Evaluating the functional response of isolated cypress domes to groundwater alteration in west-central Florida

by

Megan Kristine Bartholomew

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Approved by

Christopher Anderson, Chair, Professor of Wetland Ecology
Robert Boyd, Professor of Plant Ecology
Jacob Berkowitz, Research Soil Scientist

Abstract

The hydrology of a wetland is the single most important determinant of its function and slight alterations can lead to significant changes in plant communities and biogeochemistry within the wetland. Therefore, understanding the influence of hydrology on vegetative and soil processes is pivotal to restoration efforts. This study investigated how hydrologic alteration and recovery influenced wetland vegetation and soil processes in Starkey Wilderness Park (SWP), a well-field in west-central Florida. Vegetation responses to groundwater alterations were observed using long term species and hydrologic data collected from SWP. The results from the vegetation study suggest that hydrologic recovery has restored vegetative functions and measures, such as species richness and hydrophytic assemblages, in a relatively short (5-7 year) period. However, differences in species composition and community variation persist in wetlands of various degrees of hydrologic alterations. A field study was also conducted to determine how hydrologic alterations continue to affect wetland decomposition rates and other soil processes. After eight years of hydrologic recovery, altered wetlands experienced faster decomposition than reference wetlands and rates seemed to be linked to differences in both inundation and percent soil organic matter. The findings from this study suggest functional restoration of vegetation and soils should be determined on an individual wetland basis and over longer periods (>5 years). In some cases, overall restoration goals may need to be reassessed as ecosystem development progresses.

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CHAPTER ONE: INTRODUCTION

WETLAND DEFINITION, FUNCTIONS, AND SERVICES

Wetlands which are defined as ecosystems that depend on constant or recurrent, shallow inundation or saturation at or near the surface of the substrate (National Research Council 1995), are unique and often dynamic ecosystems that provide important ecological services (e.g., flood protection, carbon storage, water quality improvement, wildlife habitat, recreation, etc.) These ecosystem services (i.e. water quality improvement) derive from wetland functions (i.e. the retention and removal of dissolved substances) which result from wetland processes (i.e. microbial processes, National Research Council 1995). These processes and their manifestations result from the integration of three factors (wetland hydrology, hydrophytic vegetation, and hydric soils) and the interaction of those factors with the environment (National Research Council 1995). While hydric soil and hydrophytic vegetation are important components, experts agree that wetland hydrology represents the most influential factor affecting the overall wetland health and function. The degree to which the hydrology fluctuates has resounding impacts on both its vegetation and soil functions (De Steven et al. 2010). As a result, wetland hydrology plays a pivotal role in determining wetland type and process, and should guide restoration and management practices (Mitsch and Gosselink 2007, De Steven and Lowrance 2011, Foulquier et al. 2013). However, the importance of maintaining natural patterns of wetland hydrology was not historically recognized, resulting in substantial loss of wetlands and associated ecological functions and services in the United States (Mitsch and Gosselink 2007).

HISTORICAL WETLAND DEGRADATION

It has been estimated that, since European settlement, wetland degradation in the U.S. has progressed at an average rate of half a million hectares per year (Mitsch and Gosselink 2007). While wetland losses have decreased in recent years due to regulation (e.g., executive orders, "no net loss" policies, and Section 404 of the Clean Water Act), wetland functions continue to be lost and "no net loss" objectives remain elusive at local, regional, and national scales (Zedler 2000, Rain et al. 2013). This is especially true in the southeastern coastal plain, where extensive wetland loss continues despite the implementation of protective regulations (Mitsch and Gosselink 2007, Stedman and Dahl 2008, Rains et al. 2013). Rapid urbanization represents a significant driver of wetland degradation in the region (Kirkman et al. 2000, Mitsch and Gosselink 2007, Pittman and Waite 2009, McCauley et al. 2013). For instance, the population of the greater Tampa, Florida area has more than doubled, from 1.1 million to 2.4 million, in the last two decades (Metz 2011). This influx of people permanently changes the landscape and puts stress on natural resources including freshwater supply (Tampa Bay Water 2009, Metz 2011, Rains et al. 2013, Lewis et. al 2015).

In the Tampa Bay area, groundwater has been widely used for public supply. From the early 1930s to the mid-1990s, groundwater was the sole source of municipal water, with extraction occurring from eleven well-fields within the northern Tampa Bay area (Metz 2011). As the population grew in and around Tampa, so did reliance on groundwater, and pumping rates within the well-fields increased to meet public demand. This prolonged groundwater withdrawal negatively affected water levels in rivers, lakes, and wetlands, in and around the well-fields (Rochow 1994, Southwest Florida Water Management District 1996). During this time, roughly one-third of the wetlands in the Tampa Bay area were lost and those that remained experienced some degree of impact (Stedman and Dahl 2008, Rains et al. 2013).

STUDY AREA

J.B. Starkey Wilderness Park (SWP), located in New Port Richey, Florida, is one of the eleven well-fields in the Tampa Bay area. The park encompasses 3,200 hectares in the southwest corner of Pasco County and hosts a mosaic of ecosystems including pine flatwoods, sandhills, and various types of wetlands (Hutchinson 1984, Rochow 1998). There are 14 wells within SWP that began producing groundwater in the mid-1970s. At first pumping rates were moderate (3.79-22.71 million liters of water a day) and was focused in the western portion of the park. Pumping steadily increased and expanded to meet municipal demand, first to the central section of the park and then in the eastern section, until the mid-2000s pumping peaked at 49.21 million liters a day (Metz 2011). Significant environmental change has been observed in the park due to lower water tables associated with extended groundwater withdrawal (Hutchinson 1984, Southwest Florida Water Management District 1996).

Wetlands, especially isolated cypress domes, within the central and western sections of the park were visibly altered, vegetation shifted to upland and drier communities, cypress trees began to lean and die, and soil subsidence was observed

(Dooris et al. 1990, Metz 2011). However, the eastern-most wetlands remained largely unchanged, creating a gradient of hydrologically altered wetlands within the park (Rochow 1998). As ecosystem degradation due to groundwater withdrawal, became apparent, local agencies worked to reduce the area's reliance on groundwater and in 2008 alternative water sources were secured. That year withdrawals within SWP were reduced by nearly 75% from 41.64 million liters a day in 2007 to 15.14 million liters a day in 2008 (Metz 2011). Per regulatory requirements and as part of their well-field monitoring, Tampa Bay Water and Southwest Florida Water Management District collected wetland hydrology, vegetation, and soil data in isolated cypress domes within SWP since the 1970s (Southwest Florida Water Management District and Tampa Bay water 2005). Data from this long-term monitoring effort suggested that decreased pumping allowed for some recovery of the aquifer with partial restoration of wetland hydrology in this region, including some of the wetlands within SWP (Tampa Bay Water 2009, Metz 2011).

CYPRESS DOMES

Isolated cypress domes are depressional wetlands that occur within the upland-wetland mosaic of the pine flatwoods in the southeastern coastal plain of the United States (Marios and Ewel 1983). The use of the term isolated cypress domes has been used widely in literature. However, recent investigations (Mushet et al. 2015) and the alterations observed in multiple wetlands in SWP due to groundwater extraction suggest that these wetlands are not isolated from a hydrological or ecological perspective. Therefore, to imply hydrologic (Tiner 2003) and ecological connectivity, isolated cypress domes will be referred to as cypress dome wetlands. Cypress dome wetlands are the most prevalent wetland type in west-central Florida and account for greater than 34% of all Florida wetlands (Heimburg 1984, Casey and Ewel 1998, Doherty et al. 2000). The hydroperiods in these systems, which usually range from 180 to 270 days per year (Casey and Ewel 1998), are dictated by precipitation and groundwater, making them particularly susceptible to hydrologic alterations due to groundwater extraction. The center of cypress dome wetlands experience the greatest depth and duration of inundation are generally dominated by *Taxodium* sp. These wetlands exhibit a range of species dictated by hydroperiod, hydropattern, and basin morphology. Recent judicial decisions (e.g., SWANCC v U.S. Army Corps of Engineers 2001) resulted in decreased regulatory authority to protect some depressional wetlands under Section 404 of the Clean Water Act. Consequently, cypress dome wetlands may be more susceptible to impact from urbanization and other activities (McCauley et al. 2013).

STARKEY WETLAND PARK (SWP) RESTORATION

Wetland restoration is a common strategy for mitigating wetland loss in the United States, with the general goal of reestablishing degraded biological, chemical, and physical properties (National Research Council 2001, Moreno-Mateos et al. 2012). As was the case in SWP, a common approach to wetland restoration projects begins by re-establishing natural patterns of wetland hydrology (Zedler 2000). Researchers have advocated for passive wetland hydrologic restoration (De Steven 2010), which has been implemented at SWP, because it remains more cost effective and requires less maintenance than more invasive approaches. Additionally, passive restoration exhibits the potential to restore and sustain natural processes (National Research Council 2001, Halle 2007) through wetland self-design (Mitsch and Wilson 1996). Despite the benefits of this approach, many restoration initiatives fail to achieve pre-impact ecosystem structural and functional conditions (Zedler and West 2008, Copeland 2010, De Steven et al. 2010, Moreno-Mateos et al. 2012). Failures result largely because the restored wetland hydrology is inappropriate or insufficient and the complex interactions between hydrology and other aspects of the wetland ecosystem are not fully understood (De Steven et al. 2010, Caldwell et al. 2011, Foulquier et al. 2013).

Vegetation responses to hydrologic recovery have been investigated, as restoration success criteria are often based on the ability of a project to reestablish pre-impact plant communities (Kentula 2000, De Steven and Gramling 2013). However, an in-depth understanding of the variability in plant community establishment and succession in the context of restored wetland hydrology is still lacking and few studies quantify ecological changes over extended periods (Caldwell et al. 2011). Most studies are limited to short (2-3 year) periods which remain insufficient to capture potential response lags and ensure continued restoration success. Additionally, vegetation-centric approaches may be inadequate in gauging overall restoration of wetland function because they overlook soil properties and functions (Shaffer and Ernst 1999) that affect important wetland processes (Stolt 2000, Mitsch and Gosselink 2007). Documenting restoration impacts on soil properties, typically requires longer time periods than identifiable changes in wetland vegetation and hydrology (Brinson et al. 1981, Lockaby et al. 1996, Mitsch and Gosselink 2007). As a result, the relationship between wetland hydrologic restoration and soil functional improvements, such as organic matter accumulation and nutrient cycling are not as clear (Atkinsons and Cairns 2001, Taylor and Middleton 2005, Gingrich and Anderson 2011).

The history of hydrologic alteration and recovery at SWP, as well as the long-term vegetation and hydrology data available, provides a unique opportunity to observe how wetland functions respond to hydrologic alteration and recovery. Using historical data to evaluate vegetation patterns and a field study to assess soil processes, this research addresses knowledge gaps associated with the restoration of wetland functions in the context of hydrologic alterations. Specifically, this thesis investigates 1) how wetland communities respond to hydrologic recovery, 2) how cypress dome vegetation differs among wetlands of various hydrologic regimes, and 3) how decomposition and other important soil functions differ between wetlands with various degrees of hydrologic alteration. The primary goal of this research is to provide information that can advance the science of wetland restoration and inform natural resource managers about restoration capabilities and the limitations of hydrologic recovery in cypress dome wetlands.

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CHAPTER 2: VEGETATION RESPONSE TO GROUNDWATER PUMPING AND HYDROLOGIC RECOVERY IN CYPRESS DOME WETLANDS OF WEST-CENTRAL FLORIDA

ABSTRACT

An investigation of wetland vegetation response to groundwater alteration was conducted in J.B. Starkey Wilderness Park, a municipal wellfield, in New Port Richey, Florida. Historic groundwater withdrawal and subsequent hydrologic recovery created a gradient of impacted wetlands. Twenty-seven wetlands were grouped, based on their hydrologic histories, as either altered (low inundation), marginally altered (intermediate inundation), or least altered (normal inundation) wetlands. Historic vegetation and hydrologic data were used to assess vegetation responses to hydrology between and among these groups. Species richness, prevalence index scores, importance percentages, and non-metric multidimensional scaling (NMDS) were used to evaluate vegetation responses in pre-and post-hydrologic recovery periods. Altered and marginally altered wetlands, responded to hydrologic restoration with increased species richness and hydrophytic tendency (i.e., lower prevalence index scores) among plant communities. Prevalence index scores and species richness in altered wetlands (2.37±0.13 – 1.97±0.03 and 9.2±1.07 – 12.11±1.27 respectively) were comparable to least altered wetlands (2.14±0.11 – 1.87±0.09 and 8.78±1.14 – 13.53±1.28, respectively) following hydrologic restoration. Marginally altered wetlands had lower prevalence index scores (1.96±0.05 – 1.56±0.05) and greater species richness (14.17±0.72 – 15.17±0.74) than least altered wetlands. Species

importance percentages and NMDS suggest variation in vegetation of altered and marginally altered sites was greater than least altered sites. Additionally, species composition in altered and marginally altered wetlands remained different than the composition found in least altered wetlands. These results suggest that reduction in groundwater extraction can cause enough passive hydrologic recovery to elicit vegetation responses but species composition in historically altered wetlands remain different than least altered wetlands in most cases.

INTRODUCTION

Hydrology influences wetland type and process (Mitsch and Gosselink 2007) such that it is considered the most important factor in the establishment and persistence of wetlands. The hydrologic regime of a wetland is pivotal to its ecological services because among other effects it dictates the vegetation functions and plant communities (De Steven et al. 2010). The degree to which hydrology fluctuates within a wetland has resounding impacts on local flora. Extreme inundation or drawdown can cause local extinctions and the timing and duration of flooding is pivotal to wetland plant community assembly. Rapid changes in hydrology can disrupt patterns of succession and result in undesirable (invasive, inappropriate, or monotypic) plant communities. Conversely, gradual changes in wetland hydrology allows vegetation succession to evolve and equilibrate over time (Howard and Wells 2009, Froend and Sommer 2010, Palanisamy and Chui 2013). This is especially relevant in cypress dome wetlands where communities are dictated by seasonal hydroperiods and wetland geomorphology (DeSteven et al. 2010). Local groundwater withdrawals can affect wetland hydroperiods (Kirkman et al. 2000), potentially resulting in the invasion of mesophytic species, altering the understory composition, or even allowing for complete community replacement (Florida Natural Areas Inventory 2006).

The role of hydrology in wetland vegetation development makes it critical to wetland restoration efforts. Restoration success is usually based on reestablishing wetland pre-impact vegetation (Kentula 2000, De Steven and Gramling 2013). Failures occur when the target communities are unsuitable for the restored hydrology or are replaced by better-adapted alternative plant communities (Erwin 1991, Caldwell et al. 2011). While plant communities within cypress dome wetlands have been established (Kirkman 2000, Casey and Ewel 2006), studies recognize the lack of in-depth understanding of the variability of plant communities in the context of hydrology (Caldewell et al. 2011). Additionally, others (Mitsch and Wilson 1996, Kentula 2000, Zedler 2000, Mitsch et al. 2012) emphasize that the timing of vegetation response to restoration varies from years to decades and the relatively short (5 year) monitoring periods associated with restoration projects may not adequately capture these responses. Recently, passive restoration, which allows for wetland self-design (Mitsch and Wilson 1996), has been advocated in cypress domes wetlands (De Steven 2010). Even though passively restored systems sometimes lack characteristic vegetation (van der Valk 1996, De Steven et al. 2006) managers see it as a viable option because it is less expensive, requires lower maintenance, and has a greater potential to sustain natural processes than more invasive approaches (National Research Council 2001, Halle 2007).

In rapidly growing areas, like the Tampa Bay, Florida area, cypress domes wetlands are at risk of degradation due to groundwater withdrawal associated with urbanization (Pittman and Waite 2009, Metz 2011, McCauley et al. 2013). Until the late-1990s, groundwater extractions from 11 wellfields were used to meet public water demand. Long-term groundwater withdrawal has lowered the water table in and around these wellfields (Lee et al. 2009) and caused negative impacts, such as shorter hydroperiods, vegetation shifts to upland communities, the introduction of invasive species, soil subsidence, falling trees, and the loss of wetland dependent wildlife (Dooris et al. 1990, Rochow 1998, Lee et al. 2009). Due in part to the local degradation experienced in the wetlands within the wellfields, alternative water sources were implemented (Southwest Florida Water Management District 2009) and in the early 2000s groundwater extraction within the wellfield decreased. Since then wetlands within the wellfield have experienced various degrees of passive hydrologic restoration (Metz 2011).

The current study seeks to inform natural resource managers about the restoration capabilities and limitations of hydrologic recovery in wetlands. Vegetation measures such as species richness, prevalence index (PI) scores, importance values, and ordination were used to evaluate how wetland communities respond to hydrologic recovery and how cypress dome vegetation differs among wetlands of various hydrologic conditions. We hypothesized that the vegetation in marginally altered and altered wetlands would respond to hydrologic recovery and that altered and marginally altered wetlands would be comparable to the least altered wetlands following 4 – 6 years of hydrologic restoration.

METHODS

Study Site

J.B. Starkey Wilderness Park (SWP) is a 3,200-ha property located in New Port Richey, in the northern Tampa Bay area of Pasco County, Florida (Fig. 2.1). It has historically been used as a ground water wellfield to the Northern Tampa Bay area. There are over eighty cypress dome wetlands within the wellfield that have experienced various degrees of alteration due to groundwater extraction (Metz 2011). The park has 14 wells that began pumping groundwater in the mid-1970s. At first, pumping was moderate (3.79-22.71 million liters of water a day) and was focused in the western portion of the park. However, as population numbers increased in the Tampa Bay region, pumping steadily increased and expanded to meet municipal demand, first in the central section of the park and then in the eastern section (Fig. 2.1). Increased pumping continued until the mid-2000s when pumping peaked at 49.21 million liters a day (Metz 2011). As the environmental damage of prolonged groundwater withdrawal became apparent within the wetlands, local government and water management districts worked to reduce groundwater extraction. In 2008, alternative water sources were secured and groundwater withdrawal within SWP was reduced to 15.14 million liters a day. The continued decrease in pumping allowed for some recharge of the aquifer and partial restoration of wetland hydrology (Tampa Bay Water 2009, Metz 2011, Lewis et al. 2015). Long-term hydrologic and ecological monitoring of SWP has been conducted by the Southwest Florida Management District (SWFWMD) and Tampa Bay Water (TBW) as part of their regulatory water use permits (SWFWMD and TBW 2005). The hydrologic history as well as the ongoing data collection at SWP provides an opportunity to examine how vegetation has responded to hydrologic changes across a gradient of alteration.

Wetland Hydrologic Data Collection and Classification

In SWP, permanent staff gauges and groundwater wells have monitored wetland hydrology for the last four decades. Using these long-term data sets, twenty-seven cypress dome wetlands were selected for investigation (Fig. 2.1). Selected sites had monthly hydrologic data collection since 1990 with more recent collections occurring bi-weekly (Fig. 2.2). Based on these data, two different hydrologic periods were selected. The first hydrologic period, referred to as the pre-hydrologic recovery period, occurred from 2005 to 2007 when groundwater production rates were highest (Metz 2011). The second period, referred to as the post-hydrologic recovery period, occurred from 2012 to 2014, four to six years after groundwater extraction dramatically decreased.

