

Monitoring of Tree Island Condition in the Southern Everglades

(Cooperative Agreement #: W912HZ-14-2-0022) Year-5 Report (2014-2019)



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Cover photo: Satinleaf island after Hurricane Irma (Courtesy: Jed Redwine, ENP)

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General Background

Tree islands, an integral component of the Everglades, are abundant in both the marl prairie and ridge and slough landscapes. They are also likely to be sensitive to large-scale restoration actions associated with the Comprehensive Everglades Restoration Plan (CERP) authorized by the Water Resources Development Act (WRDA) of 2000 to restore the south Florida ecosystem. More specifically, changes in hydrologic regimes associated with restoration projects, including the construction of two Tamiami Bridges and Central Everglades Planning Project (CEPP) components (USACE, 2014), will probably alter the impact of existing local and landscape-level stressors, such as hydrology, invasive exotics, windstorms, and fire (Wetzel et al., 2017).

While such alterations in the impact of these stressors influence the spatial distribution pattern of tree islands within the landscape, the hydrologic alterations also affect the internal water economy of islands, which in turn influences plant community structure and function by affecting species composition, tree regeneration and growth. In the Greater Everglades Conceptual Ecological Model (CEM), researchers have identified plant community composition and structure of tree islands as one of several ecological attributes that are affected by changes in hydrologic characteristics, fire regimes and other stressors. When these stresses become severe, the forest's structure and function can be in peril, leading to tree island loss. For restoration purposes, it is important to project when natural and/or management-induced hydrologic conditions and other stressors will surpass the ability of islands to remain ecologically functional. Several examples of such adverse episodes have been reported. For instance, Everglades researchers showed that loss of tree islands in the Water Conservation Areas was primarily caused by management-related highwater levels due to compartmentalization of the system after 1960 (Patterson & Finck, 1999; Brandt et al., 2000). Likewise, one analysis of multi-year historical aerial photography suggested that a decline in the areal extent of tree islands also occurred within Everglades National Park (ENP) between 1952 and 2004 (Sklar et al., 2013). Though the reasons for this decline in ENP islands have not yet been fully explored, one possibility is that it reflects the effects of alterations in the Everglades' hydrologic regime either directly, or through their impact on other stressors such as fire and windstorms. Thus, for the RECOVER monitoring program, a strategy for tree island work that focuses on effects at both local and landscape-scale effects is recommended.

To better understand inter-annual variability as well as long-term trends and mechanisms that drive them, it is essential to delineate patterns of community composition and configuration at high spatial precision that allow for detection of short-term fluctuations and to differentiate them from persistent long-term change. An approach that concentrates most effort on linking intensive ground surveys with extensive community patterns derived from satellite data and aerial photography is likely to help in reaching a more nuanced understanding of past change in tree island structure, as well as in projecting responses to future changes in water level.

To strengthen our ability to assess the "performance" of tree island ecosystems and to predict how hydrologic alterations translate into ecosystem responses, an improved understanding of plant community structure and function, and their interactive responses to disturbances such as fires and hurricanes is important. Built on a baseline study of vegetation structure and composition and associated biological processes over three years (1999-2002) on three tree islands in Shark River Slough (Ross & Jones, 2004), a broader study was initiated in 2005 and has been continued through today. While the initial (1992-2003) tree island work was supported by the National Park Service (NPS) through the Department of Interior's Critical Ecosystems Study Initiative (CESI), for four years (2005-2009) the project was funded alternatively by the US Army Corps of Engineers (USACE) and South Florida Water Management District (SFWMD), directly or indirectly through ENP. Since 2009, the study has been funded by the US Army Corps of Engineers (USACE) through its contracting office US Army Engineers Research and Development Center (ERDC). Until Fall of 2014, the study was led by Dr. Michael Ross. Thereafter, the study has been led by Dr. Jay Sah, while Dr. Michael Ross and Dr. Daniel Gann are actively involved as the Co-PIs in the study. The comprehensive results of works accomplished through 2014 are described in Ruiz et al. (2011, 2013a) and Sah et al. (2012, 2015). This document describes the work primarily accomplished between 2014 and 2019 (Cooperative Agreement # W912HZ-14-2-0022).

The major goal of ongoing monitoring of southern tree islands is to assess structural and compositional responses of tree island vegetation to natural and management-induced hydrologic change, alterations in relative proportion of forest communities on the islands, and the expansion or contraction of islands within their surrounding marshes. This research addresses the relevant RECOVER performance measures (PM), (1) GE-15: 'Ridge and Slough Sustainability', and (2) 'Total System Performance Measure (RECOVER, 2011). The working hypothesis of the study is expressed as 'the loss of elongated patterns of ridges, sloughs, and tree islands in the direction of water flow in the ridge and slough landscape of the Everglades is attributed to disrupted sheet flow and related changes in water depth' identified in the hypothesis cluster of the sub-section 3.3.7.1 of the 2009 CERP Monitoring and Assessment Plan (RECOVER, 2009). Since 2012, the study has linked field sampling (in a network of permanent plots and along transects) and remote sensing activities to establish a more complete, spatially explicit inventory of vegetation patterns within individual tree islands, one that can be used to monitor vegetation change in a consistent and repeatable way.

The specific objectives of our ongoing research are:

- 1) To characterize the relationship between hydrologic regime and tree demography in the hardwood hammock portions of tree islands.
- 2) To assess change in the structure and composition of both swamp forest and hardwood hammock.
- 3) To develop a tree island vegetation classification scheme based on canopy and understory vegetation types.

- 4) To develop and validate methods that consistently differentiate tree island vegetation assemblages, and that delineate their boundaries from spectral signatures of biseasonal satellite data and aerial photography.
- 5) To detect changes and trends in the relative proportion of different tree island vegetation types
- 6) To investigate the correlation of vegetation change to changes in hydrological regime.

The document is organized in four sections. Section 1 focuses on tree layer dynamics in hardwood hammock portions of eight tree islands of a 16-island network in ENP. The period of study (2014-2019) includes Hurricane Irma (2017) and the recovery thereafter. Section 2 summarizes vegetation changes in bayhead forest and bayhead swamp portions of four Shark River Slough tree islands. Section 3 examines the understory vegetation composition, and the relative influence of canopy cover and hydrology on understory vegetation along transects representing topographic gradients in 12 tree islands. Section 4 summarizes the development and evaluation of a methodology to detect Everglades tree island plant communities (hardwood hammock, bayhead forest , bayhead swamp, and mixed communities of the surrounding marshes) and their changes over time. The methodology was developed in three phases while mapping communities across eight islands, five in ENP, and three in WCA3B using bi-seasonal WorldView2 (WV2) multi-spectral satellite data augmented with Light Detection and Ranging (LiDAR) data available for some of those islands.

1. Effects of hydrology and hurricane on vegetation dynamics in tree island hardwood hammocks of the southern Everglades

1.1 Introduction

Tree islands are a prominent feature in both the marl prairies (MP) and ridge and slough (R&S) landscapes of the Everglades. In the R&S landscape, flow-induced teardrop-shaped tree islands often include different plant communities - tropical hardwood hammock, bayhead forest (hereafter called 'bayhead') and bayhead swamp - arranged along topographic, hydrologic and soil nutrient gradients (Armentano et al., 2002; Sah, 2004; Espinar et al., 2011; Sah et al., 2018). Despite the small areas they cover, the hardwood hammock-dominated heads are of great ecological significance, as both biodiversity and phosphorus 'hotspots' within the homogeneous oligotrophic landscape (Ross & Jones, 2004; Wetzel et al., 2008). While hydrology plays an important role in the development and maintenance of the ridge-slough-tree island patterned landscape, the associated plant communities also influence the hydrodynamics and spatial distribution of soil resources, which in turn affect ecological processes on tree islands (Ross & Jones, 2004; Ross et al., 2006; Givnish et al., 2008; Hanan & Ross, 2010; Espinar et al., 2011; Ross & Sah, 2011; Sullivan et al., 2011, 2013; Wetzel et al., 2005, 2017; Sah et al., 2018;) (Figure 1.1).



Figure 1.1 A Conceptual model: vegetation dynamics in tree islands and surrounding marsh.

Beyond the physiographic template, the species assemblages and areal extent of different plant communities on the R&S tree islands, and between tree islands and marsh fluctuate significantly over time depending on the climate and anthropogenically induced changes in flooding and fire regimes (Stone & Chimura, 2004; Bernhardt & Willard, 2009). In R&S tree islands, the swamp forests and tails are usually the areas that respond most noticeably to hydrologic changes, whereas on an inter-annual scale the response of the tree island heads to

windstorms overshadows any detectable hydrologic response (Ruiz et al., 2013a, 2011; Sah et al., 2018).

In the hardwood hammocks, which are rarely flooded and often have a mean annual water table below 40 cm, tree species composition is probably more the legacy of long-term interaction between water levels and other physical processes, including recurrent tropical storms. In these islands, plant communities recover within a few years after a hurricane. However, vegetation recovery also depends on the post-hurricane environmental conditions. On September 10, 2017, Hurricane Irma made landfall in the Florida Keys as a Category 4 hurricane, and then struck the southwest coast of Florida as a Category 3 hurricane (Cangialosi et al., 2018). However, its impact was felt in most of south Florida. A preliminary analysis of 2017 (WY 2017/18) and 2018 (WY 2018/19) tree data revealed a severe damage to trees in eight tree islands for which pre-Irma data were available. Several questions presented themselves: (a) How widespread and severe were the effects of Hurricane Irma on Everglades tree islands, based on the eight tree islands we studied? (b) Which species were most vulnerable to damage? and (c) Was the tree damage pattern related to tree size, e.g. DBH and height? Post-Irma assessment of tree damage in these hardwood hammocks served as baseline data to follow the vegetation recovery from the damage. This assessment of recovery from hurricane damage will reveal the islands' resilience, i.e. their capacity to recover since the last disturbance.

This section of the report includes the results of the continued monitoring of tree layer vegetation structure and composition within hardwood hammocks on a subset of four tree islands within a 16-island network established in ENP for long-term monitoring and assessment (Shamblin et al., 2008; Ruiz et al., 2011). It also includes the post-Irma assessment of vegetation on those four islands and an additional four tree islands for which pre-Irma vegetation composition data were available.

1.2 Methods

1.2.1 Study Area

The eight tree islands for which pre-hurricane data were available represent a subset of those studied between 2005 and 2010. These islands included one prairie island (Grossman Hammock) along the eastern border of the Park, four islands (Black Hammock, Gumbo Limbo, Satinleaf, and Vulture Hammock) in Shark River Slough, and three (Chekika, Irongrape and SS-81) in Northeast Shark Slough (Figure 1.2). SS-81 and Chekika are located immediately downstream from the 1-mile (eastern) and 2.7-mile (western) bridges on Tamiami Trail, respectively, and they are likely to exhibit the impacts of increased flow from the WCAs into the Park as time goes on.



Figure 1.2 Location map of tree islands that have permanent plots in hardwood hammocks. The plots have been sampled during various periods between 2001 and 2019. Vegetation in the plots on eight tree islands were re-sampled in WY 2017/2018 and 2018/2019 after hurricane Irma.

1.2.2 Data Collection

1.2.2.1 Vegetation sampling

The vegetation sampling in the hardwood hammock plots was organized in a nested design that accounted for all the major forest strata (trees & saplings, shrubs, seedlings, and herbaceous macrophytes). The sampling protocol followed the methodology described by Sah and Ruiz et al. (2011). Between WY 2011/12 and 2016/17, trees and saplings were sampled in the hardwood hammock plots on four islands: Black Hammock (BL), Gumbo Limbo (GL), Satinleaf (SL) and SS-81 (Heartleaf: HL). However, in the two years following Hurricane Irma (WY 2017/18 and 2018/19), tree layer sampling was conducted in those four islands as well as in another four islands (Chekika Island (CH), Grossman Hammock (GR), Irongrape (IG), and Vulture Hammock (VH)). The size of monitoring plots on these eight islands ranged between 300 m² in SS-81 to 625 m² in Gumbo Limbo and Satinleaf (Table 1.1).

Tree Island	Easting NAD83 (UTM_Z17N)	Northing NAD83 (UTM_Z17N)	Plot Size (m2)	Mean (± 1 S.D.) Plot Elevation (m NAVD 88)	Minimum Plot Elevation (m NAVD 88)	Maximum Plot Elevation (m NAVD 88)	Island height (cm)**
Black Hammock	531295	2832630	400	2.330 ± 0.166	1.988	2.584	99.1
Chekika	534372	2847485	400	2.624 ± 0.035	2.545	2.712	113.8
Grossman	541819	2833205	400	2.042 ± 0.144	1.386	2.238	44.5
Gumbo Limbo	525999	2834793	625	2.059 ± 0.071	1.916	2.24	87.8
Irongrape	533651	2836523	400	2.240 ± 0.050	2.092	2.345	92.0
Satinleaf	524499	2838019	625	2.221 ± 0.076	2.082	2.368	89.3
Heartleaf (SS-81)	547639	2848113	300	2.168 ± 0.304	1.592	2.649	80.0
Vulture	528918	2841667	400	2.663 ± 0.191	2.338	2.977	127.7

Table 1.1 Location and topographic data (mean, minimum, and maximum) of hardwood hammock plots on eight tree islands sampled in 2018/19.

Each plot is gridded into $5\times5m$ cells, whose corners and midpoint are marked by 30 cm long flags and $\frac{1}{2}$ " PVC stakes affixed to the ground, respectively. When the plots were first established on these islands, the plot and cells were set up using compass, measuring tape, sighting pole(s), and right-angle prism. In these plots, all trees (≥ 5 cm) are tagged with numbered aluminum tags, and the location of each tagged tree is recorded to the nearest 0.1m using the SW corner of the plot as a reference (0, 0). Furthermore, if a tree has multiple stems ≥ 5 cm diameter (cm) at breast height (DBH), each stem is tagged with a unique ID that allows it to be cross-referenced back to its 'parent' stem. Status (live and dead) and DBH of each individual tree was first recorded when plots were established (in Black Hammock, Gumbo Limbo and Satinleaf in 2001, and in SS-81, Chekika, Grossman, Irongrape and Vulture Hammock in 2007).

During both 2017/18 and 2018/19 samplings, the status (live and dead) of tagged trees and the presence of any tree that had grown into the >5cm DBH class (hereafter called 'ingrowth') since the previous survey were recorded. In-growth trees were identified to species, tagged, and its DBH was measured. The density and species of all tree saplings (stems 1-5 cm in DBH) within each 5 x 5 m cell were also recorded, and assigned to one of two DBH size classes: 1-3 cm or 3-5 cm. The density of woody seedlings (stems < 1 m) and shrubs (stems > 1 m and < 1 cm DBH) was estimated in nested circular plots of 1.0 m² and 3.14 m², respectively, centered on the midpoint of each cell. Seedlings present within the 1 m² (0.57 m radius) plots were counted and identified to species and assigned to one of three height categories (1-30, 30-60, & 60-100 cm). Shrubs rooted within the 3.14 m² (1 m radius) plots were counted and identified to species. The total cover of each shrub species was also estimated using a modified Braun-Blanquet scale based on the following six cover categories: Cat 1: <1%; 2: 1-4%; 3: 4-16%; 4: 16-32%; 5: 32-66%; & 6: >66% (Sah, 2004). Within the 1 m radius plot, the total cover of all herbaceous macrophytes, which includes seedlings, shrubs (< 1 m tall), epiphytes, vines and lianas, was also estimated by species, using the same cover scale.

Canopy closure was estimated by taking two densiometer readings, one facing north and one facing south, at the midpoint of each cell (Lemmon, 1956). The densiometer estimates of

forest canopy closure were supplemented with hemispherical canopy photographs. At the midpoint of each cell, a hemispherical photo of the canopy directly overhead was taken using a Nikon 950 digital camera with a Nikon FC-E8 fisheye lens adapter (NIKON Inc., Melville, NY), placed and leveled 1.5 m above the ground. Leaf area index (LAI) was calculated by processing the hemispherical canopy photos with the Gap Light Analyzer program, GLA 2.0 (Frazer et al., 1999). For each hemispherical image, we calculated the percent canopy openness and the 4-ring LAI – the ratio of the total one-sided leaf area to the projected ground area (Parker, 1995).

1.2.2.2 Hurricane damage

The effects of Hurricane Irma on hardwood hammocks of the tree islands mentioned above were assessed in 2017/18. Vegetation structure and composition of these islands were available for varying periods prior to Hurricane Irma. While vegetation data was available for all eight islands for water years (WY) 2006/07 to 2011/12, four islands (Black Hammock, Gumbo Limbo, Satinleaf and SS-81) had tree census data until 2016/2017, just one year before the hurricane.

Hurricane damage was recorded by assigning each tree to five categories: a) uprooted, b) broken main stem, c) broken branch, d) defoliation, and e) not damaged. If the tree was uprooted, its DBH and the direction of tip-up treefall were recorded. For trees with broken trunks, the height of the break and the DBH of the broken trunk or the largest branch of tree were recorded. In addition, the diameter of all woody debris (≥ 2.5 cm diameter) was measured and recorded.

1.2.2.3 Hydrology

For hardwood hammock plots in each of the study islands, ground elevation data were available from detailed topographic survey conducted using auto-level from either a 1st order vertical control monument (benchmark) or from a reference benchmark established in the marsh, followed by an estimate of benchmark elevation by differential GPS; in some cases, benchmark elevation was calculated by relating water depth at the benchmark to the estimate of water surface elevation at that location and time from EDEN (Everglades Depth Estimation Network) (Ruiz et al., 2011). In conjunction with the daily EDEN water surface elevation data (http://sofia.usgs.gov/eden), elevation of the ground surface within the plots was then calculated.

1.2.3 Data Analysis

1.2.3.1 Hydrologic conditions

EDEN estimates were also used to transform field measurements of water depth to projections of hydrologic conditions over time at each sampling site. EDEN acquires water level data from a network of stage recorders throughout the Everglades and produces interpolated daily water surface estimates (Palaseanu & Pearlstine, 2008). Ground elevation for each plot was estimated by subtracting the mean water depth from the EDEN water surface elevation for the marsh adjacent to each tree island on the day it was sampled. Mean annual water depth (hereafter

called relative water level (RWL), and discontinuous hydroperiod (the number of days in a year when water is above the ground surface) were then estimated based on ground elevation and the time series data of water surface elevation extracted from EDEN database. Previous studies have found that prairie and marsh vegetation composition are well predicted by the previous 3-5 years of hydrologic conditions (Armentano et al., 2006; Ross et al., 2006; Zweig & Kitchens, 2009), whereas tree island vegetation was found strongly correlated with 7-year average hydroperiod and water depth (Sah, 2004; Espinar et al., 2011; Sah et al., 2018). Thus, in this study, we averaged hydroperiod and mean annual RWL for 4-7 water years (May 1st – April 30th) prior to each sampling event to examine the relationships between hydrologic parameters and change in vegetation characteristics.

1.2.3.2 Hurricane damage

We assessed the effect of tree size (DBH) and hydrologic variables on the likelihood of tree damage using simple logistic regression. The response variables were the damage categories described earlier. P-values for regression coefficients were determined using Z-tests.

1.2.3.3 Tree-layer vegetation dynamics

Tree census data were summarized by calculating annual mean tree mortality and ingrowth, two important indicators of woody vegetation dynamics. In addition, tree density and basal area for each species were calculated, and summed to produce totals for each island.

Differential mortality and/or in-growth among species over time can result in changes in species composition. These changes were analyzed using non-metric multidimensional scaling (NMDS) ordination. Species abundance data used in the ordination was the species importance value (IV). Tree density and basal area for each species were summed for each plot, relativized as a proportion of the plot total, and used to calculate Importance Value (IV) of species using the following equation: $IV = 100 \cdot ((Rd + Rba) / 2)$, where Rd is the species relative density and Rba is the species relative basal area. Importance value (IV) data of each species were standardized to species maxima and the Bray-Curtis (B-C) dissimilarity index was used as a measure of dissimilarity in the ordination.

1.2.3.4 Herb/Shrub layer vegetation dynamics

Changes in herb and shrub species composition were also examined with non-metric multidimensional scaling (NMDS) ordination. In this case, abundance data were species cover estimates. Cover data of each species were standardized to site total, and the Bray-Curtis (B-C) dissimilarity index was again used as the measure of dissimilarity.

1.3 Results

1.3.1 Hydrologic conditions

Hydrologic condition in tree island hammocks varies depending on the location of tree islands within the R&S landscape and tree island height above the surrounding marshes. On the eight tree islands, the annual mean (\pm SD) relative water level (RWL) over thirteen years, (2006/07 to 2018/19: the period for which tree data were available for all eight islands) ranged between -94.9 \pm 8.95 cm in Chekika and -60.6 \pm 8.75 cm in Gumbo Limbo (Table 1.2). The mean RWL in Chekika, Vulture, and Black Hammock was much (17-34 cm) lower than in other tree islands. However, the mean RWL was not uniform within the plot on each hammock. Within plot variation (Coefficient of variation, CV) in water level was the highest in SS-81 (CV = 35.0%), and the lowest in Chekika (CV = 2.7%).

	Relative water level (RWL) (cm)					
Tree Island	Annual Mean (± S.D.)	Annual Range (Min-Max)	Within plot variation (CV %)			
Black Hammock	-86.3 ± 8.3	-100.7 to -73.6	16.8			
Chekika	-94.3 ± 8.9	-113.9 to -80.7	2.7			
Grossman	-69.9 ± 12.8	-91.1 to -54.0	13.0			
Gumbo Limbo	-60.6 ± 8.8	-74.5 to -45.1	10.7			
Irongrape	-67.5 ± 8.9	-82.1 to -53.4	6.0			
Satinleaf	-65.0 ± 8.8	-78.0 to -48.2	11.2			
Heartleaf (SS-81)	-63.4 ± 10.3	-84.6 to -49.0	35.0			
Vulture	-91.2 ± 10.0	-107.8 to -73.3	18.9			

Table 1.2 Annual mean $(\pm SD)$ relative water level (RWL) averaged over 13 years (WY 2006/07 to 2018/19) in the hardwood hammock plots on eight tree islands.

Over the last thirteen years, the RWL varied annually. Between WY 2006/07 and WY 2011/12, the annual mean RWL, was 1 to 25 cm lower than 28-year (1991-2019) average (Figure 1.3). In contrast, the RWL in the most recent 7-year period (from WY 2012/13 to 2018/19) was 2 to 16 cm above the 28-year average, except in WY 2014/15 and 2015/16, when mean RWL was 2 to 11 cm below the long-term average. For most years, none of the hammock plots on these islands were inundated during these periods, except in WY 2017/18, when the highest water level on these islands was observed. In that year, characterized by the extremely high water levels in Hurricane Irma's aftermath, plots on 5 of 8 islands were partly inundated for varying periods. One sub-plot in Black Hammock was inundated for only one day, whereas a sub-plot in SS-81 was inundated up to 167 days. In fact, the aforementioned sub-plot on SS-81 in NESRS was inundated for 5 to 167 days in 9 of 13 years between WY 2006/07 and 2018/19.



