EFFICACY OF CONDITION ASSESSMENTS TO PREDICT ECOSYSTEM FUNCTION IN ISOLATED WETLANDS

By

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A THESIS PRESENTED TO THE GRADUATE SCHOOL
OF THE UNIVERSITY OF FLORIDA IN PARTIAL FULFILLMENT
OF THE REQUIREMENTS FOR THE DEGREE OF
MASTER OF SCIENCE

UNIVERSITY OF FLORIDA

2009

ACKNOWLEDGMENTS

Numerous individuals contributed to the completion of this project. Specifically, I would like to thank Siobhan Fennessy who sparked my interest in this subject. I also thank Chad Foster, Jorge Guevara, Jeff Hull, Larry Korhnak, Dina Liebowitz, Danielle Liebenow, Lauren Long, Danielle Watts, and Justin Vogel for their assistance in the field and laboratory, and moral support to accomplish this endeavor. In particular, I want to thank my major advisor, Matthew Cohen, for his unconditional encouragement, advice, and wisdom, and members of my supervisory committee: Kelly Reiss, Katherine Ewel, and Masato Miwa. Finally, I want to thank my parents, John and Peggy Deimeke, and fiancé, Gregory Parent; I would not be here without you.

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CC	Coefficients of Conservatism
DI	De-ionized
EMAP	EPA's Monitoring and Assessment Program
EMC	Event Mean Concentration
EPA	Environmental Protection Agency
EPC	Equilibrium Phosphorus Concentration
FDEP	Florida Department of Environmental Protection
FQAI	Floristic Quality Assessment Index
FWCI	Florida Wetland Condition Index
IBI	Index of Biological Integrity
LDI	Landscape Development Intensity
LIDAR	Light Detection and Ranging
LOI	Loss-on-ignition
NWI	National Wetland Inventory
ORAM	Ohio Rapid Assessment Method
P	Phosphorus
PSI	Phosphorus Sorption Index
SD	Standard Deviation
SRP	Soluble Reactive Phosphorus
TKN	Total Kjeldahl nitrogen
TP	Total phosphorus
UMAM	Uniform Mitigation Assessment Method
WRAP	Wetland Rapid Assessment Procedure

Abstract of Thesis Presented to the Graduate School
of the University of Florida in Partial Fulfillment of the
Requirements for the Degree of Master of Science

EFFICACY OF CONDITION ASSESSMENTS TO EVALUATE WETLAND ECOSYSTEM FUNCTION

By

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December 2009

Chair: Matthew Cohen
Major: Forest Resources and Conservation

Wetland condition indices are increasingly used to evaluate the quality of wetland habitat and the compensatory mitigation required when wetlands they are impacted. There remain several uncertainties about the use of condition indices in wetland management decisions, and improving wetland condition indices requires that we understand how stable they are in time, and how well they capture wetland functions (e.g., nutrient retention). To address temporal stability of wetland condition indices and using these as predictors of ecosystem function, wetland condition and phosphorus retention were evaluated in isolated cypress domes of northern Florida to establish a clearer connection between ecosystem condition (a measure of ecosystem departure from ecological integrity) and function. Two available condition assessment methods, Florida Wetland Condition Index (FWCI) and the Uniform Mitigation Assessment Method (UMAM), were completed for 23 wetlands located in reference (n = 13), urban (n = 6), and agricultural (n = 4) land use settings in 2001 and 2008. Despite low vegetative similarities between sampling years (23-29% similar using the Jaccard index), FWCI scores did not systematically shift between sampling periods (2001 mean of 37.4 (±18.3 SD); 2008 mean of 34.8 (±12.8 SD)). FWCI scores varied by land use, such that reference sites (42.9 ± 5.4 SD) were significantly different from both agricultural (17.4 ± 12.1 SD) and urban (29.0 ± 7.7 SD). FWCI scores did

occupy a narrower range (10-50) in 2008 (decreasing high scores and increasing low scores) (slope < 1.0), which may be due to moderation of floristic quality: the ratio of coefficients of conservatism (CC) scores of species lost to gained was positively correlated with FWCI scores in 2001 (linear regression, F=12.3, p=0.002). UMAM scores also varied significantly by land use with reference (0.86 ± 0.06 SD) sites different from agricultural (0.50 ± 0.24 SD) and urban (0.47 ± 0.11 SD) sites. Moreover, UMAM scores were positively correlated with FWCI scores (F=35.5, R^2=0.63, p<0.005), illustrating the utility of UMAM as a rapid assessment method.

Standard phosphorus (P) sorption methods (phosphors sorption isotherm (PSI) and total mass P retained) did not show significant differences by land use. The amount of P adsorbed in an assay more closely resembling *in situ* conditions (load = 0.5 ppm P) did vary by land use (Factorial ANOVA, F=25.2, p<0.0001), such that agricultural sites (-10.3 ± mg/kg) tended to release P while urban (1.6 ± mg/kg) and reference (1.2 ± mg/kg) sites retained P. P sorption in these assays was less than that for standard methods. Significant associations exist between P sorption and condition scores such that sites with higher FWCI and UMAM condition scores tended to retain greater amounts of P, but the relationship is driven by severely degraded sites. Removal of sites actively used as cattle pastures renders the association insignificant. Although reference and urban sites had significantly different condition scores, they had similar P retention capacity. Assuming maximum P removal and standing water, these results suggest that condition assessments either overestimate reference site condition or underestimate urban site condition. In either case, the use of these condition assessments in predicting mitigation requirements could lead to an overall reduction in landscape P retention as urban sites are preferentially impacted due to reduced mitigation requirements.

CHAPTER 1
ECOLOGICAL CONDITION AND ECOSYSTEM FUNCTION

Wetlands are valuable ecosystems for the services they provide. Biodiversity, extractable resources, nutrient assimilation, and water regulation are just a few of the numerous services from which humans benefit (MEA 2000, Dodds et al. 2008). Variability among wetlands makes specifying the exact number and magnitude of ecosystem services difficult, but sixteen distinct functions have been identified (Costanza et al. 1997) (Table 1-1). As wetland acreage and quality are increasingly threatened due to changing land use intensity (Brinson and Malvarez 2002, Kentula et al. 2004), the provision of these services is jeopardized. In order to better protect wetland ecosystems and their valuable services, more research is needed in understanding how wetland degradation is affecting ecosystem service provision.

Ecological condition is the departure of an ecosystem from ecological integrity, or a "…balanced integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to the natural habitat of the region" (Karr and Dudley 1981, p. 56). Ecological condition is based on a site's similarity, over some suite of selected attributes, to minimally impacted sites of similar type. Minimally impacted wetlands are referred to as the reference standard, and so every other wetland of similar type within a given region is judged relative to these reference sites. For example, an urban cypress dome located near a golf course would be assessed relative to a cypress dome located in a protected area free of hydrologic impacts and surrounded by intact native ecosystem. The assumption underlying this process is that systems have the highest attainable condition when they are without human disturbances. Previous studies in Florida (Reiss 2006, Reiss and Brown 2007) and Ohio (Lopez and Fennessy 2002, Fennessy et al. 2007, Mack 2007) have shown that condition is responsive to environmental stressors and disturbances such that wetlands located in

minimally disturbed landscapes have the highest condition scores. Modern tools to assess wetland condition include quantification at three levels of measurement intensity (i.e., Level I, II, and III), which are differentiated by expertise and time necessary for completion. Methods requiring less detail and time (Level I and II) are calibrated to and verified by more comprehensive assessments (Level III) (Fennessy et al. 2004).

Species composition is widely used as the basis for assessing ecological condition because it is expeditious and unambiguous to sample. Species diversity, richness, and abundance are the most common metrics used to distill species composition (e.g.,Wilhelm and Ladd 1988, Lopez and Fennessy 2002, Cohen et al. 2004, Reiss 2006, Miller et al. 2006). Additionally, United States Environmental Protection Agency (US EPA) suggests the use of indicators, such as the number of native, rare, threatened, endangered, or nuisance species, for evaluating biotic integrity (Novitzki 1995). For example, the Florida Wetland Condition Index (FWCI) for forested wetlands (Reiss 2006) is comprised of six metrics, five of which are indicators (Table 1-2). Still, other methods assess composition via visual interpretation in which the absence or dominance of invasive species can increase or reduce ecological condition, respectively (e.g., Ohio Rapid Assessment Procedure (ORAM), Mack 2001).

As new species are established and others displaced (i.e., by competition, herbivory, or disturbances), one would expect condition scores to shift as well. In Pennsylvania, wetlands dominated by *Phalaris arundinacea*, an invasive weedy grass, have lower condition scores than systems without it (Miller et al. 2006) suggesting condition is sensitive to vegetation changes; however, there is a lack of knowledge regarding temporal consistency of condition scores due to the absence of long-term monitoring of condition. A shift in species composition should lead to a shift in condition score if overall vegetative indicators or mean site coefficients of conservatism

(CC) scores change. It is plausible as well that condition scores remain consistent despite substantial change in vegetative composition if the species lost over time are replaced by species of similar indicator categories or with similar CC scores. Ultimately, if the natural variability in composition yields small temporal variation in assessment scores compared with the variation among sites exposed to different stressors, then conditional assessment is a robust tool in differentiating between wetland quality.

Florida recently adopted the Uniform Mitigation Assessment Method (UMAM) in February 2004 as the statewide rapid assessment (i.e., Level II) procedure mandated for use in determining wetland mitigation (FDEP 2007). UMAM scores determine mitigation credits for impacted and mitigated wetlands (i.e., restored, enhanced, preserved, or created) and are calculated based on both on-site characteristics, such as vegetation, soil, and water flow patterns, and off-site characteristics, such as surrounding land use and downstream habitats. Although UMAM incorporates metrics common to numerous other assessment methods, its correlation with biotic integrity is unknown as the method has yet to be verified by an intensive, Level III assessment. It is imperative to validate its use as a rapid assessment procedure as its accuracy has implications in protection and replacement of wetland resources. If UMAM fails to predict condition obtained using Level III tools, the implications are large for mitigating losses in wetland acreage and function.

Many condition assessment methods also recognize the importance of structural and functional attributes in contributing to full ecological integrity. The Washington State wetland rating system evaluates the amount of inundation and flow as well as stream connectivity, to evaluate hydrologic functions (WA Stat Dept of Ecology 1993). Similarly, the Montana wetland assessment method rates three hydrologic functions: flood attenuation, surface water storage, and

ground-water discharge/recharge based on indicators including periodic flooding, frequency and duration of flooding, and springs, seeps and inlets (Burglund 1999). In Ohio, ORAM uses horizontal interspersion, or presence and dominance of varying strata to evaluate wetland structure and spatial patterns (Mack 2001). While these characteristics are not included explicitly in Karr and Dudley's (1981) widely recognized definition of ecological integrity, they are considered vital to maintaining ecological integrity consistent with reference standard ecosystems (Fennessy et al. 2004). Ideally, conditional assessment is not just a measure of species composition, but also the "relative ability of a wetland to support and maintain its complexity and capacity for self-organization with respect to species composition, physico-chemical characteristics and function processes as compared to wetlands of similar class without human alterations" (Fennessy et al. 2004, p.3).

It is important to note that measures of condition and function are not necessarily the same thing. Condition, as mentioned above, is a measure of an ecosystem's departure from reference ecosystem attributes typically measured by deviation in community structure. Function refers to the habitat, biological, or system processes of an ecosystem, and consequently, includes a vast array of potential measurements (Costanza et al. 1997). It is plausible that wetland condition measures fail to adequately express the variable levels of wetland function. Further, as every wetland type does not perform every function equally, universal measures of wetland condition across wetland type may over- or under-estimate wetland function. For example, denitrification rates, considered an ecosystem function, are similar between reference and non-reference wetland ecosystems, and thus not reflected in conditional assessment (Stander and Ehrenfeld 2009a).

One difficulty associated with quantifying wetland functions is the potential for off-setting or conflicting functions. In constructed wetlands, for example, shallow depths and large surface areas are associated with increased nitrogen retention and avian biodiversity, whereas small, deep sites are correlated with P retention, but not biodiversity (Hansson et al. 2005). Similarly, water movement and flooding support biogeochemical processes and water storage functions, but also introduce debris and exotic species, which inhibit habitat and biodiversity functions (Ehrenfeld 2004). Wetland resource regulators and managers typically choose to assess condition instead of function because it is comparatively easier to measure, provides a relative score in a larger sample size, and avoids the costs associated with quantifying and integrating multiple potentially off-setting functions into a single score (Carletti et al. 2004, Fennessy et al. 2004).

The relationship of ecosystem function to condition is not well established. In one study, condition was shown to correlate with indicators of function (Spencer et al. 1998). Measurements including bank stability, vegetation width and continuity, water turbidity, conductivity, and color were attributes that informed ecosystem condition assessment in floodplain wetlands in southeastern Australia and were found to effectively predict wetland functions such as nutrient storage and substrate development for biogeochemical processes (Spencer et al. 1998). While this one study found an association exists between ecological condition and indicators of ecosystem function, a well-defined relationship between condition metrics and function has been constrained by a lack of detailed measurements of individual functions and the sheer magnitude of possible functions to measure (Costanza et al. 1997).

Moreover, the relationship between ecosystem function and biodiversity is not linear. Although areas of high biodiversity are resistant to disturbances, and thus more stable

(McNaughton 1977, Chapin et al. 1997), high biodiversity does not always lead to increases in function (Chapin et al. 2000, Schwartz et al. 2000). Rather, measures of ecosystem function tend to maximize at relatively low levels of biodiversity, at least over short duration experiments (Schwartz et al. 2000). Furthermore, tracking species presence or absence may be insufficient in predicting changes to ecosystem processes as interactions between species (e.g., mutualism, predation, herbivory, and competition) can grossly modify energy and material flows (Chapin et al. 2000).

The notion that condition can be used as an indicator of function, or that high condition ecosystems are more valuable because they provide an optimal suite of ecosystem functions, is questionable. For example, a wetland located in an urban setting will usually score low in conditional assessments because of altered hydrology, intensive surrounding land use, and shifts in community composition (e.g., presence of invasive and/or exotic species); however, there is limited evidence to support the claim that urban wetlands fail to provide valuable ecosystem functions (e.g., biogeochemical cycling, water supply and regulation, gas regulation, pollination, or waste treatment). In fact, contrary evidence has been documented: urban and rural riparian ecosystems have the same denitrification potential despite drastically different landscapes (Groffman and Crawford 2003), and denitrification rates are similar between reference and non-reference riverine sites, while higher in non-reference than reference mineral flats (Stander and Ehrenfeld 2009b). Moreover, the functional value of urban sites in removing nitrogen is realized precisely because those wetlands are more heavily loaded than reference systems (i.e., they are "working wetlands"). This presents an important conceptual distinction between site functional potential and the realized function. Sites judged to be in poor condition may provide vital

ecosystem services as a result of their urban or agricultural landscape setting, and therefore, may be just as important to watershed function as those high scoring wetlands in natural settings.