Cypress dome wetlands exhibit zonation due to fluctuations in annual hydroperiods (Haag et al. 2005). The SWFWMD and TBW (2005) identified three wetland zones (deep, outer deep, and transition) based on elevational differences from the historic normal pool (long term average wetland water level prior to hydrologic alterations). Historic normal pool levels were previously established in each wetland by correlating the elevations associated with biological and physical indicators of hydrology such as tree buttressing, moss collars, and the spatial distribution of saw palmetto (*Serenoa repens)* (Table 2. A1, SWFWMD and TBW 2005). The current study examined the transition zone, which spans from the wetland edge to 0.15 meters below the historic normal pool level and the outer deep zone, which spans from 0.15 meters below the historic normal pool elevation to 0.3 meters below the historic normal pool level (Fig. A1). Healthy cypress domes wetlands are generally inundated for 180 or more days each year (Casey and Ewel 1998). Therefore, the severity of hydrologic alteration across wetlands was determined based on the average days inundated over a 15-year period and the frequency of healthy yearly inundation (> 180 days a year) before and after hydrologic recovery (Table 2.A2, Table 2.A3). Based on the degree of alteration, wetlands were placed into three hydrologic groups (altered, marginally altered, and least altered). The least altered wetlands exhibited an average yearly inundation of 180 days or more, like healthy cypress dome wetlands, in their outer deep zone and were considered the most reference like wetlands available in SWP. Marginally altered and altered wetlands experienced on average less than 180 days of inundation a year, as outlined in Table 2.1. These hydrologic groups were used to assess how hydrologic conditions affect wetland vegetation.

Vegetation Data Collection

Because of extensive monitoring and assessment being conducted at wellfields throughout the region, SWFWMD and Tampa Bay Water developed the Wetland Assessment Procedure (WAP) for evaluating cypress dome wetlands (SWFWMD and TBW 2005). The WAP was designed, using historical data and common wetland assessment practices and parameters, to characterize the condition and health of cypress dome wetlands and document the ecological changes associated with groundwater withdrawal (SWFWMD and TBW 2005). As part of the WAP, vegetation surveys occur in each zone of SWP wetlands on an annual basis since 2005. These surveys occur in permanent assessment areas that are ten meters wide and span from the edge of one wetland zone to the edge of the next wetland zone. All groundcover vegetation (defined as all herbaceous and woody species less than 1 m tall) within the assessment are identified to species (or lowest taxonomic level defined by WAP) and their absolute percent cover is determined using visual estimation. Thus, the WAP data provides annual species lists and percent cover data within each wetland zone. Prior to analysis species lists were updated to reflect current species names and taxa that were not identified to species level were removed from the data set (less than 4% of all data). The absolute percent cover for species that were reported as less than 5 percent were converted to a standard 2.5 percent.

Wetland Plant Data Analyses

Using the annual WAP vegetation data, species richness, hydrophytic prevalence, and community assemblage were investigated in the both the outer deep zone and the transition zone of each wetland. Vegetation measures, such as species richness, prevalence index (PI) scores, distribution of WAP vegetation classification, importance percentages, and non-metric multidimensional scaling (NMDS), were used to compare communities within and between wetland hydrologic groups (altered, marginally altered, and least altered) and over the two hydrologic periods (pre- and post-hydrologic recovery).

Species Richness

Species richness was determined by counting the number of species listed in annual WAP reports. Yearly average species richness values were determined for each zone of each hydrologic group (altered, marginally altered, and least altered). Richness values were compared between the pre- (2005-2007) and post-hydrologic (2012-2014) recovery periods as well as between hydrologic groups.

Prevalence Index

The hydrophytic tendency of the plant community within each wetland was determined using a Prevalence Index (PI) score. Changes in the hydrophytic tendency were evaluated pre- and post-hydrologic recovery based on changes in PI scores in each zone and between hydrologic groups. Prevalence indices, as described in Peet et al. (1988), combine relative abundance and wetland indicator status ratings to create a weighted average that describes the predominance of hydrophytic species at a location based on the following equation:

$$Prevalence\ Index = \sum {}^{A_i W_i}\!/_{\sum W_i}$$

Where A_i = abundance of species i, W_i = wetland indicator status rating for species i

All species on the WAP vegetation species lists were classified per their Florida Department of Environmental Protection (DEP) wetland indicator status rating which includes five categories related to vegetation tolerance for saturated soil conditions: obligate upland species almost never occur in wetlands (5), facultative upland species rarely occur in wetlands (4), facultative species exhibit equal distribution in both wetlands and uplands (3), facultative wetland species frequently occur in wetlands (2), and obligate wetland species almost always occur in wetlands (1) (US Army Corps of Engineers, 2012; Tiner 2012). Using this information as well as the percent cover reported in the WAP vegetation data, a PI score was calculated for each wetland zone of during pre- and post-hydrologic recovery periods. Lower scores indicate more hydrophytic communities with scores of three or less indicating the presence of a wetland plant community (Peet et al. 1988, US Army Corps of Engineers 2012).

Statistical Analyses

Analysis of variance (ANOVA) and Tukey's honest significant difference (HSD) were used to compare vegetation variables (species richness, PI score) between groups during the pre- (2005-2007) and post-hydrologic recovery (2012-2014) periods. Additionally, paired t-tests were used to compare a group's vegetation variables between the hydrologic periods. In instances where the assumption of normality was violated, species richness data in both the outer deep and transition zone, data were natural log transformed prior to analysis. All statistical analysis was run in the program R and significance was determined at the $\alpha = 0.05$ level.

Vegetation Community Analyses

WAP Species Category Distributions

The WAP recognizes 111 wetland species characteristic of cypress domes wetlands. Further, WAP species are classified based on the zone (e.g., transition, outer deep) of the WAP transect where they are commonly found in healthy cypress dome wetlands. The WAP vegetation classifications include: upland (species commonly found in the upland), adaptive (found throughout the wetland, ruderal species), transition (species commonly found in the transition zone), outer deep (species commonly found in the outer deep zone), and deep (species commonly found in the deep zone, Table 2.2). To evaluate the suitability of species in the transitional and outer deep zones, plant species data were analyzed to determine what portion of the species cover was represented by each WAP vegetation classification. Any species detected that was not designated in the WAP zone classifications was listed in a non-WAP category created for our analysis. Yearly zone classifications over a ten-year period (2005 – 2014) represented in each wetland were averaged per hydrologic group (altered, marginally altered, and least altered) and used to investigate the community composition of each zone by weighted average using the following equation:

$$Distribution\ of\ Zone_x = {}^{\sum C_x}/_{C_w} \times 100\%$$

Where $\sum C_x$ = the summation of the cover of all the species of Zone, C_w = the total coverage in the wetland zone, and x = vegetation zone classification (Upland, Adaptive, Transition, Outer Deep, Deep, and Non-WAP; Table 2.2)

Species Importance Percentages

Importance percentages for each groundcover species in each hydrologic group during the pre- and post-hydrologic periods were determined using a modified version of the method described in Kirkman et al. (2000). Average percent cover values and frequency of occurrence for each species was determined for each hydrologic group during the pre- and post-hydrologic recovery periods. For example, the importance values of species found in the outer deep zone of the altered wetlands were determined prior to and after hydrologic recovery. Importance percentages were calculated by multiplying the average percent cover value of a species by its frequency of occurrence. To standardize the importance percentages per time period and group, values were divided by the total vegetation cover reported for the specific group during a specific period. These numbers were then multiplied by 100%, converting results to a percentage basis for comparison. For reporting purposes or performing additional analyses (see below) any species with an importance value of < 2.5% was excluded. The following equation was used:

$$Importance\ Percentage = {}^{F_i \times C_i}/_{TC} \times 100\%$$

Where F_i = frequency of species i, C_i = average cover of species i, and TC = summation of all species coverage

Community Comparison among Hydrologic Groups

The NMDS ordination was used to describe differences in vegetation species composition (as measured by importance percentage) between hydrologic groups. Species with importance percentages < 2.5% were removed and then data were square-root transformed to reduce the influence of highly abundant species (Quinn and Keough 2002). Data were then put in a Bray-Curtis dissimilarity matrix and an ordination was run on the dissimilarity matrix. Stress coefficients represented the goodness of fit and NMDS models were acceptable for interpretation when stress was <0.2 (McCune and Grace 2002). An analysis of similarities (ANOSIM) was used on the Bray-Curtis dissimilarity matrix of the species importance percentage data by hydrologic group to assess the significance of any differences between the communities of these groups. NMDS and ANOSIM were run using the program R.

RESULTS

Species Richness

Over the monitoring period, species richness generally remained similar in the transition zone of the least altered wetlands (Fig. 2.3C). However, the transition zone in both the altered and marginally altered wetlands showed a marked increase in species richness two years after (2010) hydrologic restoration (Fig. 2.3A, 2.3B). Species richness in the transition zone of the altered wetlands increased by five species post-hydrologic recovery, a statistically significant increase (Fig. 2.3D). Species richness also displayed significant increases, with eight additional species occurring in the transition zone of the marginally altered wetlands (Fig. 2.3E). Least altered wetlands experienced no significant increase in species richness (Fig. 2.3F) in the transition zone post-hydrologic recovery.

Similar trends were also observed in the outer deep zone. All three hydrologic groups exhibited significant increases in species richness in the outer deep zone following hydrologic recovery (Fig. 2.4). Altered (Fig. 2.4D) and marginally altered (Fig. 2.4E) wetlands had seven more species in the outer deep zone while the least altered wetlands had six more species post-hydrologic recovery (Fig. 2.4F).

In the wetland transition zone, during the pre-recovery period (2005-2007), no significant differences in species richness occurred between the altered, marginally altered, and least altered groups (Fig. 2.5A). However, differences in species richness were observed between wetland groups (Fig. 2.5B), with marginally altered wetlands containing significantly more species (14.17±0.72) than both altered (9.2±1.07, $p < 0.001$) and least altered (8.78±1.14, $p = 0.002$) wetlands during the post-hydrologic recovery.

Species richness was significantly different between the hydrologic groups in the outer deep zone during the pre-recovery period (Fig. 2.5C). Altered wetlands displayed significantly fewer species (5.33±0.49) than the least altered wetlands (8.13±0.79, $p = 0.021$) while no differences were observed between the marginally altered (7.53±0.57) and altered wetlands ($p = 0.062$) or the marginally altered and least altered ($p = 0.775$) wetlands. Post-hydrologic recovery no significant differences in species richness occurred between the altered, marginally altered, and least altered wetlands (Fig. 2.5D).

Prevalence Index

The PI scores within each hydrologic group were compared pre- and post-hydrologic recovery. There were no observed changes in PI scores in the transition zone for any of the wetland groups post-hydrologic recovery. However, in the outer deep zone, altered (Fig. 2.6A) and marginally altered wetlands (Fig. 2.6B) displayed lower PI scores post-hydrologic recovery while the least altered wetlands maintained similar PI scores throughout the monitoring period (Fig. 2.6C).

In the transition zone, PI scores remained similar among the three groups (altered: 2.28±0.19, marginally altered: 2.02±0.11, and least altered: 2.00±0.03) prior to hydrologic recovery (Fig. 2.7A). However, post-hydrologic recovery hydrophytic tendency varied between the three groups. In the transition zone, marginally altered wetlands had lower PI scores (1.96±0.05) than altered wetlands (2.37±0.13, $p = 0.006$). The PI scores in the least altered (2.14±0.11) wetlands remained comparable with altered and marginally altered wetlands (Fig. 2.7B).

During the pre-hydrologic recovery period, there were significant differences in the PI scores in the outer deep zone (Fig. 2.7C). Altered wetlands had significantly higher PI scores (2.34±0.16) than the least altered wetlands (1.82±0.05, $p = 0.003$). Marginally altered wetlands (2.09±0.09) had PI scores similar to both the altered and least altered wetlands. Significant differences in PI scores, post-hydrologic recovery, were also observed (Fig. 2.7D). Post-hydrologic recovery, marginally altered wetlands (1.56±0.05) had significantly lower PI scores than altered (1.97±0.14, $p = 0.003$) and least altered (1.87±0.09, $p = 0.047$) wetlands. Post-hydrologic recovery altered and least altered wetlands had similar PI scores.

Community Composition and Comparison

WAP Category Distributions

The WAP plant classifications (upland, transition, adaptive, etc.) were used to observe category distributions in the transition (Fig. 2.8) and outer deep (Fig. 2.9) zones. Vegetation in the outer deep and transition zones were largely composed of WAP-classified outer deep plants in all the hydrologic groups during the pre-hydrologic recovery period. Following hydrologic recovery, in both the transition and outer deep zones, wetlands experienced an increase in non-WAP species. However, these changes were less substantive in the least altered wetlands when compared to changes in the altered and marginally altered wetlands.

The relative coverage of outer deep WAP classified species declined in the transition zone of the marginally altered wetland during the hydrologic recovery period because of an increase in non-WAP species as well as an increase and subsequent decline in upland and adaptive species (Fig. 2.8). In the transition zone of the altered wetlands, the relative cover of outer deep WAP species as well as adaptive species declined post-hydrologic recovery due to the encroachment of non-WAP species (Fig. 2.8). Post-hydrologic recovery, the coverage of outer deep and adaptive species declined in the outer deep zone of the marginally altered wetlands due to non-WAP species encroachment (Fig 2.9). In the outer deep zone of the altered wetlands, the coverage of outer deep WAP species declined due to an increase and subsequent decline in upland and adaptive WAP plants as well as encroachment by non-WAP species during the post-hydrologic recovery period (Fig 2.9).

Species Importance Percentages

Importance percentages were calculated for each hydrologic group during the various hydrologic periods (Table 2.3 and Table 2.4). *Amphicarpum muhlenbergianum* was the most important species in the transition zone during the pre-recovery period. Post-hydrologic recovery, *A. muhlenbergianum* remained the most important in the least altered and marginally altered wetlands. However, in the altered wetlands, *Galactia elliotti* became the most important species post-hydrologic recovery.

In the outer deep zone, during the pre-hydrologic recovery *A. muhlenbergianum* was the most important species in the marginally altered and least altered communities while *Eupatorium capillifolium* was the most important species in the altered wetlands. Post-hydrologic recovery, *A. muhlenbergianum* was still the most important species in the least altered wetlands. However, *Woodwardia virginica* became the most important species in the marginally altered and altered wetlands after hydrologic recovery.

Community Comparison among Hydrologic Groups

Results from NMDS show that plant species assemblages in the transition zones of the various wetlands generally occupied the same ordination space during the pre-hydrologic recovery period (Fig. 2.10A) and ANOSIM results indicated that the communities were not different between the hydrologic groups ($R = 0.04$, $p = 0.29$). The NMDS of post-hydrologic recovery transition data showed marginally altered and altered communities diverging from the least altered communities (Fig. 2.10B). However, ANOSIM indicated that the communities within each group were still not significantly different from one another ($R = 0.03$, $p = 0.36$). In both the pre- and post-hydrologic recovery periods the least altered wetlands clustered more tightly than the altered or marginally altered wetlands suggesting less variability in community composition.

Ordination plots generated for the outer deep zone during the pre-hydrologic restoration showed clustering based on the hydrologic groups (Fig. 2.11A). It appears that altered and least altered sites occupy different ordination spaces. ANOSIM indicated a near significant difference in plant communities of the hydrologic groups prior to hydrologic restoration ($R = 0.12$, $p = 0.09$). After hydrologic recovery, ordination plots indicated that the outer deep communities of the various hydrologic groups were converging (Fig. 2.11B) and ANOSIM found no differences in the communities based on the hydrologic groups ($R = 0.01$, $p = 0.47$). Prior to hydrologic restoration variability within the wetlands of each hydrologic group was similar, a much tighter clustering of the least altered sites following hydrologic recovery suggest increased variability in community compositions among the altered and marginally altered sites.

DISCUSSION

Vegetation responses to passive hydrologic recovery were observed in SWP and the results from this study suggest that increased inundation influenced vegetative measures, such as species richness and hydrophytic assemblages, in relatively short (5-7 year) periods. However, this study also found differences in species composition and community variation in wetlands of various degrees of hydrologic alterations suggesting that more degraded wetlands may require additional recovery time prior to achieve reference conditions.

Species richness increased in both the outer deep and transition zone of the altered and marginally altered wetlands as well as in the outer deep zone of the least altered wetlands following hydrologic recovery. This increase in species richness, even in the least altered wetlands, is unsurprising as past studies have illustrated that richness can increase after hydrologic recovery due to the persistence of some less flood tolerant species along with the recruitment of more flood tolerant species (Battaglia and Collins 2006, De Steven et al. 2010). Species richness responded immediately in the outer deep zone and within two years of hydrologic intervention in the transition zone suggesting a potential 0-2-year lag in vegetation response following hydrologic changes. Persistence of some species, and the recruitment of others, was most substantial in the altered and marginally altered wetlands. These wetlands had species richness that was comparable to the least altered wetlands and in some case exceeded the least altered wetlands at the end of the study period.

We hypothesized that vegetation shifts in altered sites would trend toward least altered conditions after restoration, including changes in the PI scores over time. In general, as water levels recede in a wetland, plant communities shift toward more upland and facultative upland species. Conversely, as water becomes more available wetland plant communities shift toward obligate and facultative wetland species that are more tolerant of saturated soil conditions (Hammersmark et al. 2009, De Steven and Gramling 2013, Turner et al. 2015) During both hydrologic periods, wetlands had PI scores of ≤ 3.0 indicating that all study wetlands met hydrophytic vegetation criteria (Peet et al. 1998, U.S. Army Corps of Engineers 2012). Post-hydrologic recovery PI scores declined in the outer deep zones of the altered and marginally altered wetlands suggesting that the increase in inundation was sufficient to support an increase in the wetland character of understory plant communities, which are more responsive to changes in hydrology (Young et al. 1995, Chapin and Page 2013).

The cypress dome wetlands in SWP showed a gradient of vegetation responses to passive hydrologic recovery. Least altered communities exhibited greater stability and less variation in community composition, species richness, and PI scores during both hydrologic periods. The species found in the least altered communities were generally appropriate for cypress dome wetlands. Pre-hydrologic recovery transition zones in least altered communities were dominated by *Amphicarpum muhlenbergianum*, which was common throughout the SWP wetlands and considered a ubiquitous species, with lesser amounts of *Hypericum fasciculatum*, and *Aristida stricta* var. *beyrichiana*. Post-hydrologic recovery, *A. muhlenbergianum* remained the most important species followed by *A. beyrichiana*, and *Morella cerifera*. *Amphicarpum muhlenbergianum* and *A.*

beyrichiana are generally found in association with one another and occur where saturated soils persist throughout part of the year (Sheahan et al. 2011). Additionally, these species are indicative of intermediate shallow flooding, shorter seasonal hydroperiods, and greater light penetration (Marios and Ewel 1983, Florida Natural Areas Inventory 2010, Thurman 2016), which are conditions typically found in the outer edge or transition zone of cypress dome wetlands. Least altered communities in the outer deep zone were also dominated by *A. muhlenbergianum* as well as *Stillingia aquatica* and *Hypericum fasciculatum.* The presence of *S. aquatica*, an obligate species, and the loss of *A. beyrichiana*, a facultative species, suggest that inundation in the outer deep is greater than that of the transition zone. *Amphicarpum muhlenbergianum* remained the dominant species in the least altered wetlands following the hydrologic recovery.