Figure 1.3 Twenty-eight year (Water Year 1991/92-2018/19) average and annual mean (±SE) relative water level (RWL) in the hardwood hammock plots on eight tree islands, (a) Black Hammock (BH), (b) Chekika (CH), (c) Gumbo Limbo (GL), (d) Grossman (GR), (e) Heartleaf/SS-81 (HL), (f) Irongrape (IG), (g) Satinleaf (SL), and (h) Vulture (VH).

In general, the annual mean water level in these hammocks followed the regular dry (low) and wet season (high) pattern. However, in some years, water level in the hammock plots was much higher in the dry season than the wet season due to either an anomaly in weather pattern, management-induced changes in hydrologic regime, or both. For instance, over the last 13 years, the most remarkable discrepancy between dry and wet season pattern was in 2010/11 and 2015/16 when the water level was higher, i.e. much closer to the ground surface in the dry season than in the wet season (Figure 1.4). This was caused by unusual high winter rainfall followed by the very dry wet season as well as the increased water deliveries into the Park. Moreover, in those years, the discrepancies in dry and wet season water level were more distinct in NESRS and Prairie islands (Figure 1.4 b, d, g) than in SRS islands.



Figure 1.4 Seasonal mean (±SE) relative water level (RWL) in the hardwood hammock plots on eight tree islands, (a) Black Hammock, (b) Chekika, (c) Gumbo Limbo, (d) Grossman, (e) Irongrape, f) Satinleaf, g) SS-81, and h) Vulture.

In South Florida, including the Everglades, winter rainfall is strongly linked to El Niño events. In the 2016 dry season, a strong 2015-2016 El Niño dumped more than 30 cm of rainfall resulting in high water conditions throughout South Florida. In fact, both December 2015 and January 2016 produced the highest rainfall for those months in several decades (Abtew & Ciuca, 2017). Much higher than normal dry season rainfall in the Everglades basin, especially the Water Conservation Areas, resulted in extreme water level in WCA-3A, prompting emergency operations to move water to the south, i.e. into Everglades National Park (ENP). During the 90-day emergency operation period (February 12-May 11, 2016), the Park received one-half million acre-ft of water, 60% of which flowed through SRS and 40% through NESRS (Abtew & Ciuca, 2017). This emergency operation period also overlapped with the Modified Water Deliveries (MWD) Increment 1 Field Test period. Increased water delivery to the Park resulting from both MWD Increment 1 Field test and unusual emergency operation during the dry season contributed to the spatial and temporal differences in water conditions within the Everglades tree island hammocks. In fact, mean annual relative water level in these islands in SRS and NESRS is hardly in tandem with the total annual rainfall in that region. For this analysis, stage recorder P33

located in SRS (Figure 1.2), for which long-term rain data are available on DBHYDRO (www.sfwmd.gov/science-data/dbhydro), was used. Between 2006/07 and 2018/19, a correlation between annual total rainfall at P33 and mean RWL on each of these eight islands was non-significant (Figure 1.5; p-value>0.05), because the hydrologic conditions in these islands depend in large part on the water delivery from the adjacent WCA 3A and 3B.



Figure 1.5 Relationship between annual total rainfall at P33 stage recorder and mean annual relative water level (RWL) in two groups of tree islands: (A) four tree islands sampled annually, and (B) four tree islands first sampled between 2006/07 and 2010/11, and then again in 2017/18 and 2018/19.

1.3.2 Hurricane damage

In September 2017, Hurricane Irma, a category 4 hurricane, hit the southwest coast of Florida. However, its impact was felt throughout south Florida. An analysis of 2017 (WY 2017/18) and 2018 (WY 2018/19) tree data revealed varying degrees of damage to trees on eight tree islands for which pre-Irma data were available (Figure 1.6).



Figure 1.6 Photos of three tree islands (Black Hammock, Gumbo Limbo and Satinleaf) showing the damage from Hurricane Irma. (Photo credit – Jed Redwine, NPS)

On the eight islands that were sampled, 68.9 % of trees sustained one or another form of damage from Hurricane Irma. The percentage of trees damaged by the hurricane was higher (>75%) on Chekika, Gumbo Limbo, Irongrape and Vulture than on the other four islands (Figure 1.7). The form of wind damage varied among tree islands. Black Hammock had the highest percent (11.8%) of uprooted trees, whereas damage on other islands consisted mostly of broken trunks and branches (Table 1.3).

On the studied islands, not all species were equally susceptible to hurricane damage (Table 1.4). For instance, all individuals (100%) of the less abundant species (1-3 trees), spicewood, (*Calyptranthes pallens*), willow-bustic (*Sideroxylon salicifolium*), and paradise tree (*Simarouba glauca*) had broken branches. Among the five most abundant species, white stopper

(*Eugenia axillaris*) and mastic (*Sideroxylon foetidissimum*) had 85% trees that sustained some kind of damage (Table 1.4), while 50% or less of gumbo limbo, (*Bursera simaruba*), sugarberry (*Celtis laevigata*) and satinleaf (*Chrysophyllum oliviforme*) trees sustained damage.



Figure 1.7 Tree damage percent on eight tree islands sampled 1-6 years before and 1-3 months after Hurricane Irma.

	Percent of damaged trees					
Tree Island	Uprooted	Broken Trunk	Broken Branch	Defoliated		
Black Hammock	11.8	11.8	20.0	24.5		
Chekika	7.2	11.6	37.7	24.6		
Grossman	5.4	15.5	43.4	3.1		
Gumbo Limbo	9.4	31.3	34.4	7.8		
Irongrape	5.6	33.3	38.9	0.0		
Satinleaf	1.7	11.9	24.9	22.0		
SS-81	3.3	6.7	21.7	0.0		
Vulture	0.0	20.0	37.3	24.0		

Table 1.3 Types of tree damage by Hurricane Irma on eight tree islands sampled two to four months after the hurricane in WY 2017/18.

Table 1.4 Plant species that sustained different types of tree damage by Hurricane Irma on eight tree islands sampled two to four months after the hurricane in WY 2017/18.

	Number of Total Trees	Percent of damaged trees (%)				
Tree species		Uprooted	Broken Trunk	Broken Branch	Defoliated	
Annona glabra	4	0.0	0.0	0.0	0.0	
Ardisia escallonioides	3	0.0	0.0	66.7	0.0	
Bursera simaruba	149	5.4	14.1	28.9	2.0	
Calyptranthes pallens	1	0.0	0.0	100.0	0.0	
Celtis laevigata	106	2.8	7.5	32.1	9.4	
Chrysobalanus icaco	29	3.4	17.2	17.2	20.7	
Chrysophyllum oliviforme	42	0.0	7.1	42.9	0.0	
Coccoloba diversifolia	50	6.0	10.0	50.0	4.0	
Eugenia axillaris	204	8.3	21.1	18.1	39.7	
Ficus aurea	11	0.0	18.2	36.4	0.0	
Myrsine floridana	3	0.0	66.7	0.0	0.0	
Nectandra coriacea	24	16.7	12.5	45.8	4.2	
Sambucus canadensis	1	0.0	0.0	0.0	0.0	
Schinus terebinthifolius	14	7.1	0.0	21.4	0.0	
Sideroxylon foetidissimum	55	0.0	25.5	56.4	12.7	
Sideroxylon salicifolium	1	0.0	0.0	100.0	0.0	
Simarouba glauca	3	0.0	0.0	100.0	0.0	
Zanthoxylum fagara	2	0.0	50.0	0.0	0.0	

The logistic regression model revealed that tree size (DBH) was a significant predictor of the tree damage, particularly broken branches and defoliation, caused by wind. Since the sampling was done 2-4 after the hurricane, some damages, especially defoliation, occurred after the hurricane, might have also been included in the analysis. Larger trees were likely to have

more broken branches, whereas percent of defoliation was higher for smaller trees (Figure 1.8). However, tree size did not have significant effects on uprooting or the breakage of the main trunk.



Figure 1.8 Probability of hurricane damage based on tree DBH.

1.3.3 Tree mortality and in-growth

Tree layer vegetation dynamics are a function of tree mortality and in-growth, two important indicators of woody vegetation dynamics on tree islands. During 2007-2010 (WY 2007/08 to 2010/11), when the hardwood hammock on all 16 islands were studied, mean annual tree mortality on those islands was 3.6%, and both NESRS and R&S islands had higher mortality than MP islands (Figure 1.9). During those years, mean tree in-growth was significantly higher (paired t-test, P <0.001) than mean tree mortality. On average, the mean tree ingrowth was 104 trees ha⁻¹ year⁻¹ whereas tree mortality was 44 trees ha⁻¹ year⁻¹. In-growth on some islands was higher also because of recovery from Hurricane Wilma in 2005. However, the ingrowth on Irongrape was exceptionally high (> 200%) in the post-Irma year, mostly due to papaya (*Carica papaya*) - an ephemeral semi-woody pioneer that recruits profusely from the seedbank, but will likely disappear from the canopy within a decade or less.

Between 2011 (WY 2011/12) and 2016 (WY 2016/17), when hammocks on four islands (Black Hammock, Gumbo Limbo, Satinleaf, and SS-81) were monitored annually, both the mean tree ingrowth and mortality showed little variation, except on SRS islands in 2014. In general, annual mean mortality was slightly higher than in-growth. On these four islands, the mean

mortality rate was almost the same during both periods, before 2011 (3.41%) and between 2011 and 2016 (3.22%), whereas the mean in-growth rate dropped from 7.0% year⁻¹ to 2.72% year⁻¹.



Figure 1.9 Annual mean (±) tree in-growth (A) and mortality (B) on the tree islands monitored in Shark River Slough (SRS) and Northeast Shark River Slough (NESRS) within the Everglades National Park between WY 2007/08 and 2018/19. The number of tree islands studied varied among years. Between 2011/12 and 2016/17, hardwood hammocks were studied on only four islands (Black Hammock, Gumbo Limbo, Satinleaf and SS-81).

On some of the studied tree islands, tree mortality drastically increased in 2017/18, mostly caused by Hurricane Irma. After the hurricane in WY 2017/18, we sampled vegetation on four additional tree islands (Chekika, Grossman, Irongrape and Vulture), all from the same network of 16 islands within ENP (Ruiz et al., 2011). On five of the eight islands, tree mortality was higher in 2017/18 than in previous years (Figure 1.10). In 2017/18, i.e. within 2-4 months after Hurricane Irma, increased tree mortality was observed in Black Hammock, Grossman, Satinleaf and SS-81. In the year after the hurricane, tree mortality on Black Hammock, Chekika, Grossman, Gumbo Limbo, Irongrape and SS-81 was relatively high. An elevated rate of mortality on these islands one year after a hurricane, including exceptionally high mortality in Irongrape, suggests delayed, hurricane-related mortality.



Figure 1.10 Post-Irma tree damage and annual mean tree mortality (%) on eight tree islands before and after Hurricane Irma. On four islands (Chekika, Grossman, Irongrape and Vulture), pre-Irma tree mortality data were available for only 2010/11 and 2011/12.

In concurrence with the trend in tree mortality and ingrowth on four islands that were monitored in all years, total tree basal area first increased until 2009/10, and then decreased over the next 7 years, between 2011 and 2018 (Figure 1.11). The lowest mean value of BA in WY

2015/6 was because only three tree islands were sampled in that year. Black Hammock, which has higher (38% of total) BA than three islands, was not sampled in 2015/16. On these islands, the total basal areas in post-Irma years was lower than in 2006/07 (one year after Hurricane Wilma).



Figure 1.11 Boxplot showing the trend in tree basal on four tree islands monitored within the Everglades National Park between 2007 (WY 2007/08) and 2018 (WY 2018/19). In WY 2015/16, tree basal area was low, as only three tree islands were sampled.

As reflected by variation in annual mean tree mortality and ingrowth, the short-term trend of tree dynamics observed in the hardwood hammocks is in accord with variation in hydrologic condition, though, the relationship between mean annual RWL and tree mortality on these islands was weak (r = 0.05; Figure 1.12).



Figure 1.12 Scatterplot showing the relationship between relative water level and tree in-growth and mortality on four tree islands monitored within the Everglades National Park between 2007 (WY 2007/08) and 2018 (WY 2018/19).

1.3.4 Tree layer vegetation dynamics

Among the eight islands, tree layer vegetation composition on Grossman and SS-81, located within the MP landscape and NESRS, respectively, was quite different from the SRS tree islands. A nonparametric multi-dimensional scaling (NMDS) ordination, based on tree species' IVI and B-C dissimilarity, revealed that tree species composition has changed slightly in the hammocks of these eight islands (Figure 1.13). The changes were obvious on three tree islands (Black Hammock, Gumbo Limbo and Satinleaf) that have been monitored since 2002. On Black Hammock, the IVI of sugarberry (*Celtis laevigata*) decreased, whereas the IVI of white stopper (*Eugenia axillaris*) increased significantly. Likewise, the IVI values of mastic (*Sideroxylon foetidissimum*) and satinleaf (*Chrysophyllum oliviforme*) in Gumbo Limbo and Satinleaf, respectively, doubled in 15 years (Figure 1.14). On these islands, however, there was not much shift in species composition during WY 2011-2017, until after Hurricane Irma. A minimal shift in species composition in the tree layer was also observed in the hammock of SS-81, which has been monitored annually since WY 2008.



Figure 1.13 Scatterplot of NMDS ordination based on tree species IVI in eight tree island hammocks sampled between Water Year (WY) 2002 and 2019. Fitted vectors are relative water level (RWL), tree island height (TI_Ht), soil nitrogen (TN), soil phosphorus (TP) and total organic carbon (TOC).



Figure 1.14 Importance value index (IVI) of tree species in hardwood hammocks of four tree islands monitored annually. ANNGLA = *Annona glabra*; BURSIM = *Bursera simaruba*; CARPAP = *Carica papaya*; CELLAE = *Celtis laevigata*; CHRICA = *Chrysobalanus icaco*; CHROLI = *Chrysophyllum oliviforme*; COCDIV = *Coccoloba diversifolia*; EUGAXI = *Eugenia axillaris*; FICAUR = *Ficus aurea*; NECCOR = *Nectandra coriacea*: SAMCAN = *Sambucus canadensis*; SCHTER = *Schinus terebinthifolius*; SIDFOE = *Sideroxylon foetidissimum*; SOLERI = *Solanum erianthum*; TREMIC = *Trema micranthum*.

Four islands were sampled annually until 2011/12, and then in 2017/18 and 2018/19. Among these, vegetation composition was relatively stable on Grossman, whereas the other three islands had a significant change in species abundances. One year after Hurricane Irma, the IVI of sugarberry (*Celtis laevigata*) on Chekika was 1/10th of its IVI seven years ago in 2012 (Figure 1.15). Similarly, on Vulture, IVI was 33% less in 2017/18 and 2018/19 than in 2012. In contrast, mastic (*Sideroxylon foetidissimum*) and white stopper (*Eugenia axillaris*) increased on these two islands, respectively. Moreover, a major change was observed in tree layer species on Irongrape where papaya (*Carica papaya*) significantly increased (Figure 1.15 c).



Figure 1.15 Importance value index (IVI) of tree species in hardwood hammocks of four tree islands that were monitored until 2011/12, and then again in 2017/18 and 2018/19. ARDSEC =; BURSIM = Bursera simaruba; CALPAL = *Calyptranthes pallens*; CARPAP = *Carica papaya*; CELLAE = *Celtis laevigata*; CHROLI = *Chrysophyllum oliviforme*; CITAUR = *Citrus aurantifolia*; COCDIV = *Coccoloba diversifolia*; EUGAXI = *Eugenia axillaris*; FICAUR = *Ficus aurea*; MYRFLO = *Myrsine floridana*; NECCOR = *Nectandra coriacea*; SCHTER = *Schinus terebinthifolius*; SIDFOE = *Sideroxylon foetidissimum*; SIDSAL = *Sideroxylon salicifolium*; SIMGLA = *Simarouba glauca*; and ZANFAG = *Zanthoxylum fagara*.

1.3.4.1 Herb and shrub layer vegetation dynamics

As in the tree layer, understory species composition on Grossman and SS-81 tree islands was somewhat different from the vegetation on other islands (Figure 1.16). In the understory of SS islands, the tree seedlings of white stopper (*Eugenia axillaris*) reached high densities (Figure 1.17). In contrast, the understory on Grossman, an island within the MP landscape, had high cover of lancewood (*Nectandra coriacea*) seedlings, whereas on SS-81, few tree seedlings were observed (Figure 1.18).

The environmental vectors representing RWL and soil depth were significantly and marginally correlated with ordination configuration, respectively. However, the relationship

between tree island elevation above marsh (TI_Ht) and canopy cover (Canocov) with ordination configuration was not significant (Table 1.5). Moreover, the results of NMDS ordination revealed that variation in understory species composition over time within an island was more divergent than the tree layer composition on the same island, suggesting that there was little variation in tree species' importance value (IVI) over the sampled years whereas relative abundance of understory species varied greatly within the same period.



Figure 1.16 Scatterplot of NMDS ordination based on herb and shrub species cover on eight tree island hammocks sampled between Water Year (WY) 2002 and 2019. Fitted vectors are relative water level (RWL), tree island height (TI_Ht), canopy cover (Canocov), and soil depth (SoilDep).

Table 1.5 Correlation (r) and statistical significance of fitted environmental vectors with species coverbased 3-dimensional non-metric multiscaling ordination configuration.

Vectors	\mathbb{R}^2	p-value	
Relative water level	0.14	< 0.05	
Soil depth	0.59	<0.1	
Canopy cover	0.60	NS	
Tree island height	0.62	NS	

A change in understory vegetation composition was more obvious in post-hurricane years. For instance, on the five islands (Black Hammock, Gumbo Limbo, Satinleaf, Chekika, and Grossman) which had pre-Wilma data, the difference in understory vegetation composition between pre-hurricane and the 2nd post-hurricane year was relatively high. On these islands total understory cover increased until 2-3 years after Hurricane Wilma, then started to decrease (Figures 1.16, 1.17). The total understory cover was lower just after Hurricane Irma, then on four of these five islands again increased one year after Hurricane Irma, suggesting the influence of changes in tree canopy cover in response to the hurricane. On Grossman, however, the understory layer is much more distinct on Irongrape, which was relatively open 13 years ago. Now, it has a dense canopy due to both increase in tree basal area (from 19.5 m² ha⁻¹ in 2007 to 25.5 m² ha⁻¹ in 2019) as well as an extensive growth of woody climber yellow nicker (*Caesalpinia bonduc*).



Figure 1.17 Percent cover of herb and shrub species in hardwood hammocks of four tree islands. BIDALB= *Bidens alba*; CARPAP = *Carica papaya*; CELLAE = *Celtis laevigata*; EUGAXI = *Eugenia axillaris*; NEPEXA= *Nephrolepis exaltata*; PARFLO= *Parietaria floridana*; PARQUI= *Parthenocissus quinquefolia*; PASSUB= *Passiflora suberosa*; RIVHUM= *Rivina humilis*; SIDFOE = *Sideroxylon foetidissimum*; THEKUN= *Thelypteris kunthii*.



Figure 1.18 Percent cover of herb and shrub species in hardwood hammocks of four tree islands. CAEBON= *Caesalpinia bonduc*; EUGAXI = *Eugenia axillaris*; MYRFLO = *Myrsine floridana*; NECCOR = *Nectandra coriacea*; NEPEXA= *Nephrolepis exaltata*; PSYNER= *Psychotria nervosa*; RIVHUM= *Rivina humilis*; SIDFOE = *Sideroxylon foetidissimum*; THEKUN= *Thelypteris kunthii*; VERVIR= *Verbesina virginica*.

1.4 Discussion

In the Everglades, in the hardwood hammock portions of tree islands that we studied, plant communities respond to changes in hydrologic conditions and the periodic disturbances such as tropical storms. Our results show that periodic fluxes in the hydrologic regime directly and/or interacting with tropical storms affect the tree demography and both the tree layer and understory (herb and shrub) vegetation composition. However, their effects vary among islands depending on the position of islands within the landscape and existing hydrologic conditions.

Hydrologic condition in tree island hammocks varies depending on the location of tree islands within the R&S landscape and tree island height above the surrounding marshes. Based on their locations, islands in the NESRS area have been drier than the western and central SRS islands and might be expected to have the lowest water level below the ground. However, our results showed that while Chekika had the lowest RWL, not all islands in NESRS had a lower RWL than islands in other regions. Possibly, hydrologic conditions on tree islands are not simply the function of regional marsh hydrology, but also the geomorphological characteristics of tree islands, such as the tree island height - the difference in elevation between the surface of the tree

island and the surrounding marsh. In a study of 76 slough and prairie tree islands within the Park and WCA3B, RWL was negatively correlated with tree island height (Ross & Sah, 2011). Among the eight islands studied, Chekika and Vulture had lower RWL than other islands and also had the greatest tree island height (Table 1.1). In addition to its influence on RWL, tree island height may reveal the mode and timing of landform development, thus affecting soil characteristics on the islands. For instance, soil phosphorus on tree islands was positively correlated with island height (D'Odorico et al., 2011; Ross & Sah, 2011).

