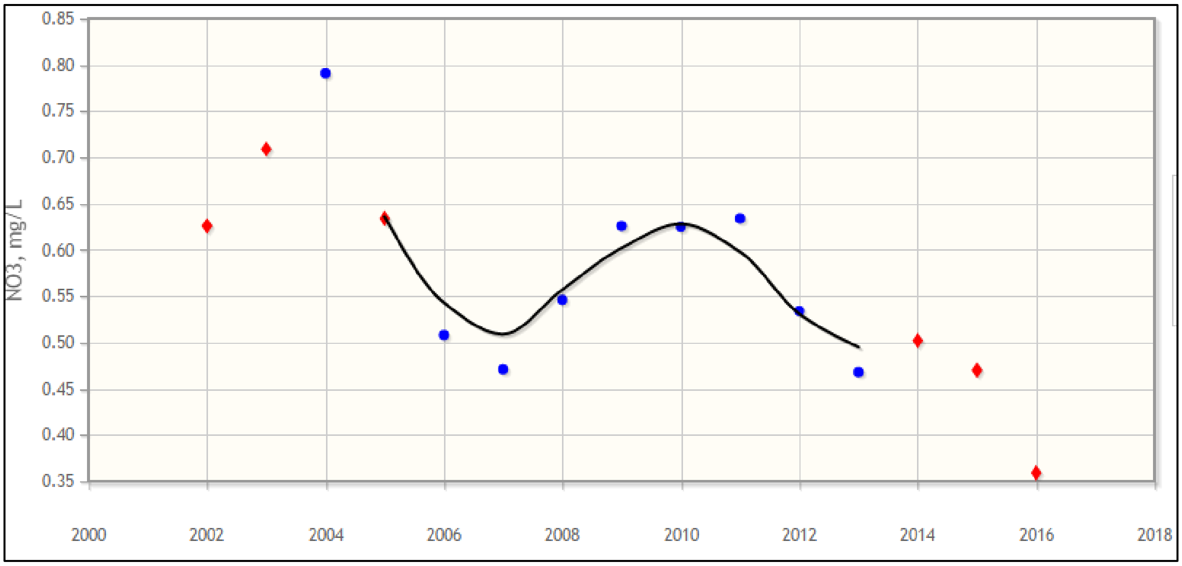


Figure 72. Annual weighted mean concentration of nitrate in wet deposition from Sapelo Island (NTN Site GA33), approximately 47 km (29.2 mi) north of CUIS (NADP 2016c). The black line represents a smoothed 3-yr moving average. Red diamonds represent years when NADP's data completeness criteria (valid samples for 75% of the period) were not met and, therefore, were not included in trend line calculations.

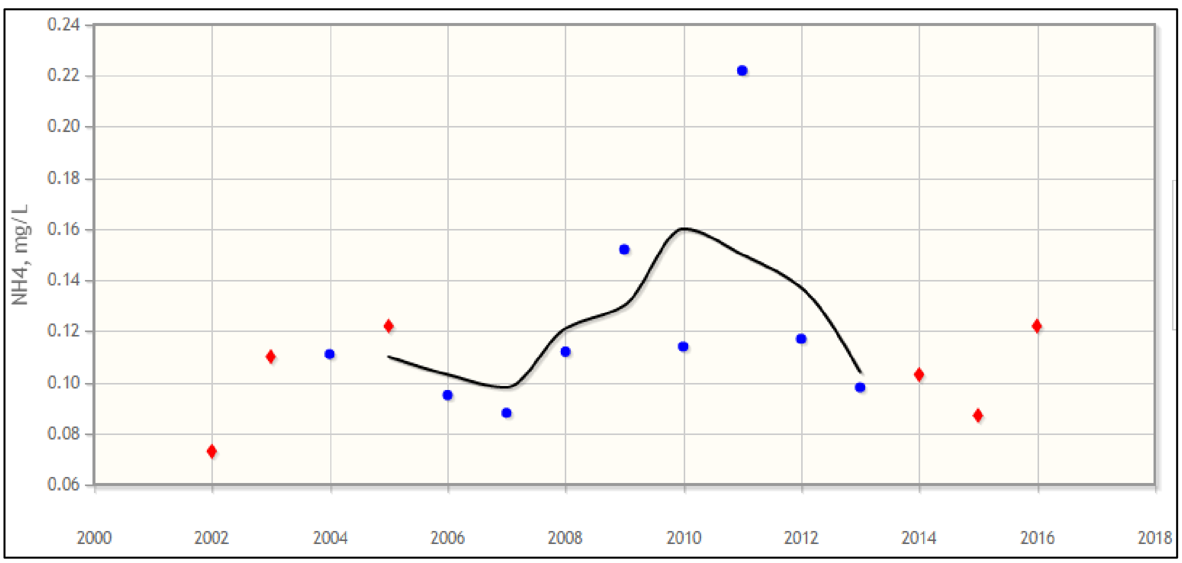


Figure 73. Annual weighted mean concentration of ammonium in wet deposition from Sapelo Island (NTN Site GA33) (NADP 2016c). The black line represents a smoothed 3-yr moving average. Red diamonds represent years when NADP's data completeness criteria were not met and, therefore, were not included in trend line calculations.

Sulfur Deposition

Five-year interpolated averages of sulfur (from sulfate) wet deposition are used to estimate condition for deposition. The most recent 5-year (2011–2015) estimate for sulfur wet deposition at CUIS is 2.6 kg/ha/yr (NPS 2016b). This falls in the *Moderate Concern* range. A comparison to previous estimates suggests that sulfur deposition is decreasing at CUIS, and has just improved from Significant to Moderate Concern levels in recent years (Figure 74).

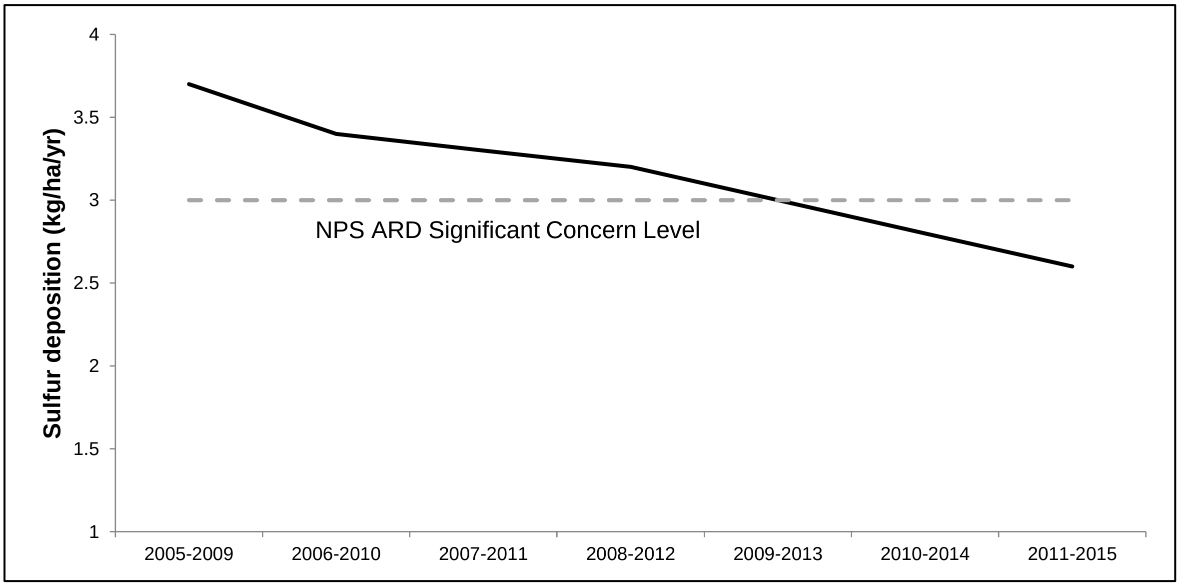


Figure 74. Estimated 5-year averages of sulfur wet deposition (kg/ha/yr) at CUIS (NPS 2016b).

As with nitrogen, concentrations (mg/L) of sulfur compounds in wet deposition can also be used to evaluate overall trends in deposition. Figure 75 suggests that the sulfate concentration in the Georgia coastal region has declined over time, with levels below 0.7 mg/L for the past 4 years (NADP 2016c).

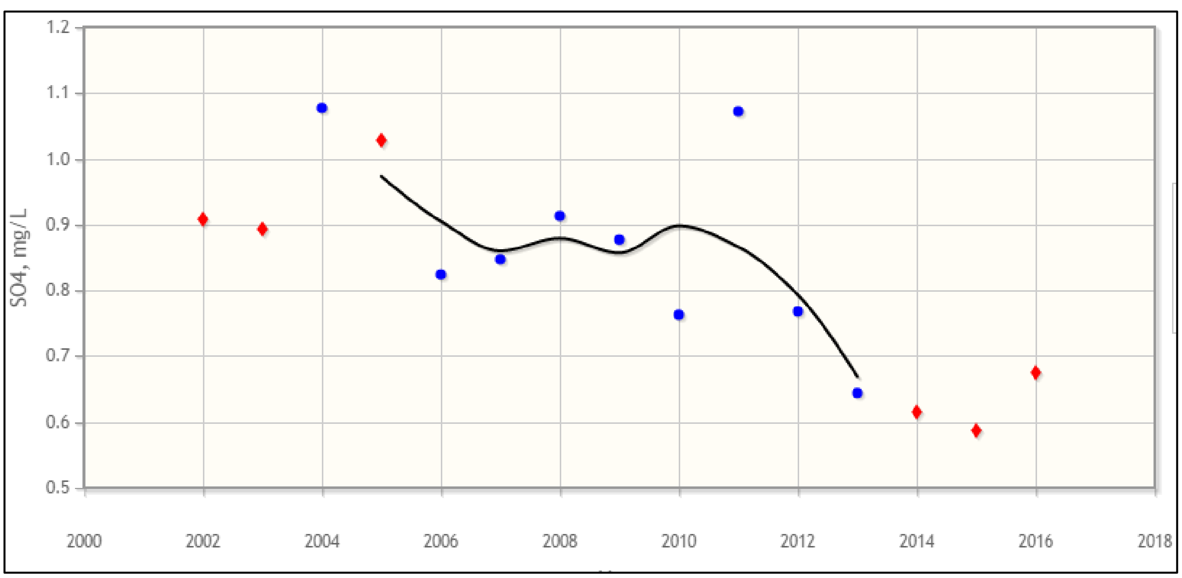


Figure 75. Annual weighted mean concentration of sulfate in wet deposition from Sapelo Island (NTN Site GA33) (NADP 2016c). The black line represents a smoothed 3-yr moving average. Red diamonds

represent years when NADP's data completeness criteria (valid samples for 75% of the period) were not met and, therefore, were not included in trend line calculations.

Mercury Deposition

The 2013-2015 wet mercury deposition estimate was very high for CUIS, at 13.2 μg/m^2/yr. Predicted methylmercury concentrations in surface waters were also very high, at an estimated 0.51 ng/l (NPS 2016a). When compared to the NPS ARD mercury status assessment matrix (Table 69), these estimates result in a condition of *Significant Concern*. However, confidence in this assignment is low, given a lack of park-specific contaminant data.

During the summer of 2011, Sutton (2012) measured total air mercury concentration at four locations within CUIS. The resulting mean concentration was 0.012 μg/m^3, which was higher than air concentrations on other barrier islands (Sapelo and Ossabaw Islands) and in urban study areas (Jacksonville, FL, and Savannah, GA). Sutton (2012) hypothesized that air mercury concentrations may be higher in coastal areas than in urban areas due to the prevalence of wetlands (freshwater and salt marshes), as wetlands have been known to accumulate mercury and re-release it into the atmosphere through plant transpiration (Zillioux et al. 1993, Lindberg et al. 2002).

Based on interpolations by the MDN, mercury deposition levels in the CUIS area in 2015 were likely in the 13-15 μg/m^2 range (Figure 76) (NADP 2017). Based on interpolations displayed in Figure 77, total mercury concentrations in the area were likely 10-12 ng/L (NADP 2017).

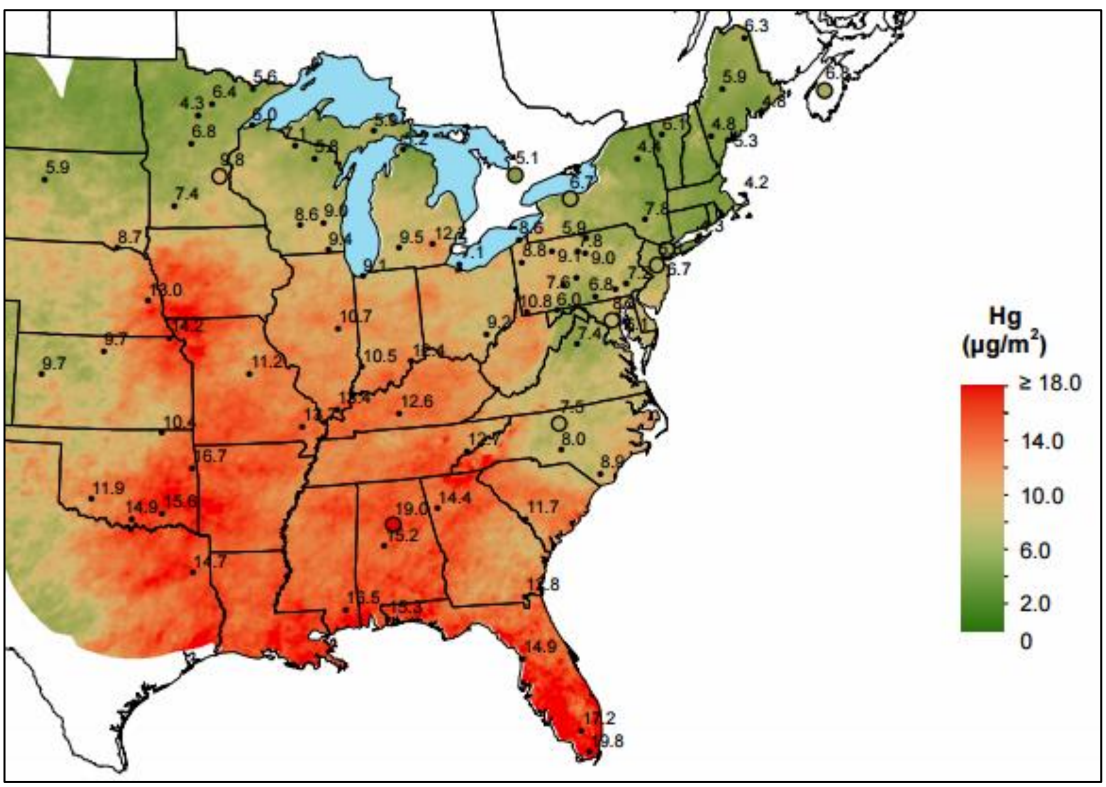


Figure 76. Total annual mercury wet deposition in 2015, based on interpolations by the MDN (NADP 2017).

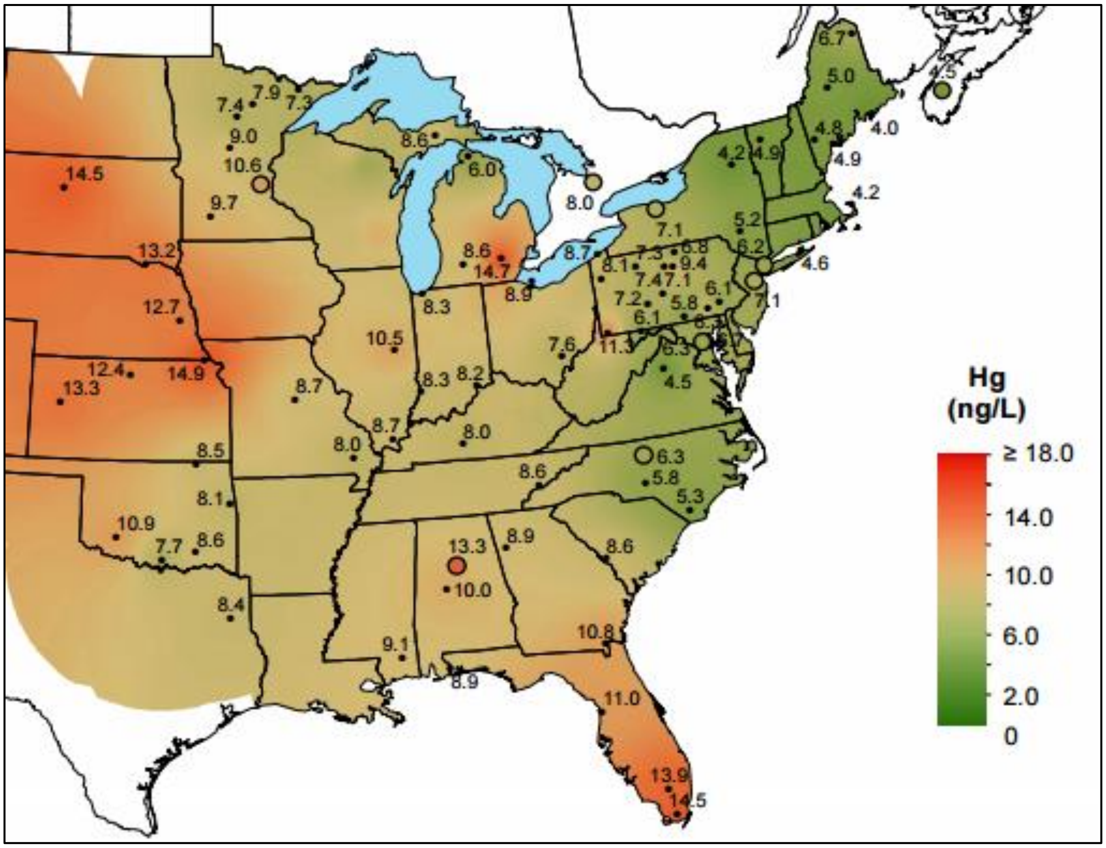


Figure 77. Total mercury concentrations in 2015, based on interpolations by the MDN (NADP 2017).

The NPS ARD has measured mercury wet deposition at 16 national parks across the U.S. (NPS 2013a). The location closest to CUIS where monitoring has occurred is Congaree National Park in central South Carolina, approximately 320km (200 mi) north of CUIS. According to an analysis of mercury concentrations in precipitation, concentrations at Congaree have improved slightly, declining by 0.37 ng/L/yr between 2000 and 2009 (NPS 2013a).

Ozone

Historically, ozone has not been a particular concern in the CUIS region. Kohut (2004) determined that the risk of ozone exposure at the park was low, with concentrations estimated (through kriging) to exceed 80 ppb only occasionally between 1995 and 1999. During these same years, the estimated W126 value remained above 13 ppm-hrs and exceeded 25 ppm-hrs in 1998 (Kohut 2004).

The condition of human risk from ozone in NPS units is determined by calculating the 5-year average of the 4th-highest daily maximum of 8-hour average ozone concentrations measured at each monitor within an area over each year (NPS 2013b). The most recent 5-year (2011–2015) estimated average for 4th-highest 8-hour ozone concentration at CUIS was 60.9 ppb (NPS 2016b). This is within the *Moderate Concern* range. A comparison to previous estimates suggests that ozone conditions are improving around CUIS, as 5-year average estimates have declined from a high of 70.5 ppb for 2005-2009 (Figure 78).

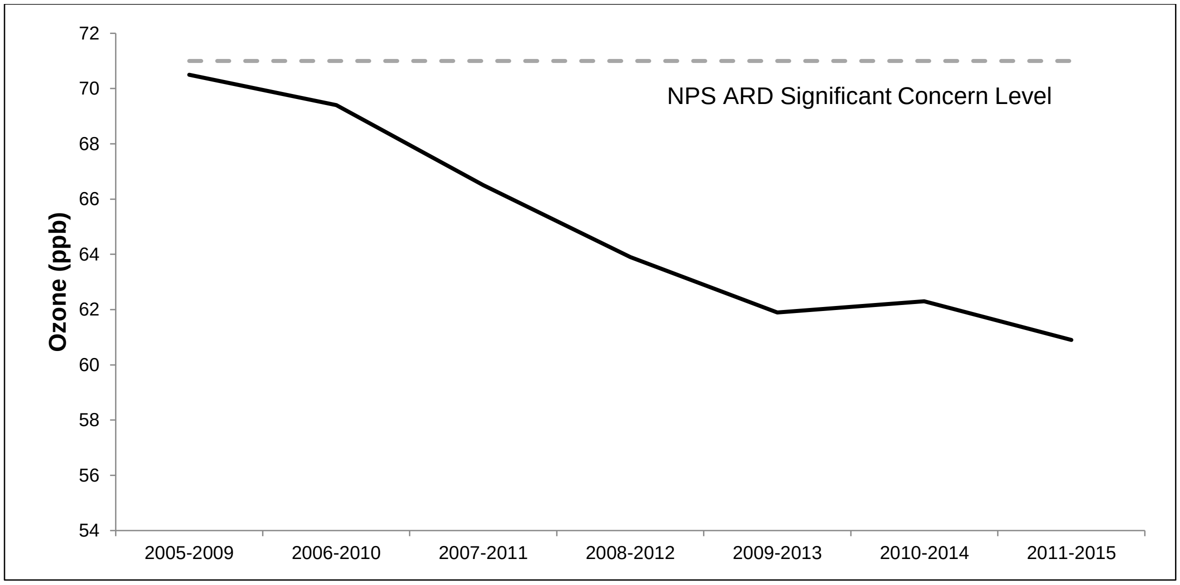


Figure 78. Estimated 5-year averages of the 4th-highest daily maximum of 8-hour average ozone concentrations for CUIS (NPS 2016b).

The apparent improvement in ozone condition is supported by data from the nearest year-round ozone monitor (in nearby Brunswick, GA), which show ozone concentrations fluctuating over time but with a general decreasing trend (Figure 79) (EPA 2016a).

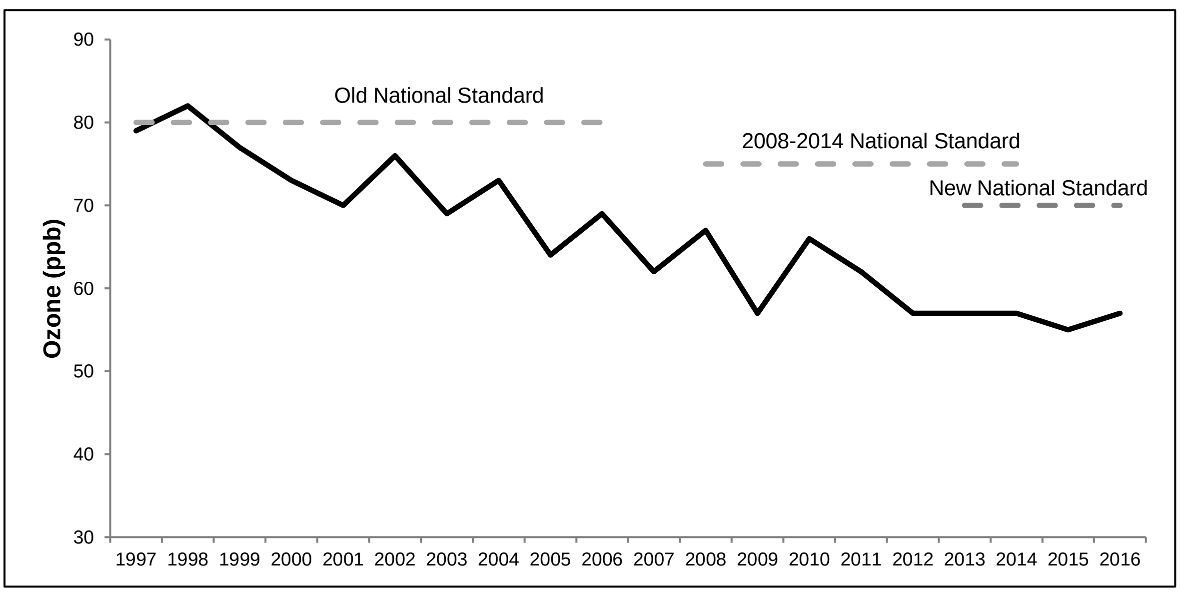


Figure 79. Annual 4th-highest 8-hour maximum ozone concentrations (ppb) at the Risley Middle School monitoring site (Site ID: 13-127-0006) in Brunswick, GA, 1997-2016 (EPA 2016a).

Vegetation health risk from ground-level ozone condition is determined by estimating a 5-year average of annual maximum 3-month, 12-hour W126 values. The 2011–2015 estimated W126 metric for CUIS of 4.7 falls in the *Good Condition* category (NPS 2016b). Again, a comparison to previous estimates suggests that ozone conditions are improving, with W126 decreasing from a high of 8.6 for 2005-2009 (Figure 80).

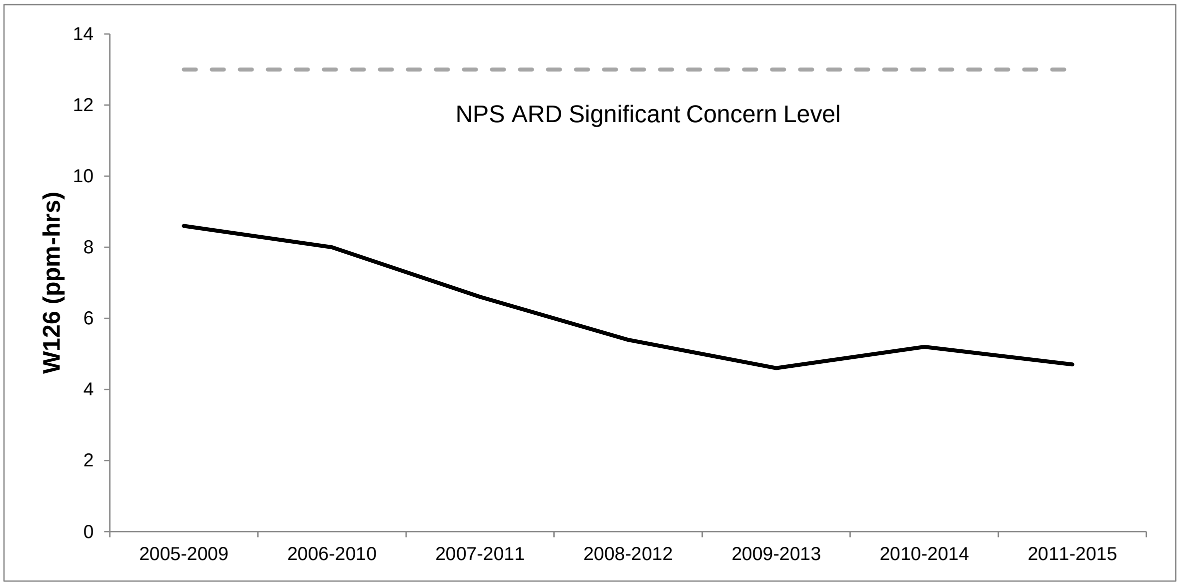


Figure 80. Estimated 5-year averages of the W126 ozone metric for CUIS (NPS 2016b).

Visibility

Five-year estimated averages of visibility on mid-range days minus natural condition visibility on mid-range days are used to estimate condition for visibility. The 2011–2015 estimated visibility on mid-range days for CUIS was 9.0 dv above estimated natural conditions (NPS 2016b). This estimate falls into the *Significant Concern* category based on NPS criteria for air quality assessment.

Comparing the most recent mid-range estimate to previous NPS ARD estimates of visibility suggests that conditions may be improving at CUIS. The 5-year average has declined every year since 2009, when estimated visibility was 11.5 dv above estimated natural conditions, although it is still above the significant concern threshold (>8 dv) (Figure 81). Based on monitoring data from the nearby OKEF1 station, conditions also appear to be improving over time on the 20% haziest and 20% clearest days (Figure 82) (NPS 2016b).

$PM_{2.5}$ is a major contributor to visibility impairment. Annual average 24-hour $PM_{2.5}$ concentrations are available from the Risley Middle School station in Brunswick, GA for 1999-2016. Annual concentrations at this station have fluctuated, falling below 30 µg/m^3 in most years but with occasional higher spikes (Figure 83) (EPA 2016a). The EPA NAAQS for $PM_{2.5}$ uses the 3-year average 98th percentile 24-hour $PM_{2.5}$ concentration to assess human health risk. The most recent 3-year average (2014-2016) concentration for this station is 23.7 µg/m^3. This meets the EPA standard of <35 µg/m^3.

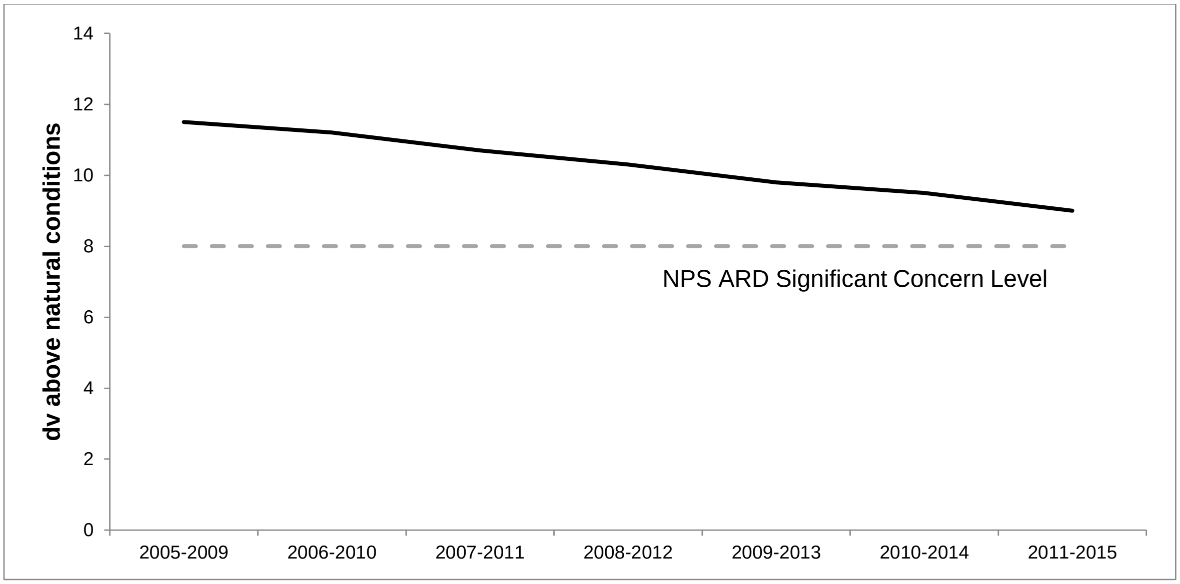


Figure 81. Estimated 5-year averages of visibility (dv above natural conditions) on mid-range days at CUIS (NPS 2016b).

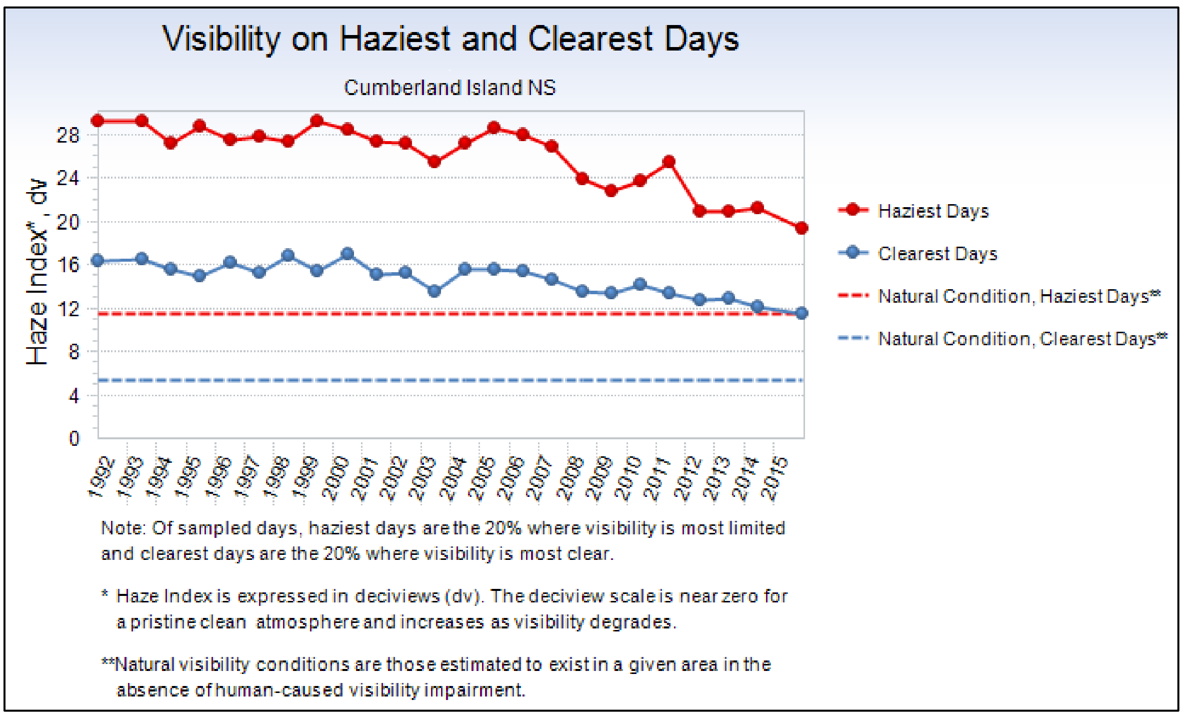


Figure 82. Long-term trends in visibility in the CUIS region, based on measurements from the OKEF1 monitoring station at Okefenokee National Wildlife Refuge in southeast Georgia (reproduced from NPS 2016b).