Compared to least altered wetlands, altered wetlands experienced significant vegetation changes post-hydrologic recovery. Species richness increased and PI scores decreased, trending toward least altered conditions. However, altered wetlands displayed more variability between individual site compositions than least altered wetlands and importance values suggest substantial community turnover in both the outer deep and transition zones following hydrologic restoration. Pre-hydrologic recovery, *A. muhlenbergianum* dominated the transition zones of the altered wetlands with lesser amounts of *Eupatorium leptophyllum*, *Galactia elliottii*, *Cladium mariscus*, and *Andropogon virginicus. Eupatorium leptophyllum* and *C. mariscus* are commonly found in wetlands but the importance of *G. elliotti* (an upland plant) and *Andropogon virginicus* (a plant indicative of drier soils and reduced hydroperiods; Hitchcock 1950, Thurman 2016) suggests that the transition zones of these wetlands were experiencing less inundation and drier conditions. Post-hydrologic recovery, the communities show signs of the drier pre-recovery period as well as increased inundation. Interestingly, *A. muhlenbergianum* disappeared as a dominant species in the transitional zone of altered sites (Table 2.3). *Galactia elliotti* was the most important species post-hydrology recovery and like *Vitis rotundifolia* (an adaptive facultative plant) was most likely established during the pre-recovery period and has been able to tolerate hydrologic recovery. The importance of *Rhynchospora microcephala* (a facultative wetland plant), *Diodia virginiana* (an obligate wetland plant) and *Woodwardia virginica* (an obligate wetland plant) suggest an increase in inundation following hydrologic recovery.

Prior to hydrologic recovery altered outer deep communities included *Eupatorium capillifolium, G. elliotti,* and *Panicum hemitomon* suggesting drier conditions and shorter but possibly deeper hydroperiods (Thurman 2016). *Galactia elliotti* remained important post-hydrologic recovery. However, *W. virginica*, which was the most important species has been identified as a common and ubiquitous cypress dome species (Thurman 2016). Other important species included *A. muhlenbergianum*, which was also present in the least altered wetlands, and *Rhynchospora inundata,* which is indicative of wetter and restored communities.

Post-hydrologic recovery, marginally altered wetlands, which have experienced a greater increase in inundation than altered wetlands, had greater species richness and more hydrophytic communities than the least altered wetlands. Species composition was somewhat comparable to least altered communities because *A. muhlenbergianum* was the most important species both pre-and post-hydrologic recovery. There was also good agreement in the portion of various WAP (and non-WAP) species classifications for the outer deep zone (Fig. 2.9). However, following hydrologic recovery, marginally altered wetlands also recruited numerous species that were not present in the least altered wetlands and a number of non-WAP species. Although some common important species are shared, the recruitment of non-WAP species has contributed to wider variation in the assemblage of species in altered and marginally altered wetlands (see NMDS results; Figs. 2.10 and 2.11). If achieving a community composition comparable to least altered wetlands is the goal of restoration, the marginally altered wetlands may require additional time and remediation to achieve least altered conditions.

Hydrologic recovery remains ongoing in SWP (Table 2.1) and will likely continue to influence the trajectory of wetland restoration. As hydrologic recovery continues we might see additional changes in wetland plant communities, especially at the most altered locations. We might also see altered communities diverge from the reference communities in species richness and PI scores as illustrated in the marginally altered communities, if inundation and depth of inundation increases, due to historical subsidence as observed in the altered and marginally altered sites. Differences in community composition and variability in the altered and marginally altered wetlands, which hosted both upland and obligate species within the same zone, suggests that restoration and vegetation response is ongoing, progresses on individual timelines, and extends beyond the general restoration monitoring period of most projects.

Wetland restorations aim to return communities to pre-impact conditions (Kentula 2000, De Steven and Gramling 2013). However, in some cases pre-impact conditions cannot be obtained because either abiotic or biotic thresholds have been crossed (Beisner et al. 2003, Briske et al. 2006). In these cases, alternative communities, that are different from reference or unimpacted conditions but provide functions and services that are just as ecologically valuable (De Steven 2010), may be more realistic. In SWP, the goal of returning communities to least altered like conditions may be unachievable for the marginally altered and altered wetlands. Species composition and community variation, as observed by the importance percentages and ordination, suggest that communities in the marginally altered and altered wetlands are composed of greater numbers of species that are considered non-WAP and are not found in the least altered sites. In the case of the marginally altered sites, communities are even more hydrophytic than the least altered wetlands. As the hydrology in the park continues to restore, further divergence in community compositions among these groups may occur. As these wetlands, which provide important ecological services, continue to diverge restoration goals may need to be reassessed and adjusted as restoring assemblages consistent with least altered wetlands becomes unrealistic.

CONCLUSIONS

Study results suggest that the recovering wetland hydrology, due to reduced groundwater withdrawal, plays an important role in wetland restoration and re-establishment of vegetation. Plant communities in altered and marginally altered wetlands responded to an increase in inundation depth and hydroperiod. Following seven years of passive hydrologic restoration, species richness and PI scores in the altered wetlands are comparable to those of the least altered communities. However, marginally altered wetlands, which experienced greater hydrologic recovery, had greater species richness and more hydrophytic assemblages than the least altered wetlands examined. Additionally, vegetation composition was different and more variable in the altered and marginally altered wetlands than in the least altered wetlands. Available evidence suggests that measures of species composition require more than 2-5 years to stabilize following restoration activities. As a result, increased monitoring time frames may be required following hydrologic restoration to evaluate steady state conditions. Continued monitoring is necessary as hydrologic restoration continues. The vegetation communities examined, while different, may prove as ecologically valuable as least altered communities and restoration goals may need to be evaluated and adjusted as projects progress.

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Table 2.1. Average inundation period in each hydrologic group as well as the annual frequency at which reference inundation (>180 days/ year in outer deep zone and > 105 days/year in the transition zone) occurred before and after hydrologic recovery.

	Transition Zone			Outer Deep Zone		
	Average Inundation Period (days per year)	Frequency of Reference Inundation		Average Inundation Period (days per year)	Frequency of Reference Inundation	
Hydrologic Group	1990 -2014	1990 - 2007	2008 - 2014	1990 -2014	1990 - 2007	2008 - 2014
Least Altered	> 105	60%	60%	> 180	60%	60%
Marginally Altered	55 -105	30%	50%	80 - 180	30%	45%
Altered	< 55	5%	15%	<80	2%	15%

Table 2.2. Description of WAP species plant classifications for cypress dome wetlands as described in SWFWMD and Tampa Bay Water (2005) and associated national wetland indicator status rating and definition (Lichvar et al. 2016).

WAP Classification	Location in cypress dome	Species habit and associated wetland indicator rating
Upland	Not expected	Plants that occupy mesic to xeric habitats and almost never occur in saturated soils, typically found surrounding cypress dome wetlands (i.e., UPL)
Adaptive	Limited in Transition	Plants that can grow in hydric, mesic, or xeric soils/plants that occupy mesic to xeric soils, typically opportunist or ruderal species (i.e., UPL - FAC)
Transition	Transition	Plants that grow where water saturates the soils or floods the soil surface a least seasonally/plants that almost always occur in standing water, typically plants that tolerate shallow and shorter hydroperiod (i.e., FACW/OBL)
Outer Deep	Outer Deep	Plants that grow where water saturates the soils or floods the soil surface a least seasonally/plants that almost always occur in standing water, typically plants that tolerate deeper and greater hydroperiods than transition species (i.e., FACW/OBL)
Deep	Deep	Plants that grow where water saturates the soils or floods the soil surface a least seasonally/plants that almost always occur in standing water, typically plants that tolerate deeper and greater hydroperiods than outer deep species (i.e., FACW/OBL)
Non-WAP	Varies	Upland and wetland plant species that are not typically associated with cypress domes of west-central Florida and/or are not sensitive to hydrologic changes (i.e., UPL – OBL)

Table 2.3. Species and their importance percentages (Im %) and national wetland indicator status rating (NWISR) in the transition zone for each hydrologic group during the two hydrologic periods. Species with importance percentages less than 5% are not included. Non-WAP species are in boldface.

Hydrologic Group	Species pre-recovery	Im %	NWISR	Species post-recovery	Im %	NWISR
Altered						
	Amphicarpum muhlenbergianum	52.8	FACW	*Galactia. elliotti*	14.6	FACU
	Eupatorium leptophyllum	7.9	FACW	*Diodia virginiana*	7.0	FACW
	Galactia. elliotti	6.0	FACU	*Vitis rotundifolia*	7.0	FAC
	Cladium mariscus	5.6	OBL	***Rhynchospora microcephala***	6.4	FACW
	Andropogon virginicus	5.1	FAC	***Woodwardia virginica***	5.3	OBL
Marginally Altered						
	Amphicarpum muhlenbergianum	44.9	FACW	*Amphicarpum muhlenbergianum*	20.7	FACW
				Andropogon glaucopsis	7.0	FACW
				Woodwardia virginica	6.3	OBL
				Eriocaulon decangulare	5.6	OBL
Least Altered						
	Amphicarpum muhlenbergianum	53.4	FACW	*Amphicarpum muhlenbergianum*	52.9	FACW
	Hypericum fasciculatum	9.0	FACW	***Aristida stricta var. beyrichiana***	7.6	FAC
	Aristida stricta var. beyrichiana	6.0	FAC	*Morella cerifera*	5.3	FAC

Table 2.4. Species and their importance percentages (Im %) and national wetland indicator status rating (NWIS) in the outer deep zone for each hydrologic group during the two hydrologic periods. Species with importance percentages less than 5% are not included. Non-WAP species are in boldface.

Hydrologic Group	Species pre-recovery	Im %	NWISR	Species post-recovery	Im %	NWISR
Altered						
	Eupatorium. capillifolium	15.27	FACU	***Woodwardia virginica***	12.46	OBL
	Galactia. elliotti	9.89	FACU	*Galactia. elliotti*	9.79	FACU
	Panicum hemitomom	5.59	OBL	*Amphicarpum muhlenbergianum*	9..07	FACW
				Rhynchospora inundata	6.41	
Marginally Altered						
	Amphicarpum muhlenbergianum	28.97	FACW	***Woodwardia virginica***	11.45	OBL
	Stillingia. aquatica	5.13	OBL	*Amphicarpum muhlenbergianum*	10.84	FACW
				Carex verrucosa	7.93	OBL
Least Altered						
	Amphicarpum muhlenbergianum	33.19	FACW	*Amphicarpum muhlenbergianum*	25.46	FACW
	Stillingia. aquatica	7.08	OBL			
	Hypericum fasciculatum	6.94	FACW			

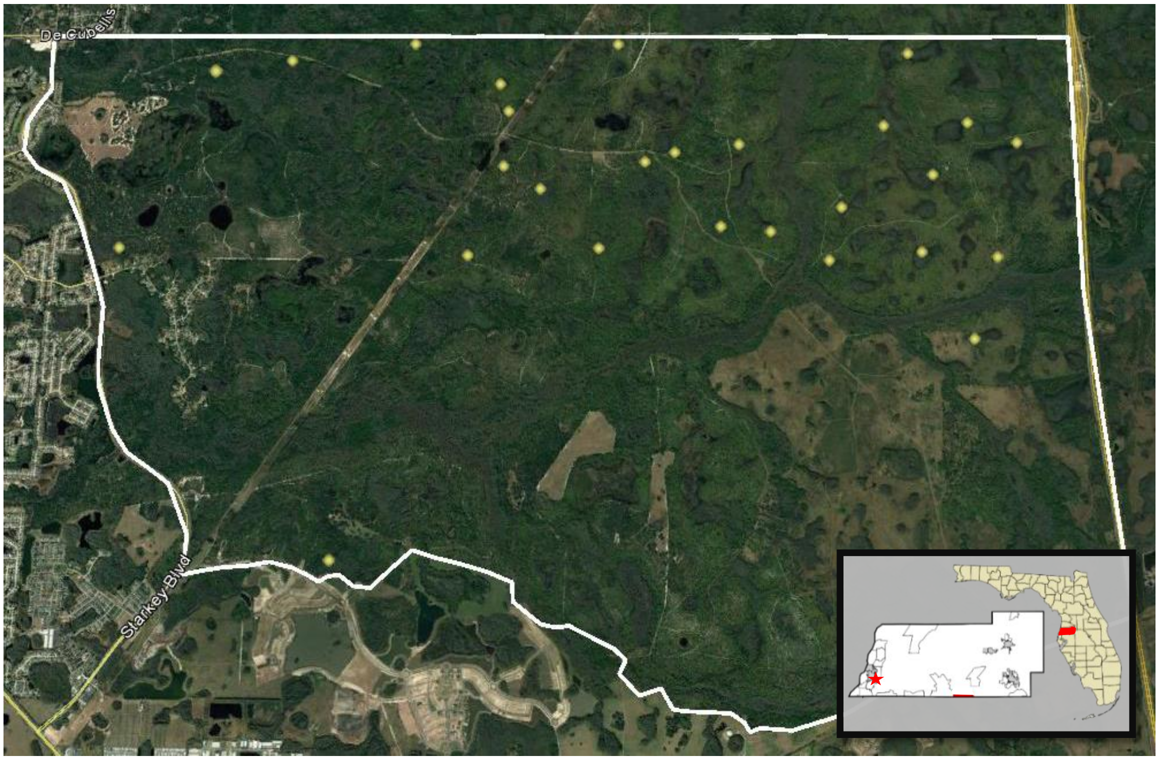

Figure 2.1. J.B. Aerial image of J.B. Starkey Wilderness Park. White line denotes park boundaries and white circles represent the wetlands examined in the current study. Aerial image from Google Earth. Inset on bottom right highlights the location of SWP in Pasco County, FL.

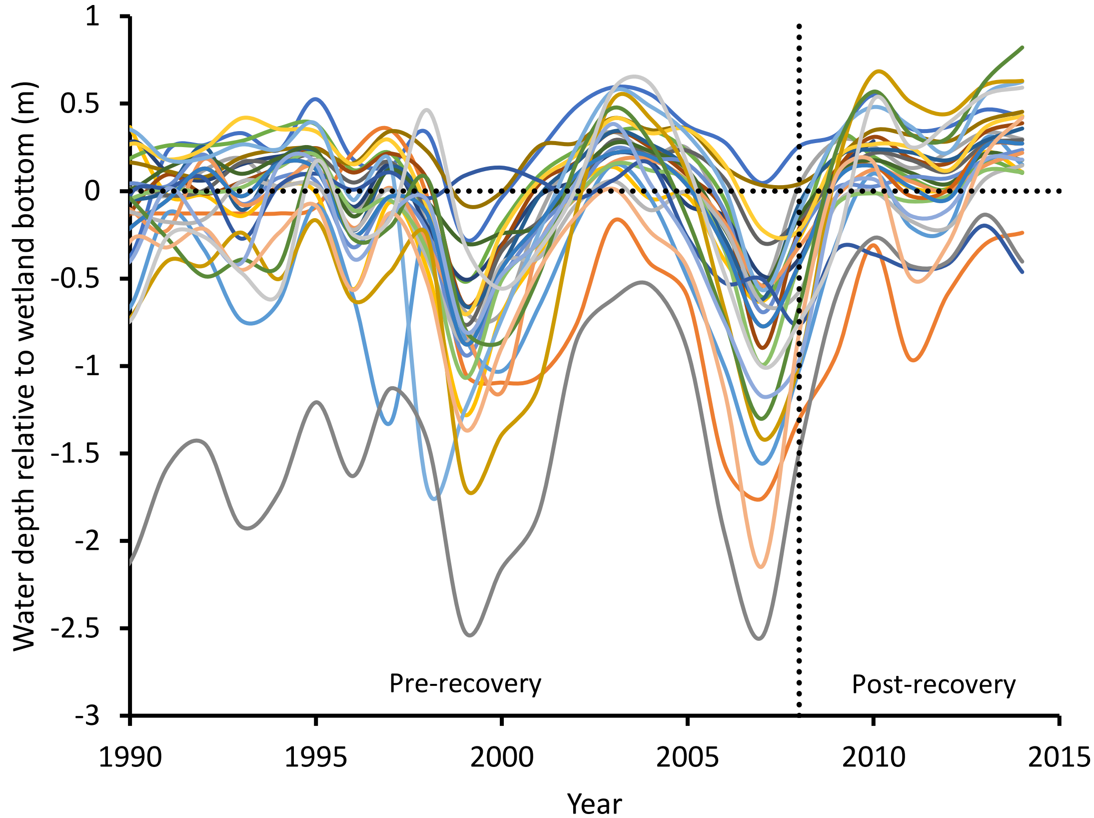

Figure 2.2. Average yearly water depth in each selected wetland from 1990 - 2014. Depths are relative to individual wetland bottoms. Reduced groundwater extraction and hydrologic recovery began occurred in January 2008.