In the hardwood hammocks of the studied islands, the annual mean RWL remained well below the soil surface, suggesting that limited increase in marsh hydroperiod or water depth in ENP are unlikely to have significant adverse impact on tropical hardwood hammock communities on these islands. As outlined in the Central Everglades Planning Project (CEPP) and Combined Operations Plan (COP), restoration activities are expected to increase water deliveries from WCA 3A to ENP through NESRS (USACE, 2014, 2020). Using modeled water surface elevations for different scenarios described in CEPP, relative water level on tree islands in Western/Central SRS and NESRS was expected to increase by 5-10 cm and 15-20 cm, respectively (Wetzel et al., 2017). Despite model predictions of relatively wet conditions in SRS and NESRS in different restoration scenarios compared to current conditions, vegetation success models using the Everglades Landscape Vegetation Succession (ELVeS) also showed minimal or no change in plant community types on those islands (Wetzel et al., 2017). Water conditions throughout the Everglades, including ENP, depend on the gradual implementation of restoration plan components. Under the preferred plan (ALTQ+) identified in the COP, water delivery into ENP (both northeast and western SRS combined) is projected to increase by 25%, and the delivery into NESRS is projected to increase by approximately 162,000 acre-feet per year on average (USACE, 2020). Similarly, during the process of revisions to the 2005 Interim Goals and Targets for CERP, out of four simulations, the 2032PACR simulation projects the flow into NESRS to increase by a total of 528,000 acre-feet per year (RECOVER, 2020). In fact, water level in NESERS has already been relatively high because of the increased water delivery due to Increment Field Tests that began in October 2015 and continued through 2019 (USACE, 2020). Under these tests, a large volume of water was delivered every year into the Park. In addition, the 1-mile Tamiami Bridge completed in 2013 also has enhanced the water flow into the Park resulting in high water in NESRS. The additional water delivery into ENP would affect the tree islands in SRS. However, after an analysis of possible inundation of 36 tree islands for which elevation data were available, it has been concluded that none of those islands will be inundated more than 10% of modeled time period, a performance indicator used to evaluate the Interim Goals scenarios (RECOVER, 2020; USACE, 2020). However, an incremental upward shift in the RWL could cause a shift in species composition and productivity of plant communities on these islands. Over the period of this study, such a shift in vegetation in response to hydrologic change commonly occurred in wetter communities (bayhead and bayhead swamp)(Sah et al., 2018) (See Section 2).

In general, hydrology is the major driver of differences in species composition among various plant communities arranged along topographic gradients within a tree island (Armentano et al., 2002; Wetzel, 2002; Ross & Jones, 2004; Espinar et al., 2011; Sah et al., 2018). However, in the hardwood hammocks which rarely get flooded, and where the mean annual water table is often 40 cm or more below the ground surface (Table 2; Figure 1.5), tree species composition dynamics is probably more the legacy of long-term interaction between hydrology and other physical processes, including recurrent disturbances. On some of these islands, high tree mortality was observed until 3-4 years after Hurricane Wilma in 2005. In the hammocks of those islands, higher mortality in the 3rd and 4th post-hurricane years after Hurricane Wilma was attributed to the interaction of multiple disturbances, e.g., hurricane and drought (Ruiz et al., 2011). Immediately after Hurricane Irma, we also observed severe damage to the tree layer vegetation on some of the islands for which we had pre-Irma data. Our results suggest that larger trees were likely to have more broken branches, though tree size did not have significant effects on uprooting or the breakage of the main trunk. Moreover, tree mortality after the hurricane was higher than the background mortality i.e. mortality before the hurricane. As a result of tree mortality caused by the hurricane, a shift in vegetation composition was noticed on four islands, Black Hammock, Chekika, Gumbo Limbo and SS-81 (Figure 1.13).

Hardwood hammocks have primarily flood-intolerant species. Water level above or near the ground surface for longer periods, especially during the dry season, adversely impacts the survival and growth of those tree species (Stoffella et al., 2010). During the 2016 (WY 2015/16) dry season, water level on the SRS tree islands was higher than that in wet season, and was very close (<40 cm) to the ground surface for a longer period (Figure 1.5) than other years. which may have affected tree growth and increased mortality in subsequent years. In this study, however, the RWL estimates are based on a flat water table at the same elevation as in the marsh for which the EDEN estimates are derived. Though, the studies have suggested that the water table under the tree island can be drawn down further during the dry season and mounded during the wet season (Sullivan et al. 2011). Thus, the water level may not be flat throughout the year as assumed, but this assumption is useful to have an approximate estimate. In 2017/18 also, the water level throughout the area was high, primarily because of the effects of Hurricane Irma. At least in one island, SS-81, part of the study plot was flooded for more than three months for three years since 2015 when water delivery to NESRS increased. On this island, high tree mortality, especially of a flood-intolerant species, sugarberry (Celtis laevigata) was observed in 2017 and 2018. In the post-hurricane years, another two islands, Black Hammock and Irongrape, also experienced relatively high mortality (Figure 1.10). However, hydrologic conditions in those two islands are not very different from the other SRS tree islands, and they, together with thirty-five other SRS islands, are not expected to experience flooding more than 10% of the year, even in the preferred scenario of Combined Operation Plan (USACE 2020). Nonetheless, since those islands are in the path of water flow through NESRS, the increase in their water levels is expected to be steeper than in other parts of SRS. The response of NESRS islands therefore requires care, and an effective link between science and management.

Beside water level and windstorms, fire is another stressor that affects tree island vegetation, especially when it consumes peat soils and lowers surface elevation (Wetzel et al., 2008). On our studied islands, hardwood hammocks had not been burned between 2001 and 2019. Though, a fire in 2008 had burned up to very close to the hardwood hammock on Black Hammock possibly affecting the boundary between tree island and surrounding marsh (Sah et al. 2018). Thus, the observed dynamics of plant communities in the hardwood hammocks were primarily the result of hydrologic changes and impact of tropical storms.

In summary, community dynamics in the hardwood hammock portions of the study islands, which are rarely flooded and have not burned for decades or more, tree species composition is primarily the legacy of the long-term interaction between hydrology and tropical storms, though short-term responses in tree demography or understory species composition may result from flooding events and/or tropical storms.

2. Hydrologically driven vegetation successional dynamics in bayhead and bayhead swamp portions of tree islands

2.1 Introduction

In the Everglades, where slough-ridge-tree island mosaic forms a topographic gradient, plant communities are sensitive to hydrologic changes. This is especially true in tree islands present within the ridge and slough (R&S) landscape. The R&S tree islands are complex, and have different plant communities arranged along topographic, hydrologic and soil nutrient gradients (Armentano et al., 2002; Ross & Jones, 2004; Espinar et al., 2011; Sah et al., 2018). In these islands, alterations in hydrologic regimes, together with periodic disturbances (hurricane, fire), result in changes in species composition, that ultimately influence the vegetation successional processes. For instance, under prolonged dry conditions in the R&S landscape, expansion of sawgrass in sloughs is usually followed by the establishment and growth of trees in the peat environment that drives successional processes towards dominance of woody plants, often in patches (e.g. tree islands) (Johnson, 1958; Kolipinski & Higer, 1969; Willard et al., 2006). Paleoecological evidence also suggests that establishment and proliferation of woody vegetation in sawgrass marshes or on ridges occurred during periods of sustained drought (Willard et al., 2002, 2006; Bernhardt, 2011). Likewise, the location of boundaries between tree island communities and surrounding low-stature marsh vegetation also shifted in the past, depending on hydrology, climate, or fire induced changes in surface elevation(Stone & Chimura, 2004), or, since the 20th century, as a result of water management (Willard et al., 2006; Bernhardt & Willard, 2009).

Substantial changes in hydrologic conditions, whether natural or management-induced, are likely to cause quantitative and qualitative changes in tree island plant community structure and composition, and with extreme and prolonged changes even leading to complete degradation of forest structure and extensive change in ecosystem function. For instance, management-related extreme and prolonged high water level caused loss of tree island number and coverage in Water Conservation Areas (Patterson & Finck, 1999; Brandt et al., 2000; Sklar & van der Valk, 2002; Hofmockel et al., 2008). In contrast, shorter hydroperiod than prevailed during the pre-drainage era have resulted in the continued rapid development and succession of tree islands into well-developed forested communities in other regions (Johnson, 1958; Kolipinski & Higer, 1969; Willard et al., 2006).

The climatological records and hydrologic data from the Shark River Slough (SRS) region suggest that water level during most of the last decade of the 20th century was well above the 28-year average. In contrast, both the mean annual rainfall and water level were relatively low between 2001 and 2012. A comprehensive analysis of data collected in 2001/2002, and again in 2011/2012 on three SRS tree islands suggests that there was little change in vegetation composition in the head portion of tree island, whereas in the tail portion of the islands, the relative abundance of flood-tolerant species declined, while that of moderately flood-tolerant species increased over the study period (Sah et al., 2018). In contrast to the hydrological

conditions observed during 2001-2012 period, the conditions have been wetter than normal during the recent years. For instance, between 2012 and 2019, mean annual water level was higher than 28-year average in five of seven years (Section 1; Figure 1.3). However, the water level was not as high as in mid- to late 1990s. In addition, in 2014/15, mean water level was about 5 cm lower than 28-year average. Such a variation in water conditions over 18 years has provided an opportunity to assess the response of SRS tree island vegetation to the shift in hydrologic regime in the region. The question is whether the short-term changes in hydrologic conditions influence the vegetation successional process from more hydric towards relatively dry communities in SRS tree islands.

This study examines vegetation successional dynamics over a 18-year period between 2001 and 2019 within the hydric portion of SRS tree islands by i) assessing the response of species composition and life forms to the changes in hydrologic regime over time, and iii) quantifying changes in relative importance of woody species. We hypothesize that hydrologic differences have resulted in i) an increase in dominance of woody plants over herbaceous, and ii) increase in relative abundance of flood-intolerant woody species over flood-tolerant species. Moreover, both increase in dominance of woody species over herbaceous species, and flood-intolerant species over flood-tolerant species will suggest that tree island growth, development, and succession is dependent on hydrologic fluxes, particularly during periods of prolonged droughts or below average hydroperiod.

2.2 Methods

2.1.1 Study Area

The tree islands represent three islands (Black Hammock, Gumbo Limbo, and Satinleaf) in the Shark River Slough, and one island (SS-81) in Northeast Shark River Slough (Figure 2.1). The SS-81 island is located immediately downstream from the 1-mile (eastern) bridge on Tamiami Trail and is likely to be impacted by increased flow from the WCAs into the Park.

On the three islands within SRS, the bayhead and bayhead swamp plots were first sampled in 2001 or 2002, and then in WY 2011/12 and 2018/19. In the tail (tall sawgrass) region of the Gumbo Limbo island, an additional plot was also established and first sampled in 2002, and then was sampled in WY 2002/03, 2011/12 and 2018/19. On SS-81, a bayhead swamp plot was first-time established in 2012, and sampled only twice, in WY 2011/12 and 2018/19.


Figure 2.1 Location map of tree islands that have permanent plots in bayhead and/or bayhead swamp plots. The plots have been sampled in varying periods between 2001 and 2019. Vegetation in the plots on three tree islands (Black Hammock, Gumbo Limbo and Satinleaf) were first sampled in WY 2001/02, and then re-sampled in WY 2011/12 and 2018/19, whereas on SS-81, a bayhead swamp plot was established and first sampled in 2011/12 and re-sampled in 2018/19.

2.1.2 Data Collection

2.1.2.1 Vegetation sampling

The vegetation sampling in the bayhead and bayhead swamp plots consisted of a nested sampling design that accounted for all the major vegetation strata (trees & saplings, shrubs, seedlings, and herbaceous macrophytes) present within the plots. The sampling protocol followed the methodology described by Sah (2004) and Ruiz et al. (2013a). The size of bayhead and bayhead swamp plots are 400 m² and 225 m², respectively (Table 2.1). The size of sawgrass tail plot in Gumbo Limbo is the same as of the bayhead swamp plot.

Table 2.1 Location and topographic data (mean, minimum, and maximum) of bayhead (BH), bayhead swamp (BS) and sawgrass tail plots on four tree islands.

Tree Island	Plot	Easting NAD83 (UTM_Z17N)	Northing NAD83 (UTM_Z17N)	Plot Size (m2)	Mean (± 1 S.D.) Plot Elevation (m NAVD 88)	Minimum Plot Elevation (m NAVD 88)	Maximum Plot Elevation (m NAVD 88)	Burned (Years: 2001-2018))
Black	BH	531246	2832598	400	1.572 ± 0.062	1.435	1.729	2006, 2008
Hammock	BS	531053	2832372	225	1.450 ± 0.088	1.354	1.828	2006, 2009
Gumbo Limbo	BH	525986	2834724	400	1.499 ± 0.084	1.336	1.701	2017
	BS	525741	2834101	225	1.244 ± 0.034	1.186	1.302	2017
	Tail	525319	2833597	225	na	na	na	2017
Satinleaf	BH	524454	2837943	400	1.564 ± 0.109	1.444	1.827	2017
	BS	524421	2837834	225	1.456 ± 0.074	1.383	1.640	2017
SS-81	BS	547596	2847668	225	1.600 ± 0.029	1.570	1.660	2018

Each plot is gridded into $5\times5m$ cells, whose corners and midpoint are marked by 30 cm long flags and $\frac{1}{2}$ " PVC stakes affixed to the ground, respectively. In these plots, all trees (≥ 5 cm) are tagged with numbered aluminum tags, and the location of each tagged tree is recorded to the nearest 0.1m using the SW corner of the plot as a reference (0, 0). Furthermore, if a tree has multiple stems ≥ 5 cm diameter (cm) at breast height (DBH), each stem is tagged with a unique ID that allows it to be cross-referenced back to its 'parent'. Status (live and dead) and DBH of each individual tree was first recorded when plots were established (In BL, GL and SL in 2001, and in SS-81 in 2012).

During the 2011/12 and 2018/19 samplings, the tree census included the record of the status (live and dead) of tagged trees, and any tree that had grown into the >5cm DBH class (hereafter called 'in-growth') since the previous survey. In-growths were identified to species and tagged. The DBH of each tree was also measured. The density and species of all tree saplings (stems 1-5 cm in DBH) within each 5 x 5 m cell was also recorded, and assigned to one of two DBH size classes: 1-3 cm or 3-5 cm. At the midpoint of each cell, the density of woody seedlings (stems < 1 m height) and shrubs (stems > 1 m and < 1 cm DBH) was estimated using nested circular plots of 1.0 m² and 3.14 m², respectively. Seedlings present within the 1 m² (0.57 m radius) plots were counted and identified to species and assigned to one of three height categories (1-30, 30-60, & 60-100 cm). All shrubs rooted within the 3.14 m² (1 m radius) plots were counted and identified to species. The total cover of each shrub species was also estimated using a modified Braun-Blanquet scale based on the following six cover categories: Cat 1: <1%; 2: 1-4%; 3: 4-16%; 4: 16-32%; 5: 32-66%; & 6: >66% (Sah, 2004). The total cover of all herbaceous macrophytes, which includes seedlings, shrubs (< 1 m tall), epiphytes, vines and lianas, within the 1 m radius plot was similarly estimated by species, using the same cover scale outlined above.

2.2.2.2 Hydrology

For bayhead and bayhead swamp plots in three SRS tree islands, ground elevation data were available from a detailed topographic survey conducted using auto-level from either a 1st order vertical control monument (benchmark) or a reference benchmark established in marsh followed by their elevation estimation by differential GPS or calculating from the EDEN (Everglades Depth Estimation Network) water surface elevation for that particular location (Ruiz et al., 2011). For the bayhead swamp plot in SS-811, ground elevation was calculated using field-based water depth and EDEN water surface elevation data. In this plot, water depths were measured in each 5x5m sub-plot in 2018, when there was standing water. Ground elevation for each sub-plot was then estimated by subtracting the mean water depth from the EDEN water surface elevation data (http://sofia.usgs.gov/eden), those elevation data were used to calculate mean annual water depth, and discontinuous hydroperiod (i.e., the number of days per year when the location had water depth >0 cm for each plot).

2.1.3 Data Analysis

2.1.3.1 Hydrologic conditions

Mean annual water depth and discontinuous hydroperiod were calculated using ground elevation and the time series data of water surface elevation extracted from EDEN database. Previous studies have found that prairie and marsh vegetation composition are well predicted by the previous 3-5 years of hydrologic conditions (Armentano et al., 2006; Ross et al., 2006; Zweig & Kitchens, 2009) , whereas tree island vegetation was found strongly correlated with 7-year average hydroperiod and water depth (Espinar et al., 2011; Sah, 2004; Sah et al., 2018). Thus, in this study, we averaged hydroperiod and mean annual water depth for 4-7 water years (May 1st – April 30th) prior to each sampling event to examine the relationships between hydrologic parameters and change in vegetation characteristics.

2.1.3.2 Tree and Sapling-layer vegetation dynamics

Tree census data were summarized by calculating tree density and basal area. Changes in tree density and basal area together with differential mortality and/or in-growth among species over time usually result in different species composition. A change in tree species composition was analyzed using non-metric multidimensional scaling (NMDS) ordination. Species abundance data used in the ordination was species' basal area without any standardization and the Bray-Curtis (B-C) dissimilarity index was used as a measure of dissimilarity in the ordination.

Changes in relative density and basal area of different species usually result in changes in relative importance of species in the stand. To examine whether there has been a shift in relative importance of flood tolerant species in those plots, tree density and basal area for each species were summed for each plot, relativized as a proportion of the plot total, and used to calculate

Importance Value (IV) of species using the following equation: $IV = 100 \cdot ((Rd + Rba) / 2)$, where Rd is the species relative density and Rba is the species relative basal area.

2.1.3.3 Shrub and herb-layer vegetation dynamics

Shrub and herb census data were summarized by calculating annual mean percentage cover of all herb layer species including seedlings, shrubs (< 1 m tall), epiphytes, vines and lianas. We characterized change in shrub and herb species composition and examined vegetation:environment relationships using NMDS ordination. Species abundance data used in the ordination was species' mean percentage cover. The cover values for each species were standardized to plot total cover and the Bray-Curtis (B-C) dissimilarity index was used as a measure of dissimilarity in the ordination. Species present in less than 5% of sites were excluded from the analysis. Relationships between species composition and environmental vectors representing hydro-edaphic characteristics (hydroperiod and soil depth) were examined using a vector-fitting procedure incorporated in the computer R package VEGAN (Oksanen et al., 2019). Vector fitting is a form of multiple linear regression that finds the direction along which sample coordinates have maximum correlation with the fitted vector within the ordination space. Ordination axes were rotated so that Axis 1 was aligned with the hydroperiod.

2.3 Results

2.2.1 Hydrologic conditions

Hydrologic condition in the hydric portion of tree islands varies within each island as well as among islands depending on the location of tree islands within the R&S landscape. Over 28 years (1991/92 to 2018/19) for which EDEN data are available, the bayhead plot on Black Hammock island was drier than the other two islands, whereas the hydrologic condition in bayhead plots on Gumbo Limbo and Satinleaf were not much different. In the bayhead plots on Black Hammock, Gumbo Limbo and Satinleaf, the 28-year average hydroperiod were 135, 166 and 174 days, and annual mean (\pm SD) relative water level (RWL) were -7.9 \pm 2.6 cm, -2.6 \pm 5.2 cm and -1.0 \pm 5.4, respectively (Figure 2.2). The bayhead swamp plot was the wettest on Gumbo Limbo, whereas those plots on the Black Hammock and SS-81 islands were not significantly different in wetness. In bayhead swamp plots, the 28-year average hydroperiod were 208, 271, 294 and 342 days, and annual mean (\pm SD) RWL were 2.1 \pm 2.9 cm, 2.6 \pm 3.8 cm, 12.5 \pm 4.4 and 21.8 \pm 1.4 on SS-81 on Black Hammock, SS-81, Satinleaf and Gumbo Limbo, respectively (Figure 2.3).

Bayhead



Figure 2.2 Twenty-eight year (Water Year 1991/92-2018/19) average and annual mean (\pm SE) relative water level (RWL) in bayhead forests on three tree islands (a) Black Hammock, (b) Gumbo Limbo, and (c) Satinleaf.

The mean RWL in both bayhead and bayhead swamp plots varied annually. The hydrologic condition was much wetter in the mid-90s, i.e. prior to the first sampling in 2001/02 than the rest of the study period. In contrast, between 2006/07 and 2011/12, the annual mean RWL was lower than 28-year average, though such differences varied greatly (1 cm to 24 cm) among years (Figures 2.2, 2.3). In contrast, during the most recent 7-year period (from WY 2012/13 to 2018/19), the RWL was again above the 28-year average in 5 of 7 years. Variation in annual pattern differed among islands. For instance, after a surge in water level during 2017/18, primarily due to hurricane Irma, RWL decreased in the following year (2018/19) in three islands. But, SS-81, located in NESRS, experienced an increase in RWL probably caused by an increase in amount of water delivery into the Park resulted from a series of Modified Water Deliveries (MWD) Incremental Field Tests (USACE, 2020).





Figure 2.3 Twenty-eight year (Water Year 1991/92-2018/19) average and annual mean (±SE) relative water level (RWL) in bayhead swamp plots on four tree islands (a) Black Hammock, (b) Gumbo Limbo, (c) Satinleaf, and d) SS-81.

There is a time lag between changes in hydrologic conditions and their effects on vegetation composition. Researchers have shown that tree island vegetation is strongly correlated with 7-year average hydroperiod and water depth (Sah, 2004; Espinar et al., 2011; Sah et al., 2018). The periodic fluctuations in hydrologic conditions observed over 28 years were also manifested in hydroperiod and annual mean RWL averaged over seven years before each sampling event. In both bayhead and bayhead swamp plots, 7-year average hydroperiod and RWL were significantly (Paired t-test; p < 0.01) lower before 2011/12 sampling than 2001/02 and 2018/19 samplings. Typically, 7-year average hydroperiod was 60-90 days shorter and RWL was 10-12 cm lower during 2011/12 sampling than other two samplings (Figures 2.4).



Figure 2.4 Box Plots showing hydroperiod and relative water level (RWL) averaged over seven years prior to each of three vegetation sampling events (2001/02, 2011/2012 and 2018/19) in bayhead and bayhead swamp plots on Shark River Slough tree islands. Bayhead swamp plots were sampled on four islands (Black Hammock (BL), Gumbo Limbo (GL), Satinleaf (SL) and SS-81 (HL)), whereas bayhead plots were sampled on only three islands (BL, GL and SL).

2.2.2 Tree/Sapling-layer vegetation dynamics

Variation in tree and sapling layer vegetation in the hydric (bayhead and bayhead swamp) portion of tree islands are well illustrated by NMDS ordination (stress = 0.139). Tree and sapling layer woody vegetation in bayhead swamp plots of SS-81 and in both bayhead and bayhead swamp plots on Gumbo Limbo were different from the vegetation in the bayhead and bayhead swamp plots on other islands, and were indicative of much wetter type (Figure 2.5). In contrast, woody vegetation in bayhead swamp plots of Black Hammock and Satinleaf islands were similar in species composition to bayhead plots on those islands.

The NMDS ordination also revealed changes in vegetation composition over time. However, the change pattern varied among islands. For instance, shift in position of Gumbo Limbo bayhead and bayhead swamp plots first towards the drier end of the hydrologic vector, and then in opposite direction was in concurrence with changes in hydrologic pattern over the study period, but a shift in position of plots along the gradient on other islands was not so distinct.



Figure 2.5 Scatterplot of 3-D NMDS ordination based on tree species BA in bayhead and bayhead Swamp of four islands sampled 2-3 times between Water Year (WY) 2002 and 2019. Fitted environmental vectors represent the direction of maximum correlation between the variable and ordination configuration.