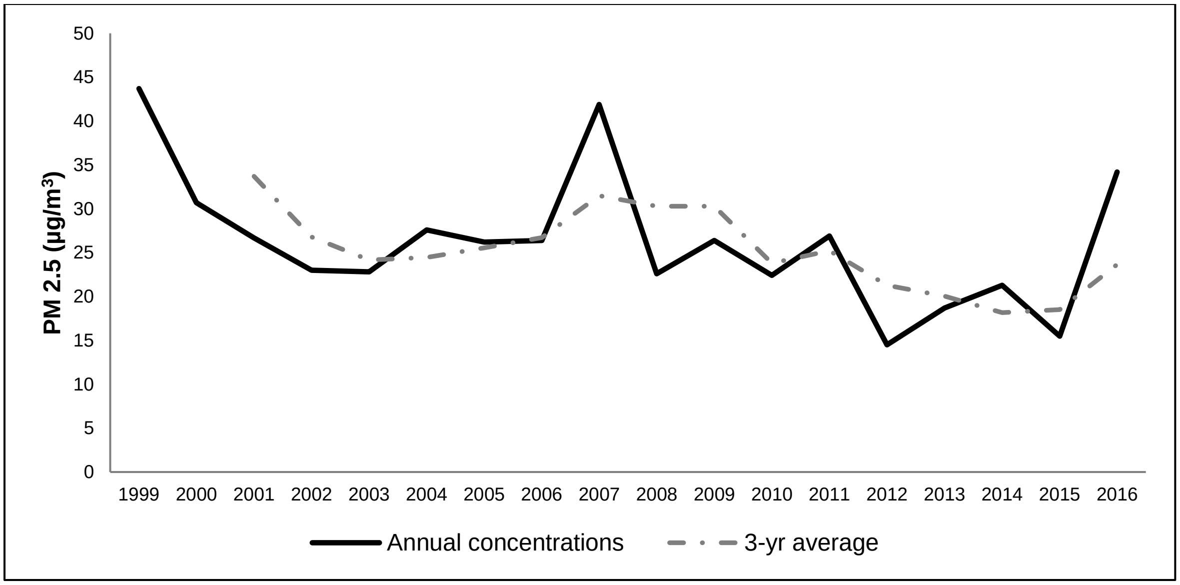


Figure 83. Annual 24-hour particulate matter ($PM_{2.5}$) concentrations (98th percentile) and 3-year running averages for the CUIS region, 1999-2016 (EPA 2016a). The monitoring station is located at Risley Middle School (Site ID: 13-127-0006) in Brunswick, GA.

Threats and Stressor Factors

Threats to CUIS's air quality include power plants and industrial facilities (especially paper mills), a Superfund site in nearby Brunswick, vehicle emissions, and wildland fires. If approved and constructed, the proposed Spaceport in Camden County would also threaten air quality (Figure 84). The NPS expressed concerns over the potential impacts of this commercial space launch site, including the effects it may have on air quality, shortly after the FAA announced its intent to prepare an EIS for the facility (Austin 2015). Little is known about the impacts of rocket launches on ground-level air quality, although concerns have been raised that metal particles and other chemicals released into the air during launches may settle and accumulate in surrounding wetlands (Bowden et al. 2014, Konkel 2014).

There are currently three operational paper/pulp mills within 25 km (15.5 mi) of CUIS: two in Fernandina Beach, FL, approximately 3.2 km (2 mi) south of the park, and one in Brunswick, GA, around 23 km (14.3 mi) to the north (Figure 84) (NPS 2015a). Paper and pulp mill emissions are known to include air contaminants such as particulate matter, sulfur dioxide, and reduced sulfur compounds (e.g., hydrogen sulfide, methyl mercaptan), which typically produce a distinct unpleasant odor (WBG 1999). Depending on the pulping process used, these mills may also produce nitrogen oxides, VOCs, and heavy metals (e.g., mercury, cadmium, lead) (WBG 1999, EPA 2012b). According to a 2012 assessment by the EPA (2012b), the emission levels of these compounds from paper and pulp mills is relatively low and is not expected to cause any detrimental environmental effects. The primary impact of paper/pulp mills has been described as "the reduction of aesthetic air quality" (Murray 1992, p. 5). This is likely the case at CUIS, as park staff rarely smell pulp emissions on the island (Fry, written communication, 18 July 2017), but smoke rising from the mill's stacks is visible from the southern end of the island (Figure 85).

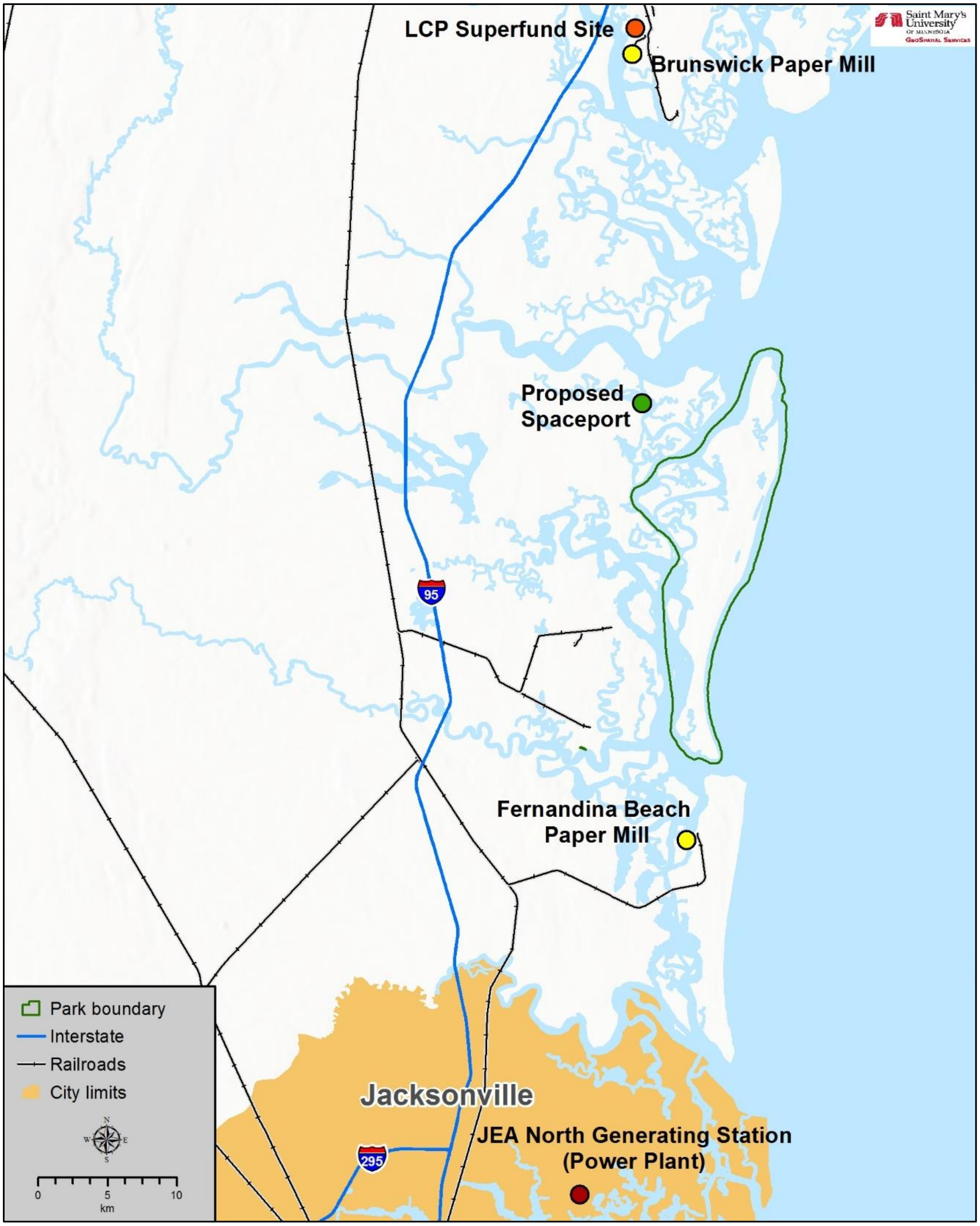


Figure 84. Locations of potential air quality stressors in the vicinity of CUIS.

Figure 85. Pulp mill stacks in Fernandina Beach visible from the southern tip of CUIS (SMUMN GSS photo).

Power plants are a major source of greenhouse gas emissions in the U.S. (EPA 2012a). The Jacksonville Electric Authority (JEA) operates a large power plant, known as the Northside Generating Station, approximately 32 km (20 mi) south of CUIS (Figure 84). According to JEA (2017), the plant utilizes a mix of natural gas, fuel oil, coal, and petroleum coke to power three large steam units and four smaller diesel-powered units. Natural gas and fuel oil are the current primary fuel sources for the larger units (JEA 2017). While considered less polluting than coal, natural gas combustion produces nitrogen oxides, carbon monoxide and carbon dioxide, VOCs, and methane (EPA 1995).

A former industrial site in Brunswick operated by LCP Chemicals until 1994 may still be contributing mercury emissions into the atmosphere (Sutton 2012, ATSDR 2014). While in operation, the chlor-alkali plant released mercury and mercury-containing wastes onto the ground and into 270 ha (670 ac) of tidal marshlands west of the site. The EPA has estimated that 380,000 pounds of mercury were “lost” in the area between 1955 and 1979 alone (ATSDR 2014). The marshlands still contain residual mercury today, which can evaporate into the air and be transported long distances (ATSDR 2014). Sutton (2012) found elevated levels of mercury in the air and in Spanish moss tissues up to 1.3 km (0.8 mi) away from the LCP site.

Transportation sources account for a significant portion of nitrogen oxide and VOC emissions in the U.S. and also produce some particulate pollution and sulfur dioxides (Small and Kazimi 1995). These emissions can contribute to ozone formation and impact visibility. While vehicle traffic at CUIS is limited to use by park management and private landowners, Interstate 95 is a major north-south travel corridor along the coast, just 17 km (10.6 mi) west of the park. Railroads in the vicinity can also contribute emissions (Figure 84).

Prescribed burning and wildfires produce air pollutants, including PM, carbon monoxide, and VOCs, which can contribute to ozone formation (Wotawa and Trainer 2000, Lee et al. 2005). Air pollution from fires typically impairs visibility and can travel long distances. For example, forest fires in Canada have been shown to impact air quality in the eastern U.S., including areas as far south as Tennessee (Wotawa and Trainer 2000, Lee et al. 2005). During a large, long-lasting wildfire at Okefenokee Swamp (over 60 km [38 mi] west of CUIS) in 2011, ash was transported as far as CUIS (Fry, oral communication, 8 March 2017) and air quality advisories were issued in Jacksonville, FL (Scanlan 2011).

Air pollutants such as ozone and particulates are strongly influenced by weather shifts (e.g., heat waves, droughts) (EPA 2012a). According to the EPA and the Intergovernmental Panel on Climate Change (IPCC), warmer temperatures associated with global climate change are expected to negatively affect air quality (EPA 2012a). For example, the EPA (2012a) projects that climate change could increase summertime average ground-level ozone concentrations in many areas by 2-8 ppb.

<u>Data Needs/Gaps</u>

The park's recent Foundation Document (NPS 2014a) recognized the need for collection of air quality data specifically within CUIS boundaries. Monitoring on the southern end of the island would be most likely to capture any impacts from the Fernandina Beach paper mill and JEA's Northside Generating Station. Monitoring on the north end could reflect impacts from the Brunswick paper mill and the LCP Chemicals former industrial site. In-park monitoring and analysis of air pollution composition could also help identify additional industrial facilities to the west of CUIS that may be impacting park air quality through long-range transport (Fry, written communication, 18 July 2017).

Studies regarding the potential effects of air pollutants on island resources are also lacking (NPS 2014a). Given the concern over elevated mercury levels in the area, an in-depth assessment of mercury levels in the park's air, sediment, and organisms (e.g., plants, aquatic animals, birds) appears justified.

<u>Overall Condition</u>

Nitrogen Deposition

The project team assigned this measure a *Significance Level* of 3. The most recent 5-year (2011–2015) estimate for nitrogen deposition at CUIS is 2.7 kg/ha/yr, which falls in the *Moderate Concern* range identified by the NPS ARD (NPS 2016b). Sullivan et al. (2011d) ranked CUIS as at very high risk of nutrient enrichment from nitrogen deposition, but current levels are below the minimum ecosystem critical load for the ecoregion (Pardo et al. 2011), suggesting that sensitive vegetation elements may not currently be at risk for harmful effects. This measure is assigned a *Condition Level* of 2, indicating moderate concern.

Sulfur Deposition

Sulfur deposition was also assigned a *Significance Level* of 3. As with nitrogen, the most recent 5-year (2011–2015) estimate for sulfur wet deposition at CUIS of 2.6 kg/ha/yr falls in the *Moderate*

Concern range (NPS 2016b). Conditions appear to be improving, but this measure is assigned a *Condition Level* of 2 as well, for moderate concern.

Mercury Deposition

A *Significance Level* of 3 was assigned for mercury deposition. Based on ARD's 2013-2015 estimates for wet mercury deposition of 13.2 $\mu g/m^2/yr$ and predicted methylmercury concentration in surface waters of 0.51 ng/l, this measure falls in the *Significant Concern* range. This may be related to the prevalence of wetlands (especially salt marshes), as wetlands have been known to accumulate mercury and re-release it into the atmosphere (Zillioux et al. 1993, Lindberg et al. 2002). Therefore, mercury deposition is assigned a *Condition Level* of 3.

Ozone

The project team also assigned this measure a *Significance Level* of 3. Ozone levels appear to be declining around CUIS, with the most recent 5-year (2011–2015) estimated average of 60.9 ppb falling in the *Moderate Concern* range (NPS 2016b). The 2011–2015 estimated W126 metric of 4.7 ppm-hrs falls in the *Good Condition* category. Overall, this measure is assigned a *Condition Level* of 2.

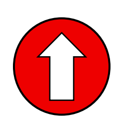
Visibility

Visibility was assigned a *Significance Level* of 3. The 2011–2015 estimated visibility on mid-range days for CUIS was 9.0 dv above estimated natural conditions, or within the *Significant Concern* category identified by the NPS ARD (NPS 2016b). Although visibility appears to be improving over time, it is currently assigned a *Condition Level* of 3 for high concern.

Weighted Condition Score

The *Weighted Condition Score* for air quality at CUIS is 0.80, indicating significant concern (Table 78). However, it is important to acknowledge that the factors influencing this condition are almost entirely beyond the control of park management. Since several measures (sulfur deposition, ozone, visibility) appear to be improving and none are declining, the overall trend is considered to be improving. A medium confidence border is applied due to the use of estimates/interpolations, as park-specific data is lacking.

Table 78. Weighted Condition Score for Air Quality in CUIS.

Air Quality			
Measures	**Significance Level**	**Condition Level**	**WCS = 0.80**
Nitrogen Deposition	3	2	
Sulfur Deposition	3	2	
Mercury Deposition	3	3	
Ozone	3	2	
Visibility	3	3	

4.9.6. Sources of Expertise

John Fry, CUIS Chief of Resource Management

4.10. Barrier Island Geomorphology

4.10.1. Description

Geomorphology refers to the arrangement of physical features on the land's surface, including their structure, origin, and development. Barrier islands, such as Cumberland Island, experience nearly constant geomorphological change due to natural erosion and accretion (i.e., accumulation) processes (Griffin 1982, Alber et al. 2005, Calhoun and Riley 2016). The physical structure of barrier island shores is largely shaped by wave action and tidal currents, with the retreat or advance of shorelines depending on sand/sediment availability and sea level changes (Griffin 1982, Bellis 1995). When sea levels are relatively stable, shorelines advance/accrete when sediment availability is high and often retreat/erode when sediment supply is limited. If sea levels drop, barrier shorelines appear to advance, particularly on the ocean side. When sea levels rise, barrier shorelines retreat and appear to shift landward (e.g., towards the mainland) (Bellis 1995). Cumberland Island is currently considered to be in what is called a "regressive state" (Calhoun and Riley 2016). This state occurs when the elevation and width of an island prevent drastic ocean shoreline erosion (e.g., overwash or breach during storms) but a lack of sediment supply on the back-barrier side allows steady erosion, so that the barrier island is narrowing from the landward side (Calhoun and Riley 2016).

The ocean-side (east) and back-barrier (west) coasts of CUIS vary in both appearance and in the physical processes that shape them. The east side of the island is dominated by sweeping beaches backed by nearly continuous dune fields, with crests reaching over 10 m (33 ft) high in some areas (Figure 86) (NPS 1984, Alber et al. 2005). The back-barrier coast has a more irregular outline, with streams meandering through tidal salt marshes (Figure 87).

Figure 86. The ocean-side (east) shoreline of CUIS (NPS photo).

Figure 87. The back-barrier (west) shoreline of CUIS (Photo by Western Carolina University, from Peek et al. 2016).

While the eastern shoreline is shaped primarily by the open-ocean forces of wave action and winds, the western shoreline is shaped largely by tidal stream and inlet dynamics (Jackson 2010). Shorelines are also influenced by geologic makeup (e.g., resistance of substrate to erosion) and human activity (e.g., dredging, boat wakes, shoreline stabilization structures) (Jackson 2006). For at least a decade, the back-barrier shoreline of CUIS has been experiencing substantial erosion, which is threatening natural resources and cultural features as well as park infrastructure (Alber et al. 2005, Jackson 2006). This erosion washes away established marsh and upland habitat and destroys prehistoric archeological sites along the shore (Alber et al. 2005, NPS 2014a).

In addition to shorelines, dune fields are a common dynamic geographic feature of barrier islands. Dunes form when sands deposited above sea level by wave action are blown together by onshore winds to form hills of varying sizes (Griffin 1982). These dunes protect inland habitats and structures from wind and ocean waves, particularly during storms. The growth or deflation of island dunes depends on a number of factors, including climatic conditions (e.g., moisture, winds), the presence of stabilizing vegetation, and the supply of sand to the shore (McLemore et al. 1981, Cofer-Shabica 1993a). For example, if stabilizing vegetation is absent, high winds can cause dunes to migrate further inland, encroaching on other island habitats (McLemore et al. 1981). While dune erosion and migration are natural processes, they can be accelerated or slowed by human influence (e.g., recreational use, livestock grazing) (Hillestad et al. 1975).

4.10.2. Measures

- Back-barrier shoreline change
- Ocean shoreline change
- Dunefield dynamics

4.10.3. Reference Condition/Values

Selecting a reference condition for barrier island geomorphology is challenging, given that barrier island shorelines shift frequently as a result of natural processes. The earliest available documentation of shoreline positions at Cumberland Island are from the mid- to late-1800s. These sources have been used by researchers to calculate shoreline change over time, but the historic shoreline positions do not necessarily represent a "desirable" or "target" condition. For the purposes of this NRCA, recent CUIS shoreline change rates will be compared to historic change rates and to recent change rates from other Georgia barrier islands to provide insight regarding current condition. Given the limited information regarding barrier island dunefield dynamics, a reference condition for this measure is undetermined.

4.10.4. Data and Methods

During the late 1970s, the Georgia Geologic Survey conducted investigations of geologic resources and processes within CUIS to collect basic data that would aid the NPS in planning and management of the seashore (McLemore et al. 1981). While much of this effort focused on hydrogeology and subsurface features, the report briefly describes the dunes and dune migration rates at the time of the study. Additional studies of Cumberland Island's geomorphology were summarized in a 1993 Georgia Geological Society publication (Farrell et al. 1993). Topics included ocean shoreline change (Cofer-Shabica 1993b), the backdune ridge complex (Cofer-Shabica 1993a), and back-barrier shoreline change (Cofer-Shabica 1993c).

Pendleton et al. (2004) assessed the vulnerability of CUIS ocean coastline to SLR. As part of this assessment, historic ocean shoreline positions were mapped and erosion/accretion rates were calculated using Digital Shoreline Analysis System (DSAS) software and data provided by the USGS National Assessment of Coastal Change Hazards project.

Jackson (2006) studied spatial and temporal trends in back-barrier erosion at CUIS from 1857-2002. The earliest available information regarding Cumberland Island shoreline positions comes from 19th century U.S. Coast and Geodetic Survey topographic sheets. An 1857 survey covered the southern third of the island, the middle portion was covered by an 1867 survey, and the northern third was covered in an 1870 survey (Jackson 2006). Additional historic maps and imagery allowed Jackson (2006) to compare back-barrier over four periods: 1857/70-1933, 1933-1983, 1983-2002, and 1857/70-2002 (long-term net change). Shoreline positions were digitized from the historical sources into a spatial dataset so that shoreline change over time could be analyzed using GIS tools. Change was analyzed across 848 transects grouped into 10 "zones" along the back-barrier shoreline (Figure 88) (Jackson 2006). Later, Jackson (2010) used a statistical analysis package called AMBUR (Analyzing Moving Boundaries Using R) to assess shoreline change for multiple Georgia barrier islands, including CUIS, from 1855-2004.

At the request of the NPS, the USGS conducted an analysis of the CUIS ocean shoreline's vulnerability to inundation during a direct hurricane landfall (Stockdon et al. 2007). Vulnerability was assessed by comparing storm-induced mean-water levels (i.e., storm surge) to the elevation of the first dune crest beyond the beach. Dune elevations were derived every 20 m along the coast from a LiDAR topographic survey conducted by the USACE in January 2006 (Stockdon et al. 2007). The resulting report also presented rates of ocean shoreline change between October 1999 and January 2006.

Figure 88. The division of the back-barrier shoreline into zones for Jackson's (2006) shoreline change analysis (colored segments), and general locations of segments surveyed by Calhoun and Riley (2016) (black circles).

In late 2011/early 2012 and early 2013, the USGS conducted two surveys to quantify the magnitude of erosion along five segments of the back-barrier shoreline of CUIS: Cumberland Wharf (CW), Brickhill Bluff (BB), Plum Orchard (PO), Dungeness Wharf (DW), and Raccoon Keys (RK) (Figure 88) (Calhoun and Riley 2016). Segment lengths varied from 170 m (558 ft) at Cumberland Wharf to 400 m (1,312 ft) at Raccoon Keys. During survey site selection, priority was given to locations where habitats or cultural/historical resources are threatened by continued erosion. Researchers used AMBUR to quantify shoreline change between the two surveys and to project the shoreline positions along each segment in 2050 and 2100 (Calhoun and Riley 2016). At four of the survey locations (excluding Cumberland Wharf), additional monitoring equipment (standard bank pins and Photo-Electronic Erosion Pins [PEEPs]) was placed to gather more detailed measurements of erosion. Data collection from standard bank pins began in February 2012 and from PEEPs in May 2012. Data were also collected on water levels (i.e., wave height/tidal fluctuations) and boat traffic (using hydrophones and acoustic recording devices) in an effort to associate potential causes of erosion with back-barrier shoreline changes (Calhoun and Riley 2016).

Photos from two of the Calhoun and Riley (2016) survey locations: Cumberland Wharf (left, A) and Dungeness Wharf (right, D).

4.10.5. Current Condition and Trend

<u>Back-barrier Shoreline Change</u>

Some of the earliest measurements of back-barrier shoreline change at CUIS were reported by Cofer-Shabica (1993c). Surveys in October 1987 and June 1993 in the area between Sea Camp and Dungeness Dock found shoreline retreat as high as 9.2 m (30.2 ft) for an average loss rate of -1.6 m/yr (-5.2 ft/yr). Just north of Dungeness Dock, the shoreline receded 7.9 m (25.9 ft) from 1987-1993 for an average of -1.4 m/yr (-4.6 ft/yr) (Cofer-Shabica 1993c).

Jackson (2006, 2010) showed that net erosion has been the dominant trend along the CUIS back-barrier shoreline since the mid-1800s (Table 79). From 1855-2004, the overall change rate along the back-barrier has averaged -0.19 m/yr (-0.62 ft), with 77% of the shore experiencing net erosion (Jackson 2010). This was a slightly higher rate of erosion than was experienced along the Georgia coast overall, which showed a mean of -0.10 m/yr (-0.33 ft). In some erosion “hotspots” at CUIS, such as near the Dungeness Dock, the long-term erosion rate reached as much as -1.76 m/yr (-5.77 ft/yr) (Jackson 2010). From 1974-2004 alone, the CUIS back-barrier shoreline receded at an average rate of -0.48 m/yr (-1.57 ft/yr), a higher rate of erosion than any of the historical periods studied (Table 79), with 81% of the shoreline experiencing net erosion.

Table 79. A comparison of mean back-barrier shoreline change rates (m/yr) and the percent of back-barrier shoreline experiencing erosion between Cumberland Island and Georgia coast-wide rates over various historical periods and long-term (1855-2004) (Jackson 2010). Negative change rates represent erosion and positive rates represent accretion.

	Cumberland Island		Georgia Coast-Wide	
Era	Shoreline Change Rate	% Erosion	Shoreline Change Rate	% Erosion
1855-1933	-0.34	80	-0.15	64
1933-1951	0.26	32	0.27	42
1951-1974	0.25	43	0.20	41
1974-2004	-0.48	81	-0.39	78
1855-2004	-0.19	77	-0.10	65

A closer look at back-barrier shoreline change by zone (as mapped by Jackson 2006) reveals several erosional “hotspots”, often near tidal inlets. The most notable erosion was documented along Zones II, VI, VIII, X (Figure 89). The highest long-term erosion rate was along Zone II (-0.68 m/yr [-2.23 ft/yr]), followed by Zone X (-0.34 m/yr [-1.12 ft/yr]) (Table 80) (Jackson 2006). For the most recent period assessed (1983-2002), Zone IX experienced the highest erosion rate at -2.48 m/yr (-8.14 ft/yr). Two other segments – Zones II and VIII – showed erosion rates near -2.0 m/yr (-6.6 ft/yr) during this period. From 1983-2002, 85% of the back-barrier shoreline experienced erosion and seven of the 10 zones showed higher erosion rates than during previous periods (Jackson 2006). Only Zone I experienced substantial accretion (0.64 m/yr [2.10 ft/yr]) during this time (Table 80). Figure 89 below displays the long-term trends (1855-2002) by zone (colored segments) and the rates of change for the most recent period (1983-2002) (numbers along the shore). Ground and aerial photos showing back-barrier shoreline erosion at CUIS are included in Appendix J.

Calhoun and Riley’s (2016) study focused further on the erosion hotspots identified by previous research, with repeat surveys and sampling from late 2011 through 2013. Erosion was highly variable across selected segments, ranging from no net erosion to a maximum of 2.5 m (8.2 ft) of retreat over the year-long study period (Calhoun and Riley 2016). Mean erosion rates detected through repeat surveys varied from 0.25 m/yr (0.82 ft/yr) at Cumberland Wharf to 0.77 m/yr (2.53 ft/yr) at Raccoon Keys (Table 81). Standard bank pin measurements detected erosion ranging from 0.59 m/yr (1.94 ft/yr) at Dungeness Wharf to 1.0 m/yr (3.28 ft/yr) at both Brickhill Bluff and Raccoon Keys. Mean erosion rates for each site based on the two study methods (repeat surveys and bank pins) were generally similar (i.e., within 0.5 m/yr [1.6 ft/yr]), although bank pins detected higher maximum erosion rates for two of the locations as well as the overall highest erosion rate of the two methods (Calhoun and Riley 2016).

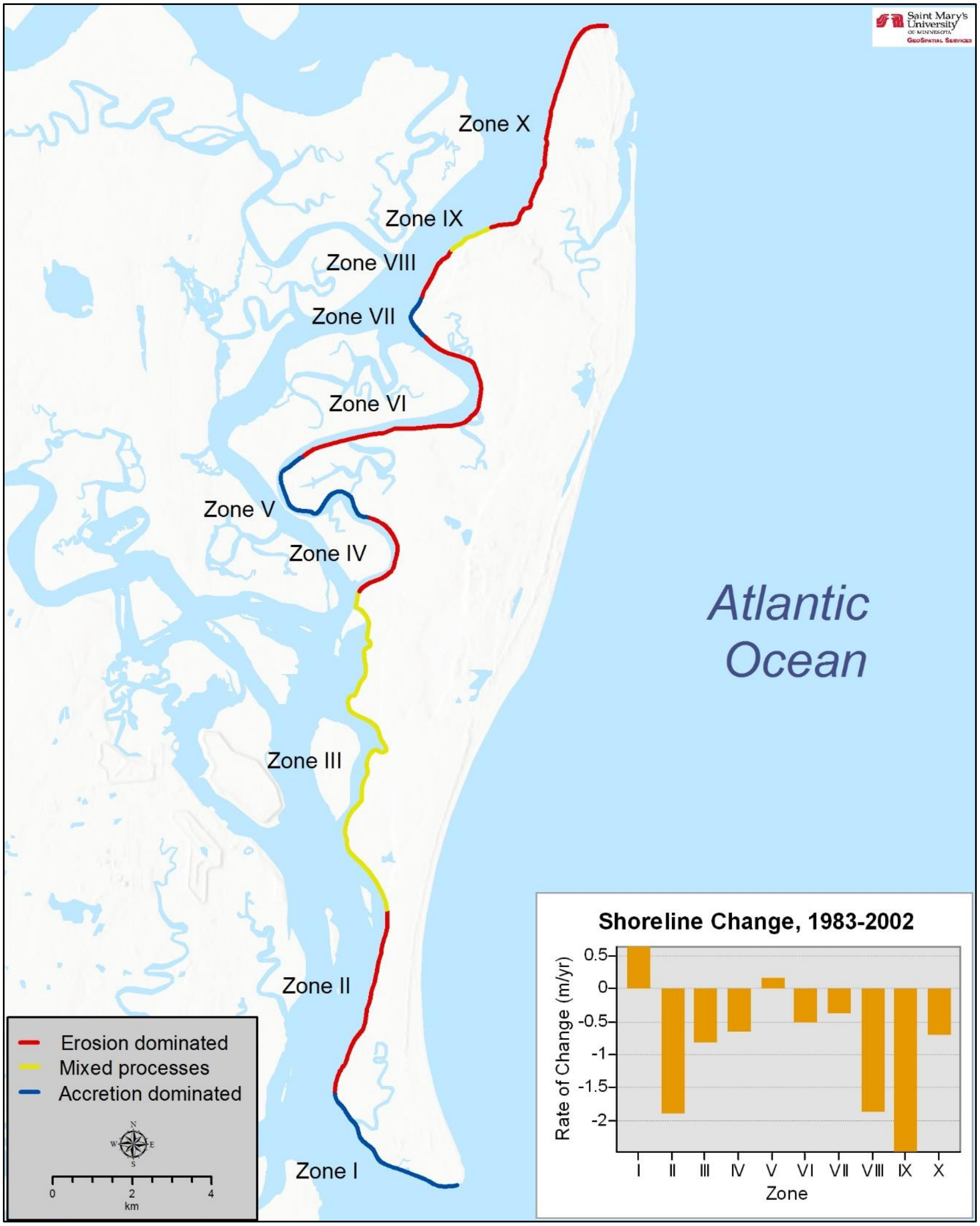


Figure 89. Shoreline change along the CUIS back-barrier, as reported by Jackson (2006). The colored segments represent the long-term trend (1855-2002) by zone, while the bar graph shows the rate of change for the most recent study period (1983-2002). Negative numbers = erosion, positive numbers = accretion.