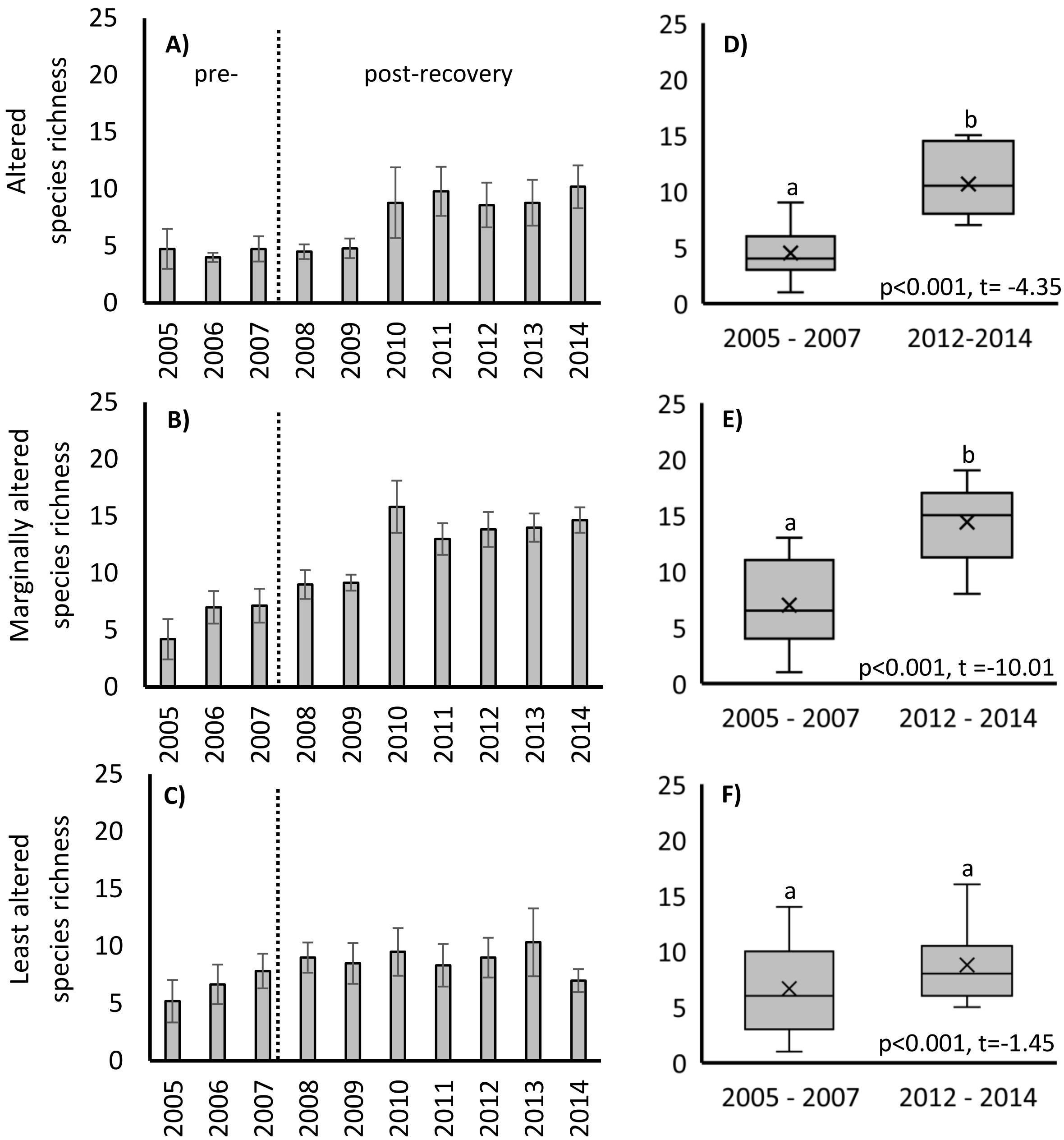

Figure 2.3. Mean (±SE) annual species richness in the transition zone of A) altered, B) marginally altered, and C) least altered wetlands. Box-plot comparison of species richness between pre-and post-hydrologic recovery of D) altered, E) marginally altered, and F) least altered wetlands. Differences between hydrologic periods, per paired t-test, indicated by lowercase letters.

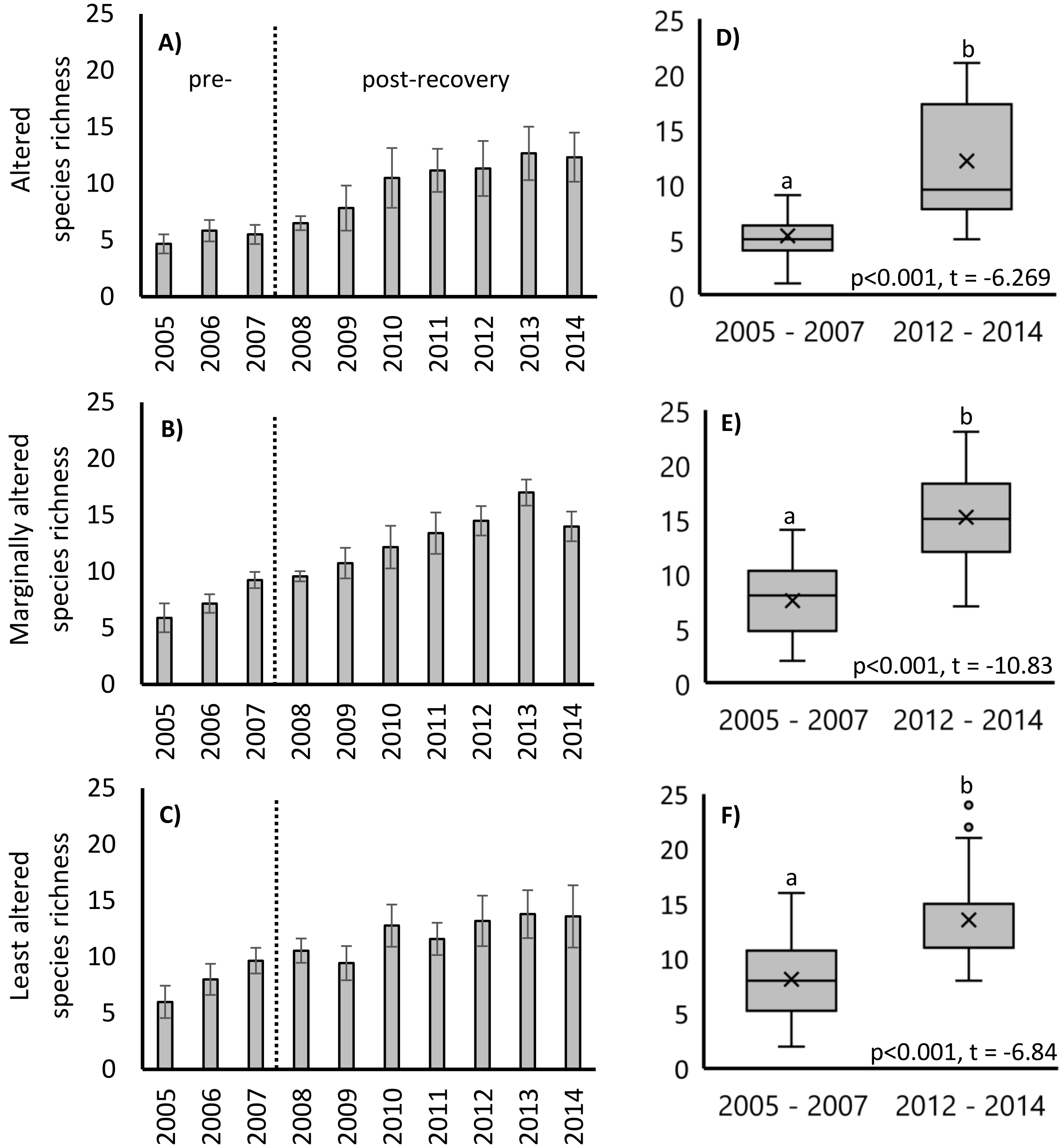

Figure 2.4. Mean (±SE) annual species richness in the outer deep zone of A) altered, B) marginally altered, and C) least altered wetlands. Box-plot comparison of species richness between pre-and post-hydrologic recovery of D) altered, E) marginally altered, and F) least altered wetlands. Differences between hydrologic periods, per paired t-test, indicated by lowercase letter.

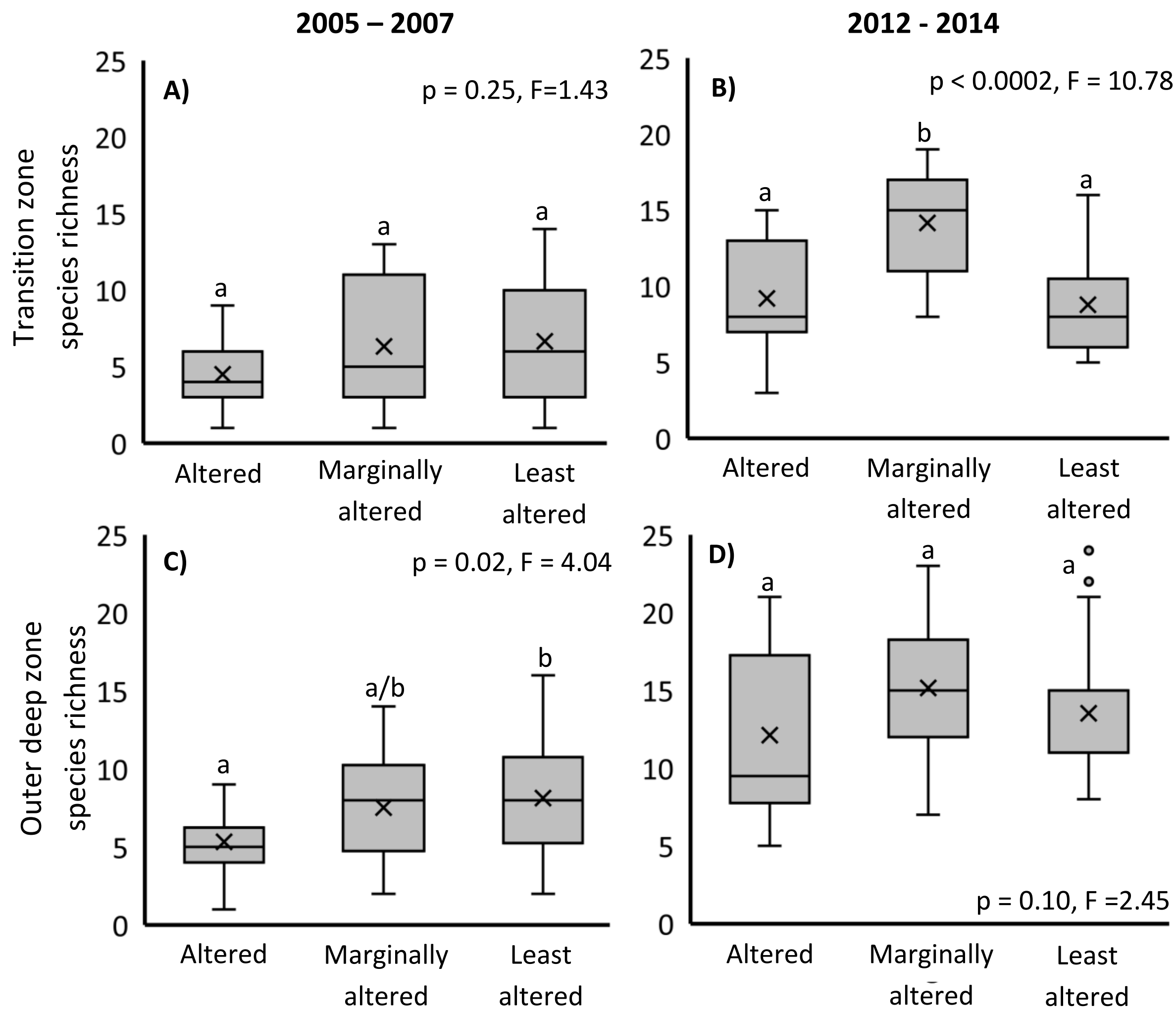

Figure 2.5. Box-plot comparison of species richness in each wetland hydrologic group during pre- and post-hydrologic recovery periods for the transition zone (A and B) and the outer deep zones (C and D). Lower case letters identify groups with statistically significant differences.

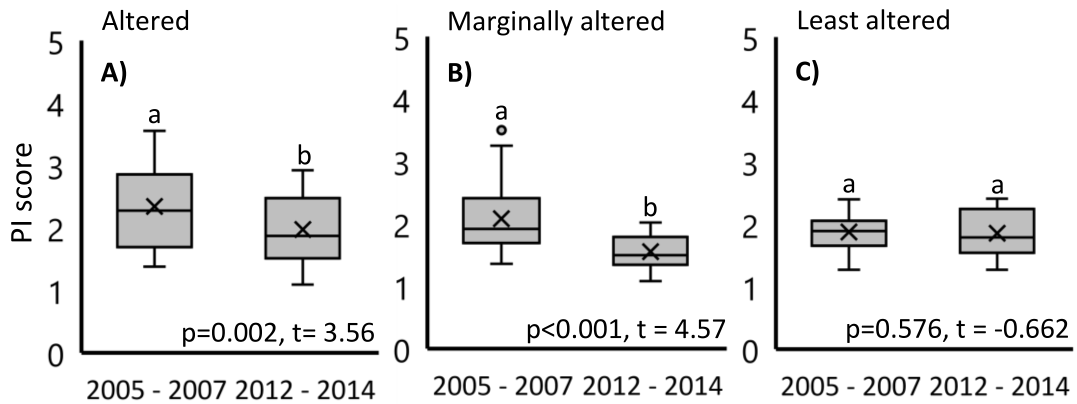

Figure 2.6. Prevalence index scores in the outer deep zone during pre- (2005 -2007) and post-hydrologic (2012 -2014) recovery periods. Lower case letters.

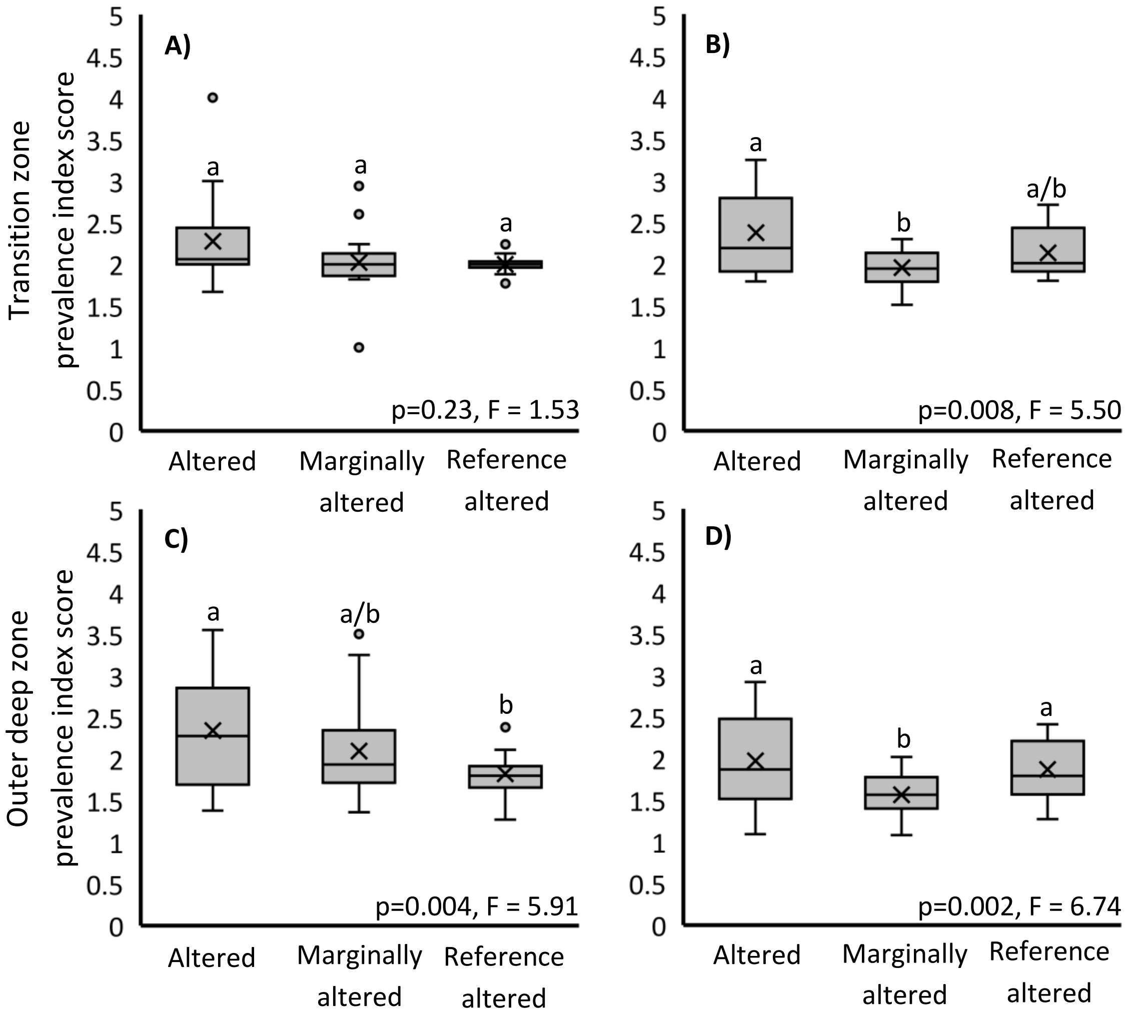

Figure 2.7. Box-plot comparison of PI scores in each wetland hydrologic group pre- and post-hydrologic recovery for the transition zone (A and B) and the outer deep zone (C and D). Lower case letters.

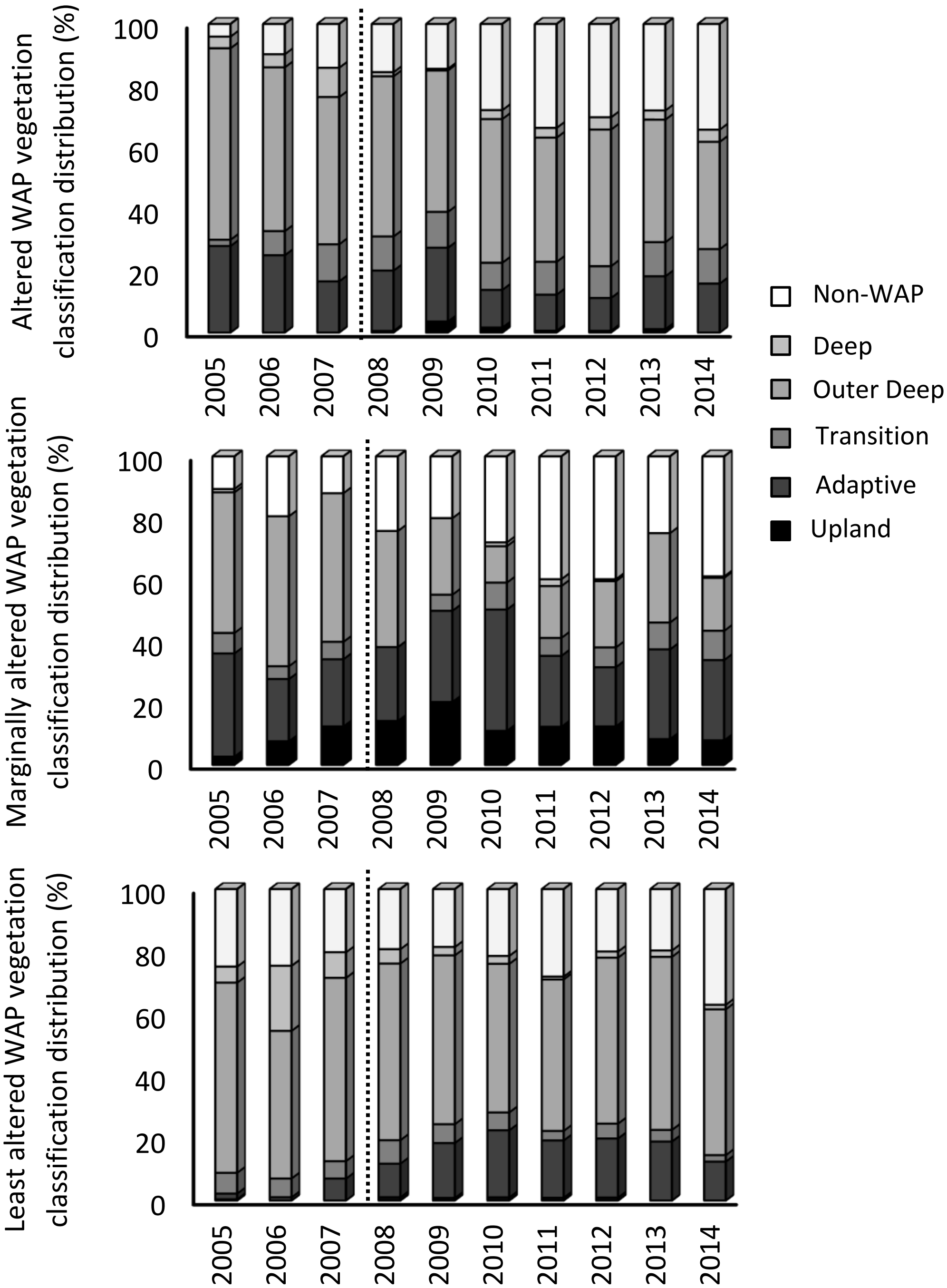

Figure 2.8. WAP and non-WAP vegetation category distribution in the transition zone of wetlands in each hydrologic group. Dashed line indicates the initiation of hydrologic recovery.