The key knowledge gap this research addresses is the link between existing measures of ecological condition and the multiple ecological functions that wetlands provide. Specifically, we focus on two condition assessment methods, namely the Florida wetland condition index (FWCI, Reiss 2006) and the uniform mitigation assessment method (UMAM, FDEP 2007), and one measure of function: P retention. FWCI has been used in previous wetland surveys, and re-evaluation of the same ecosystems provides answers to the temporal stability of condition scores question. In addition, as the state-mandated procedure in establishing functional loss and gain in the mitigation process, UMAM needs to be verified. In accordance with US EPA guidelines, rapid assessment methods can be verified by more intensive assessment methods (Fennessy et al. 2004). As an intensive vegetation analysis, FWCI can be used to verify UMAM as a condition assessment method. Finally, P retention is a valuable ecosystem service that wetlands provide. The ability of condition assessments to predict P retention will assure minimal landscape scale loss of this service in the course of mitigation.

Table 1-1. Ecosystem services and functions (adapted from Costanza et al. 1997)

Ecosystem Service	Ecosystem Function
Gas Regulation	Regulation of atmospheric chemical composition.
Climate Regulation	Regulation of global temperature, precipitation, and other biologically mediated climatic processes at global or local levels.
Disturbance Regulation	Capacitance, damping, and integrity of ecosystem response to environmental fluctuations.
Water Regulation	Regulation of hydrological flows.
Water Supply	Water storage and retention.
Erosion Control and Sediment Retention	Retention of soil within an ecosystem.
Soil Formation	Soil formation processes.
Nutrient Cycling	Phosphorus storage, cycling, processing and acquisition.
Waste Treatment	Recovery of mobile nutrients and removal or breakdown of excess or xenic nutrients and compounds.
Pollination	Movement of flora gametes.
Biological Control	Trophic-dynamic regulations of populations.
Refugia	Habitat for resident and transient populations.
Food Production	That portion of gross primary production extractable as raw materials.
Genetic Resources	Sources of unique biological materials and products.
Recreation	Providing opportunities for recreational activities.
Cultural	Providing opportunities for non-commercial uses.

Table 1-2. Examples of condition assessment methods and their metrics.

Assessment Method	Level	Metrics
FWCI (Reiss 2006)	Level III, Intensive	1. Tolerant indicator species 2. Sensitive indicator species 3. Floristic quality assessment index 4. Exotic species 5. Native perennial species 6. Wetland status species
ORAM (Mack 2001)	Level II, Rapid	1. Wetland size 2. Upland buffers and surrounding land use 3. Hydrology 4. Habitat alteration and development 5. Special wetland communities 6. Vegetation, interspersion and microtopography
Pennsylvania IBI (Miller et al. 2006)	Level III, Intensive	1. Adjusted FQAI 2. Tolerant plant species 3. Annual species 4. Non-native species 5. Invasive species 6. Trees 7. Vascular cryptogams 8. *Phalaris arundinacea* cover
UMAM (FDEP 2007)	Level II, Rapid	1. Location and landscape 2. Water environment 3. Community structure
WRAP (Miller and Gunsalus 1999)	Level II, Rapid	1. Wildlife utilization 2. Wetland overstory/shrub canopy 3. Wetland vegetative ground cover 4. Adjacent upland/wetland buffer 5. Field indicators of wetland hydrology 6. Water quality input and treatment systems

CHAPTER 2
WETLAND CONDITION ASSESSMENTS: FWCI AND UMAM

Introduction

Wetlands deliver such a variety of goods and services to people that it is important to monitor changes in their condition, which may affect their usefulness. In the U.S. wetland ecosystem services are protected through the no net loss goal in which wetlands impacted by land use disturbances must be off-set through mitigation (FDEP 2009). Mitigation wetlands are preserved, enhanced, restored, or created ecosystems that are on-site or off-site (FDEP 2009). The ecosystem services provided by mitigation wetlands, theoretically, are supposed to account for those services lost by development of the original wetland, thereby satisfying the goal of no net loss.

Wetland impacts can be quantified using condition assessments, or methods that measure ecosystem departure from full ecological integrity. Condition assessments yield condition scores that vary along a disturbance gradient. For example, wetlands evaluated using an adjusted Floristic Quality Assessment Index (FQAI; Lopez and Fennessy 2002, Miller and Wardrop 2005) or vegetative index of biological integrity (IBI; Miller et al. 2006, Mack 2007) produced condition scores that were negatively correlated with indices of disturbance intensity. Similarly, wetlands evaluated using the Ohio Rapid Assessment Procedure (ORAM) had condition scores that were negatively associated with the Landscape Development Intensity (LDI) index, a ranking of human development (Brown and Vivas 2005, Mack 2006). In another study, both the Wetland Rapid Assessment Procedure (WRAP; Reiss and Brown 2007) and the Florida Wetland Condition Index (FWCI; Reiss 2006, Surdick 2006) were shown to correlate negatively with the LDI index (Brown and Vivas 2005). In general, the goals of wetland condition assessments are

to (1) understand the variety of wetlands in the landscape; (2) prioritize sites for conservation or restoration efforts; and (3) determine site mitigation value (Fennessy et al. 2004).

Condition assessment methods are categorized by the detail and scale of the evaluation (Fennessy et al. 2004), and constitute the 3-tier framework from EPA's Monitoring and Assessment Program (EMAP) (Danielson 1998). Level I, or landscape, methods require the least amount of time to complete, contain the least biological detail, and can often be accomplished without a site visit using aerial photographs, land use maps, and topographic maps (Danielson 1998). Level II assessment methods, frequently referred to as rapid assessments, typically require at least brief site visits, and focus on attributes such as landscape setting, ecosystem structure, simple diagnostics of vegetative composition (e.g., presence or absence of key indicator species), and disturbance indicators. They are used regularly for state monitoring programs because they generally require less field time (1-4 hours), sampling, and expertise (Carletti et al. 2004, Fennessy et al. 2004). Finally, Level III intensive assessments typically include detailed characterization of species composition and diversity, often across trophic levels (e.g., vascular plants, benthic algae/diatoms, fish, macroinvertebrates, amphibians, avifauna) (Danielson 1998), and may require multiple days at a site for completion. Level III assessments can include collection of physical and chemical water and soil data (Danielson 1998), but these data are not universally required. Often, Level I and II assessment methods are calibrated and verified by Level III assessments, all of which assume that wetlands respond predictably to stressors (Fennessy et al. 2004). As such, scores generated from remote or rapid assessment metrics should correlate with Level III scores (Miller et al. 2006, Reiss and Brown 2007).

Condition Assessments

Florida recently adopted the Uniform Mitigation Assessment Method (UMAM) as the statewide rapid assessment (i.e., Level II) procedure mandated for use in mitigation

determination (FDEP 2007). Scores from this assessment are critical in determining mitigation credits for impacted and mitigated (i.e., restored, enhanced, created, preserved) wetlands. UMAM evaluates wetlands based on three qualitative metrics, established using best professional judgment: (1) location and landscape support, (2) water environment, and (3) community structure. Scores generated from this assessment are affected by on-site characteristics, such as vegetation, soil, and water flow patterns, and out-of-site characteristics, such as surrounding land use and downstream habitats. Although UMAM incorporates metrics common to numerous other assessment methods, the quality of information it provides is unknown as the method has yet to be verified by an intensive, Level III assessment. It is imperative to verify its use as a rapid assessment procedure as its accuracy has implications in current and future mitigation activities. If UMAM fails to reliably approximate the assessment obtained using Level III tools it could indicate large-scale losses in wetland acreage and function.

Vegetation is a central component of most condition assessments, but it is also dynamic. As new species are established and others displaced (e.g., by competition, herbivory, or disturbances), one would expect condition scores to shift as well. In Pennsylvania, wetlands dominated by *Phalaris arundinacea*, an invasive weedy grass, have lower condition scores than systems without it (Miller et al. 2006), suggesting condition is sensitive to vegetation changes. However, temporal consistency of condition scores is unknown. A shift in species composition should lead to a shift in condition score if overall vegetative indicators or coefficients of conservatism (CC) scores change (Wilhelm and Ladd 1988, Lopez and Fennessy 2002, Miller and Wardrop 2005, Reiss 2006, Mack 2007). It is plausible as well that condition scores remain consistent despite substantial change in vegetative composition if the species lost over time are

replaced by species of similar indicator categories or by species with similar CC scores. Ultimately, if the natural variability in composition yields small variation in assessment scores compared with the variation between sites exposed to different stressors, then conditional assessment is a robust tool in differentiating between wetlands under variable stressors. Tracking condition scores over time not only will help to answer these questions, but can stimulate the refinement of current methods to be more sensitive to ecological change.

Objectives

The objectives of this study are twofold. First, the change in condition scores over time are evaluated. If land use stressors are unchanged, condition scores from 2008 will not be systematically different from scores in 2001. Contributing to the stability of condition scores, vegetation will not be significantly different between sampling years. Secondly, UMAM, a Level II, rapid assessment method will be verified. Condition scores from the UMAM assessment will be highly correlated with those from the FWCI, an intensive vegetation assessment. This would indicate that UMAM is producing condition scores that accurately reflect ecological condition. As condition assessments are widely used tools that influence mitigation processes, it is imperative to understand their nuances better (e.g., change over time and rapid assessment accuracy) in order to prevent large-scale losses in wetland acreage and function.

Methods

Study Area and Site Selection

Twenty-three palustrine depressional forested wetlands (principally cypress domes) in northern Florida were visited between May and the end of October in 2008 (Figure 2-1); all sites were sampled previously (2001/2002; hereafter 2001) as part of a statewide effort to develop wetland condition indices. Sites were located in three main land use settings identified in 2001as

reference (n = 13), agricultural (n = 4), and urban (n = 6) using GIS land use covers and ground truthing (Reiss 2004); land use was verified visually during 2008 site visits and no changes in buffer land cover was observed.

FWCI: Past and Present

In 2008, all sites were assessed using the Level III FWCI assessment method developed in 2001 using biological observations at these and other sites around the state. As in 2001, wetland boundaries were approximated using hydrology and vegetation (Reiss 2006); wetland boundaries were not formally delineated. Four belt-transects measuring 1 m wide radiating from the center to the edge of each wetland in the cardinal directions were used to evaluate community composition. Each transect was subdivided into 5-m sections in which presence/absence of vegetation was recorded, along with information about each species (growth form: aquatic, fern, grass, herb, sedge, shrub, or vine; and category: annual/perennial, evergreen/deciduous, indigenous/exotic; Wunderlin and Handsen 2008). Tree species was also recorded every 10 m along the transect using a ten factor basal area prism.

FWCI condition score was determined from six metrics: (1) tolerant indicator species; (2) sensitive indicator species; (3) FQAI; (4) exotic species; (5) native perennial species; and (6) wetland status species (Reiss 2006). FWCI metrics were computed as described in Reiss (2004); all metrics excluding FQAI were calculated as proportions in which the count of species fitting the particular category (e.g., tolerant, sensitive, exotic) (N) was divided by the species richness (R) of the site:

$$P=N/R$$

FQAI was calculated as the sum of CC scores for all species divided by the species richness of a site (Cohen et al. 2004):

$$FQAI = \sum (C_1 + C_2 + \ldots Cn)/R.$$

CC scores were determined by a survey of expert Florida botanists in 2001 according to species fidelity to particular wetland conditions; CC range from 0 (invasive and/or exotic species) to 10 (species with high affinity for reference standard wetland conditions) (Wilhelm and Ladd 1988). The CC scores reported in Reiss (2004) were used to compute FQAI in this study. Vegetation not identified to species level was excluded from metric calculations because CC scores, exotic/native status and tolerance/sensitivity classifications, all integral to FWCI calculation, are species specific. As in Reiss (2004), each metric was scaled and normalized to fit a 0 to 10 range, and summed to yield a cumulative FWCI score between 0-60 (higher scores indicating higher condition) for each site.

UMAM

UMAM condition scores are based on three metrics: (1) location and landscape support; (2) water environment; and (3) community structure (FDEP 2007). Location and landscape support evaluates surrounding habitats, invasive species presence, wildlife access to and from the site, benefits to downstream fish and wildlife, land uses outside the assessment area, and dependency of downstream habitats on proper wetland function. The water environment metric assesses water levels and flows, water level indicators, soil moisture, soil erosion/deposition, evidence of fire history, vegetation or benthic community zonation, vegetative hydrologic stress, presence of animal species with specific hydrologic requirements, presence of plant species indicative of water quality degradation, standing water quality (discoloration, turbidity, or oil sheen), and light penetration. The community structure metric evaluates the vegetation and structural habitat; plant cover in the canopy, shrub, and ground stratum; presence of invasive species; evidence of normal and natural regeneration and recruitment; natural age and size distribution of species within the system; density and quality of coarse woody debris; land

management practices; topographic features (e.g., ponds, channels, and hummocks); and evidence of siltation or algal growth.

Each metric was rated as a whole number from 0 to 10 (0 defined as "insufficient to provide functions" and10 defined as "optimally supporting functions") employing best professional judgment in accordance with procedural rules. Scores for individual metrics were summed and divided by 30 to yield condition scores between zero and one (FDEP 2007). Calculations for time lags and mitigation credits were disregarded such that the final score reflects current condition only, and not a measure of potential impact or mitigation.

Data Analysis

Vegetation composition similarity between years and sites was evaluated using Jaccard's index of similarity (Wallwork 1976):

$$J = (j/r)$$

where j is the number of species common to both sampling years and r is the total species richness between samples. Jaccard's index of similarity was used to compare change in vegetation composition as it was independent from characteristics evaluated in FWCI and UMAM.

As the same group of wetlands was sampled in 2001 and 2008, paired t-tests were used to ascertain systematic differences in mean FWCI scores between sampling years. Additionally, linear regressions were used to determine presence of correlation between FWCI data in 2001/2002 and 2008, and between FWCI and UMAM data (total scores and individual metrics). Analysis of variance (ANOVA) was used to test differences in mean condition scores by land use for both FWCI and UMAM data; post-hoc Scheffe tests were used to determine differences between individual pairs (e.g., reference and urban). Calculations were completed using Microsoft Excel.

To test whether species changes over time were neutral with respect to floristic quality, the relationship between the CC scores of species lost vs. species gained as a function of site FWCI condition was evaluated. A positive trend would indicate that high condition sites saw a loss of species with high floristic quality, while low condition sites saw a lost of species with low floristic quality. A change of this nature would indicate that sites tend to converge on the median value.

Results

Land use did not change for any sites between sampling periods. Although LDI scores were not recalculated, visual inspection confirmed land use categories as assigned in 2001/2002 (Appendix A). Overall, sites had an average species richness of 27; reference sites with an average of 22 (±7 SD) species, agricultural sites with 35 (±10 SD), and urban sites with 33 (±8 SD) species (ANOVA, $F=6.7$, $p=0.006$). Post-hoc analysis using Scheffe interval test showed differences between reference and urban, and reference and agricultural sites, but no difference between agricultural and urban sites.

Vegetation between sites for different sampling years was not similar. Overall, similarities between sampling years were 0.29 for reference sites, 0.23 for agricultural sites, and for 0.27 urban sites where 1 represents perfect similarity and 0 represents no shared species (not significantly different by land use, Table 2-1, ANOVA, $p=0.4$). Site-site comparisons within the same sampling years were also not similar. Reference sites maintained the highest mean site-site similarity at 0.23 (2001) and 0.20 (2008), followed by urban sites at 0.22 (2001) and 0.20 (2008), and lastly by agricultural sites at 0.17 (2001) and 0.15 (2008). Essentially, sites were more similar to themselves over the eight year time lapse, than with other sites in the same land use category and sampling year. Furthermore, similarities between land use were greater for the reference-urban (0.19) comparison than agricultural-urban (0.16) or reference-agricultural (0.11).