Table 80. Back-barrier shoreline change rates (m/yr) for CUIS by zone, 1857-2002 (Jackson 2006).

Segment	1857/70-1933	1933-1983	1983-2002	1857/70-2002
Zone I	-0.15	1.66	0.64	0.08
Zone II	-0.87	0.10	-1.90	-0.68
Zone III	0.20	0.25	-0.82	0.07
Zone IV	-0.14	-0.17	-0.65	-0.23
Zone V	0.36	0.16	0.17	0.27
Zone VI	-0.55	0.31	-0.50	-0.22
Zone VII	0.03	0.90	-0.36	0.30
Zone VIII	-0.32	0.43	-1.87	-0.28
Zone IX	0.33	1.00	-2.48	0.16
Zone X	-0.68	0.26	-0.70	-0.34

Table 81. Shoreline position change (m/yr) at selected CUIS back-barrier segments, as measured by two different methods (Calhoun and Riley 2016).

	Repeat Surveys (2011 & 2013)			Standard Bank-Pins (2012-2013)		
Site	**Minimum**	**Maximum**	**Mean**	**Minimum**	**Maximum**	**Mean**
Cumberland Wharf	0	1.41	0.25	N/A	N/A	N/A
Brickhill Bluff	0	1.99	0.47	0.08	2.50	1.00
Plum Orchard	0	1.94	0.26	0.01	2.19	0.72
Dungeness Wharf	0	2.40	0.37	0	1.71	0.59
Raccoon Keys	0	2.34	0.77	0.32	2.27	1.00

Back-barrier shoreline erosion north of the Dungeness seawall that has exposed an archeological site (shell middens) (SMUMN GSS photo).

Calhoun and Riley (2016) noted two different patterns in terms of the timing of erosion along the CUIS back-barrier shoreline. Of the four sites sampled intensively, erosion at three (BB, PO, and DW) was dominated by "punctuated erosional events that were coincident with above-average high tides and elevated wind speeds" (Calhoun and Riley 2016, p. 1). Only one site (RK) showed steady, low-magnitude erosional retreat across the study, with little response to particular events. This difference may be related to both natural and anthropogenic factors. The three locations that showed punctuated erosion consist of sand-dominated substrates while the RK substrate consists of a dense sand and clay mixture with high organic content, topped by peat; the latter substrate is much more resistant to abrupt erosion events than the former (Calhoun and Riley 2016). In addition, the RK site experienced the highest boat traffic of any of the sites, which may be exposing the shore to elevated wave impacts from boat wakes.

Ocean Shoreline Change

Since the mid-1800s, shoreline accretion has been more common than erosion along the CUIS oceanfront (Jackson 2010). Overall, the shoreline prograded (i.e., grew or advanced) at an average rate of 1.28 m/yr (4.20 ft/yr) over the period 1855-2004 (Jackson 2010). This was above the mean shoreline change rate for the Georgia coast as a whole of 0.64 m/yr (2.10 ft/yr) (Table 82). During this time, only 20% of the CUIS ocean shoreline experienced net erosion, compared to 40% for Georgia coast-wide. From 1974-2004 alone, the CUIS shoreline advanced at a rate of 1.72 m/yr (5.64 ft/yr), with only 38% of shoreline experiencing net erosion. In comparison, the Georgia coast shoreline advanced an average of 0.55 m/yr (1.80 ft/yr), with 50% of the shoreline experiencing erosion (Table 82) (Jackson 2010).

Table 82. A comparison of mean oceanfront shoreline change rates (m/yr) and the percent of ocean shoreline experiencing erosion between Cumberland Island and Georgia coast-wide rates over various historical periods and long-term (1855-2004) (Jackson 2010). Negative change rates represent erosion and positive rates represent accretion.

	Cumberland Island		Georgia Coast-Wide	
Era	Shoreline Change Rate	% Erosion	Shoreline Change Rate	% Erosion
1855-1933	1.17	38	-0.03	56
1933-1951	2.20	25	3.44	39
1951-1974	0.79	43	0.70	52
1974-2004	1.72	38	0.55	50
1855-2004	1.28	20	0.64	40

The most notable accretion at CUIS occurred on the southern end of the island, associated with construction of the jetty, which began in 1881 (Figure 90). From 1855-2004, the shoreline adjacent to the jetty accreted around 1.6 km (~1 mi) for an average rate of 10.78 m/yr (35.37 ft/yr) (Jackson 2010). The majority of this advancement occurred after 1933. Shoreline changes along the northern end of CUIS were more variable due to the dynamics of the Christmas Creek Inlet. This area has experienced periods of erosion and accretion over time, with a net loss of 107 m (351 ft) between 1855-2004 (Jackson 2010). This averages to an erosion rate of -0.52 m/yr (-1.71 ft/yr).

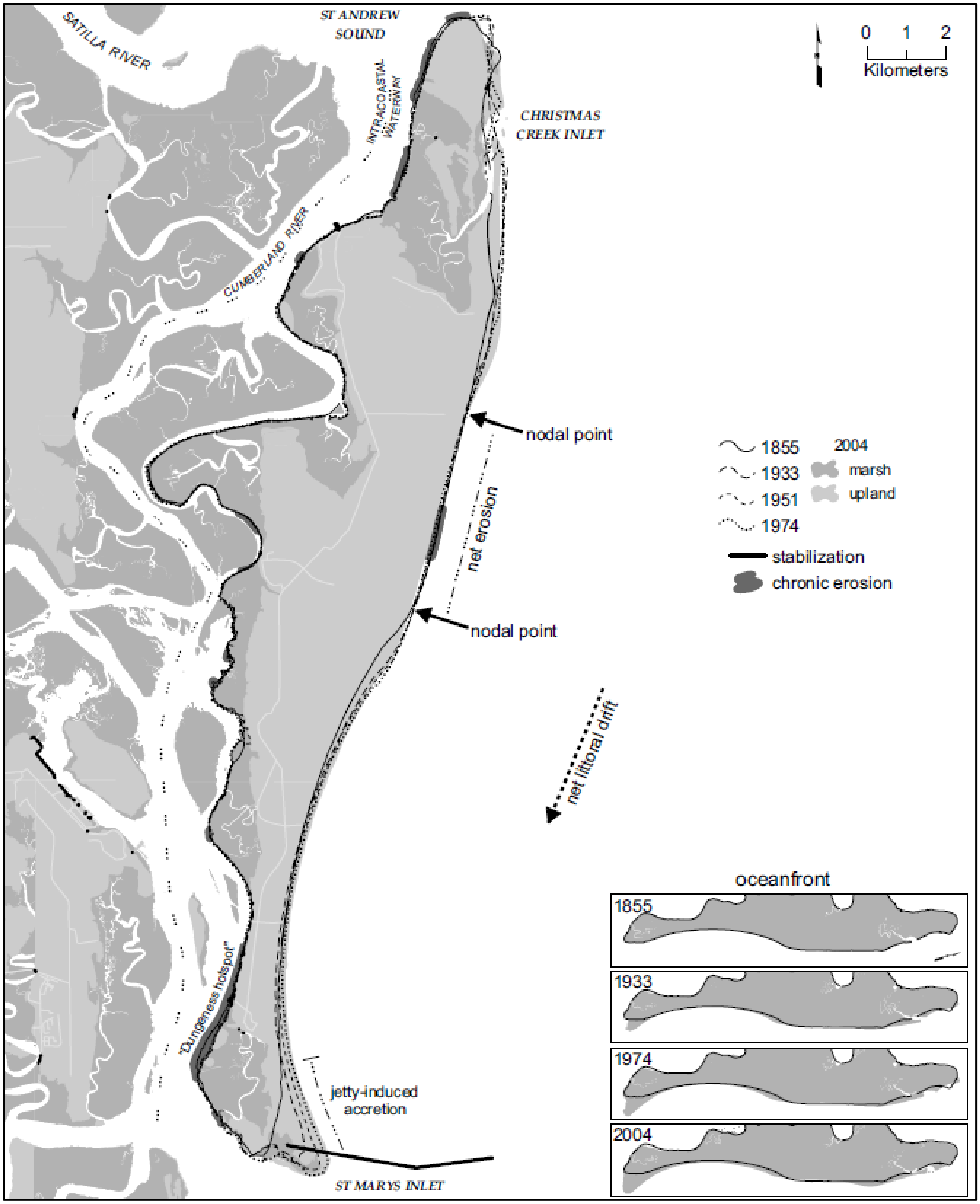


Figure 90. Historical shoreline change at CUIS, 1855-2004. The light and medium gray shading represent the 2004 extent of Cumberland Island while the dark gray shading highlights areas of chronic shoreline erosion. In the "oceanfront" insets, the black border represents the 1855 shoreline position (Jackson 2010).

One ocean shoreline segment in the central portion of the island has experienced primarily erosion (Figure 90). Over the nearly 150-year period through 2004, this area showed a mean erosion rate of -0.89 m/yr (-2.92 ft/yr). A more detailed illustration of CUIS oceanfront shoreline change over time (through 1993) is included in Appendix K.

A study of CUIS ocean shoreline change that focused on the period 1999-2006 found similar results. During this time, the mean shoreline change rate was 1.98 m/yr (6.50 ft/yr) (excluding Little Cumberland Island which experienced inlet effects), with 80% of the entire shoreline experiencing accretion (Stockdon et al. 2007). Again, a 5-km (3.1-mi) stretch of shoreline in the north-central portion of the island was the largest area to experience net erosion (Figure 91). On Cumberland Island proper (excluding Little Cumberland Island), the magnitude of shoreline change along the coast over these 6 years ranged from 120.70 m (396.00 ft) of accretion, for a rate of 19.31 m/yr (63.35 ft/yr), to 105.31 m (345.51 ft) of erosion, at a rate of -16.85 m/yr (-55.28 ft/yr) (Stockdon et al. 2007).

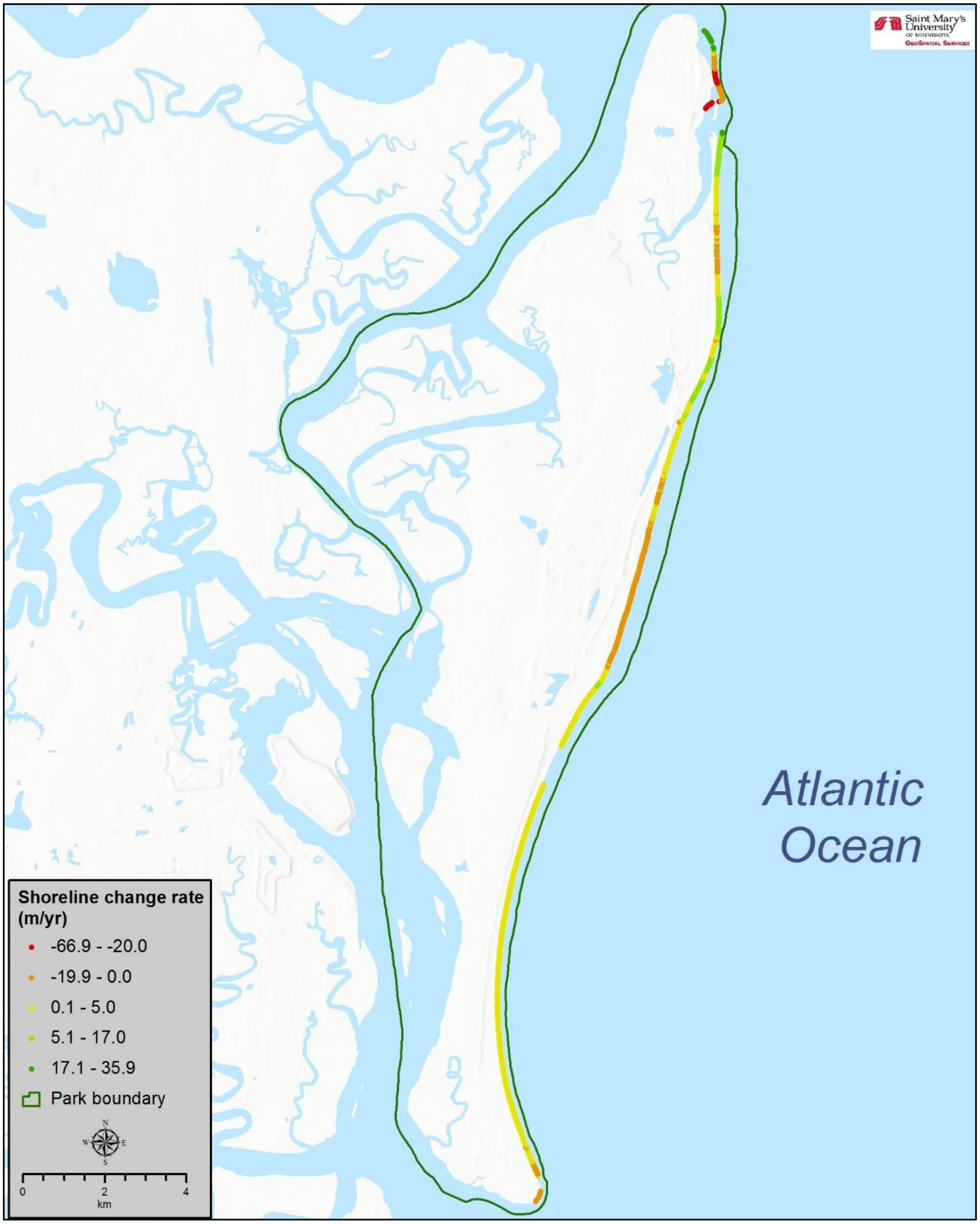


Figure 91. Shoreline change rates (m/yr) for the CUIS ocean coast, 1999-2006 (Stockdon et al. 2007). Red and orange indicate net erosion while yellow and greens show net accretion.

Dunefield Dynamics

The dunes at CUIS run nearly the entire length of the island, just behind the beach on the Atlantic shoreline, with crests reaching as high as 13.7 m (45 ft) towards the north end (NPS 2014a). The quartz sand comprising these dunes originates along the beaches, often when inlet shoals from the wave zone migrate onto the beach as sand bars (Hillestad et al. 1975). When these sands dry, prevailing easterly winds carry sand westward to replenish and/or expand the dunes. Over time, the dunes also migrate landward (west) in response to the prevailing winds (McLemore et al. 1981, Cofer-Shabica 1993a). Little scientific study has been conducted into dunefield dynamics at CUIS. The limited information available will be presented here, but provides little insight into the current condition of the park's dunefields.

The dune ridge at CUIS (NPS photo).

Dune height, stability, and migration rates vary across the island (Hillestad et al. 1975, McLemore et al. 1981). Historically, foredunes were absent in the south and dunes there were generally lower than in the central and northern portions of the island, due to a lack of stabilizing vegetation (Hillestad et al. 1975). At the time of park establishment, large, unstable dunes were observed encroaching into adjacent forests and other habitats in some areas of CUIS. The most notable examples were in the vicinity of Whitney Lake and Sweetwater Lake in the north and central portions of the island and near the Dungeness Beach Field in the south (Hillestad et al. 1975). Sands were reportedly filling important wetland habitat in the Whitney and Sweetwater Lake areas.

According to McLemore et al. (1981), dune migration in an area adjacent to forest in the southern portion of the island (north of Dungeness, near Nightingale Avenue) was measured at 0.4 m (1.4 ft) in just 9 months. This equates to approximately 0.6 m/yr (1.9 ft/yr). However, dune migration at Beach Fields was reportedly much higher, at 0.9 m/yr (3.0 ft/yr). This higher rate was likely due to the sparse, herbaceous vegetation at the Beach Fields, which offers little resistance to sand movement

(McLemore et al. 1981). A decade later, Cofer-Shabica (1993a) reported that dune migration at Beach Fields was 1.2-1.9 m/yr (3.9-6.2 ft/yr). Black-and-white photos illustrating dune migration in the Dungeness area between 1980 and 1993 are shown below in Figure 92.

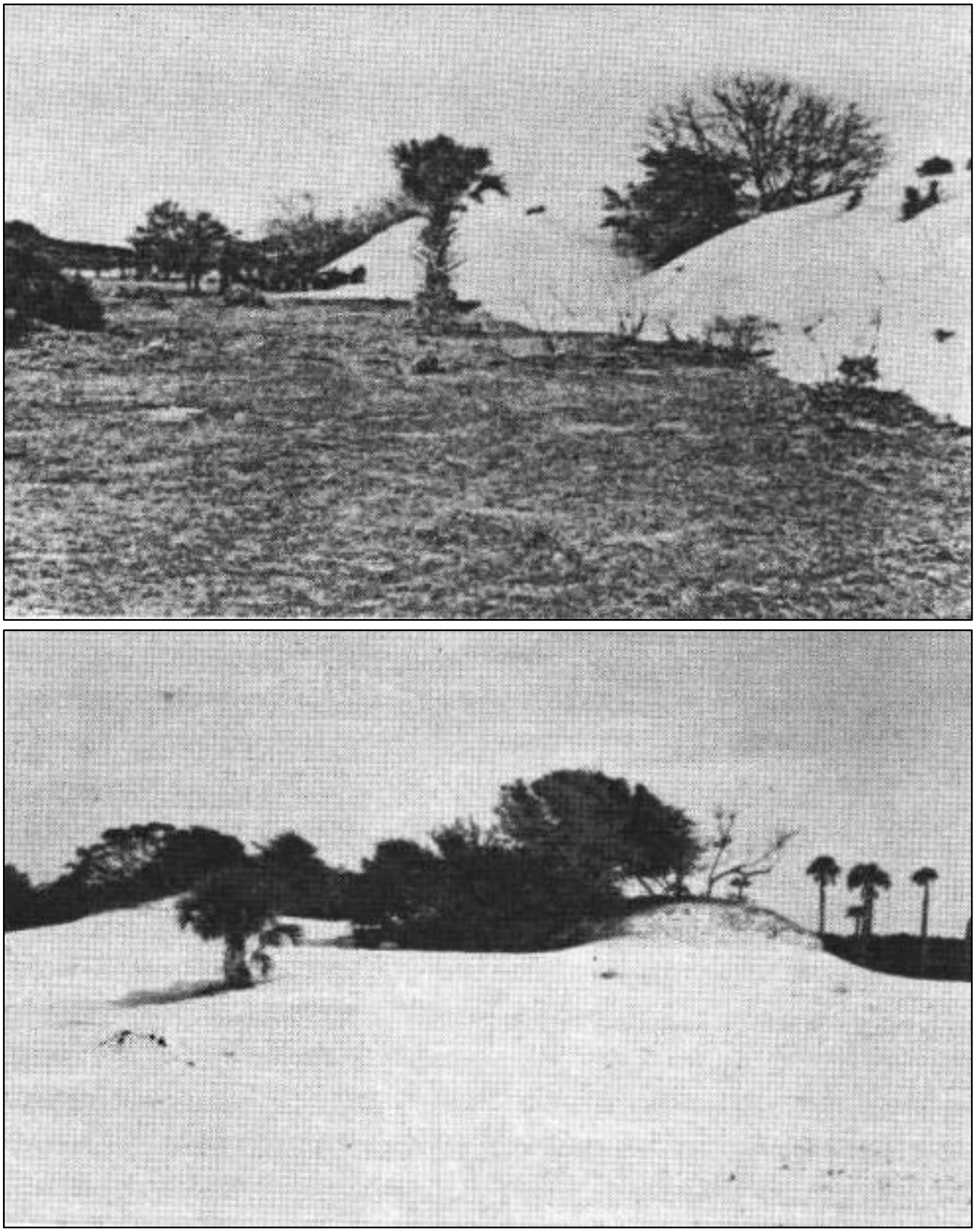


Figure 92. Photos of the back dune at Dungeness Crossing in 1980 (above) and in October 1993 (below). Note the location of the palm at the dune edge in 1980 and engulfed by the dune in 1993 (Cofer-Shabica 1993a).

Cofer-Shabica (1993a) also reported on changes in back dune ridge height at the Dungeness Dune Crossing. In 1978, researchers measured the elevation of the dune ridge crest at 7.91 m (25.95 ft) above Beach Field ground level. In September 1993, the ridge crest was measured again and was found to be just 3.86 m (12.66 ft) above Beach Field ground level (Cofer-Shabica 1993a). Over 15 years, the dune ridge was reduced by just over 4.0 m (~13 ft), or at a rate of 0.27 m/yr (0.89 ft/yr). The sand supply to this back dune had clearly been reduced during this time. Cofer-Shabica (1993a) hypothesized that this may have been due to the maturation of the foredune complex and interdune meadow to the east, so that these areas trapped more sand before it reached the back dune ridge.

As part of their study of CUIS coastal vulnerability to hurricanes, Stockdon et al. (2007) derived crest elevations for the easternmost dune ridge from 2006 LiDAR data. Dune crest elevations ranged from 1.28 m (4.20 ft) to 13.53 m (44.39 ft) with a mean of 4.24 m (13.91 ft). The vast majority of elevation measurements (97%) were below 7 m (23 ft). A graphical representation of dune ridge elevations and location can be found in Appendix L.

<u>Threats and Stressor Factors</u>

Threats to CUIS's shorelines and/or dunefields include erosion, natural ocean/inlet processes (e.g., wind, waves, tides), storm events, hardened shoreline structures (e.g., seawalls, jetties, rip-rap), feral animals (e.g., horse grazing and trampling, hog rooting), dredging (mostly historic), boat traffic, and increased visitor use. Many of these threats have been discussed in previous components and will be only briefly reviewed here.

At a basic level, the factor thought to most influence coastal erosion is wave action (Calhoun and Riley 2016). Waves can directly remove sediment through physical contact, or at lower water levels they may undercut banks and bluffs, causing the eventual collapse of the soil above. Wave action can be magnified by natural events (e.g., storms) or by human influence (e.g., boat wakes), and can be compounded when certain factors coincide (e.g., when strong, sustained winds occur during high tide). For example, high storm tides are suspected to be undercutting the bluff in the vicinity of Cumberland Wharf, causing slumping events along the bluff (Calhoun and Riley 2016). As a result of continuing SLR (discussed in Chapter 2), high tide levels are expected to become even higher along the Atlantic coast (Dahl et al. 2017), exposing additional shoreline to wave action. At the Fernandina Beach tidal gauge just south of CUIS, tidal flooding events are projected to increase from an average of 1.9 events/year (2000-2015) to 9.2 events/year by 2030 and 40.1 events/year by 2045 (Dahl et al. 2017).

Hurricanes and strong storms (e.g., northeasters) can have significant impacts on barrier island shorelines and dunefields, even if the storms make landfall some distance away (Shabica et al. 1993, Jackson 2006, Stockdon et al. 2007). The sustained winds from hurricanes can elevate tides and wave heights for at least 100 km (62 mi) from the center of the storm (Jackson 2010). In 1964, Hurricane Dora reportedly washed away dunes and lowered some beaches in the Cumberland Island vicinity by 1.5 m (4.9 ft) (Shabica et al. 1993). In 2004, some breaching occurred within CUIS's primary dune system due to an active storm season (Alber et al. 2005). Figure 93 shows erosion that occurred during Hurricane Matthew in 2016 and Figure 94 shows shoreline change after Hurricanes Matthew and Irma. While storm events might be expected to primarily impact the oceanfront coast of the island, Calhoun and Riley (2016) found that storm-driven erosion was the dominant factor in shoreline change at three of the four back-barrier erosion hotspots studied at CUIS. As a result of global climate change, the intensity of hurricanes is projected to increase over the next century (Knutson et al. 2010), which may in turn increase storm impacts on barrier islands such as CUIS.

Figure 93. A view of the CUIS ocean shoreline and beach on the northeast side of the island, before (27 Sept 2016, left) and after (19 Nov 2016) Hurricane Matthew (NPS photos courtesy of Lisa Baron). The grassy foredunes present in the September picture were lost due to hurricane impacts.

Figure 94. Photos of the CUIS ocean shoreline from an established point near the southern end of the island (NPS photos). Photos are from 2012, 2016 (before Hurricane Matthew), and 2017 (after Hurricanes Matthew and Irma). Top photos are facing south, bottom photos are facing north. Shoreline vegetation clearly advanced between 2012 and 2016 but was lost, along with actual dunes, during the two storms.

Stockdon et al. (2007) assessed the vulnerability of CUIS's ocean coastline to inundation during hurricanes of varying strengths. Inundation was predicted to occur when the modelled storm-induced mean-water level exceeds the elevation of the most seaward sand dune crest. During inundation, sediment (i.e., sand) transport is likely to occur, which often results in larger magnitude beach erosion and shoreline retreat (Stockdon et al. 2007). The assessment showed that just 10% of CUIS shoreline is vulnerable to inundation during a Category I storm, but 97% would be vulnerable during a Category V storm (Figure 95). The central portion and the northern end of the island are most susceptible to inundation due to the low dune elevations in these areas (Stockdon et al. 2007).

In some areas, the shoreline has been artificially stabilized with hardened structures to prevent erosion. One of the largest examples is the Dungeness seawall, a 260-m (853-ft) long structure first constructed in the early 1900s (Figure 96) (Jackson 2006). While structures such as this protect some shore segments from erosion, they can focus and magnify erosion along adjacent segments (i.e., "end-around effects"). At Dungeness, the shoreline just north and south of the seawall have retreated by 67 m (220 ft) and 132 m (433 ft), respectively, since 1857 (Jackson 2006). Similar end-around effects have been observed around a 15-m (49-ft) long wooden bulkhead near Plum Orchard (see photos in Appendix K).

Channel alterations within Cumberland Sound and St. Marys Inlet have influenced natural flow and sediment dynamics in a way that has likely impacted island geomorphology (Shabica et al. 1993, Jackson 2006). As discussed in Chapter 2, south end jetty construction and subsequent dredging have altered tidal prism (the amount of water that flows in and out between high and low tides) and sediment deposition patterns (Shabica et al. 1993, Jackson 2006). Channel alteration to accommodate submarines at the naval base was also shown to alter current velocity in Kings Bay, just west of CUIS (Alber et al. 2005, Calhoun and Riley 2016). The jetty, which was constructed to stabilize the inlet channel for boat traffic, has caused an unnatural amount of accretion to occur at the southern end of CUIS, extending the tip of the island by approximately 1.6 km (~1 mi) since the late 1850s (Pendleton et al. 2004, Jackson 2010).

Grazing and trampling by feral horses and rooting by feral hogs along the shore and in the dunes can remove stabilizing vegetation, making these areas more vulnerable to erosion (NPS 2014a, Calhoun and Riley 2016). The grazing of free-ranging livestock was thought to be the primary factor in dune instability and encroachment around the time of park establishment (Hillestad et al. 1975).

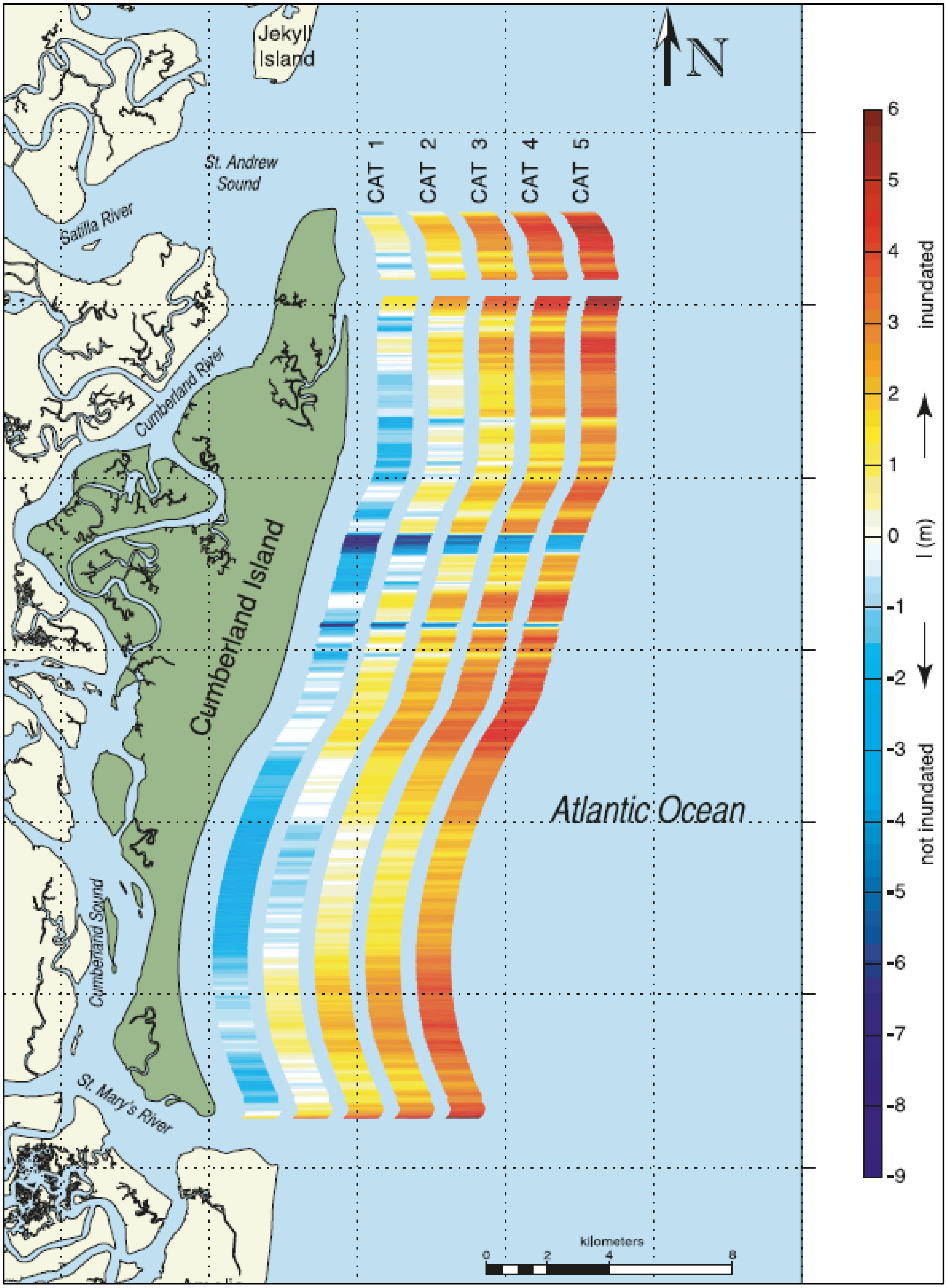


Figure 95. The potential for inundation of the CUIS beach system during Category 1-5 hurricanes. Positive numbers (red to yellow) indicate that modeled storm surge exceeds dune crest elevation, making that portion of the coast more vulnerable to extreme changes due to inundation (Stockdon et al. 2007).

Figure 96. The Dungeness seawall at low tide (SMUMN GSS photo).

Visitor use, both on land and water, can contribute to shoreline and dune erosion. Recreational boat traffic, particularly during high tides, increases wave frequency and height, which can accelerate shoreline erosion (Calhoun and Riley 2016). Proposed developments along the coast near CUIS could increase recreational boating along the island's shores, particularly on the south end (Alber et al. 2005, Jackson 2006). The beaches on the south tip of the island are one of the few places where boats can reliably land without a dock or structure (Fry, personal communication, March 2017). Occasionally during the summer, up to 30 boats have been seen on shore during a single day (NPS 2010). The grounding and anchoring of boats in this area may physically alter the shoreline and disrupt sediment dynamics in the vicinity (Alber et al. 2005). In the dunefields, vehicle and foot traffic can lower dunes and/or remove stabilizing vegetation, potentially causing a "blow-out" (McLemore et al. 1981). A blow-out occurs when a small dune area is lowered by repeated traffic and the prevailing winds are concentrated and funneled through this opening. Sand transport increases in this area, causing continued lowering of the dune crest and widening at its base. This lowering reduces the dune's ability to buffer the island from storm impacts (McLemore et al. 1981).

