



Southeast Environmental Research Center
FLORIDA INTERNATIONAL UNIVERSITY

Monitoring of Tree Island Condition in the Southern Everglades
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General Background

Tree islands, an integral component of the Everglades in both the marl prairie and ridge and slough landscapes, are complex ecosystems. They are sensitive to activities associated with the Comprehensive Everglades Restoration Plan (CERP) authorized by the Water Resources Development Act (WRDA) 2000 to restore the south Florida ecosystem. More specifically, changes in hydrologic regimes associated with restoration projects are likely to affect the internal water economy of the islands, which in turn will influence tree island plant community structure and function. To strengthen our ability to assess the “performance” of tree island ecosystems and predict how these hydrologic alterations would translate into ecosystem response, an improved understanding of plant community structure and function, and their responses to major drivers and stressors 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 and Jones 2004), a more extensive study was initiated in 2005 with initial funding from Everglades National Park and South Florida Water Management District (SFWMD). The study has been continued through 2017 with funding from US Army Corps of Engineers (USACOE).

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 that result in the changes in aerial extent or relative proportion of forests on tree islands and the boundary between islands and marshes. This research addresses the relevant performance measures (PM): 1) GE-15: ‘Ridge and Slough Sustainability’ 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 2009 CERP Monitoring and Assessment Plan (RECOVER 2009).

The specific objectives of our ongoing research are:

- 1) To characterize relationships among the hydrologic regimes with demographic fluctuations in the hardwood forests,

- 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 to consistently differentiate vegetation assemblage and to delineate their boundaries from spectral signatures of bi-seasonal satellite data and aerial photography
- 5) To detect changes and trends in aerial extent of the relative proportion of different vegetation communities
- 6) To investigate the correlation of vegetation changes in response to hydrological regime changes.

This document describes the work accomplished in 2014-2018, and emphasizes particularly the work done in 2017-2018. The document is organized in three sections. Section 1 summarizes vegetation structure and composition along hydrologic gradients in tree islands, and demographic changes in tree layer composition in the hardwood hammock portions of a subset of eight tree islands among a 16-island tree island network established in Everglades National Park for long-term monitoring and assessment. Section 2 examines the understory vegetation composition, and relative influence of canopy cover and hydrology on it along transects representing topographic gradient in 12 tree islands. Section 3 explores the methodology for detecting Everglades tree island community types (hardwood hammock, bayhead forest, and bayhead swamp) in three tree islands (WCA3B-07, WCA3B-24 and WCA3B-25) in Water Conservation Area 3B, and differentiating them from the surrounding marsh communities using WorldView2 (WV2) multi-spectral satellite data augmented with LiDAR data available for those islands.

1 Plant communities along hydrologic gradient in tree islands

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, where they have undergone extensive damage from extreme flooding, drought, fire, and tropical storms (Patterson and Fink 1999; Sklar and van der Valk 2002; Wetzel et al. 2008; Ruiz et al. 2011, 2013a). These islands are also 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 Project Planning (CEPP) project components (USACE 2014), are likely to alter the impact of local and landscape-level stressors such as hydrology, invasive exotics, windstorms, and fire on tree islands (Wetzel et al. 2017). While such alterations in the impact of these stressors at the broader scale 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 Ecological Conceptual Model (ECM) for Greater Everglades, 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 and fire regimes. It is therefore important to understand how the tree island plant community varies along the existing hydrologic gradient, and whether changes in community structure and composition influenced by natural and/or management-induced hydrologic conditions have surpassed the ability of the islands to remain functional. When plant community composition responds to changes in environmental drivers beyond a certain threshold, the tree island structure and function can be in peril, leading to the loss of the islands.

In the ridge and slough (R&S) landscape, flow-induced teardrop-shaped tree islands often include different plant communities - tropical hardwood (hammock), bayhead and bayhead swamp forests – arranged along topographic, hydrologic and soil nutrient gradients (Armentano et al. 2002; Sah 2004; Espinar et al. 2011; Sah et al. 2018). The hardwood hammock-dominated heads are of great ecologic significance, as both biodiversity and phosphorus ‘hotspots’ within the homogeneous oligotrophic landscape (Ross and 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 and Jones 2004; Wetzel et al. 2005; Ross et al. 2006, Givnish et al. 2008; Hanan and Ross 2010; Ross and Sah 2011; Sullivan et al. 2011, 2014, 2016) (**Figure 1.1**).

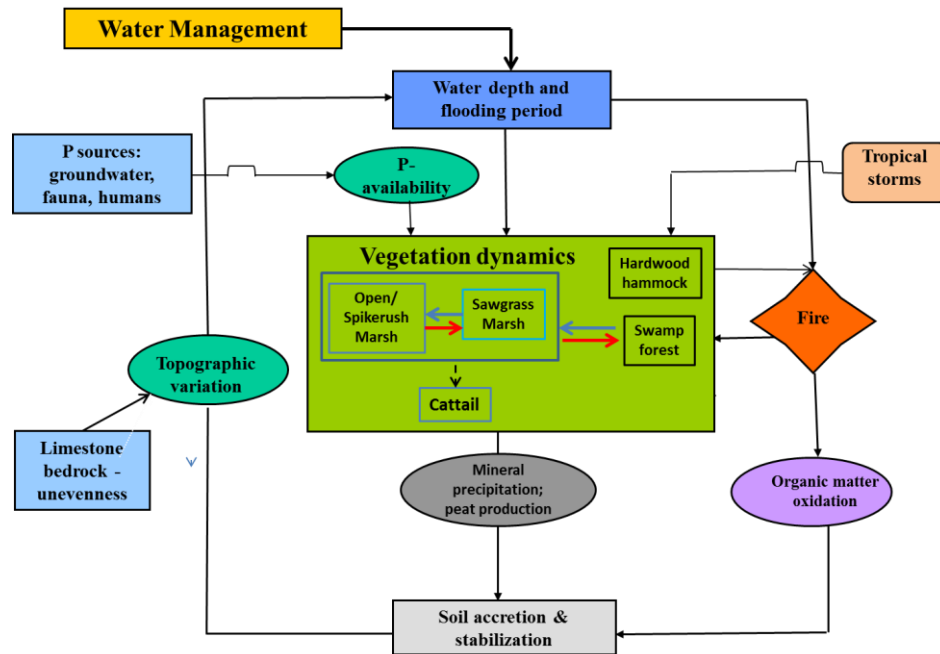


Figure 1.1: A Conceptual model: vegetation dynamics in tree islands and surrounding marsh.

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 hydrology and fire frequencies (Stone and Chmura 2004; Bernhardt and Willard 2009). In R&S tree islands, the swamp forests and tails are usually the areas that respond most noticeably to hydrology, whereas on an inter-annual scale the response of the tree island heads to windstorms overshadows any detectable hydrologic response (Ruiz et al. 2011; 2013b; Sah et al. 2018). Structural and compositional responses of the hardwood hammocks to hydrologic alterations become evident primarily in extreme events. In contrast, an analysis of the multiple year historical imagery suggests that a decline in the areal extent of tree islands within Everglades National Park occurred over the last half-century (Sklar et al. 2013), and the loss was

mostly concentrated in the swamp-forest dominated tail regions of the islands. Though the reason for this pattern has not yet been fully explored, one possibility is that they reflect alterations in the Everglades' hydrologic regime prior to the establishment of Everglades National Park (ENP), 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 both the local and landscape-scale effects, especially those taking place along the elevation gradient of the tree islands, is important. Ecological stressors influence tree island vegetation through their effects on species composition, growth, physiological processes, soil accretion and oxidation, fire frequency and intensity. Depending on the extent of hydrologic alterations, differential recruitment and growth of flood-tolerant and flood-intolerant species of different growth forms (e.g., herbaceous or woody) are expected to exhibit spatial change, resulting in shifts in species assemblages along the hydrologic gradient.

Therefore, to understand inter-annual variability, long-term trends and the mechanisms that drive them, it is essential to delineate spatially explicit patterns of community composition and configuration at high spatial precision that allow for detection of short-term fluctuations as well as persisting long-term change. An approach that concentrates most effort on linking extensively distributed ground surveys with community patterns derived from satellite data and aerial photography interpretations (Section 3) 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.

In the hardwood hammocks, which rarely get flooded and often have mean annual water table below 40 cm, tree species composition is probably more the legacy of long-term interaction between hydrology and other physical processes, including recurrent tropical storms. In these islands, plant communities recover within few years after a hurricane. However, vegetation recovery depends on also the post-hurricane environmental conditions. In September 2017, Hurricane Irma, a category 4 storm hurricane hit the southwest coast of Florida. However, its impact was felt in most of south Florida. A preliminary analysis of 2017 (WY 2017/18) tree data revealed a severe damage to the tree layer vegetation in eight tree islands, for which we had pre-Irma data. The questions were, a) how widely was the effects of Hurricane Irma on the tree islands in the Everglades, b) which species were likely to have damaged, and c) was the tree damage pattern related to tree size, e.g. DBH and height.

The effects of hurricane Irma were assessed on eight tree islands which represented a subset of 16-island tree island network established in Everglades National Park for long-term monitoring and assessment (Shamblin et al. 2008; Ruiz et al. 2011). In the hardwood hammock of these eight islands, vegetation structure and composition data were available for varying periods prior to the Hurricane Irma. While all the eight islands had vegetation data available for five water years (WY), 2006/07 to 2011/12, four islands had tree census data until 2016/2017, just one year before the hurricane. Post-Irma assessment of tree damage in these hardwood hammocks will serve as baseline data to follow the vegetation recovery from the damage. This will help to understand the islands' resilience, i.e. their capacities to recover since the past disturbance.

This section of the report includes the results of the continued monitoring of tree layer vegetation structure and composition within hardwood hammock on a subset of four tree islands among a 16-island network established in Everglades National Park for long-term monitoring and assessment (Shamblin et al. 2008; Ruiz et al. 2011). It also includes the post-Irma assessment of vegetation on additional four tree islands in the network.

1.2 Methods

1.2.1 Study Area

The tree islands represent a subset of tree islands studied between 2005 and 2010. Eight islands for which pre-hurricane data were available included one prairie island (Grossman Hammock) along the eastern boarder 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 2). SS-81 is located immediate downstream from the 1-mile bridge on Tamiami Trail, and is likely to be impacted by increased flow from WCA into the Park.

1.2.2 Data Collection

1.2.2.1 Vegetation sampling

The vegetation sampling in the hardwood hammock 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. (2011). Between WY 2011/12 and 2016/17, tree layer vegetation sampling was done in the hardwood hammock plots on only four tree islands, Black Hammock (BL), Gumbo Limbo (GL), Satinleaf (SL) and SS-81 (Heartleaf: HL). However, in WY 2017/18, after hurricane Irma, the tree layer sampling was done also in other 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 (Table 1.1).