The importance value (IV) of species in both tree and sapling layers showed a great variability between plots and census periods. Across all bayhead plots, the average IV of several tree species declined between 2001/02 and 2011/12 (Table 2.2). However, the IV of three tree species dahoon holly (*Ilex cassine*), coastal plain willow (*Salix caroliniana*), and cocoplum (*Chrysobalanus icaco*) increased during this period. The most notable increase was in the IV value of *C. icaco*. Its IV increased across all bayhead plots. In contrast to the increase in IV of *C. icaco*, the flood-tolerant species pond apple (*Annona glabra*) decreased in the bayhead plot on Black Hammock, but remained relatively unchanged on Gumbo Limbo and Satinleaf (Figure 2.6). On Satinleaf, however, IV of sweetbay magnolia (*Magnolia virginiana*) significantly decreased.

Table 2.2 Mean (± 1 S.E.) tree and sapling importance value (IV) in bayhead and bayhead swamp plots sampled in 2001/02, 2011/12 and 2018/19. The IV values for bayhead plots were averaged over three islands, and for bayhead swamp plots over four islands.

Smaalaa	Species	Bayhead			Bayhead swamp		
Species	Code	2001_02	2011_12	2018_19	2001_02	2011_12	2018_19
Trees							
Annona glabra	ANNGLA	44.6 ± 24.5	35.7 ± 7.6	42.4 ± 4.3	14.3 ± 24.7	49.6 ± 50.0	47.8 ± 50.1
Chrysobalanus icaco	CHRICA	4.4 ± 7.6	18.2 ± 17.1	25.6 ± 19.4		2.2 ± 3.8	3.0 ± 5.2
Ficus aurea	FICAUR	12.7 ± 22.0	9.9 ± 16.3	0.7 ± 1.2			
Ilex cassine	ILECAS	2.9 ± 3.0	3.3 ± 3.4	5.8 ± 8.5		4.6 ± 7.9	8.0 ± 13.9
Magnolia virginiana	MAGVIR	15.4 ± 17.2	10.6 ± 9.9	11.6 ± 10.0		6.6 ± 11.4	7.8 ± 13.5
Morella cerifera	MORCER	2.9 ± 2.8	2.1 ± 1.0	1.7 ± 1.4	4.2 ± 7.2	3.8 ± 6.6	
Persea borbonia	PERBOR	0.8 ± 1.4					
Salix caroliniana	SALCAR	14.7 ± 4.9	19.8 ± 0.6	12.3 ± 5.6	14.9 ± 25.8		
Sambucus canadensis	SAMCAN	1.5 ± 1.5	0.4 ± 0.7				
		Saplin	gs/Woody vi	ines			
Annona glabra	ANNGLA	23.5 ± 18.7	10.7 ± 5	6.6 ± 6	49.6 ± 32.6	38.8 ± 36.3	42.8 ± 35.3
Cephalanthus occidentalis	CEPOCC	0.2 ± 0.3					1.0 ± 2.1
Chrysobalanus icaco	CHRICA	37.2 ± 45.4	50.7 ± 41	48.8 ± 39.1	0.2 ± 0.4	0.6 ± 1.2	1.3 ± 2.7
Dalbergia ecastaphyllum	DALECA			1.0 ± 1.7			
Ficus aurea	FICAUR	0.2 ± 0.3	14.4 ± 24.9	8.6 ± 10.5	0.1 ± 0.1	0.1 ± 0.1	
Ilex cassine	ILECAS	3.3 ± 4.9	2.6 ± 3.4	6.0 ± 8.5		0.4 ± 0.8	1.4 ± 1.5
Magnolia virginiana	MAGVIR	8.0 ± 13.8	9.4 ± 11.1	8.5 ± 8	14.9 ± 24	8.8 ± 16	8.5 ± 16
Morella cerifera	MORCER	21.3 ± 21.9	3.1 ± 3.3	0.5 ± 0.8	6.1 ± 7.3	23.7 ± 32.6	10.0 ± 12.4
Persea borbonia	PERBOR		4.0 ± 7.0		0.2 ± 0.3	0.3 ± 0.4	
Salix caroliniana	SALCAR	6.2 ± 3.2		19.3 ± 18.6	28.8 ± 48.1	27.4 ± 38.3	34.8 ± 40.9
Sambucus canadensis	SAMCAN	0.2 ± 0.3	5.1 ± 6.3				
Vitis sp.	VITSPP			0.7 ± 1.3			

In bayhead swamp plots, there were no trees on Gumbo Limbo and SS-81 (Figure 2.7). However, there were saplings of several species in the bayhead swamp plots of these islands . Between 2001/02 and 2011/12, the IV of the two dominant sapling species, *A. glabra* and *M. virginiana* increased in tree layer, but significantly decreased in sapling layers (Table 2.2). In the sapling layer, however, an increase in IV of wax myrtle (*Morella cerifera*) was noticeable, while IV of *S. caroliniana* species remained almost unchanged between these two census dates, as did most of the other sapling species.

Between 2011/12 and 2018/19, the change pattern observed in IV of several species was opposite of what was observed between the first two samplings. For instance, across all bayhead plots, the IV of *A. glabra* increased by 20% from 35.7% to 12.4%. In contrast, the IV of *S. caroliniana* decreased by 35%. An opposite trend was observed in the IV of these two species in the sapling layer. Surprisingly, the IV of two major species, *C. icaco* and *I. cassine* continued to

increase during this period also. In bayhead swamp plots, while the IV of *I. cassine* doubled on Black Hammock, the IV of other species did not change much during this period, except in sapling layer on SS-81. On this island, the IV of *M. cerifera* decreased by 50% while the IV of *S. caroliniana* significantly increased.



Figure 2.6 Importance value index (IVI) of tree and sapling species in Bayhead plots on four tree islands. ANNGLA= Annona glabra; CEPOCC= Cephalanthus occidentalis; CHRICA= Chrysobalanus icaco; DALECA= Dalbergia ecastaphyllum; FICAUR= Ficus aurea; ILECAS= Ilex cassine; MAGVIR= Magnolia virginiana; MORCER= Morella cerifera; PERBOR= Persea borbonia; SALCAR= Salix caroliniana; SAMCAN= Sambucus canadensis; VITSPP= Vitis spp.



Figure 2.7 Importance value index (IVI) of tree and sapling species in bayhead Swamp on four tree islands. ANNGLA= *Annona glabra*; CEPOCC= *Cephalanthus occidentalis*; CHRICA= *Chrysobalanus icaco*; ILECAS= *Ilex cassine*; MAGVIR= *Magnolia virginiana*; MORCER= *Morella cerifera*; PERBOR= *Persea borbonia*; SALCAR= *Salix caroliniana*.

2.2.3 Shrub and herb layer vegetation dynamics

Variation in shrub and herbaceous species composition in tree island plots was well summarized by the NMDS ordination (stress = 0.165), which revealed that bayhead, bayhead swamp and marsh vegetation were different in shrub and herb composition (Figure 2.8). Bayhead and bayhead swamp plots separate to both sides of NMDS Axis-1 which was strongly associated with hydrology (7Yrs RWL, r = 0.45, p-value<0.05) (Figure 2.8). The soil depth vector finds a position nearly perpendicular to the first axis, increasing along both the positive side of Axis 1 and the negative side of Axis 2.



Figure 2.8 Plots of axis scores derived from shrub and herbs cover–based three-dimensional non-metric multidimensional ordination (NMDS) of plots sampled on in 4 tree islands. Fitted environmental vectors represent the direction of maximum correlation between the variable and ordination configuration.

There is an increment in RWL and hydroperiod length towards the positive side of Axis 1 indicating that bayhead swamp plots have higher water levels and longer hydroperiods and are characterized by graminoids and forbs vegetation. Also, seedlings of two flood-tolerant tree species, pond apple (*Annona glabra*) and coastal plain willow (*Salix caroliniana*) were present (Figure 2.8). The bayhead plots located to the negative side of Axis 1 associated with lower water levels and shorter hydroperiods were dominated by the seedlings of moderate flood-

tolerant tree species cocoplum (*C. icaco*) and three ferns (giant leather fern, *Acrostichum danaeifolium*, swamp fern, *Blechnum serrulatum* and hottentot fern, *Thelypteris interrupta*) (Figure 2.8).

In herb and shrub layer vegetation, changes in species composition over time (2001/02-2018/19) varied among islands, and among the plot types within an island. Bayhead swamp plots show more variation than bayheads. The sawgrass tail sampled in Gumbo Limbo had shown a significant shift in composition. The plot had dense sawgrass in 2011/12 and 2018/19, but not during 2001-2003 samplings, when less dense sawgrass competition had allowed other hygrophilous species like saltmarsh umbrella-sedge (*Fuirena breviseta*), swamp smartweed (*Polygonum hydropiperoides*), pickerelweed (*Pontederia cordata*), green arrow arum (*Peltandra virginica*) and leafy bladderwort (*Utricularia foliosa*) to dominate.

2.4 Discussion

Tree island plant communities, especially those in the hydric portions of tree islands, are dynamic, and respond to changes in hydrologic conditions. Our results show that periodic fluxes in the hydrologic regime, resulting in below average water levels and shorter hydroperiods over a period as short as one decade, promote the establishment and growth of woody plants within the tree islands.

In tree islands within the R&S landscape, three distinct plant communities, bayhead, bayhead swamp and sawgrass marsh are commonly considered as phases of chronosequence of vegetation succession. Along this sequence, while the proportion of woody plants and herbaceous species varies, woody composition in bayhead and bayhead swamp also differs (Armentano et al., 2002; Sah et al., 2018). Bayhead forest typically has a mix of flood-tolerant and flood-intolerant tree species. Several flood-tolerant tree species e.g., pond apple (Annona glabra), wax myrtle (Morella cerifera), sweetbay magnolia (Magnolia virginiana), and coastal plain willow (Salix caroliniana) that occur in bayhead also occur in bayhead swamp portions of tree islands. However, their growth remains stunted in the latter. Because of relatively dry conditions during 2001/02-2011/12, our expectation was that woody plant abundance would increase in cover in both bayhead and bayhead plots. During that period, we saw increases in tree density and basal area in both bayhead and bayhead swamp plots. In bayhead plots, floodtolerant species like pond apple (Annona glabra) and coastal plain willow (Salix caroliniana) saw their IV decline while moderately flood-tolerant species like cocoplum (Chrysobalanus *icaco*) and dahoon holly (*Ilex cassine*) increased. The increase in abundance of woody plants, especially flood-intolerant species, during the relatively dry period supports the concept that tree islands are dynamic communities and such changes in community composition in response to hydrologic fluxes may result in successional changes in plant communities (Stone & Chimura, 2004). In those plots, an increase in the number of trees and a new cohort of saplings indicate a slow but steady progression in the succession of the bayhead swamp into a bayhead community.

With an increase in wetness in tree islands, one would expect a decline in abundance of woody plants or at least an increase in relative proportion of flood-tolerant species over

moderately flood-tolerant and flood-intolerant species. The period between 2011/12 and 2018/19 samplings was wetter than the 7-year period before 2011/12. In 2018/19, we observed an increase in IV of some flood-tolerant species, like *A. glabra* in the tree layer of bayhead plots. In contrast to our expectation, however, the IV of moderately flood-tolerant species like *C. icaco* and *I. cassine* also increased and the IV of a flood-tolerant species, *S. caroliniana* decreased in those plots. While on average, the 7-year period between 2011/12 and 2018/19 was relatively wet, in fact, after two wet years of 2012/13 and 2013/14, South Florida had experienced a severe drought in 2014/2015, that might have ameliorated the effects of increasing wetness. After 2015, water level in the study area was higher than the 28-year average for three years, but the RWL decreased again in 2018/19. The mixed results observed in changes in woody plant abundance, especially in bayhead plots, are mainly due to inter-annual variability in water level.

In contrast to three SRS islands, SS-81 in NESRS experienced an increasing wetness until 2018/19 (Figure 2.3). High water level in NESRS was mainly due to Increment Field Tests that began in October 2015 and continued through 2019 (USACE, 2020). Under these tests, a large volume of water was delivered every year into the Park, especially in the NESRS area. In addition, the 1-mile Tamiami Bridge completed in 2013 also has enhanced the water flow into the Park resulting in high water in NESRS the effects of which were observed on vegetation in the bayhead swamp plot of SS-18. Between 2011/12 and 2018/19, in this plot the IV of flood tolerant species, coastal plain willow (Salix caroliniana), increased by more than 100% while the IV of moderately flood-tolerant wax myrtle (Morella cerifera) significantly decreased (Figure 2.7). In coming years also, water delivery into ENP (both northeast and western SRS combined) is projected to increase by 25%, and the delivery into NESRS is projected to increase by approximately 162,000 acre-feet per year on average (USACE, 2020). If the trend in wetness observed on SS-81 between 2015 and 2019 and its effects on bayhead swamp vegetation is an indication, we can expect much more similar changes in the hydric portions of tree islands in both NESRS and SRS. Since this is the part of R&S tree islands, where expansion and contraction of islands in response to hydrologic changes commonly occur (Stone and Chmura 2004), restoration activities under the Comprehensive Everglades Restoration Plan (CERP) to increase the water delivery to ENP are likely to affect tree island dynamics. Depending on the magnitude of increase in water delivery into the Park, the balance between flood-tolerant and flood-intolerant woody and herbaceous vegetation on these tree islands and surrounding marshes will change, and that may result in a shift in boundary between tree islands and marshes in this part of ENP.

Distribution of tree species in tree islands has normally been viewed in relation to prolonged low or high-water conditions. However, tree island plant communities are also susceptible to the direct and indirect effects of disturbance, such as fire, particularly during drought conditions like those observed in 2007-2008. Fires not only kill trees, but also consume the rich organic soils, in the process altering the water regime by lowering the surface elevation (Wetzel et al., 2008). Under these circumstances, immediate post-fire flooding can be detrimental to tree island recovery, and may lead to their recession or elimination (Ruiz et al.,

2013b). Furthermore, fire is also known to sharpen the edges of both ridges and tree islands (Givnish et al., 2008; Wetzel et al., 2008). Between 2001/02 and 2011/12 samplings, three fires burned the marshes close to Black Hammock (Table 2.1). However, these fires did not burn the part of our study plots, though both bayhead and bayhead swamp plots were within the fire boundary. Thus, vegetation within the bayhead and bayhead swamp portion of Black Hammock was not affected. Between 2011/12 and 2018/19, a fire in 2017 burned portions of Satinleaf and Gumbo Limbo, and a fire in 2018 burned partly the bayhead swamp plot on SS-81 (Table 2.1). Changes in abundance of woody plant species in the bayhead swamp portion of those three islands and in the sawgrass tail of the Gumbo Limbo were probably also affected by these fires and their interactions with hydrology. A more detailed analysis of the effects of those fires and their interaction with hydrology on vegetation in tree islands will help to better understand the plant community dynamics in the hydric portion of tree islands.

3. Overstory-understory interactions along flooding gradients in tree islands

3.1 Introduction

In forested communities, where vegetation is arranged in different height strata, understory vegetation accounts for a substantial part (up to 90%) of plant diversity (Gilliam, 2007). However, that also depends on the availability of resources such as soil water, nutrients, and light (Small & McCarthy, 2005). Spatial and temporal alterations in these resources, caused by natural events or anthropogenic disturbances, affect the diversity and cover of understory plant species, which may impact tree regeneration and forest dynamics (Royo & Carson, 2006). A general assumption in forest ecosystem management practices is that dominant species have the largest impact on ecosystem function, and therefore the study of tree layer associations will lead to an understanding of ecosystem structure and function. However, several researchers (McCune & Antos, 1981; Sagers & Lyon, 1997) have questioned whether all vegetation layers in the forests respond similarly to the same environmental gradients. This question is more relevant for the Everglades, where vegetation in general, and the forested communities in particular are the manifestation of topographic variation along which hydrology and soil nutrients vary systematically. In the Everglades, tree islands, which are tree-dominated patches interspersed in a background of marshes and prairies, vary in shape, size, hydrology, soil characteristics and plant community structure and composition (Armentano et al., 2002; Kozlowski, 2002; Sklar & van der Valk, 2002; Ross & Sah, 2011; Sah et al., 2018). Thus, a general question is how the vegetation in different strata - overstory and understory - interact along environmental gradients among and within tree islands in the Everglades.

Tree islands often include one or more types of plant community, namely tropical hardwood hammock, bayhead forest, bayhead swamp and tall sawgrass, arranged along a gradient of increasing wetness. (Olmsted & Loope, 1984; Gunderson, 1994; Armentano et al., 2002; Sah, 2004; Sah et al., 2018). The distinctness among different communities along the hydrology gradient is most evident in tear drop-shaped tree islands within the ridge and slough (R&S) landscape. At the most elevated portion (also termed as 'head') of these tree islands, hardwood hammocks are rarely flooded, broad-leaved forests composed of flood-intolerant tropical and temperate tree species. In contrast, the 'tail' portion is dominated at its upper end by a mixed-species assemblage of flood-tolerant trees, ferns, vines and graminoids, and further downstream by tall sawgrass. As the dominance of woody plants decreases along the gradient from head to tail, tree height and canopy cover also decrease (Sah et al., 2018), resulting in an increase in light availability for the ground layer (also called, 'understory') vegetation (Figure 3.1). Moreover, soils in the hardwood hammock at the head are alkaline, mineral soils with extremely high P concentrations, while soils in the seasonally-flooded tail communities are mostly organic, with low P concentrations (Ross et al., 2006; Espinar et al., 2011). We expect that changes in overstory characteristics and underlying environmental conditions in tree islands will affect understory plant species composition and biomass, which in turn will influence future composition in the tree-layer (Figure 3.2).



Understory diversity & productivity

Figure 3.1 Schematic diagram representing the variation in tree height, overstory productivity, understory light and vegetation characteristics (diversity and productivity) along hydrology gradient.





In the Everglades, hydrologic conditions have been greatly modified by implementation of a complex water management system (Light & Dineen, 1994), resulting in altered water flows and changes in frequency and duration of flooding and drying events. Along with disturbances

like fires and windstorms, these management-induced changes in hydrologic regimes act as major stressors that impact tree island vegetation structure and composition (Figure 3.2). However, the nature of effects of hydrologic modifications on the tree island vegetation depends on both the severity and duration of extreme environmental events. For instance, longer hydroperiod due to prolonged flooding, first causes physiological changes in flood-intolerant trees before they exhibit any visible changes or suffer mortality, altering tree layer composition (Kozlowski, 2002). In contrast, a disturbance that physically impacts tree canopy structure, caused by either breakage of branches or tree mortality, results in a modified understory light environment, and in turn affects the ground layer vegetation, tree seedling recruitment, and growth of surviving trees, eventually impacting overstory composition. This is true especially for hammocks that are potentially affected by the hydrologic changes associated with restoration efforts under the Comprehensive Everglades Restoration Plan (CERP), authorized by the Water Resources Development Act (WRDA) 2000. Within CERP, changes in water management associated with hydrologic restoration are likely to impact local and landscape-level tree island stressors such as hydrology, invasive exotics, windstorms, and fire. While broad-scale alterations in these stressors will impact the spatial distribution pattern of tree islands, the local and landscape scale hydrologic alterations are likely to affect the internal water economy of islands, which in turn will influence tree island plant communities, including understory vegetation composition. Since the response of the tree layer to hydrologic alteration typically lags behind that of the herb layer, it is possible that understory vegetation can serve an early indicator of anticipated changes in tree island conditions caused by restoration activities.

In this section, our objectives were to describe the understory vegetation composition, and to examine how canopy cover and hydrology interact to influence understory species composition and diversity along a flooding gradient, and whether there is a shift in the relative importance of these two factors along the gradient. We expected that the effects of canopy cover on understory species composition variation would be stronger in elevated portions of the topographic gradient, with shorter periods of inundation, than in areas with prolonged hydroperiod.

3.2 Methods

3.2.1 Study area

The study area included a subset of tree islands within Everglades National Park and Water Conservation Areas 3A and 3B. In the Everglades National Park, the study tree islands were in both ridge & slough (R&S) and marl prairie (MP) landscapes, whereas in the WCAs, they were only in the R&S landscape (Figure 3.3). The R&S tree islands are mostly tear drop-shaped, oriented parallel to the direction of water flow (north-northeast to south-southwest). At the most elevated portion (also called 'head) of these tree islands, hardwood hammock are rarely flooded, broad-leaved forests, whereas the 'tail' portion is dominated at its upper end by a mixed-species assemblage of flood-tolerant trees, ferns, vines and graminoids (Armentano et al.,

2002; Sah et al., 2018). In contrast, tree islands in the MP landscape are mostly hardwood hammocks, which vary in shape and size depending on the underlying bedrock sculpture.



Figure 3.3 Location map of tree islands on which both overstory and understory vegetation was sampled in permanent plots and/or in a series of plots on transects.

The forest flora in both hammock types mostly include tree species of tropical, West Indian origin, though the prairie hammocks are more species-rich and include several trees whose U.S. distributions are otherwise restricted to the Florida Keys. The hammocks in these two landscapes also have distinct soil characteristics (Ross & Sah, 2011). The mineral soils in the R&S tree island hammocks are not deep (usually < 1 m), but so rich in phosphorus that they

are considered hotspots embedded in the phosphorus-poor, oligotrophic landscape. The prairie hammocks, however, are characterized by shallower organic, relatively low-P soils. Within an individual tree island, both soil characteristics vary along the topographic gradient (Espinar et al., 2011; Ross et al., 2006).

3.2.2 Vegetation sampling

Vegetation sampling was done in permanent plots and at a series of sites along transects on tree islands. Within ENP, permanent plots were established in a network of 16 tree islands in the R&S (10) and MP (6) landscapes. Plots ranged in size from 225 to 625 m², and vegetation structure and composition were studied periodically. The plots in three tree islands were established in 2000-2001, and the others were established and first sampled in 2005-2007. Each plot was gridded into 5 x 5 m cells. Beside the plots, vegetation was sampled 1-2 times along transects on 12 islands, 9 within ENP, 2 in WCA3A and 1 in WCA3B (Figure 3.3).

3.2.2.1 Vegetation sampling in permanent plots

In permanent plots, vegetation structure and compositional parameters were measured in both canopy and ground layers. Vegetation parameters and sampling methods are described in detail in (Sah, 2004; Ruiz et al., 2011; Sah et al., 2018). Soil characteristics, including soil depth and soil nutrient parameters, were determined. Soil analysis methods are described in detail in (Ross & Sah, 2011).