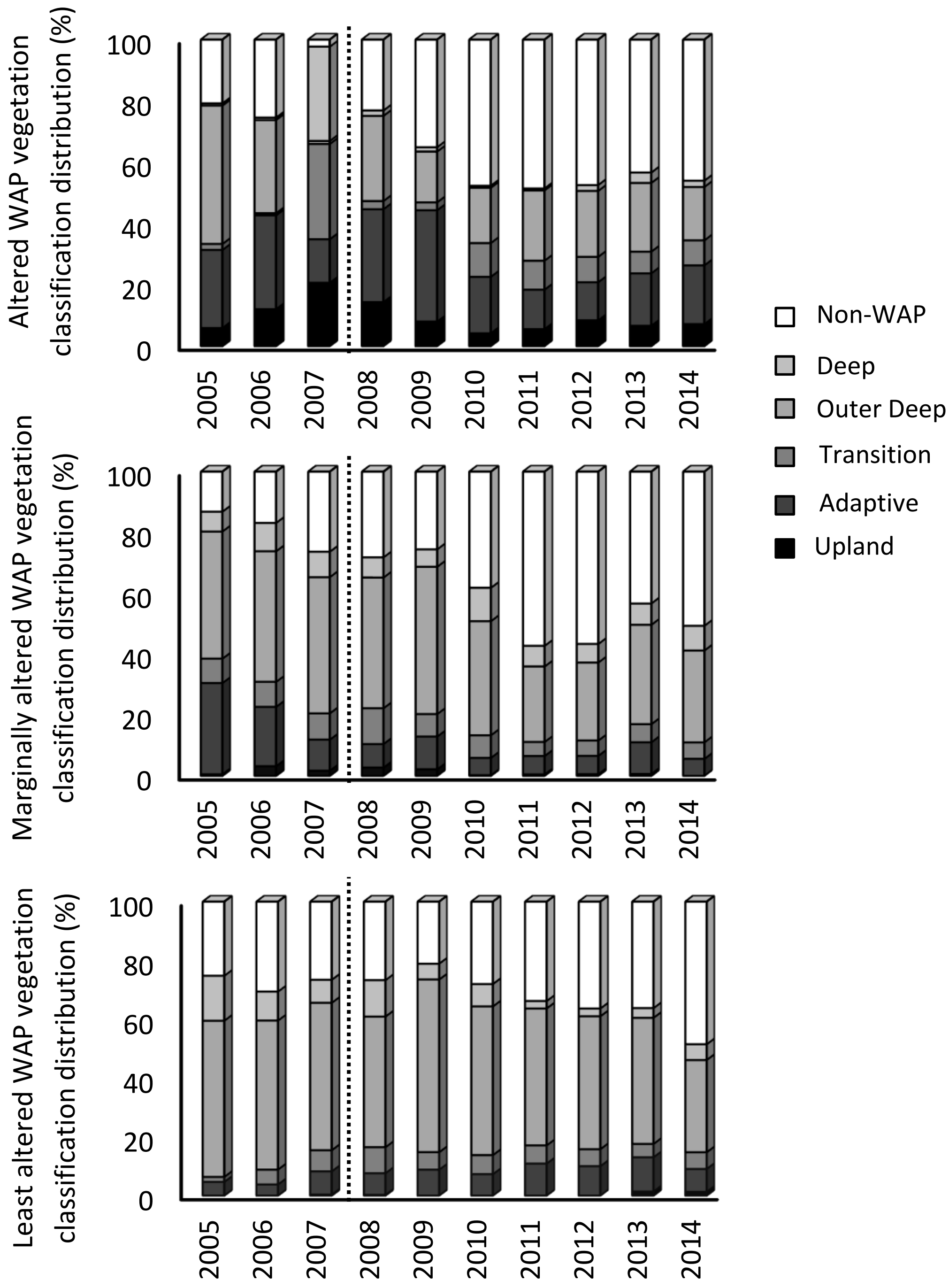

Figure 2.9. WAP and non-WAP vegetation category distribution in the outer deep zone of wetlands in each hydrologic group. Dashed line indicates the initiation of hydrologic recovery.

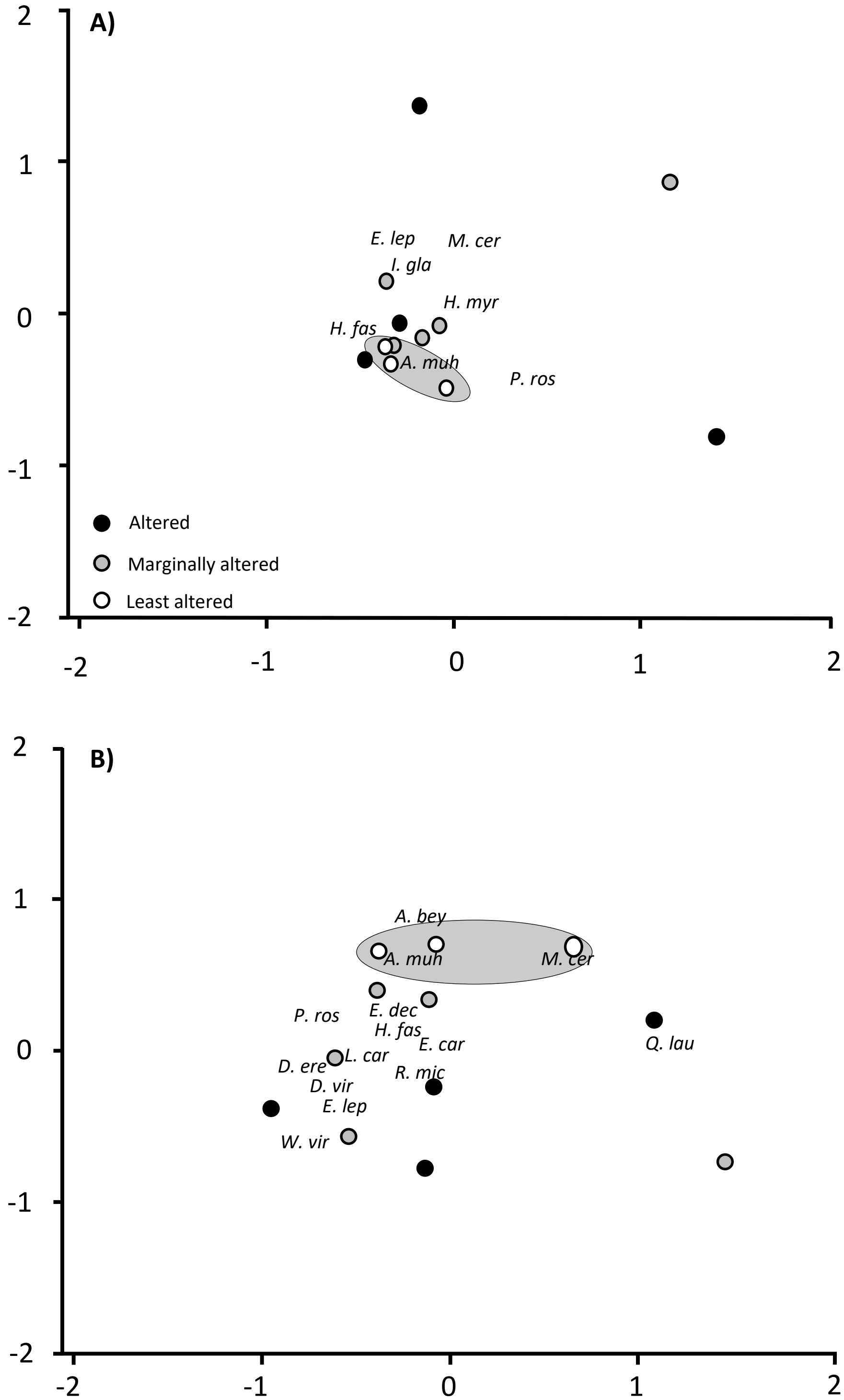

Figure 2.10. NMDS ordination of the transition zone data during the A) pre-hydrologic recovery and B) post-hydrologic recovery periods. The variation in reference wetlands is highlighted with grey ellipses. All species were used in ordination but only species that occurred in multiple wetlands were denoted in plots. Full species names are listed in Appendix Table 2. A4.

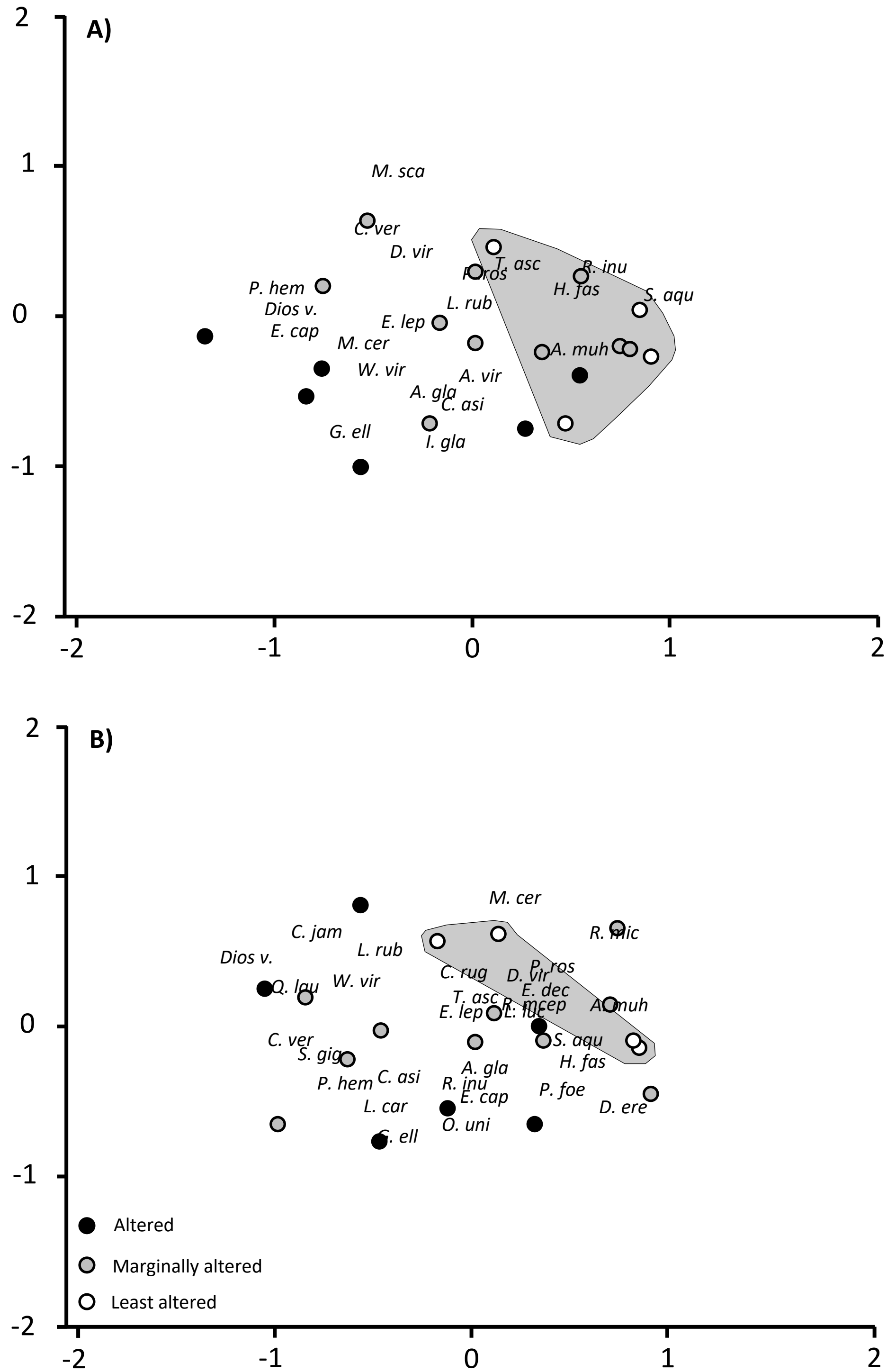

Figure 2.11. NMDS ordination of the outer deep zone data during the A) pre-hydrologic recovery and B) post-hydrologic recovery periods. The variation in reference wetlands is highlighted with grey ellipses. All species were used in ordination but only species that occurred in multiple wetlands were denoted in plots. Full species names are listed in Appendix Table 2. A4.

CHAPTER 3: IMPACTS OF GROUNDWATER ALTERATION AND RECOVERY ON WETLAND SOIL CONDITIONS IN CYPRESS DOMES WETLANDS OF WEST-CENTRAL FLORIDA

ABSTRACT

Soils and soil processes are often overlooked in wetland restoration efforts. We investigated if hydrologic alterations associated with groundwater pumping in a municipal wellfield affected decomposition rates in wetland soils. Using 12 cypress dome wetlands, above- and belowground decomposition rates were determined in transition and outer deep wetland zones using surrogate organic material. In each zone, flood duration, flood frequency, anaerobic conditions, pH, and soil organic matter were evaluated to investigate decomposition patterns within three hydrologic groups of various hydrologic conditions (altered, marginally altered, and least altered). Altered sites exhibited fewer days of inundation in transition and outer deep zones (46±22 days and 132±33 days, respectively) compared to marginally altered (187±26 days and 314±19 days, respectively) and least altered wetlands (218±40 days and 326±17 days, respectively). Altered wetlands exhibited greater decomposition, measured as percent mass remaining, in the aboveground outer deep zone material (73.3±4.0%) and in the belowground transition zone material (62.0±4.5%). Soils in altered wetlands were composed of significantly greater percent soil organic matter in both transition and outer deep zones (13.7±2.8% and 15.9±2.5%, respectively) than least altered wetlands (6.0±1.7% and

9.1±1.6%, respectively). Pairwise comparison suggests decomposition was greater belowground (69.8%±1.5% percent mass remaining) than aboveground (75.8%±1.7% percent mass remaining). Analyses indicate that marginally altered wetlands were comparable to least altered wetlands. Multivariate analysis suggests that soil decomposition in altered wetlands may have differed from other wetlands due to continued differences in soil characteristics and wetland hydrology. The most impacted wetlands at the wellfield may require additional time and/or augmentation to reach least altered conditions.

INTRODUCTION

The primary goal of wetland restoration is to re-establish the conditions and functions of natural systems (Nation Research Council 1992). The restoration of hydrology is generally the first step in a wetland restoration project (Zedler 2000) and success is often assessed based upon hydrologic and vegetative criteria. However, this approach remains inadequate to gauge functional restoration because it overlooks soil properties (Shaffer and Ernst 1999) which determine important wetland processes (Stolt 2000, Mitsch and Gosselink 2007). For instance, it is understood that many of the functional attributes of wetlands that relate to water quality and ecosystem energetics are directly tied to soil conditions (Stolt et al. 2000, Mitsch and Gosselink 2007, Ballantine and Schneider 2009). For this reason, increased focus on edaphic conditions should be considered in efforts to restore wetlands. Decomposition is the physical and chemical breakdown of complex organic molecules into simple inorganic molecules (Juma 1998). Many wetland soils are characterized by periods of inundation and subsequent anaerobic conditions that slow the decomposition process, leading to soil carbon accumulation (Reddy and Patrick 1974, Brinson 1981). In cypress dome wetlands, a common depressional wetland in the southeastern United States, decomposition is driven by microbial and invertebrate communities that are strongly influenced by hydrology and changes in the soil redox potential (Hefting et al. 2004, Siljanen et al. 2011).

Studies have been conducted on the effects of hydrology and restoration on decomposition rates with varying results. It is widely acknowledged that patterns of wetland hydrology strongly influence decomposition. Mitsch and Gosselink (2007) suggested that optimal decomposition occurs when soil moisture is adequate but slows under continually dry or continually flooded conditions, although results have varied. For instance, Day et al. (1982) found no effect of inundation on decomposition, while others have found that decomposition was fastest in permanent waters due leaching (Herbst and Reice 1982, Hietz 1984). It has also been shown that flood frequency (i.e., the number of flood pulses) combined with duration can affect decomposition rates. Brinson et al. (1981) found accelerated decomposition in freshwater wetlands that experienced cyclic wetting and drying while Lockaby et al. (1996) found that a single brief flooding event, as opposed to more frequent and or longer flooding events, resulted in the greatest decomposition in floodplain wetlands in Georgia. Battle and Golladay's (2001) study within cypress swamps concurs with Brinson et al. (1981), suggesting that multiple flooding events or cycles of wetting and drying may often result in the greatest rates of decomposition in wetlands.

The effect of hydrologic restoration on decomposition is less certain. Some studies suggest that restoration may have no effect on soil functions like decomposition (Balcombe et al. 2005, Alvarez and Becares 2006, Gingrich and Anderson 2011),whereas other studies argue that decompositions rates are affected by differences in hydroperiod and hydropattern (Atkinson and Cairns 2001, Spieles and Mora 2007), wetland age (Spieles and Mora 2007), litter quality (Crawford 2007, Fennessey et al. 2008), and soil conditions such as pH and soil organic matter (Taylor and Middleton 2004, Crawford 2007, Fennessey et al. 2008) that exist between degraded, restored and natural wetlands. However, a consensus as to which type of wetlands (i.e., altered, restored, or natural) experience faster decomposition has not been reached, with some studies finding higher rates in reference wetlands (Atkinson and Cairns 2001, Fenessey et al. 2008) while others report greater decomposition rates in restored or created wetlands (Taylor and Middleton 2004, Crawford et al. 2007, Spieles and Mora 2007).