FWCI

The average FWCI score for all twenty-three sites was 34.8 (±12.8 SD) on a scale where 60 represents the reference standard wetland condition. The range of scores was 43.8, with an agricultural site (A1) scoring lowest at 7.2, and a reference site (R10) scoring highest at 51.0. In 2001, the average FWCI score across sites was similar at 37.4 (±18.3), but the range (0 to 58.4) and standard deviation was greater. The lowest scoring site scored zero, while the highest scoring site scored 58.4.

Paired t-test analysis suggested that the mean FWCI scores were not systematically different between sampling periods ($t= -1.5$, $df=22$, $p=0.14$). Within sampling years, FWCI scores were different between land uses (Figure 2-2). In 2008, agricultural sites as a group scored lowest with an average condition of 17.4 (±12.1), followed by urban sites with an average of 29.0 (±7.7), and reference sites with the highest condition average of 42.9 (±5.4) (ANOVA, $F=20.6$, $p<0.005$). Scores by land use were similar in 2001, but with a wider range. In 2001, agricultural sites scored lower, 12.0 (±17), and reference sites scored higher, 49.2 (±9.2), but urban sites scored similarly, 28.7 (±11.6) (ANOVA, $F=18.9$, $p<0.005$). FWCI scores for urban and agricultural sites did not show differences between sampling years, but reference sites did. The average FWCI score for reference sites was 49.2 (±9.2) in 2001 and 42.9 (±5.4) in 2008 (ANOVA, $F=4.5$, $p=0.043$).

Moreover, scores were highly correlated between sampling periods (Figure 2-3, linear regression, $F=158.6$, $R^2=0.88$, $p<0.005$). The 95% CI of the slope of the fitted line (0.63 plus/minus) does not include a slope of one, indicating a systematic convergence in scores towards the median value despite a high correlation. The standard error of the estimate generated by the regression was 4.32 suggesting scores can be accurately predicted within ± 4.32 FWCI points 95% of the time.

CC scores of species lost or gained between sampling years were not different (Table 2-2, ANOVA, $p>0.05$). Overall, the average CC scores of species lost and gained was 3.8 and 3.7, respectively. Species remaining constant between sampling periods had an average CC score slightly higher at 4.0. Interestingly, CC scores of species lost or gained are different across land uses. For example, the average CC score of species lost and gained in reference wetlands was 4.5 and 4.3, whereas in urban land uses the average CC score of species lost and gained was 3.3 and 3.5, and 2.8 and 2.8 in agricultural land uses (ANOVA, $F=29.8$, $p<0.0001$).

There does appear to be a trend suggesting CC scores convergence. The ratio of CC scores of species lost to gained is positively correlated with FWCI scores in 2001 (Figure 2-4, linear regression, $F=12.3$, $p=0.002$). Sites with high FWCI scores in 2001 tended to lose species with higher CC scores than they were replaced with. Similarly, sites with low FWCI scores in 2001 tended to lose species with lower CC scores than they were replaced with.

Species lost or gained according to wetland status (FDEP 2009) was analyzed, in part because the 2001 survey that drove development of the FWCI was during a regionally significant drought, and the 2008 survey was during a period of more normal rainfall. Overall, nearly 60% of species gained were of facultative-wetland or obligate status, whereas 35% were facultative or upland (5% of species status was undeterminable). Thirty-three percent of species lost from 2001 to 2008 were facultative-wetland or obligate, while 19% were facultative or upland (49% of species status was undeterminable). Land use comparisons failed to show significant differences in species lost or gained expect in the gain of facultative or upland species in which case reference systems tended to gain 10% more than urban wetlands; agricultural sites showed no difference from reference or urban land uses (ANOVA, $F=4.07$, $p=0.033$, Scheffe interval).

Finally, the average number of species lost or gained per site was similar. The average facultative/upland species gained was 4.5, while the average lost was 4.1. The average facultative-wetland/obligate species gained was 7.8, while the average lost was 7.0.

UMAM

The average UMAM score across all sites was 0.69 (±0.22). Reference systems, as expected, scored highest with an average score of 0.86 (±0.06), while agricultural and urban wetlands scored similarly at 0.50 (±0.24) and 0.47 (±0.12), respectively (Figure 2-5, ANOVA, $F=29.5$, $p<0.005$). Overall, scores ranged from 0.30 to 0.97. Reference wetlands had the highest and narrowest range (0.77-0.97). The sites in other land uses occupied a lower and wider range: urban wetlands scored between 0.30 and 0.60, while agricultural wetlands scored between 0.30 and 0.77.

UMAM scores correlated with FWCI scores from both 2008 and 2001 (linear regression; $F=35.5$, 34.0; $R^2=0.63$, 0.62; $p<0.005$, <0.005 respectively, Figure 2-6). Standard error of the regression with 2008 FWCI scores was 7.4. Reference sites were tightly grouped in the upper range of both condition assessments, while urban and agricultural sites were loosely distributed along the lower to mid-range of assessments. Individual UMAM metrics were also correlated with total FWCI scores from 2008. UMAM community metric showed the strongest correlation (linear regression, $R^2=0.61$, $F=33.2$, $p<0.0001$), followed by the location and landscape support metric (linear regression, $R^2=0.56$, $F=26.5$, $p<0.0001$) and water environment metric (linear regression, $R^2=0.47$, $F=18.8$, $p=0.00029$).

Discussion

As predicted, sites did not show substantial change in condition as measured by FWCI. All land use categories assigned in 2001 from LDI analysis were verified visually in the field as unchanged in 2008. The absence of land use change is likely a determining factor that kept

condition relatively constant between the sampling years, as human land use is considered a driver of change in wetland condition. Additionally, precipitation did not vary annually between 2001 and 2008, suggesting climate was not a factor in shifting condition scores (SERCC 2007; ANOVA, $F=0.5$, $p>0.05$).

Dissimilarity in vegetation composition between sampling years suggests a moderate, if not substantial, shift in vegetation species composition, contrary to our prediction. An overall mean similarity of 0.27 indicates that vegetation differed by more than 70% between 2001 and 2008. No one land use category change substantially more or less than the others. Rather, vegetation in all three land uses was at least 70% different in 2008 from 2001. In most cases, sites tended to be more similar to themselves across the seven year span than to other sites within the land use, as indicated by a greater similarity for the between year comparison than within year comparison.

Although a 70% difference in vegetation composition between sampling years might suggest a substantial shift in species, other studies report comparable dissimilarities for wetlands. Hartzell et al. (2007) reported Jaccard index similarities for vegetation of 38% between created and natural depression wetlands. While that may indicate a large discrepancy in vegetation, only one of twelve vegetation metrics was significantly different between the two systems. Additionally, invasion of non-native *Phragmites australis* and *Typha* spp. in Great Lakes coastal wetlands was thought to cause a 19% similarity (Jaccard's index) between 2001 and 2004 (T'ulbure 2007). While the authors suggest this is a striking dissimilarity, FQAI and CC scores were greater after the two species were established. Consequently, using Jaccard's index to track vegetation change can provide an alternative measure of community composition and detect changes where condition scores did not.

Of interest is the possibility of a convergence to the mean of current FWCI scores. Sites scoring high in 2001 tended to score lower in 2008, while sites scoring very low in 2001 tended to score higher in 2008. This convergence effect results in a loss of extreme condition wetlands, or those scoring excessively low or high. There are two explanations for this observation. The first is statistical over-fitting of the original scores. Individual FWCI metrics were calibrated to LDI and normalized to occupy the entire score range of 0-10 (Reiss 2004). While the current scores were normalized, we did not set the minimum or maximum scores as end points at 0 or 10, thereby stretching the data to fit the entire range. Doing so would have reduced the observed convergence effect, but also would remove meaning from the scores. Furthermore, the 2001 FWCI assessment was based on a statewide survey, while this study was limited to the northern region.

Secondly, metric analysis indicated a systematic shift in vegetation that would contribute to an overall convergence of FWCI scores. While the mean CC scores of species lost or gained was similar at 3.8 and 3.7, respectively, the ratio of CC scores showed a distinct pattern. The ratio of CC scores for species lost:gained was positively correlated with 2001 FWCI scores. Sites with high FWCI scores tended to lose species of high CC scores, which were replaced by species of lower CC scores. Sites with low FWCI scores tended to lose species of low CC scores, which were replaced by species of higher CC scores. The ultimate effect of this shift in vegetation was a loss of species with extreme CC scores, which contributes to an overall convergence of FWCI scores in 2008. Therefore, while the effect may be due to statistical overfitting of the original FWCI scores in 2001, there is evidence suggesting an overall moderation of vegetative quality in these sites and a loss of sensitive species in reference

standard wetlands. It is unclear if other assessment methods show similar convergence over time as research on the temporal stability of assessment metrics is lacking.

As predicted, UMAM scores were positively correlated with FWCI scores (R^2=0.63) indicating that the information from the rapid assessment comports with assessments provided by FWCI. Similarly, Mack (2000) verified and calibrated ORAM to vegetative IBI methods unique to Ohio. Verification was supported by a simple linear regression of rapid assessment scores to intensive assessment scores. Although the two methods were highly correlated (R^2=0.85), Mack does not state a cut off point at which the method would have been unverifiable. Providing a universal test of verification can standardize the process and lend greater credibility to rapid assessment methods.

It is imperative to note, though, a UMAM score of 0.8 does not automatically indicate an FWCI score in the upper 80%. In fact, our data indicate that a UMAM score of 0.8 can have a widely variable FWCI score between 20 and 50. Similarly, wetlands with FWCI scores in the 20's have UMAM scores ranging between 0.33 and 0.77. Linear regression of UMAM scores with 2008 FWCI scores produced a standard error of 7.4 suggesting UMAM can predict FWCI scores to within 15 points of the actual score. Such large variability was unexpected and suggests a potential need of refining UMAM to produce less score variability (Mack 2007). This confirms the implicit trade-off in using rapid assessments versus intensive assessments. UMAM is fast and requires less detail, whereas FWCI is time intensive and may require plant botanical expertise. While these methods produce similar information when compared over the population of wetlands, UMAM scores may not be useful in predicting FWCI scores for a particular wetland as the error range is larger than expected.

Analyzing UMAM metrics did not substantially improve the strength of the correlation with FWCI, contrary to our prediction. A score of 2 out of 10 in the location and landscape support metric produced FWCI scores ranging from 7 to 40. The water environment metric was a bit less variable: a score of 4 produced FWCI scores from 7 to 30. The vegetation metric showed little variability in predicting low FWCI scores, but increased variability on the higher end, with a score of 8 corresponding to FWCI scores from 17 to 51. While the vegetation metric was the best individual metric to predict FWCI scores, it was not better than the total UMAM score. This reinforces FDEP's (2007) assertion that UMAM should be used as a whole because breaking it down into its constituents is not an appropriate use of the method.

Fennessy et al. (2004) list four essential characteristics of rapid assessments methods: (1) must measure condition; (2) must be rapid (<5 hours in the field); (3) must be an on-site assessment; and (4) must be verified. UMAM successfully satisfied the first three requirements, and although UMAM scores correlate well with FWCI scores, the repeatability of the method is questionable as it makes use of best professional judgment to reach final condition scores. Final condition score could vary with evaluator. To satisfy the final requirement, UMAM would need greater guidelines and substantially less weight on professional judgment. For example, ORAM metrics are clearly defined such that a wetland with an undisturbed buffer of 50 m receives 9 more points than a site with a road running along its border (Mack 2001). While the official UMAM rule provides examples of metric criteria that satisfy certain point breaks at 10, 7, 4, and 0 (FDEP 2007), they are not specific enough to minimize professional interpretation. Limiting evaluator interpretation may be as simple as providing formal UMAM training for professionals, as is available for ORAM (OEPA 2009). Even though the method would still allow for best

professional judgment and exclude it from Fennessy et. al's (2004) definition of a rapid assessment, training sessions would ensure data quality and implementation consistency.

It is rare that wetlands undergo repeated condition assessments over time without a change in disturbance or environmental impacts. Consequently, most literature available can only speak to condition due to a driving force (i.e., land use change or logging). This study provides a valuable analysis of condition variability over time absent new driving forces, and demonstrates a need for long-term monitoring of condition and vegetation beyond mitigation wetlands (i.e., created, restored, preserved or enhanced wetlands). Understanding the natural variability in condition and vegetation will create a benchmark against which mitigation wetlands can be compared.

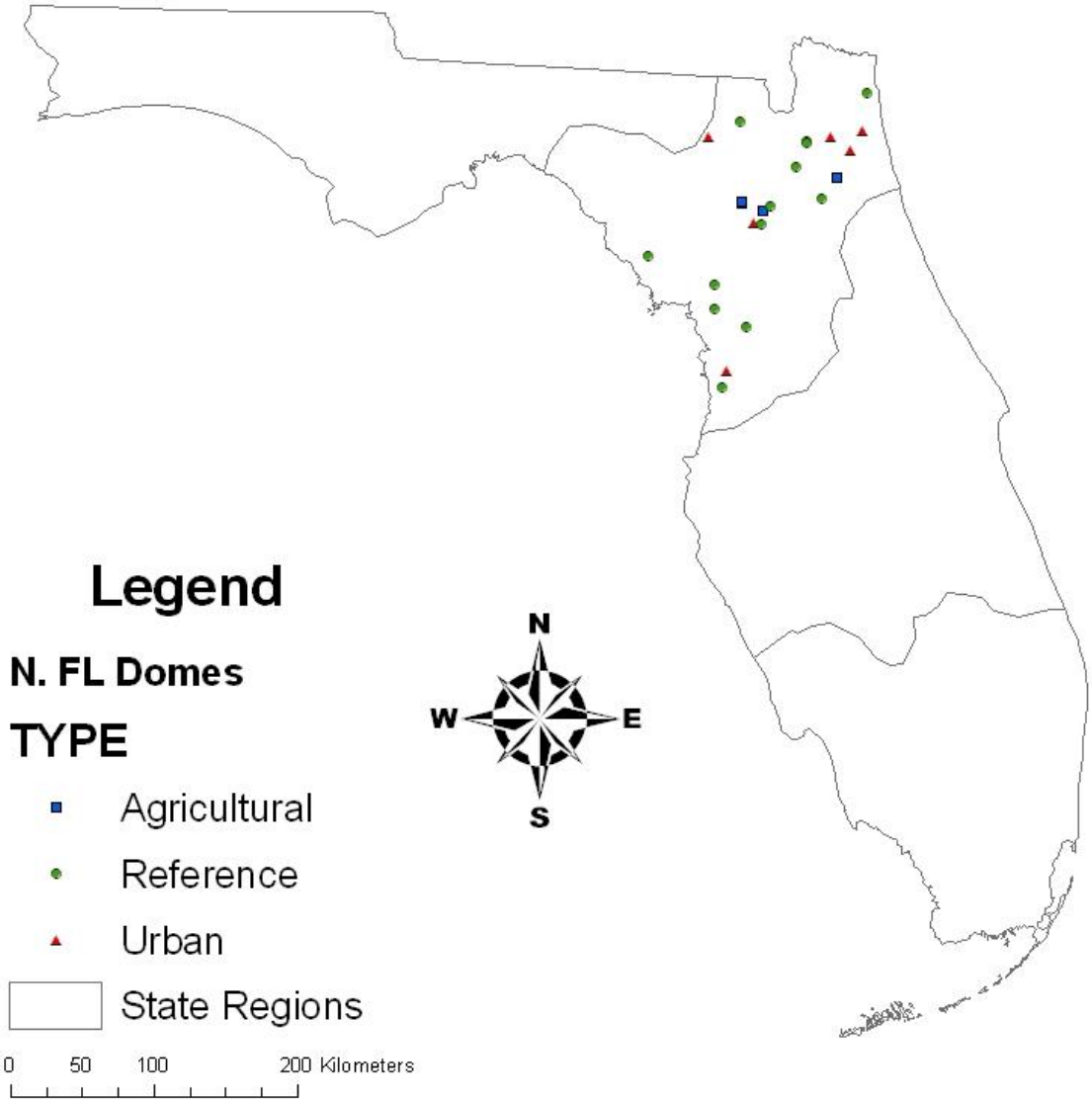


Figure 2-1. Location of 23 wetlands sites in northern Florida, U.S.A. Blue squares represent agricultural sites, green circles are reference standard sites, and red triangles are urban sites.