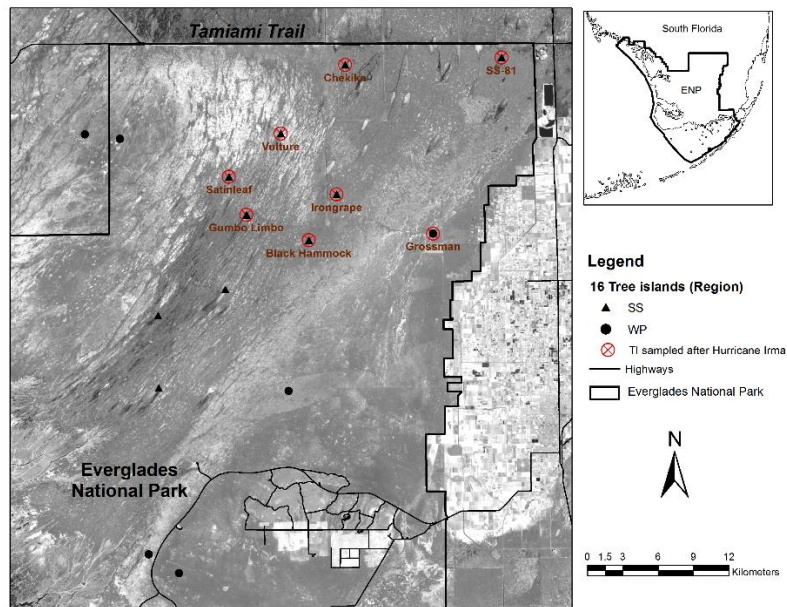


Figure 1.2: Location map of tree islands that have permanent plots in hardwood hammock. The plots have been sampled varying period between 2001 and 2018. Vegetation in the plots on eight tree islands were re-sampled in WY 2017/2018 after hurricane Irma.

Table 1.1: Location and topographic data (mean, minimum, and maximum) of hardwood hammock plots on eight tree islands sampled in 2017/18.

Tree Island	Easting NAD83 (UTM_Z17N)	Northing NAD83 (UTM_Z17N)	Plot Size (m ²)	Mean (\pm 1 S.D.) Plot Elevation (m NAVD 88)	Minimum Plot Elevation (m NAVD 88)	Maximum Plot Elevation (m NAVD 88)
Black Hammock	531295	2832630	400	2.330 \pm 0.166	1.988	2.584
Chekika	534372	2847485	400	2.624 \pm 0.035	2.545	2.712
Grossman	541819	2833205	400	2.042 \pm 0.144	1.386	2.238
Gumbo Limbo	525999	2834793	625	2.059 \pm 0.071	1.916	2.24
Irongrape	533651	2836523	400	2.240 \pm 0.050	2.092	2.345
Satinleaf	524499	2838019	625	2.221 \pm 0.076	2.082	2.368
SS-81	547639	2848113	300	2.168 \pm 0.304	1.592	2.649
Vulture	528918	2841667	400	2.663 \pm 0.191	2.338	2.977

Each plot is gridded into 5×5m cells, whose corners and midpoint are marked by 30 cm long ½" PVC stakes affixed to the ground. 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 allowed 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, Chekika, Grossman, Irongrape and Vulture Hammock in 2007).

In WY 2017/18, the tree census in eight islands included the record of the status (live and dead) of tagged trees, and any tree that had grown into the >5 cm DBH class (hereafter called 'in-growth') since the previous survey. In-growths were identified to species and tagged. 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) 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.

The forest canopy closure of the plot was estimated by taking two densiometer readings, one facing north and one facing south, at the midpoint of each cell within the plot (Lemmon 1956). The densiometer estimates of forest canopy closure were supplemented biannually 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.

Tree island leaf area index (LAI) was calculated by processing the bi-annual hemispherical canopy photos taken using Gap Light Analyzer, GLA 2.0 (Frazer et al. 1999). For each hemispherical image taken, we calculated the percent canopy openness and the 4-ring leaf area index (LAI) – the ratio of the total one-sided leaf area to the projected ground area (Parker 1995).

1.2.2.2 Hurricane damage

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 angle of tip-off were recorded. For the trees with main stem (trunk) broken, the height of the broken trunk and the DBH of broken trunk or the largest branch of tree were recorded. In addition, diameter of all woody debris (≥ 2.5 cm diameter) was measured and recorded.

1.2.2.3 Hydrology

For hardwood hammock plots in each 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 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). In conjunction with the daily 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).

1.2.3 Data Analysis

1.2.3.1 Hydrologic conditions

In conjunction with EDEN (Everglades Depth Estimation Network), water surface elevation data (<http://sofia.usgs.gov/eden>), field measurements of water depth were used to estimate the hydrologic conditions 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 and 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, and discontinuous hydroperiod 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 and 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 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 above-defined damage categories (uprooted, main trunk broken, branches broken, and undamaged). P-values for regression coefficients were determined using Z-tests.

1.2.3.2 Tree-layer vegetation dynamics

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

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' 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: $IV = 100 \cdot ((R_d + R_{ba}) / 2)$, where R_d is the species relative density and R_{ba} is the species relative basal area. Importance value (IV) data of each species were standardized to species maxima and the Bray-Curtis dissimilarity index was used as a measure of dissimilarity in the ordination.

1.3 Results and Discussions

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 (Ross and Jones 2004; USACE 2009; Wetzel et al. 2017). On the eight tree islands, re-sampled after hurricane Irma in 2017/2018, the annual mean relative water level (RWL) over twelve years (2006/07 to 2017/18), ranged between -95.5 ± 9.09 cm in Chekika Island and -60.7 ± 10.7 cm in Gumbo Limbo (Figure 1.3). The mean RWL in hammock plot was much (20 to 26 cm) lower in Chekika Island, Vulture Hammock, and Black Hammock 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 = 35.3%), and the lowest in Chekika Island Hammock (CV = 9.5%).

Based on their locations, SS-81 in the NESRS area that is comparatively drier than SRS, might be expected to have the lowest water level below the ground. However, it seems that hydrologic conditions on tree islands are not simply the function of regional hydrology, but also the geomorphological characteristics of tree islands, such as the tree island height, that is, the difference in elevation between the surface of the tree island and the surrounding marsh. In a study of 69 slough and prairie tree islands within the Park and WCA3B, relative water level (RWL) was negatively correlated with tree island height (Ross and Sah 2011). Among the four islands studied, Black Hammock has higher tree island height (99.1 cm) than Satinleaf (89.3 cm), Gumbo Limbo (87.8 cm) and SS-81 (79.9 cm). 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 (Ross and Sah 2011).

Between WY 2006/07 and WY 2011/12, the annual mean RWL, was 1 to 25 cm lower than 27-year (1991-2018) average (Figure 1.3). In contrast, the water level in the most recent 6-year period (from WY 2012/13 to 2017/18) was 2 to 16 cm above the 27-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 of years, none of the hammock plots on these islands was inundated during these periods, except in WY 2017/18. In this year, plots on 5 of 8 islands were partly inundated for varying period. 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 sub-plot on SS-81 in NESRS was inundated for 5 to 167 days in 8 of 12 years between WY 2006/07 and 2017/18.

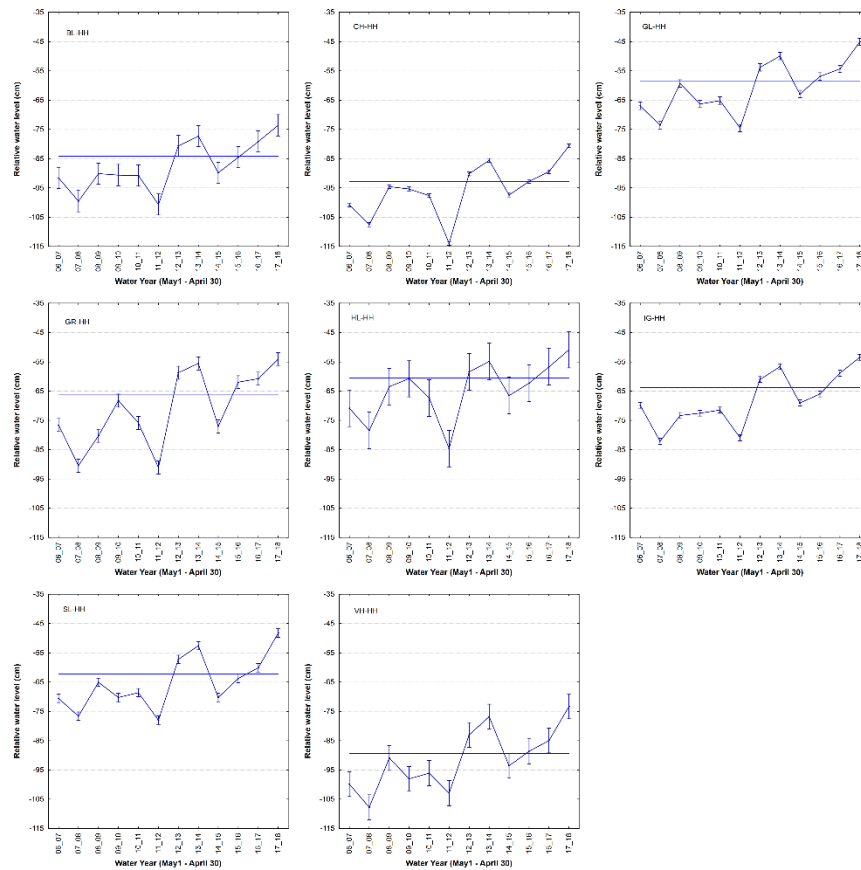


Figure 1.3: Twenty-six year (Water Year 1992–2017) average and annual mean (\pm SE) relative water level (RWL) in the hardwood hammock plots on four tree islands, (a) Black Hammock, (b) Gumbo Limbo, (c) Satinleaf, and (d) SS-81.

In general, the annual mean water level in these hammocks followed the regular dry (low) and wet season (high) pattern. However, in some years, the relative water level in the hammock plots was much higher in the dry season than the wet season due to either anomaly in weather pattern, management-induced changes in hydrologic regime or both. For instance, over the last 12 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.