Data Needs/Gaps

According to Jackson (2010), Georgia is the least studied coastal state with regard to shoreline change, and until recently most studies in the state focused on oceanfront coastlines. The processes that cause back-barrier erosion and the mechanisms for accretion along barrier islands such as CUIS are poorly understood (Jackson 2006, Calhoun and Riley 2016). Continued data collection and analysis are needed to better understand the factors contributing to back-barrier erosion and to develop mitigation/management strategies (Jackson 2006, NPS 2014a). Specifically, Jackson (2006) recommended establishing permanent monitoring stations on the back-barrier shoreline; conducting RTK (real time kinematic) GPS surveys of the back-barrier, oceanfront, and inlet shorelines on an annual basis and immediately after storm events; and collecting LiDAR data (to construct digital terrain models of the island) and a complete set of digital orthophotographs that includes both shoulders of the bordering inlets every 5 years. In addition, Jackson (2006) suggested that the NPS investigate shoreline stabilization alternatives such as a "living shoreline", which could be tested at erosion hotspots such as Plum Orchard. Living shorelines utilize natural materials such as live vegetation, sand, rock, or oyster shells to protect and stabilize coastal areas (NOAA 2017b). CUIS is also in need of an official shoreline management plan (Jackson 2006, NPS 2014a).

Park staff have conducted photo point shoreline surveys on both sides of the island "to monitor and interpret the degree of erosion and accretion which occurs along the shoreline" (NPS 2013c, p. 1). The first photo point survey was conducted in 2000 but was not repeated until 2012, at which time the monitoring protocol was revised to better capture erosion/accretion activity at selected sites. Permanent reference markers are established 1 km (0.6 mi) apart on both shorelines, and the survey protocol involves taking two photos at each marker (NPS 2013c). To date, no in-depth evaluation or analysis of this photo monitoring has been completed. Exploration and comparison of these photos may provide useful visual evidence and some insight into shoreline change at CUIS in recent decades. Photos from the east shoreline may also capture changes in the island's dunefields. A map of photo point monitoring locations and a sampling of repeat photos from selected points are included in Appendix M.

Very little is known about dunefield dynamics at CUIS. However, it may be possible to determine historic dune characteristics such as ridge locations from historic aerial imagery. GIS analysis could then be used to analyze changes in the dunefield over time. Supplemental information such as LiDAR data showing dune elevations, if available for different time periods, could also be used to explore dunefield changes (e.g., dune building, erosion, migration). Such analysis is beyond the scope of this NRCA project.

Overall Condition

Back-barrier Shoreline Change

The NRCA project team assigned this measure a *Significance Level* of 3. Jackson (2006) described the widespread erosion along CUIS's back-barrier shoreline in recent decades (a relatively short period) as alarming. While the long-term (1855-2004) and most recent available (1974-2004) mean shoreline change rates for the back-barrier as a whole are just above Georgia coast-wide means, some segments along the back-barrier have experienced erosion rates near or exceeding 2 m/yr (6.6 ft/yr) (Jackson 2006, Calhoun and Riley 2016). Erosion is likely to increase if the frequency and magnitude of tidal flooding and strong storms also increases. Therefore, this measure is assigned a *Condition Level* of 3, indicating significant concern.

Ocean Shoreline Change

This measure was also assigned a *Significance Level* of 3. The general long-term trend for the ocean shoreline of CUIS has been accretion, with only small areas of the shore experiencing net erosion (Stockdon et al. 2007, Jackson 2010). Very high rates of accretion, averaging 10.78 m/yr (35.37 ft/yr) from 1855-2004, occurred at the southern end of the island near the jetty protecting the St. Marys Inlet (Jackson 2010). As a result, this portion of the island advanced approximately 1.6 km (~1 mi) over nearly 150 years (Figure 90). There is some concern that this unnaturally high accretion and advance may influence other aspects of the island's geomorphology in this area. A *Condition Level* of 2, indicating moderate concern, has been assigned for this measure.

Dunefield Dynamics

A *Significance Level* of 3 was assigned for this measure. While some data is available regarding historic dune migration rates and more recent dune crest elevations, this is not enough information to assess current condition. Therefore, a *Condition Level* cannot be assigned for this measure. It is likely that the island's dunes are more stable at present than they were at the time of park establishment (mid-1970s) due to reduced grazing pressure, but no scientific studies have confirmed this.

Weighted Condition Score

The *Weighted Condition Score* for CUIS barrier island geomorphology is 0.83, indicating significant concern. An overall trend could not be determined for several reasons, including the limited information regarding dunefield dynamics and a lack of recent information (post-2006) on ocean shoreline change. This also resulted in the use of a moderate confidence border (Table 83).

Table 83. Weighted Condition Score for Barrier Island Geomorphology in CUIS.

Barrier Island Geomorphology			
Measures	**Significance Level**	**Condition Level**	**WCS = 0.83**
Back Barrier Shoreline Erosion	3	3	
Ocean Shoreline Erosion	3	2	
Dunefield Dynamics	3	n/a	

4.10.6. Sources of Expertise

Lisa Baron, SECN Coastal Ecologist

John Fry, CUIS Chief of Resource Management

Linda York, NPS Regional Coastal Geomorphologist

5. Discussion

Chapter 5 provides an opportunity to summarize assessment findings and discuss the overarching themes or common threads that emerged for the featured components. The data gaps and needs identified for each component are summarized and the role these play in the designation of current condition is discussed. Also addressed is how condition analysis relates to the overall natural resource management issues of the park.

5.1. Component Data Gaps

The identification of key data and information gaps is an important objective of NRCAs. Data gaps or needs are those pieces of information that are currently unavailable, but are needed to help inform the status or overall condition of a key resource component in the park. Data gaps exist for most key resource components assessed in this NRCA. Table 84 provides a detailed list of the key data gaps by component. Each data gap or need is discussed in further detail in the individual component assessments (Chapter 4).

Table 84. Identified data gaps or needs for the featured components.

Component	Data Gaps/Needs
Upland Forest Community	➢ Further study of tree regeneration in oak maritime forests and longleaf pine communities ➢ Monitoring of redbay to determine how the species is recovering from LWD, and study to detect impacts of redbay loss/reduction on other components of maritime forests ➢ Research into the ecological roles of soil microbiota and mycorrhizae in upland forests, as well as the role of lichens in maritime forests
Freshwater Wetlands	➢ Additional research into CUIS's wetland vegetation communities, soils, hydrologic regime, and ecosystem processes ➢ Measurements of surface water dynamics and groundwater transmissivity, and modeling of groundwater dynamics ➢ Soil quality data collection and additional water quality monitoring
Salt Marshes	➢ Re-evaluation of the percent of salt marsh area grazed by horses versus ungrazed ➢ Further investigation of community stressors (e.g., boat wakes, SLR, coastal storms, grazing) ➢ Monitoring of high fringing salt marsh to determine if cordgrass low marsh is encroaching
Interdune Communities	➢ Focused survey to better understand the distribution and interaction of various vegetation communities and plant species ➢ Documentation of environmental variables (e.g., climate, soils, surface water presence) to understand their influence on the community

Table 84 (continued). Identified data gaps or needs for the featured components.

Component	Data Gaps/Needs
Mammals	➢ Continuation of annual trail camera surveys and scent station surveys ➢ Investigation of the marine mammal community of the park's waters ➢ Semi-annual inventories to update species richness and abundance estimates, with particular attention to the reintroduced bobcat population ➢ Annual bat surveys to document abundance and to monitor for WNS
Birds	➢ Continuation of SECN landbird monitoring efforts, including identification of potential trends in species presence or abundance ➢ Broad study of nesting shorebird population as a whole and fledging success ➢ Research into wading bird nesting numbers and fledging success
Herpetofauna	➢ Continuation of SECN amphibian monitoring ➢ Continued monitoring of the CUIS gopher tortoise population and burrow counts; development of a management plan for the species ➢ Research into the impacts of climate change on reptiles and amphibians, including sea turtles and their nesting habitat
Water Quality	➢ Additional data for all selected measures to detect any changes over time ➢ Study of water quantities and freshwater hydrology on the island, as they have significant influence on water quality ➢ Collection of water quality data from similar barrier island freshwater systems, to offer points for comparison
Air Quality	➢ Data collection specifically within CUIS boundaries ➢ Studies regarding the potential effects of air pollutants, particularly mercury, on island resources
Barrier Island Geomorphology	➢ Continued back-barrier erosion data collection and analysis to better understand contributing factors and to develop management/mitigation strategies (e.g., permanent monitoring stations, regular collection of LiDAR and digital orthophotos) ➢ Evaluation/analysis of photo point shoreline survey results ➢ Study of dunefield dynamics using historic aerial imagery and GIS analysis

Many of the park's data needs involve the continuation of current monitoring in order to fully document and describe the vegetation and wildlife communities of CUIS. Continued monitoring will also assist researchers and managers in identifying changes to these resources. Other components would benefit from research into how environmental factors or various threats influence park resources.

5.2. Component Condition Designations

Table 85 displays the conditions assigned to each resource component presented in Chapter 4 (definitions of condition graphics are located in Table 86 and Table 87). It is important to remember that the graphics represented are simple symbols for the overall condition and trend assigned to each component. Because the assigned condition of a component (as represented by the symbols in Table 85) is based on a number of factors and an assessment of multiple literature and data sources, it is strongly recommended that the reader refer back to each specific component assessment in Chapter 4 for a detailed explanation and justification of the assigned condition. Condition designations for some components are supported by existing datasets and monitoring information and/or the expertise of NPS staff, while other components lack historic data, a clear understanding of reference conditions (i.e., what is considered desirable or natural), or even current information.

For featured components with available data and fewer data gaps, assigned conditions varied. Three components are considered to be in good condition: salt marshes, mammals, and herpetofauna. Just one component (upland forest community) is of moderate concern. Air quality and barrier island geomorphology are of high concern, primarily due to outside influences upon the park. Condition could not be assigned for four of the ten components (Table 85).

Table 85. Summary of current condition and condition trend for featured NRCA components.

Category	Component	WCS	Condition
Ecological communities	Upland Forest Community	0.43	
	Freshwater Wetlands	N/A	
	Salt Marshes	0.33	
	Interdune Communities	N/A	
Wildlife	Mammals	0.13	
	Birds	N/A	
	Herpetofauna	0.24	
Environmental Quality	Water Quality	N/A	
	Air Quality	0.8	

Table 85 (continued). Summary of current condition and condition trend for featured NRCA components.

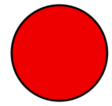
Category	Component	WCS	Condition
Physical Characteristics	Barrier Island Geomorphology	0.83	

Table 86. Symbols used to indicate condition, trend, and confidence in the assessment.

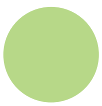
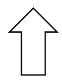
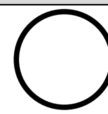
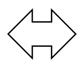
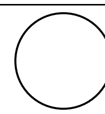
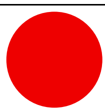
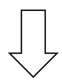
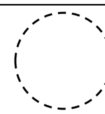
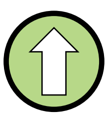
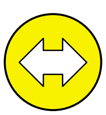
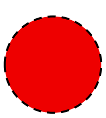
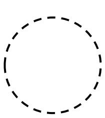
Condition Status		Trend in Condition		Confidence in Assessment	
Condition Icon	**Condition Icon Definition**	**Trend Icon**	**Trend Icon Definition**	**Confidence Icon**	**Confidence Icon Definition**
	Resource is in Good Condition		Condition is Improving		High
	Resource warrants Moderate Concern		Condition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

Table 87. Example indicator symbols and descriptions of how to interpret them in WCS tables.

Symbol Example	Description of Symbol
	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.

5.3. Park-wide Condition Observations

Despite the variety in vegetation and physical features at CUIS, many of the resources discussed in this report are interrelated and share similar management concerns (e.g., data gaps, threats from outside the park).

5.3.1. Vegetation Communities

The native vegetation communities of CUIS are vital resources for the park, providing habitat for wildlife and performing critical ecological functions, while attracting many visitors to the area. The park's salt marshes are currently in good condition, which may be partially related to the general inaccessibility of the marshes (due to tidal flooding) and the lack of historic human use. Much of the low salt marsh vegetation is relatively tolerant of harsh and variable environmental conditions (Peek et al. 2016), which may provide the community with some resiliency to stressors. However, salt marsh areas accessible to feral horses have been negatively impacted by grazing activity (Turner 1986, Dolan 2002).

The upland forest community is of moderate concern, largely due to the loss of fire-dependent longleaf pine habitats over time. The park's maritime forests, with their impressive live oaks, are in better condition but managers are concerned about a lack of oak regeneration. The upland forest and other CUIS vegetation communities will likely benefit from the recent reintroduction of fire as a management tool (NPS 2015a). The condition of the park's freshwater wetlands and interdune communities is currently unknown. Freshwater wetland acreage measures were determined to be of moderate concern, but due to data gaps for plant species diversity and water and soil quality in the wetlands, overall condition could not be assigned. Neither of the measures selected for interdune communities (acreage, plant diversity) had enough available information to determine condition.

5.3.2. Other Biotics

Animals featured as NRCA components were mammals, birds, and herpetofauna. The current condition of birds at CUIS is unknown due to a lack of contemporary data, particularly regarding wading birds. While several shorebird species have been studied, there is not enough information to assess condition for the selected measures (nesting numbers, fledging success). The many habitats of CUIS are critically important for many migratory bird species, including several species of conservation concern (NPS 2016f), and further study of the park's bird populations should be a high priority.

The park's mammals and herpetofauna are currently considered to be in good condition. The mammalian species composition is about what should be expected for an island the size and location of Cumberland Island. Mesocarnivore species richness is of low concern and deer population size is currently of no concern. Among herpetofauna, sea turtle measures are either of no concern (species richness) or low concern (nesting numbers, hatching success). Based on recent research (Moore 2016), the park's gopher tortoise population is in good condition and will likely benefit from the return of fire to the landscape.

5.3.3. Environmental Quality

Environmental quality is important for maintaining healthy functioning ecosystems. The health of terrestrial and aquatic organisms in parks can be affected substantially by the condition of air and water quality. The current condition of CUIS's water quality could not be determined due to data gaps. A water quality study of the park's freshwater wetlands conducted in 1999-2000 gathered baseline data (Frick et al. 2002), but no recent water quality monitoring results are available for comparison.

Air quality is currently of significant concern, based largely on the NPS ARD rating methodology. The visibility and mercury deposition measures are of significant concern, while ozone, nitrogen deposition, and sulfur deposition are of moderate concern (NPS 2016b). The factors contributing to the concern over air quality (e.g., power plant and industrial emissions) are almost entirely beyond the control of park management. However, conditions appear to be improving, particularly for sulfur deposition and ozone.

5.3.4. Physical Features (Geomorphology)

As discussed in Chapter 4, barrier islands experience nearly constant geomorphological change due to natural erosion and accretion processes (Griffin 1982, Alber et al. 2005, Calhoun and Riley 2016). Island geomorphology is influenced by wave action, tidal currents, sediment availability, and sea level changes (Griffin 1982). At CUIS, the two geomorphological processes of most interest at this time are shoreline change and dune dynamics (e.g., growth, erosion, migration). Very little is currently known about the dynamics of the park's dunes, despite their importance as a protective barrier for the rest of the island. Shoreline change has been studied, particularly on the back-barrier (west) side of CUIS, where substantial erosion has been observed in several areas (Alber et al. 2005, Jackson 2006). Although shorelines shift naturally over time, the changes that have occurred at CUIS in recent decades appear to be exacerbated by human activities (e.g., dredging, jetty and seawall construction, boat traffic) (Shabica et al. 1993, Jackson 2006). As a result, barrier island geomorphology is currently a significant concern for CUIS.

5.3.5. Park-wide Threats and Stressors

Several threats and stressors influence the condition of multiple resources at CUIS. These include feral wildlife, fire suppression, and a range of climate change impacts (e.g., SLR, extreme weather events). Feral horses and hogs, originally brought to the island by European settlers as livestock, have had a significant impact on the CUIS ecosystem (Dolan 2002, Dilsaver 2004). These species have competed with native wildlife for resources and their grazing and foraging activities have influenced the park's vegetation communities, particularly freshwater wetlands (Hillestad et al. 1975, Turner 1986, Noon and Martin 2004). Feral hogs also prey upon the eggs of shorebirds, sea turtles, and other reptiles (Plauny 2000, Kammermeyer et al. 2011).

Fire suppression is believed to have impacted many of the vegetation communities at CUIS. Historically, fires were frequent in the northern and central portions of the island, especially in longleaf pine communities (Frost et al. 2011). Over the past century, fires have become less frequent across the southeastern U.S., largely due to human suppression efforts (Frost 1993). A lack of fire allows the density of woody species to increase, encroaching upon grass and forb-dominated communities, including wetlands. In the park's freshwater wetlands, fire suppression has allowed organic matter to accumulate and fill in depressions, resulting in the reduction of water retention and an increased probability of wetland drying (Heath and Byrne 2014). In forests, the increased density is often among mid- and understory species, which can inhibit the regeneration of fire-adapted tree canopy species (e.g., oaks and pines) (Brockway et al. 2000, Frost et al. 2011). As mentioned previously, vegetation communities at CUIS will likely benefit from the recent reintroduction of fire as a management tool, but it will take some time for managers to burn all the areas that are currently in need of fire, and improvements may not be seen immediately.

As discussed in Chapters 2 and 4, temperatures are projected to increase across the southeastern U.S. over the next century as a result of global climate change (Carter et al. 2014). Warmer air temperatures will increase evaporation and plant transpiration rates, meaning that even if annual precipitation remains constant or slightly increases, overall conditions could still become drier in the future. These changes have the potential to impact the distribution and health of both vegetation and wildlife species, particularly those sensitive to environmental change (Bates et al. 2008, Fisichelli 2015, GA DNR 2015a). Warmer air and ocean temperatures are also projected to increase the intensity of tropical storms and hurricanes (Knutson et al. 2010), which would threaten many park resources, as well as park infrastructure and visitor experience. In September 2017, Hurricane Irma "sideswiped" CUIS after it had been downgraded to a tropical storm and caused substantial damage to the park. The island was evacuated on 7 September as the storm approached and remained closed to visitors through 11 November, largely due to the destruction of the mainland ferry dock (NPS 2017c). Many scientists hypothesized that warmer ocean temperatures related to climate change intensified Hurricane Irma (Drash 2017).

Climate change also contributes to SLR due to both the thermal expansion of water and the melting of continental ice (IPCC 2013, Peek et al. 2016). Rising water levels inundate coastal areas, including beaches and tidal marshes, and may alter coastal dynamics (e.g., shoreline erosion/accretion). The rate of SLR is expected to increase throughout the remainder of this century, so that the total rise from 2000 through 2100 will be 0.28-0.98 m (0.9-3.2 ft) (IPCC 2013).

5.4. Overall Conclusions

Despite its relatively small size, CUIS is a diverse park with a variety of rare or unique resources, from rare sea turtles and shorebirds to expansive salt marshes and moss-draped live oak trees. This assessment serves as a review and summary of available data and literature for featured natural resources in the park. The information presented here may serve as a baseline against which any changes in condition of components in the future may be compared. Current condition could not be determined for many components due to data gaps; for resources where condition could be assessed, the majority were in good condition or of moderate concern with a stable trend. Understanding the condition of these resources can help managers prioritize management objectives and better focus conservation strategies to maintain the health and integrity of these ecosystems.

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Appendix A. Aquatic wildlife species (fish and crustaceans) documented at CUIS.

Table A.1. Aquatic wildlife species (fish and crustaceans) documented at CUIS (NPS 2016f).

Category	Scientific Name	Common Name
Fish	*Abudefduf saxatilis*	sergeant major
	Alosa pseudoharengus	alewife, bigeye herring
	Ameiurus natalis	yellow bullhead
	Anchoa hepsetus	striped anchovy
	Anchoa mitchilli	bay anchovy
	Ancylopsetta ommata	ocellated flounder
	Anguilla rostrata	American eel
	Archosargus probatocephalus	sheepshead
	Bairdiella chrysoura	silver perch
	Bathygobius soporator	frillfin goby
	Brevoortia tyrannus	Atlantic menhaden
	Caranx hippos	crevalle jack
	Caranx latus	horse-eye jack
	Carcharhinus limbatus	blacktip shark
	Centropomus pectinatus	tarpon snook
	Centropristis striata	black sea bass
	Chaetodipterus faber	Atlantic spadefish
	Chasmodes bosquianus	striped blenny
	Chilomycterus schoepfii	burrfish, porcupinefish
	Chloroscombrus chrysurus	Atlantic bumper
	Citharichthys spilopterus	bay whiff
	Cynoscion nebulosus	spotted seatrout
	Cynoscion nothus	silver seatrout
	Cynoscion regalis	gray trout, weakfish
	Cyprinodon variegatus	sheepshead minnow
	Dasyatis sabina	Atlantic stingray
	Dormitator maculatus	fat sleeper
	Elops saurus	ladyfish
	Etropus crossotus	fringed flounder
	Eucinostomus argenteus	spotfin mojarra
	Eucinostomus lefroyi	longfinned silverbiddy
	Fundulus confluentus	marsh killifish
	Fundulus heteroclitus	mummichog
	Fundulus majalis	striped killifish
	Gambusia affinis	eastern mosquitofish

Category	Scientific Name	Common Name
Fish (continued)	*Gobiesox strumosus*	skilletfish
	Gobiosoma bosc	naked goby
	Gobiosoma ginsburgi	seaboard goby
	Haemulon aurolineatum	tomtate
	Histrio histrio	sargassum frogfish
	Hypleurochilus geminatus	crested blenny
	Hyporhamphus unifasciatus	silverstripe halfbeak
	Hypsoblennius hentz	feather blenny
	Labrisomus nuchipinnis	hairy blenny
	Lagodon rhomboides	pinfish
	Leiostomus xanthurus	spot
	Lepomis gulosus	warmouth
	Lepomis macrochirus	bluegill
	Lutjanus griseus	gray snapper
	Megalops atlanticus	tarpon
	Menidia beryllina	inland silverside
	Menidia menidia	Atlantic silverside
	Menticirrhus americanus	southern kingfish
	Menticirrhus littoralis	Gulf kingfish
	Menticirrhus saxatilis	northern kingfish
	Micropogonias undulatus	Atlantic croaker
	Micropterus salmoides	largemouth bass
	Mugil cephalus	striped mullet, gray mullet
	Mugil curema	white mullet
	Myrichthys ocellatus	goldspotted eel
	Myrophis punctatus	speckled worm eel
	Oligoplites saurus	leatherjack
	Opisthonema oglinum	Atlantic thread herring
	Opsanus tau	oyster toadfish
	Orthopristis chrysoptera	pigfish
	Paralichthys dentatus	fluke, summer flounder
	Paralichthys lethostigma	southern flounder
	Peprilus paru	harvestfish
	Peprilus triacanthus	butterfish
	Poecilia latipinna	sailfin molly
	Pogonias cromis	black drum
	Pomatomus saltatrix	bluefish
	Prionotus scitulus	leopard searobin
	Sardinella aurita	round sardinella

Category	Scientific Name	Common Name
Fish (continued)	*Scartella cristata*	molly miller
	Sciaenops ocellatus	red drum
	Scomberomorus maculatus	Atlantic Spanish mackerel
	Selene vomer	lookdown
	Sphoeroides maculatus	northern puffer
	Sphyrna lewini	scalloped hammerhead
	Stephanolepis hispida	planehead filefish
	Strongylura marina	Atlantic needlefish
	Syngnathus fuscus	northern pipefish
	Syngnathus louisianae	chain pipefish
	Synodus foetens	inshore lizardfish
	Trachinotus carolinus	Florida pompano
	Trachinotus falcatus	permit
	Trinectes maculatus	hogchoker
	Urophycis floridana	southern hake
Crustaceans	*Arenaeus cribrarius*	speckled swimming crab
	Balanus amphitrite	striped barnacle
	Balanus eburneus	ivory barnacle
	Chthamalus fragilis	fragile barnacle
	Clibanarius vittatus	thinstripe hermit
	Conopea galeata	seawhip barnacle
	Emerita talpoida	Atlantic sand crab
	Lepidopa websteri	sand crab
	Lysmata wurdemanni	peppermint shrimp
	Menippe mercenaria	Florida stone crab
	Pagurus longicarpus	long-armed hermit crab
	Pagurus pollicaris	flatclaw hermit
	Palaemonetes sp.	grass shrimp
	Uca pugilator	Atlantic sand fiddler
	Uca pugnax	Atlantic marsh fiddler

Appendix B. Plant species documented in CUIS upland forest vegetation communities.

Table B.1. Plant species documented in CUIS upland forest vegetation communities. * indicates non-native species.

Scientific Name	Common Name	Hillestad et al. 1975	Zomlefer et al. 2008, 2011	Heath & Byrne 2014
Acalypha gracilens	slender threeseed mercury	–	x	–
Acer rubrum	red maple	–	–	x
Ageratina aromatica	lesser snakeroot	x	–	–
Ageratina jucunda	hammock snakeroot	–	x	–
Allium sp.	onion	–	–	x
Ambrosia artemisiifolia	common ragweed	–	x	–
Ampelopsis arborea	peppervine	x	–	x
Andropogon brachystachyus	shortspike bluestem	–	x	–
Andropogon glomeratus	bushy bluestem	–	–	x
Andropogon gyrans var. *gyrans*	Elliott's bluestem	–	x	x
Andropogon ternarius	splitbeard bluestem	–	x	–
Andropogon virginicus	broomsedge bluestem	x	–	x
Andropogon virginicus var. *decipiens*	broomsedge bluestem	–	x	–
Andropogon virginicus var. *glaucus*	chalky bluestem	–	x	–
Aralia spinosa	devil's walkingstick	–	x	–
Aristida lanosa	woollysheath threeawn	x	x	–
Aristida longespica var. *geniculata*	red threeawn	–	x	–
Aristida purpurascens var. *purpurascens*	arrowfeather threeawn	–	x	–
Aristida purpurascens var. *virgata*	arrowfeather threeawn	–	x	–
Aristida spiciformis	bottlebrush threeawn	–	x	x
Aristolochia serpentaria	Virginia snakeroot	x	–	–
Arnoglossum ovatum	ovateleaf cacalia	–	x	–
Aronia arbutifolia	red chokeberry	x	x	x
Arundinaria gigantea	giant cane	–	–	x
Arundinaria tecta	switchcane	x	–	–
Asclepias amplexicaulis	clasping milkweed	–	x	–
Asclepias humistrata	pinewoods milkweed	–	x	–
Asclepias pedicellata	savannah milkweed	–	x	–

Scientific Name	Common Name	Hillestad et al. 1975	Zomlefer et al. 2008, 2011	Heath & Byrne 2014
Asimina longifolia	slimleaf pawpaw	–	x	–
Asimina parviflora	smallflower pawpaw	x	x	x
Asimina pygmea	dwarf pawpaw	x	–	–
Axonopus sp.	carpetgrass	–	–	x
Axonopus furcatus	big carpetgrass	–	x	–
Baccharis halimifolia	eastern baccharis	–	–	x
Bambusa sp.*	bamboo	–	–	x
Bejaria racemosa	tarflower	x	x	x
Berchemia scandens	Alabama supplejack	x	x	–
Bignonia capreolata	crossvine	x	–	x
Boehmeria cylindrica	small-spike false nettle	–	–	x
Bulbostylis ciliatifolia	capillary hairsedge	–	x	–
Burmannia capitata	southern bluethread	–	x	–
Callicarpa americana	American beautyberry	–	–	x
Carex sp.	sedge	x	–	–
Carex floridana	Florida sedge	–	x	–
Carphephorus odoratissimus	vanillaleaf	–	x	x
Carya glabra	pignut hickory	x	–	–
Celtis laevigata	sugar hackberry	x	–	–
Cenchrus tribuloides	sanddune sandbur	x	–	x
Cephalanthus occidentalis	common buttonbush	–	–	–
*Centella asiatica**	spadeleaf	–	–	x
Cercis canadensis	eastern redbud	x	–	–
Chasmanthium laxum ssp. *sessiliflorum*	longleaf woodoats	x	x	x
Cirsium horridulum	yellow thistle	–	–	x
Cirsium nuttallii	Nuttall's thistle	–	–	x
Cladium jamaicense	Jamaica swamp sawgrass	–	x	x
Clematis reticulata	netleaf leather flower	–	x	–
Cnidoscolus urens var. *stimulosus*	finger rot	x	–	x
Commelina erecta	erect dayflower	x	–	–
Corallorhiza wisteriana	spring coralroot	–	x	–
Cornus asperifolia	toughleaf dogwood	–	x	–
Cornus foemina	stiff dogwood	–	x	–
Crinum sp.	swamplily	–	–	x
Crotalaria rotundifolia	rabbitbells	–	x	–
Croton punctatus	gulf croton	–	–	x
Cuscuta pentagona	fiveangled dodder	–	x	–

Scientific Name	Common Name	Hillestad et al. 1975	Zomlefer et al. 2008, 2011	Heath & Byrne 2014
Cyperus sp.	sedge	x	–	x
Cyperus filiculmis	wiry flatsedge	–	x	–
Cyperus plukenetii	Plukenet's flatsedge	–	x	–
Cyperus retrorsus	pine barren flatsedge	x	x	–
Cyperus tetragonus	fourangle flatsedge	–	x	–
Desmodium sp.	ticktrefoil	–	–	x
Desmodium paniculatum	panicledleaf ticktrefoil	–	x	–
Dicerandra linearifolia var. *linearifolia*	coastalplain balm	–	x	–
Dichanthelium sp.	rosette grass	–	–	x
Dichanthelium aciculare	needleleaf rosette grass	x	x	–
Dichanthelium acuminatum var. *longiligulatum*	rough panicgrass	–	x	–
Dichanthelium dichotomum	cypress panicgrass	–	x	–
Dichanthelium ensifolium	cypress witchgrass	–	x	–
Dichanthelium oligosanthes	Heller's rosette grass	–	x	–
Dichanthelium portoricense	hemlock witchgrass	–	x	–
Dichanthelium strigosum var. *leucoblepharis*	roughhair rosette grass	–	x	–
Dichondra carolinensis	Carolina ponysfoot	–	x	x
Diodia virginiana	Virginia buttonweed	–	–	x
Diospyros virginiana	common persimmon	x	–	–
Dulichium arundinaceum	threeway sedge	–	x	–
Dyschoriste oblongifolia	oblongleaf snakeherb	x	–	–
Echinochloa muricata	rough barnyardgrass	–	x	–
Eleocharis robbinsii	Robbins' spikerush	–	x	–
Elephantopus tomentosus	devil's grandmother	–	–	x
Epidendrum magnoliae	green fly orchid	x	x	x
Eragrostis elliottii	field lovegrass	–	x	–
Eragrostis secundiflora ssp. *oxylepis*	red lovegrass	–	x	–
Erechtites hieraciifolius	American burnweed	–	x	–
*Eremochloa ophiuroides**	centipede grass	x	–	x
Eubotrys racemosa	swamp doghobble	–	x	–
Eupatorium capillifolium	dogfennel	–	–	x
Eupatorium compositifolium	yankeeweed	x	–	–
Eupatorium mohrii	Mohr's thoroughwort	–	x	–
Euphorbia polygonifolia	seaside sandmat	–	–	x
Eustachys petraea	pinewoods fingergrass	–	x	x
Fimbristylis sp.	fimbry	–	–	x