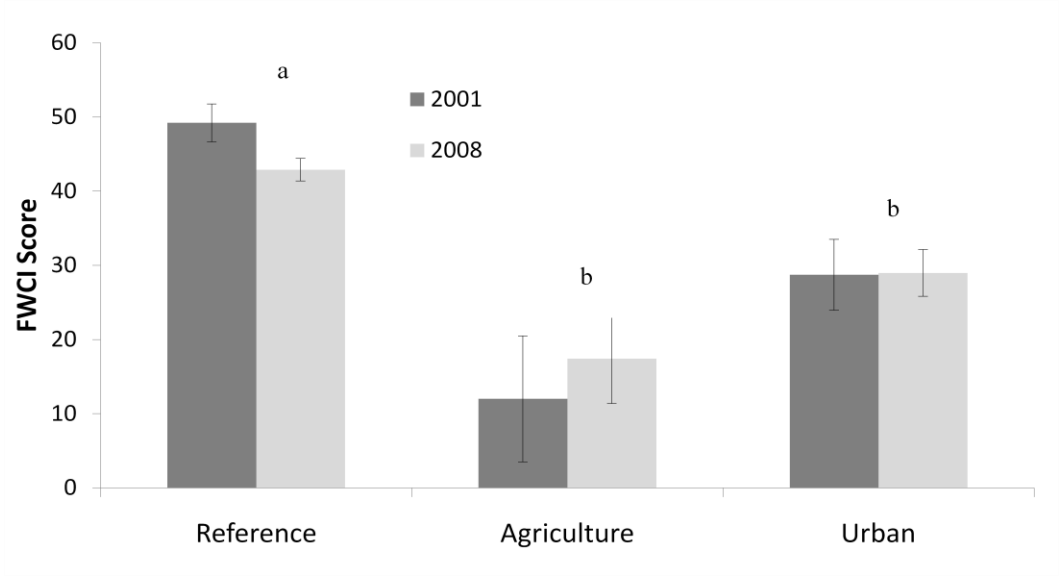


Figure 2-2. Mean FWCI scores by land use from the 2001 and 2008 assessments. For both years, letters indicate significant differences between land uses. Error bars represent standard error. (n=23; 2001/2002: ANOVA, F=18.9, p<0.005; 2008:ANOVA, F=20.6, p<0.005).

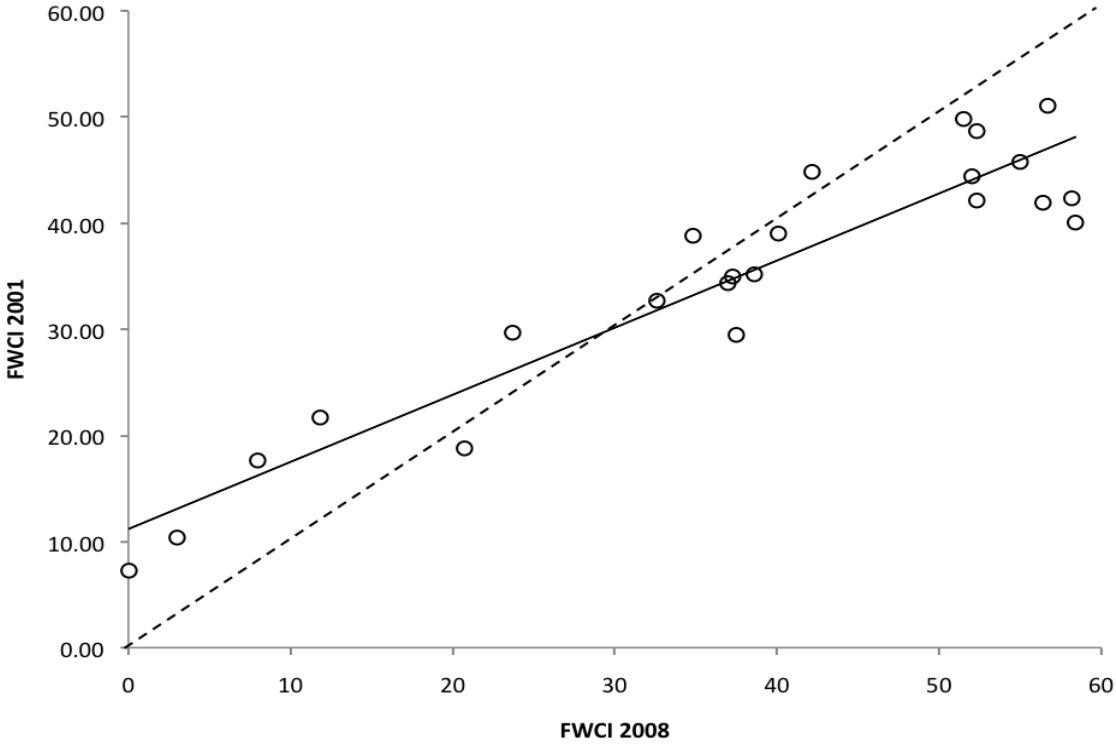


Figure 2-3. Correlation of FWCI scores between 2001 and 2008. Dotted line represents slope of 1 (linear regression, t=12.6, FWCI 2001=0.63*(FWCI 2008) +11.2, R^2=0.88, p<0.005). Fitted line significantly less than one.

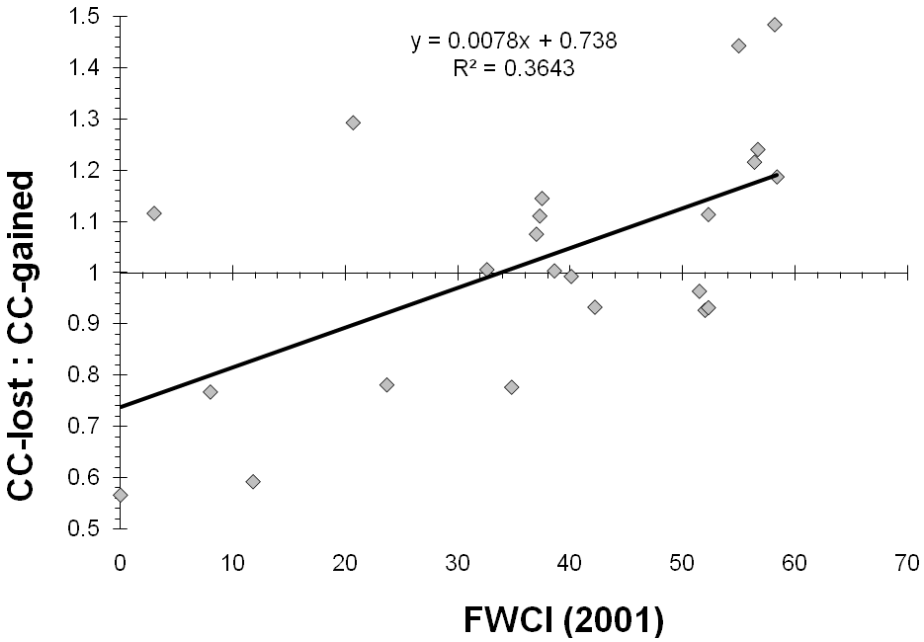


Figure 2-4. FWCI scores from 2001 and the ratio of CC scores of species lost to species gained over seven years later (linear regression, F=12.0, p=0.002).

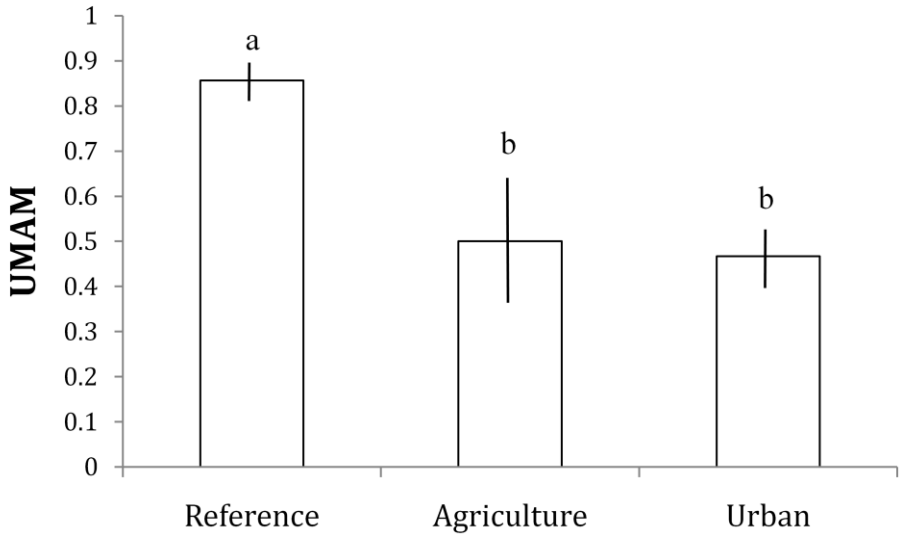


Figure 2-5. Mean UMAM scores by land use; scores were different between land uses. Error bars represent standard error (n=23; ANOVA, F=29.5, p<0.005).

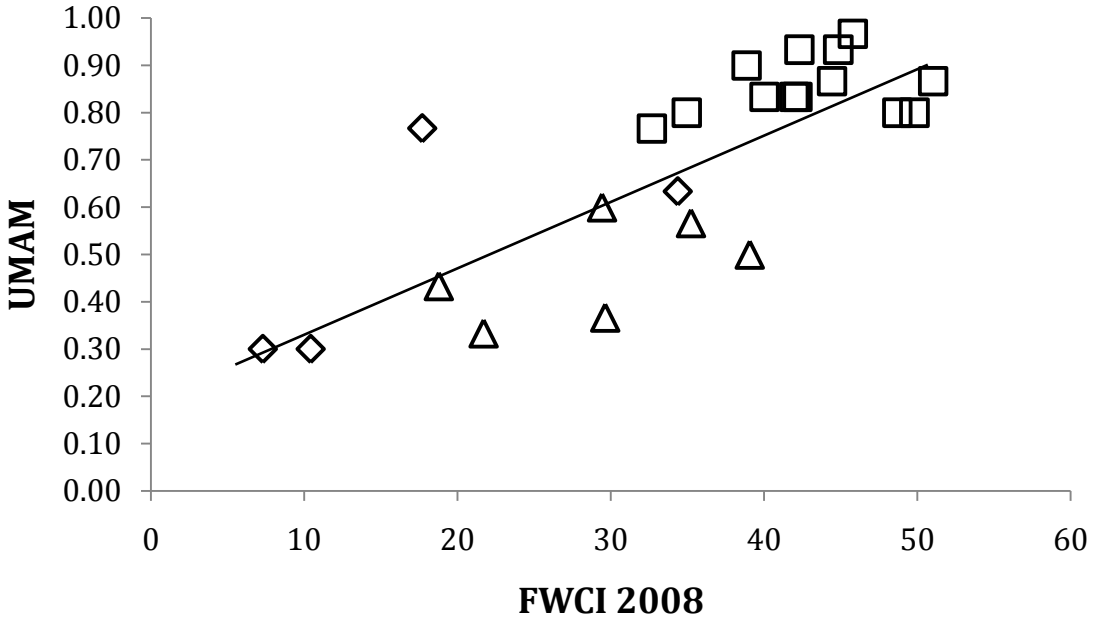


Figure 2-6. Correlation of FWCI and UMAM scores from 2008, sub-divided by land use. Diamonds are agricultural sites, triangles are urban, and squares are reference (Regression; F=35.5, UMAM = 0.014*(FWCI 2008) + 0.198, R^2=0.63, p<0.005)

Table 2-1. Jaccard similarity indicies by land use between 2001 and 2008 (not significantly different by land use, ANOVA, p=0.4). Values represent mean (±SD).

Land Use	Jaccard Similarity Index
Agricultural	0.23 (0.10)
Reference	0.29 (0.07)
Urban	0.27 (0.07)

Table 2-2. CC scores of species lost, gained, or constant between 2001 and 2008. Scores were different by land use (ANOVA, F=29.8, p<0.0001). Values are mean (±SD).

Land Use	Gained	Lost	Constant
Agriculture	2.86 (1.04)	2.80 (1.36)	3.07 (0.46)
Reference	4.30 (0.57))	4.47 (0.53)	4.60 (0.37)
Urban	3.46 (0.51)	3.32 (0.54)	3.87 (0.60)
Overall	3.7 (0.81)	3.8 (1.1)	4.0 (0.77)

CHAPTER 3
PREDICTING WETLAND FUNCTION USING CONDITION

Introduction

Critical among the myriad services that wetlands provide (Costanza et al 1997, MEA 2000, Tong et al. 2007) is the capacity to remove excess phosphorus (P) from runoff (Bruland and Richardson 2006, Dunne et al. 2006, Cohen et al. 2007, Reddy and DeLaune 2007). While P is an important natural nutrient, overabundance of the element can cause water quality problems and downstream eutrophication (Carpenter et al. 1998). Lake Okeechobee in southern Florida has a history of P loading due to agricultural practices in its basin (SFWMD et al. 2004). One mitigation strategy to reduce P loads in the lake included the restoration and creation of historically isolated wetlands throughout its basin (Dunne et al. 2006). These wetlands act as P sinks and reduce the amount of P reaching the lake. Work in Lake Okeechobee basin is still underway, but data suggest that the presence of wetlands aid in reducing the P concentration in the lake (Dunne et al. 2006). Therefore, wetlands are vital ecosystems due to their ability to remove excess P from water, protecting downstream water bodies.