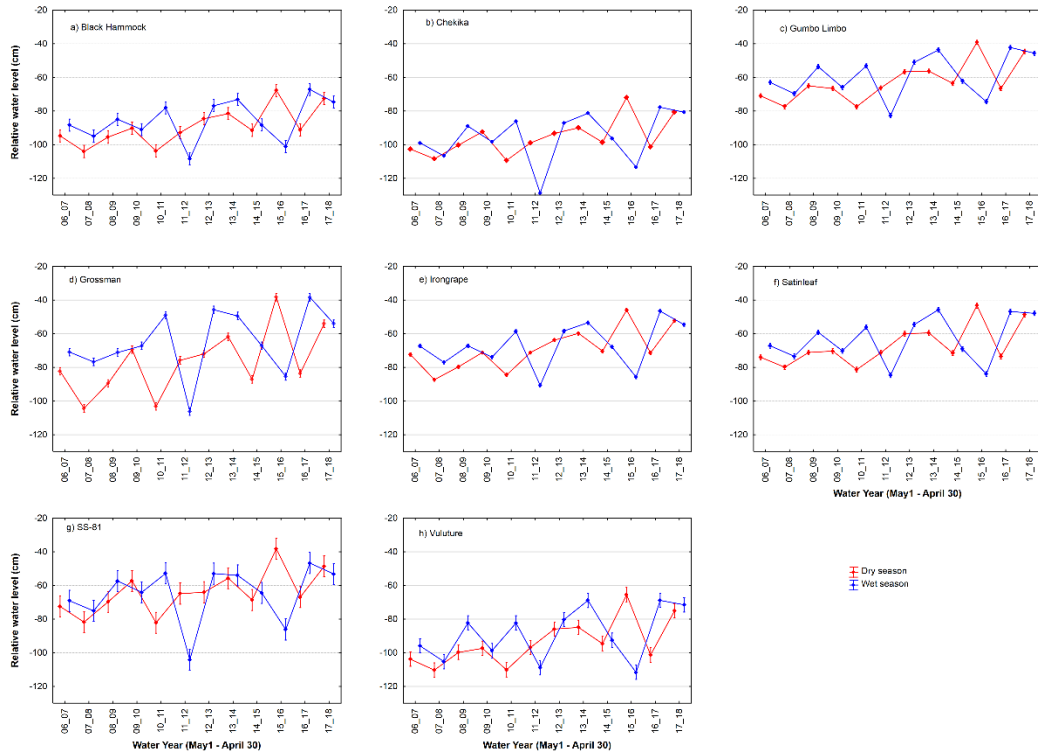


Figure 1.4: Seasonal mean (\pm SE) relative water level (RWL) in the hardwood hammock plots on eight tree islands, (a) Black Hammock, (b) Chekika, (c) Gumbo Limbo, (d) Grossman, (e) Irongrape, (f) Satinleaf, (g) SS-81, and h) Vulture.

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 were the wettest months in several decades (Abteu and Ciuca 2017). Much higher than normal dry season rainfall in Everglades basin, especially Water Conservation Areas, resulted in extreme water level in WCA-3A, prompting emergency operations for moving 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 flow through SRS and 40% through NESRS (Abteu and Ciuca 2017). This emergency operation period has 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, the stage recorder, P33 for which a long-term rain data are available on DBHYDRO (<https://www.sfwmd.gov/science-data/dbhydro>) data resource was used. A correlation between annual total rainfall at P33 between 2007 and 2017 and mean RWL on each of these for islands was insignificant (Figure 1.5; $p\text{-value} > 0.05$), because the hydrologic conditions in these islands partly depend on the water delivery from the adjacent WCA 3A and 3B. Thus, rain driven water delivery operational plans as envisioned in the Combined Operation Plan (COP) might help to provide natural hydrological conditions in these islands.

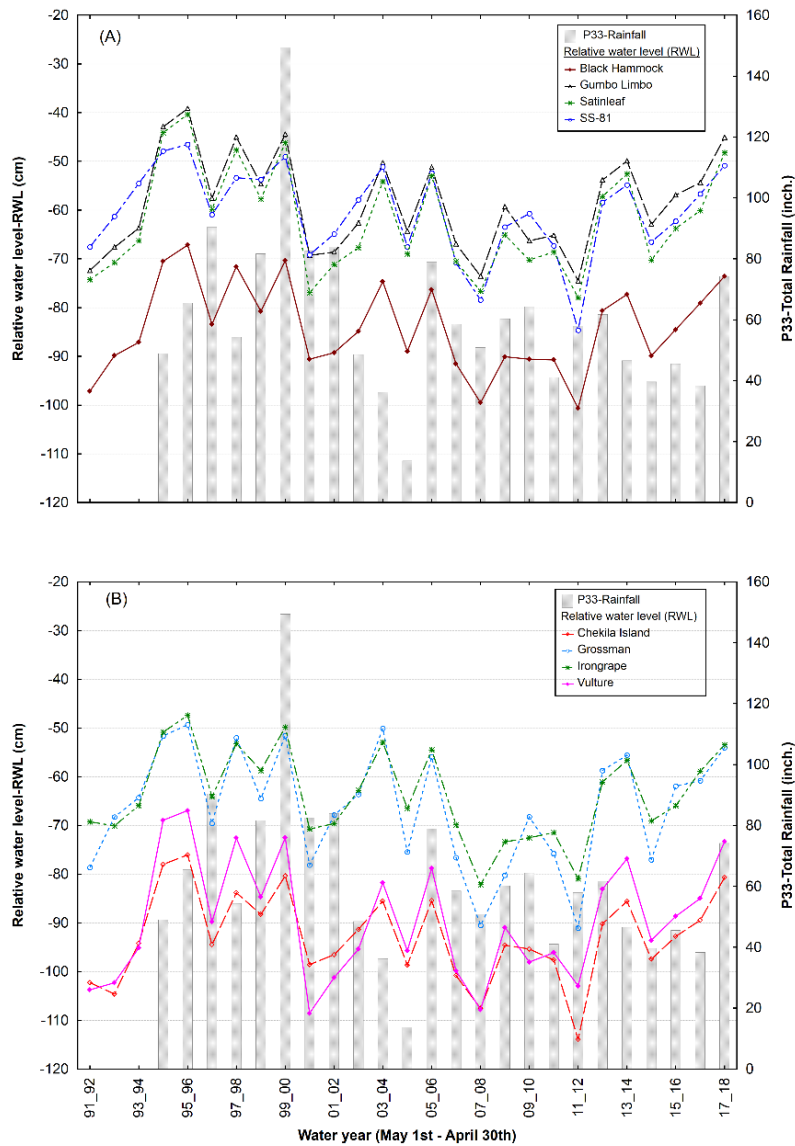


Figure 1.5: Relationship between annual total rainfall at P33 stage recorder and mean annual relative water level (RWL) in two groups of tree islands: (A) four tree islands sampled annually, and (B) four tree islands first sampled between 2006/07 and 2010/11, and then again in 2017/18.

1.3.1 Hurricane damage

In September 2017, Hurricane Irma, a category 4 storm hurricane hit the southwest coast of Florida. However, its impact was felt in most of south Florida. An analysis of 2017 (WY 2017/18) tree data has revealed varying degree of damage to the tree layer vegetation in eight tree islands for which we had pre-Irma data (Figure 1.6).

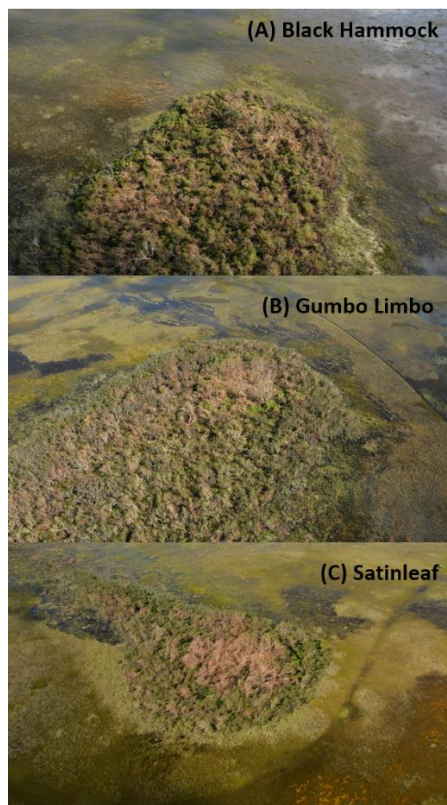


Figure 1.6: Photo of three tree islands (Black Hammock, Gumbo Limbo and Satinleaf) showing the damage from Hurricane Irma. (Photo credit – Jed Redwine, NPS)

In eight islands that we sampled, 68.9 % of trees sustained one or other form of damage from the Hurricane Irma. Percent of trees damaged from the hurricane was the higher (>75%) in Chekika Island, Gumbo Limbo, Irongrape and Vulture than in other four islands (Figure 1.7). The cause of damage by wind varied among tree islands. Black Hammock had the highest percent (11.8%) of uprooted trees, whereas other island had the trees damaged mostly due to broken branches (Table 1.2).

In the study islands, not all species were equally susceptible to the damage by hurricane (Table 1.3). For instance, all individuals (100%) of three less abundant species (1-3 trees), such

as Spicewood, (*Calyptrotrichus pallens*), Willow-bustic (*Sideroxylon salicifolium*), Paradise tree (*Simarouba glauca*) had broken branches. Among the most abundant species, while much higher percent (>85%) of White stopper (*Eugenia axillaris*) and Wild mastic (*Sideroxylon foetidissimum*) sustained some kind of damage than any other species (Table 1.3), only half of Gumbo limbo, (*Bursera simaruba*), Sugarberry (*Celtis laevigata*) and Satinleaf (*Chrysophyllum oliviforme*) tree were damaged.

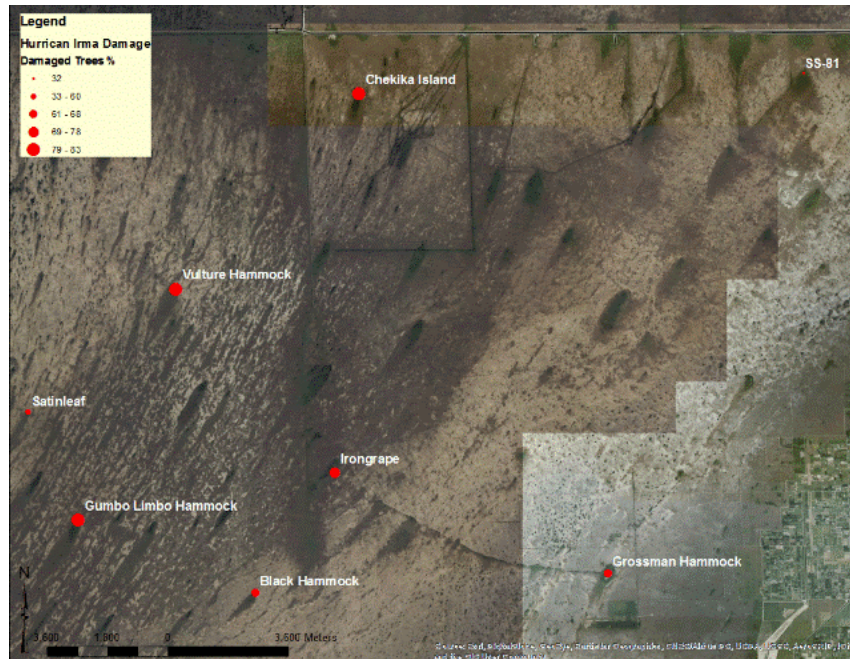


Figure 1.7: Tree damage percent on eight tree islands sampled three months after Hurricane Irma in WY 2017/18.

