



Phosphorus Retention and Storage by Isolated and Constructed Wetlands in the Okeechobee Drainage Basin

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EXECUTIVE SUMMARY 2006

Wetlands are known to accrete nutrients and other contaminants. Small historically isolated wetlands, which are a common feature throughout the Okeechobee Basin cover about 12,000 ha of the four priority sub-basins. These systems (about 50%) are presently ditched and drained. Hydrologic restoration of these wetlands may help to provide water storage and long-term phosphorus (P) retention within Okeechobee's four priority basins. In addition, constructed wetlands strategically placed in the watershed can also aid in increasing P retention. Therefore, it is important to understand the role and effectiveness these systems have in storing water and subsequently retaining P from incoming water such as agricultural runoff.

In this 2006 report, we outline the final update of our multi-year, multidisciplinary project to determine the phosphorus storage and retention capacity of historically isolated wetlands and to demonstrate the efficacy of pilot-scale constructed wetlands to store and retain incoming phosphorus.

The project tasks, objectives and percent completion to date are outlined in table below.

Table 1. Description of tasks and objecties.

Task	Objectives
1.0	Introduction
1.1	Project objectives and workplan
2.0	Identification of cooperators
3.0	Evaluation of isolated wetlands for phosphorus attenuation
3.1	Quantify P storage in historically isolated wetland soils at the landscape-scale.
3.2	Determine soil characteristics and processes responsible for effective P storage in historically isolated wetland soils
3.3	Determine efficacy of hydrologically restored isolated wetlands and constructed wetlands to store and retain P
3.4	Measure P assimilation rates and equilibrium P concentrations of wetland soils
3.5	Quantify the effect of fluctuations in hydrology on P flux
3.6	Monitor subsurface phosphorus transport
4	Optimize on farm constructed wetlands to increase P removal performance using pilot scale studies
5	Review current hydrologic and P models for adaptation to the Basin and use these models to simulate P retention capacity
6	Communicate the findings and experiences gained during this project to dairy farmers and beef cattle ranchers through extension media

The objective of Task 3.1 the project was to quantify P storage in historically isolated wetland soils at the landscape-scale

About half of the 12,000 ha of historically isolated wetlands are drained. In a landscape-scale soil P survey, we found that isolated wetland surface soils (0-10 cm) generally contained more P (250 kg ha^{-1}) than surrounding their upland pasture surface soils (130 kg ha^{-1}). Wetlands located in dairies contained more P than those located in improved pastures, which contained more P than wetland soils in unimproved pastures/rangelands. We predicted using spatial and spectral data (spectral data reflecting vegetation and soil moisture were the most important predictors) that on average, surface soils (0-10 cm) of all historically isolated wetlands within the four priority basins stored 292 kg P ha^{-1} .

The objective of Task 3.2 was to determine soil characteristics and processes responsible for effective P storage in historically isolated wetland soils

The intensive assessment of 20 isolated wetlands suggests that land use impacts P storage in soils and water. We found that surface waters of dairy wetlands had greater temperatures, dissolved solids, pH and lower redox than wetlands within improved pasture and unimproved rangelands. Similarly, surface water total P concentrations were greatest in dairy wetlands (2.51 mg L^{-1}), relative to wetland waters of improved pasture (0.28 mg L^{-1}) and waters of unimproved/rangelands (0.04 mg L^{-1}).

Dairy wetland surface soils had much greater concentrations of total P ($1,229 \text{ mg kg}^{-1}$), inorganic P (696 mg kg^{-1}) and water extractable P (22.4 mg kg^{-1}) than surface soils collected from improved and unimproved pasture wetland soils. Inorganic P was more than 15 times greater in dairy wetland soils relative to soils collected from wetlands in improved or unimproved pasture. Labile organic P ranged between 8 and 12% of soil total P, with no difference between land uses. Improved pasture and unimproved pasture wetland soils had greater amounts of soil total P stored as slow to unavailable fractions of organic P (54%) relative to dairy soils, which stored 30 %. We found significant relationships between soil organic matter and wetland soil total P in soils collected from unimproved and improved, whereas in dairy wetland soils, total P was more related to soil Ca content and other soil mineral components that contained Mg, Fe, and Al.

The objectives of Tasks 3.3 and 3.6 were to determine efficacy of hydrological restored isolated wetlands to store and retain P from cow-calf operations, and to determine subsurface phosphorus transport

Findings from our hydrological restoration site-specific (Larson Dixie and Beaty ranches) studies indicate that isolated wetlands collect water from surrounding uplands for a short time (a matter of days) during the year and then slowly discharge to groundwater (duration of weeks to months). Deep marsh areas of these wetlands (two wetlands at Larson Dixie and two wetlands at Beaty) had a hydroperiod that ranged between 120 and 275 days, while upland areas were wet for less than 20 days. Site

surface waters tend to have variable concentrations of P. During two and a half years of site monitoring, surface water TP ranged between 0.3 to 1.7 mg P L⁻¹.

Wetland-groundwater exchange was characterized in a constrained water budget framework and groundwater recharge for the wetlands ranged from 0.5 to 0.8 cm/d. The hydraulic resistivity (R), of the wetlands was passively measured during natural drawdown events over three years, with approximately 7 to 9 events per wetland. The R values for the four study wetlands were between 49 and 96 days. These R values were higher than those reported at other wetland sites, because of differing soil hydraulic properties. Darcy's Law was used to accurately predict the groundwater outflow from the wetlands using the calibrated R values. This approach is useful to obtain long-term (weeks and months) estimates of surface water-groundwater exchange in these systems.