Surrounding land uses affect the composition, structure, and function of wetland ecosystems. Agricultural land uses alter nutrient retention and leaching from riparian ecosystems (Patty et al. 1997, Kuusemets and Mander 2002, Johnson and Rejmankova 2005), increase nitrogen and P concentrations in wetland water column and soil (Berka et al. 2000, Cuffney et al. 2000), and affect the diversity of vegetation, fish, invertebrates, and algae in wetlands (Cuffney et al. 2000, Lopez and Fennessy 2002). Similarly, the introduction of roads near or within wetlands reduced species richness of bird, mammal, herptile, and vegetation assemblages (Findlay and Houlahan 1997, Trombulak and Frissell 1999, Findlay and Bourdages 2000). In Florida, shifts in avian communities from insectivores, ground nesters, and cavity

nesters to ground gleaners, canopy nesters, and exotic species occurred where wetlands were surrounded by high intensity land uses, such as industrial parks (Surdick 2005).

Wetlands positioned in different land uses exhibit clear functional differences in P retention. In general, agricultural land uses tend to have a reduced capacity to retain P, while wetlands located in minimally disturbed locations have higher P retention capabilities (Bruland and Richardson 2006, Dunne et al. 2006, Cohen et al. 2007). Urban and industrial land uses have varied effects on P retention, occasionally showing values not different from reference sites (Cohen et al. 2007). Understanding functional differences between wetlands located in diverse landscapes will lead to a clearer understanding of how land uses impact the provision of ecosystem services.

As P does not have a gaseous form, pathways of P retention are constrained to organic soil accretion and mineral adsorption (Reddy and DeLaune 2007). Wetlands can retain P in both organic and inorganic forms, and the relative importance of the two storage mechanisms varies with wetland type, redox conditions, regional mineralogy, and hydrology (Mitsch and Gosselink 2000). Inorganic forms of P (e.g., phosphate, polyphosphate) are ionically sorbed to minerals, metals, and clays (all with positive surface charge) and can be available or unavailable depending on the affinity of the sorption site for P and the concentration in the water. The quantity and distribution of iron (Fe) and aluminum (Al) influence the number of P binding sites, and the temporal patterns of wet and dry cycles will determine P presence and binding affinity (Reddy and DeLaune 2007).

Additionally, the equilibrium phosphorus concentration (EPC) controls the direction of P movement. The EPC value is the result of sediment mineralogical properties as well as long term P loading; as such, sites with a history of high nutrient loading are predicted to have high EPC

values, and thus, reduced capacity to retain P for a given runoff concentration. Alternatively, sites with little or no history of nutrient loading will have low EPC values, and therefore, higher capacity to retain P. EPC refers to the water column P concentration at which the net flux of P to or from the sediment is zero (Figure 3-1a). When water column P concentration is greater than EPC, P is adsorbed by the sediment (Figure 3-1b, Figure 3-2). Alternatively, when the water column P concentration is less than the EPC, desorption occurs and the sediment will act as a P source to the overlying water column (Figure 3-1c, Figure 3-2). Given the same P concentration in water entering a wetland, a site with a low EPC has a higher potential to retain P than does a site with a high EPC (Reddy and DeLaune 2007).

Multiple methods are available for measuring P retention, but P sorption isotherm (PSI) methods are used for their ease and affordability. Soil samples are subjected to high doses of P for 24 hours, and the difference in P between the initial and final solution represents the amount of P retained by the soil (Bruland and Richardson 2006). One constraint of this method is that the initial P concentration used (130 mg/L as P) seriously overestimates the amount of P wetlands would experience under normal circumstances. As P retention is partially driven by concentration gradients (Reddy and DeLaune 2007), this method produces inflated measures of P retention that would unlikely be realized *in situ*. Although it is useful in ascertaining the number of P binding sites, it is an overestimate of the P retention service actually provided. Consequently, using reduced P concentrations is preferred in order to arrive at realistic measures of P retention.

The goal in quantifying P retention as an ecosystem service is relating it to measurements of ecosystem condition. Condition assessments measure site departure from ecological integrity (Karr and Dudley 1981). In Florida, two methods of measuring ecosystem condition are in use.

The first is the Florida Wetland Condition Index (FWCI, Reiss 2004), which evaluates condition based on six macrophyte metrics. The second is the Uniform Mitigation Assessment Method (UMAM, FDEP 2007), which evaluates condition based on three metrics: location and landscape support, water environment, and community structure. UMAM is the state mandated method that establishes mitigation credits in the course of wetland impacts and development. Both methods yield a cumulative score that relates information on ecological condition. The use of condition assessments in establishing mitigation credits assumes that full ecological integrity is associated with high functionality, particularly in the case of UMAM (FDEP 2007). As condition is easier and more cost effective to measure than ecosystem functions, it is often the method of choice when understanding wetland quality (Fennessy 2004). However, recent studies suggest that attributes evaluated in condition assessments do not reflect wetland function in terms of nitrogen removal (Stander and Ehrenfeld 2009a, b).

The goal of this study is to understand the relationship between conditional assessments and ecosystem function focusing on P retention. Specifically, the use of two condition assessments, FWCI and UMAM, are tested in terms of predicting P retention. It is expected that wetlands with low condition scores will be associated with low measures of P retention and high EPC values, both of which indicate low service provision. Alternatively, wetlands with high condition scores will be associated with high measures of P retention and low EPC values indicative of high service provision. These findings would support the use of condition assessments in protecting wetland P retention in the course of mitigation.

Methods

Study Area and Site Selection

Twenty-two palustrine depressional forested wetlands in northern Florida were evaluated from May through the end of October in 2008 (Figure 3-3). Sites were located in three main

land use settings identified in 2001 as reference (e.g., State Parks, silvicultural lands), agricultural (e.g., pasture, row crops), and urban (e.g., residential, industrial) and verified through the use of landscape development intensity (LDI) index (Reiss 2004). Although differences exist between sites located in State Parks and silvicultural lands, the decision to treat the wetlands as one group was consistent with previous work (Reiss 2006) and representative of expected background variation. Of the sites evaluated, four were located in agricultural landscapes, twelve in reference, and six in urban.

P Retention

P retention was estimated using two methods, and both involved incubating soil samples with de-ionized (DI) water or concentrated P solutions. During and after incubation the overlying water column was sampled and soluble reactive P (SRP) measured.

The first method is a standard soil P sorption metric called the P sorption index (PSI; Richardson 1985, Axt and Walbridge 1999, Bridgham et al. 2001, Bruland and Richardson 2006). Samples are subjected to a high initial concentration P (130 mg P L^{-1}) solution for 24 hours to ensure that all sorption sites are saturated by the end of the test. Five soil cores (2 cm by 10 cm) were sampled from each site spanning the entire width from wetland edge-to-edge through the wetland center for all sites. Soil cores were kept at 4 deg C until further analysis. Within ten days of field collection, approximately 2 g of field-wet soil was loaded with 14 mL of either DI water or 130 ppm P solution and shaken for 24 hours. At the end of that period, samples were centrifuged and filtered. Post-incubation water was diluted (1:400) and tested for SRP using colorimetric analysis (EPA 365.1) on an Aquamate spectrophotometer. The index was calculated as

$$PSI = S * (\log C)^{-1}$$

where S is the amount of P sorbed (mg P 100 g soil^{-1}) and C = the final inorganic P concentration in solution (mg PO_4-P L^{-1}) (Bruland and Richardson 2006).

General soil characteristics were also measured. Moisture content was calculated based on the difference in sample mass after drying for 24 hours at 70 degrees C. Loss on ignition (i.e., organic matter content) was measured by ashing samples in a muffle furnace for 4 hours at 500 degrees C (Bridgham et al. 1998, Reddy et al. 1998, Zhu and Ehrenfeld 1999). Soils were tested for Fe and Al using Mehlich 3 extraction and inductively coupled argon plasma (ICAP) spectroscopy at the Analytical Research Laboratory at the University of Florida.

The second method used to estimate P retention was designed to more closely resemble *in situ* conditions (i.e., diffusion, organic P mineralization) of P sorption. Six cores 15 cm long and 10 cm in diameter were extracted from each of thirteen sites (4 agricultural, 5 reference, 5 urban). Sites were selected based on ease of accessibility and proximity to the University of Florida. Cores were sealed with a plastic cap in the field and caulked to prevent leakage; to prevent artificial drying, cores were not dewatered before treatment so *in situ* redox conditions were preserved to the greatest extent possible. Two cores from each site were loaded with 1 L of a 10 ppm, 0.5 ppm and 0 ppm (DI water) P solution. Each day during a seven-day incubation, 20 mL of water were sampled from each core; samples were filtered, diluted (if necessary), and tested for SRP using colorimetric analyses described above.

Condition

Wetland condition was evaluated using two methods. FWCI was based on vegetative composition evaluated using 4 belt transects in each of the cardinal directions. Along each transect species presence was recorded every five meters. FWCI condition score was determined from six metrics: (1) tolerant indicator species; (2) sensitive indicator species; (3) FQAI; (4) exotic species; (5) native perennial species; and (6) wetland status species (Reiss 2006). FWCI

metrics were computed as described in Reiss (2004); all metrics excluding FQAI were calculated as proportions in which the count of species fitting the particular category (e.g., tolerant, sensitive, exotic) (N) was divided by the species richness (R) of the site:

$$P=N/R$$

FQAI was calculated as the sum of CC scores for all species divided by the species richness of a site (Cohen et al. 2004):

$$FQAI = \sum (C_1 + C_2 + \ldots Cn)/R.$$

CC scores were determined by a survey of expert Florida botanists in 2001 according to species fidelity to particular wetland conditions; CC range from 0 (invasive and/or exotic species) to 10 (species with high affinity for reference standard wetland conditions) (Wilhelm and Ladd 1988). The CC scores reported in Reiss (2004) were used to compute FQAI in this study. Vegetation not identified to species level was excluded from metric calculations because CC scores, exotic/native status and tolerance/sensitivity classifications, all integral to FWCI calculation, are species specific. As in Reiss (2004), each metric was scaled and normalized to fit a 0 to 10 range, and summed to yield a cumulative FWCI score between 0-60 (higher scores indicating higher condition) for each site.

The second condition assessment method was UMAM. UMAM condition scores are based on three metrics: (1) location and landscape support; (2) water environment; and (3) community structure (FDEP 2007). Location and landscape support evaluates surrounding habitats, invasive species presence, wildlife access to and from the site, benefits to downstream fish and wildlife, land uses outside the assessment area, and dependency of downstream habitats on proper wetland function. The water environment metric assesses water levels and flows, water level indicators, soil moisture, soil erosion/deposition, evidence of fire history, vegetation or benthic community

zonation, vegetative hydrologic stress, presence of animal species with specific hydrologic requirements, presence of plant species indicative of water quality degradation, standing water quality (discoloration, turbidity, or oil sheen), and light penetration. The community structure metric evaluates the vegetation and structural habitat; plant cover in the canopy, shrub, and ground stratum; presence of invasive species; evidence of normal and natural regeneration and recruitment; natural age and size distribution of species within the system; density and quality of coarse woody debris; land management practices; topographic features (e.g., ponds, channels, and hummocks); and evidence of siltation or algal growth.

Each metric was rated as a whole number from 0 to 10 (0 defined as "insufficient to provide functions" and10 defined as "optimally supporting functions") employing best professional judgment in accordance with procedural rules. Scores for individual metrics were summed and divided by 30 to yield condition scores between zero and one (FDEP 2007). Calculations for time lags and mitigation credits were disregarded such that the final score reflects current condition only, and not a measure of potential impact or mitigation

Data Analysis and Model

To estimate EPC values from the time series of water column P concentrations, we used a first order model of P sorption (Kadlec and Knight 1996). This model predicts three parameters based on observed P concentrations over time. The first order model predicts the concentration at time t (C_t) as a function of the initial concentration (C_0), a background equilibrium concentration (C*, also the EPC), and a rate coefficient, k:

$$C_t = C^* + C_0 \exp(-k^*t)$$

One of the assumptions of this model is that there is no P in the overlying water. Although this does not hold for our methods, we did use a DI incubation to quantify water-soluble soil P. Moreover, we estimated C* from low and high P levels, and used multiple replicates to make

sense of noisy and uncertain estimates. The model assumes C* and k are constants, but they are affected by temperature and seasonality (Kadlec 2000). Because we sampled in a single season and cores were climate controlled, these variables should behave like constants. Finally, because the model extrapolates to infinity and is subject to minute whims of the data, we placed a time limit on the parameters. By limiting the time variable to seven days, we were able to arrive at estimates that were within the predictive power of our soil core profiles, which were allowed to equilibrate over seven days.

Linear regression was used to determine the relationship between the two different methods of P retention. As both, in principle, measure the same phenomenon, we expected PSI values to be positively correlated with values of P retention from the core assay. ANOVA was used to establish P retention differences between methods and land uses, as well as estimates of EPC and k. Two-way ANOVA was used to test for interactions between land uses and P concentrations in the core assay. Post-hoc Scheffe interval tests were used to distinguish between land use groups in all ANOVA tests. Functional measures (PSI, core assay, and EPC) were compared with condition scores (FWCI and UMAM) using linear regressions to test for the expected positive correlation. Measures of P retention from the core assay were transformed using an inverse exponential function, while EPC values were transformed using an exponential function.

Results

Measures of P retention

PSI values did not vary by land use (ANOVA, $F = 0.067$, $p = 0.93$) with an overall average of 19.9 (±20.9) S/logCt (Table 3-1). Similarly, the amount of P adsorbed during the PSI assay (mg P/kg soil) did not vary by land use (Table 3-2, ANOVA, $F = 0.400$, $p = 0.67$). When loaded with 130 ppm P solution, reference sites adsorbed a mean of 321.1 (±267) mg P/kg, urban sites

adsorbed 346.3 (±282), and agricultural sites 227.3 (±387). When loaded with DI water, every sample for every site released less than 100 mg P/kg soil. Although not statistically significant, reference wetlands showed the lowest P release of 19.9 mg P/kg soil (±8.6), followed by urban sites at 22.8 mg P/kg soil (±11.9) and agricultural wetlands at 25.7 mg P/kg soil (±13) (ANOVA, $F = 0.53$, $p=0.59$).

PSI values did not correlate significantly with P release with DI water (linear regression, $F=0.08$, $p=0.78$), nor did the amount of P adsorbed (linear regression, $F=0.25$, $p=0.62$). Soil Fe concentrations were significantly correlated with PSI values (linear regression, $F=15.5$, $p=0.003$), but not correlated with total amount of P adsorbed (linear regression, $F=0.22$, $p=0.64$) or released (linear regression, $F=0.62$, $p=0.45$). Moreover, Al was not significantly correlated with PSI (linear regression, $F=0.58$, $p=0.46$), P sorption (linear regression, $F=1.12$, $p=0.30$), or P release (Linear regression, $F=0.055$, $p=0.82$).

Soils were sampled across a transect spanning the entire width of the wetland. Organic matter and moisture were different between edge and interior samples, but measures of P retention were not. Soil moisture in edge samples was 28.6% (± 16.4), contrasted with 43.2% (± 19.8) in the interior samples (ANOVA, $F=14.7$, $p=0.0002$). Similarly, organic matter content was 13.2% (± 11.9) in the edge samples, versus 23.3% (± 22.7) in interior samples (ANOVA, $F=5.9$, $p=0.016$). However, measures of P retention (ANOVA, $F=2.5$, $p=0.12$) and release (ANOVA, $F=0.14$, $p=0.70$) were not statistically significant between wetland edge and interior sampling.