Table 1.2: Types of tree damage by Hurricane Irma on eight tree islands sampled three months after the hurricane in WY 2017/18.

Tree Island	Percent of damaged trees			
	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: Plant species that sustained different types of tree damage by Hurricane Irma on eight tree islands sampled three months after the hurricane in WY 2017/18.

Tree species	Number of Total Trees	Percent of damaged trees			
		Uprooted	Broken Trunk	Broken Branch	Defoliated
<i>Annona glabra</i>	4	0.0	0.0	0.0	0.0
<i>Ardisia escallonioides</i>	3	0.0	0.0	66.7	0.0
<i>Bursera simaruba</i>	149	5.4	14.1	28.9	2.0
<i>Calypttranthes pallens</i>	1	0.0	0.0	100.0	0.0
<i>Celtis laevigata</i>	106	2.8	7.5	32.1	9.4
<i>Chrysobalanus icaco</i>	29	3.4	17.2	17.2	20.7
<i>Chrysophyllum oliviforme</i>	42	0.0	7.1	42.9	0.0
<i>Coccoloba diversifolia</i>	50	6.0	10.0	50.0	4.0
<i>Eugenia axillaris</i>	204	8.3	21.1	18.1	39.7
<i>Ficus aurea</i>	11	0.0	18.2	36.4	0.0
<i>Myrsine floridana</i>	3	0.0	66.7	0.0	0.0
<i>Nectandra coriacea</i>	24	16.7	12.5	45.8	4.2
<i>Sambucus canadensis</i>	1	0.0	0.0	0.0	0.0
<i>Schinus terebinthifolius</i>	14	7.1	0.0	21.4	0.0
<i>Sideroxylon foetidissimum</i>	55	0.0	25.5	56.4	12.7
<i>Sideroxylon salicifolium</i>	1	0.0	0.0	100.0	0.0
<i>Simarouba glauca</i>	3	0.0	0.0	100.0	0.0
<i>Zanthoxylum fagara</i>	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. Larger trees were likely to have more broken branches, whereas percent of defoliation was higher in smaller trees (Figure 1.8). However, tree size did not have significant effects on uprooting or the breakage of main trunk.

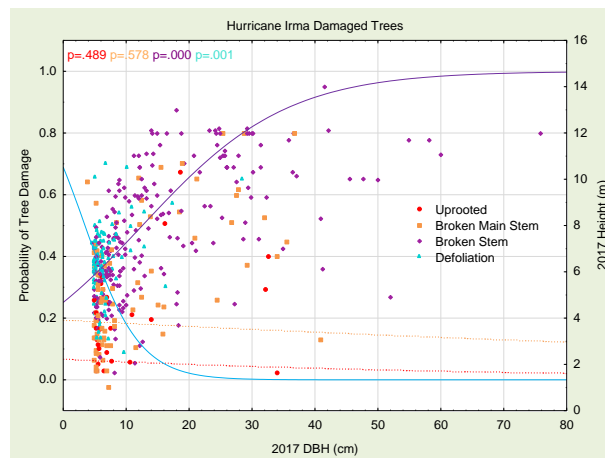


Figure 1.8: Probability of hurricane damage based on tree DBH.

1.3.2 Tree mortality and in-growth

Tree density and basal area are the functions 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 in 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 in-growth was 104 trees $\text{ha}^{-1} \text{year}^{-1}$ whereas tree mortality was 44 trees $\text{ha}^{-1} \text{year}^{-1}$. In-growth on some islands was higher also because of recovery from Hurricane Wilma in 2005. In subsequent years, i.e. between 2011 (WY 2011/12) and 2016 (WY 2016/17), when hammocks on only four islands (Black hammock, Gumbo Limbo, Satinleaf, and SS-81) were annually monitored, the both mean tree in-growth and mortality showed little variation, except in SRS islands in 2014. 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} .

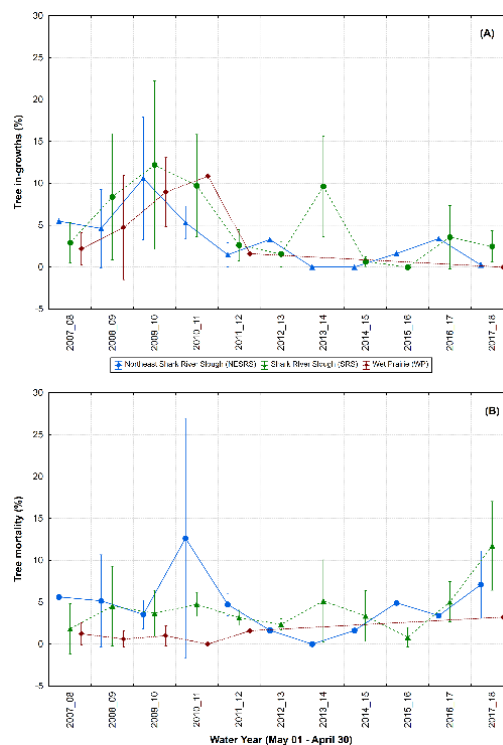


Figure 1.9: Annual mean (\pm) tree in-growth (A) and mortality (B) on the tree islands monitored in Shark River Slough (SRS) and Northeast Shark River Slough (NESRS) within the Everglades National Park between 2007 and 2017. The number of tree islands studied varied among years. Since 2011, hardwood hammocks on four islands (Black Hammock, Gumbo Limbo, Satinleaf and SS-81) were studied.

In some of the studied tree islands, tree mortality drastically increased in 2017/18, mostly due to damage caused by Hurricane Irma. After the hurricane in WY 2017/18, vegetation on four (Chekika, Grossman, Irongrape and Vulture Hammocks) additional tree islands, all from the same network of 16 islands within ENP (Ruiz et al. 2011), were sampled. In five of eight islands, tree mortality was higher in 2017/18 than in previous years (Figure 1.10), for which tree mortality data were available. In 2017/18, higher tree mortality was observed in Black Hammock, Chekika, Grossman, Irongrape, Satinleaf and SS-81.

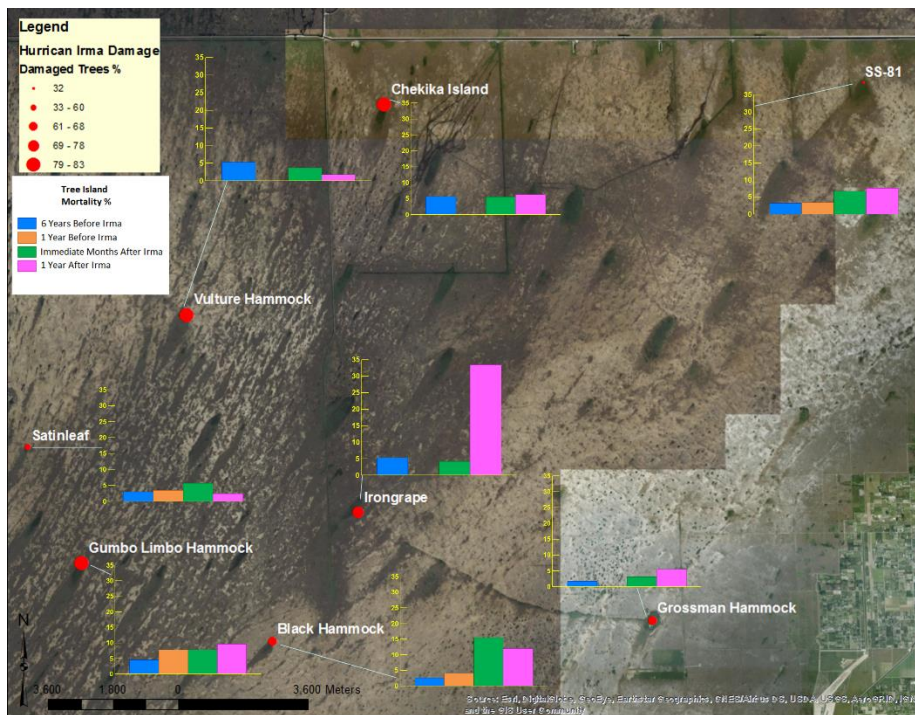


Figure 1.10: Annual mean tree mortality (%) on eight tree islands before and after Hurricane Irma. On four islands, tree mortality data were available for only 2010/11 and 2011/12.

In concurrence with the trend in tree mortality and in-growths, total tree basal area on four islands slightly decreased in 5 years, between 2011 and 2017 (Figure 1.11), though the change in BA was not statistically significant. The lowest mean value of BA in WY 2015/6 was because only three tree islands were sampled in that year.

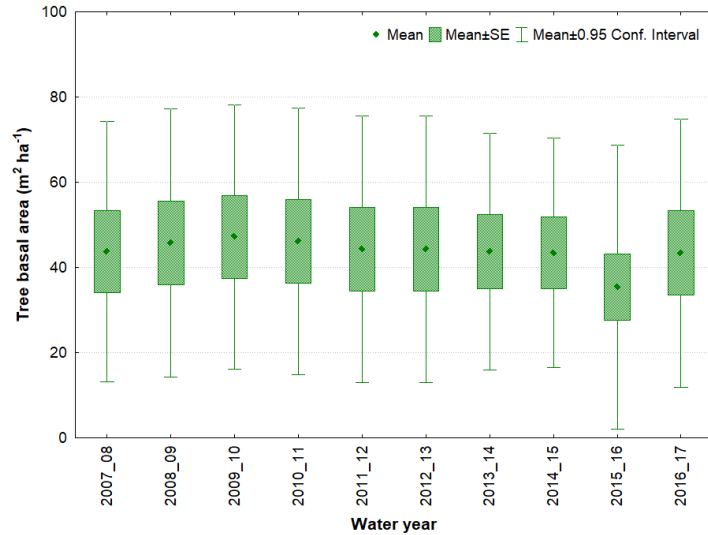


Figure 1.11: Box-plot showing the trend in tree basal on four tree islands monitored within the Everglades National Park between 2007 (WY 2007/08) and 2017 (WY 2017/18). In WY 2015/16, tree basal area was low, as only three tree islands were sampled.