Canopy openness (the percentage of the canopy gaps for a specified sky region) and leaf area index (the ratio of the total one-sided leaf area to the projected ground area) (Parker, 1995), were used as surrogate measures of understory light availability. Canopy openness was measured at the center of each 5 x 5 m cell using a spherical densiometer (Lemmon, 1956). In addition, a hemispherical photograph was taken with a digital camera (Nikon Coolpix 995; Nikon, Japan) and a hemispherical lens (Nikon Fisheye Converter FC-E8 0.21x) placed at 1 m height above the ground at the center of the plot. To minimize sun flecks, the photos were taken when the sun was not directly above the canopy and the north was always aligned with the bottom of the photo. Canopy openness and the 4-ring leaf area index (LAI) were determined from hemispherical photographs using the software Gap Light Analyzer (GLA), version 2.0 (Frazer et al., 1999).

3.2.2.2 Vegetation sampling on transects

Vegetation sampling along transects were done in three stages (Table 3.1). Between 2000 and 2002, vegetation was sampled along four transects on three tree islands (Black Hammock, Gumbo Limbo and Satinleaf). One transect, hereafter termed as 'NS transect', followed each island's long axis, and the other three transects were established in the west-east direction (hereafter, WE transects), at right angles to the NS transect. The three WE transects were re-sampled in the spring of 2011. A total of 554 plots (2.5 m radius) were sampled along these transects, with plots spaced at 5-10 m intervals. Between 2012 and 2014, vegetation data were collected along the N-S axis of nine tree islands; six were in ENP, two in WCA3A and one in

WCA3B. In these islands, a total of 309 plots were sampled, and the plots were spaced at 30 to 42 m intervals to coincide with the centroid coordinate of Landsat TM 30 x 30 m pixels. Between 2015 and 2018, we conducted vegetation surveys along transects on five islands, four (Gumbo Limbo, NP-202, SS-93 and SS-94) in ENP and one (WCA3B-12) in WCA3B (Figure 3.3). While sampling was done along N-S transect on SS-93 and SS-94, on other three islands, transects were randomly oriented, and sampling focused on the bayhead and bayhead swamp communities, for two reasons, 1) to provide a more complete representation of vegetation assemblages on tree islands within the R&S landscape, and 2) to augment fine scale vegetation mapping by providing the ground data in the areas of tree islands where the interpretation of spectral signature was difficult. On these transects, sampling was done at 10 -20 m intervals.

Tree Island	Region	Easting	Northing	Transects	Number of Plots				
Tree island transects sampled in 2001-2002, and again in 2011									
Black Hammock	ENP	531300	2832630	NS (1), WE (3)	165				
Gumbo Limbo	ENP	526020	2834820	NS (1), WE (3)	259				
Satinleaf	ENP	524490	2838030	NS (1), WE (3)	130				
	Tree island transects sampled between 2012 and 2014								
Black Hammock	ENP	531300	2832630	NS	18				
Gumbo Limbo	ENP	526020	2834820	NS	42				
SS-81	ENP	547620	2848170	NS	29				
PSU 66 TI	WCA3A	523710	2867430	NS	50				
Satinleaf	ENP	524490	2838030	NS	20				
WCA3B-12	WCA3B	546300	2857380	NS	49				
Chekika Island	ENP	534360	2847510	NS	40				
Johnny Buck	ENP	528270	2834700	NS	41				
WCA3A-266	WCA3A	518070	2853150	NS	20				
Tree island transects sampled between 2014 and 2018									
Gumbo Limbo	ENP	525999	2834793	-	59				
NP-202	ENP	529770	2838836	-	61				
WCA3B-12	WCA3B	546325	2857390	-	16				
SS-93	ENP	535151	2848529	NS	14				
SS-94	ENP	535517	2848771	NS	14				

Table 3.1 Tree island location and summary of transect sites on 12 islands on which vegetation was surveyed along transects between WY 2001/02 and 2018/19.

Vegetation sampling on the transects was done using a nested plot design. On all tree islands, the vegetation sampling protocols included: (1) an estimate of maximum height and cover class of trees and vines by species within a 2-2.5 m radius plot; and (2) an estimate of the cover class of herbs and shrubs by species within a 1 m radius plot around each transect point. The cover classes used to estimate species cover in each stratum were: 1, 0-1%; 2, 1-4%; 3, 4-

16%; 4, 16-33%; 5, 33-66%; and 6, > 66%. At each plot center, mean canopy cover was determined as the mean of four densiometer readings facing in each of the four cardinal directions (i.e., North, East, South, and West) (Lemmon, 1956).

3.2.3 Data Analysis

3.2.3.1 Hydrologic characterization

Along the transect, three representative measurements of water depth were taken within each vegetation plot by measuring the distance between the ground surface and the water table surface above the ground. At the few plots where the water table was below the ground surface, a small 3-cm radius hole was dug and allowed to equilibrate while the vegetation sampling within the plot was completed. The water table elevation, at these plots, was then estimated by measuring down from the soil surface to the top of the water table. These measurements were recorded as negative values to indicate that the water table was below the ground surface. Measurements of water depth were not taken at the "head" of the study tree islands, where the water table was sometimes inaccessibly far below the ground surface; thus, water depths at the "head" of each tree island were recorded as zero.

Mean annual water depth, termed as relative water level (RWL), and hydroperiod were used to characterize the hydrologic regime of the plots. The RWL for each hammock plot was calculated using ground elevation of the plots and surface water level adjacent to each island over the period from January 1, 2000 to December 31, 2018. Discontinuous hydroperiod (i.e., the number of days per year when the location had water depth >0 cm for each plot) were then estimated based on ground elevation and the time series data of water surface elevation extracted from EDEN database. Previous studies have found that prairie and marsh vegetation composition are well predicted by the previous 3-5 years of hydrologic conditions (Armentano et al., 2006; Ross et al., 2006; Zweig & Kitchens, 2009), whereas tree island vegetation was found strongly correlated with 7-year average hydroperiod and water depth (Sah, 2004; Espinar et al., 2011; Sah et al., 2018). Thus, in this study, we averaged hydroperiod and mean annual water depth for 4-7 years water years (May 1st – April 30th) prior to each sampling event to examine the relationships between hydrologic parameters and change in vegetation characteristics.

3.2.3.2 Understory vegetation composition

Multivariate techniques, including non-metric multidimensional scaling (NMDS) ordination, were used to examine the effects of environmental factors on understory vegetation. We did this analysis separately for two datasets, once collected from permanent plots and the other from transect sites. For permanent plots in which herb and shrub layer species cover data were collected within each 5x5 m grid cell, we calculated frequency and mean percent cover of each species for each island. Then, importance values (IV) for each species were calculated from the relative frequency and cover values by using the equation: IV = (Relative frequency + Relative cover)/2. To characterize the abundance of plant species in tree and sapling layers, we

calculated density and basal area, which then were used to calculate importance values (IV). However, for the transect sites, where species data were collected in nested plots at 5-30 m intervals along the transects, we summarized herb and shrub species data as the mid-point of the cover class, and used them as a measure of species abundance at each site.

The relationships between understory plant species composition and environmental variables, including hydrology, soil characteristics, and canopy cover were examined using a vector-fitting procedure incorporated in the computer program DECODA (Minchin, 1998). The significance of the environmental vectors was assessed using a Monte-Carlo procedure permutation test with 10,000 permutations of the species data, as samples in the given ordination space are not independent (Minchin, 1998). The differentiation of understory species along the environmental vectors was assessed by calculating species centroids as weighted averages of sample scores, and plotting them in the NMDS ordination space with sample points and fitted environmental vectors.

We used redundancy analysis (RDA) variance partitioning to examine the portion of understory vegetation composition explained by environmental and spatial variables in the transect data, because RDA allowed partialling out of the spatial variation and separation of the effect of canopy cover from that of hydrology by sub-grouping environmental variables (Borcard et al., 1992). Specifically, we used a partial RDA variance partitioning technique that allowed us to examine variation in understory species composition explained by canopy cover in the presence of the hydrology covariable, as well as for the hydrologic variable while accounting for canopy cover. In the first RDA analysis, we constructed four components of predictors, such as (1) environmental predictor only, (2) the spatial variable only, (3) environmental variables as predictor and spatial variable as the covariable, and (4) the spatial variable as the predictor and environmental variables as covariables. Based on these analyses, the variance partitioning technique allowed us to determine the percentage of variance accounted by environmental and spatial components. Later, we partitioned the pure and shared effects of canopy cover in relation to hydrology. To test the significance of the first and second RDA axes and the overall model we used a Monte Carlo test with 999 restricted permutations. At present, the RDA analysis was done using the data from only one island, Gumbo Limbo. To address the landscape-scale spatial variation and effects of individual tree islands, we are in the process of analyzing the data from all islands together, and the results will be presented in a manuscript (in progress).

3.3 Results

Along the hydrologic gradient of 12 tree islands, a total of 121 plant species were recorded in the understory. These included 10 fern, 23 graminoid, 50 forb, 5 shrub, 10 vines, and 23 tree (seedling) species. Based on species' mean cover, the most abundant species was sawgrass, *Cladium jamaicense* (CLAJAM). The next three dominant species included two ferns (giant leather fern, *Acrostichum danaeifolium* (ACRDAN) and swamp fern, *Blechnum serrulatum* (BLESER)) and seedlings of a tree species (Cocoplum, *Chrysobalanus icaco* (CHRICA)) that is abundant in bayhead forest. Several plant species had restricted distributions, as 34 species were present in fewer than three sampling plots.

Variation in understory species composition along transects on tree islands was well summarized by a species' cover-based 3-d NMDS ordination (stress = 0.13), which revealed that hardwood hammock, bayhead forest, and bayhead swamp and marsh vegetation were different in understory composition (Figure 3.4). However, within an individual vegetation type, especially in the wet part of the islands, there was a wide variation in species composition, as revealed by the spread of sites along the 2nd axis. In general, sites in bayhead swamp were richer in the number of understory species per plot than sites in hardwood hammock, bayhead or tall sawgrass. Mean (\pm SD) species richness was 7.0 (\pm 3.0), 6.2 (\pm 3.0) and 3.5 (\pm 2.0) species/plot (3.14 m⁻²) in bayhead swamp forest, bayhead forest and hardwood hammock, respectively.



Figure 3.4 Plots of axis scores derived from understory species' cover–based three-dimensional nonmetric multidimensional ordination (NMDS) of 805 plots sampled on transects in 12 tree islands. Fitted environmental and species richness vectors represent the direction of maximum correlation between the variable and ordination configuration.

The environmental vectors representing hydrologic condition (relative water level, RWL) and light availability (Tree cover) were strongly correlated (r = 0.853 and 0.712, respectively) with ordination configuration (Table 3.2), suggesting that understory vegetation on the study

islands is arranged along a gradient defined by these two major environmental variables. Since overstory tree cover, a surrogate measure of light availability, is negatively correlated with relative water level, no surprise, tree cover was the highest in the hardwood hammock part of the islands. Soil depth was also significantly correlated with ordination configuration, orthogonal to the light-water gradient, but the correlation of these variables with the ordination was less strong (r = 0.427, p < 0.001).

Variable	n	r	р
Relative water level (RWL)	548	0.853	< 0.001
Tree canopy cover (CanCov)	705	0.712	< 0.001
Soil depth (SoilDep)	548	0.427	< 0.001
Species Richness (SppRich)	805	0.458	< 0.001

Table 3.2 Correlation (r) and statistical significance of fitted community (species richness) and environmental vectors with species cover-based 3-dimensional ordination configuration.

In hardwood hammocks where tree cover was the highest, the understory vegetation was species poor, and mainly characterized by the dominance of tree seedlings (gumbo limbo, Bursera simaruba; sugarberry, Celtis laevigata (CELLAE); pigeon plum, Coccoloba diversifolia (COCDIV); white stopper, Eugenia axillaris (EUGAXI); Myrsine floridana (MYRFLO) and mastic, Sideroxylon foetidissimum (SIDFOE)), shrub (rouge plant, Rivina humilis (RIVHUM)), and ferns (southern shield fern, Thelypteris kunthii (THEKUN); sword fern, Nephrolepis exaltata (NEPEXA)) (Figure 3.4). The seedlings that were commonly present in hardwood hammocks were typically of flood-intolerant tree species (Jones et al., 2006; Stoffella et al., 2010). In contrast, in the bayhead forests, the understory vegetation was dominated by the seedlings of several flood-tolerant to moderately flood-tolerant tree species (Jones et al., 2006; Stoffella et al., 2010) (such as, cocoplum, C. icaco; strangler fig, Ficus aurea (FIUAUR); dahoon holly, Ilex cassine (ILECAS), wax myrtle, Morella cerifera (MORCER); red bay, Persea borbonia (PERBOR)). In addition, two ferns (giant leather fern, A. danaeifolium and swamp fern, B. serrulatum) were also common (Figure 3.4). In the understory of bayhead swamp forest, vegetation was mainly characterized by graminoids and forbs. Since this part of the island remains inundated for more than six months, seedlings of only two flood-tolerant tree species, pond apple (Annona glabra) and coastal plain willow (Salix caroliniana), were commonly present.

For the transect data, the portions of understory vegetation composition explained by environmental variables and spatial variables were examined using the RDA variance partitioning method. At present, we have used this method to analyze the data from only one tree island, Gumbo Limbo. Partitioning of environmental (Tree cover and RWL) and spatial variable (distance along transect) revealed that the spatial effect, combining both pure and shared, accounted for about 10.93%, and environmental variables accounted for about 6.14% of the understory compositional variance (Figure 3.5). While the pure spatial effect was 9.49% (p =

0.001), and the pure environmental effect was 5.66% (p = 0.003), the shared effect was only 2.3% (Table 3.3).



Figure 3.5 Venn diagram of variance partitioning of understory vegetation composition explained by a group of environmental variables (tree cover, TreeCov. and relative water level, RWL) and spatial variable (distance along transect). The details of symbols X1, X2, a, b and c are in Table 2.3.

Table 3.3 Breakdown of the variance partitioning of understory vegetation composition explained by a group of environmental variables (tree cover and relative water level) and spatial variable (distance along transect).

	df	Variance explained (%)	p-value
Environmental variables only (X1 = a+b) (TreeCov & RWL)	2	5.7	0.003
Spatial variable only $(X2 = b+c)$	1	9.5	0.001
Total, including shared $(X1+X2 = a+b+c)$	3	17.2	0.001
Environmental variable ('spatial' as co-variable)	2	6.1	0.001
Spatial variable ('environmental' as co-variable)	1	10.9	0.001
Shared	0	2.3	
Residual (adjusted)		85.2	

We examined the relative importance of canopy cover and RWL using partial RDA variation partitioning. Our results showed that along the gradient, tree cover explains relatively higher variation in understory composition than the RWL. While spatial effect, pure and shared, in both cases was approximately 10.5%, tree canopy cover explained 5.24% of the variance, which was slighter higher than 4.37% of variance explained by RWL (Figure 3.6; Table 3.4). Thus, when only tree cover was used as an environment variable, the total adjusted variance explained was 13.9%, whereas it was 13.0% when RWL was used as the environmental variable.



Figure 3.6 Venn diagrams of variance partitioning of understory vegetation composition explained by two subsets of environmental variables (tree canopy cover and relative water level) in combination with the spatial variable (distance along transect). The details of symbols X1, X2, a, b and c are in Table 2.4.

Table 3.4 Breakdown of the variance partitioning of understory vegetation composition explained by two subsets of environmental variables ((A) tree cover, TreeCov and, (B) relative water level, RWL) in combination with the spatial variable (distance along transect).

	10	Env. variable: Tree cover (TreeCov)		Env. variable: Rel. water level (RWL)	
	df	Variance explained (%)	p-value	Variance explained (%)	p-value
Environmental variable only $(X1 = a+b)$	2	4.2	0.002	3.5	0.003
Spatial variable only $(X2 = b+c)$	1	9.5	0.001	9.5	0.001
Total, including shared $(X1+X2 = a+b+c)$	3	15.5	0.001	14.6	0.001
Environmental variable (spatial as covariable)	2	5.2	0.001	4.4	0.001
Spatial variable (environmental as co-variable)	1	10.6	0.001	10.4	0.001
Shared	0	1.9		1.8	
Residual (adjusted)		86.2		87.0	

3.4 Discussion

In the understory/herb layer vegetation on the study tree islands, sawgrass was the most omnipresent species, primarily because a high number of sites on the transects were in the tail region of the islands, where bayhead swamp forest and tall sawgrass vegetation with sawgrass as a major component are prevalent (Sah, 2004; Ruiz et al., 2011; Sah et al., 2018). On these islands, tree seedlings were an important component of understory vegetation. Moreover, species composition of tree seedlings present in the understory vegetation along hydrologic gradient mimic the overstory tree composition observed along the same gradient on those tree islands (Sah et al., 2015, 2018), even though seedlings of some tree species may have higher hydroperiod optima than mature trees of the same species (Sah, 2004). However, the high abundance of strangler fig (Ficus aurea) seedlings in bayhead forest was surprising, given that it is a flood-intolerant species, and is common at the high elevation part of tree islands. The seasonally inundated bayhead forest is the most topographically heterogeneous region in SRS tree islands. Microtopography in this forest is sometimes quite striking, with the larger trees frequently occupying (and helping to form) local peaks (Sah, 2004). Thus, strangler fig seedlings in the bayhead forest were most common on the local mounds, following a flood avoidance strategy noted for seedlings of other tree species in the bayhead forests (Reed, 2007). Beside overstory tree composition, occurrence and abundance of tree seedlings in the understory also varies among seasons (Sah, 2004; Reed, 2007). Since all islands were not sampled in the same season of the year, the time when these islands were sampled might also have impacted the variation in tree seedling composition.

Two major environmental variables (hydrology and tree cover) explained less than 6% variation in understory species composition, whereas the spatial component explained about 10% variation in species composition. The present analysis of variance partitioning included data from only one island (Gumbo Limbo). A more comprehensive analysis of data from all islands (that is underway) is likely to yield more robust and improved results. Moreover, environmental drivers other than hydroperiod and tree cover might be also important for understory vegetation. For instance, in an analysis of data collected from a small number of sites on three SRS tree islands, (Espinar et al., 2011) had found that environmental variables that also included soil nutrients (total nitrogen (TN), total phosphorus (TP), and N:P ratio) explained 18.8% of variation in understory vegetation It should be noted that spatial effects were non-significant in (Espinar et al., 2011), but were significant in the current study.

An important finding in our study was the stronger effect of canopy cover than RWL on understory vegetation composition. Since tree cover along the topographic gradient on these islands is also a function of the hydrologic characteristics, the effects of RWL on understory vegetation in the present analysis might also have indirectly mediated understory composition through its influence on tree cover, confounding the results. Thus, analyses such as path analysis or structural equation modeling (SEM) that can decouple the direct and indirect effects of environmental drivers on understory composition might be called for. Vegetation layers within forests may vary in their response to environmental gradients. However, the response of vegetation layers to the same gradient may also differ spatially, affecting the correspondence between them (Sagers & Lyon, 1997). At the moment, though we have not analyzed the response of overstory and understory vegetation responses to hydrology and light availability together, previous analysis of hardwood hammock plot data has shown that both overstory and understory vegetation differ in their response to relative water layer, our metric of hydrology (Sah et al., 2012). The discrepancy in such responses were attributed to differences in water use pattern by the plants in these layers. Trees usually use rain water during the wet season and ground water in the dry season (Saha et al., 2010). Patterns of water use in shrubs and herbaceous plants of south Florida hammocks has not yet been fully explored. However, in other ecosystems, researchers have found that understory herbs and shrubs are more dependent on rainwater regardless of their topographic position (Sagers & Lyon, 1997).

The relationship between overstory and understory vegetation is also shaped by their differential responses to disturbance (Sagers & Lyon, 1997; Tóbisch & Standovár, 2005). In the Everglades, researchers have documented the effects of disturbances on tree layer structure and composition in the tree island hammocks (Loope et al., 1994; Armentano et al., 1995, 2002; Ruiz et al., 2011). However, disturbances such as tropical storms, which are common in South Florida, might affect the tree layer and understory vegetation differently. In general, the dynamics of many understory plants in forests is driven by changes in canopy cover above them. Hence, some of the storms that may have little impact on tree species composition can create openings by causing physical damage to upper forest canopy, in turn affecting environmental conditions and species composition in the understory. In 2005, several islands were impacted by Hurricanes Katrina and Wilma, which made landfall in South Florida with maximum sustained winds of 70 and 105 knots, respectively (Knabb et al., 2006; Pasch et al., 2006). More recently, Hurricane Irma, a category 4 storm hurricane struck the southwest coast of Florida in September 2017, but its impact was felt throughout south Florida. While the 2005 storms had minimal effects on the tree layer composition on the ENP tree islands (Ruiz et al., 2011), Hurricane Irma caused severe damage in the tree layer on some islands within ENP (See Section 1). The understory vegetation data collected three months and one and two years after Hurricane Irma are being analyzed, and detailed results will be included in the manuscript (in progress).

In summary, understory vegetation composition in the southern Everglades tree islands varies along hydrology gradients. Since overstory and understory vegetation associations may differ in their response to similar environmental factors, species composition in the overstory is not always a good predictor of understory associations. However, as understory vegetation, especially those dominated by tree seedlings, are tightly linked through competitive interactions to the success of tree species in reaching the forest canopy, understory vegetation composition and dynamics has the potential to influence overstory stand structure, and shape broader ecosystem responses to major natural and anthropogenic ecological drivers.

4. Plant Community Detection on Tree Islands in ENP and WCA3B from multispectral WorldView 2 and G-LiHT LiDAR data

4.1 Introduction

Tree islands are essential and intricate components of the Everglades ecosystem. Plant communities in tree islands are arranged along hydrologic and nutrient gradients (Sah et al., 2018). Compartmentalization of the Everglades and modifications of hydrologic regimes have caused changes to the vegetation structure and composition of prairies, marshes and tree islands. In some cases, the hydrological alterations have resulted in the loss of tree islands (Willard et al., 2006). Therefore, as a result of the implementation of the Comprehensive Everglades Restoration Plan (CERP), further impact to tree island vegetation is expected. To understand how the structure and composition of plant communities in tree islands respond to hydrologic change, we need mapping techniques that can accurately detect clearly defined plant communities at a resolution that represents the scales at which change occurs along hydrologic and nutrient gradients. The objective of this project component was to develop tree island plant community detection algorithms based on spectral reflectance patterns and vegetation height and to detect vegetation changes over long time periods. We evaluated three aspects that affect tree island vegetation monitoring using remote sensing methods. The three aspects were, (1) the possibility of spectral signature extension across tree islands, (2) the effect of canopy height models on community classification accuracy, and (3) the detection of long-term plant community changes with a mixed methods approach combining high-resolution remote sensing with photogrammetry.