In Florida and throughout the southeastern United States, groundwater pumping has the potential to alter surface waters and affect wetlands (McCauley et al. 2013). Groundwater pumping is a risk to cypress domes wetlands in west central Florida because groundwater and precipitation drive their hydrology (Rochow 1994, SWFWMD 1996, Rains et al. 2013). In west-central Florida, cypress domes are particularly abundant (Casey and Ewel 1998) and groundwater pumping has historically been the primary source for municipal water (Rains et al. 2013). On wellfields where pumping occurs, there has been a documented decline in water table level associated with groundwater pumping and wetland alterations have been observed (Dooris et al. 1990, Rochow 1998, Metz 2011). One such wellfield is the Starkey Wilderness Park (SWP) wellfield in New

Port Richey, Florida which has historically provided water for the greater Tampa Bay region (Tampa Bay Water 2010). SWP has 14 wells that went into groundwater production in the mid-1970s. Pumping within the park varied in location and rate, thus creating a gradient of hydrologic alteration across the site. Pumping also generally increased and expanded to meet municipal demand, peaking at 49 million liters per day in the early 2000s (Metz 2011). However, alternative water sources were secured in 2008, reducing water withdrawals within SWP nearly 75%. Decreased pumping allowed for some recharge of the aquifer, partially restoring the hydrology of wetlands in SWP (Tampa Bay Water 2009, Metz 2011, Lewis et al. 2015).

Soil subsidence, especially in wetlands near groundwater pumping wells, has been documented within SWP (Tampa Bay Water 2013). The gradient of impact on the wetlands within SWP, in conjunction with long-term hydrologic data, provided an opportunity for an in-depth investigation into how hydrology affects the restoration of soil processes. Investigations into how decomposition rates differ between wetlands with historically different hydrology were conducted using the gradient of hydrologically altered wetlands in SWP. We hypothesized that despite some hydrologic recovery, decomposition rates would differ between wetlands and that the most altered wetlands, which have historically experienced the least inundation, would have the greatest decomposition rates.

METHODS

Site Description

The SWP spans over 3,200 ha in the southwest corner of Pasco County, Florida (Fig. 3.1) and includes over eighty cypress dome wetlands. As a wellfield, the hydrology and wetlands of SWP have historically been affected by decades of groundwater pumping . However, since 2008, SWP has undergone some hydrologic recovery due to a nearly 75% reduction in groundwater extraction pumping. Local agencies, such as the Southwest Florida Water Management District (SWFWMD) and Tampa Bay Water (TBW), monitor the hydrology of the wetlands within the park using groundwater wells and staff gauges. Historically, wetland monitoring occurred monthly, however since the mid-1990s wetland hydrology has been measured on a bi-weekly schedule (Fig. 3.2).

Wetland Selection and Hydrologic Groups

Cypress dome wetlands were selected as a common wetland type to evaluate potential differences in soil processes. For this study, two zones (transition and outer deep) were identified based on differences in landscape position relative to historic normal pool elevation. Historic normal pool elevation, which represents the long-term average water level, was established by correlating the elevations associated with biological and physical indicators of hydrology such as tree buttressing, moss collars, and the location of saw palmetto (*Serenoa repens)* which form a fringe surrounding cypress dome wetlands (SWFWMD and TBW 2005, Table 3. A1). The transition zone spans from the historic normal pool elevation inward to an elevation 0.15 m below the historic normal pool level. The outer deep zone begins 0.15 m below the historic normal pool level and extends inward to 0.3 m below the historic normal pool elevation (Fig. A1). Using wetland elevation (Table 3. A2) and long term hydrologic data, twelve wetlands were selected for study based on their hydrologic alteration histories (Fig. 3.1). Unaltered cypress dome wetlands remain inundated for 180 or more days each year (Casey and Ewel 1998). Therefore, hydrologic groups (altered, marginally altered and least altered) were determined based on the average days of inundation over a 15-year period and the frequency of reference-like yearly inundation (>180 days/year) before and after hydrologic restoration (Table 3.1, Table 3. A3, Table 3. A4). Past and current field observations suggest that these wetlands have also experienced varying degrees of subsidence ranging from 0 to 152 cm (TBW 2013 and personal observations), with altered wetlands experiencing the most subsidence.

Decomposition Analysis

Above- and belowground decomposition was determined in both the transition and outer deep zones in each of the twelve selected wetlands. Aboveground decomposition was monitored using litter bags (Swift et al. 1979) containing uniform material (tongue depressors, Puritan 704, made from *Betula papyrifera*, henceforth 'popsicle sticks'). The use of a uniform material demonstrates the effects of the decomposition microenvironment (Baker et al. 2001) in each site and allows for meaningful comparison of rates among and between sites. Sticks were oven dried at 70°C for 48 hours to a constant dry-weight. Once dried, five sticks were weighed collectively (total weight 10 - 11g) and placed in a nylon mesh bag with 5 cm openings. Nylon rope, fixed with 10 mesh bags, was deployed in each zone of each wetland on 15-16 October 2015 and a single bag was collected from each position at weeks 0, 3, 7, 12, 15, 18, 27, 35, and 55, ending on 6 November 2016. Upon collection, bags were stored on ice until they were transported back to Auburn University where each stick was gently cleaned with distilled water and then frozen until processing. Frozen sticks were later thawed and oven dried at 70°C until constant weight. Dry mass was measured to the nearest 0.01 g.

Belowground decomposition was also monitored using the same uniform material. A 1-m^2 quadrat was established in each wetland zone. The quadrat was divided into ten 0.2-m^2sub-quadrats (two rows of five columns). Individual popsicle sticks, were inserted vertically into the soil at the upper left hand corner of each sub-quadrat. A single belowground stick was retrieved from each quadrat and processed using the schedule and methods described above.

Final popsicle dry mass in both above- and belowground experiments were used to determine the percent of the original popsicle mass that remained. The percentage of mass that remained was determined by:

$$Percent\ Mass\ Remaining = X_t/X_0 \times 100\%$$

Where X_t is the mass of the popsicle stick at time t and X_0 is the initial mass of the popsicle stick

In each zone of the aboveground study, mass lost was the collective change in the weight of the 5 sticks/bag, while belowground mass lost for each zone was determined by averaging the changes in individual sticks that were retrieved in each quadrat. Decay constants (*k)* were determined using the equation from Olson (1963):

$$X_t/X_0 = e^{-kt}$$

Where X_t is the mass of the popsicle sticks at time t, X_0 is the initial mass of the popsicle sticks, k is rate of mass loss yr^{-1}, and t is the proportion of time in a year that the bag was deployed

Soil Organic Matter and pH

Additional soil measures were collected within each wetland zone in April 2016. Three 15-cm deep and 5.5-cm diameter soil cores were taken at random locations within the transition and deep pool zones. Cores were composited into one sample for each wetland zone. Composite samples were transported on ice and stored at 4°C until preparation and analysis.

Soil pH was determined using a soil to water ratio of 1:2.5 (weight to weight). Ten g of air dried soil (50°C for 72 hours) was taken from each composite sample and added to 25 mL of deionized water. This mixture was then stirred for one minute, allowed to settle for 30 minutes, stirred for one minute, and allowed to settle for 30 minutes before the pH was measured using a pH electrode (McLean 1982, Thomas 1996). Percent soil organic matter (SOM) was determined using the loss on ignition method. Ten grams of oven dried soil (70°C for 72 hours) were ground, sieved, and ignited at 550°C for four hours. Material was re-weighed and mass lost was calculated as the percent soil organic matter as in Schulte and Hopkins (1996). Three replicate measures were taken from each composite for each analysis (pH and SOM).

Soil Oxidation-Reduction Potential

Indicator of reduction in soil (IRIS) tubes, which are PVC tubes coated in Fe oxide paint composed of solid phase ferric (Fe^{3+}) iron, were used as an indicator of in-situ soil oxidation-reduction (redox) conditions in the wetland soils (National Technical Committee of Hydric Soils 2005, Jenkinson and Franzmeier 2006, Catenson and Rabenhorst 2006). Under anaerobic soil conditions, the iron coating is reduced to soluble ferrous (Fe^{2+}) iron revealing the white PVC underneath. The removal of paint is quantified (Jenkinson and Franzmeier 2006, Rabenhorst 2008) and has been used to indicate the extent of iron-reduction in the soil profile.

The IRIS tubes were deployed in all wetlands within both transition and deep pool zones on 8 January 2016. To minimize paint lost via scraping during installation, a 7/8-inch diameter soil probe was used to drill a 30-cm deep pilot hole into which the IRIS tube was then inserted. Three replicate IRIS tubes deployed within a square meter plot and were retrieved after 15 weeks.

In the field, tubes were carefully extracted from the soil and wrapped in newspaper and transported back to Auburn University. At Auburn, the tubes were gently rinsed under distilled water to wash away any remaining soil and air-dried. Once dried, tubes were re-wrapped and sent to the US Army Corps of Engineers Engineer Research and Development Center, Vicksburg, MS for further analysis. Each tube was analyzed using a 1 cm^2 dot matrix to identify the area in which greater than 50% of the Fe^{3+} was removed. Results were reported as total area removed (cm^2) and used to calculate total percent paint removed. Anaerobic conditions were identified when there was >30% paint removal within a 15cm zone that begins within 15cm of the surface (National Technical Committee of Hydric Soils 2015).

Hydrologic Conditions

The inundation period in each of the study wetlands was determined using elevation data and hydrologic records, collected via groundwater wells and staff gauges, from October 2015 to November 2016 (388-day monitoring period, 35 to 36 hydrologic observations per wetland). As an indication of wetland hydropattern, the number of hydrologic pulses, measured as occurrences from wet to dry and dry to wet, was also calculated for each wetland zone during the study.

Statistical Analyses

To determine differences in decomposition rates, the average percent masses of the popsicle sticks remaining in the aboveground and belowground experiments at the end of the 388 days were compared using a Welch's t-test. Decomposition, measured as the percent mass remaining at 388 days, was compared between each hydrologic group using ANOVA and Tukey's HSD. Prior to ANOVA, decomposition data were natural log transformed to meet the normality assumption. The various soil, and hydrologic, conditions were determined for each zone in all wetlands and then compared using ANOVA and Tukey's HSD. In instances where the assumption of normality was violated for soil parameters, IRIS, and hydrologic pulse data, square root transformations were performed prior to analysis. All statistics were run in R and were considered significant when $p < 0.05$.

Principal component analysis (PCA) was used to determine potential associations between decomposition rates, environmental data and wetland hydrologic groups. Data used in the PCA included percent mass remaining at the conclusion of the study, soil pH, percent SOM, and the number of days inundated during the decomposition experiment.

Based on variability in the units of the soil parameters, a correlation matrix was developed and only components with eigenvalues >1 were considered (Quinn and Keough 2002). The data from the above- and belowground experiments were initially ordinated separately in environmental space. However, ordination was similar in both experiments so data were combined to better detect overall trends in decomposition, environmental parameters, and hydrology.

RESULTS

Decomposition Analyses

At the end of the 388-day experiment there were notable differences in the percent mass remaining between the aboveground (75.8±1.7) and belowground (69.8±1.5) decomposition experiments based Welch's t-test ($t = 2.10$, $p = 0.04$). Both experiments displayed declines in mass over time, however, the belowground material experienced greater decomposition than the aboveground experiment (Fig. 3.3).

Both transition and deep pool wetland zones of each hydrologic group exhibited decomposition over time. There were no significant differences in the final percent mass remaining between the three hydrologic groups in the transition zone for aboveground material however, there were significant differences in the outer deep zone ($F = 6.26$, $p < 0.01$). Altered (73.3±4.0%) and marginally altered wetlands (71.0±3.5%) experienced greater decomposition than least altered wetlands (87.1±0.5%) (Fig. 3.4; $p = 0.03$ and $p < 0.01$, respectively).

In the belowground experiment, there were significant differences in the final percent mass remaining in the transition zone ($F = 5.65$, $p = 0.01$). Altered sites (62.0±4.5%) had significantly greater decomposition than least altered (79.3±3.0%, $p < 0.01$) wetlands. There were no differences in decomposition between the three groups in the outer deep zone of the belowground experiment (Fig. 3.5). Decay coefficients for the individual wetlands ranged from 0.11 yr^{-1} to 0.78 yr^{-1} and average decay coefficients (k) were calculated for each group in all experiments (Table 3.2).

Edaphic and Hydrologic Conditions

Soil and hydrologic parameters for each hydrologic wetland group displayed several significant differences (Table 3.3). Total paint removal from the IRIS tubes ranged from 0 to 100% the study and anaerobic conditions required for hydric soil identification were met (30% removal within a 15-cm zone) in all but one of the altered wetlands (Berkowitz and Nobel 2015). Soil pH values across all wetlands were acidic, as observed in many cypress dome wetlands, ranging from 3.71 to 4.00. There was no significant difference in the pH between the transition and outer deep zone ($F = 0.92$, $p = 0.34$). Additionally, there were no significant differences in pH between the wetlands of various hydrologic alteration history ($F = 1.70$, $p = 0.19$). Soil organic matter (SOM) in the wetlands ranged from 1.5 to 16.0%. There were significant differences in the SOM between the two wetland zones, with average SOM of 10.2±0.8% in the outer deep and 6.4±0.6% in the transition zone ($F = 46.69$, $p < 0.001$). In the transition zone, altered wetlands had significantly higher SOM than marginally altered ($p = 0.03$) or least altered ($p = 0.02$) wetlands. In the outer deep zone altered wetlands, SOM remained significantly higher than least altered wetlands ($p = 0.04$).

Average inundation in the 12 wetlands ranged from 46 - 218 days during the study period (12% - 56% of the experiment) in the transition zone and 132 - 326 days (34% - 84% of the experiment) in the outer deep zone. Average hydrologic pulses ranged from 1.8 to 7.8 times in the transition zone and occurred, on average, 4 times in the outer deep zone (Table 3.4). Least altered and marginally altered wetlands experienced significantly greater inundation than altered wetlands in both the transition ($p < 0.01$ and $p = 0.02$, respectively) and outer deep zones ($p < 0.001$ and $p < 0.001$, respectively). There were no observed differences in the number of hydrologic pulses between the three hydrologic groups in the outer deep zone. However, in the transition zone altered wetlands experienced fewer hydrologic pulses than the marginally altered ($p = 0.01$) or least altered wetlands ($p <0.01$).

Multivariate Decomposition Analyses

Using wetland zone-level data, the PCA described 62.18% of the total variance. Component one was negatively associated with inundation (loading: 0.66) and final percent mass remaining (loading: 0.71) and accounted for 30.85% of the total variations. Component two accounted for 30.42% of the total variation and was negatively associated with pH (loading: 0.73) and positively associated with percent SOM (loading: 0.60) (Fig. 3.6). From the ordination plot, it was observed that there was substantial overlap between least altered and marginally altered wetlands, however altered wetlands often occupied a different portion of the ordination space along both PC axes (Fig. 3.6).

DISCUSSION

The influence of past hydrologic alterations and restoration on soil processes such as decomposition are likely important but have been less considered in efforts to restore wetland function (Shaffer and Ernst 1999, Stolt 2000, Zedler 2000, Ballantine and Schneider 2009). Our results indicated that decomposition rates tended to be highest in the altered wetlands, as we hypothesized, suggesting that differences in inundation patterns may be influencing decomposition rates. However, results suggest that a combination of historic and current hydrologic conditions influence differences in decomposition rates, with inundation in the altered wetlands remaining significantly less than inundation in the least altered wetlands. Additionally, greater soil organic matter accumulation has occurred in the altered wetlands than in least altered wetlands. These two factors, less inundation and greater SOM, may be interacting with one another and may explain why altered wetlands experienced accelerated decomposition.

There were strong trends in soil condition among the three wetland hydrologic groups as demonstrated by the results of this study. Soil and hydrologic conditions generally overlapped between the marginally altered and least altered sites, suggesting similar soil conditions. However, the altered sites had much greater variability in their soil and hydrologic conditions and based on the PCA results (Fig. 3.6) appeared to be diverging into two groups related to decomposition. The first group includes altered wetlands with high percent soil organic matter and intermediate decomposition and inundation, and the second group includes drier wetlands with intermediate to low soil organic matter and high decomposition. It appears that among the altered wetlands, those with higher organic matter, group one, exhibited decomposition rates comparable to the other hydrologic groups. Altered wetlands with the highest decomposition rates, group two, (i.e., >2 on PC1; Fig. 3.6) had lower amounts of SOM. In this case, the additional SOM may be holding more moisture (Hudson 1994) which could compensate for less inundation and slow decomposition in altered group one more than in altered group two Soil organic matter accumulation, which is influenced by decomposition, hydroperiods, and primary production (Debusk and Reddy 1998, Collins and Kuehl 2000, Atkinson and Cairns 2001, Hernandez and Mitsch 2004) seems to be playing an important role in these wetlands and should continue to be monitored as wetland restoration progresses. As these altered wetlands age or if hydroperiods continue to increase we might expect to see declines in decomposition rates to that of least altered like conditions in altered group one. An increase in SOM is also anticipated as wetland hydrology continues trending toward least altered conditions.

It is well established that prolonged inundation inhibits decomposition (Reddy and Patrick 1975, Barlocher et al. 1978). During our study, least altered wetlands experienced inundation 53% of the experiment in the transition zone and 84% of the experiment in the outer deep zones. This degree of inundation caused anaerobic conditions, as demonstrated in the IRIS tube results, which likely inhibited decomposer activity (Foulquier et al. 2013) and resulted in lower decomposition rates. Altered wetlands experienced less inundation than least altered conditions even though groundwater pumping has been reduced by nearly 75% (Metz 2011) since 2008. The reduction in pumping appears to have resulted in greater hydrologic recovery in the marginally altered wetlands. Mean measures of inundation in the marginally altered wetlands were statistically comparable to the least altered wetlands during the experimental period. This was noteworthy given the hydrologic differences between marginally and least altered wetlands before 2008 (Table 3.1). The hydrologic similarity between least altered and marginally altered wetlands is likely the reason wetland decomposition rates between the two groups were comparable in all but the aboveground material deployed in the outer deep zone. Conversely, on average, altered wetlands only experience inundation 12% of the time in the transition zone and 34% of the time in the outer deep zone. There continue to be important hydrologic differences between altered wetlands and the other groups. As a result, altered sites tended to have higher decomposition rates, particularly at sites where there was less SOM. Differences were also detected between the least altered and marginally altered wetlands in the aboveground material of the outer deep zone. While percent inundation between marginally and least altered wetlands were comparable, there may still be important differences in the proximity of the water table to surface soils when wetlands are not flooded that may elicit differences in the soil environment. Drier soils in the marginally altered wetlands could have allowed for greater gas diffusion which influences the microbial activity that drives decomposition (Skopp et al. 1990, Liang et al. 2003, Bossio et al. 2006).

Previous studies suggest that soil conditions respond to restoration at different rates and on different time scales than measures of wetland hydrology or vegetation (Zedler and Callaway 1999, Craft et al. 2003, Spieles 2006, Ballantine and Schneider 2009, Streeter et al. 2017), which seems to also be the case with this study. Measures of pH and redox conditions were not different between the altered and least altered sites suggesting that some soil properties are either 1) not easily altered by changes in wetland hydrology, or 2) respond rapidly following hydrologic restoration (Berkowitz 2013). However, significant differences among wetland hydrologic groups were observed in decomposition and SOM, indicating that these parameters have not reached least altered conditions and may require more time or additional interventional to achieve least altered conditions.