In contrast, measures of P retention using the core assay did show significant differences between land uses (Factorial ANOVA, $F=25.2$, $p<0.0001$) and P concentration (Factorial ANOVA, $F=13.7$, $p<0.0001$), but land use and P concentration did not show evidence of an

interaction effect (Factorial ANOVA, F=1.5, p=0.20; Table 3-2). Over the seven-day incubation, agricultural sites released on average 10.3 (±12.7) and 6.6 (±15.7) mg P/kg soil for the 0.5 and 10 ppm P solution cores, respectively. Reference sites retained on average 1.2 (±1.0) and 11.5 (±8.7) mg P/kg soil for the 0.5 and 10 ppm P solution cores, while urban sites retained on average 1.6 (±0.31) and 17.3 (±8.4) mg P/kg soil for the 0.5 and 10 ppm P solution cores. All sites released P when loaded with DI water (0 ppm P solution). Specifically, agricultural sites released 10.8 (±10.2) mg P/kg soil, reference sites 0.10 (±0.19) mg P/kg soil, and urban sites 0.10 (±0.14) mg P/kg soil. Post-hoc Scheffe tests distinguished between agricultural and reference sites, and agricultural and urban sites, but not between reference and urban sites. Similarly, Scheffe tests revealed no difference in P sorption between cores loaded with DI water and 0.5 ppm P.

For the core assays, the amount of P adsorbed was far less than that calculated from PSI methods. PSI P sorption (mg P/kg soil) values overestimated the amount of P adsorbed from the 10 ppm cores by as much as 37 times, and more so with the 0.5 ppm P cores. The values for P release upon loading with DI water were similar, with values exaggerated in the PSI method by as much as 20 times the 0 ppm core assay.

PSI values were not significantly correlated with the amount of P sorbed from core assays (linear regressions, F=0.6, 0.5, 1.19, p=0.51, 0.63, 0.26 for 0, 0.5, and 10 ppm P cores respectively). Similarly, estimates of P sorption using the PSI methodology (mg/kg) were not correlated with the amount of P adsorbed from any soil cores (linear regressions, F=0.08, 0.47, 2.53, p=0.8, 0.51, 0.14 for 0, 0.5, and 10 ppm P cores, respectively). Estimates of P release (measured in mg/kg) using the PSI method with DI water effectively predicted sorption from all soil core concentrations (Figure 3-4, a-c). As would be expected, P release from the PSI assay

was negatively correlated with P sorption in the cores. The strongest correlation was with 0.5 ppm P cores (linear regression, F=13.2, R^2=0.571, p=0.004), followed by 10 ppm P cores (linear regression, F=7.7, R^2=0.435, p=0.02), and lastly 0 ppm P cores (linear regression, F=6.24, R^2=0.384, p=0.03). However, removal of two outliers (A1 and A4) renders the association between PSI P release and core assay P sorption insignificant at the 0 ppm (linear regression, F=0.35, p=0.57), 0.5 ppm (linear regression, F=0.009, p=0.92), and 10 ppm (linear regression, F=0.13, p=0.73) P concentrations.

Time series profiles from soil cores suggest convergence to equilibrium for cores loaded with 0 and 0.5 ppm P solution. For reference sites (R1, R2, R3, R5, R6), P equilibrium concentrations appear to be between 0.015 and 0.068 ppm. Urban sites (U1, U3, U4, U6) showed a wider range, but similar equilibrium concentrations of between 0.002-0.124 ppm. Agricultural site time series were drastically different. Most agricultural sites released P when loaded with 0 and 0.5 ppm P solution; as such, equilibrium concentrations were much higher than reference and urban sites. Specifically, two agricultural sites (A2 and A3) showed similar ranges, with equilibrium concentrations between 0.4 and 1.4 ppm. In contrast, A1 and A4 equilibrium concentrations were at least two and four times greater: 4.65-10.5 and 2.3-7.4 ppm, respectively, after seven days of incubation.

EPC

EPC values were significantly different by land use (Factorial ANOVA, F=26.7, p<0.0001) and initial P concentration (Factorial ANOVA, F=74.7, p<0.0001; Table 3-3), but there was no interaction effect between land use and P concentration (Factorial ANOVA, F=1.9, p=0.12). EPC values were similar between reference and urban soil cores, but substantially higher in agricultural soil cores for all initial P concentrations (post-hoc Scheffe test, 95% confidence). Notably, when loaded with DI water, agricultural soils showed an average EPC of 3.16 ppm,

over 200 times greater than reference (0.017 ppm) and urban (0.010 ppm) soils. Similarly, at 0.5 ppm P, the mean agricultural EPC (3.46 ppm) was 30 times greater than reference (0.143 ppm) and urban (0.018 ppm) soils; and at 10 ppm P, it was twice as large in agricultural (12.49 ppm) sites as in reference (6.97 ppm) and urban (5.2 ppm) sites. EPC values were similar for the DI and 0.5 ppm cores, but distinct for 10 ppm cores (post-hoc Scheffe test).

Condition

Comprehensive analyses on condition scores are published in Chapter 2.

Linear regressions were used to determine the association between condition scores and measures of P retention. Overall, no measure of P retention using the PSI methodology (index, mass of P sorption, or mass of P release) was significantly correlated with ecosystem condition as measured with FWCI (linear regressions, PSI: F=0.827, p=0.37; mass P sorption: F=0.371, p=0.55; mass P release: F=0.59, p=0.453). Similarly, UMAM and PSI P retention lacked meaningful relationships (linear regressions; PSI: F=0.32, p=0.58; mass P sorption: F=0.04, p=0.84; mass P release: F=0.002, p=0.96).

Measures of total P retention using the core method, however, showed significant correlation with ecosystem condition. Specifically, transformed values of P retention were negatively correlated with FWCI scores (linear regression; F=9.4, R^2=0.46, p=0.01) and UMAM scores (linear regression; F=6.1, R^2=0.36, p=0.03) for the 0 ppm P core suggesting that sites with high condition scores can adsorb more P. However, removal of the two outliers (A1 and A4) renders the relationship insignificant. P retention values were not significantly correlated with FWCI or UMAM condition scores at the 0.5 ppm P cores (linear regression; F=4.1, 3.2, R^2=0.27, 0.26, p=0.07, 0.11, respectively).

In general, EPC estimates were negatively correlated with measures of condition such that sites with high condition scores were associated with low EPC values. In particular, transformed

EPC values were negatively correlated with FWCI (linear regression; F=8.9; R^2=0.45, p=0.012) and UMAM (linear regression; F=6.8, R^2=0.38, p=0.02) at 0 ppm P load. The two outliers were noted (A1 and A4) and removal of these sites from the analysis did not change the trend of the condition-EPC, but the relationship was no longer significant for the 0 ppm load (linear regression; F=2.8, 0.09, R^2=0.24, 0.01, p=0.13, 0.77 for FWCI and UMAM, respectively). FWCI and UMAM were not significantly correlated with EPC values at the 0.5 ppm P load (linear regression; F=4.1, 2.7, R^2=0.27, 0.19, p=0.07, 0.13, respectively.).

Discussion

Wetlands are valuable ecosystems for the services they provide. As services are numerous and usually difficult to quantify, land managers typically use ecosystem condition to assess wetland quality, the assumption being that wetlands of high condition are operating, or providing functions, at a similarly optimum level. We set out to test this assumption by measuring P retention, one such ecosystem service, and evaluating wetlands using two condition assessment methods.

P retention

There are many ways to measure P retention. The standard PSI method overestimates P sorption, and provides estimates of P reactivity that fail to correspond with more realistic measurements of P sorption. The ability of soils to retain P at levels approaching 450 mg/kg using PSI methods does not indicate that they function at that level in the field. First, P retention is concentration dependent, so as the P concentration of the water column increases the quantity of P retained also increases (Reddy and DeLaune 2007). Loading soils with 130 ppm P solution poorly approximates what would be expected under typical environmental conditions, even in areas that are highly P enriched. Lin (2004) documented inflows to wetlands located in intensive agricultural areas at 0.344-0.476 ppm P, substantially less than the concentration used in PSI

methods. Therefore, the resulting sorption (in mg/kg) represents an exaggeration of plausible P retention. Moreover, shaking soils during the 24-hour incubation maximizes binding sites, further exaggerating P retention rates. As such, we interpret the PSI results as measurements of maximum functional capacity, that is, the upper bound of P retention possible under the most extreme conditions.

Although PSI values can be viewed as the maximum P retention for a wetland, it was surprising not to find differences between wetlands of different landscapes. Our data suggests that P retention does not vary between wetlands of reference, urban, or agricultural landscapes when quantifying using PSI methods. Cohen et al. (2007) reported a significant effect of the interaction between ecoregion and condition (impacted v. reference), suggesting some influence of land use on P sorption. Bruland and Richardson (2006) reported highly variable PSI values, but do not distinguish between land uses. Rather, they attribute PSI variability to OM, Fe, and exchangeable calcium. Consequently, the lack of PSI differences between wetlands of varying land use may be consistent with other work as OM and Fe were similar across land use categories.

PSI values reported here were lower than that from other studies. While our PSI range was -0.03 - 56.7 with a mean of 19.9, Bruland and Richardson (2006) reported values between 14.5-184 with an average of 101.2, and Dunne and Reddy (2006) report values between 62 and 303 with an average of 167.0. One likely explanation for this discrepancy is differences in the study sites. This study sampled isolated cypress domes, while the other studies sampled floodplains and marshes with frequent hydrologic connectivity to surrounding landscapes, and therefore more frequent sediment delivery and high anion exchange capacity; both of which may lead to higher P retention. Moreover, these studies reported OM contents higher than were measured in

our sites, a factor that can contribute to greater P retention. Total P sorption measurements using the PSI method in this study were similar to that found in Cohen et al. (2007). The authors reported a wide range P sorption measures, or -75 - 995 mg P/kg soil, using a similar single-point isotherm technique with 100 ppm P solution, a range comparable to that found here, or -17 - 675 mg P/kg soil.

As expected, values of P retention from the core assay were far less than that from the PSI method, and showed differences by land use. The former phenomenon can be attributed to solution P concentration, as high P concentration drives higher P retention (Reddy and DeLaune 2007). It is not surprising then, that the PSI method, which uses a P concentration of 130 ppm P, registers higher P retention values than the core assays in which the highest P concentration was 10 ppm P. Dunne et al. (2006) reported ambient P sorption values in which P retention was measured after loading with ambient water samples. Water column concentrations ranged from 0.07 - 1.2 ppm P, conditions similar to our core assay. Sorption values observed under these conditions were -2.0 - 13.6 mg/kg, while our sorption values for the 0.5 ppm P cores ranged from -30.0 - 1.91 mg/kg. Although the ranges appear dissimilar, they are in fact more similar with each other than with estimates of P retention using 1000 ppm P, 130 ppm P, or 10 ppm P solutions. The core method produced a range of P retention measures that represents the range observed under extant environmental conditions.

Not only were retention values significantly less than the PSI methods, but the core method also showed significant sorption differences between wetlands of different land uses. Agricultural sites showed a reduced capacity to retain P relative to reference and urban wetlands. This reduced functionality in agricultural wetlands could be due to a history of P loading, which reduces P binding sites limiting the ability for present and future P retention function.

Furthermore, all agricultural sites in our sample were actively used for cattle and/or row crops, and likely received a higher P input than urban or reference sites. Consequently, the agricultural sites may not be capable of removing P when concentrations are low (e.g., 0.5 ppm). The ability of the core method to demonstrate statistically significant differences by land use is an important step in identifying functional differences between wetlands.

Moreover, allowing the cores to incubate over seven days provided time series profiles for each site, which were used to estimate realistic expectations for EPC values. EPC values varied with P concentration such that the models predicted higher equilibrium values when cores were loaded with elevated P solutions. Wetlands in agricultural settings have substantially higher EPC values than urban and reference wetlands at all P concentrations. However, it does not follow that high intensity land use is always associated with high EPC values. As with measures of P sorption, urban wetlands were more similar to reference wetlands than agricultural sites. This would suggest that lumping wetlands surrounded by high intensity land uses into one category (i.e., "impacted") may not be useful to understanding functional variability.

The absence of functional differences between urban and reference site P retention is not surprising. As was observed previously in Cohen et al. (2007), measures of P retention failed to show even minimal differences between urban and reference wetlands. In that case the authors speculated the lack of difference was due to high P and high binding capacity in urban sites impacted by erosion, and low P and low binding capacity in reference sites. However, soil samples in this study showed comparable TP, Fe, and Al levels (Table 3-1). This would suggest, at least in terms of P retention, urban and reference sites are capable of functioning on similar levels due to similar background chemistry.

However, actual differences in ecosystem scale P removal are predicated on two attributes: hydrologic conditions and ambient P concentrations. While our estimates suggest urban and reference sites are capable of functioning at similar levels, a more realistic measure of P retention would be based on a) how long the wetlands are wet and b) how much P they receive from their surroundings. For example, reference wetlands generally have longer and more stable hydroperiods than urban wetlands (Stander and Ehrenfeld 2009a), while urban wetlands generally receive higher concentrations of P in urban runoff (Lin 2004). Actual differences in functiona between urban and reference sites cannot be documented until hydrologic and P concentrations are known. Consequently, while urban and reference wetlands may have similar functional capacity, they may realize that function differently in different land use settings.

Fortunately, event mean concentrations (EMC) can be used to understand how wetlands in different landscapes function differently. EMCs are the concentration of contaminants, nutrients, and other molecules found in runoff specific to a particular landscape. In Florida, agricultural sites have been evaluated as receiving EMCs 0.344-0.476 mg/L P; urban sites have concentrations of 0.225 mg/L P; and reference sites have EMCs of 0.053 mg/L P (Lin 2004). Using these measures as estimates of ambient P concentration found in wetlands and EPC values, directionality of P retention can be predicted. When EMC is greater than EPC, wetlands will sorb P; when EMC is less than EPC, wetlands will release P. With an EPC value of 3.46 mg/L (Table 3-3), agricultural wetlands will tend to release P to the water column at the published loading concentrations. Similarly, reference sites will also release P using this estimate of EMC and our EPC value of 0.143 mg/L. Urban sites, on the other hand, would show P sorption as their EPC value, or 0.078 mg/L, is less than the EMC. Consequently, these values

suggest that urban wetlands tend to be more functional in terms of removing P from runoff than both agricultural and reference wetlands.

One problem with these predictions is the assumption of static ambient P concentrations and EPC values. Both site parameters are dynamic and respond to environmental variables. For example, water column P concentration changes with season and management due to fertilizer application, vegetation growth, and human impacts. Additionally, EPC values can change over time as P is loaded or removed from a site. Thus, the direction of P sorption/release is variable. While our results suggest that urban wetlands are the only sites providing this function, it is likely that all wetlands are capable of P retention. However, the degree of P removal is predicated on the magnitude of difference between dynamic EMC and EPC values.

Condition

Differentiating between functional capacity and actual function proved to be important in understanding the relationship between ecosystem condition and function. The lack of correlation between any estimate of P retention using the PSI method and condition scores would suggest that condition cannot predict wetland function at all. However, estimates of P retention from the core assays and model predictions showed otherwise.