Scientific Name	Common Name	Hillestad et al. 1975	Zomlefer et al. 2008, 2011	Heath & Byrne 2014
Fimbristylis spadicea	marsh fimbry	–	x	–
Gaillardia aestivalis	lanceleaf blanketflower	–	x	–
Galactia elliottii	Elliott's milkpea	x	x	x
Galactia regularis	eastern milkpea	x	–	–
Galium hispidulum	coastal bedstraw	x	–	x
Galium obtusum	bluntleaf bedstraw	–	–	x
Galium pilosum	hairy bedstraw	–	x	–
Gaylussacia tomentosa	hairytwig huckleberry	x	x	–
Gelsemium sempervirens	Carolina jessamine	x	x	x
Gonolobus suberosus var. *suberosus*	angularfruit milkvine	x	x	–
Gratiola ramosa	branched hedgehyssop	–	x	–
Hamamelis virginiana	American witchhazel	x	x	–
*Hedera helix**	English ivy		x	–
Helianthemum corymbosum	pinebarren frostweed	x	–	x
Heterotheca subaxillaris	camphorweed	–	–	x
Houstonia procumbens	roundleaf bluet	x	–	x
Hydrocotyle bonariensis	largeleaf pennywort	–	–	x
Hypericum hypericoides	St. Andrew's cross	–	–	x
Hypericum mutilum	dwarf St. Johnswort	–	x	–
Hypericum tetrapetalum	fourpetal St. Johnswort	–	x	–
Hypoxis juncea	fringed yellow star-grass	–	x	–
Ilex ambigua	Carolina holly	x	x	x
Ilex cassine	dahoon	–	–	x
Ilex glabra	inkberry	x	x	x
Ilex opaca	American holly	x	x	x
Ilex vomitoria	yaupon	x	x	x
Indigofera caroliniana	Carolina indigo	–	x	x
Juniperus virginiana var. *silicicola*	southern redcedar	x	–	x
Kalmia hirsuta	hairy laurel	–	x	–
Krigia virginica	Virginia dwarfdandelion	–	–	x
Lechea torreyi	Piedmont pinweed	–	x	–
Lespedeza sp.	lespedeza	x	–	–
Lespedeza hirta	hairy lespedeza	–	–	x
Liatris laevigata	shortleaf blazing star	–	x	–
Liatris tenuifolia var. *tenuifolia*	shortleaf blazing star	–	x	–
*Ligustrum japonicum**	Japanese privet	–	–	x

Scientific Name	Common Name	Hillestad et al. 1975	Zomlefer et al. 2008, 2011	Heath & Byrne 2014
Liquidambar styraciflua	sweetgum	x	x	–
Lupinus villosus	lady lupine	–	x	x
Lyonia ferruginea	rusty staggerbush	x	x	x
Lyonia fruticosa	coastal plain staggerbush	–	x	–
Lyonia lucida	fetterbush lyonia	x	x	x
Magnolia grandiflora	southern magnolia	x	–	–
Magnolia virginiana	sweetbay	x	–	–
Melica mutica	twoflower melicgrass	–	x	–
Micranthemum umbrosum	shade mudflower	–	x	–
Mikania scandens	climbing hempvine	–	–	x
Mimosa microphylla	littleleaf sensitive-briar	x	–	–
Mitchella repens	partridgeberry	–	x	–
Monotropa uniflora	Indian pipe	–	x	x
Morella cerifera	wax myrtle	x	–	x
Morus rubra	red mulberry	x	x	–
Muhlenbergia capillaris	hairawn muhly	–	–	x
Muhlenbergia schreberi	nimblewill muhly	–	x	–
Nuttallanthus canadensis	Canada toadflax	–	–	x
Nyssa biflora	swamp tupelo	x	–	x
Oenothera humifusa	seabeach evening primrose	x	–	x
Ophioglossum petiolatum	stalked adder's-tongue	–	x	–
Oplismenus hirtellus	bristle basketgrass	x	x	x
Opuntia humifusa	devil's-tongue	–	–	x
Opuntia pusilla	cockspur pricklypear	x	–	–
Orthosia scoparia	leafless swallow-wort	x	x	–
Osmanthus americanus	devilwood	x	x	x
Osmunda regalis	royal fern	–	–	x
Osmundastrum cinnamomeum	cinnamon fern	–	–	x
Oxalis stricta	common yellow oxalis	–	–	x
*Panicum repens**	torpedo grass	–	x	–
Parietaria floridana	Florida pellitory	–	–	x
Parthenocissus quinquefolia	Virginia creeper	x	–	x
Paspalum setaceum var. *ciliatifolium*	fringe-leaf paspalum	x	–	–
Paspalum urvillei	Vasey grass	–	x	–
Passiflora lutea	yellow passionflower	–	x	–
Pediomelum canescens	buckroot	–	x	–
Persea borbonia	redbay	x	x	x

Scientific Name	Common Name	Hillestad et al. 1975	Zomlefer et al. 2008, 2011	Heath & Byrne 2014
Persea palustris	swamp bay	x	–	x
Phlebodium aureum	golden polypody	–	x	–
Phoradendron serotinum ssp. *serotinum*	oak mistletoe	–	x	–
Phyla nodiflora	turkey tangle fogfruit	–	–	x
Physalis sp.	groundcherry	–	–	x
Phytolacca americana	American pokeweed	–	–	x
Pinus echinata	shortleaf pine	–	x	–
Pinus elliottii	slash pine	x	x	–
Pinus glabra	spruce pine	–	–	x
Pinus palustris	longleaf pine	x	x	x
Pinus serotina	pond pine	x	x	x
Pinus taeda	loblolly pine	x	x	x
Piptochaetium avenaceum	blackseed speargrass	x	–	–
Pityopsis graminifolia var. *graminifolia*	narrowleaf silkgrass	x	–	–
Plantago virginica	Virginia plantain	–	–	x
Platanus occidentalis	American sycamore	–	x	–
Pleopeltis polypodioides	resurrection fern	x	x	x
Pluchea sp.	camphorweed	–	–	x
Pluchea baccharis	rosy camphorweed	–	x	–
Polygala lutea	orange milkwort	–	x	–
Polygonella gracilis	tall jointweed	–	x	–
Polygonum sp.	knotweed, smartweed	–	–	x
Prunus serotina var. *serotina*	black cherry	x	x	x
Pseudognaphalium obtusifolium	rabbit tobacco	–	–	x
Pteridium aquilinum	bracken fern	x	–	x
Pterocaulon pycnostachyum	dense-spike blackroot	–	x	–
Quercus austrina	bastard white oak	–	x	–
Quercus chapmanii	Chapman's oak	x	–	x
Quercus geminata	sand live oak	–	–	x
Quercus incana	bluejack oak	x	–	–
Quercus laevis	turkey oak	x	x	–
Quercus laurifolia	laurel oak	x	x	x
Quercus margarettae	sand post oak	–	x	–
Quercus myrtifolia	myrtle oak	x	–	x
Quercus nigra	water oak	x	–	–
Quercus virginiana	live oak	x	x	x

Scientific Name	Common Name	Hillestad et al. 1975	Zomlefer et al. 2008, 2011	Heath & Byrne 2014
Rhus copallinum	winged sumac	x	–	x
Rhynchospora sp.	beaksedge	–	–	x
Rhynchospora megalocarpa	sandyfield beaksedge	–	x	–
Rhynchospora wrightiana	Wright's beaksedge	x	–	–
Rubus sp.	blackberry	x	–	–
Rubus argutus	sawtooth blackberry	–	x	x
Rubus cuneifolius	sand blackberry	–	x	x
Rubus trivialis	southern dewberry	–	–	x
Sabal palmetto	cabbage palmetto	x	–	x
Sabatia sp.	rose gentian	–	–	x
Saccharum giganteum	sugarcane plumegrass	–	–	x
Sageretia minutiflora	smallflower mock buckthorn	–	x	–
Salvia lyrata	lyreleaf sage	–	–	x
Sanicula canadensis	Canadian blacksnakeroot	–	x	–
Sapindus saponaria	wingleaf soapberry	–	x	–
Sassafras albidum	sassafras	–	x	–
Schizachyrium scoparium var. *littorale*	shore little bluestem	–	–	x
Scleria oligantha	littlehead nutrush	–	x	–
Scleria triglomerata	whip nutrush	x	x	x
Senna obtusifolia	Java-bean	–	–	x
Serenoa repens	saw palmetto	x	x	x
Sideroxylon tenax	tough bumelia	x	–	x
Smilax auriculata	earleaf greenbrier	x	–	x
Smilax bona-nox	saw greenbrier	x	x	x
Smilax glauca	cat greenbrier	x	–	–
Smilax laurifolia	laurel greenbrier	–	x	x
Smilax pumila	sarsparilla vine	x	–	–
Smilax tamnoides	bristly greenbrier	–	x	–
Solidago sp.	goldenrod	–	–	x
Solidago odora ssp. *chapmanii*	Chapman's goldenrod	–	x	–
Sorghastrum elliottii	slender Indiangrass	–	x	–
Sorghastrum secundum	lopsided Indiangrass	x	x	–
Spartina bakeri	sand cordgrass	–	–	x
Spartina patens	saltmeadow cordgrass	–	–	x
Spiranthes praecox	greenvein ladies'-tresses	–	x	–
Sporobolus clandestinus	rough dropseed	–	x	–
Sporobolus junceus	pineywoods dropseed	–	x	–

Scientific Name	Common Name	Hillestad et al. 1975	Zomlefer et al. 2008, 2011	Heath & Byrne 2014
Stenotaphrum secundatum	St. Augustine grass	x	–	x
Stillingia sylvatica	queen's-delight	x	–	x
Stylisma patens	coastalplain dawnflower	–	x	–
Stylosanthes biflora	sidebeak pencilflower	x	–	–
Symplocos tinctoria	sweetleaf	x	–	x
Tilia americana var. *caroliniana*	Carolina basswood	–	x	–
Tillandsia recurvata	small ballmoss	–	–	x
Tillandsia usneoides	Spanish moss	x	x	x
Toxicodendron radicans	eastern poison ivy	–	x	x
Tragia urens	wavyleaf noseburn	x	x	–
*Triadica sebifera**	Chinese tallow	–	x	–
Trichostema setaceum	narrowleaf bluecurls	–	x	–
Tridens carolinianus	Carolina fluffgrass	–	x	–
Ulmus americana	American elm	–	–	x
Vaccinium arboreum	farkleberry	x	x	x
Vaccinium corymbosum	highbush blueberry	x	x	x
Vaccinium myrsinites	shiny blueberry	x	x	x
Vaccinium stamineum	deerberry	x	x	x
Verbascum thapsus	common mullein	–	–	x
Vicia acutifolia	fourleaf vetch	–	x	–
Vicia minutiflora	pygmyflower vetch	–	x	–
Viola sp.	violet	–	–	x
Viola sororia	common blue violet	–	x	–
Vitis aestivalis	summer grape	x	–	–
Vitis rotundifolia	muscadine grape	x	–	x
Vittaria lineata	shoestring fern	–	x	–
*Vulpia myuros**	rattail fescue	–	x	–
Woodwardia areolata	netted chainfern	–	x	–
*Youngia japonica**	oriental false hawksbeard	–	x	–
Yucca filamentosa	Adam's needle	–	x	–
Zanthoxylum clava-herculis	Hercules' club	x	–	x
Zornia bracteata	viperina	–	x	–
Total	**–**	**102**	**161**	**132**

Appendix C. Extent of freshwater (palustrine, non-tidal) wetlands by type at CUIS.

Table C.1. Extent of freshwater (palustrine, non-tidal) wetlands by type at CUIS, as mapped by the NWI (USFWS 2012).

Wetland Type/Code	# of Wetlands	Total Area (ha)	Wetland Type/Code	# of Wetlands	Total Area (ha)
PAB4	4	10.7	PFO3/1B	1	2.6
PEM1/FOF	2	5.4	PFO3/2C	1	1.5
PEM1/SS3C	3	8.2	PFO3/4B	10	228.3
PEM1B	2	6.2	PFO3/4C	2	22.7
PEM1C	71	227.4	PFO3B	2	8.6
PEM1F	29	196.1	PFO4/1B	1	1.8
PFO1/2C	9	17.5	PFO4/1C	2	13.0
PFO1/3B	2	76.8	PFO4/3B	2	8.0
PFO1/3C	5	76.1	PFO4/3C	2	0.5
PFO1/4B	3	65.7	PFO4B	2	15.3
PFO1/4C	5	91.6	PFO4C	2	6.4
PFO1B	1	12.0	PSS1/3A	1	8.3
PFO1C	32	135.7	PSS1B	1	8.5
PFO1F	3	10.8	PSS1C	4	28.8
PFO2/1C	3	2.3	PSS1F	2	12.2
PFO2/1F	2	5.1	PSS3/1C	2	5.3
PFO2/3C	1	0.7	PSS3/4B	1	2.1
PFO2/4C	1	4.8	PSS3B	1	1.9
PFO2F	4	14.4	PUBH	9	5.1

Appendix D. Plant species observed in freshwater wetlands at CUIS.

Table D.1. Plant species observed in freshwater wetlands at CUIS. * indicates non-native species. In the Heath and Byrne (2014) column, CP = South Atlantic Coastal Pond, IS = Atlantic Coast Interdune Swale, SF = Atlantic/East Gulf Coastal Plain Sweetbay - Tupelo Streamhead Forest.

Scientific Name	Common Name	Hillestad et al. 1975 (wooded)	Zomlefer et al. 2008 & 2011			Heath & Byrne 2014
			Marsh	Pond/ Slough	Swamp (wooded)	
Acer rubrum	red maple	x	–	x	–	IS
Agalinis fasciculata	beach false foxglove	–	–	x	–	–
*Alternanthera philoxeroides**	alligatorweed	–	–	x	–	–
Ammannia latifolia	pink redstem	–	–	x	–	–
Ampelopsis arborea	peppervine	x	–	–	–	–
Andropogon virginicus	broomsedge bluestem	–	–	–	–	SF, CP
Arundinaria tecta	switchcane	x	–	–	–	–
Azolla filiculoides	Carolina mosquitofern	–	–	x	–	–
Baccharis halimifolia	eastern baccharis	–	–	–	–	IS
Bacopa monnieri	herb-of-grace	–	–	x	–	–
Bidens laevis	bur marigold	–	–	x	–	–
Bulbostylis ciliatifolia	capillary hairsedge	–	–	x	–	–
Burmannia biflora	northern bluethread	–	–	–	x	–
Calystegia sepium	hedge false bindweed	–	–	x	–	–
Cardamine pensylvanica	Pennsylvania bittercress	–	–	x	–	–
Carex atlantica ssp. *capillacea*	Howe sedge	–	–	–	x	–
Carex elliottii	Elliott's sedge	–	–	–	x	–
Carex fissa var. *aristata*	hammock sedge	–	–	–	x	–
Carex lupulina	hop sedge	–	–	–	x	–
Carex stipata	owlfruit sedge	–	–	–	x	–
Carex verrucosa	warty sedge	–	x	–	–	–
Celtis laevigata	sugar hackberry	x	–	–	–	–
Cephalanthus occidentalis	common buttonbush	x	–	x	–	–
Chasmanthium laxum	slender woodoats	–	–	–	–	CP

Scientific Name	Common Name	Hillestad et al. 1975 (wooded)	Zomlefer et al. 2008 & 2011			Heath & Byrne 2014
			Marsh	Pond/ Slough	Swamp (wooded)	
Cladium jamaicense	Jamaica swamp sawgrass	–	x	–	–	–
Coleataenia longifolia ssp. *rigidula*	redtop panicum	–	–	x	–	–
Cuphea carthagenensis	Colombian waxweed	–	–	–	x	–
Cyperus haspan	haspan flatsedge	–	–	x	–	IS
Cyperus odoratus	fragrant flatsedge	–	–	x	–	–
Cyperus polystachyos	manyspike flatsedge	–	–	x	–	–
Cyperus pseudovegetus	marsh flatsedge	–	x	–	–	–
Cyperus virens	green flatsedge	–	–	x	–	–
Decodon verticillatus	swamp loosestrife	–	–	x	–	–
Dichanthelium sp.	rosette grass	–	–	–	–	SF, CP
Dichanthelium acuminatum var. *longiligulatum*	rough panicgrass	–	x	–	–	–
Digitaria serotina	dwarf crabgrass	–	x	x	–	–
Drosera intermedia	spoonleaf sundew	–	x	–	–	–
*Echinochloa crus-galli**	barnyardgrass	–	–	–	x	–
Echinochloa muricata	rough barnyardgrass	–	–	x	–	–
Echinochloa walteri	coast cockspur	–	–	x	–	–
Eclipta prostrata	false daisy	–	–	–	x	–
Eleocharis albida	white spikerush	–	x	–	–	–
Eleocharis equisetoides	jointed spikesedge	–	x	x	–	–
Eleocharis flavescens var. *flavescens*	yellow spikerush	–	–	–	x	–
Eleocharis flavescens var. *olivacea*	bright green spikerush	–	x	–	–	–
Eleocharis vivipara	viviparous spikerush	–	x	–	–	–
*Eremochloa ophiuroides**	centipede grass	–	–	–	–	SF
Eupatorium sp.	thoroughwort	–	–	–	–	IS
Eupatorium leptophyllum	false fennel	–	–	x	–	–
Eupatorium serotinum	lateflowering thoroughwort	–	–	–	x	–

Scientific Name	Common Name	Hillestad et al. 1975 (wooded)	Zomlefer et al. 2008 & 2011			Heath & Byrne 2014
			Marsh	Pond/ Slough	Swamp (wooded)	
Fimbristylis autumnalis	slender fimbry	–	–	x	–	–
Fimbristylis caroliniana	Carolina fimbry	–	–	x	x	–
Fuirena pumila	dwarf umbrellasedge	–	–	x	–	–
Fuirena squarrosa	hairy umbrellasedge	–	–	x	–	–
Gelsemium sempervirens	Carolina jessamine	–	–	–	–	SF
Gordonia lasianthus	loblolly bay	x	–	–	x	–
Habenaria repens	waterspider false reinorchid	–	–	x	–	–
Helianthemum corymbosum	pinebarren frostweed	–	–	–	–	SF
Hibiscus grandiflorus	swamp rosemallow	–	x	x	–	–
Hydrocotyle bonariensis	largeleaf pennywort	–	–	–	–	IS
Hydrocotyle ranunculoides	floating marshpennywort	–	x	–	–	–
Hydrocotyle umbellata	manyflower marshpennywort	–	–	x	–	–
Hydrocotyle verticillata var. *triradiata*	whorled marshpennywort	–	–	x	–	–
Hypericum cistifolium	roundpod St. Johnswort	–	–	x	–	–
Ilex cassine	dahoon	–	x	x	–	–
Ilex glabra	inkberry	x	–	–	–	–
Ilex opaca	American holly	x	–	–	–	CP, SF
Iva frutescens	Jesuit's bark	–	–	–	–	IS
Juncus acuminatus	tapertip rush	–	–	x	–	–
Juncus coriaceus	leathery rush	–	–	–	x	–
Juncus dichotomus	forked rush	–	–	x	–	–
Juncus effusus ssp. *solutus*	lamp rush	–	–	–	x	–
Juncus repens	lesser creeping rush	–	–	x	–	–
Juncus scirpoides	needlepod rush	–	–	x	–	–
Kosteletzkya pentacarpos	Virginia saltmarsh mallow	–	–	x	–	–
Lemna aequinoctialis	lesser duckweed	–	–	x	–	–
Lemna valdiviana	Valdivia duckweed	–	–	x	x	–

Scientific Name	Common Name	Hillestad et al. 1975 (wooded)	Zomlefer et al. 2008 & 2011 Marsh	Zomlefer et al. 2008 & 2011 Pond/ Slough	Zomlefer et al. 2008 & 2011 Swamp (wooded)	Heath & Byrne 2014
Limnobium spongia	American spongeplant	–	–	x	–	–
Ludwigia arcuata	piedmont primrose-willow	–	–	x	–	–
Ludwigia lanceolata	lanceleaf primrose-willow	–	–	–	x	–
Ludwigia leptocarpa	anglestem primrose-willow	–	–	x	–	–
Ludwigia palustris	marsh primrose-willow	–	–	x	–	–
Lyonia ferruginea	rusty staggerbush	x	–	x	–	SF, CP
Lyonia lucida	fetterbush lyonia	x	–	–	–	–
Magnolia grandiflora	southern magnolia	x	–	–	–	–
Magnolia virginiana	sweetbay	x	–	–	x	–
Mitreola petiolata	lax hornpod	–	–	–	x	–
Morella cerifera	wax myrtle	x	–	–	–	SF, IS
Muhlenbergia capillaris	hairawn muhly	–	–	x	–	–
Nelumbo lutea	American lotus	–	–	x	–	–
Nymphaea odorata	American white waterlily	–	–	x	–	–
Nymphoides aquatica	big floatingheart	–	–	x	–	–
Nyssa biflora	swamp tupelo	x	–	x	–	–
*Oldenlandia corymbosa**	flattop mille graines	–	–	x	–	–
Osmundastrum cinnamomeum	cinnamon fern	–	–	x	–	–
Osmunda regalis var. *spectabilis*	royal fern	–	–	x	–	–
Panicum sp.	panicgrass	–	–	–	–	CP
Panicum hemitomon	maidencane	–	–	x	–	–
Panicum verrucosum	warty panicgrass	–	–	x	–	–
Paspalum vaginatum	seashore paspalum	–	–	x	–	–
Pentodon pentandrus	Hale's pentodon	–	–	x	–	–
Persea borbonia	redbay	–	–	–	–	CP
Persea palustris	swamp bay	x	x	–	–	–
Persicaria glabra	denseflower knotweed	–	–	x	–	–
Persicaria hirsuta	hairy smartweed	–	–	x	–	–
Persicaria hydropiperoides	swamp smartweed	–	–	x	–	–

Scientific Name	Common Name	Hillestad et al. 1975 (wooded)	Zomlefer et al. 2008 & 2011			Heath & Byrne 2014
			Marsh	Pond/ Slough	Swamp (wooded)	
*Persicaria posumbu**	oriental lady's thumb	–	–	–	x	–
Persicaria setacea	bog smartweed	–	–	–	x	–
*Phyllanthus urinaria**	chamber bitter	–	–	–	x	–
Pinguicula pumila	small butterwort	–	x	–	x	–
Pinus elliottii	slash pine	x	–	–	–	–
Pinus taeda	loblolly pine	x	–	–	–	SF, CP
Pleopeltis polypodioides	resurrection fern	–	–	–	–	SF
Pluchea foetida	stinking camphorweed	–	–	x	–	–
Polygala lutea	orange milkwort	–	–	x	–	–
Pontederia cordata	pickerelweed	–	–	x	–	–
Proserpinaca pectinata	combleaf mermaidweed	–	–	–	x	–
Prunus serotina	black cherry	–	–	–	–	CP
Ptilimnium capillaceum	threadleaf mockbishopweed	–	–	x	–	–
Quercus laurifolia	laurel oak	x	–	–	–	SF, CP
Quercus margarettae	sand post oak	–	–	–	x	–
Quercus nigra	water oak	x	–	–	x	–
Quercus virginiana	live oak	x	–	–	–	SF, CP
Rhexia mariana	Maryland meadowbeauty	–	–	x	–	–
Rhynchospora capitellata	slender-fruit beaksedge	–	–	–	x	–
Rhynchospora corniculata	shortbristle horned beaksedge	–	–	x	–	–
Rhynchospora inexpansa	nodding beaksedge	–	x	–	–	–
Rhynchospora microcephala	smallhead beaksedge	–	–	x	–	–
Rhynchospora miliacea	millet beaksedge	–	–	x	–	–
Rhynchospora mixta	mingled beaksedge	–	–	–	x	–
Ruppia maritima	widgeongrass	–	–	x	–	–
Sabal palmetto	cabbage palmetto	x	–	–	–	–
Sabatia stellaris	rose of Plymouth	–	–	x	–	–
Sacciolepis striata	American cupscale	–	–	x	–	–
Sagittaria filiformis	threadleaf arrowhead	–	–	x	–	–

Scientific Name	Common Name	Hillestad et al. 1975 (wooded)	Zomlefer et al. 2008 & 2011			Heath & Byrne 2014
			Marsh	Pond/ Slough	Swamp (wooded)	
Sagittaria graminea ssp. *graminea*	grassy arrowhead	–	–	–	x	–
Sagittaria lancifolia	bulltongue arrowhead	–	–	x	–	–
Sagittaria latifolia	broadleaf arrowhead	–	–	x	–	–
Salix caroliniana	coastal plain willow	x	–	x	–	–
Sambucus nigra ssp. *canadensis*	American black elderberry	–	–	x	–	–
Samolus valerandi	seaside brookweed	–	–	x	–	–
Saururus cernuus	lizard's tail	–	–	x	–	–
Schoenoplectus pungens var. *pungens*	common threesquare	–	–	x	–	–
Scleria reticularis	netted nutrush	–	–	x	–	–
Scleria triglomerata	whip nutrush	–	–	–	–	SF, CP
Serenoa repens	saw palmetto	x	–	–	–	SF
Setaria corrugata	coastal bristlegrass	–	–	x	–	–
Setaria magna	giant bristlegrass	–	–	–	–	IS
Setaria parviflora	yellow bristlegrass	–	–	x	–	–
Smilax auriculata	earleaf greenbrier	x	–	–	–	IS
Smilax glauca	cat greenbrier	x	–	–	–	–
Smilax laurifolia	laurel greenbrier	x	–	–	–	–
Solanum americanum	American black nightshade	–	–	–	x	–
Solidago sempervirens	seaside goldenrod	–	–	–	–	IS
Spartina bakeri	sand cordgrass	–	–	x	–	IS
Sporobolus virginicus	seashore dropseed	–	–	–	–	IS
*Stellaria media**	common chickweed	–	–	–	x	–
Strophostyles helvola	trailing fuzzy-bean	–	–	x	–	–
Symphyotrichum subulatum	eastern annual saltmarsh aster	–	–	–	x	–
Thalia geniculata	bent alligator-flag	–	–	–	x	–
Tillandsia usneoides	Spanish moss	–	–	–	–	SF
Toxicodendron radicans	eastern poison ivy	–	–	–	–	IS
Triglochin striata	three-rib arrowgrass	–	–	x	–	–
Typha domingensis	southern cattail	–	–	x	–	–
Typha latifolia	broadleaf cattail	–	–	x	–	–
Ulmus americana	American elm	x	–	–	x	–
Utricularia gibba	humped bladderwort	–	–	x	–	–

Scientific Name	Common Name	Hillestad et al. 1975 (wooded)	Zomlefer et al. 2008 & 2011			Heath & Byrne 2014
			Marsh	Pond/ Slough	Swamp (wooded)	
Utricularia inflata	swollen bladderwort	–	–	x	–	–
Utricularia purpurea	purple bladderwort	–	–	x	–	–
Utricularia subulata	zigzag bladderwort	–	x	–	–	–
Vaccinium arboreum	farkleberry	–	–	–	–	SF, CP
Vicia acutifolia	fourleaf vetch	–	–	x	–	–
Vigna luteola	hairypod cowpea	–	–	x	–	–
Viola lanceolata	bog white violet	–	x	–	x	–
Vitis rotundifolia	muscadine grape	x	–	–	–	–
Websteria confervoides	algal bulrush	–	–	x	–	–
Woodwardia areolata	netted chainfern	–	–	x	–	–
Woodwardia virginica	Virginia chainfern	–	–	–	x	–
Xyris elliottii	Elliott's yelloweyed grass	–	x	–	–	–
Xyris fimbriata	fringed yelloweyed grass	–	–	x	–	–
Xyris platylepis	tall yelloweyed grass	–	–	–	x	–
Total	**–**	**28**	**18**	**94**	**37**	**32**

Appendix E. Maps of salt marsh areas grazed by feral horses, as identified by Dolan (2002).

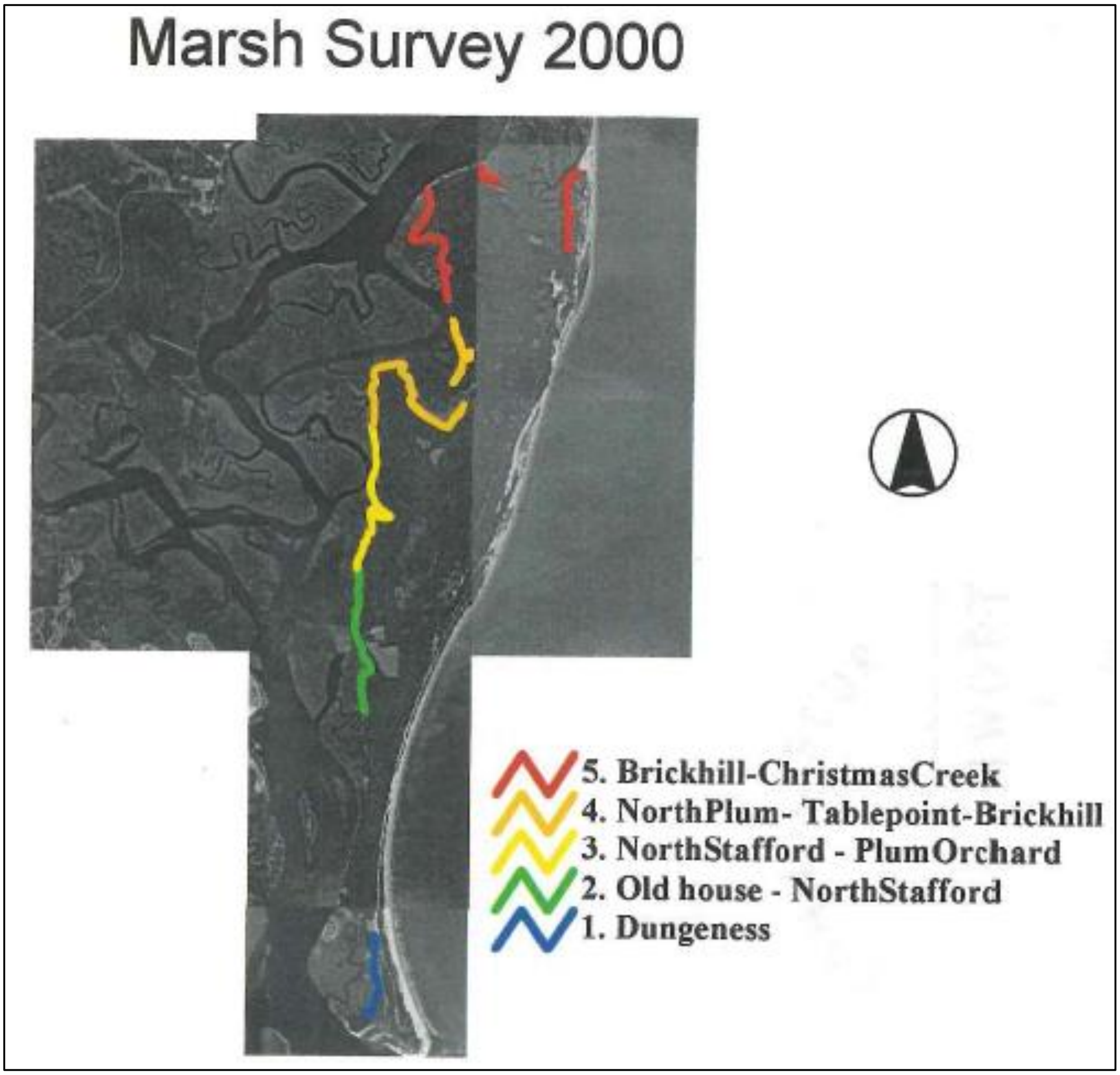

Figure E.1. The location of Dolan's (2002) salt marsh sampling segments on CUIS.