As reflected by variation in annual mean tree mortality and in-growth, 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). Hardwood hammocks have primarily flood-intolerant species. Water level above or near the ground surface for longer periods, especially in the dry season, adversely impacts the survival and growth of those tree species (Stoffella et al. 2011). During the 2016 (WY 2015/16) dry season, water level in the SRS tree islands was very close (<40 cm) to the ground surface for a longer period (Figure 1.5), which may have affected tree growth and mortality in subsequent years.

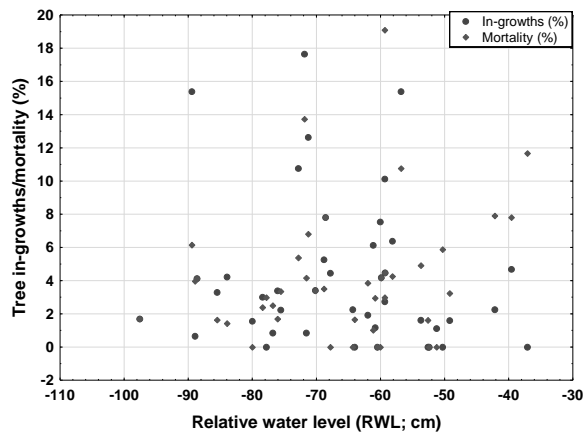


Figure 1.12: Scatter-plot showing the relationship between relative water level and tree in-growth and mortality on four tree islands monitored within the Everglades National Park between 2007 (WY 2007/08) and 2017 (WY 2017/18).

1.3.3 Tree layer vegetation dynamics

Among the eight islands sampled in 2017/2018, tree layer vegetation composition in Grossman Hammock and SS-81, located within MP landscape and NESRS, respectively, was quite different from rest of SRS tree islands. A nonparametric multi-dimensional scaling (NMDS) ordination, based on tree species' IVI and B-C similarity, revealed that in the hammocks on these eight islands, tree species composition, as represented by species' important value index (IVI), has somewhat changed over time. The changes were obvious on three tree islands that have been monitored since 2002. In these three tree islands, the species composition in 2002/2003 was slightly different from the composition in recent years (Figure 1.13). For instance, over the sampling period, in Black Hammock, the IVI of Sugarberry (*Celtis laevigata*) decreased, whereas the IVI of White stopper (*Eugenia axillaris*) significantly increased. Likewise, the IVI values of Wild mastic (*Sideroxylon foetidissimum*) and Satinleaf (*Chrysophyllum oliviforme*) in Gumbo Limbo and Satinleaf hammock, respectively, doubled in 15 years (Figure 1.14). In these islands, however, there has not been much shift in species composition in recent years (Water Years: 2010-2017), until after the Hurricane Irma. A minimal shift in species composition in the tree layer was also observed in the hammock of SS-81, which has been annually monitored since WY 2008.

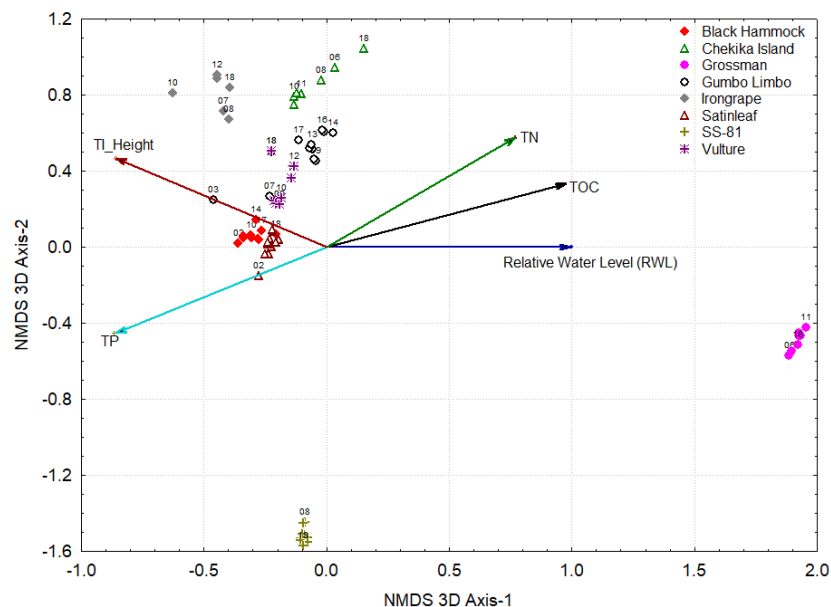


Figure 1.13: Scatter-plot of 3-D NMDS ordination based on tree species IVI in eight tree island hammocks sampled between Water Year (WY) 2002 and 2017. Fitted vectors are relative water level (RWL), tree island height (TL_Ht), soil nitrogen (TN), soil phosphorus (TP) and total organic carbon (TOC).

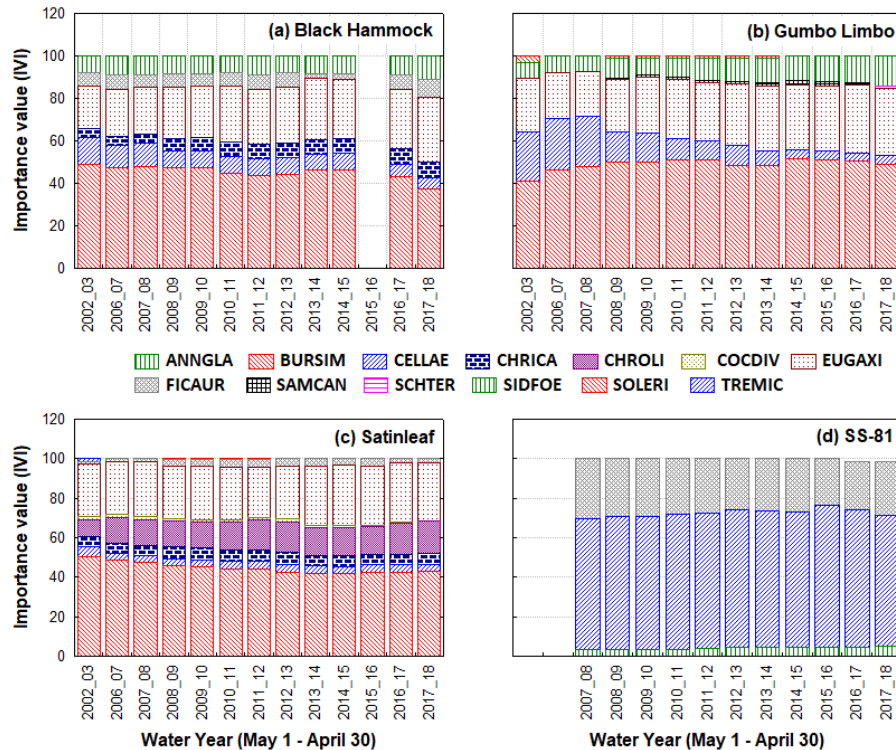


Figure 1.14: Importance value index (IVI) of tree species in hardwood hammocks of four tree islands. ANNGLA = *Annona glabra*; BURSIM = *Bursera simaruba*; CELLAE = *Celtis laevigata*; CHRICA = *Chrysobalanus icaco*; CHROLI = *Chrysophyllum oliviforme*; COCDIV = *Coccoloba diversifolia*; EUGAXI = *Eugenia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

In general, hydrology is the major driver of differences in species composition among various plant communities arranged along topographic gradient within a tree island (Armentano et al., 2002; Wetzel 2002; Ross and Jones, 2004; Espinar et al., 2011; Sah et al. 2018). However, in the hardwood hammocks which rarely get flooded, and mean annual water table is often below 40 cm (Figure 1.5), tree species composition is probably more the legacy of long-term interaction between hydrology and other physical processes, including recurrent disturbances. In some of these islands, high tree mortality was observed until 3-4 years after hurricane Wilma in 2005. In those hammocks, higher mortality in the 3rd and 4th than in the 1st and 2nd post-hurricane years was attributed to the interaction of multiple disturbances, e.g., hurricane and drought (Ruiz et al. 2011). In September 2017, Hurricane Irma, a category 4 storm hurricane hit the southwest coast of Florida. However, its impact was felt in most of south Florida. 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). After Hurricane Irma, the IVI of

Celtis laevigata in Chekika island was 1/4th of the its IVI six years ago in 2012 (Figure 1.15). Similarly, in Vulture Hammock, IVI was 33% less in 2017, after the hurricane than in 2012. In contrast, White stopper (*Eugenia axillaris*) increased on this island.

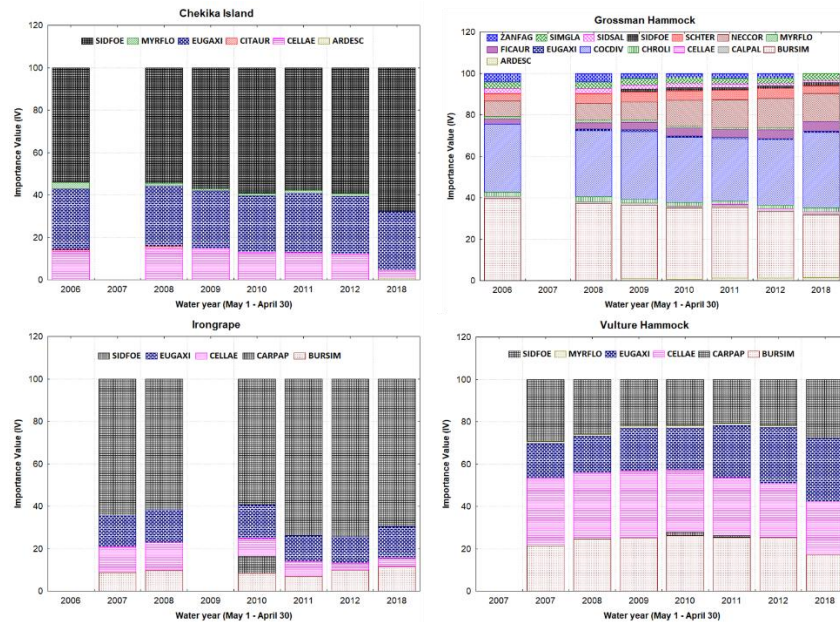


Figure 1.15: Importance value index (IVI) of tree species in hardwood hammocks of four tree islands. ARDESC =; BURSIM = *Bursera simaruba*; CALPAL = *Calypttranthes pallens*; CARPAP = *Carica papaya*; CELLAE = *Celtis laevigata*; CHROLI = *Chrysophyllum oliviforme*; CITAU = *Citrus aurantifolia*; COCDIV = *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*.