Accurate detection of wetland plant communities from high-resolution spectral data has been demonstrated across different Everglades landscapes (Gann et al., 2012, 2015, 2019; Wendelberger et al., 2018). The rationale for signature extension is that similar environmental conditions (including hydrology) lead to quasi-identical or similar plant communities across the larger landscape, and that these quasi-identical communities have comparable spectral reflectance patterns of electromagnetic radiation. Consequently, locally extracted spectral signatures of plant communities can be used to successfully detect communities with similar species distributions across larger spatial extents. However, if the spectral signatures of different plant communities resemble each other, then minor differences in island-specific signatures for community classes are more likely to be confused with different plant communities on other islands.

For communities that have varying seasonal signatures during dry and wet seasons, the use of bi-seasonal spectral data improved detection accuracy of mangrove communities in coastal wetlands (Wendelberger et al., 2018) and of freshwater marsh communities (Gann et al., 2012). However, confusion between shrubs and trees that display the same phenological cycle cannot be resolved with bi-seasonal data only. For classes that are defined by height and do not differ in species composition, reflectance patterns of individual trees (≥ 5 m) or shrubs (< 5 m) are expected to closely resemble each other in spectral reflectance. Hence, to further reduce

classification error between trees and shrubs and all other classes, we evaluated the inclusion of vegetation height estimates on community detection accuracy. We derived canopy height models (CHM) from Light Detection and Ranging (LiDAR).

Detecting gradual change of tree island communities over a specified time period requires detection of plant communities through at least two time steps. The existence of high spatial and spectral resolution satellite data facilitates the precise spatial detection of communities, but mapping the fine scale progression of vegetation changes beginning at a time before high-resolution spectral satellite data were available means that a change in data source and method are required. The only spatially explicit and exhaustive datasets for the 1900s are overlapping aerial photography, that allow for digitization of communities using analog or digital stereoscopy (Lydersen & Collins, 2018). We explored the integration of high spatial resolution datasets with manually digitized vegetation layers that have much larger minimum mapping units.

The main objective of this report is to describe and summarize the findings of the work accomplished during three phases between 2014 and 2019. In phase I, the validity of spectral signature extension across tree islands and surrounding marsh communities was evaluated by comparing tree island community map accuracies and areal coverage derived from two mapping strategies. The first method established a spectral classifier based on only training data samples located on one large tree island, which was expected to represent major regional tree island plant communities. We tested the representativeness of the signatures, extending them to one nearby smaller island. The second approach established a classifier from spectral training samples across both islands. Both classifiers were then applied to predict vegetation types across both islands and class differences and classification errors were evaluated. In phase II, the spectral signature extension methodology for detecting Everglades tree island community types was enhanced by including LiDAR data-derived vegetation height in the classification algorithm. We then tested the same method in an area with different hydrological management legacies. Phase III addressed community changes over the past ~40 years and the challenges of combining plant community classification and detection methods. We determined changes in five major tree island communities of five tree islands within Everglades National Park (ENP). Digitized communities from stereo photography of 1973 were compared to morphologically filtered spatially explicit maps derived from WorldView (WV) data.

4.2 Methods

4.2.1 Study Area

Phase I of this study covered two tree islands, located within 4 km of each other within ENP: <u>Gumbo Limbo</u> (island head located at 25.631 N 80.741W) and <u>NP202 (25.667 N 80.703</u> W). In phase II we expanded the study to six tree islands located within ENP and Water Conservation Area 3B (WCA3B): <u>Chekika</u>, (25.745 N 80.657W), <u>SS93 (25.757 N 80.646 W), <u>SS94</u> (25.754 N 80.649 W), <u>WCA3B-24</u> (25.843 N 80.631 W), <u>WCA3B-07</u> (25.823 N 80.626 W), and <u>WCA3B-25</u> (25.809 N 80.626 W) (Figure 4.1).</u>

Islands were selected on the criterion that a set of islands had to be located within the footprint of a single day and path acquisition of WV data. Using images acquired on the same day assures that spectral differences across islands are a result of variability in plant communities and spectral variability of the communities and not seasonal effects associated with phenological cycles of the vegetation or variability in atmospheric conditions. For Phase II, an additional island selection criterion was added, requiring that LiDAR data be available within a few years of the spectral WV data acquisition. Ideally, the LiDAR data were collected at the same time as spectral data to ensure that the vegetation height is correlated to the current vegetation conditions, but this is usually not possible unless acquisition schedules were coordinated. Limiting the window of acquisition dates of different data sources to a few years provided some confidence that most of the vegetation had not changed over the period, unless a major hurricane event impacted the study area between data acquisition dates.

4.2.2 Classification Scheme

After initial reconnaissance field visits to each tree island, community classes were defined at the species level whenever possible, but species-specific plant communities with similar spectral signatures were iteratively grouped under more generic classes to capture community morphological characteristics (Table 4.1), which also reduced classifier confusion. Community classes were representative of tree island and marsh community types located within a 200 m buffer around the approximate boundaries of the eight islands. Tree island vegetation classes included hardwood hammock (tH) with trees or woody species with heights greater than 5 m tall that are not typically found in standing water. Bayhead forests (tB) were occupied by trees or shrubs greater than 5m tall that tolerate extended periods of standing water, including a *Schinus terebinthifolia* (Brazillian pepper) dominated class (tB_Shin) on one of the islands. Bayhead swamp shrubs included the same woody species as bayhead forest but were less than 5 m tall (sB). This class also included shrub species that frequently do not grow to tree stature such as *Chrysobalanus icaco* (cocoplum). *Salix caroliniana* (willow) was also mapped as a dominated shrub class (sSa) on some islands.

Marsh classes were divided into classes dominated by a single species or by a mix of species. The dominant marsh classes included regular to dense *Cladium jamaicense* (gMCl), sparse *Cladium jamaicense* (gMCl_S), a broadleaved floating *Nymphaea odorata* (waterlily) dominated marsh class (blFNy), and *Typha domingensis* (cattail) dominated class (gMTy). Three mixed classes included a generic sparse graminoid class with a mixture of *Eleocharis spp.*, *Panicum spp.*, and/or *Rhynchospora spp.*, typically found in deeper and longer hydroperiod marshes or sloughs (gM_S); a mixed graminoid class of denser graminoids (gM); and two mixed broadleaved emergent classes with low shrubs, typically found in tree island tails and edges. These classes often had *Cephalanthus occidentalis* (buttonbush) and frequently included fern species (blE_s). We further differentiated this class to include broadleaved emergent communities with strong graminoid presence (typically sawgrass) (gM_blE_s). Two additional categories for classes found commonly within tree island tails and edges were a class of

herbaceous marsh (hM) and herbaceous marsh mixed with shrubs, typically *Cephalanthus occidentalis* (s_hM) (Table 4.1).

4.2.3 Image Data Processing

Due to its high spatial and spectral resolution properties, we chose the WorldView 2 (WV2) sensor to detect the plant communities. Previous studies within the larger Everglades (Gann & Richards, 2009; Gann et al., 2012, 2015; Wendelberger et al., 2018) showed that biseasonal data increased mapping accuracies significantly. For phases I and II, two images with minimal cloud cover containing the islands of interest were obtained for a wet and dry season. Optimally the two dates were far enough apart to capture the highest variability of phenologies of the vegetation. For phase I, we mapped Gumbo Limbo and NP202 (Figure 4.1) from two images containing both islands that were obtained for the dry season of 2011 (2011-05-01) and the wet season of 2013 (2013-09-22). For phase II, two images containing islands Chekika, SS93 and SS94 (Figure 4.1) were obtained for the wet season of (2012-10-20) and the dry season of 2013 (2013-04-06). For islands WCA3B-07, -24 and -25 located in WCA3B (Figure 4.1) two nearly cloud-free images were acquired in mid-January of 2017 (2017-01-19) and mid-February of 2018 (2018-02-16); both of these images represented the wet season condition period in WCA-3B. Nevertheless, we included both images in the analysis, since they were acquired one year apart and because seasonality and hydrological conditions vary with year, and because additional information on plant community was expected to be added with a second image.

All images were first geometrically, then radiometrically calibrated and atmospherically corrected in ENVI (Exelis Visual Information Solutions, Inc., 2013). Atmospheric correction for each image was completed using the Fast Line-of-sight Atmospheric Analysis of Hypercubes (FLAASH) module in ENVI (Exelis Visual Information Solutions, Inc., 2013). Selection of an atmospheric model in FLAASH was based on local air temperature at the time of image acquisition while the aerosol model chosen was based on wind direction (coastal vs. inland) and time of year. Local texture layers were generated for each pixel during the stacking procedures in R (R Core Team, 2013) using the focal function in the raster package (Hijmans & van Etten, 2010). We calculated local mean and range including the surrounding 8 pixels for each pixel of each of the 8 WV2 bands. Finally, the Normalized Difference Vegetation Index (NDVI) was calculated for both images and stacked with all other data layers, which resulted in a data cube of 51 layers or variables.

Vegetation height was generated from LiDAR data collected in May 2015 using Goddard's LiDAR, Hyperspectral & Thermal Imager (G-LiHT) sensor operated by the National Aeronautics and Space Administration (NASA). The originally processed digital surface (DSM) and terrain (DTM) models were created using first and last returns from the point cloud, and gridded at 1 m spatial resolution. This NASA product was aggregated at 2 m spatial resolution to match the resolution of WV2 images. A canopy height model (CHM), a representation of the height of the vegetation, was derived by subtracting the DTM from the DSM. Because of the dense ground vegetation in the tree islands, the LiDAR ground data was not representative of the ground elevation. The canopy height bias was corrected using forty tree heights collected *in-situ*. The forty heights were measured in the field using a telescopic pole, in a North-South transect on Chekika island. A circular 2 m buffer was drawn around each of the forty ground elevation points, then four LiDAR points were selected in each cardinal point at the edge of the 2 m circular buffer. From these points, the mean LiDAR canopy height was estimated. Vegetation heights calculated from the ground elevations measurements and the LiDAR derived elevation data were compared to estimate the potential bias in the LiDAR data. The difference of 1.07 m (bias) between the two methods was added to the LiDAR derived canopy height layer.

4.2.4 Digitization

The approximate boundaries of each tree island were digitized from 2009 (ENP) and 2011 (WCA3B) color infrared aerial imagery (CIR) at a scale of 1:1,000. A buffer of 200 m was added to each island to include the ecotone between tree island and surrounding marsh communities, and to allow for analysis of expansion and contraction of tree islands over time during phase III. Outlines of clouds, cloud shadows, and man-made structures (small boardwalks and buildings) overlapping the tree island and buffer areas were digitized. The areas masked by man-made structures were classified as missing data (NA), while the areas covered with clouds were classified using the corresponding cloud free image of each image pair.

Training samples for each vegetation class were digitized using a combination of field surveys, aerial surveys by helicopter and ground surveys, and high resolution CIR aerial photography in digital stereo view (DAT/EM Systems International, 2013). Training points were not located in areas of digitized clouds, cloud shadows, or man-made structures or shadows. For aerial and ground surveys, coordinates of points of interest were determined using satellite images and aerial photographs. Survey points were located in the field using a real-time kinematic Trimble GPS unit (R8) with satellite-based augmented systems (SBAS).

In total, 2,093 training points were digitized on Gumbo Limbo and an additional 555 on NP202 for the signature extension analysis. On Chekika (1,753), and SS93 and SS94 (5,372) a combined 7,125 samples were digitized, and for WCA3B 4,639 training points were digitized across WCA3B-24 (2,143), WCA3B-07 (1,545) and WCA3B-25 (951).

4.2.5 Spectral Signature Evaluation

For all training samples, signatures were extracted from the 51 (phase I), and 52 (phase II) layer data cubes. During phase I, two random forest classifiers were established for all community types; one with consideration of all training samples across both islands (method 2) and one using training samples of Gumbo Limbo only (method 1). For phase II, a random forest classifier was trained for all community types with consideration of all training samples across all three tree islands for both sub regions. Data analysis was performed with Microsoft R Open (MRO) version 3.5.1 (R Core Team, 2013). For classification we used the unifying modeling framework of the 'caret' package (Kuhn et al., 2015, 2019), using its 'rf' random forest function (Breiman, 2001). We determined from test runs that 1,000 decision trees were sufficient to

maximize model-based classification accuracy. Beyond 1,000 trees, no significant increase in accuracy was observed ($\alpha = 0.05$) (Kuhn et al., 2015, 2019). The parameter for the optimal number of random variables selected at each split ("mtry") was established for each random forest model through built-in tuning routines.

4.2.6 LiDAR Data Evaluation

During phase II we further evaluated the effect of LiDAR data in the accuracy of tree and shrub classes using a design-based accuracy assessment. We evaluated the change in the commission and omission errors in the tree and shrub classes when including LiDAR derived vegetation canopy height. For the LiDAR data analysis, all shrub containing classes were combined. The same was done for the classes containing trees. Then, the difference in accuracy for tree and shrub classes, between classifications using only bi-seasonal data versus bi-seasonal and LiDAR data was evaluated.

We evaluated four models for each island set. Models for Chekika and SS93 and SS94 were constructed for (1) wet season image only, (2) dry season image only, (3) bi-seasonal (wetdry), and (4) bi-seasonal plus LiDAR. For WCA3B islands four models were constructed using (1) the January 2017 image only, (2) the February 2018 image only, (3) the bi-annual set, and (4) the bi-annual plus LiDAR data sets. From each of the four models established for the data subsets the model with the highest model-based accuracy was used to predict the final classified maps. The minimum mapping unit (MMU) was established on a community basis ranging from 4 m² for *Salix* shrubs to 16 m² for all other classes (Table 4.1). The MMU was enforced with an iterative filtering function. Iteratively clumping and sieving classes to replace patch values below the MMU threshold with the neighboring class values eliminated small patch noise. The iterative filtering function was coded in R using clump and sieve functions from the 'raster' package (Hijmans & van Etten, 2010). Clumps were generated taking into account eight adjacent cells.

4.2.7 User-Based Accuracy Assessment

To evaluate the accuracy of all final plant community maps, we conducted a design-based accuracy assessment based on a stratified random sample design. We calculated the number of samples required for each map assuming a multinomial distribution of errors for a desired map accuracy confidence of 95% with a 5% precision of the estimate (Congalton & Green, 1999) and distributed the number of samples equally across all classes of a map.

Pixel centroids selected for accuracy assessment were greater than 1 m distant from training pixels (no overlap of pixels). Each sample was evaluated from aerial photography in stereo view at a fixed zoom of 1:500 and a reference class label was assigned. Confusion matrices were constructed from predicted and reference class labels, and overall and class-specific user's and producer's accuracies were calculated and adjusted for inclusion probabilities associated with the stratified random sample design (Olofsson et al., 2013, 2014; Stehman, 2013). Finally, bias adjusted areas were calculated for each class (Olofsson et al., 2013). All

sampling, accuracy assessment and bias adjusted area calculation was scripted in R (R Core Team, 2013).

4.2.8 Historic Change Analysis

Historic change of vegetation communities was assessed at a much coarser scale than that used to generate the fine resolution maps. To determine long-term changes for the five tree island within ENP (Gumbo Limbo, NP202, Chekika, SS93 and SS94), we digitized major community outlines for hardwood hammock tree (tH), bayhead tree (tB), bayhead shrub (sB), and mixed shrub, graminoid, and emergent broadleaf (s_gM_eBl) vegetation from 1973 stereoscopic near-infrared aerial photography. The minimum mapping unit was established based on a community basis ranging from 2 ha for the mixed shrub, graminoid, and emergent broadleaf (s_log Label 2). Polygons were digitized in ArcGIS with lines overlaid in Summit Evolution photogrammetry software from DAT/EM Systems (DAT/EM Systems International, 2013). Polygons were then labeled as one of the four vegetation classes, and their areal extent was calculated. The digitization and classification of polygons was based on characteristics like tone or color, shape, texture and canopy height. The final polygon layer was rasterized to the origin and outline of the digitized tree islands mapped from WV2 data.

To compare the 1973 map to the high-resolution detailed-class vegetation maps of 2012 to 2016, the WV2 derived maps were reclassified into the four coarse classes according to the reclassification schema of the 1973 map (Table 4.2), and clipped to tree island boundaries that were digitized from 2016 aerial photography using the ArcGIS basemap aerial photography of 2016. To match the MMU of the manually digitized 1973 maps we then applied a morphological filter that reclassifies areas smaller than the MMU to the spatially closest class in an iterative fashion until all polygons smaller than the MMU were absorbed by the surrounding classes.

In a final step, union of the two maps generated the intersections between the two map community classes. To reduce small area changes that were either insignificant or products of image misalignment, we applied a minimum mapping unit of 100 m^2 applying a morphological filter considering the four neighbor pixels. Tree island areas that did not overlap between the two years were classified as "Not Tree Island to Class x", for cases where tree islands expanded, and "Class x to Not Tree Island" in cases of island contraction. The spatially explicit changes of major communities across the four islands was mapped and tabulated for each island. The final class changes by island were tabulated in hectares and as percent of overall change by island.

4.3 Results and Discussion

4.3.1 Gumbo Limbo & NP202

Visually comparing plant community maps for Gumbo Limbo and NP202 (Figure 4.2) classified using training points restricted to Gumbo Limbo (Method 1) and training points located on both islands (Method 2) shows high similarity, which suggests signature extension is possible. A comparison of the design-based accuracies showed that training the classifier with samples from Gumbo Limbo only, resulted in an overall map accuracy of $95.1 \pm 2.2\%$ for
Gumbo Limbo (N = 736) and 92.7 \pm 2.5% for NP202 (N = 696). Overall accuracy increased to 97.3 \pm 0.97% (N = 736) and 94.9 \pm 1.6% (N = 656) for Gumbo Limbo and NP202, respectively, when training sample points on NP202 were included. While overall accuracies were very high for both methods and only increased by about 2%, class-specific user's accuracies (commission errors) and producer's accuracies (omission errors) varied in response to the addition of training points across islands.

On Gumbo Limbo the highest user's accuracy was achieved for hardwood hammock, reaching 100% for both methods (Table 4.3). All other classes, except for bayhead trees, reached accuracies of 90% or greater for method 1. Despite a nominal increase of 5.6% for method 2, lowest user's accuracy was observed for bayhead trees for both methods ($61.9 \pm 10\%$ method 1, and $67.5 \pm 9.6\%$ method 2) and the difference was not significant at the 95% confidence level (Table 4.3). Graminoid marsh and bayhead shrub were the only two classes on Gumbo Limbo that experienced a decrease in user's accuracy of 3.2% and 6.5%, while maintaining high accuracies of 93.5 ± 5.1% and 89.2 ± 6.4% respectively (Table 4.3). Producer's accuracies on Gumbo Limbo decreased for four of the eight classes when including training samples of NP202. The highest reduction was observed for hardwood hammock (-15%) followed by bayhead trees (-9.9%) and the broadleaved short shrub mix class (-7.1%) (Table 4.3).

As expected, more drastic differences in accuracy were observed for island NP202. The class that benefited most from the additional training points was hardwood hammock, increasing by 33.3% from $50 \pm 10.6\%$ to $83.3 \pm 8.1\%$ in user's accuracy. Bayhead tree and bayhead shrub classes did not improve with the addition of training points, maintaining a low $70 \pm 9.9\%$ and $80.5 \pm 8.6\%$ respectively (Table 4.3). A large increase in producer's accuracy of 15.1% was observed for bayhead swamp while bayhead trees saw an even larger reduction by 23.5% (Table 4.3). The differences in accuracy between the two methods led to less than 2% differences in class cover for both islands (Table 4.4). Rare classes that had cover of less than 2% (hammock trees and herbaceous marsh with shrubs) had the highest confusion rates, but since they are so rare, they did not affect the overall map accuracy or percent cover by class.

Even when just considering the woody classes, the differences for these low accuracy classes changed by 0.1% for herbaceous marsh shrub mix and 0.3% for hammock trees on Gumbo Limbo and no cover change was still below 0.1% change for both classes on NP202 (Table 4.5). Of all woody classes the most abundant class on Gumbo Limbo was the graminoid marsh broadleaf shrub mix that is very abundant in the low tails and edges of both islands (Figure 4.2, Table 4.5). The second most abundant class was bayhead shrub at about 24% on Gumbo Limbo and between 17-19% on NP202 (Figure 4.2, Table 4.5).

On Gumbo Limbo, switches in classes were most abundant between regular and sparse graminoid marsh with about 20% of sparse graminoid marsh switching to intermediate density when training from samples of both islands, and ~15% switched the other direction when leaving out NP202 samples (Table 4.6). This was not surprising because classes are spectrally very heterogeneous and are composed of the same species at different relative abundances. Likewise, for woody classes, the highest categorical switch occurred between classes with identical species

but different heights, and therefore, possibly different understory vegetation. Eighteen percent of bayhead trees switched to bayhead shrub when samples from NP202 were included and 15% of bayhead shrub switched to bayhead trees when leaving out the training samples of NP202 (Table 4.6). Hardwood hammock lost about 8% of its cover when mapped by Gumbo Limbo samples only (Table 4.6).

For NP202 the most common class switch when including training samples from NP202 was recorded for bayhead trees switching to bayhead shrubs (54%), followed by 36% of herbaceous marsh shrub mix switching to bayhead shrub and an additional 27% switching to the graminoid-broadleaf march shrub mix class (Table 4.7). This is an expected shift because the signature for bayhead shrub was refined and the producer's accuracy for bayhead shrub had increased by 15% with the inclusion of NP202 training samples (Table 4.3). For woody classes, leaving out samples of NP202 led to a 62.5% switch of hammock trees to bayhead trees, and 38% of bayhead shrubs to bayhead trees (Table 4.7), which explains the low 50% accuracy for the hammock tree class when not using the NP202 training samples (Table 4.3).

4.3.2 Chekika, SS93, SS94

The results for phase II showed that single season models only had a model-based overall accuracy of 85% (wet = $84.8 \pm 1.8\%$; dry = $85 \pm 1.4\%$), whereas the bi-seasonal data increased overall accuracy to $90.6 \pm 1.3\%$. Adding the canopy height variable increased accuracy by an additional 1.2% (Table 4.8). This statistically insignificant increase, however, increased accuracy for rare classes by as much as 10%. Hammock trees, bayhead trees and bayhead shrubs experienced a 7%, 10% and 10% increase, respectively, whereas the mixed-woody and non-woody classes did not show a significant difference in accuracy (Table 4.8). User's accuracy for all shrub classes combined increased by 2.6% and tree classes by 6.7%, when canopy height was included in the classifier, while producer's accuracy increased by 4.5% and 4.6% for shrub and tree classes, respectively (Table 4.9). Hence, vegetation communities on Chekika, SS93, and SS94 were derived from bi-seasonal+LiDAR data using training samples located on all three islands (Figure 4.3).