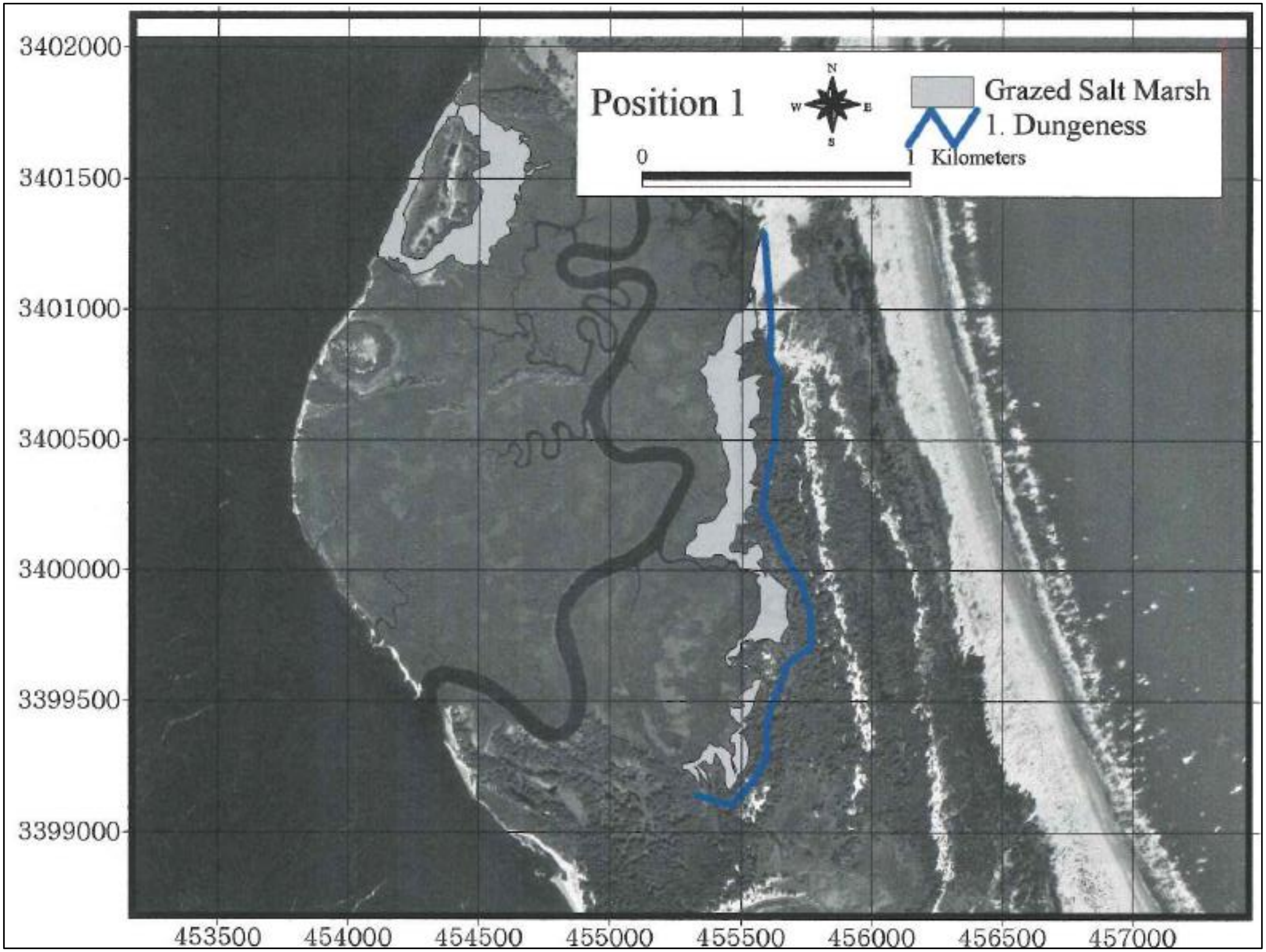

Figure E.2. Salt marsh area grazed by horses in the Dungeness sampling segment (Dolan 2002).

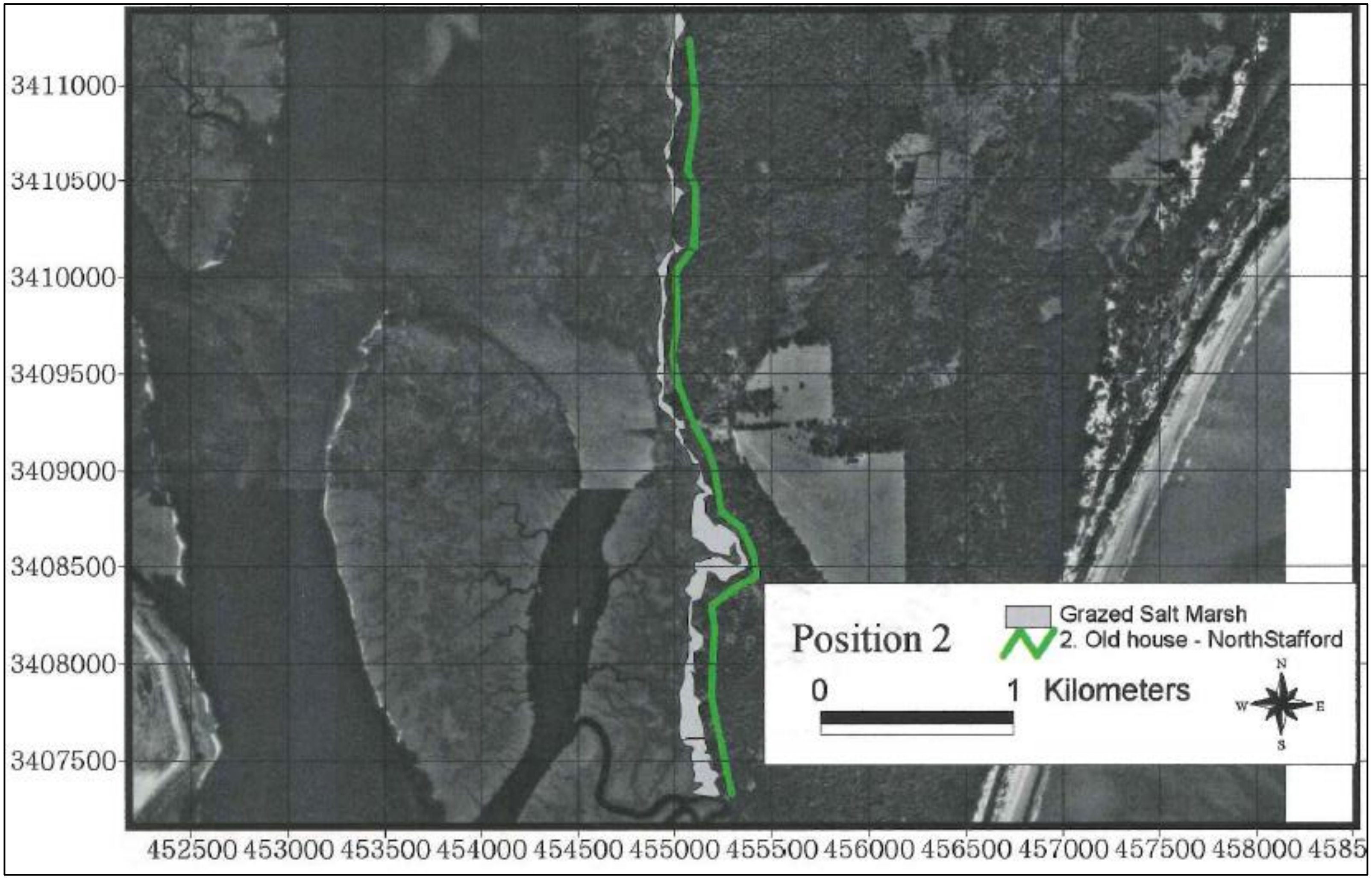

Figure E.3. Salt marsh area grazed by horses in the North Stafford sampling segment (Dolan 2002).

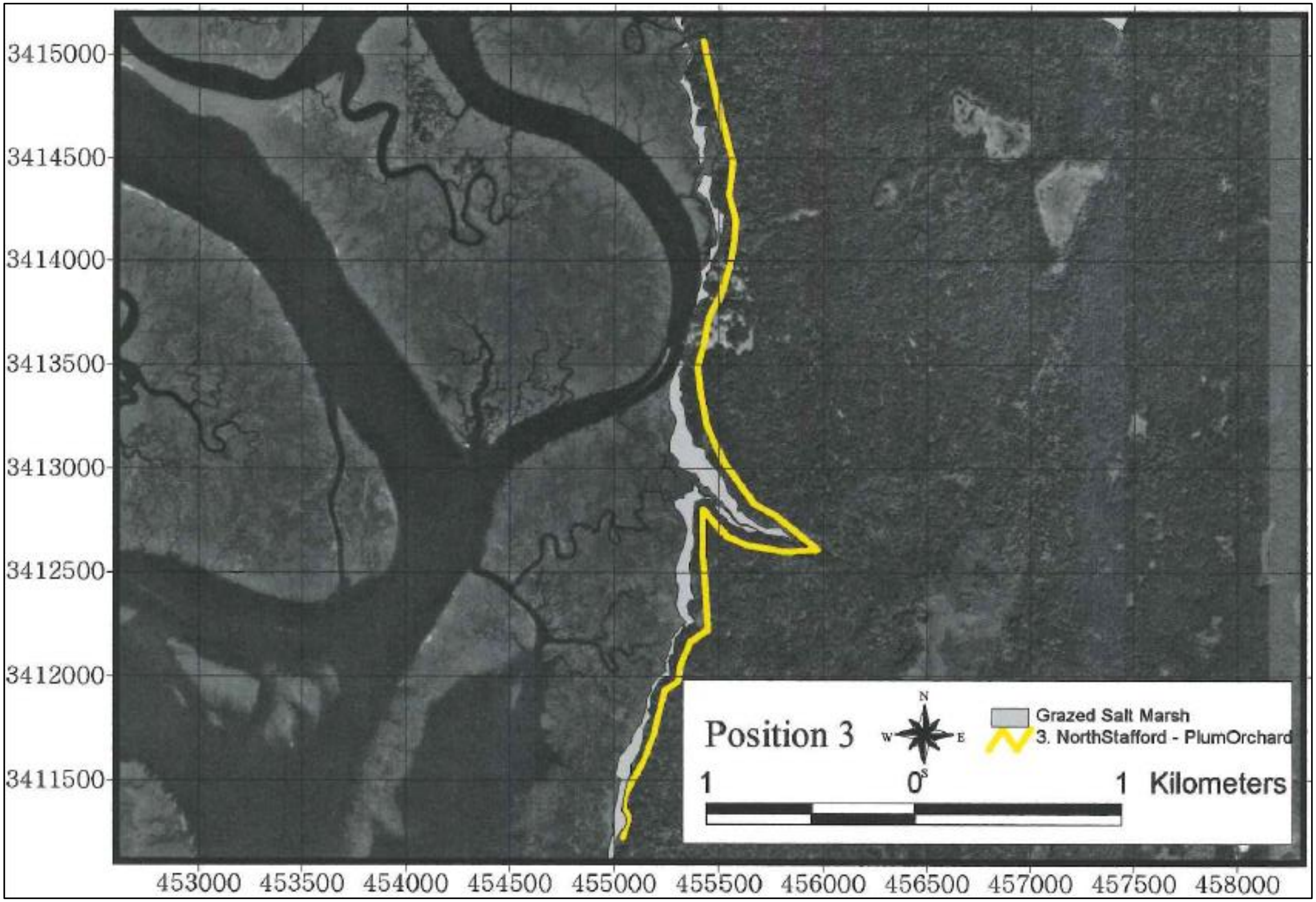

Figure E.4. Salt marsh area grazed by horses in the Plum Orchard sampling segment (Dolan 2002).

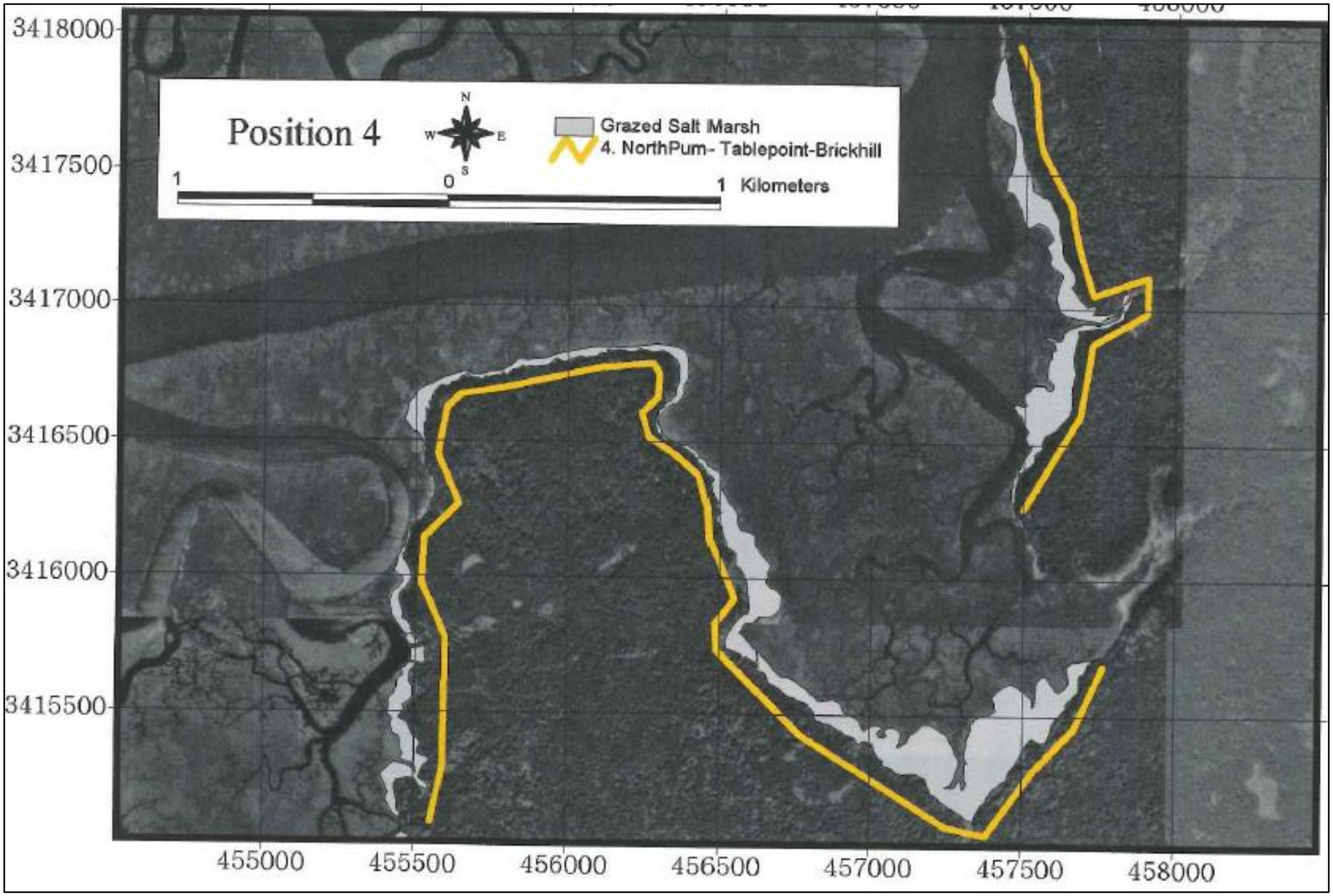

Figure E.5. Salt marsh area grazed by horses in the Tablepoint-Brickhill sampling segment (Dolan 2002).

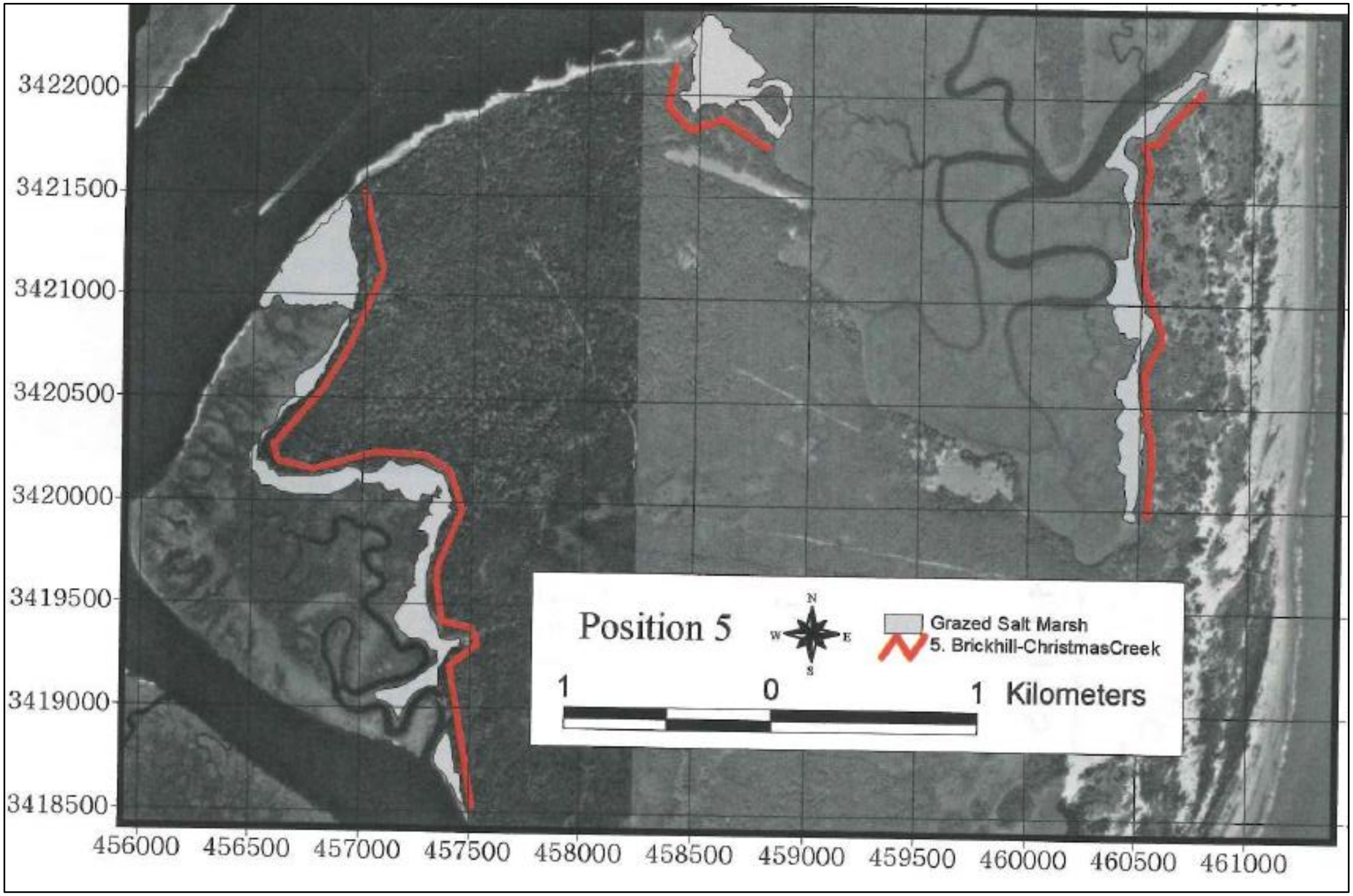

Figure E.6. Salt marsh area grazed by horses in the Brickhill-Christmas Creek sampling segment (Dolan 2002).

Appendix F. Plant species observed in interdune vegetation communities at CUIS.

Table F.1. Plant species observed in interdune vegetation communities at CUIS.

Scientific Name	Common Name	Hillestad et al. 1975	Dolan 2002	Zomlefer et al. 2008 & 2011
Agalinis fasciculata	beach false foxglove	–	–	x
Andropogon glomeratus var. *glaucopsis*	purple bluestem	–	–	x
Andropogon longiberbis	hairy bluestem	–	x	–
Atriplex cristata	crested saltbush	–	–	x
Buchnera americana	American bluehearts	–	–	x
*Bulbostylis barbata**	watergrass	x	x	–
Cakile edentula	American searocket	–	–	x
Carex sp.	sedge	–	x	–
Cenchrus echinatus	southern sandbur	–	–	x
Cenchrus tribuloides	sanddune sandbur	x	x	x
Cirsium horridulum	yellow thistle	–	x	x
Cirsium sp.	thistle	–	x	–
*Cocos nucifera**	coconut palm	–	–	x
Coleataenia anceps	beaked panicum	–	–	x
Coleataenia longifolia ssp. *longifolia*	long-leaved panic grass	–	–	x
Conyza canadensis	Canadian horseweed	x	x	x
Crotalaria rotundifolia	rabbitbells	–	x	–
Croton punctatus	gulf croton	x	x	x
*Cynodon dactylon**	Bermudagrass	x	x	–
Cyperus sp.	sedge	x	x	–
Cyperus croceus	Baldwin's flatsedge	–	–	x
Cyperus erythrorhizos	redroot flatsedge	–	–	x
*Cyperus esculentus**	yellow nutsedge	–	x	x
Cyperus haspan	haspan flatsedge	–	–	x
Cyperus pseudovegetus	marsh flatsedge	–	–	x
Cyperus retrorsus	pine barren flatsedge	–	x	–
*Cyperus rotundus**	purple nutsedge	–	–	x
Cyperus virens	green flatsedge	–	x	–
Dichanthelium aciculare	needleleaf rosette grass	–	–	x
Dichanthelium commutatum	variable panicgrass	–	x	–
Dichanthelium sphaerocarpon var. *floridanum*	roundseed panicgrass	–	x	–

Scientific Name	Common Name	Hillestad et al. 1975	Dolan 2002	Zomlefer et al. 2008 & 2011
Digitaria serotine	dwarf crabgrass	–	–	x
Diodella teres	poorjoe	x	x	–
Dysphania ambrosioides	Mexican tea	x	x	–
Eleocharis sp.	spikerush	x	x	–
Eleocharis fallax	creeping spikerush	–	–	x
Eleocharis montevidensis	sand spikerush	–	–	x
Eragrostis elliottii	field lovegrass	–	–	x
Eragrostis pectinacea	tufted lovegrass	–	–	x
*Eragrostis pilosa**	Indian lovegrass	–	–	x
Eragrostis refracta	coastal lovegrass	x	x	–
Eragrostis secundiflora ssp. *oxylepis*	red lovegrass	–	–	x
*Eremochloa ophiuroides**	centipede grass	x	x	x
Erigeron quercifolius	oakleaf fleabane	–	–	x
Eupatorium capillifolium	dogfennel	x	x	–
Eupatorium compositifolium	yankeeweed	x	–	–
Euphorbia bombensis	Dixie sandmat	–	–	x
Euphorbia maculata	spotted sandmat	–	x	–
Euphorbia polygonifolia	seaside sandmat	–	x	–
Eustachys petraea	pinewoods fingergrass	x	x	x
Euthamia graminifolia	flat-top goldentop	–	x	–
Fimbristylis spadicea	marsh fimbry	x	x	–
Gamochaeta antillana	Caribbean purple everlasting	–	–	x
Gamochaeta pensylvanica	Pennsylvania everlasting	–	–	x
Heterotheca subaxillaris	camphorweed	–	–	x
Houstonia procumbens	roundleaf bluet	–	x	–
Hydrocotyle bonariensis	largeleaf pennywort	x	x	x
Hydrocotyle umbellata	umbrella pennyroyal	x	–	–
Ipomoea pes-caprae	bayhops	–	x	x
Ipomoea imperati	beach morning-glory	–	x	x
Iva imbricata	seacoast marsh elder	x	x	x
Juncus sp.	rush	x	x	–
Juncus bufonius	toad rush	x	–	–
Juncus dichotomus	forked rush	–	–	x
Juncus marginatus	grassleaf rush	–	–	x
Juncus megacephalus	bighead rush	–	x	–
Juncus scirpoides	needlepod rush	–	–	x
Kyllinga brevifolia	shortleaf spikesedge	–	x	–

Scientific Name	Common Name	Hillestad et al. 1975	Dolan 2002	Zomlefer et al. 2008 & 2011
Lipocarpha micrantha	smallflower halfchaff sedge	–	–	x
Ludwigia virgata	savannah primrose-willow	–	x	–
Mikania scandens	climbing hempvine	x	x	–
Mollugo verticillata	carpetweed	x	x	–
Morella cerifera	wax myrtle	x	x	–
Muhlenbergia capillaris	hairawn muhly	–	x	x
Oenothera humifusa	seabeach evening primrose	x	x	x
Oenothera laciniata	cutleaf evening primrose	–	–	x
Opuntia pusilla	cockspur pricklypear	–	x	–
Panicum amarum	bitter panicgrass	–	x	x
Paronychia baldwinii	Baldwin's nailwort	–	–	x
Paspalum sp.	paspalum	–	x	–
Paspalum setaceum	thin paspalum	x	x	x
Paspalum vaginatum	seashore paspalum	x	x	x
Phyla nodiflora	turkey tangle fogfruit	x	x	x
Phyllanthus abnormis	Drummond's leafflower	–	–	x
Physalis sp.	groundcherry	x	x	–
Physalis walteri	Walter's groundcherry	–	–	x
Pinus taeda	loblolly pine	–	–	x
Plantago sp.	plantain	–	x	–
Plantago virginica	Virginia plantain	x	x	x
Pluchea sp.	camphorweed	x	x	–
Polygonum glaucum	seaside knotweed	x	x	–
Polypremum procumbens	juniper-leaf	–	–	x
*Portulaca oleracea**	little hogweed	x	x	–
Rhexia mariana	Maryland meadowbeauty	–	–	x
Rhynchospora colorata	starrush whitetop	x	x	–
Rumex hastatulus	heartwing dock	–	x	x
Sabal palmetto	cabbage palmetto	x	x	–
Sabatia stellaris	rose of Plymouth	x	x	x
Sageretia minutiflora	smallflower mock buckthorn	–	–	x
*Salsola kali**	Russian thistle	–	–	x
Sapindus saponaria var. *saponaria*	wingleaf soapberry	–	–	x
Schizachyrium scoparium var. *littorale*	shore little bluestem	–	x	–
Schoenoplectus americanus	American bulrush	x	x	–
*Scleranthus annuus**	German knotgrass	x	x	–

Scientific Name	Common Name	Hillestad et al. 1975	Dolan 2002	Zomlefer et al. 2008 & 2011
Sesuvium portulacastrum	shoreline seapurslane	–	–	x
Setaria sp.	bristlegrass	x	x	–
Setaria parviflora	yellow bristlegrass	–	–	x
Smilax auriculata	earleaf greenbrier	–	x	x
Smilax glauca	cat greenbrier	–	–	x
Solidago sempervirens	seaside goldenrod	–	–	x
Spartina patens	saltmeadow cordgrass	x	x	x
Sphenopholis obtusata	prairie wedgescale	–	–	x
Sporobolus virginicus	seashore dropseed	x	x	–
Strophostyles helvola	trailing fuzzy-bean	–	–	x
Strophostyles umbellata	pink fuzzybean	–	x	–
Suaeda linearis	annual seepweed	–	x	–
*Tamarix parviflora**	small-flower tamarisk	–	–	x
Tradescantia ohiensis	Ohio spiderwort	–	–	x
Triplasis purpurea	purple sandgrass	–	x	x
Typha sp.	cattail	x	x	–
Uniola paniculata	sea oats	–	x	x
*Verbascum thapsus**	common mullein	–	x	–
Vigna luteola	hairypod cowpea	–	–	x
Yucca aloifolia	aloe yucca	–	–	x
Yucca gloriosa	moundlily yucca	x	x	–
Total	**–**	**42**	**71**	**75**

Appendix G. Mammal species observed in CUIS during the various inventories, surveys, and stranding reports over time.

Table G.1. Mammal species observed in CUIS during the various inventories, surveys, and stranding reports over time.

Common Name	Scientific Name	NPS (2016f)	Bangs (1898)	Hillestad et al. (1975)	Webster (2010)	Castleberry and Morris (2017)	GA DNR Strandings (1996-2016)
African wild ass	*Equus africanus*	–	–	–	H	–	–
American beaver	*Castor canadensis*	–	–	–	H	–	–
American black bear	*Ursus americanus*	H	X	H	H	–	–
Atlantic spotted dolphin	*Stenella frontalis*	–	–	–	–	–	X
Atlantic white-sided dolphin	*Lagenorhynchus acutus*	–	–	–	–	–	X
big brown bat	*Eptesicus fuscus*	X	–	X	–	X	–
black rat	*Rattus rattus*	X	X	X	P	–	–
Bobcat[c]	*Lynx rufus*	X	–	X	X	–	–
bottlenose dolphin	*Tursiops truncatus*	X	–	–	–	–	–
common gray fox[c]	*Urocyon cinereoargenteus*	H	–	H	–	–	–
common raccoon[c]	*Procyon lotor*	X	X	X	X	–	–
cotton mouse	*Peromyscus gossypinus*	X	X	X	X	–	–
Coyote[c]	*Canis latrans*	X	–	–	X	–	–
dwarf sperm whale	*Kogia sima*	P	–	–	–	–	X
eastern fox squirrel	*Sciurus niger*	H	–	X	H	–	–
eastern gray squirrel	*Sciurus carolinensis*	X	X	X	X	–	–
eastern harvest mouse	*Reithrodontomys humulis*	H	–	X	–	–	–
eastern mole	*Scalopus aquaticus*	X	X	X	X	–	–

[a] Indicates a species listed as endangered on the USFWS Endangered Species List

[b] Indicates a species listed as threatened on the USFWS Endangered Species List

[c] Indicates a mesocarnivore species

Table G.1 (continued). Mammal species observed in CUIS during the various inventories, surveys, and stranding reports over time.

Common Name	Scientific Name	NPS (2016f)	Bangs (1898)	Hillestad et al. (1975)	Webster (2010)	Castleberry and Morris (2017)	GA DNR Strandings (1996-2016)
eastern red bat	*Lasiurus borealis*	–	–	–	–	X	–
evening bat	*Nycticeius humeralis*	–	–	–	–	X	–
feral cat[c]	*Felis catus*	H	–	–	P	–	–
feral cattle	*Bos taurus*	–	–	–	H	–	–
feral hog	*Sus scrofa*	X	–	–	X	–	–
feral horse	*Equus caballus*	X	–	–	X	–	–
goose-beaked whale	*Ziphius cavirostris*	P	–	–	–	–	–
hispid cotton rat	*Sigmodon hispidus*	X	X	X	X	–	–
hoary bat	*Lasiurus cinereus*	–	–	–	–	X	–
humpback whale[a]	*Megaptera novaeangliae*	–	–	–	–	–	X
least shrew	*Cryptotis parva*	X	–	X	–	–	–
little brown bat	*Myotis lucifugus*	X	–	–	–	–	–
marsh rabbit	*Sylvilagus palustris*	X	X	X	X	–	–
marsh rice rat	*Oryzomys palustris*	X	X	X	X	–	–
Mink[c]	*Mustela vison*	X	–	X	X	–	–
nine-banded armadillo	*Dasypus novemcinctus*	X	–	X	X	–	–
North Atlantic right whale[a]	*Eubalaena glacialis*	–	–	–	–	–	X
northern yellow bat	*Lasiurus intermedius*	X	–	X	–	X	–
oldfield mouse	*Peromyscus polionotus*	–	–	H	–	–	–
pygmy sperm whale	*Kogia breviceps*	P	–	–	–	–	X

[a] Indicates a species listed as endangered on the USFWS Endangered Species List

[b] Indicates a species listed as threatened on the USFWS Endangered Species List

[c] Indicates a mesocarnivore species

Table G.1 (continued). Mammal species observed in CUIS during the various inventories, surveys, and stranding reports over time.

Common Name	Scientific Name	NPS (2016f)	Bangs (1898)	Hillestad et al. (1975)	Webster (2010)	Castleberry and Morris (2017)	GA DNR Strandings (1996-2016)
river otter[c]	*Lontra canadensis*	X	–	X	X	–	–
rough-toothed dolphin	*Steno bredanensis*	P	–	–	–	–	–
seminole bat	*Lasiurus seminolus*	X	–	X	–	X	–
short-finned pilot whale	*Globicephala macrorhynchus*	P	–	–	–	–	–
silver-haired bat	*Lasionycteris noctivagans*	–	–	–	–	X	–
southeastern pocket gopher	*Geomys pinetis*	H	X	H	H	–	–
southern short-tailed shrew	*Blarina carolinensis*	X	–	X	X	–	–
tri-colored bat (eastern pipistrelle)	*Perimuotis subflavus*	X	–	X	–	X	–
Virginia opossum	*Didelphis virginiana*	X	–	H	X	–	–
West Indian manatee[b]	*Trichechus manatus*	X	–	–	–	–	–
white-tailed deer	*Odocoileus virginianus*	X	X	X	X	–	–
Confirmed	**–**	**26**	**11**	**21**	**17**	**8**	**6**
Probably Present	**–**	**5**	**0**	**0**	**2**	**0**	**0**
Historically Present	**–**	**6**	**0**	**5**	**6**	**0**	**0**

[a] Indicates a species listed as endangered on the USFWS Endangered Species List

[b] Indicates a species listed as threatened on the USFWS Endangered Species List

[c] Indicates a mesocarnivore species

Appendix H. Bird species that have been documented in CUIS by the various surveys and inventory efforts.

Table H.1. Bird species that have been documented in CUIS by the various surveys and inventory efforts. X=confirmed, P=probably present, U=unconfirmed, H=historic record.