1.4 Summary

In the hardwood hammocks on SRS tree islands between 2007 and 2017 (WY2008 and 2018), the annual mean RWL remained well below the soil surface. Thus, slight increases in marsh hydroperiod or water depth in Everglades National Park are unlikely to have any significant impact on tropical hardwood hammock communities. However, an incremental upward shift in the RWL could cause a shift in species composition and productivity of these islands. In contrast, over the period of this study, such a shift in vegetation in response to hydrologic change commonly occurred in the wetter communities (Bayhead and Bayhead swamp) (Sah et al. 2018). In the four islands that were annually monitored during this period, bayhead and bayhead swamp plots were sampled in WY 2018/19, and the data analysis is in progress and results which will include more comprehensive response of tree island vegetation to hydrologic changes in these years, will be described in Year-5 data. Moreover, most of south

Florida, including these tree islands felt the impact of Hurricane Irma, a category 4 storm hurricane hit the southwest coast of Florida. An analysis of 2017 (WY 2017/18) tree data has revealed severe damage to the tree layer vegetation in eight tree islands for which we had pre-Irma data. While our results suggest that tree mortality after the hurricane was higher than the background mortality in the previous years. Moreover, larger trees were likely to have more broken branches, however, tree size did not have significant effects on uprooting or the breakage of main trunk. While there has been a slight shift in tree layer vegetation after hurricane, the more detailed account of the effects of Hurricane Irma on delayed mortality and vegetation composition on those islands will be described in the next (Year-5) report.

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

2.1 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), depending on the availability of resources such as soil water, nutrients, and light (Small and 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 and 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 and Antos 1981; Sagers and 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; Sklar and 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 and 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 comprised 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 ground layer (also called, ‘understory’) vegetation (Figure 2.1). 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 2.2).

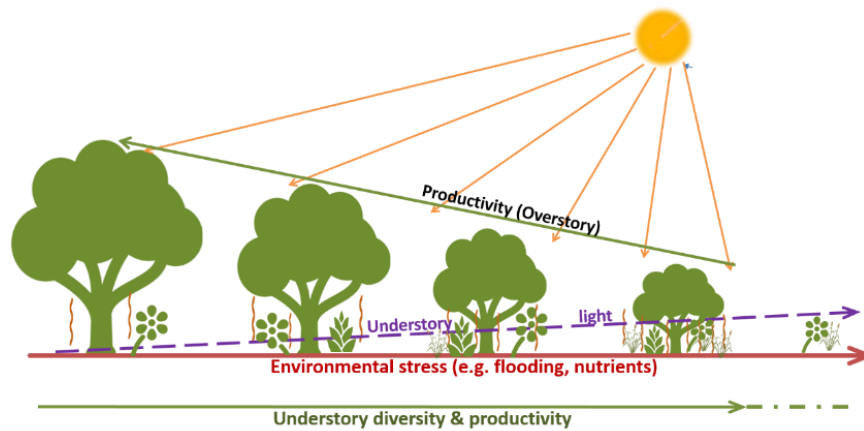


Figure 2.1: Schematic diagram representing the variation in tree height, overstory productivity, understory light and vegetation characteristics (diversity and productivity) along hydrology gradient.

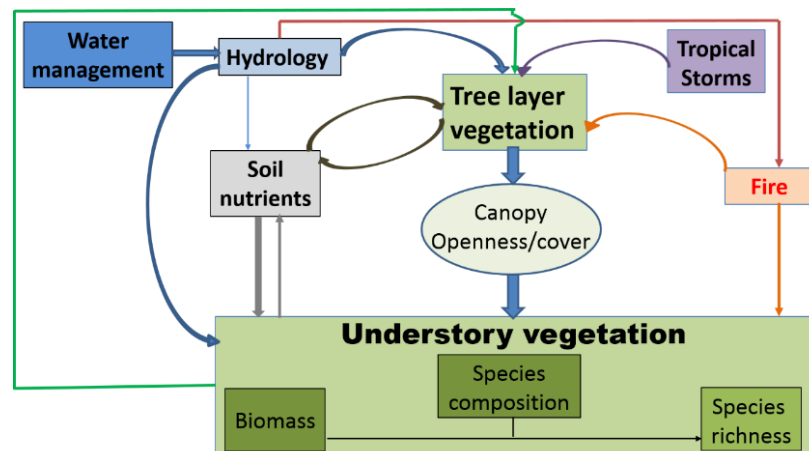


Figure 2.2: Conceptual model showing the relationship among environmental factors, tree layer vegetation, and understory vegetation characteristics.

In the Everglades, hydrologic conditions have been greatly modified by implementation of a complex water management system (Light and 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 2.2). 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 especially so for hammocks 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.

2.2 Methods

2.2.1 Study area

The study area included a subset of tree islands within Everglades National Park (ENP) and Water Conservation Areas 3A and 3B (WCA3A and WCA3B). 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 2.3). The R&S tree islands are mostly tear drop-shaped, oriented parallel to the direct 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.

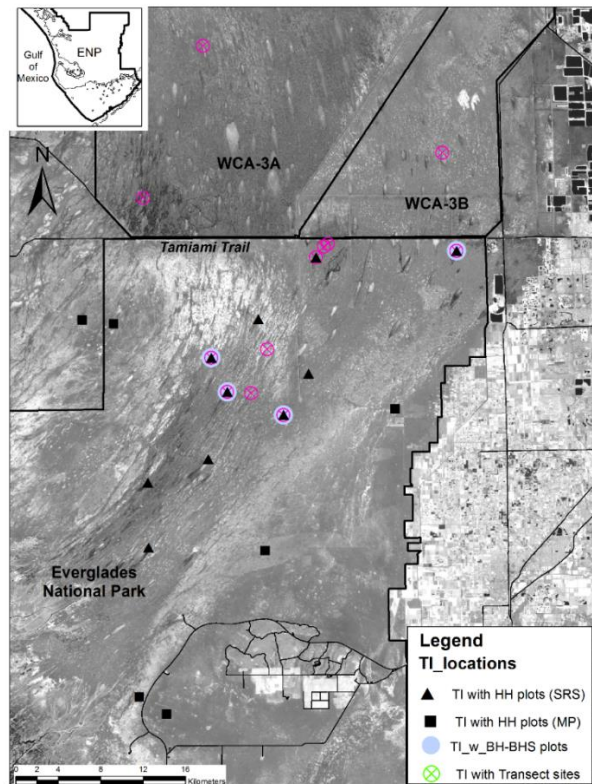


Figure 2.3: Location map of tree islands on which both overstory and understory vegetation was sampled in permanent plots and/or in a series of plots on transects.

The forest flora in both hammock types are comprised mostly of 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 and Sah 2011). The mineral soils in the ridge-slough hammocks are not deep (usually < 1 m), but so rich in phosphorous 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 (Ross et al. 2006; Espinar et al. 2011).

2.2.2 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 2005-2006. 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 2.3).

2.2.2.1 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) and Sah et al. (2018). Soil characteristics, including soil depth and soil nutrient parameters, were determined. Soil analysis methods are described in detail in Ross and 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 a 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).

2.2.2.2 Vegetation sampling on transects

Vegetation sampling along transects were done in three stages (Table 2.1). 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 2015-2017, 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 2.3). 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.

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).

Table 2.1: Tree island location and summary of transect sites on 12 islands on which vegetation was surveyed along transects between WY 2001 and 2017.

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 2017					
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

2.2.3 Data Analysis

2.2.3.1 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 Jan 1, 2000 to Dec, 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 and 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.

2.2.3.2 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: $IV = (\text{Relative frequency} + \text{Relative cover})/2$. 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 partialing 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 our next report.

2.3 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* (CHRIC)) 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 2.4). 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/m² (plot) in bayhead swamp forest, bayhead forest and hardwood hammock, respectively.

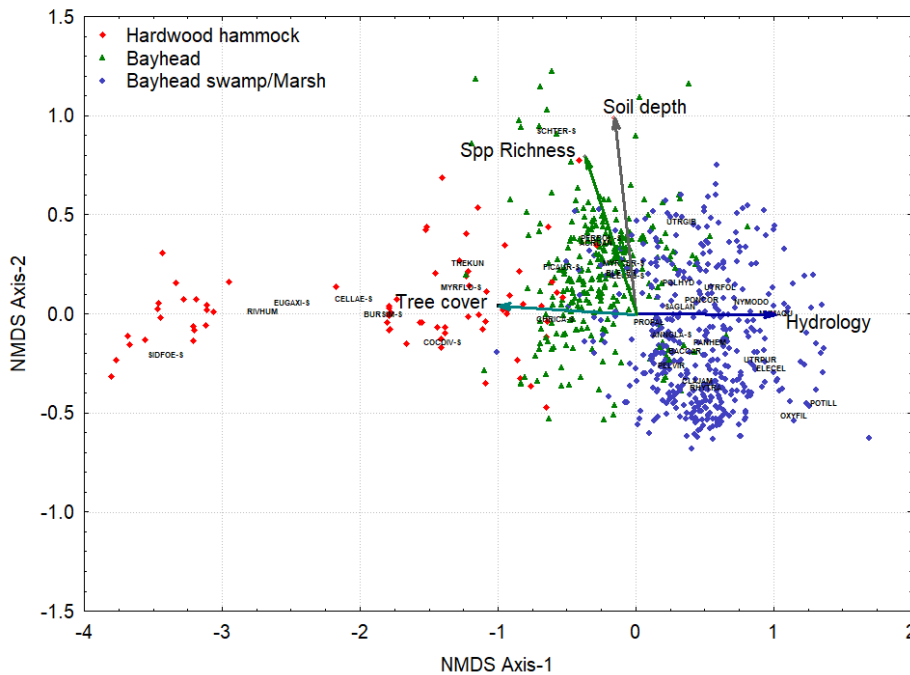


Figure 2.4: Plots of axis scores derived from understory species' cover-based three-dimensional non-metric multidimensional ordination (NMDS) of 805 plots sampled on transects in 12 tree islands. Fitted environmental and species richness vector represent the direction of maximum correlation between the 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 2.2), 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 ($r = 0.427$, $p < 0.001$).

Table 2.2: Correlation (r) and statistical significance of fitted community (species richness) and environmental vectors with species cover-based 3-dimensional ordination configuration.

Variable	n	r	p
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

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 Wild Mastic, *Sideroxylon foetidissimum* (SIDFOE)), shrub (Rougeplant, *Rivina humilis* (RIVHUM)), and ferns (Southern shield fern, *Thelypteris kunthii* (THEKUN); Sword fern, *Nephrolepis exaltata* (NEPEXA)) (Figure 2.4). 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 2.4). 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 2.5). 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 2.3).