The use of popsicle sticks as a standardized organic material and measures of decay coefficients allows for the comparison with other studies. The average decay coefficients in these wetlands ranged from 0.17 to 0.47 yr^{-1} with an overall average of $0.32 \pm 0.03 yr^{-1}$. These rates were lower than decay coefficients reported in Baker et al. (2001) which used popsicle sticks to examine decomposition rates in floodplain soils in the southeastern United States. However, this is expected since cypress dome wetlands generally exhibit longer inundation and slower rates of decomposition rates (Golladay et al. 1999) than the floodplain communities which experience frequent cycles of wetting and drying. The overall average decay rate form the current study correspond well with the 0.327 yr^{-1} rate reported by Day (1982) in cypress dome wetlands of the Great Dismal swamp. The range of decay rates observed fall within the range of the decay coefficients reported for cypress litter and cypress dome soils (Nessel 1978, Yates and Day 1983, Duever et al. 1984, Brown et al. 1984, Dierber and Ewe 1984).

Although not a specific goal related to this research, by comparing (pairwise) decomposition rates above- and belowground, it was found that decomposition varied between the two locations in the soil profile. Linear regressions described decomposition rates in both the above- and belowground experiments (R^2 = 0.92 and 0.99), however, the belowground material experienced greater decomposition than the aboveground experiment (Fig. 3.2). This was somewhat surprising and likely represents differences in soil moisture regimes and decomposer activities in the soil. While not measured in this study, differences in desiccation during drawdown may differ in the soil compared to on its surface. During drawdown periods, soil organic matter can retain sufficient moisture (Hudson 1994) and support greater soil biological activity than conditions at the surface where increased desiccation occurs. It is also noteworthy that the mesh bags may exclude some macroinvertebrates (Tiegs et al. 2009) in the aboveground experiment. However, visual evidence suggested that this was not the case. Other factors, including differences in litter quality (Finzi et al. 1998, Bardgett 2005) and differences in microbial communities at different levels in the soil profile (Fierer et al. 2003, Bossi et al. 2006, Unger et al. 2009), could also account for the differences in decomposition at and below the soil surface. Regardless of the mechanism, the results of this study illustrate that researchers should consider the source of organic matter and the pathway for its inclusion into wetland soils when interpreting the results and applicability of in-situ decomposition studies.

Following seven years of hydrologic restoration differences remained in decomposition rates in wetlands of various hydrologic histories. Altered wetlands have historically been most affected by groundwater pumping and increased decomposition (as evident from past measured soil subsidence). It appears that these wetlands have been partially restored, however, it is uncertain if these sites will become comparable with least altered wetlands. This, as well as other studies, indicates the need for longer monitoring periods and greater consideration of edaphic conditions to determine wetland restoration outcomes. Based on current soil conditions, least altered conditions may not be attainable or appropriate as restoration goals for highly altered sites.

CONCLUSIONS

Soil conditions and processes are useful indicators of wetlands functionality and should be assessed in wetland restorations. We found that after eight years of passive hydrologic restoration, there were still differences in decomposition rates between altered and least altered wetlands, with the altered wetlands experiencing faster decomposition. Results suggest that differences in decomposition were linked to shorter hydroperiods and differences in SOM content. High SOM content in some altered wetlands may compensate for lower observed rates of decomposition. Researchers should consider the specific pathways that allow SOM to accumulate in the soil profile. Differences in decomposition rates at different locations in the soil profile (above- and below the surface) suggest that different mechanisms and associated environmental factors may be important to restorative processes. Individual wetland response to hydrologic recovery varied in both degree and time-scales suggesting that restoration activities require adaptive management strategies throughout the restoration process.

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Table 3.1. Average inundation period in each hydrologic group as well as the annual frequency at which reference inundation (>180 days/ year in outer deep zone and > 105 days/year in the transition zone) occurred before and after hydrologic recovery.

	Transition Zone			Outer Deep Zone		
	Average Inundation Period (days per year)	Frequency of Reference Inundation		Average Inundation Period (days per year)	Frequency of Reference Inundation	
Hydrologic Group	1990 -2014	1990 - 2007	2008 - 2014	1990 -2014	1990 - 2007	2008 - 2014
Least Altered	> 105	60%	60%	> 180	60%	60%
Marginally Altered	55 -105	30%	50%	80 - 180	30%	45%
Altered	< 55	5%	15%	<80	2%	15%

Table 3.2. Mean (±SE) decay coefficient (*k*, yr^{-1}) at the end of the 388-day study for each hydrologic group for aboveground and belowground material.

	Outer Deep Zone	Transition Zone
Above Ground Experiment		
Altered	0.303 (0.056)	0.368 (0.099)
Marginally Altered	0.331 (0.048)	0.203 (0.024)
Least Altered	0.130 (0.056)	0.332 (0.058)
Below Ground Experiment		
Altered	0.328 (0.044)	0.462 (0.071)
Marginally Altered	0.369 (0.139)	0.334 (0.040)
Least Altered	0.308 (0.109)	0.222 (0.037)

Table 3.3. Mean (±SE) soil and hydrologic parameters. Soil organic matter (SOM) is reported in percent, inundation is reported in days, and pulses are reported as occurrences over the 388-day experiment. F and p-values as determined by ANOVA are listed in the table and different letters represent statistically significant differences, via Tukey HSD, at p<0.05.

	Altered	Marginally Altered	Least Altered	F-value	p-value
IRIS (% paint removal)	49. (10.0)^{a}	88 (4)^{b}	51 (7)^{a}	9.97	< 0.001
pH	4.10 (0.06)^{a}	4.28 (0.08)^{a}	4.23 (0.07)^{a}	1.70	0.19
SOM Transition (%)	13.7 (2.8)^{a}	6.4 (1.1)^{b}	6.0 (1.7)^{b}	5.38	0.01
SOM Outer Deep (%)	15.9 (2.5)^{a}	10.1 (1.5)^{ab}	9.1 (1.6)^{b}	3.67	0.04
Inundation- Outer Deep (# days)	132 (33)^{a}	314 (19)^{b}	326 (17)^{b}	20.46	<0.001
Inundation- Transition (# days)	46 (22)^{a}	187 (26)^{b}	218 (40)^{b}	9.26	<0.01
Pulses- Outer Deep (# pulses)	4 (1)^{a}	4 (3)^{a}	4 (2)^{a}	0.03	0.97
Pulses- Transition (# pulses)	1.8 (0.6)^{a}	6.8 (1.1)^{b}	7.5 (0.7)^{b}	10.78	<0.01

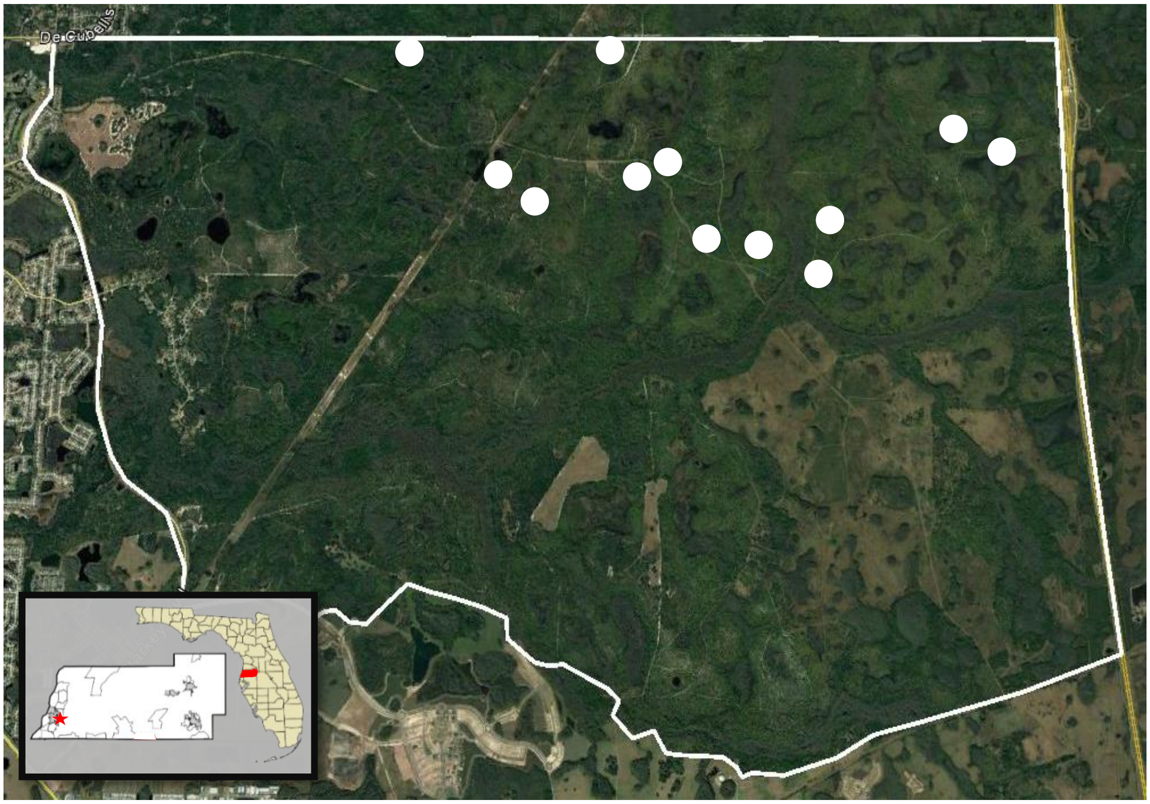

Figure 3.1. Map of J.B. Starkey Wilderness Park (SWP), the white line denotes park boundary; circles identify the location of wetlands used for the study. Inset on bottom left highlights the location of SWP in Pasco County, FL.

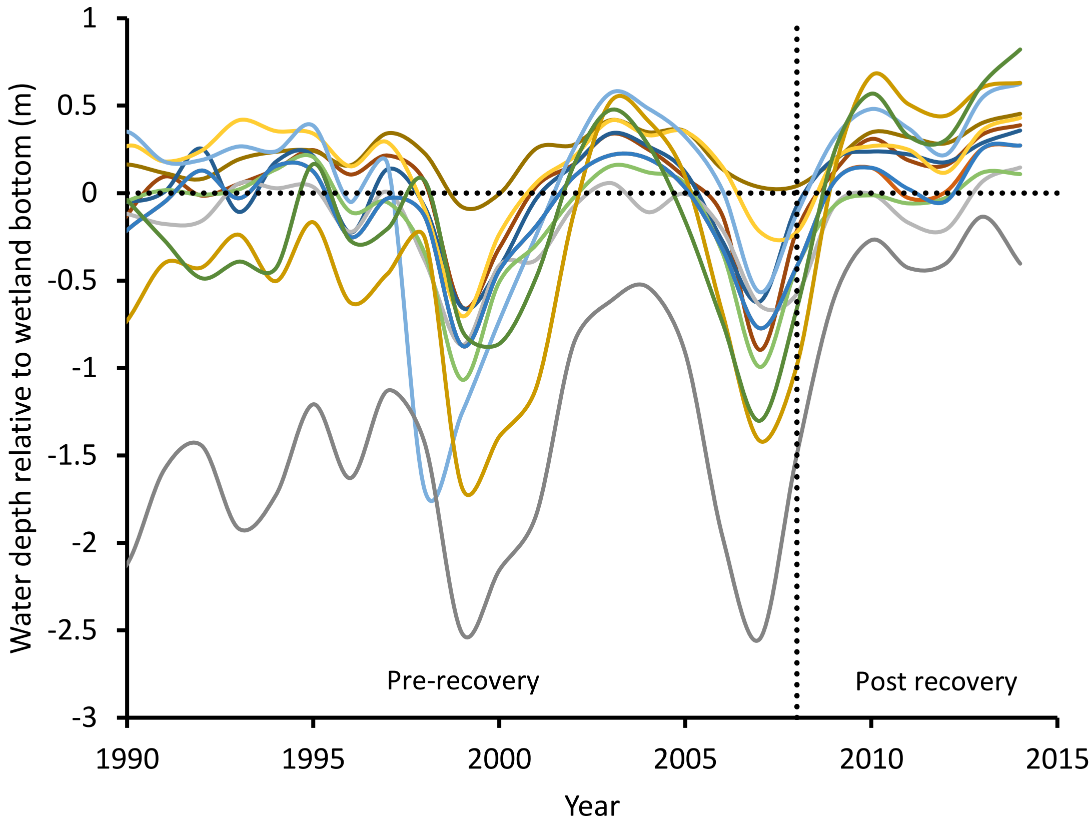

Figure 3.2. Average yearly water depth in each selected wetland from 1990 - 2014. Depths are relative to individual wetland bottoms. The change in groundwater extraction and hydrologic period occurred in January 2008.

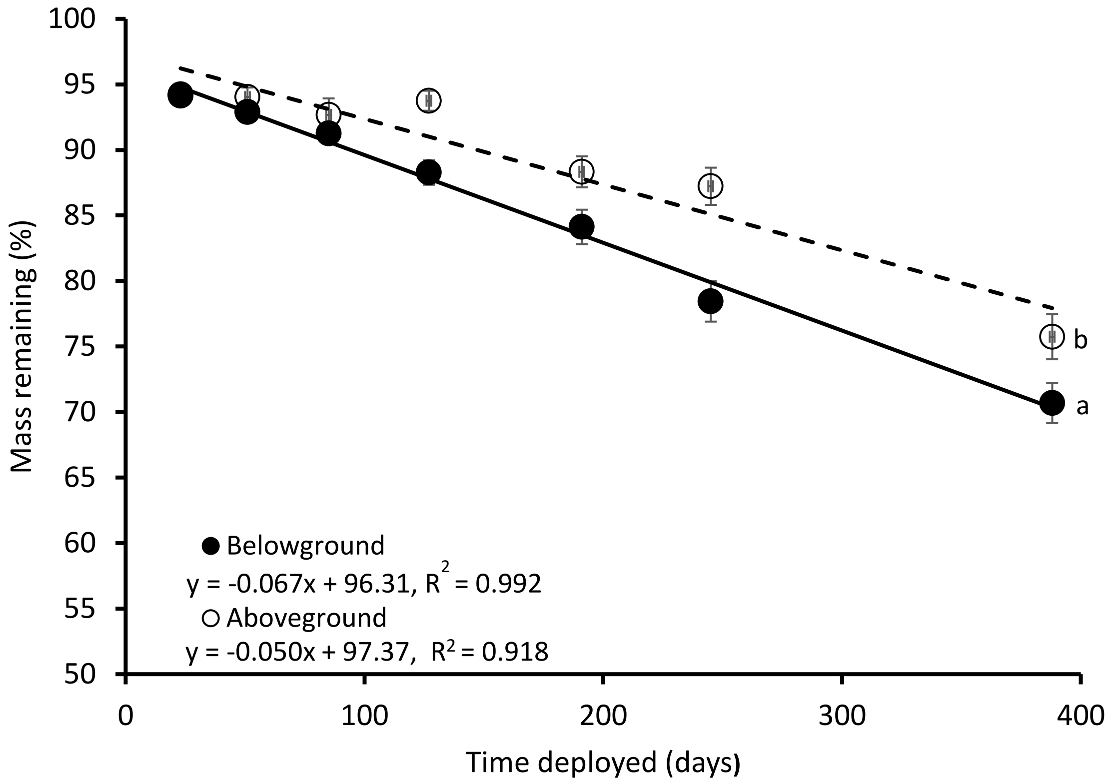

Figure 3.3. Average mass loss and standard error of popsicle sticks in the above- (open circles and dashed line) and belowground (closed circles and solid line) decomposition experiments. Mass loss is expressed as the percentage of the original popsicle mass remaining and for each aboveground (n = 24) and belowground (n=24) point. Different letters represent statistically significant differences in the percent mass remaining at the $p<0.05$ level based on Welch's t-test.

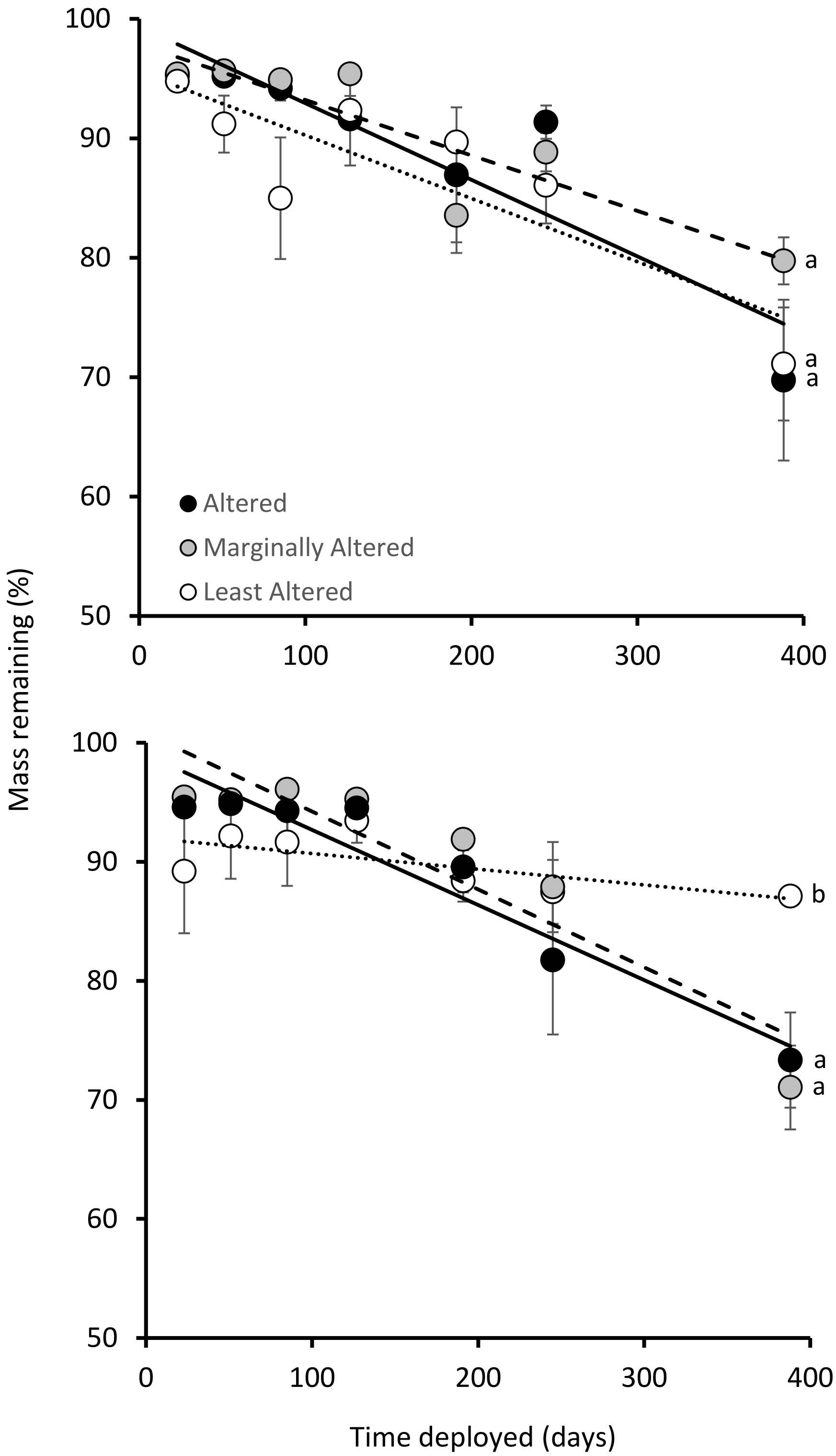

Figure 3.4. Mean (±SE) percent mass loss of the aboveground popsicle sticks in wetlands of different hydrologic groups. Left panel represents decomposition in the transition zone and right panel represents decomposition in the outer deep zone. Different letters represent statistically significant differences in the final percent mass remaining, determined via ANOVA and Tukey's HSD, at the p<0.05 level.