Common Name	Scientific Name	NPS (2016f)	CBC (1986, 1992-2015)	Dlugolecki (2012)	MWBS (98-17)	Pearson (1922)	Sprunt (1936)	Byrne et al. (2011)	Kurimo-Beechuk and Byrne (2016)
Acadian flycatcher	*Empidonax virescens*	X	–	–	–	–	X	X	X
American avocet	*Recurvirostra americana*	X	X	–	X	–	–	–	–
American bittern	*Botaurus lentiginosus*	X	X	X	–	–	X	–	–
American black duck	*Anas rubripes*	X	X	–	–	–	–	–	–
American coot	*Fulica americana*	X	X	–	–	–	X	–	–
American crow	*Corvus brachyrhynchos*	X	X	–	–	–	–	X	X
American golden-plover	*Pluvialis dominica*	X	–	–	–	–	–	–	–
American goldfinch	*Carduelis tristis*	X	X	–	–	–	X	–	X
American kestrel	*Falco sparverius*	X	X	–	–	X	X	–	–
American oystercatcher	*Haematopus palliatus*	X	X	–	X	X	X	–	–
American pipit	*Anthus rubescens*	X	X	–	–	–	–	–	–
American redstart	*Setophaga ruticilla*	X	X	–	–	–	X	–	–
American robin	*Turdus migratorius*	X	X	–	–	X	–	–	–
American white pelican	*Pelecanus erythrorhynchos*	X	X	–	–	–	X	–	–
American wigeon	*Anas americana*	X	X	–	–	–	–	–	–
American woodcock	*Scolopax minor*	X	X	–	–	–	–	–	–
anhinga	*Anhinga anhinga*	X	X	X	X	X	X	X	–
Audubon's shearwater	*Puffinus lherminieri*	P	–	–	–	–	–	–	–
Bachman's sparrow	*Aimophila aestivalis*	X	X	–	–	–	X	–	–
Bachman's warbler	*Vermivora bachmanii*	H	–	–	–	–	–	–	–

Common Name	Scientific Name	NPS (2016f)	CBC (1986, 1992-2015)	Dlugolecki (2012)	MWBS (98-17)	Pearson (1922)	Sprunt (1936)	Byrne et al. (2011)	Kurimo-Beechuk and Byrne (2016)
Baird's sandpiper	*Calidris bairdii*	X	–	–	–	–	–	–	–
bald eagle	*Haliaeetus leucocephalus*	X	X	–	–	X	X	–	–
Baltimore oriole	*Icterus galbula*	X	X	–	–	–	–	–	–
bank swallow	*Riparia riparia*	X	–	–	–	–	X	–	–
barn owl	*Tyto alba*	X	–	–	–	–	–	–	–
barn swallow	*Hirundo rustica*	X	–	–	–	X	–	–	–
barred owl	*Strix varia*	X	X	–	–	X	–	–	–
belted kingfisher	*Megaceryle alcyon*	X	X	X	–	X	X	–	–
Bewick's wren	*Thryomanes bewickii*	X	–	–	–	–	–	–	–
black rail	*Laterallus jamaicensis*	X	–	–	–	–	–	–	–
black scoter	*Melanitta nigra*	X	X	–	X	–	–	–	–
black skimmer	*Rynchops niger*	X	X	–	X	X	X	–	–
black tern	*Chlidonias niger*	X	–	–	–	–	–	–	–
black vulture	*Coragyps atratus*	X	X	X	–	X	X	X	–
black-and-white warbler	*Mniotilta varia*	X	X	–	–	X	X	X	–
black-bellied plover	*Pluvialis squatarola*	X	X	–	X	–	X	–	–
black-billed cuckoo	*Coccyzus erythropthalmus*	X	–	–	–	–	–	–	–
blackburnian warbler	*Setophaga fusca*	X	–	–	–	–	–	–	–
black-crowned night-heron	*Nycticorax nycticorax*	X	X	X	–	X	X	–	–
black-headed gull	*Chroicocephalus ridibundus*	–	X	–	–	–	–	–	–
black-legged kittiwake	*Rissa tridactyla*	P	–	–	–	–	–	–	–
black-necked stilt	*Himantopus mexicanus*	X	–	–	–	–	–	–	–
blackpoll warbler	*Setophaga striata*	X	–	–	–	X	–	–	–
black-throated blue warbler	*Setophaga caerulescens*	X	X	–	–	–	X	–	X
black-throated green warbler	*Setophaga virens*	X	X	–	–	–	X	–	–
blue grosbeak	*Guiraca caerulea*	X	–	–	–	–	–	–	–

Common Name	Scientific Name	NPS (2016f)	CBC (1986, 1992-2015)	Dlugolecki (2012)	MWBS (98-17)	Pearson (1922)	Sprunt (1936)	Byrne et al. (2011)	Kurimo-Beechuk and Byrne (2016)
blue jay	*Cyanocitta cristata*	X	X	–	–	X	X	X	X
blue-gray gnatcatcher	*Polioptila caerulea*	X	X	–	–	X	X	X	X
blue-headed vireo	*Vireo solitarius*	X	X	–	–	–	X	–	–
blue-winged teal	*Anas discors*	X	X	X	–	–	X	–	–
blue-winged warbler	*Vermivora pinus*	X	–	–	–	–	X	–	–
boat-tailed grackle	*Quiscalus major*	X	X	–	–	X	X	–	X
bobolink	*Dolichonyx oryzivorus*	X	–	–	–	X	–	–	–
Bonaparte's gull	*Larus philadelphia*	X	X	–	X	–	X	–	–
brant	*Branta bernicla*	X	X	–	–	–	–	–	–
bridled tern	*Sterna anaethetus*	P	–	–	–	–	–	–	–
broad-winged hawk	*Buteo platypterus*	X	–	–	–	–	–	–	–
brown creeper	*Certhia americana*	X	X	–	–	–	–	–	–
brown noddy	*Anous stolidus*	P	–	–	–	–	–	–	–
brown pelican	*Pelecanus occidentalis*	X	X	–	X	X	X	–	–
brown thrasher	*Toxostoma rufum*	X	X	–	–	X	X	–	X
brown-headed cowbird	*Molothrus ater*	X	X	–	–	–	–	–	X
brown-headed nuthatch	*Sitta pusilla*	X	X	–	–	X	X	X	X
buff-breasted sandpiper	*Tryngites subruficollis*	X	–	–	–	–	–	–	–
bufflehead	*Bucephala albeola*	X	X	–	X	–	–	–	–
burrowing owl	*Athene cunicularia*	X	–	–	–	–	–	–	–
Canada goose	*Branta canadensis*	X	X	–	–	–	X	–	–
Canada warbler	*Wilsonia canadensis*	X	–	–	–	–	–	–	–
canvasback	*Aythya valisineria*	X	X	–	–	–	–	–	–
Cape May warbler	*Setophaga tigrina*	X	–	–	–	–	–	–	–
Carolina chickadee	*Poecile carolinensis*	X	X	–	–	–	X	X	X
Carolina wren	*Thryothorus ludovicianus*	X	X	–	–	X	X	X	X
Caspian tern	*Sterna caspia*	X	X	–	X	–	–	–	–

Common Name	Scientific Name	NPS (2016f)	CBC (1986, 1992-2015)	Dlugolecki (2012)	MWBS (98-17)	Pearson (1922)	Sprunt (1936)	Byrne et al. (2011)	Kurimo-Beechuk and Byrne (2016)
cattle egret	*Bubulcus ibis*	X	X	X	–	–	–	–	–
cedar waxwing	*Bombycilla cedrorum*	X	X	–	–	–	X	X	–
chestnut-sided warbler	*Setophaga pensylvanica*	X	–	–	–	–	–	–	–
chimney swift	*Chaetura pelagica*	X	–	–	–	X	X	–	X
chipping sparrow	*Spizella passerina*	X	X	–	–	–	–	–	–
chuck-will's-widow	*Caprimulgus carolinensis*	X	–	–	–	X	X	X	–
clapper rail	*Rallus longirostris*	X	–	–	–	X	X	–	–
clay-colored sparrow	*Spizella pallida*	U	X	–	–	–	X	–	–
cliff swallow	*Petrochelidon pyrrhonota*	X	–	–	–	–	–	–	–
common gallinule	*Gallinula galeata*	–	X	X	–	X	X	–	–
common goldeneye	*Bucephala clangula*	X	X	–	X	–	–	–	–
common grackle	*Quiscalus quiscula*	X	X	–	–	X	X	X	X
common ground-dove	*Columbina passerina*	X	X	–	–	X	X	–	–
common loon	*Gavia immer*	X	X	–	X	–	–	–	–
common merganser	*Mergus merganser*	X	–	–	–	–	–	–	–
common moorhen	*Gallinula chloropus*	X	–	–	–	–	–	X	–
common nighthawk	*Chordeiles minor*	X	–	–	–	X	X	–	X
common snipe	*Gallinago gallinago*	X	–	–	–	–	–	–	–
common tern	*Sterna hirundo*	X	–	–	–	X	X	–	–
common yellowthroat	*Geothlypis trichas*	X	X	–	–	–	X	X	X
connecticut warbler	*Oporornis agilis*	X	–	–	–	–	–	–	–
cooper's hawk	*Accipiter cooperii*	X	X	–	–	–	X	–	–
cory's shearwater	*Calonectris diomedea*	P	–	–	–	–	–	–	–
dark-eyed junco	*Junco hyemalis*	X	X	–	–	–	–	X	–
double-crested cormorant	*Phalacrocorax auritus*	X	X	–	X	X	X	–	–
dovekie	*Alle alle*	P	–	–	–	–	–	–	–
downy woodpecker	*Picoides pubescens*	X	X	–	–	X	X	X	X
dunlin	*Calidris alpina*	X	X	–	X	–	X	–	–

Common Name	Scientific Name	NPS (2016f)	CBC (1986, 1992-2015)	Dlugolecki (2012)	MWBS (98-17)	Pearson (1922)	Sprunt (1936)	Byrne et al. (2011)	Kurimo-Beechuk and Byrne (2016)
eastern bluebird	*Sialia sialis*	X	X	–	–	X	–	–	X
eastern kingbird	*Tyrannus tyrannus*	X	–	–	–	X	–	–	–
eastern meadowlark	*Sturnella magna*	X	X	–	–	X	X	–	–
eastern phoebe	*Sayornis phoebe*	X	X	–	–	–	–	X	X
eastern screech-owl	*Megascops asio*	X	X	–	–	–	X	X	–
eastern towhee	*Pipilo erythrophthalmus*	X	X	–	–	X	X	X	X
eastern wood-pewee	*Contopus virens*	X	–	–	–	X	X	X	X
Eurasian collared-dove	*Streptopelia decaocto*	P	X	–	–	–	–	–	–
Eurasian wigeon	*Anas penelope*	X	–	–	–	–	–	–	–
European starling	*Sturnus vulgaris*	X	X	–	–	–	–	–	–
field sparrow	*Spizella pusilla*	X	X	–	–	–	X	–	–
fish crow	*Corvus ossifragus*	X	X	–	–	X	X	–	X
Forster's tern	*Sterna forsteri*	X	X	–	X	–	X	–	–
fox sparrow	*Passerella iliaca*	X	X	–	–	–	–	–	–
Franklin's gull	*Leucophaeus pipixcan*	–	X	–	–	–	–	–	–
fulvous whistling-duck	*Dendrocygna bicolor*	X	–	–	–	–	–	–	–
gadwall	*Anas strepera*	X	X	–	–	–	–	–	–
glaucous gull	*Larus hyperboreus*	X	X	–	X	–	–	–	–
glossy ibis	*Plegadis falcinellus*	X	X	X	–	–	X	–	–
golden eagle	*Aquila chrysaetos*	X	–	–	–	–	–	–	–
golden-crowned kinglet	*Regulus satrapa*	X	X	–	–	–	X	–	–
golden-winged warbler	*Vermivora chrysoptera*	X	–	–	–	–	–	–	–
grasshopper sparrow	*Ammodramus savannarum*	X	X	–	–	–	–	–	–
gray catbird	*Dumetella carolinensis*	X	X	–	–	X	X	X	–
gray kingbird	*Tyrannus dominicensis*	X	–	–	–	–	–	–	–
gray-cheeked thrush	*Catharus minimus*	X	–	–	–	–	–	–	–

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great black-backed gull	*Larus marinus*	X	X	–	X	–	–	–	–
great blue heron	*Ardea herodias*	X	X	X	X	X	X	X	X
great cormorant	*Phalacrocorax carbo*	X	–	–	–	–	–	–	–
great crested flycatcher	*Myiarchus crinitus*	X	–	–	–	X	X	X	X
great egret	*Ardea alba*	X	X	X	X	X	X	–	–
great horned owl	*Bubo virginianus*	X	X	–	–	X	X	–	–
greater scaup	*Aythya marila*	X	X	–	X	–	–	–	–
greater shearwater	*Puffinus gravis*	P	–	–	–	–	–	–	–
greater yellowlegs	*Tringa melanoleuca*	X	X	X	X	–	X	–	–
green heron	*Butorides virescens*	X	X	X	–	X	X	–	X
green-winged teal	*Anas crecca*	X	X	–	–	–	–	–	–
gull-billed tern	*Sterna nilotica*	X	–	–	–	–	X	–	–
hairy woodpecker	*Picoides villosus*	X	X	–	–	–	X	–	–
henslow's sparrow	*Ammodramus henslowii*	X	–	–	–	–	–	–	–
hermit thrush	*Catharus guttatus*	X	X	–	–	–	X	–	–
herring gull	*Larus argentatus*	X	X	–	X	X	X	–	–
hooded merganser	*Lophodytes cucullatus*	X	X	X	X	–	X	–	–
hooded warbler	*Wilsonia citrina*	X	–	–	–	–	X	X	X
horned grebe	*Podiceps auritus*	X	X	–	X	–	–	–	–
house finch	*Carpodacus mexicanus*	U	X	–	–	–	–	–	X
house sparrow	*Passer domesticus*	X	X	–	–	–	–	–	–
house wren	*Troglodytes aedon*	X	X	–	–	–	X	–	–
iceland gull	*Larus glaucoides*	X	X	–	X	–	–	–	–
indigo bunting	*Passerina cyanea*	X	–	–	–	–	–	–	–
Kentucky warbler	*Oporornis formosus*	X	–	–	–	–	–	–	–
killdeer	*Charadrius vociferus*	X	X	X	X	–	X	–	–
king eider	*Somateria spectabilis*	X	–	–	–	–	–	–	–

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king rail	*Rallus elegans*	X	X	X	–	–	X	–	–
Kirtland's warbler	*Setophaga kirtlandii*	X	–	–	–	–	–	–	–
lapland longspur	*Calcarius lapponicus*	X	–	–	–	–	–	–	–
lark bunting	*Calamospiza melanocorys*	X	–	–	–	–	–	–	–
lark sparrow	*Chondestes grammacus*	X	–	–	–	–	–	–	–
laughing gull	*Larus atricilla*	X	X	–	X	X	X	X	X
Le Conte's sparrow	*Ammodramus leconteii*	X	–	–	–	–	–	–	–
Leach's storm petrel	*Oceanodroma leucorhoa*	P	–	–	–	–	–	–	–
least bittern	*Ixobrychus exilis*	X	–	–	–	–	–	–	–
least flycatcher	*Empidonax minimus*	X	–	–	–	–	–	–	–
least sandpiper	*Calidris minutilla*	X	X	–	X	X	X	–	–
least tern	*Sternula antillarum*	X	–	–	–	X	–	–	–
lesser black-backed gull	*Larus fuscus*	X	X	–	X	–	–	–	–
lesser scaup	*Aythya affinis*	X	X	–	X	–	X	–	–
lesser yellowlegs	*Tringa flavipes*	X	X	–	X	X	X	–	–
limpkin	*Aramus guarauna*	U	–	–	–	–	–	–	–
Lincoln's sparrow	*Melospiza lincolnii*	X	X	–	–	–	–	–	–
little blue heron	*Egretta caerulea*	X	X	X	X	X	X	X	–
loggerhead shrike	*Lanius ludovicianus*	X	X	–	–	X	X	–	X
long-billed curlew	*Numenius americanus*	X	X	–	X	–	–	–	–
long-billed dowitcher	*Limnodromus scolopaceus*	X	X	–	–	–	–	–	–
long-billed marsh wren	*Telmatodytes palustris*	U	–	–	–	–	–	–	–
long-eared owl	*Asio otus*	H	–	–	–	–	–	–	–
Louisiana waterthrush	*Seiurus motacilla*	X	–	–	–	–	X	–	–
magnificent frigatebird	*Fregata magnificens*	X	X	–	–	–	–	–	–
magnolia warbler	*Setophaga magnolia*	X	–	–	–	X	–	–	–
mallard	*Anas platyrhynchos*	X	X	–	X	–	–	–	–
manx shearwater	*Puffinus puffinus*	–	X	–	–	–	–	–	–

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marbled godwit	*Limosa fedoa*	X	X	–	X	–	–	–	–
marsh wren	*Cistothorus palustris*	X	X	–	–	X	X	–	–
merlin	*Falco columbarius*	X	X	–	–	–	X	–	–
Mississippi kite	*Ictinia mississippiensis*	X	–	–	–	–	–	–	–
mottled duck	*Anas fulvigula*	X	X	–	–	–	–	–	–
mountain plover	*Charadrius montanus*	X	–	–	–	–	–	–	–
mourning dove	*Zenaida macroura*	X	X	–	–	X	X	X	X
Nashville warbler	*Vermivora ruficapilla*	X	X	–	–	–	–	–	–
Nelson's sharp-tailed sparrow	*Ammodramus nelsoni*	X	X	–	–	–	–	–	–
northern bobwhite	*Colinus virginianus*	U	X	–	–	–	–	–	–
northern cardinal	*Cardinalis cardinalis*	X	X	–	–	X	X	X	X
northern flicker	*Colaptes auratus*	X	X	–	–	–	X	–	X
northern gannet	*Morus bassanus*	X	X	–	X	–	X	–	–
northern harrier	*Circus cyaneus*	X	X	X	–	X	X	–	–
northern mockingbird	*Mimus polyglottos*	X	X	–	–	X	X	–	X
northern parula	*Parula americana*	X	X	–	–	X	X	X	X
northern pintail	*Anas acuta*	X	X	–	–	–	–	–	–
northern rough-winged swallow	*Stelgidopteryx serripennis*	X	–	–	–	–	–	–	–
northern saw-whet owl	*Aegolius acadicus*	X	–	–	–	–	–	–	–
northern shoveler	*Anas clypeata*	X	X	–	–	–	–	–	–
northern waterthrush	*Seiurus noveboracensis*	X	–	–	–	–	–	–	–
oldsquaw	*Clangula hyemalis*	X	X	–	–	–	–	–	–
orange-crowned warbler	*Vermivora celata*	X	X	–	–	–	–	–	–
orchard oriole	*Icterus spurius*	X	–	–	–	–	X	–	–
osprey	*Pandion haliaetus*	X	X	X	–	X	X	X	X
ovenbird	*Seiurus aurocapillus*	X	X	–	–	X	X	X	–

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painted bunting	*Passerina ciris*	X	X	–	–	X	X	–	X
palm warbler	*Setophaga palmarum*	X	X	–	–	X	X	–	–
parasitic jaeger	*Stercorarius parasiticus*	P	X	–	X	–	–	–	–
pectoral sandpiper	*Calidris melanotos*	X	X	–	–	–	–	–	–
peregrine falcon	*Falco peregrinus*	X	X	–	–	X	–	–	–
Philadelphia vireo	*Vireo philadelphicus*	X	–	–	–	–	–	–	–
pied-billed grebe	*Podilymbus podiceps*	X	X	X	–	–	X	–	–
pileated woodpecker	*Dryocopus pileatus*	X	X	–	–	X	X	X	X
pine siskin	*Carduelis pinus*	X	X	–	–	–	–	–	–
pine warbler	*Setophaga pinus*	X	X	–	–	X	X	X	X
piping plover	*Charadrius melodus*	X	X	–	X	–	X	–	–
pomarine jaeger	*Stercorarius pomarinus*	P	X	–	X	–	–	–	–
prairie warbler	*Setophaga discolor*	X	X	–	–	–	X	X	X
prothonotary warbler	*Protonotaria citrea*	X	–	–	–	X	X	–	–
purple finch	*Carpodacus purpureus*	X	X	–	–	–	–	–	–
purple gallinule	*Porphyrula martinica*	X	–	–	–	X	X	–	–
purple martin	*Progne subis*	X	–	–	–	–	–	–	X
purple sandpiper	*Calidris maritima*	X	X	–	X	–	–	–	–
razorbill	*Alca torda*	P	X	–	–	–	–	–	–
red crossbill	*Loxia curvirostra*	X	–	–	–	–	–	–	–
red knot	*Calidris canutus*	X	X	–	X	–	X	–	–
red phalarope	*Phalaropus fulicaria*	X	–	–	–	–	–	–	–
red-bellied woodpecker	*Melanerpes carolinus*	X	X	–	–	X	X	X	X
red-breasted merganser	*Mergus serrator*	X	X	–	X	–	X	–	–
red-breasted nuthatch	*Sitta canadensis*	X	X	–	–	–	–	–	–
red-cockaded woodpecker	*Picoides borealis*	X	–	–	–	–	X	–	–

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reddish egret	*Egretta rufescens*	X	X	–	X	–	–	–	–
red-eyed vireo	*Vireo olivaceus*	X	–	–	–	X	X	X	X
redhead	*Aythya americana*	X	X	–	–	–	–	–	–
red-headed woodpecker	*Melanerpes erythrocephalus*	X	X	–	–	X	X	–	X
red-necked grebe	*Podiceps grisegena*	X	–	–	–	–	–	–	–
red-necked phalarope	*Phalaropus lobatus*	X	–	–	–	–	–	–	–
red-shouldered hawk	*Buteo lineatus*	X	X	X	–	X	X	X	X
red-tailed hawk	*Buteo jamaicensis*	X	X	–	–	X	–	–	–
red-throated loon	*Gavia stellata*	X	X	–	X	–	–	–	–
red-winged blackbird	*Agelaius phoeniceus*	X	X	–	–	X	X	X	X
ring-billed gull	*Larus delawarensis*	X	X	–	X	–	X	–	–
ring-necked duck	*Aythya collaris*	X	X	–	–	–	–	–	–
rock dove	*Columba livia*	X	X	–	–	–	–	–	–
roseate spoonbill	*Ajaia ajaja*	X	X	X	–	–	–	–	–
rough-legged hawk	*Buteo lagopus*	X	–	–	–	–	–	–	–
royal tern	*Sterna maxima*	X	X	–	X	–	X	–	X
ruby-crowned kinglet	*Regulus calendula*	X	X	–	–	–	X	–	–
ruby-throated hummingbird	*Archilochus colubris*	X	–	–	–	X	X	X	–
ruddy duck	*Oxyura jamaicensis*	X	X	X	–	–	–	–	–
ruddy turnstone	*Arenaria interpres*	X	X	–	X	X	X	–	–
rusty blackbird	*Euphagus carolinus*	X	X	–	–	–	X	–	–
Sabine's gull	*Xema sabini*	X	–	–	–	–	–	–	–
sanderling	*Calidris alba*	X	X	–	X	–	X	–	–
sandhill crane	*Grus canadensis*	X	–	–	–	–	–	–	–
sandwich tern	*Sterna sandvicensis*	X	X	X	X	–	–	–	–
savannah sparrow	*Passerculus sandwichensis*	X	X	–	–	–	X	–	–
scarlet tanager	*Piranga olivacea*	X	–	–	–	X	–	X	–

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seaside sparrow	*Ammodramus maritimus*	X	X	–	–	–	X	–	–
sedge wren	*Cistothorus platensis*	X	X	–	–	–	–	–	–
semipalmated plover	*Charadrius semipalmatus*	X	X	–	X	X	X	–	–
semipalmated sandpiper	*Calidris pusilla*	X	–	–	–	X	X	–	–
sharp-shinned hawk	*Accipiter striatus*	X	X	–	–	–	X	–	–
saltmarsh sparrow	*Ammodramus caudacutus*	X	X	–	–	–	–	–	–
short-billed dowitcher	*Limnodromus griseus*	X	X	–	X	–	X	–	–
short-eared owl	*Asio flammeus*	X	–	–	–	–	–	–	–
snow bunting	*Plectrophenax nivalis*	X	–	–	–	–	–	–	–
snow goose	*Chen caerulescens*	X	X	–	–	–	–	–	–
snowy egret	*Egretta thula*	X	X	X	X	X	X	X	–
snowy owl	*Bubo scandiacus*	H	–	–	–	–	–	–	–
solitary sandpiper	*Tringa solitaria*	X	–	–	–	X	–	–	–
song sparrow	*Melospiza melodia*	X	X	–	–	–	X	–	–
sooty shearwater	*Puffinus griseus*	P	–	–	–	–	–	–	–
sooty tern	*Sterna fuscata*	P	–	–	–	–	–	–	–
sora	*Porzana carolina*	X	X	X	–	–	–	–	–
spotted sandpiper	*Actitis macularia*	X	X	X	X	X	X	–	–
sprague's pipit	*Anthus spragueii*	X	–	–	–	–	–	–	–
stilt sandpiper	*Calidris himantopus*	X	–	–	X	–	–	–	–
summer tanager	*Piranga rubra*	X	–	–	–	X	X	X	X
surf scoter	*Melanitta perspicillata*	X	X	–	X	X	–	–	–
Swainson's thrush	*Catharus ustulatus*	X	–	–	–	–	–	–	–
Swainson's warbler	*Limnothlypis swainsonii*	X	–	–	–	–	–	–	–
swallow-tailed kite	*Elanoides forficatus*	X	–	–	–	–	–	–	–
swamp sparrow	*Melospiza georgiana*	X	X	–	–	X	X	–	–
Tennessee warbler	*Vermivora peregrina*	X	–	–	–	–	–	–	–
tree swallow	*Tachycineta bicolor*	X	X	–	–	X	X	–	–

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tricolored heron	*Egretta tricolor*	X	X	X	X	X	X	–	–
tufted titmouse	*Baeolophus bicolor*	X	X	–	–	–	X	X	X
tundra swan	*Cygnus columbianus*	X	–	–	–	–	X	–	–
turkey vulture	*Cathartes aura*	X	X	X	–	X	X	X	–
upland sandpiper	*Bartramia longicauda*	X	–	–	–	–	–	–	–
veery	*Catharus fuscescens*	X	–	–	–	–	–	–	–
Vermilion flycatcher	*Pyrocephalus rubinus*	X	–	–	–	–	–	–	–
vesper sparrow	*Pooecetes gramineus*	X	X	–	–	–	–	–	–
virginia rail	*Rallus limicola*	X	X	X	–	–	–	–	–
western kingbird	*Tyrannus verticalis*	X	X	–	–	–	–	–	–
western meadowlark	*Sturnella neglecta*	X	–	–	–	–	–	–	–
western sandpiper	*Calidris mauri*	X	X	–	X	–	X	–	–
whimbrel	*Numenius phaeopus*	X	X	–	X	X	X	–	–
whip-poor-will	*Caprimulgus vociferus*	X	X	–	–	–	–	–	–
white ibis	*Eudocimus albus*	X	X	X	X	X	X	–	–
white-breasted nuthatch	*Sitta carolinensis*	X	X	–	–	–	–	–	–
white-crowned sparrow	*Zonotrichia leucophrys*	X	–	–	–	–	–	–	–
white-eyed vireo	*Vireo griseus*	X	X	–	–	X	X	X	X
white-rumped sandpiper	*Calidris fuscicollis*	X	–	–	–	–	–	–	–
white-tailed tropicbird	*Phaethon lepturus*	U	–	–	–	–	–	–	–
white-throated sparrow	*Zonotrichia albicollis*	X	X	–	–	–	X	–	–
white-winged dove	*Zenaida asiatica*	X	X	–	–	–	–	–	–
white-winged scoter	*Melanitta fusca*	X	X	–	X	–	–	–	–
wild turkey	*Meleagris gallopavo*	X	X	X	–	X	X	X	X
willet	*Catoptrophorus semipalmatus*	X	X	–	X	X	X	–	X
willow flycatcher	*Empidonax traillii*	X	–	–	–	–	–	–	–

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Wilson's phalarope	*Phalaropus tricolor*	X	X	–	–	–	–	–	–
Wilson's plover	*Charadrius wilsonia*	X	X	–	X	X	X	–	–
Wilson's snipe	*Gallinago delicata*	–	X	–	–	–	X	–	–
Wilson's storm-petrel	*Oceanites oceanicus*	P	–	–	–	–	–	–	–
Wilson's warbler	*Wilsonia pusilla*	X	–	–	–	–	–	–	–
winter wren	*Troglodytes troglodytes*	X	X	–	–	–	–	–	–
wood duck	*Aix sponsa*	X	X	X	–	X	X	–	–
wood stork	*Mycteria americana*	X	X	X	X	–	X	–	–
wood thrush	*Hylocichla mustelina*	X	–	–	–	–	–	–	–
worm-eating warbler	*Helmitheros vermivorus*	X	–	–	–	–	–	–	–
yellow rail	*Coturnicops noveboracensis*	X	–	X	–	–	–	–	–
yellow warbler	*Setophaga petechia*	X	–	–	–	X	–	X	–
yellow-bellied flycatcher	*Empidonax flaviventris*	X	–	–	–	–	–	–	–
yellow-bellied sapsucker	*Sphyrapicus varius*	X	X	–	–	–	–	–	–
yellow-billed cuckoo	*Coccyzus americanus*	X	–	–	–	X	X	–	X
yellow-breasted chat	*Icteria virens*	X	–	–	–	X	–	–	X
yellow-crowned night-heron	*Nyctanassa violacea*	X	X	X	–	X	X	–	–
yellow-headed blackbird	*Xanthocephalus xanthocephalus*	X	–	–	–	–	–	–	–
yellow-rumped warbler	*Setophaga coronata*	X	X	–	–	–	X	X	–
yellow-throated vireo	*Vireo flavifrons*	X	–	–	–	–	–	–	X
yellow-throated warbler	*Dendroica dominica*	X	X	–	–	–	X	X	X
Total Number of Species	**–**	**331**	**211**	**36**	**63**	**97**	**147**	**50**	**55**
Confirmed	**–**	**307**	**–**	**–**	**–**	**–**	**–**	**–**	**–**
Probably Present	**–**	**15**	**–**	**–**	**–**	**–**	**–**	**–**	**–**

Common Name	Scientific Name	NPS (2016f)	CBC (1986, 1992-2015)	Dlugolecki (2012)	MWBS (98-17)	Pearson (1922)	Sprunt (1936)	Byrne et al. (2011)	Kurimo-Beechuk and Byrne (2016)
Unconfirmed	**–**	**6**	**–**	**–**	**–**	**–**	**–**	**–**	**–**
Historic Record	**–**	**3**	**–**	**–**	**–**	**–**	**–**	**–**	**–**

Appendix I. Summary of sea turtle nest numbers at CUIS, 1974-2016.

Table I.1. Summary of sea turtle nest numbers at CUIS, 1974-2016.

Year	Surveyed Length (km)	Loggerhead Nests	Green Nests	Leatherback Nests	Unknown sp. Nests
1974	8.0	195	–	–	–
1975	8.0	172	–	–	–
1976	8.0	134	–	–	–
1977	8.0	115	–	–	–
1979[a]	8.0	287	–	–	–
1980	8.0	234	–	–	–
1982	8.0	172	–	–	–
1983	≥26.0	217	–	–	–
1984	8.0	155	–	–	–
1985	≥26.0	204	–	–	–
1986	27.2	196	–	–	–
1987	27.2	172	–	–	–
1988	27.2	164	–	–	–
1989	27.2	158	–	–	–
1990	27.2	231	–	–	–
1991	27.2	245	–	–	–
1992	26.0	229	–	–	–
1993	26.0	92	–	–	–
1994	27.0	255	–	–	–
1995	28.0	203	–	–	–
1996	28.0	194	–	–	–
1997	28.0	188	–	–	–
1998	28.2	234	–	–	–
1999	28.2	260	–	–	–
2000	28.7	181	–	–	–
2001	27.6	196	–	2[b]	–
2002	26.8	189	0	0	0
2003	28.4	322	0	1	0
2004	28.4	53	0	0	0
2005	28.4	232	0	0	0
2006	28.4	325	0	0	0

[a] Information in this row from Stoneburner (1979).