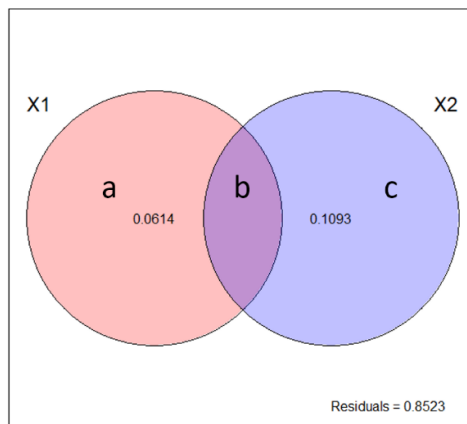


Figure 2.5: Ven diagram of variance partitioning of understory vegetation composition explained by a group of environmental variables (tree cover, TreeCov. and relative water level, RWL) and spatial variable (distance along transect). The details of symbols X1, X2, a, b and c are in Table 2.3.

Table 2.3: 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 relative water level (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 slightly higher than 4.37% of variance explained by RWL (Figure 2.6; Table 2.4). 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.

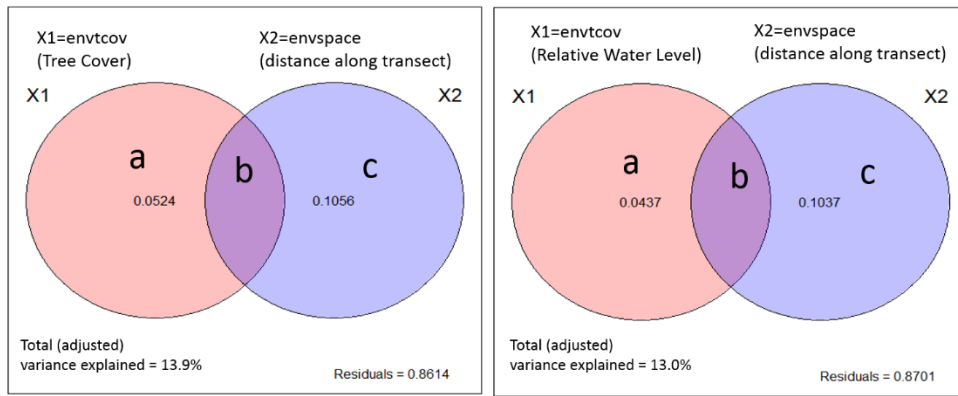


Figure 2.6: Ven diagrams of variance partitioning of understory vegetation composition explained by two subsets of environmental variables (tree canopy cover and relative water level) in combination with spatial variable (distance along transect). The details of symbols X1, X2, a, b and c are in Table 2.4.

Table 2.4: 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 spatial variable (distance along transect).

	df	Env. variable: Tree cover (TreeCov)		Env. variable: Rel. water level (RWL)	
		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	

2.4 Discussions

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; Sah et al. 2018), even though seedlings of some tree species may have higher hydroperiod optima than mature trees of the same species (Ross and Jones 2004; 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 was most common on the local mounds, following a flood avoidance strategy noted for seedlings of other tree species in the bayhead forests (Reed 2006). Beside overstory tree composition, occurrence and abundance of tree seedlings in the understory also varies among seasons (Sah 2004; Reed 2006). 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 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 and 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 and Lyon 1997).

The relationship between overstory and understory vegetation is also shaped by their differential responses to disturbance (Sagers and Lyon 1997; Tobisch and Standovar 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 year after Hurricane Irma are being analyzed, and detailed results will be included in the Year-5 report.

2.5 Summary

In summary, understory vegetation composition in the southern Everglades tree islands varies along hydrologic 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. Since 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.

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3. Plant Community Detection on Tree Islands in WCA3B from multi-spectral World View 2 and G-LiHT LiDAR data

- Ximena Mesa, Paulo Olivas and Daniel Gann

3.1 Introduction

Tree islands are essential and intricate components of the Everglades ecosystem. Plant communities in tree islands are arranged along hydrologic and nutrient gradients. 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. 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, accurate and detailed mapping techniques are needed that can detect clearly defined plant communities at a resolution that represents the scales at which change occurs along hydrologic and nutrient gradients.

The objective of developing plant community detection algorithms based on spectral reflectance patterns and vegetation height was to determine if signature extension across tree islands was possible. The rationale for this methodology is that similar environmental conditions (including hydrology) lead to quasi identical or similar plant communities across the larger landscape, and that the communities have comparable spectral reflectance patterns of electromagnetic radiation. As a result, locally extracted spectral signatures of communities can then be used to successfully map greater spatial extents. However, if the spectral signatures of some plant communities resemble each other then, minor differences in island-specific signatures for the same classes are more likely to be confused. For instance, the reflectance patterns of individuals of tree or shrub species that are shorter than 5 m (shrubs) are expected to closely resemble the spectral signatures of those that are at least 5 m tall. High confusion between tree and shrub community classes was observed for single season classifiers in the first two phases of this project. Using bi-seasonal spectral data improved the classifier in cases where the dry or wet season spectral signatures varied in their differences between seasons for community classes (Gann and Ogorcak 2016). The largest confusion of shrubs and trees with the same phenological cycle, however cannot be resolved with bi-seasonal data only. Hence, to further reduce classification error between trees and shrubs and all other classes we included vegetation height

estimates derived from Light Detection and Ranging (LiDAR) data by subtracting the last LiDAR return (expected ground) from the first return (expected top of canopy)(Gann et al. 2017).

The main objective for this report was to test the fully implemented and refined detection method and algorithms to map community classes of three tree islands within Water Conservation Area 3B (WCA 3B). Tree islands in WCA 3B were expected to differ from those within ENP because of their different hydrological management legacies.

3.2 Methods

3.2.1 Study Area

This study considered three tree islands located within 1 km of each other within WCA 3B: WCA3B-24, tree island head located at N 25.843 W 80.631 (WGS 84), WCA3B-07 located at N 25.823 W 80.626, and WCA3B-25 located at N 25.809 W 80.626 (Figure 3.1). The three islands were selected to be located within the footprint of satellite images acquired on the same day, and that LiDAR data was available within a few years of the spectral data acquisition. Using images acquired on the same day assures that spectral differences across islands are a result of differences in plant communities and spectral variability of communities and not seasonal effects associated to phenological cycles of the vegetation or variability in atmospheric conditions. Ideally, the LiDAR data should also be collected at the same time to ensure that the vegetation height extracted from the data reflects the vegetation type present.

3.2.2 Classification Scheme

After initial reconnaissance field visits, a classification schema with 10 vegetation classes was developed (Table 3.1) to represent tree island and marsh community types located within a 200 m buffer around the approximate boundaries of the three islands. Tree island vegetation classes included bayhead forest (tB) with trees or woody species, bayhead swamp representing shrubs, or woody species less than 5 m tall (sB), and *Salix caroliniana* (willow) dominated shrub (sSa). Marsh classes were divided into single-species dominated and mixed classes. 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 (bIFNy), and a *Typha domingensis* (cattail) dominated class (gMTy). Three non-dominant

mixed classes included a generic sparse graminoid class with mixes of *Eleocharis*, *Panicum*, or *Rhynchospora* species typically found in deeper and longer hydro-period marshes or sloughs (gM__S); and two mixed broadleaved emergent classes with low shrubs, typically found in tree island tails and edges. These classes often had *Cephalanthus occidentalis* and frequently included fern species (bIE_S), we further differentiated this class for strong graminoid presence (typically sawgrass) (gM_bIE_s) (Table 3.1).

3.2.3 Data Processing

For its high spatial and spectral resolution properties, we chose the WorldView 2 (WV2) sensor to detect vegetation communities. Since previous studies within the larger Everglades (Gann, Richards, & Biswas, 2012; Gann, Richards, Lee, & Gaiser, 2015; Gann & Richards, 2009) including the first two reports of this project showed that bi-seasonal data increased mapping accuracies significantly, two images with minimal cloud cover containing all three islands were obtained. Optimally the two dates are far enough apart to capture the highest variability of the phenological cycles of the vegetation, but the only two nearly cloud-free images available were acquired in mid-January of 2017 (01/19/2017) and mid-February of 2018 (02/16/2018), both in the wet condition period for that part of the system. We still included both images in the analysis, since they were acquired one year apart, and seasonality and hydrological conditions vary with year. Images were first geometrically, then radiometrically calibrated and atmospherically corrected in ENVI (Exelis Visual Information Solutions, 2013). Atmospheric correction for each image was completed using the FLAASH module in ENVI (ENVI, 2009). 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 Development Core Team & 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 the 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, respectively and gridded at 1 m spatial resolution. This NASA product was then 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.

3.2.4 Island and Training Point Digitization

The approximate boundaries of each tree island were digitized from 2009 color infrared aerial imagery (CIR) imagery at a resolution of 1:1,000. A buffer of 200 m was added to each island to include a hydrological gradient from tree island to surrounding marsh communities. Outlines of clouds, cloud shadows, and man-made structures overlapping the tree island and buffer areas were digitized in a separate mask layer. The three islands included four small boardwalks, which were also included in the digitized mask layer. The areas masked by man-made structures were classified as missing data, while the areas covered with clouds were classified using the corresponding cloud free image of the image pair.

Training points for each vegetation class were added iteratively using a combination of field surveys, both aerial surveys by helicopter and ground surveys, and high-resolution aerial photography in digital stereo view (DATEM Summit). 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). The 2009 CIR imagery was also used to identify and digitize training samples for all vegetation classes for training purposes. Training points were not located in areas of digitized clouds, cloud shadows, or man-made structures or shadows. In total, 2,143 training points were digitized on WCA3B-24, 1,545 on WCA3B-07 and 951 on WCA3B-25.

3.2.5 Spectral Signature Evaluation

For all training samples, signatures were extracted from the 51-layer data cube and the LiDAR derived CHM. A random forest classifier was trained for all community types with consideration of all training samples across all three tree islands. Data analysis was performed

with Microsoft R Open (MRO) version 3.5.1 ((R Development Core Team & R Core Team, 2018). For classification we used the unifying modeling framework of the ‘caret’ package (Kuhn et al., 2015), using its ‘rf’ function for random forest. We determined from test runs that 1,000 decision trees was a safe threshold beyond which no significant increase in model-based classification accuracy was observed ($\alpha = 0.05$) (Kuhn et al., 2015). The parameter for the optimal number of random variables selected at each split (“mtry”) was established for each random forest model through build-in tuning routines.