The extreme P retention results generated from the two agricultural sites (A1 and A4) suggest that function maximizes at relatively low levels of condition. Similar to threshold-like effects, condition scores at or below 15 on the FWCI scale and 0.3 on the UMAM scale are associated with function impairment, or in this case high levels of P release. Alternatively, sites with condition scores above 15 FWCI or 0.3 UMAM are associated with P retention. This threshold-effect is similar to the biodiversity-function relationship (Chapin et al. 2000, Schwartz et al. 2000) in which measures of function stabilize at relatively low levels of biodiversity. By maintaining a minimal level of biodiversity, or in this case, condition, wetlands can be expected

to perform a desirable level of function, or P retention. However, similar to the biodiversity-function relationship, there could be benefits associated with higher-than-minimal-condition. Chapin et al. (2000) argue that high levels of biodiversity allow for ecosystem resistance and resilience due to differences in sensitivity among functionally similar species. It is possible that the condition-P retention relationship is tracking the same phenomenon. At 15 FWCI or 0.3 UMAM, P retention functions are maximized, and sites with higher conditions are protected against environmental impacts due to functionally similar species.

The correlation of condition and P retention is predicated on the inclusion of agricultural sites within the sample. Removing these sites from the analysis voids the relationship and highlights a potential problem within the condition-function paradigm. Our results suggest that urban and reference wetlands function at similar levels, as noted by similar P retention measures and EPC values. Condition assessments, however, produce very different scores for urban and reference sites, with urban sites scoring lower. As condition assessments are used to establish mitigation requirements, it would be expected that urban wetlands, as a result of their reduced condition scores, require less mitigation than reference wetlands. However, the fact that both reference and urban wetlands appear to function at similar levels regarding P retention suggests one of two things: (1) mitigation requirements for urban wetlands are not sufficient in eliminating functional loss in course of development, or (2) reference systems are rated too high relative to their level of function. In the first case, mitigation could lead to large-scale landscape functional loss, whereas, in the second case, condition assessments are producing inappropriate estimates of wetland function. Past research reached similar conclusions using the HGM approach (Hruby 2001, Stander and Ehrenfeld 2009a and b). Specifically, reference wetlands do

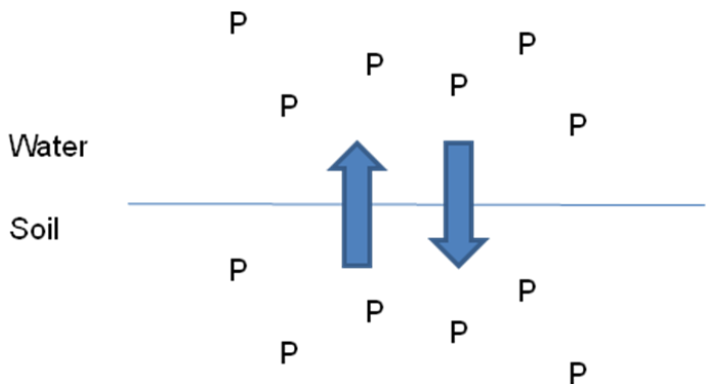
not appear to represent characteristic functioning of an entire subclass of wetlands, and thus, should not be used as a standard against which all other wetlands are judged.

When assessors judge a site as maintaining excellent condition, or full integrity, it is assumed that the functions of that ecosystem are performing at reference levels (Fennessy et al. 2004). This leaves no room for conflicting functions (Ehrenfeld 2004), or capturing the ecological value of "working wetlands" (i.e., urban or treatment wetlands). The fact that urban sites score low in condition assessments, but retain functional capacity undercuts this assumption. One cannot simply assume that desirable or valuable functions correlate with condition. More importantly, one cannot assume that low condition wetlands are not performing valuable functions for the landscape. Doing so would cause a loss of these sites to development or further degradation, and entire regional landscapes will suffer from functional loss. Fennessy et al. (2004) acknowledges this conflict and suggests that in cases where sites are performing valuable functions despite degradation, additional points be awarded to reflect functional value despite the low condition. It is unclear how evaluators are to ascertain function in the course of conditional assessments if condition does not comport with valuable ecosystem functions. Conditional assessments may not be sufficient in determining mitigation value if they cannot track important functions. Ultimately, functional assessment (i.e., P retention cores, denitrification assays, hydrological monitoring) should be a future requirement in determining wetland mitigation value (Stander and Ehrenfeld 2009a and b).

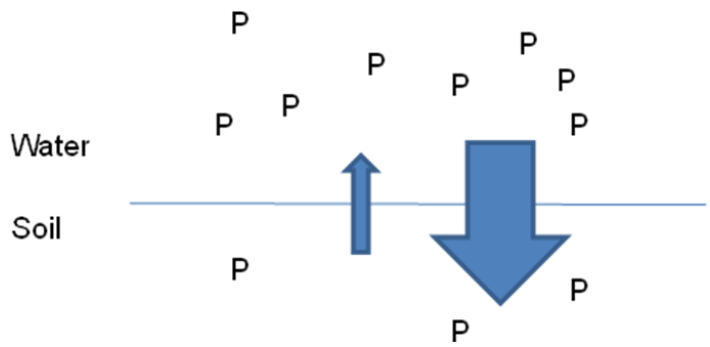
The decision to measure one wetland function, namely P retention, was due in part to time and monetary constraints. P retention is only an isolated function picked from a suite of numerous potential functions. It does not represent an integrative evaluation of all wetland functions, and it may even conflict with other functions, such as wild-life use (Surdick 2006),

denitrification, and biodiversity (Hansson 2005). Furthermore, sites with greater P retention are not more valuable than sites with less P retention as other valuable functions were not measured. However, the existence of low condition sites that retain P suggests that assessment methods may mislead regulators when it comes to evaluating valuable wetland functions. In quantifying ecosystem services, we must be clear about not only what function we are measuring, but also the realistic implications of the method employed to measure it. As anthropogenic values are ascribed to measures of ecosystem service provision, providing over- or underestimates of function can lead to false augmentation or depression of ecosystem value, and therefore inappropriate expectations for mitigation programs (Reiss et al. 2007).

As mitigation legislation was motivated by large scale, regional losses in wetland function, it is imperative that assessment methods be designed to reflect wetland functions that are regionally valuable. Identifying the functions of utmost priority first will lead to the development of condition assessments that reflect those values (Carletti et al. 2004). Condition assessments are excellent in capturing habitat ecosystem services. However, their use in evaluating process functions is still less certain. If the goal is to minimize large-scale regional process function loss from wetlands, we need condition assessment methods that adequately reflect those services.



a.



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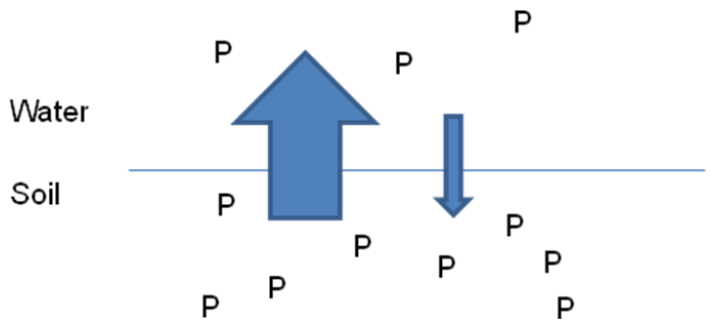


c.

Figure 3-1. Schematic of P transfer between water column and sediment; equilibrium P concentration (EPC) (a), sequestration of P into the soil (b); and release of P to the water column (c).

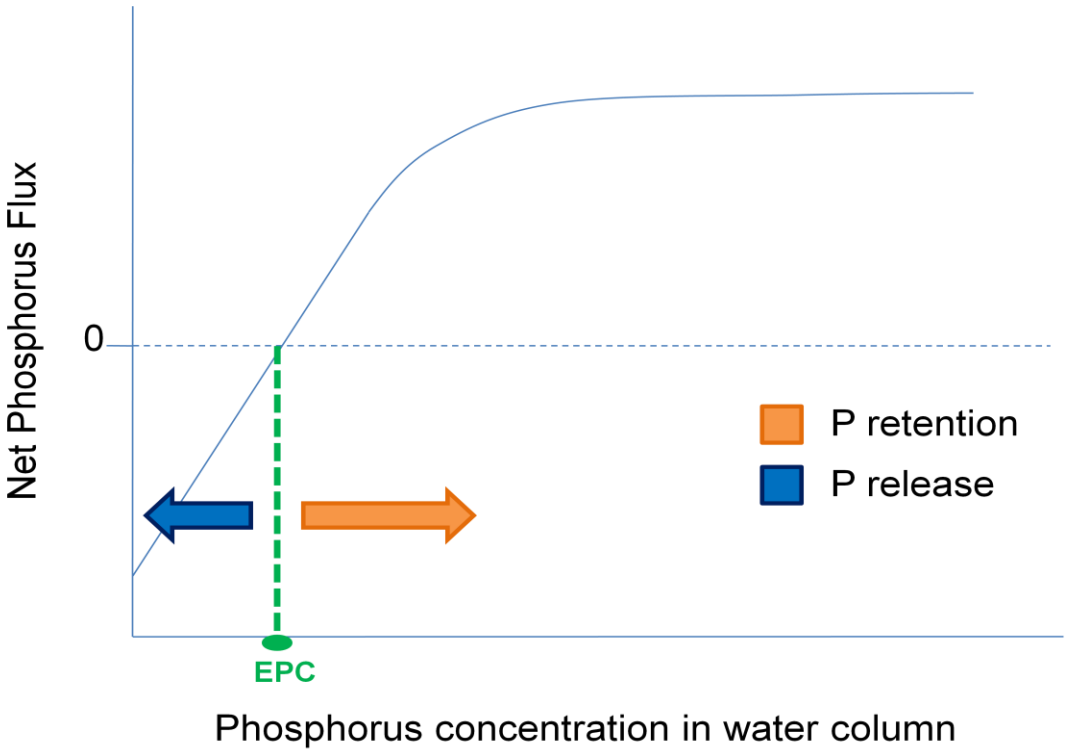


Figure 3-2. The role EPC plays in determining the direction of P flux. If the P concentration in the water column is greater than the EPC, the P flux will be into the soil; by extension, water column P concentrations less than the EPC result in P fluxes out of the sediment.

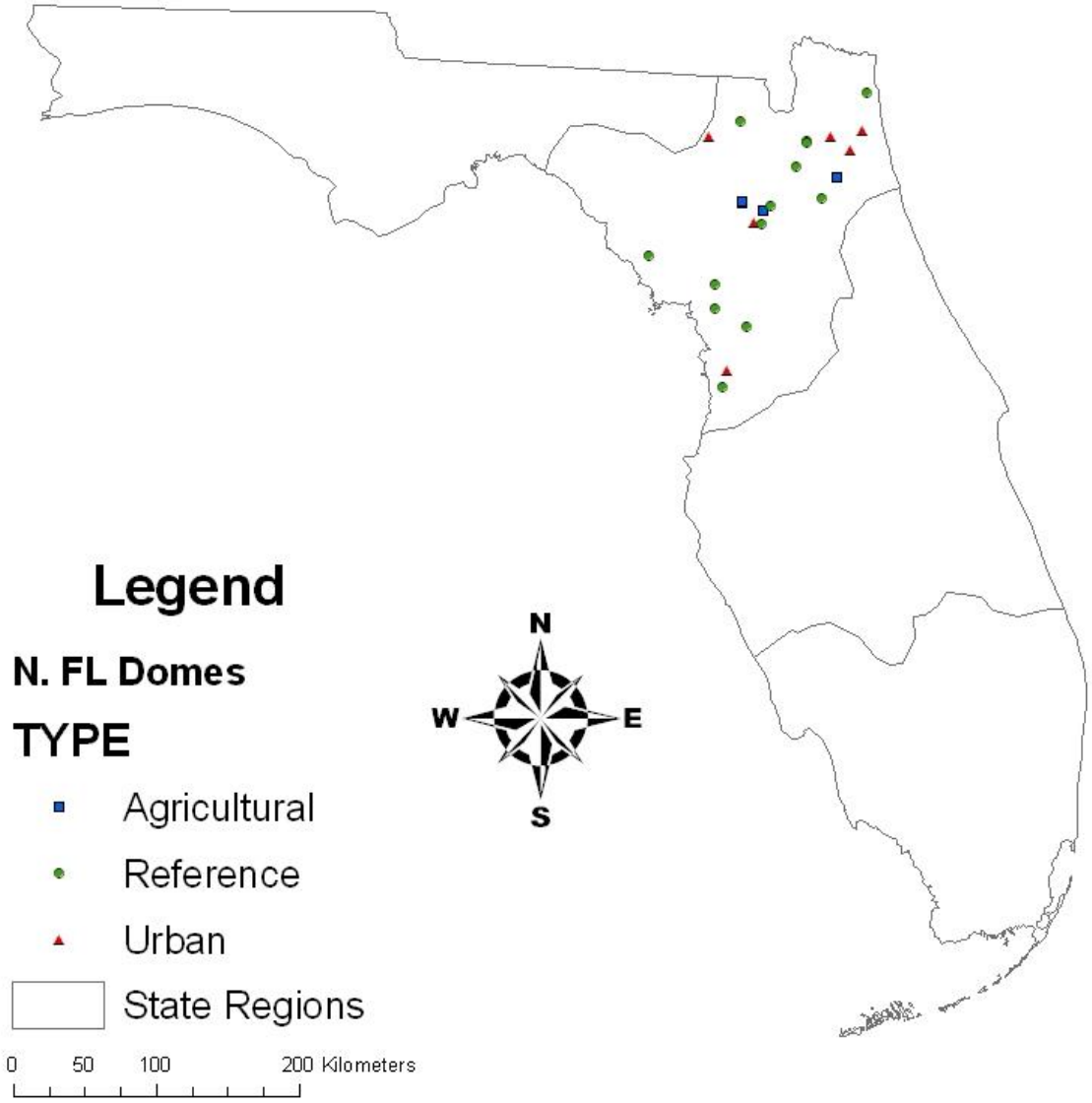
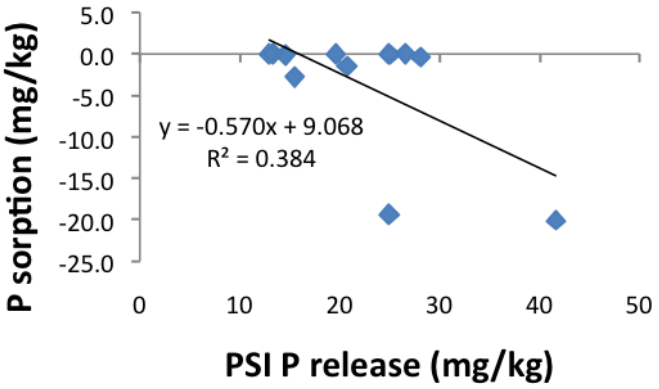
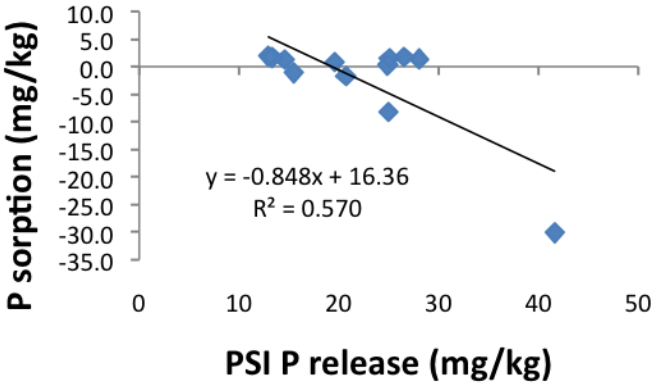


Figure 3-3. Location of 23 wetlands sites in northern Florida, U.S.A. Blue squares represent agricultural sites, green circles are reference standard sites, and red triangles are urban sites.



a.