[b] Reported by Rabon et al. (2003)

[c] Information in this row is from Seaturtle.org (2017); it is preliminary and subject to change.

Table I.1 (continued). Summary of sea turtle nest numbers at CUIS, 1974-2016.

Year	Surveyed Length (km)	Loggerhead Nests	Green Nests	Leatherback Nests	Unknown sp. Nests
2007	28.4	177	0	0	0
2008	28.4	335	0	0	1
2009	28.4	250	0	2	0
2010	28.4	483	3	0	0
2011	28.4	366	1	5	0
2012	28.4	699	0	1	0
2013	28.4	547	14	0	0
2014	28.4	318	1	0	0
2015	28.4	575	3	1	4
2016	28.4	866	1	0	0
2017[c]	28.4	513	11	1	1

[a] Information in this row from Stoneburner (1979).

[b] Reported by Rabon et al. (2003)

[c] Information in this row is from Seaturtle.org (2017); it is preliminary and subject to change.

Appendix J. Aerial and ground photos showing examples of back-barrier shoreline erosion.

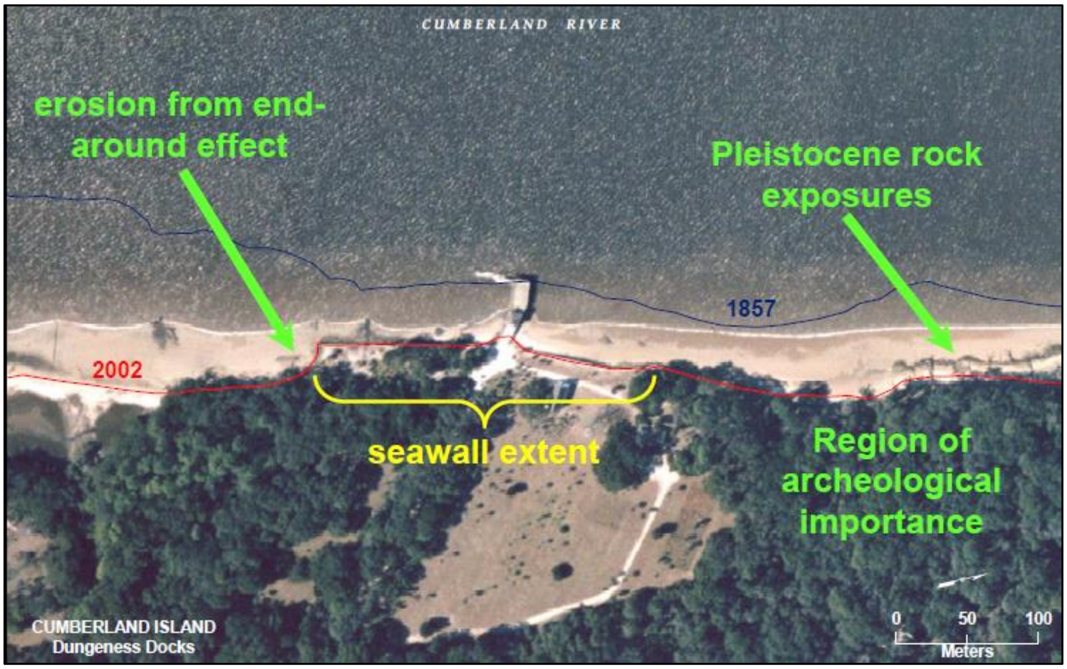

Figure J.1. Aerial photo showing shoreline position in the Dungeness Dock area in 1857 (blue line) and 2002 (red line) (Jackson 2006).

Figure J.2. Aerial photo showing shoreline position in the Plum Orchard area in 1870 (blue line) and 2002 (red line) (Jackson 2006).

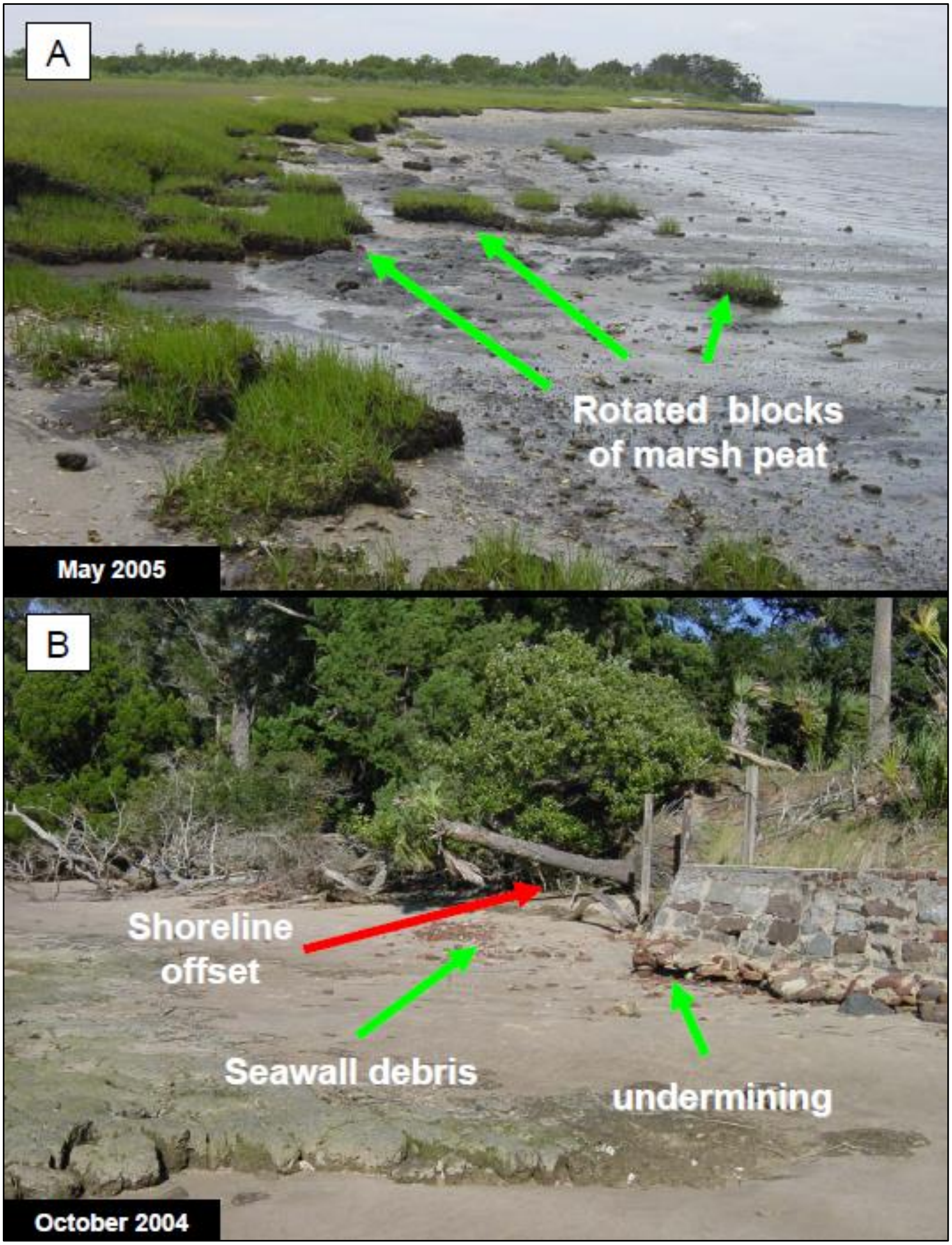

Figure J.3. Examples of shoreline erosion enhanced by tidal/inlet processes south of Dungeness Dock (A) and end-around effects of the Dungeness seawall (B) (Jackson 2006).

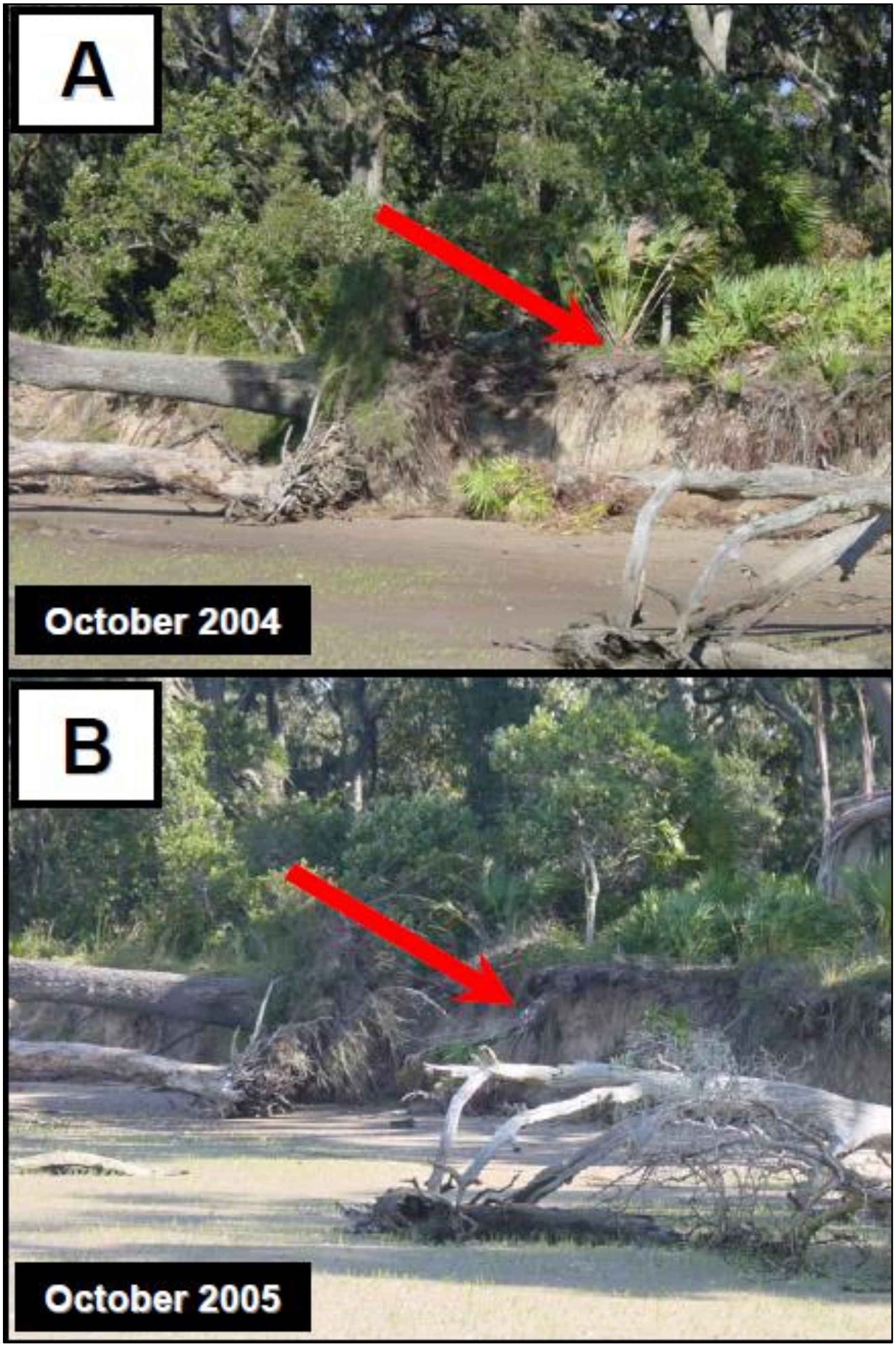

Figure J.4. Photos showing shoreline erosion in the Dungeness area from 2004 (A) to 2005 (B). The red arrow highlights a palmetto on the shore edge in 2004 that had collapsed by 2005. Also note the increased exposure of fallen tree roots left of the palmetto (Jackson 2006).

Appendix K. Oceanfront shoreline change at CUIS, 1857-1993 (Pendleton et al. 2004).

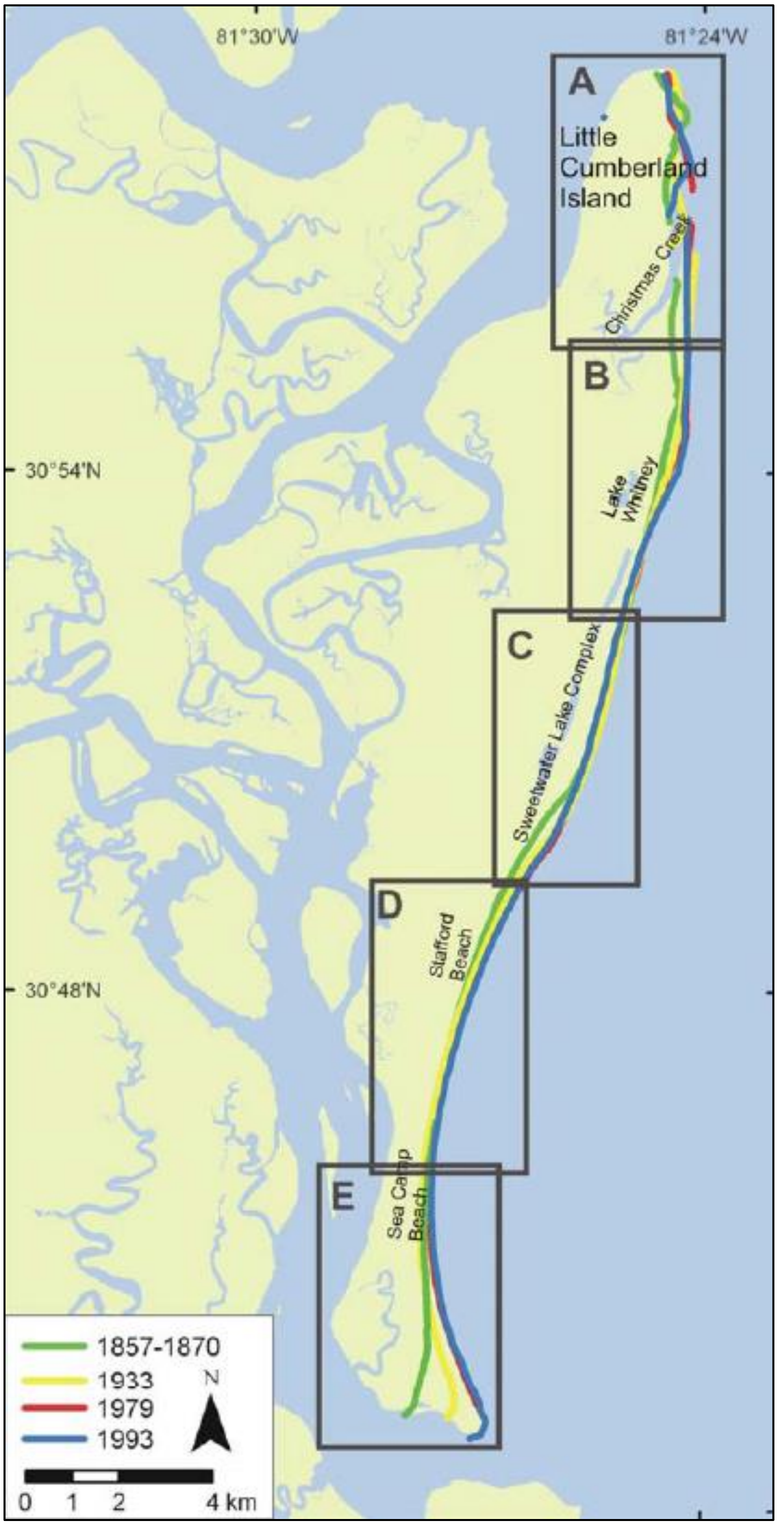

Figure K.1. Oceanfront shoreline change at CUIS, 1857-1993 (Pendleton et al. 2004).

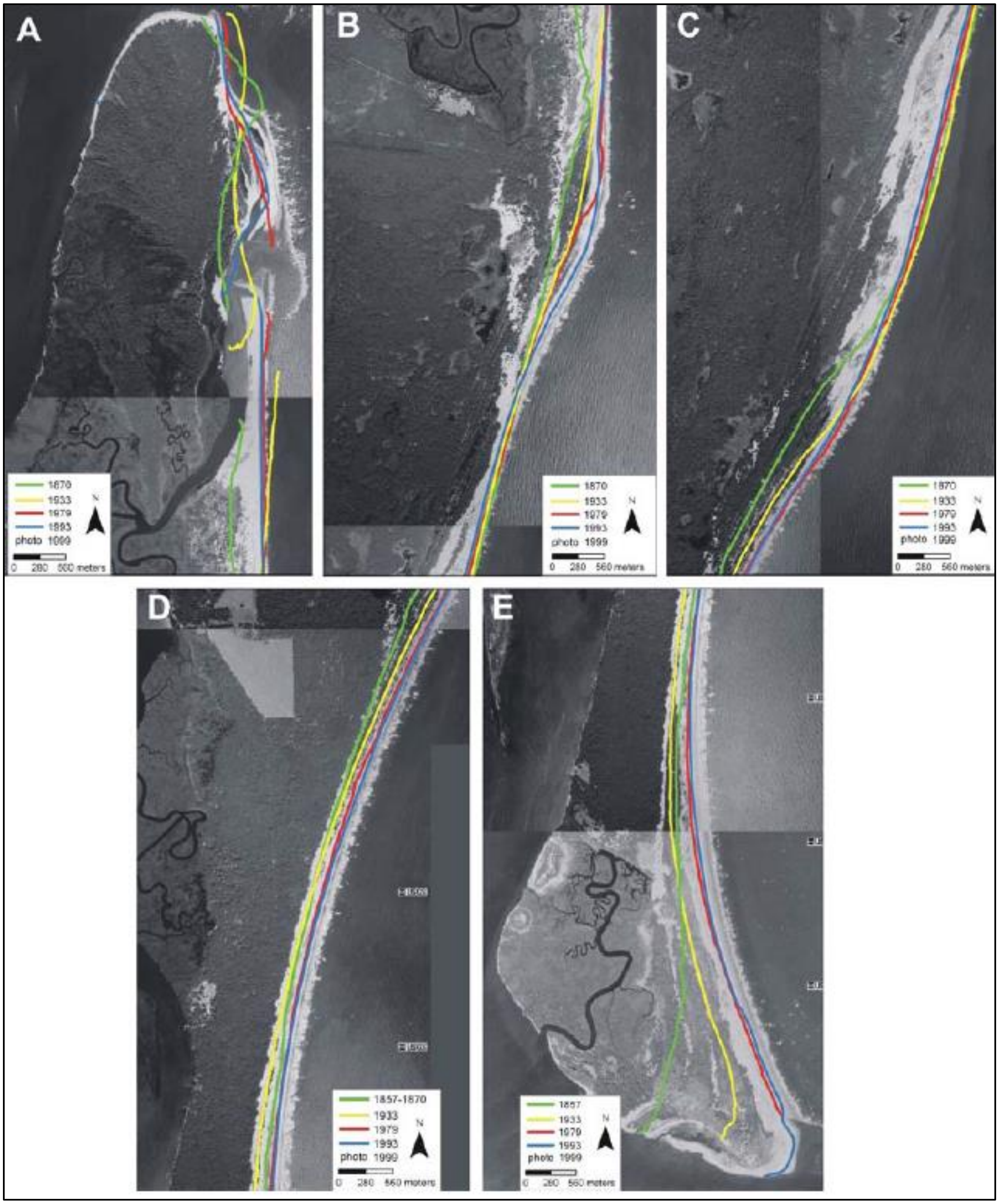

Figure K.2. Close-up aerial photos showing oceanfront shoreline change at CUIS from north (A) to south (E) (Pendleton et al. 2004). The small area of net erosion is seen towards the top of image C, while the accretion related to the jetty is illustrated in image E.

Appendix L. Dune crest elevations for the easternmost dune ridge at CUIS.

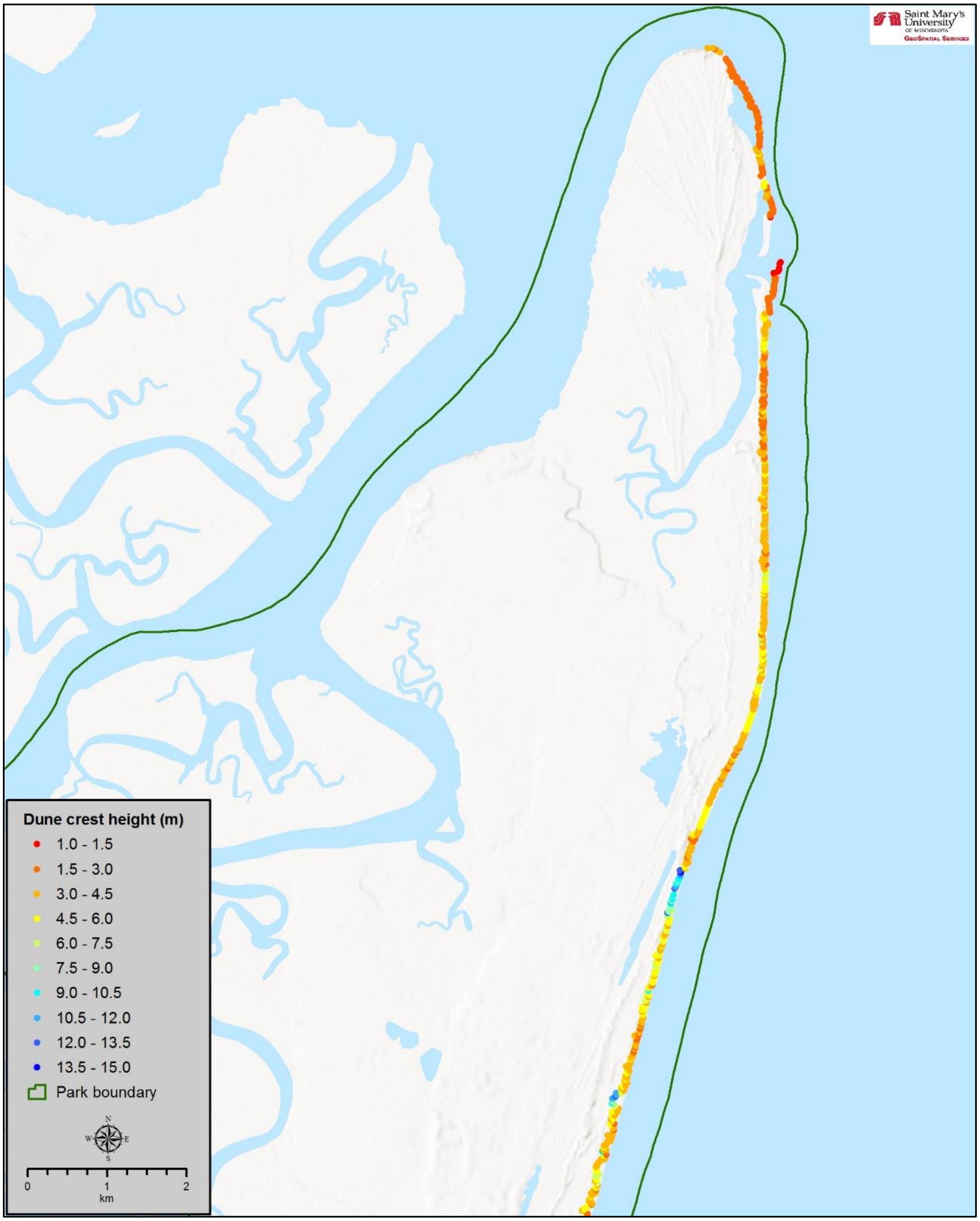

Figure L.1. Dune crest height (elevation) in the northern portion of CUIS (Stockdon et al. 2007).

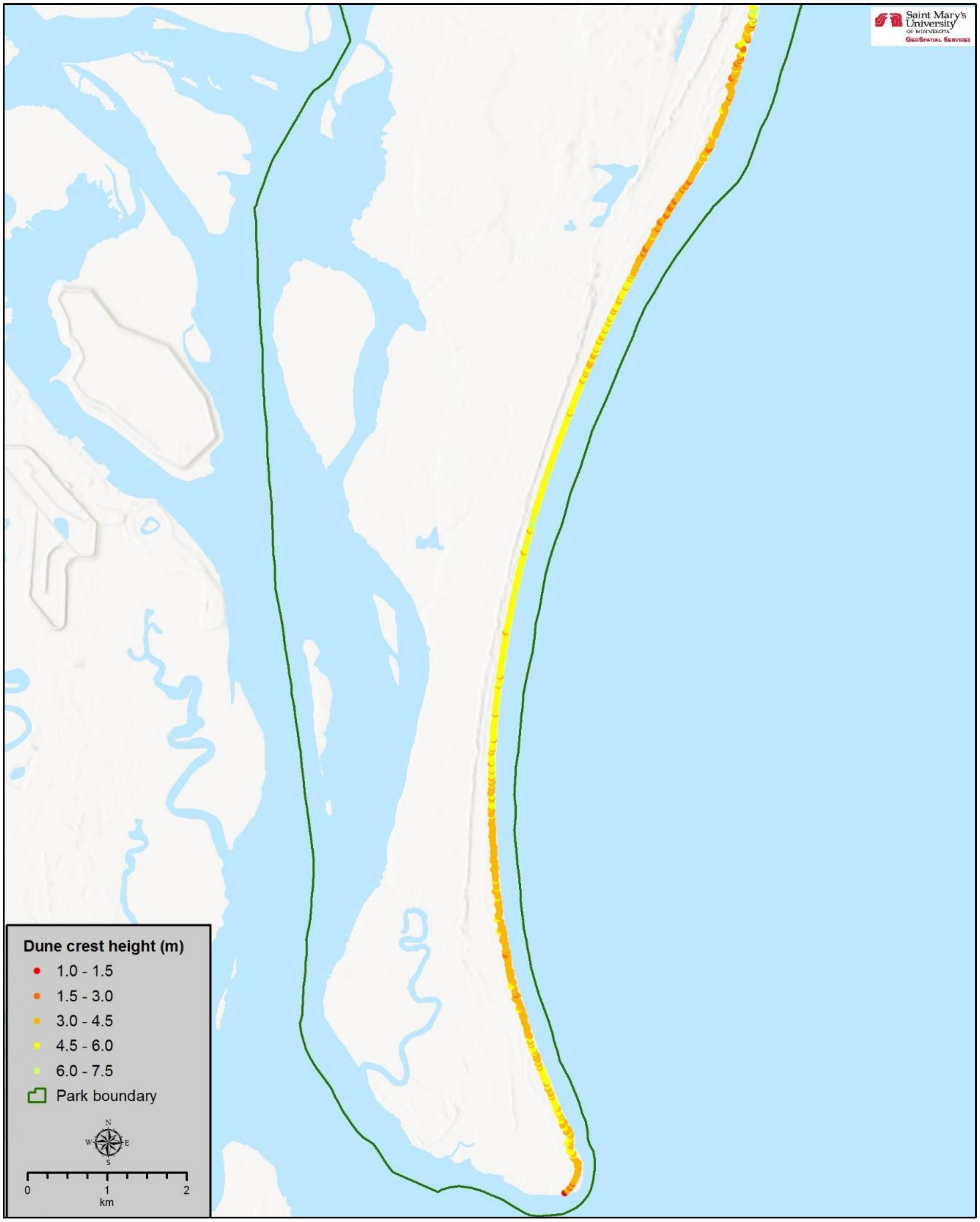

Figure L.2. Dune crest height (elevation) in the southern portion of CUIS (Stockdon et al. 2007).

Appendix M. Comparative photos from shoreline monitoring photopoints at CUIS (NPS photos).

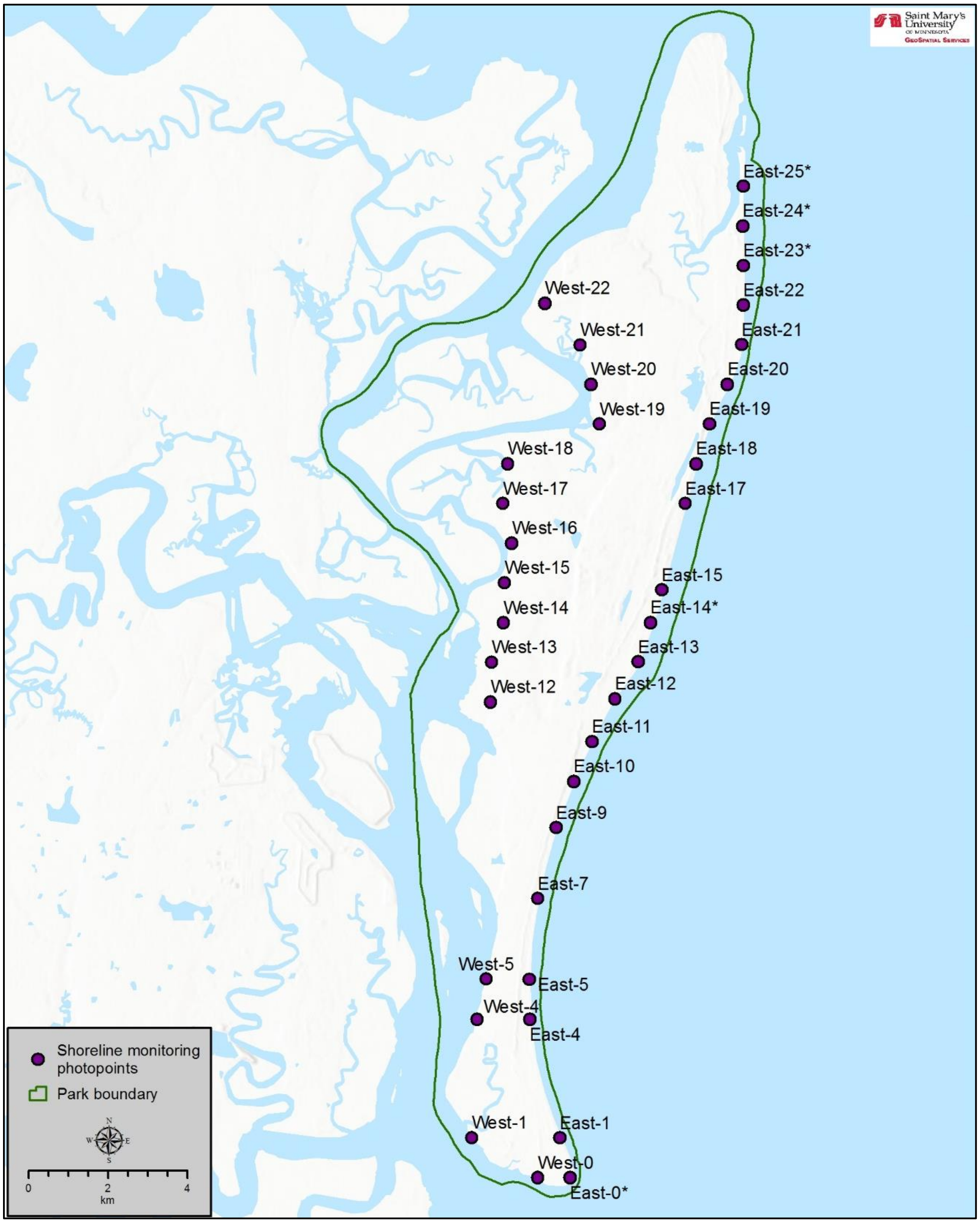

Figure M.1. Locations of CUIS shoreline monitoring photopoints (NPS 2013c). Asterisks (*) indicate markers that were replaced or newly established in 2012. West shoreline photopoints 6-8 and 11 would fall on private property and are therefore not marked or monitored. The markers for some photopoints were missing in 2012 and were not replaced.

Figure M.2. Photos from east side marker #14 (mid-island) in 2012 (top) and 2016 (bottom) showing erosion and loss of vegetation at the front edge of the dunefield (NPS photos).

Figure M.3. Photos from east side marker #1 (south end) in 2012 (top) and 2016 (bottom) showing accretion and advance of dune vegetation (NPS photos). However, advance/accretion was reversed by fall storms in 2016-2017 (see Figure 94).

Figure M.4. Photos from west side marker #12 (mid-island) in 2000 (top) and 2012 (bottom) showing changes in the salt marsh (NPS photos).

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