Overall map accuracy was $93.3 \pm 2.2\%$ (Table 4.10). The highest accuracy of 100% was achieved for gramminoid marsh dominated by *Typha domingensis*, and the bayhead tree classes including the *Schinus* class, and the lowest accuracy was $70 \pm 5.5\%$ for sparse graminoid marsh, followed by hammock trees at $80 \pm 4.8\%$ (Table 4.10). Except for the sparse sawgrass class (gMCl_S), all other classes reached accuracies of 92.6% or greater (Table 4.10). The most abundant plant community class across Chekika and SS93 and SS94 was *Cladium* with 72.8 \pm 2.2 ha, or about 40% of the mapped area. Together with sparse *Cladium*, this class, which dominates in the island tails and along the edges of islands (Table 4.10, Figure 4.3), covered about 63% of the mapped area (Table 4.10). The most abundant woody class was the herbaceous marsh shrub mix with 18.9 \pm 0.4 ha (10%), followed by bayhead shrub with 10.5 \pm 0.3 ha (6%) and bayhead trees with 7.5 \pm 0.01 ha (4%) (Table 4.10).

The similarity in spectral signatures of bayhead trees and shrubs also encountered in phase I on Gumbo Limbo and NP202 explains the strong and disproportionate increase in accuracy with the inclusion of LiDAR derived vegetation height. This result has important implications not only for the mapping of these communities but also in the detection of change, with the expansion or contraction of hardwood hammocks, bayhead forests, and shrublands, and in differentiating bayhead trees and shrubs from the shorter shrubs that are mixed with herbaceous and graminoid marsh.

4.3.3 WCA3B-07, -24,-25

Maps of vegetation communities on WCA3B-24, WCA3B-07 and WCA3B-25 show the configuration of vegetation communities on each island as classified using training points located on all three islands (Figure 4.4). Model-based overall accuracy for single season data was $69.9 \pm 1.9\%$ and $72.9 \pm 2.1\%$ for January 2017 and February 2018 data, respectively (Table 4.11). Combining the bi-annual wet-season data increased overall accuracy by 6% to $78.1 \pm 2.0\%$ and the inclusion of LiDAR by another 6% to $84.1 \pm 1.9\%$. The lack of a dry season image resulted in a cross-validated model-based accuracy of less than 90% contrary to the Chekika island group which reached 90% threshold with bi-seasonal data (Table 4.11). Accuracy again increased disproportionately for shrub and tree classes, when adding canopy height models. User's accuracy increased by 7.8% for shrub classes and by 25.4% for the tree classes (Table 4.12). For producer's accuracy shrub classes increased by 21.4% and tree classes by 11.6% (Table 4.12).

The final map for WCA3B-24, WCA3B-07 and WCA3B-25 was derived from the biannual+LiDAR data using training samples located on all three islands (Figure 4.4). Designbased overall map accuracy of the final map was $97.7 \pm 0.7\%$ (Table 4.13). This high accuracy can mainly be attributed to the very high accuracy of the most abundant marsh class *Cladium* which had an accuracy of $98.6 \pm 1.4\%$ covering just over $\frac{1}{3}$ of the mapped area (Table13). The absence of a hardwood hammock, which on Chekika decreased the tree mapping accuracy, led to almost 100% accuracy of bayhead trees, the only tree class present.

4.3.4 Forty Year Change

Stacking and cross-tabulating the high-resolution reclassified and morphologically filtered vegetation maps with the manually digitized 1973 maps, generated eight change classes and a no-change class (Table 4.14). The three change maps for Gumbo Limbo (Figure 4.5), NP202 (Fig.4.6), and Chekika and SS93 and SS94 (Figure 4.7), had seven, five and seven change classes, respectively. The distribution of change classes and the percent of each change class by island are presented in Table 4.14. During the 40-year time frame of this analysis, the majority of surface area on each island did not change in vegetation type. Least change occurred on Gumbo Limbo and Chekika where 89% (88 ha) and 88.5% (60.2 ha) did not change, respectively. The other three islands saw a change of roughly 20% on the island area.

In the case of Gumbo Limbo the highest change related to expansion of the island into the surrounding marsh, accounting for 43%. The second highest change class on Gumbo Limbo was

a 24% increase in bayhead trees that were shrubs in 1973, followed by a 22% increase in bayhead shrub as a result of increase in woodiness of mixed marsh shrub areas (Figure 4.5, Table 4.14). Only 8.7% of the change was a reduction in woodiness, 5.6% conversion of bayhead shrub to mixed marsh shrub and 3.1% converting from bayhead trees to shrubs (Figure 4.5, Table 4.14). The 0.1 ha of conversion of hammock to bayhead trees along the outer fringes of the hardwood hammock head (Figure 4.5, Table 4.14) represents an approximately 25% - 30% reduction of hammock tree on Gumbo Limbo (Table 4.3) over the past 40 years.

On NP202 the reduction of woodiness from bayhead shrub to mixed marsh with shrubs was the largest conversion, accounting for almost 50% of all changes (Figure 4.6, Table 4.14). An additional 7.7% of change reduced woodiness by bayhead trees converting to bayhead shrubs which was offset by a gain of about 5% of the change attributed to bayhead shrubs growing to trees. The largest conversion to more shrubby communities was observed with the expansion of the island into the neighboring graminoid dominated marsh areas (36%) along the southern and eastern edge of the island (Figure 4.6, Table 4.14).

Chekika saw a net gain in woodiness, predominantly driven by conversion from mixed marsh to bayhead shrub (27%), followed by a conversion from bayhead shrubs to trees (25.5%) and an expansion of the tree island into the surrounding marsh (17%) (Figure 4.7, Table 4.14). Only 28% of change was a result of reduction in woodiness, which mainly occurred as a loss of a large patch in the southern part of the tree island tail (Figure 4.7, Table 4.14).

SS93 and SS94 experienced a net gain in woodiness. About 69% of change on SS93 constituted an increase in woodiness and about 31% a decrease (Figure 4.7, Table 4.14). The largest loss occurred adjacent and just south of the bayhead trees, whereas the largest gain of about 30% change was observed in the southern part of the tail. The gain on SS94 was predominantly island expansion (41.9%) and bayhead shrubs converting to trees (14%). The same time 33.6% of change was bayhead trees converting to shrubs, which would mean a net loss in woodiness.

The recorded changes are conversions of generalized patches with a minimum size of 100m². Smaller area conversions are difficult to validate because of the coarse minimum mapping unit of the manually digitized maps and the co-registration of aerial photographs.

4.4 Conclusion

Mapping Results

For all eight islands and their surrounding marshes, sawgrass was the most abundant class covering an average of 49% (SD = 12.7%) of surface area across all islands. When considering woody classes, trees, shrub and shrubs mixed with herbaceous and graminoid species, the most abundant class across all islands were the mixed marsh shrub classes covering on average 16% (SD = 5.7%). Bayhead trees (tB) and hardwood hammock trees (tH) covered 12.4% and 1% on Gumbo Limbo and 8.9% and 0.3% on NP202 (Table 4.7 method 2). On Chekika, SS93 and SS-94, bayhead was present at 19.7% and hardwood hammock at less than 1% cover. On the three

tree islands located in WCA3B, hardwood hammock was absent, and the three shrub and tree communities were represented in almost equal parts, with bayhead shrub, *Salix caroliniana* shrub (sSa), and bayhead trees representing 35%, 34% and 31%, respectively. The strong presence of *Salix caroliniana* representing a more homogenous monotypic community was not present on the ENP islands. The tail and fringe areas covered by *Salix* on WCA3B-07 and WCA3B-25 were predominantly occupied by mixed marsh shrub and bayhead shrub on the other islands (Figs. 4.2 - 4.4).

General trends of woodiness between 1973 and 2013 across all islands showed a slight increase of 0.4 ha in shrub and tree classes, an overall increase of 0.2% in area covered by shrubs and trees, not counting herbaceous and graminoid shrub mixtures. The largest net increase of 4.2 ha was observed on Gumbo Limbo, which constituted 4.2% of the largest extent of the Island. The largest net loss of 5.9 ha occurred on NP202 representing 10.6% of the island surface area. Expansion of woodiness along edges of islands increased overall island area by 11.3 ha across the five islands or 4.9% of island surface area in reference to 1973. The largest expansion was observed along edges of NP202 (7.6%) followed by SS94 (7.3%), SS93 (4.8%) and Gumbo Limbo (4.7%) and least expansion was observed on Chekika (1.9%).

Most of the changes in woodiness were interior changes where shrubs grew into trees or when trees and shrub density was reduced and turned into more open herbaceous and graminoid marshes. Expansion of bayhead trees and shrubs outside the island boundary of 1973 was not observed (Figures 4.5 - 4.7).

Methods Development

High overall accuracy suggests that WV2 satellite data provide data with characteristics suitable for detecting and mapping tree island plant communities and their adjacent marshes. High confusion between tree and shrub community classes was observed for single season classifiers in the first two phases of this project. However, random forest classifiers applied to biseasonal and textural data, and especially where canopy height models were available, led to high class-specific accuracy. Woody tree and shrub classes were rarely confused with graminoid and broadleaved vegetation in the tails and surrounding marshes. These results indicate that the differentiation between tree islands and their tails and marsh communities is very reliable and that, given the spatial resolution of 2 meters for WV2 data, expansion or contraction of tree islands can be detected at short time intervals as they occur. Considering the spatial accuracy of 1-2 m expansion or contraction of tree island communities could be detected reliably when community boundaries shift by about 3-4 m on the ground.

Signature Extension

The rationale for signature extension was that similar environmental conditions lead to quasi-identical plant communities across the larger landscape, and that these quasi-identical communities had comparable spectral reflectance patterns. Phase I confirmed that while mapping an individual island with training points restricted to that island resulted in the highest accuracy,

applying that strategy for each island across a large study area with dozens of islands would likely be time-prohibitive. Conversely, a strategy that only trains on one island and attempts to classify vegetation on surrounding islands likely misses a range of spectral signatures for classes, resulting in lower than acceptable accuracy for some classes. In an attempt to capture the full range of spectral signatures of each class, a compromise between these approaches resulted in acceptable map and class-specific accuracies. This compromise includes spatially balanced training samples across all islands of different sizes that are mapped at the time.

Signature extension across islands was not reliable for woody classes that were spectrally very heterogeneous and had large species overlap with other classes. However, as demonstrated in phase II, inclusion of height information for vegetation drastically increased accuracies when signatures were combined across islands.

Canopy Height Models

Phase II demonstrated the value of adding LiDAR-derived canopy height models in the classification process. Confusion between tree and shrub classes as well as differentiating them from herbaceous marsh was reduced. Accuracy of graminoid and herbaceous classes did not increase significantly with the addition of LiDAR data. This was not surprising, because the species representing these classes have a high within-class and a low between-class variability of height. For marsh classes a LiDAR-derived vegetation density estimate might be more suitable, but this approach needs to be further explored.

Tree and shrub classes might also benefit from LiDAR-derived canopy texture and/or density estimates because canopy openness and understory structure for these classes vary (Figure 4.3). For instance, hardwood hammocks on all tree islands were relatively small areas in the heads of the islands, and were therefore limited to a small subset of species of the regional pool on any given island. The small area covered by this rich and ecologically important community makes it difficult to map because the number of training samples is too small to train a classifier. Confusion with bayhead trees is most likely to occur if training samples underrepresented hammock species. Spreading samples across all islands for this rare and spectrally diverse class is crucial but including LiDAR derived understory metrics will add valuable information on the variable understory density and structure of these classes (Venier et al., 2019). Accuracy for bayhead trees and shrubs on Gumbo Limbo and NP202 is expected to increase when the newly acquired green and near-infrared LiDAR data acquired by Everglades National Park in 2017 are incorporated in the vegetation detection. An updated version of the Gumbo Limbo and NP202 plant community map is planned for 2022. If signature extension with the use of LiDAR data-derived canopy height model is more feasible still needs to be evaluated. The 2017 LiDAR derived canopy height model is now available for all tree island monitoring in the eastern section of ENP and the 2019 campaign extended the canopy height model availability to the entire extent of ENP.

Change Detection

Long-term change detection for coarse morphological classes was possible when using stereo photography. Historic change detection of more detailed plant communities and for smaller minimum mapping units (higher spatial precision) on the basis of digitized community patches would be difficult because the delineation process is time consuming and interpretation of communities is not very accurate. Going further back in time than 1973 is even more difficult, because stereoscopic aerial photographs available for the study area from the 40s and 50s are panchromatic and the resolution of the scanned images is poor. A progressive analysis of timesteps between 1973 and current using aerial photography could assist in determining patterns of persistence and rates of change and the extent that apparent fluctuations can be attributed to image quality. For instance, the 25% - 30% conversion of hardwood hammock to bayhead trees could be confirmed or rejected on the basis of additional analysis of historic images during the 40 years.

Detecting short-term change between hammock, bayhead trees and bayhead shrubs will be even more challenging from a technical point of view. Data used in the detection process at any point in time come from data sources that are acquired over a period of several years. For instance, to generate a map representative for one instance, or "snapshot", in time the spectral data might be acquired during the wet and dry season of consecutive years, but the height information is updated less frequently every five to ten years, because LiDAR data acquisition and processing is expensive, and can only be conducted at longer time intervals.

To reduce the period over which data are acquired for each map instance in time, canopy height models can be supplemented from additional data sources. Using Synthetic Aperture Radar (SAR) data has proven useful in detecting forest structure types (Neumann et al., 2010; Wu & Sader, 1987) and SAR data are globally acquired by multiple satellites on a daily to weekly frequency. We will explore the use of SAR to derive canopy and understory metrics for different forest types. The spatial resolution of SAR data however is lower than that of WV data, and it might be too coarse for the small hammock forest patches. Since tree islands are relatively small phenomena across the landscape, the use of unmanned aerial systems (UAS) technology can provide frequent high-resolution data acquisitions at the island scale. UAS data delivers overlapping georeferenced aerial photography at centimeter resolution which can be used to generate canopy height models using stereoplotter software. They can also provide valuable information on canopy and understory forest structure (Li et al., 2020). The photographs would also provide valuable photographs with resolutions high enough to identify plant communities for training sample digitization and accuracy sample evaluation for accuracy assessment when mapping vegetation form lower resolution data (i.e., WV). This option should be considered and discussed with Everglades National Park management.

The recommended workflow for all future spatially exhaustive monitoring across tree islands is summarized in Figure 4.8. The spectral information of bi-seasonal data is crucial to differentiate shrub classes *Salix* and shrub herbaceous marsh mixtures when the images capture the phenology of shrubs during leaf-on and leaf-off episodes. Further the clear separation of

shrubs from trees is only accurate when canopy height models are used (Figure 4.8). LiDAR to date is the most accurate data source for canopy height models, but the infrequent acquisition schedules could be completed with SAR and/or aerial photograph derived canopy models. The canopy height model can be derived from different sources and we recommend to experiment with the use of satellite-borne SAR data and higher resolution UAS airborne photography.

With this high-precision monitoring plan, spatially explicit vegetation patterns and their changes over time can now be correlated with changes in hydrological patterns across the full mapped extent. The next phase of this project will extend the evaluation of effects of hydrological changes from the plot to the spatially explicit scale of each island and across the landscape among islands.

4.5 Tables

Table 4.1	Vegetation class codes	with corresponding	class descriptions	and island list
where the	class was mapped.			

Class Code	Vegetation Class Description	Islands
blE s	Broadleaf emergent marsh mixed with shrubs	All Islands
blFNy	Broadleaf floating dominated by <i>Nymphaea</i> odorata	WCA3B-24.
gM	Graminoid marsh, includes short graminoids and some <i>Typha spp</i> .	Gumbo Limbo, NP202, Chekika, SS93, SS94.
gM_S	Sparse graminoid marsh	All Islands
gMCl	Cladium jamaicense dominated marsh.	All Islands
gMCl_S	Sparse Cladium jamaicense dominated marsh	Chekika, SS93,SS94, WCA3B-07, WCA3B-24, WCA3B-25.
gMTy	Typha spp. dominated marsh	Chekika, SS93,SS94, WCA3B-07, WCA3B-24, WCA3B-25.
gM_blE_s	Mixed graminoid and emergent broadleaf marsh including ferns and shrubs	Gumbo Limbo, NP202, WCA3B-07, WCA3B-24, WCA3B-25.
hM	Herbaceous marsh	Gumbo Limbo, NP202, Chekika, SS93, SS94.
s_hM	Shrub and herbaceous mixed marsh with <i>Cephalanthus</i> , ferns and broadleaf emergent species	Chekika, SS93, SS94.
sB	Bayhead shrub, dominated by woody species with heights less than 5 m	All Islands
sSa	Salix caroliniana shrub	WCA3B-07, WCA3B-24, WCA3B-25.
tB	Bayhead tree, dominated by woody species with heights greater than 5 m	All Islands
tBSch	Bayhead tree, dominated by <i>Schinus</i> <i>terebinthifolia</i> with heights greater than 5 m	Chekika, SS93, SS94.
tH	Hardwood hammock tree, dominated by species that do not tolerate inundation	Gumbo Limbo, NP202, Chekika, SS93, SS94.

Class Code for High-	Major Classes	Minimum	
Resolution Maps	of the 1973 Map	Mapping Size	
blE_s			
blFNy			
gM			
gMCl			
gMS	aM hIE a	2 ha	
gMCl_S	gM_UIL_S	2 IIa	
gMTy			
gM_blE_s			
hM			
s_hM			
sB	сD	1 ho	
sSa	SD	1 IIa	
tB	t D	0.1 ha	
tBSh	LD	0.1 11a	
tH	tH	0.02 ha	

Table 4.2 Major community classes, and their reclassified detailed classes and minimummapping sizes. Class code and corresponding class descriptors as in Table 1.

Table 4.3 Gumbo Limbo (GL) and NP202 design-based class-specific user's and producer's accuracy. Method 1 = training data form GL only; Method 2 = training data from GL and NP202. Class Codes: gM = graminoid marsh, gMCl = Cladium marsh, $gM_S = sparse graminoid marsh$, $gM_blE_s = mixed graminoid and emergent broadleaf marsh including ferns and shrubs, <math>s_hM = shrub$ and herbaceous mixed marsh, sB = bayhead shrub, tB = bayhead tree, tH = hardwood hammock tree.

	0	Jumbo Limbo		NP202				
	Method 1	Method 2	Difference	Method 1	Method 2	Difference		
		User's Ac	curacy (%)	User's Accuracy (%)				
Class Code	Accuracy ± SE	Accuracy ± SE	%	Accuracy ± SE	Accuracy ± SE	%		
gM	96.7 ± 3.7	93.5 ± 5.1	-3.2	98.9 ± 2.2	80.5 ± 8.6	-18.4		
gMCl	96.8 ± 3.6	100.0 ± 0.0	3.2	97.7 ± 3.2	98.8 ± 2.4	1.1		
gM_S	95.7 ± 4.2	98.9 ± 2.1	3.2	90.8 ± 6.1	100.0 ± 0.0	9.2		
gM_blE_s	94.6 ± 4.7	99.0 ± 2.1	4.4	92.0 ± 5.7	96.3 ± 4.1	4.3		
s_hM	90.5 ± 6.0	92.9 ± 5.3	2.4	0.0 ± 0.0	20.0 ± 8.7	20.0		
sB	95.7 ± 4.2	89.2 ± 6.4	-6.5	80.5 ± 8.4	80.5 ± 8.6	0.0		
tB	61.9 ± 10.0	67.5 ± 9.6	5.6	72.5 ± 9.4	70.4 ± 9.9	-2.1		
tH	100.0 ± 0.0	100.0 ± 0.0	0.0	50.0 ± 10.6	83.3 ± 8.1	33.3		
	Produc	er's Accuracy (%))	Producer's Accuracy (%)				
Class Code	Accuracy ± SE	Accuracy ± SE	%	Accuracy ± SE	Accuracy ± SE	%		
gM	94.4 ± 4.1	98.5 ± 2.0	4.1	59.8 ± 14.4	91.8 ± 8.8	32.0		
gMCl	99.0 ± 0.9	100.0 ± 0.0	1.0	96.2 ± 2.7	93.6 ± 28.0	-2.6		
gM_S	93.7 ± 9.7	89.5 ± 7.4	-4.2	100.0 ± 0.0	98.6 ± 1.9	-1.4		
gM_blE_s	88.8 ± 12.3	99.8 ± 0.2	11.0	97.1 ± 4.4	99.6 ± 0.7	2.5		
s_hM	100.0 ± 0.0	92.9 ± 14.3	-7.1	$NA \pm NA$	100.0 ± 0.0	NA		
sB	73.3 ± 5.2	79.8 ± 5.0	6.5	74.5 ± 6.3	89.5 ± 3.3	15.0		
tB	92.9 ± 6.5	83.0 ± 9.0	-9.9	80.0 ± 7.6	56.5 ± 12.2	-23.5		
tH	100.0 ± 0.0	85.0 ± 26.3	-15.0	20.0 ± 32.5	50.0 ± 34.0	30.0		
Overall Accuracy	95.2 ± 2.2	97.3 ± 0.9		92.7 ± 2.5	94.9 ± 1.6			

Table 4.4 Class-specific area cover estimates in hectare and percent for Gumbo Limbo (GL) and NP202. Method 1 = training data GL only; Method 2 = training data GL & NP202. Class Codes: gM = graminoid marsh, gMCl = Cladium marsh, $gM_S = sparse graminoid marsh$, $gM_blE_s = mixed graminoid and emergent broadleaf, marsh including ferns and shrubs, <math>s_hM = shrub$ and herbaceous mixed marsh, sB = bayhead shrub, tB = bayhead tree, tH = hardwood hammock tree.