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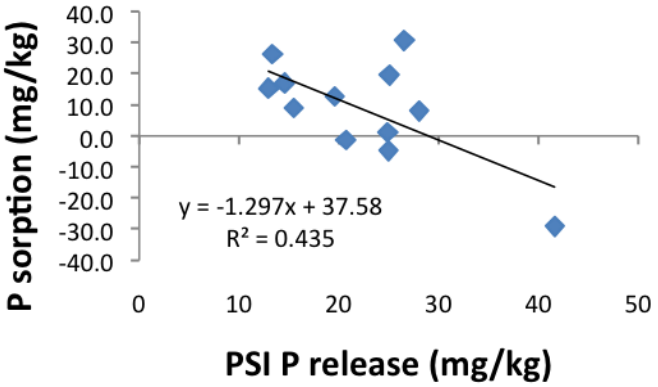

c.

Figure 3-4. Correlation of net P release from PSI methodology using DI water (x-axis) and total P sorption from core assay at each starting concentration (y-axis): are 0 ppm (a), 0.5 ppm (b); and 10 ppm (c). See text for statistics.

Table 3-1. Soil characteristics by land use. Values are mean (SD) of five replicates. Note that OM and moisture showed distinct trends in interior versus edge sampling (see text), but site means are presented here.

Land Use	PSI ($S/\log C_t$)	Organic Matter (%)	Soil Moisture (%)	TP (mg/kg)	Fe (mg/kg)	Al (mg/kg)
Reference	19.84 (18.46)	15.2 (14.28)	35.71(17.65)	42.9 (92.7)	232.3 (206.5)	919.5 (829.4)
Agriculture	17.95 (29.3)	15.85 (11.7)	35.09 (15.59)			
Urban	21.47 (18.95)	25.91(27.83)	45.59 (24.77)	53.5 (61.0)	165.5 (112.1)	1254.9 (829.4)

Table 3-2. P sorption (+) and release (-) as a function of land use and methodology. Values (mg P/kg soil) are expressed over entire incubation period such that the PSI method is over 24 hours and the core method over seven days; values are mean (SD). Letters indicate significant differences between land use (Factorial ANOVA, F=25.2, p<0.0001). Symbols represent significant differences between core concentrations (Factorial ANOVA, F=13.7, p<0.0001).

Land Use	P-sorption PSI	P-release PSI	P sorption/release at 0 ppm	P sorption/release at 0.5 ppm	P sorption/release at10 ppm
Reference	321 (267)	-19.9 (8.6)	-0.097 (0.19) ^{b}*	1.2 (1.0) ^{b}*	11.5 (8.7) ^{b}†
Agriculture	227 (387)	-25.7 (13.0)	-10.8 (10.2) ^{a}*	-10.3 (12.7) ^{a}*	-6.6 (15.7) ^{a}†
Urban	346 (282)	-22.8 (11.9)	-0.096 (0.14) ^{b}*	1.6 (0.31) ^{b}*	17.3 (8.4) ^{b}†

Table 3-3. EPC (ppm) values estimated from the first order rate model for each P concentration (0, 0.5, and 10 ppm). Values were significantly affected by land use (Factorial ANOVA, F=26.7, p<0.0001, letters) and initial P concentration (Factorial ANOVA, F=74.7, p<0.0001, symbols), but there was no interaction effect between land use and P concentration (Factorial ANOVA, F=1.9, p=0.12).

	Core Concentration	Reference	Agriculture	Urban
EPC	0 ppm	0.017^{b}*	3.16^{a}*	0.010 ^{b}*
	0.5 ppm	0.143 ^{b}*	3.46 ^{a}*	0.078 ^{b}*
	10 ppm	6.97 ^{b}†	12.49 ^{a}†	5.2 ^{b}†

APPENDIX A
SITE SPECIFIC CONDITION DATA

Site specific data for land use (R is reference, A is agricultural, and U is urban), land use activities, LDI, FWCI scores in 2001 and 2008, individual FWCI metric scores for 2008 (transformed and normalized according to Reiss (2004)), total UMAM scores, and individual UMAM metric scores. See Reiss (2006) for individual FWCI metric scores from 2001. Conserved forest* have historical silviculture operations.

Site	Land Use	Land Use Activities	LDI 2001	FWCI 2001	FWCI 2008	Tolerant Indicator Species	Sensitive Indicator Species	Exotic Species	FQAI	Wetland Status Species	Native Perennial Species	UMAM 2008	Location and Landscape	Water Environment	Community Structure
R1	R	Conserved forest*	1.1	52.0	44.4	8.2	5.2	10.0	7.0	4.0	10.0	0.867	9.0	8.0	9.0
A1	A	Active pasture	4.0	0.0	7.2	0.9	4.1	0	0	1.1	1.2	0.300	2.0	4.0	3.0
A2	A	Row crops	3.9	37.0	34.3	4.7	3.2	8.2	4.1	4.1	10.0	0.633	0.0	6.0	7.0
A3	A	Row crops	4.3	8.0	17.6	2.6	2.3	3.6	1.7	0.0	7.4	0.767	6.0	9.0	8.0
U1	U	Single family homes, school	5.0	11.8	21.7	2.4	0	6.4	2.9	0.0	10.0	0.333	2.0	4.0	4.0
R2	R	Protected park	1.0	34.8	38.8	4.9	6.0	7.6	6.4	4.0	10.0	0.900	9.0	9.0	9.0
R3	R	Protected park	1.7	56.4	41.9	7.3	7.3	10.0	6.3	1.1	10.0	0.833	9.0	8.0	8.0
U2	U	Single family residence, golf course	5.0	23.7	29.6	3.4	2.2	7.7	4.3	2.0	10.0	0.367	2.0	4.0	5.0
R4	R	Protected land	1.0	52.3	48.6	10.0	7.4	10.0	7.0	4.3	10.0	0.800	9.0	8.0	7.0
A4	A	Active pasture	3.9	3.0	10.4	0.9	4.4	0.08	0	0.0	5.0	0.300	3.0	4.0	2.0
R5	R	Conserved forest*	1.0	58.4	40.0	7.2	7.6	7.9	6.8	0.5	10.0	0.833	9.0	8.0	8.0
R6	R	Conserved forest*	1.5	52.3	42.1	8.1	6.8	10.0	6.6	0.6	10.0	0.833	9.0	8.0	8.0
U3	U	Industrial, mechanic garage	7.0	37.5	29.4	3.8	2.3	6.9	3.8	2.6	10.0	0.600	3.0	7.0	8.0
R7	R	Active silviculture	1.7	37.3	34.9	4.4	6.0	8.2	4.6	1.7	10.0	0.800	8.0	9.0	7.0

U4	U	Single family residence	6.3	20.7	18.7	1.4	1.9	2.8	1.6	1.1	10.0	0.433	2.0	7.0	4.0
U5	U	Single family residence	7.0	40.1	39.0	5.1	5.6	10.0	5.0	3.3	10.0	0.500	2.0	6.0	7.0
R8	R	Protected park	1.0	42.2	44.8	10.0	7.8	10.0	5.8	1.2	10.0	0.933	10.0	9.0	9.0
R9	R	Protected park	1.1	55.0	45.7	8.0	8.3	10.0	5.4	4.0	10.0	0.967	10.0	10.0	9.0
R10	R	Conserved forest	1.0	56.7	51.0	10.0	8.4	10.0	7.6	5.0	10.0	0.867	9.0	9.0	8.0
R11	R	Conserved forest	1.0	58.2	42.3	7.3	8.1	6.6	5.0	5.3	10.0	0.933	9.0	10.0	9.0
R12	R	Protected park	2.5	32.6	32.6	3.4	2.2	10.0	4.8	2.3	10.0	0.767	8.0	8.0	7.0
U6	U	Single family residence	4.8	38.6	35.2	4.4	4.4	8.0	3.8	4.2	10.0.	0.567	3.0	7.0	7.0

APPENDIX B
SITE SPECIFIC SOIL DATA

Negative values reported in sorption columns represent P release. Values in parentheses represent standard deviation of sample size five soil replicates, except for R11 where only three replicates were used. A agriculture, R is reference, and U is urban.

Site	Land Use	PSI (S/logCt)	PSI P sorption (mg/kg)	PSI P release (mg/kg)	0 ppm P Core (mg/kg)	0.5 ppm P Core (mg/kg)	10 ppm P Core (mg/kg)	Soil Moisture (%)	Soil Carbon (%)	TP (mg/kg)	Fe (mg/kg)	Al (mg/kg)
R1	R	18.5 (12.2)	327 (206)	22.7 (4.1)	-0.04	1.87	8.62	36.2 (16.5)	15.3 (8.2)			
A1	A	5.8 (8.1)	111 (154)	41.6 (10.1)	-19.97	-30.20	-29.20	31.1 (7.3)	14.7 (5.0)			
A2	A	0.0 (13.9)	-17.4 (277)	20.7 (3.16)	-1.45	-1.78	-1.46	33.0 (11.7)	10.8 (4.6)			
A3	A	9.4 (23.0)	139 (450)	15.5 (14.4)	-2.75	-1.10	8.88	48.6 (23.7)	23.9 (21.0)			
U1	U	13.8 (5.4)	530 (211)	13.4 (7.4)	0.02	1.66	26.30	33.5 (15.8)	18.2 (9.6)			
R2	R	26.1 (13.9)	446 (192)	14.5 (5.5)	-0.13	1.29	16.90	32.5 (11.7)	14.0 (5.0)			
R3	R	28.3 (22.3)	431 (240)	26.5 (5.7)	0.05	1.72	30.70	30.1 (3.6)	11.3 (5.0)	35.4	140.6	1650
U2	U	33.8 (11.4)	143 (130)	11.2 (5.2)				70.5 (22.5)	67.2 (38.9)	34.0	139.4	642
R4	R	7.3 (6.7)	288 (419)	6.8 (6.3)				21.0 (11.0)	9.6 (5.4)			
A4	A	34.7 (20.0)	675 (241)	24.9 (1.4)	-19.20	-8.24	-4.83	27.8 (9.0)	14.0 (6.8)			
R5	R	23.2 (35.6)	80 (291)	19.6 (8.6)	-0.01	0.81	12.60	25.1 (8.7)				
R6	R	56.7 (27.0)	16 (260)	24.9 (6.6)	-0.02	0.27	1.08	21.9 (8.9)				
U3	U	21.9 (14.4)	641 (86.2)	25.0 (2.9)	-0.07	1.51	19.50	59.4 (13.4)	21.8 (17.1)	130.3	204.8	1703
R7	R	4.7 (14.0)	303 (272)	22.8 (5.8)				27.5 (3.7)	5.8 (2.7)	5.2	531.8	399
U4	U	18.0 (4.5)	451 (84)	12.9 (3.0)	0.0	1.91	15.20	37.3 (22.8)	9.5 (9.8)	30.3	265.3	1575

U5	U	28.2 (14.3)	6 (318)	27.0 (3.9)				23.7 (8.2)	8.7 (5.9)	20.1	56.2	1020
R8	R	1.4 (12.6)	265 (94)	30.0 (6.2)				42.9 (10.6)	18.2 (9.5)	14.7	258.1	504
R9	R	44.1 (8.4)	540 (144)	28.3 (5.8)				27.9 (13.3)	10.8 (6.4)	5.6	45.8	231
R10	R	17.9 (16.8)	372 (231)	12.8 (5.3)				75.1 (7.6)	47.8 (14.7)	23.7	96.7	2368
R11	R	25.9 (5.8)	337 (75)	24.7 (2.4)				22.3 (2.7)	2.5 (1.4)	7.2	80.1	187
R12	R	1.3 (15.0)	454 (183)	40.9 (19.5)				36.1 (5.1)	11.9 (6.1)	320.0	426.1	1992
U6	U	16.5 (8.8)	306 (150)	28.0 (5.7)	-0.340	1.36	8.00	49.1 (32.1)	30.0 (23.4)	32.3	131.7	800

APPENDIX C
SITE SPECIFIC MODEL PARAMETERS

EPC (ppm) values and rate constants (K) estimated by the first order rate model for each P concentration and variance accounted for by model. Note, for site R3, day eight was eliminated from model predictions.

Site	EPC_0	K_0	$Variance_0$	$EPC_{0.5}$	$K_{0.5}$	$Variance_{0.5}$	EPC_{10}	K_{10}	$Variance_{10}$
R1	0.016	0.092	0.65	0.018	10.2	0.99	8.39	1.4	0.75
A1	5.39	0.014	0.98	8.55	0.021	0.97	18.9	0.09	0.93
A2	0.43	0.06	0.85	1.09	0.002	0.98	11.1	7.3	0.25
A3	0.86	0.41	0.82	0.904	0.33	0.57	8.2	0.42	0.79
U1	0.010	8.1	0.42	0.101	0.28	0.96	1.03	0.78	0.99
R2	0.037	0.18	0.67	0.051	1.3	0.99	6.39	0.64	0.94
R3	0.01	1.3	0.45	0.031	2.9	0.99	4.07	0.81	0.97
A4	5.9	0.88	0.89	3.3	0.27	0.99	11.8	0.18	0.72
R5	0.01	0.0001	0.33	0.11	1.4	0.99	6.4	2.02	0.98
R6	0.01	0.060	0.22	0.50	9.1	0.97	9.6	10.6	0.58
U3	0.023	0.0012	0.76	0.039	1.1	0.99	4.46	1.4	0.97
U4	0.01	0.0001	0.23	0.05	2.3	0.99	6.1	1.5	0.99
U6	0.001	0.0002	0.77	0.12	2.0	0.99	9.3	0.42	0.91

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BIOGRAPHICAL SKETCH

Elizabeth Deimeke was born and raised in St. Louis, Missouri. A Midwesterner at heart, Elizabeth attended college at Kenyon College in Gambier, Ohio. While there she participated in a research study examining the ecological condition of wetlands in the Cuyahoga River Watershed of northern Ohio. Under the guidance of Siobhan Fennessy, Elizabeth graduated with honors in 2006 with a Bachelor of Arts degree in biology and minor in environmental science. That same year she began working for the St. Louis Marathon & Family Fitness Weekend, a non-profit organization that plans a weekend of fitness activities for all generations. Additionally, Elizabeth trained, participated, and completed her first marathon in Nashville, Tennessee.

In 2007, Elizabeth was accepted and enrolled in the School of Forest Resources and Conservation at University of Florida under the guidance of Matthew Cohen. Upon completion of her M.S. degree, Elizabeth looks forward to raising her daughter and beginning a career in education.