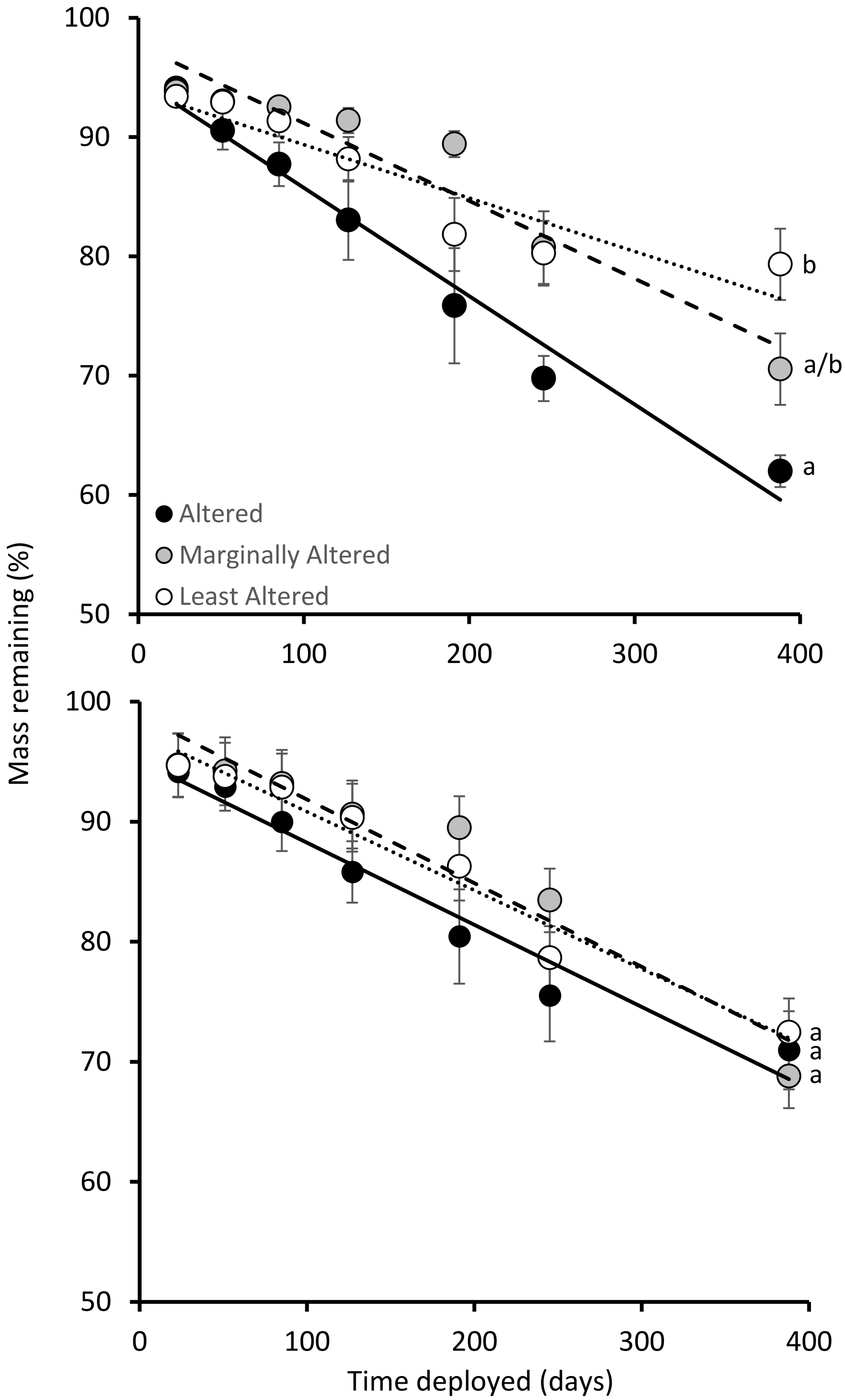

Figure 3.5. Mean (±SE) percent mass loss of the belowground popsicle sticks in wetlands of different hydrologic groups. Left panel represents decomposition in the transition zone and right panel represents decomposition in the outer deep zone. Different letters represent statistically significant differences in the final percent mass remaining, determined via ANOVA and Tukey's HSD, at the $p<0.05$ level.

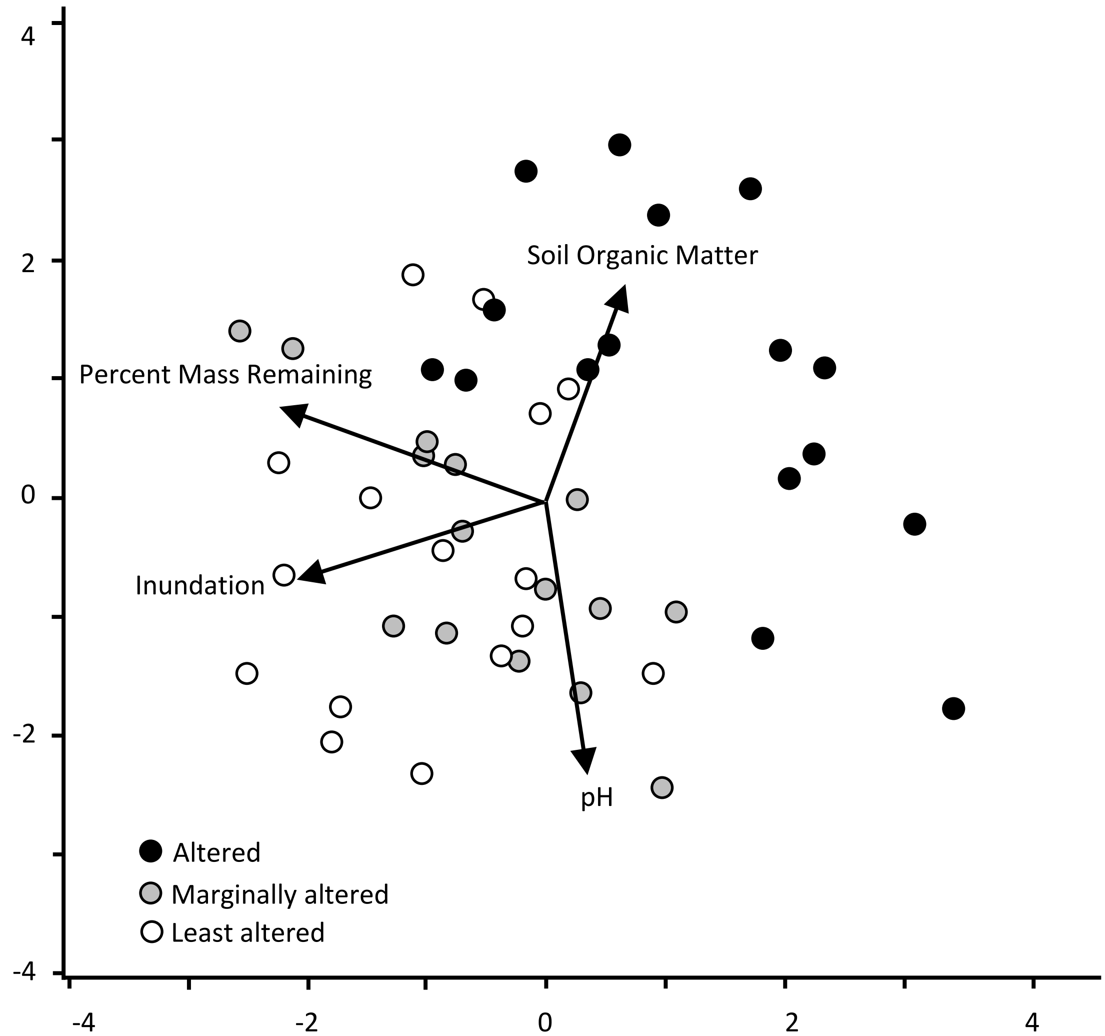

Figure 3.6. PCA ordination plot of the decomposition, soil, and inundation data from both above and belowground experiments in the outer deep and transition zone of the 12 wetlands based on a correlation matrix of association between soil parameters and inundation.

APPENDIX

Table 2. A1. Biological and physical indicators of sustained inundation elevation used to determine the historic normal pool level in cypress dome wetlands. Adapted from WAP Instruction Manual (SWFWMD and Tampa Bay Water 2005).

	Indicators of sustained inundation elevation
1	Elevation of the root crown of mature specimens of fetterbush (*Lyonia lucida*) on cypress trees or hummocks.
2	The inflection point on the buttress of cypress trees.
3	The lower limit of epiphytic byrophytes (moss collars) growing on cypress trees (*Taxodium* spp.)
4	The elevation of the rooted base of saw palmetto (*Serenoa repens*) immediately surrounding the wetland (referred to as the saw palmetto fringe). An offset factor of 0.25 feet must be added to the median value. This indicator may not be reliable for wetland if there is clear evidence that the saw palmetto fringe has been significantly altered by land management practices.
5	The ground elevation of cypress trees growing at the outside edge of the dome. An offset factor of 0.55 feet must be added to the median values.
6	Indicators of hydric soil surrounding the wetland, as determined by a qualified soil scientist. This indicator may not be reliable in a wetland with evidence of significant soil oxidation.
7	Evidence of historic escarpment. This method may not be reliable in a wetland with clear evidence of significant filling along the wetland edge.
8	If none of the above indicators exist, a historic normal pool elevation should be proposed based on any form of evidence thought to be reasonable, including other biologic indicators, aerial photographic interpretation, etc.

Table 2. A2. Annual period of inundation for transition zone wetlands in the study area (1990-2014).

Wetland Identifier	Average Inundation Period	Wetland Hydrologic Group
44	19.12	Altered
54	24.28	Altered
53	26.08	Altered
D	29.84	Altered
99	37.64	Altered
108	42.44	Altered
52	43.64	Altered
73	59.80	Marginally altered
70	73.76	Marginally altered
10	76.20	Marginally altered
65	77.52	Marginally altered
42	80.20	Marginally altered
39	82.36	Marginally altered
38	87.00	Marginally altered
109	88.12	Marginally altered
97	89.08	Marginally altered
75	94.00	Marginally altered
95	99.92	Marginally altered
SC 59	102.04	Marginally altered
76	104.84	Least altered
68	117.28	Least altered
74	133.72	Least altered
64	137.04	Least altered
69	155.6	Least altered
96	158.33	Least altered
89	175.96	Least altered
63	194.52	Least altered

Table 2. A3. Annual period of inundation for outer deep zone wetlands in the study area (1990-2014).

Wetland Identifier	Average Inundation Period	Wetland Hydrologic Group
44	26.04	Altered
D	38.60	Altered
53	39.48	Altered
54	51.84	Altered
52	68.80	Altered
99	73.44	Altered
108	89.68	Marginally altered
42	99.64	Marginally altered
38	105.08	Marginally altered
10	114.28	Marginally altered
39	126.72	Marginally altered
65	132.36	Marginally altered
97	150.28	Marginally altered
109	153.24	Marginally altered
SC 59	156.36	Marginally altered
73	157.04	Marginally altered
75	166.56	Marginally altered
95	171.24	Marginally altered
70	181.88	Least altered
64	184.76	Least altered
69	195.96	Least altered
68	196.32	Least altered
74	205.12	Least altered
76	227.84	Least altered
63	229.48	Least altered
96	229.76	Least altered
89	236.20	Least altered

Table 2. A4. Species and species codes highlighted in the ordination figures.

Plant	Plant code
Amphicarpum muhlenbergianum	A. muh
Andropogon virginicus	A. vir
Andropogon virginicus var. glaucus	A. gla
Aristida stricta var. beyrichiana	A. bey
Carex verrucosa	C. ver
Centella asiatica	C. asi
Cladium mariscus jamaicense	C. jam
Coelorachis rugosa	C. rug
Dichanthelium erectifolium	D. ere
Diodia virginiana	D. vir
Diospyros virginiana	Dios. V
Eriocaulon decangular	E. dec
Eupatorium capillifolium	E. cap
Eupatorium leptophyllum	E. lep
Euthamia caroliniana	E. car
Galactia elliottii	G. ell
Hypericum fasciculatum	H. fas
Hypericum myrtifolium	H. myr
Ilex glabra	I. gla
Lachnanthes caroliana	L. car
Lyonia lucida	L. luc
Mikania scandens	M. sca
Myrica cerifera	M. cer
Oldenlandia uniflora	O. uni
Panicum hemitomon	P. hem
Pluchea rosea	P. foe
Pluchea rosea	P. ros
Quercus laurifolia	Q. lau
Rhynchospora inundata	R. inu
Rhynchospora microcarpa	R. mic
Rhynchospora microcephala	R. mcep
Saccharum giganteum	S. gig
Stillingia aquatica	S. aqu
Taxodium ascendens	T. asc
Woodwardia virginica	W. vir

Table 3. A1. Biological and physical indicators of sustained inundation elevation used to determine the historic normal pool level in cypress dome wetlands. Adapted from WAP Instruction Manual (SWFWMD and Tampa Bay Water 2005).

	Indicators of sustained inundation elevation
1	Elevation of the root crown of mature specimens of fetterbush (*Lyonia lucida*) on cypress trees or hummocks.
2	The inflection point on the buttress of cypress trees.
3	The lower limit of epiphytic byrophytes (moss collars) growing on cypress trees (*Taxodium* spp.)
4	The elevation of the rooted base of saw palmetto (*Serenoa repens*) immediately surrounding the wetland (referred to as the saw palmetto fringe). An offset factor of 0.25 feet must be added to the median value. This indicator may not be reliable for wetland if there is clear evidence that the saw palmetto fringe has been significantly altered by land management practices.
5	The ground elevation of cypress trees growing at the outside edge of the dome. An offset factor of 0.55 feet must be added to the median values.
6	Indicators of hydric soil surrounding the wetland, as determined by a qualified soil scientist. This indicator may not be reliable in a wetland with evidence of significant soil oxidation.
7	Evidence of historic escarpment. This method may not be reliable in a wetland with clear evidence of significant filling along the wetland edge.
8	If none of the above indicators exist, a historic normal pool elevation should be proposed based on any form of evidence thought to be reasonable, including other biologic indicators, aerial photographic interpretation, etc.

Table 3. A2. Historic and zonal elevations for each selected wetland in J.B. Starkey Wilderness Park.

	Elevation (in m) at:		
Wetland	Historic Normal Pool	Transition Zone	Outer Deep Zone
42	9.67	9.52	9.36
44	10.76	10.61	10.45
52	12.16	12.00	11.85
53	12.43	12.28	12.12
54	13.02	12.87	12.71
64	13.18	13.03	12.88
65	12.97	12.81	12.66
68	13.38	13.23	13.08
69	13.60	13.44	13.29
75	14.39	14.23	14.08
76	14.33	14.18	14.02
95	11.77	11.62	11.47

Table 3. A3. Annual period of inundation for transition zone wetlands in the study area (1990-2014).

Wetland Identifier	Average Inundation Period	Wetland Hydrologic Group
44	19.12	Altered
54	24.28	Altered
53	26.08	Altered
52	43.64	Altered
65	77.52	Marginally altered
42	80.20	Marginally altered
75	94.00	Marginally altered
95	99.92	Marginally altered
76	104.84	Least altered
68	117.28	Least altered
64	137.04	Least altered
69	155.6	Least altered

Table 3. A4. Annual period of inundation for outer deep zone wetlands in the study area (1990-2014).

Wetland Identifier	Average Inundation Period	Wetland Hydrologic Group
44	26.04	Altered
53	39.48	Altered
54	51.84	Altered
52	68.80	Altered
42	99.64	Marginally altered
65	132.36	Marginally altered
75	166.56	Marginally altered
95	171.24	Marginally altered
64	184.76	Least altered
69	195.96	Least altered
68	196.32	Least altered
76	227.84	Least altered

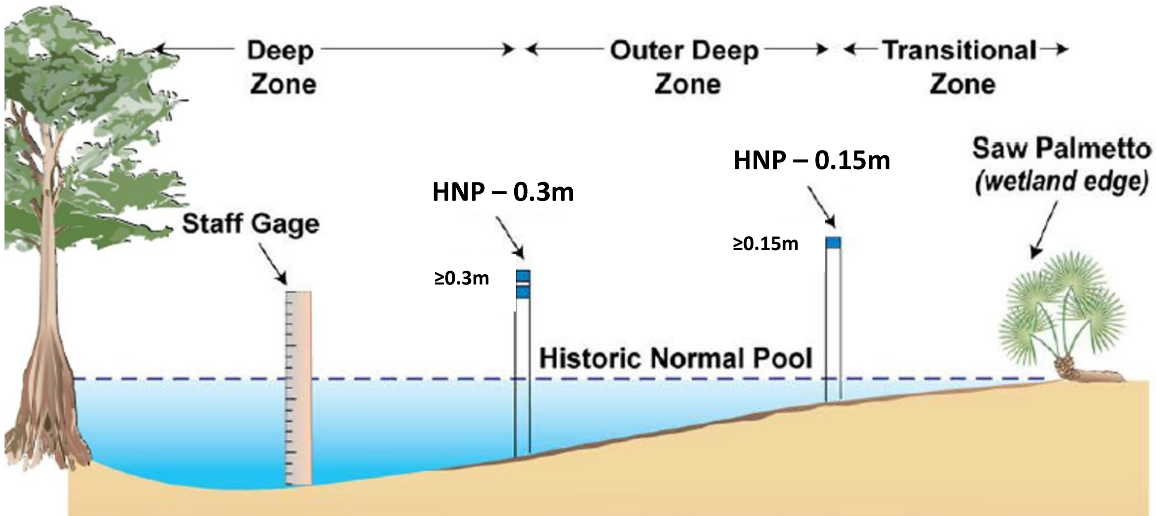

Figure A1. WAP zonation in a typical cypress dome wetland. Adapted from an image provided by SWFWMD. HNP = Historic Normal Pool.