3.2.6 Model Selection

Although, we expected that based on previous research models using bi-seasonal data would produce models with higher accuracies (Gann, Richards, & Biswas, 2012; Gann, Richards, Lee, & Gaiser, 2015; Gann & Richards, 2009), we evaluated four models constructed using: 1) January 2017 image only, 2) February 2018 image only, 3) bi-annual, and 4) bi-annual and LiDAR. From 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 was established on a community basis ranging from 4 m² for *Salix* shrubs to 16 m² for all other classes (Table 3.1). The MMU was enforced by iteratively clumping and sieving classes to replace small patch values with neighboring class values. This eliminates small patch noise. The iterative filtering function was coded in R using clump and sieve functions package ‘raster’ (Hijmans & van Etten, 2010). The clumps were generated taking into account 8 adjacent cells.

3.2.7 User-Based Accuracy Assessment

To evaluate the accuracy of the final map we conducted a design-based accuracy assessment based on a stratified random sample design. We calculated the number of samples required for each class assuming a multinomial distribution of errors for a desired map accuracy confidence of 95% with a 5% precision of the estimate (Congalton & Green, 1999). A total number of 730 samples was drawn equally distributed across all MMU filtered classes (73 each). Pixel centroids selected for accuracy assessment were greater than 1 m away from training pixels (no overlap). Each of the samples was evaluated from aerial photography in stereo view at a fixed zoom of 1:500 and a class label was assigned. Confusion matrices were constructed from predicted and references 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 (Equations based on Olofsson et al., 2014; Olofsson, Foody, Stehman, & Woodcock, 2013; 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 coded in R (R Development Core Team & R Core Team, 2018).

3.3 Results and Discussion

3.3.1 Model selection

Similar to the findings of previous research, single image models (January 2017 image: 70% and February 2018 image: 73%) presented lower accuracy than models that used bi-annual data (78%) and bi-annual and LiDAR (84%) (Table 3.2). The relatively low difference between single image results can be attributed to the lack of a true seasonal signature differences between acquisition days of the year. We found that the use of LiDAR data improved the classification of shrubs and trees, with a decrease of 25% in the omission error for shrub classes. The commission error of shrub classes was 12%.

The similarity in spectral signatures of trees and shrubs explains the strong increase in accuracy with the inclusion of LiDAR data. 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.

3.3.2 Map Accuracy

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 3.2). Overall map accuracy across all classes was $97.1 \pm 2.2\%$ (estimated standard error). The highest accuracy was achieved for broadleaf emergent species mixed with shrubs (blE_s) and bayhead forest, dominated by woody species with heights greater than 4 m (tB) reaching 100%. All other classes reached accuracies of 91% or greater.

3.3.3 Areal Coverage and Relative Abundance

For the islands and their surrounding marshes, sawgrass (gMCl) was the most abundant class, covering approximately 34% of the entire study area surface. Considering only the tree

and shrub classes, the most abundant class was the *Salix caroliniana* shrub (sSa) representing 35.1% of the study area, followed by bayhead shrub (sB) with 34.1 % cover. Bayhead forests (tB) were represented with 30.7 % (Table 3.3).

3.4 Conclusion

High overall accuracies suggest that WV2 satellite images provide data with characteristics suitable for detecting and mapping tree islands plant communities and their adjacent marshes. Random forest classifiers applied to the bi-seasonal and textural data were able to classify plant communities at high class-specific accuracies. 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 m for WV2 data, expansion or contraction of tree islands can be detected 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.

Hardwood hammocks occupied only very small areas in the heads of the three mapped islands, sometimes covering an area of approximately 4-12 m² (1-3 pixels), which makes this class difficult to map because the number of training samples is too small to train a classifier. To reliably detect hardwood hammock across large spatial extents and islands requires a significantly larger training sample set. The class representativeness of this class across the mapped extent is too small to be included in the classification scheme. Therefore, the hardwood hammock plant community in this project was grouped with the bayhead forest community, but the small hammock areas could be manually digitized, if necessary and if they are at least the size of the MMU of 16 m².

Confusion between the various graminoid classes not improving with the addition of LiDAR data was expected due to the fact that these species heights typically do not vary much. For these classes a LiDAR derived vegetation density estimate might be more suitable, but needs to be further explored. All classes might benefit from LiDAR derived canopy texture and/or density estimates. Results of this study suggest the potential to use this methodology at the landscape scale to monitor vegetation changes in tree islands in response to management decisions.

Tables and Figures (Section 3)

Table 3.1: Vegetation class codes, corresponding class descriptions and their minimum mapping units (MMU).

Class Code	Vegetation Class Description	MMU
bIE_s	Broadleaf emergent mixed with shrubs	4
bIFNy	Broadleaf floating mixed with nymphaea	4
gM_S	Sparse graminoid marsh	4
gM_bIE_s	Mixed shrub, graminoid, and emergent broadleaf, including ferns	4
gMCI	<i>Cladium jamaicense</i> dominated marsh	4
gMCI_S	Sparse <i>Cladium jamaicense</i> dominated marsh	4
gMTy	<i>Typha</i> dominated marsh	4
sB	Bayhead swamp, dominated by woody species with heights less than 4 m	4
sSa	<i>Salix caroliniana</i> shrub	2
tB	Bayhead forest, dominated by woody species at least 4 m tall	4

Table 3.2: Comparison of all model-based accuracy assessments in percent.

Vegetation Class	January, 2017(%)		February, 2018(%)		Bi-annual(%)		Bi-annual and LiDAR(%)	
	User's	Producer's	User's	Producer's	User's	Producer's	User's	Producer's
bIE_s	66.0	67.3	67.7	75.6	73.7	81.1	76.7	86.0
bIFNy	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_bIE_s	69.4	70.3	71.9	76.4	75.2	79.0	75.4	80.3
gMCI	70.9	81.3	72.1	78.5	80.4	85.9	80.5	85.5
gMCI_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(%)	70.7		73.2		78.3		84.6	

Table 3. Percent cover of classes of entire study area

Class Code	% Cover of all classes	% Cover of tree and shrub classes only
bIE_s	7.13	
bIFNy	3.06	
gM_S	6.38	
gM_bIE_s	15.24	
gMCI	33.75	
gMCI_S	9.06	
gMTy	14.82	
sB	3.61	34.10
sSa	3.71	35.10
tB	3.24	30.71



Figure 3.1: Location of WCA3B-24, WCA3B-07 and WCA3B25 tree islands

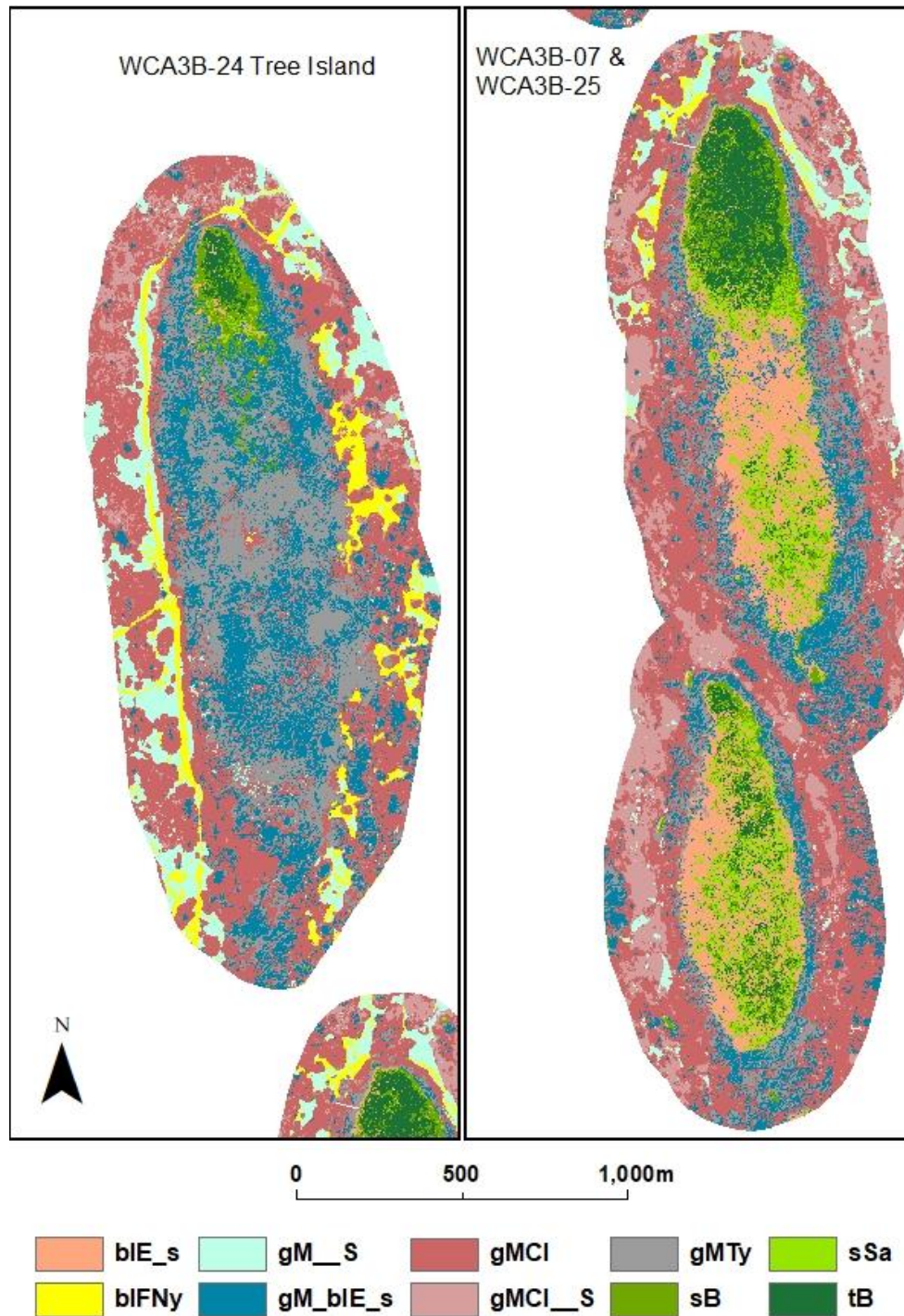


Figure 3.2: WCA3B-24, WCA3B-07 and WCA3B25 tree islands map classification. Class Codes: bIE_s = broadleaf emergent mixed with shrubs, bIFNy = broadleaf floating mixed with nymphaea, gM__S = sparse graminoid marsh, gMCI = *Cladium* dominated marsh, gMCL__S = sparse *Cladium* dominated marsh, gM_Typha = *Typha* dominated marsh, sB = bayhead shrub (swamp), sSa = *Salix caroliniana* shrub. tB = bayhead tree (forest).

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