	0	umbo Limbo	NP202					
	Method 1	Method 2	Difference	Method 1	Method 2	Difference		
	Are	a Covered (ha)		Area Covered (ha)				
Class Code	Area (ha) ± SE	Area (ha) ± SE	ha	Area (ha) ± SE	Area (ha) ± SE	ha		
gM	29.2 ± 0.8	33.0 ± 1.0	3.8	11.7 ± 1.4	14.1 ± 1.0	2.4		
gMCl	115.2 ± 2.3	117.2 ± 0.0	2.0	49.6 ± 1.1	45.7 ± 0.9	-3.9		
gM_S	25.1 ± 1.4	21.5 ± 0.9	-3.6	37.3 ± 1.3	38.9 ± 0.4	1.6		
gM_bIE_s	25.9 ± 1.9	24.0 ± 0.3	-1.9	24.4 ± 1.0	25.1 ± 0.6	0.7		
s_hM	1.2 ± 0.0	1.2 ± 0.1	0.0	0.0 ± 0.0	0.0 ± 0.0	0.0		
sB	10.2 ± 0.4	9.6 ± 0.4	-0.6	5.9 ± 3.0	6.6 ± 0.4	0.7		
tB	4.7 ± 0.4	5.0 ± 0.4	0.3	4.7 ± 0.3	3.1 ± 0.3	-1.6		
tH	0.3 ± 0.0	0.4 ± 0.1	0.1	0.1 ± 0.1	0.1 ± 0.0	0.0		
	Are	a Covered (%)		Area Covered (%)				
Class Code	Area (%)	Area (%)	%	Area (%)	Area (%)	%		
gM	13.8	15.6	1.8	5.5	6.7	1.1		
gMCl	54.4	55.3	0.9	23.4	21.6	-1.8		
gM_S	11.9	10.2	-1.7	17.6	18.4	0.8		
gM_bIE_s	0.6	0.6	-0.9	11.5	11.9	0.3		
s_hM	0.6	0.6	0.0	0.0	0.0	0.0		
sB	4.8	4.5	-0.3	2.8	3.1	0.3		
tB	2.2	2.4	0.1	2.2	1.5	-0.8		
tH	0.1	0.2	0.0	0.0	0.0	0.0		

Table 4.5 Class-specific area cover estimates in percent only considering woody classes for Gumbo Limbo (GL) and NP202 for both methods. Method 1 = training data GL only; Method 2 = training data GL & NP202. Class Codes: gM_blE_s = mixed graminoid and emergent broadleaf marsh including ferns and shrubs, s_hM = shrub and herbaceous mixed marsh, sB = bayhead shrub, tB = bayhead tree, tH = hardwood hammock tree.

		Gumbo Limbo	NP202			
	Method 1	Method 2	Difference	Method 1	Method 2	Difference
	Percent	Cover Woody Class	Percent Woody Classes			
Class Code	Area (%)	Area (%)	%	Area (%)	Area (%)	%
gM_bIE_s	61.2	59.7	-1.5	69.5	71.9	2.4
s_hM	2.8	3.0	0.1	0.0	0.0	0.0
sB	24.1	23.9	-0.2	16.8	18.9	2.1
tB	11.1	12.4	1.3	13.4	8.9	-4.5
tH	0.7	1.0	0.3	0.3	0.3	0.0

Table 4.6 Cross-tabulated class percentages (method 1 (rows) vs. method 2 (columns)) for Gumbo Limbo. Method 1 = training data GL only; Method 2 = training data GL & NP202. Class Codes: gM = graminoid marsh, gMCl = Cladium marsh, $gM_S = sparse graminoid marsh$, $gM_blE_s = mixed graminoid and emergent broadleaf marsh including ferns and shrubs, <math>s_hM =$ shrub and herbaceous mixed marsh, sB = bayhead shrub (swamp), tB = bayhead tree (forest), tH= hardwood hammock tree. Row percentages indicate what percentage of class mapped with method 1 assigned to class mapped with method 2. Column percentages indicate what percentage of class mapped with method 2 was class mapped with method 1.

	Gumbo Limbo								
	Method 1 class percentages (row)								
				1	Method 2				
		gM	gMCl	gM_S	gM_blE_s	s_hM	sB	tB	tH
	gM	98	1.9	0	0.1	0	0	0	0
	gMCl	1.5	98.1	0	0.1	0	0	0	0
Ξ	gM_S	20.7	0.2	79	0	0	0	0	0
hod	gM_blE_s	0.2	3.6	0	95.9	0.1	0.1	0	0
[et]	s_hM	0.7	1.5	0	8	83.2	6.6	0	0
2	sB	0	0	0	3.6	0.1	92.1	4.2	0
	tB	0	0	0	0	0	18.3	81.3	0.4
	tH	0	0	0	0	0	0	0	100
			Method	2 class pe	ercentages (e	column)			
				1	Method 2				
		gM	gMCl	gM_S	gM_blE_s	s_hM	sB	tB	tH
	gM	80.3	0.5	0	0.1	0	0	0	0
	gMCl	4.9	98.7	0.1	2.2	0	0	0	0

gM S

s hM

sB

tB

tH

gM_blE_s

Method 1

14.6

0.2

0

0

0

0

0

0.8

0

0

0

0

99.9

0

0

0

0

0

0

96.1

0.5

1.2

0

0

0

1.7

97.4

0.8

0

0

0

0.3

1

83.6

15

0

0

0

0

5.4

94.6

0

0

0

0

0

8.3

91.7

Table 4.7 Cross-tabulated class percentages (method 1 (rows) vs. method 2 (columns)) for NP202. Method 1 = training data GL only; Method 2 = training data GL & NP202. Class Codes: gM = graminoid marsh, gMCl = Cladium marsh, $gM_S = sparse graminoid marsh$, $gM_blE_s = mixed graminoid and emergent broadleaf marsh including ferns and shrubs, <math>s_hM = shrub$ and herbaceous mixed marsh, sB = bayhead shrub (swamp), tB = bayhead tree (forest), tH = hardwood hammock tree. Row percentages indicate what percentage of class mapped with method 1 assigned to class mapped with method 2. Column percentages indicate what percentage of class mapped with method 2 was class mapped with method 1.

	NP202								
Method 1 class percentages (row)									
					Method 2				
		gM	gMCl	gM_S	gM_blE_s	s_hM	sB	tB	tH
	gM	97.6	1	1.4	0	0	0	0	0
	gMCl	12.3	86.8	0.5	0.4	0	0	0	0
Ξ	gM_S	7.5	0.1	92.4	0	0	0	0	0
hod	gM_blE_s	0.5	3.1	0.1	96.1	0	0.2	0	0
Iet	s_hM	0	0	0	27.3	36.4	36.4	0	0
2	sB	0	0	0	14.9	0	81.6	3.5	0
	tB	0	0	0	0.4	0	54.4	44.3	1
	tH	0	0	0	0	0	0	0	100
			Method 2	2 class pe	rcentages (co	olumn)			
					Method 2				
		gM	gMCl	gM_S	gM_blE_s	s_hM	sB	tB	tH
	gM	42.8	0.2	0.3	0	0	0	0	0
	gMCl	37.2	97.9	0.7	0.7	0	0	0	0
П	gM_S	19.2	0.1	99	0	0	0	0	0
hod	gM_blE_s	0.7	1.8	0.1	95.8	0	0.8	0	0
Iet	s_hM	0	0	0	0.2	100	1.1	0	0
~	sB	0	0	0	3.1	0	60.1	7.7	0
	tB	0	0	0	0.1	0	38	92.3	62.5
	tH	0	0	0	0	0	0	0	37.5

Table 4.8 Model-based overall, and class-specific user's and producer's accuracies for Chekika, SS-92 and SS-93. Class Codes: gM = graminoid marsh, $gM_S = sparse graminoid marsh$, gMCl = Cladium marsh, $gMCl_S = sparse Cladium$ marsh, gMTy = Typha, hM = herbaceous marsh, $s_hM =$ shrub and herbaceous mixed marsh, sB =bayhead shrub (swamp), tB = bayhead tree (forest), tBSch = Schinus dominated bayhead, tH = hardwood hammock tree.

	D	ry	W	Wet		asonal	Bi-Season	al+LiDAR
Class Code	UA	PA	UA	PA	UA	PA	UA	PA
gM	86.4	85.6	84.5	86.3	92.3	91.5	92.5	92.0
gM_S	94.6	91.7	93.8	94.6	97.9	97.4	98.1	97.7
gMCl	80.6	92.8	80.8	91.8	86.3	96.1	86.3	95.8
gMCl_S	90.9	91.3	91.3	92.7	95.2	94.3	95.1	95.0
gMTy	93.0	78.1	92.0	77.1	96.3	82.1	95.3	81.9
hM	83.4	79.8	85.9	75.4	90.3	84.7	90.7	85.5
s_hM	85.3	79.9	79.6	75.8	87.7	87.0	88.5	86.1
sB	77.6	68.5	79.3	74.4	86.2	77.9	96.1	93.8
tB	73.8	79.0	73.7	77.2	81.8	84.9	91.4	94.7
tBSch	80.9	79.1	90.2	88.8	92.3	90.4	92.8	89.8
tH	89.3	91.9	91.0	86.6	90.6	94.3	97.9	95.5
Overall Accuracy	85 ±	1.4%	84.8 ±	: 1.8%	90.6 ±	1.3%	91.8 ±	= 1.8%

Table 4.9 Effect of LiDAR data on user's and producer's accuracy of tree and shrub classes for Chekika, SS-92, and SS-93 tree islands.

	U	ser's Accurac	Producer's Accuracy			
Class	Bi-Seasonal	Bi + LiDAR	Difference	Bi-Seasonal	Bi + LiDAR	Difference
Shrub Classes	94.8%	97.4%	2.6%	92.9%	97.4%	4.5%
Tree Classes	87.1%	93.8%	6.7%	89.2%	93.8%	4.6%

Table 4.10 Design-based accuracy, area cover in hectare (ha) and percent (%) across Chekika, SS-92 and SS-93. Class Codes: gM = graminoid marsh, $gM_S = sparse graminoid marsh$, gMCl = Cladium marsh, $gMCl_S = sparse Cladium$ marsh, gMTy = Typha, hM = herbaceous marsh, $s_hM =$ shrub and herbaceous mixed marsh, sB =bayhead shrub (swamp), tB = bayhead tree (forest), tBSch = Schinus dominated bayhead, tH = hardwood hammock tree.

	User's	Producer's		
Class Code	Accuracy ± SE	Accuracy ± SE	Area (ha) ± SE	%
gM	88.6 ± 3.8	82.9 ± 7.5	12.6 ± 1.2	6.97
gM_S	70.0 ± 5.5	94.3 ± 2.7	11.8 ± 0.9	6.52
gMCl	98.6 ± 1.4	97.4 ± 1.3	72.8 ± 1.4	40.24
gMCl_S	95.7 ± 2.4	90.0 ± 1.9	40.8 ± 1.3	22.55
gMTy	100.0 ± 0.0	100.0 ± 0.0	2.5 ± 0.0	1.38
hM	100.0 ± 0.0	90.1 ± 8.9	2.7 ± 0.3	1.49
s_hM	98.6 ± 1.4	96.6 ± 1.6	18.9 ± 0.4	10.45
sB	94.3 ± 2.8	100.0 ± 0.0	10.5 ± 0.3	5.80
tB	100.0 ± 0.0	99.2 ± 0.2	7.5 ± 0.0	4.15
tBSch	100.0 ± 0.0	98.0 ± 1.4	0.5 ± 0.0	0.28
tH	80.0 ± 4.8	100.0 ± 0.0	0.3 ± 0.0	0.17
Overall Accuracy	94.6 ±	: 1.0%		

Table 4.11 Model-based overall, and class-specific user's and producer's accuracies for WCA3B-07, WCA3B-24 and WCA3B-25. Class Codes: $blE_s = broadleaf$ emergent marsh mixed with shrubs, blFNy = broadleaf floating dominated by *Nymphaea odorata*, $gM_S = sparse graminoid marsh, <math>gM_blE_s = mixed$ graminoid and emergent broadleaf marsh including ferns and shrubs, gMCl = Cladium marsh, $gMCl_S = sparse$ *Cladium* marsh, gMTy = Typha, hM = herbaceous marsh, sB = bayhead shrub (swamp), sSa = Salix, tB = bayhead tree (forest).

	Januar	y, 2017	February, 2018		Bi-Annual		Bi-Annual+LiDAR	
Class Code	UA	PA	UA	PA	UA	PA	UA	PA
blE_s	66.0	67.3	67.7	75.6	73.7	81.1	76.7	86.0
blFNy	91.1	78.8	92.5	78.8	92.5	83.0	94.2	83.9
gM_S	86.5	89.5	85.6	89.7	89.5	90.8	90.3	92.3
gM_blE_s	69.4	70.3	71.9	76.4	75.2	79.0	75.4	80.3
gMCl	70.9	81.3	72.1	78.5	80.4	85.9	80.5	85.5
gMCl_S	76.2	61.3	72.6	62.2	82.6	77.7	83.4	77.7
gMTy	74.1	61.6	73.3	65.6	80.5	72.1	82.7	73.3
sB	63.9	54.7	69.7	58.0	74.7	61.4	80.6	80.1
sSa	55.4	35.4	53.8	41.6	63.2	51.5	74.6	69.7
tB	61.2	81.1	69.9	84.1	72.2	86.3	99.4	100.0
Overall Accuracy	69.9 ±	= 1.9%	$72.9 \pm 2.1\%$		78.1 ±	2.0%	84.1 ± 1.9%	

Table 4.12 Effect of LiDAR data on user's and producer's accuracy of tree and shrub classes for WCA3B-07, WCA3B-24 and WCA3B-25 tree islands.

	U	ser's Accurac	у	Producer's Accuracy					
Class	Bi-Seasonal	Bi + LiDAR	Difference	Bi-Seasonal	Bi + LiDAR	Difference			
Shrub Classes	78.6%	86.4%	7.8%	65.0%	86.4%	21.4%			
Tree Classes	74.6%	100.0%	25.4%	88.4%	100.0%	11.6%			

Table 4.13 Design-based accuracy, area cover in hectare (ha) and percent (%) across WCA3B-07, WCA3B-24 and WCA3B-25. Class Codes: $blE_s = broadleaf$ emergent marsh mixed with shrubs, blFNy = broadleaf floating dominated by *Nymphaea odorata*, $gM_S = sparse$ graminoid marsh, $gM_blE_s = mixed$ graminoid and emergent broadleaf marsh including ferns and shrubs, gMCl = Cladium marsh, $gMCl_S = sparse$ *Cladium* marsh, gMTy = Typha, hM = herbaceous marsh, sB = bayhead shrub (swamp), sSa = Salix, tB = bayhead tree (forest).

	User's	Producer's		
Class Code	Accuracy ± SE	Accuracy ± SE	Area (ha) ± SE	%
blE_s	94.5 ± 2.7	100.0 ± 0.0	24.1 ± 0.7	6.73
blFNy	93.2 ± 3.0	97.0 ± 2.9	10.5 ± 0.5	2.93
gM_S	91.8 ± 3.2	98.6 ± 1.0	21.2 ± 0.8	5.92
gM_blE_s	98.6 ± 1.4	94.7 ± 2.1	56.8 ± 1.5	15.87
gMCl	98.6 ± 1.4	99.6 ± 0.2	119.7 ± 1.7	33.44
gMCl_S	100.0 ± 0.0	91.0 ± 4.6	35.7 ± 1.8	9.97
gMTy	97.3 ± 1.9	98.6 ± 1.4	52.4 ± 1.3	14.64
sB	100.0 ± 0.0	98.6 ± 1.4	13.1 ± 0.2	3.66
sSa	97.3 ± 1.9	100.0 ± 0.0	12.9 ± 0.3	3.60
tB	100.0 ± 0.0	100.0 ± 0.0	11.6 ± 0.0	3.24
Overall Accuracy	97.7 ±	= 0.7%		

Table 4.14 Vegetation-change classes in hectares and change class as percent of all changes. Maps for vegetation changes on Gumbo Limbo, NP202, and Chekika SS93 and SS94, see Figures 8, 9, and 10 respectively.

	Gumbo Limbo		NP202		Chekika			SS93			SS94				
Change Class	Area	0/	% of Change	Area (ha)	0/	% of	Area	rea (a)	% of	Area	%	% of	Area	%	% of
	(ha)	70			70	Change	(ha)		Change	(ha)		Change	(ha)		Change
No Change	88.0	89.1	-	44.1	79.1	-	60.2	88.5	-	13.9	80.5	-	4.1	82.7	-
Not Tree Island TO Mixed Marsh with Shrub	4.6	4.7	43.1	4.2	7.6	36.3	1.3	1.9	17.2	0.8	4.8	24.3	0.4	7.3	41.9
Mixed Marsh with Shrub TO Bayhead Shrub	2.4	2.4	22.1	0.2	0.4	1.9	2.1	3.1	27.0	1.0	5.8	29.9	0.0	0.6	3.5
Mixed Marsh with Shrub TO Bayhead Tree	0.1	0.1	1.1				0.1	0.1	0.8	0.0	0.1	0.3			
Bayhead Shrub TO Mixed Marsh with Shrub	0.6	0.6	5.6	5.8	10.4	49.5	1.7	2.5	22.2	0.9	5.2	26.5	0.3	5.8	33.6
Bayhead Shrub TO Bayhead Tree	2.6	2.6	24.0	0.5	1.0	4.6	2.0	2.9	25.5	0.4	2.6	13.2	0.1	2.4	14.0
Bayhead Tree TO Bayhead Shrub	0.3	0.3	3.1	0.9	1.6	7.7	0.5	0.7	6.1	0.2	1.1	5.8	0.1	1.2	7.0
Bayhead Tree TO Hammock Tree							0.1	0.1	1.2						
Hammock Tree TO Bayhead Tree	0.1	0.1	1.0												
TOTAL	98. 7	100.0	100.0	55.7	100.0	100.0	68.0	100.0	100.0	17.2	100.0	100.0	5.0	100.0	100.0

4.6 Figures



Figure 4.1 Study area in Everglades National Park and Water Conservation Area 3B. Tree Islands that were mapped.



Figure 4.2 Tree island plant communities on Gumbo Limbo (bottom) and NP202 (top) for 2016. Classifier trained from training points on Gumbo Limbo (Method 1, left) and training points from both islands (Method 2, right). Shrub classes have average tree heights <= 5m and Tree classes have average tree heights > 5m.



Figure 4.3 Tree island plant communities on Chekika (left), SS93 and SS94 (right). Shrub classes have average tree heights <= 5m and Tree classes have average tree heights > 5m.



Figure 4.4 Tree island plant communities on WCA3B-24 (left), WCA3B-07 and WCA3B25 (right). Shrub classes have average tree heights <= 5m and Tree classes have average tree heights > 5m.



Figure 4.5 Plant community changes 1973 - 2016 for Gumbo Limbo.



Figure 4.6 Plant community changes 1973 - 2016 for NP202.



Figure 4.7 Plant community change 1973 - 2016 for Chekika, SS93 and SS94.



Figure 4.8 Schematic of workflow for mapping using WorldView 2 and LiDAR data and derived products.

5. Summary and Conclusions

Tree islands, an integral component of the Everglades, are sensitive to large-scale restoration actions undertaken within the Comprehensive Everglades Restoration Plan (CERP). More specifically, changes in hydrologic regimes associated with restoration projects, including the construction of two Tamiami Trail Bridges and components of the Central Everglades Planning Project (CEPP), are likely to alter both the short and long-term impacts of existing local and landscape-level stressors, such as hydrology, invasive exotics, windstorms, and fire. Such alterations influence plant community structure and function within an island as well as the spatial distribution pattern of tree islands within the landscape. Thus, in the RECOVER monitoring program, tree island monitoring that focuses on effects at local and landscape-scales are both important.

The primary objectives of the research described above were to assess (1) structural and compositional responses of tree island vegetation to natural and management-induced hydrologic changes and disturbances, (2) alterations in the relative proportions of forest communities on the islands, and (3) the expansion or contraction of islands within surrounding marshes. The study included both, field sampling and analysis of plots and transects, and spatially exhaustive analysis of vegetation patterns using satellite data and aerial photography interpretations. Over the last five years (2014-2019), field sampling included repeated vegetation sampling within permanent plots on eight tree islands and one-time sampling in a series of plots along transects on five islands. Data analysis in some cases included vegetation structure and composition dating back as far as 2001. Fine-scale vegetation mapping was done for eight islands using WorldView (WV) satellite data, while for six islands the spectral data were combined with LiDAR data derived canopy height models, to achieve higher classification accuracy of woody vegetation types. In addition, long-term (40-year) vegetation changes were mapped on five islands, comparing vegetation types digitized from 1973 stereoscopic near-infrared aerial photography with the vegetation units derived from WorldView 2 imagery. The research described in this Report (2014-2019) shows that connecting rigorous field sampling with spatially extensive community patterns derived from satellite data and aerial photography interpretations can produce a more complete, spatially explicit inventory of vegetation patterns within individual tree islands, and aid in projecting responses to future changes in water management.

Plant community structure and composition within the R&S tree islands are largely arranged along gradients in hydrology. In moving down-gradient from the least flood-prone to the most frequently flooded tree island communities, the relative proportion of woody and herbaceous plants change. In general, there is a decrease in the cover and canopy height of trees, while the cover of herbaceous species increases. While species composition may shift in response to short-term changes in hydrologic regime and in response to disturbances such as tropical storms and fires, the extent of such changes varies along the gradient. Chapter 1 describes community dynamics in the hardwood hammock portions of the study islands, which are rarely flooded and have not burned for decades or more. Tree species composition in these communities is primarily the legacy of the long-term interaction between hydrology and tropical storms, though short-term responses in tree demography or understory species composition may result from flooding events and/or tropical storms.

Plant communities in the hydric portion of SRS tree islands are more dynamic. Our repeated surveys over periods as short as one decade (2002-2012) revealed that below-average water levels and short hydroperiods can promote the establishment and expansion of woody plants (Chapter 2). Given that vegetation communities usually show a lag in their response to natural events or management practices, relatively high annual variability in hydrologic regime and its interaction with disturbances (tropical storms and fire), as observed between 2012 and 2019, elicited a wide range of species responses, resulting in a mixed pattern of vegetation change (Chapter 2) and making it difficult to identify a single direction of vegetation succession (Figure 1.1). One factor underlying this complexity is that tree and understory vegetation strata may respond differently to environmental conditions. As described in Chapter 3, canopy cover has a significant effect on understory vegetation composition in southern Everglades tree islands. The forest canopy thus plays a keystone role in community dynamics: it is sensitive to and impacted by environmental variability, e.g., hydrological conditions and periodic tropical storms, and in turn, it indirectly mediates the response of the understory vegetation to these same environmental factors.

Finally, Chapter 4 illustrated how vegetation maps provide additional context and better understanding of the distribution of vegetation communities on tree islands in the ENP and WCA3B. Along with extensive field surveys, the vegetation detection workflow (Figure 4.8) can help refine our understanding of how and where communities might change in response to different water management scenarios. High-precision maps such as those derived from WV satellite data preserve the shape of landscape features, which can aid greatly in detecting small changes in the landscape. Coupled with the strategically located vegetation plots on tree islands, these maps provide a foundation for modeling vegetation change in response to alternative hydrological management practices.

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