To help measure P flux in groundwater, an appropriate resin that had a high capacity to sorb P was identified. Preliminary onsite measurements indicate there is a decreasing P-flux with soil depth and values measured ranged between 0.25 and 0.90 mg m⁻¹ d⁻¹. Total P concentrations in groundwater sampled was generally below 1 mg P L⁻¹. Also, to help determine the potential diffusive flux of P from shallow soil porewater to overlying wetland waters onsite sampling using porewater sippers was undertaken. A potential diffusive flux of P from shallow pore water to overlying wetland waters was estimated as 36 mg P km⁻² d⁻¹.

To determine P stored in isolated wetlands prior hydrologic restoration P storages in plant biomass, plant litter and soil was measured. Post restoration, P storages will also be measured with the difference between pre- and post-restoration being the effect of restoration. Prior hydrologic restoration, soils (0-10 cm) stored 89% of the P relative to above ground vegetation, plant litter and below ground roots. On average, wetland soils in deep marsh and shallow marsh areas stored between 120 and 180 kg P ha⁻¹. Most of the P stored in plant biomass was below ground; however, this depended on landscape position, as surrounding pasture uplands tended to have greater amounts of P stored in below ground biomass relative to wetlands. In general, we found that these isolated wetland ecosystems contained more P (kg ha⁻¹) than their surrounding uplands. If hydrologic restoration of these wetlands increased wetland area by 20% (present wetland area ranges between 1 and 2 ha), the additional P storage gained could be up to 13 kg P ha⁻¹.

To investigate P dynamics and storages in wetland and pasture vegetation, litter and soil various laboratory- and mesocosm-scale experiments were undertaken. The first experiment lasted for one year and investigated the P stored in *Paspalum notatum* (Bahia grass) and *Hemarthria altissima* (Floralta) under various hydrologic regimes. In general, Floralta produced eight times as much biomass relative to Bahia under the various hydrologic regimes; however, P storages were somewhat similar between plant species. Bahia tended to store most P in below ground tissue, whereas Floralta stored most P in above ground tissue. Flooding tended to increase the P concentration in plant tissue. Findings suggest that Floralta may have a greater hydrologic tolerance to flooding.

In another experiment, senesced wetland and pasture plant material released between 0.01 mg P per g of plant tissue and 1 mg P per g of plant tissue to floodwaters low in P (0.09 mg P L^{-1}). Most of the P was lost from the plant material within 72 hrs. In a complimentary field study, it was found that in 1 yr 40% of plant litter is decomposed.

To investigate soil-water P dynamics, intact soil cores (0-10 cm) were collected four times during a one and a half year period from wetlands and pasture uplands at Beaty and Larson Dixie. At each time, intact site soils were incubated back in the laboratory with floodwaters that had variable P concentrations ($0.08 - 1 \text{ mg P L}^{-1}$) for seven days. Wetland and upland soils at Beaty tended to flux similar amounts of P to overlying water. Flux rates ranged between 25 and $55 \text{ g ha}^{-1} \text{ d}^{-1}$. At Larson Dixie, wetland soils fluxed less P ($13 \text{ g ha}^{-1} \text{ d}^{-1}$) than surrounding upland soils ($21 \text{ g ha}^{-1} \text{ d}^{-1}$).

The objective of Task 3.4 was to measure P assimilation rates and equilibrium P concentrations of wetland soils

Intact soil cores (0-30 cm) were collected from Larson Dixie, Beaty, and Paleaz ranches. Soils were brought back to laboratory where they were incubated under flooded conditions for 28 days at various P concentrations (0, 0.5, 1 and 5 mg P L^{-1}). At the end of 28 days, floodwaters were exchanged and soils were incubated under flooded conditions for another 28 days. Soils were flooded for five 28 days cycles. This was done to estimate at what point soils reached equilibrium (point at which soils neither released or retained P). For soils that were incubated under low P conditions ($0-0.5 \text{ mg P L}^{-1}$) soils typically released P; however, for soils that were incubated under high P conditions ($1-5 \text{ mg P L}^{-1}$) soils generally tended to retain P. Leachate from soil cores was also collected.

The objective of Task 3.5 was to determine the effect of fluctuations in hydrology on P flux

Intact soil cores (0-30 cm) were incubated under various hydrological regimes in the laboratory, and it was found that the soils with the highest nutrient status fluxed most P and this flux was independent of P concentration (0.1 and 1 mg P L^{-1}) in overlying water. As we increased the number of flooding periods, wetland soils tended to retain P ($\sim 40 \text{ g P ha}^{-1} \text{ d}^{-1}$) from overlying waters that had a P concentration of about 1 mg SRP L^{-1} . During these periods, P retention by soils was largely governed by P concentration in overlying water. On average, soils that were pre-flooded for about two months and then re-flooded retained more P ($7.3 \text{ g P ha}^{-1} \text{ d}^{-1}$) than soils that were pre-saturated for two months and then flooded ($0.7 \text{ g P ha}^{-1} \text{ d}^{-1}$) or soils that were pre-drawn down for two months and then flooded ($2.7 \text{ g P ha}^{-1} \text{ d}^{-1}$).

The objective of Task 4 was to optimize on farm constructed wetlands to increase P removal performance using pilot scale studies

During 2006, the pilot-scale constructed wetlands adjacent to Larson Barn #5 experienced a dramatic increase in inflow TP concentrations, due to drought conditions

that “concentrated” the tertiary lagoon that serves as a water source. Consequently, the P load to the wetlands doubled from an average of $34 \text{ g P m}^{-2} \text{ yr}^{-1}$ in 2005 (July –Dec) to $63 \text{ g P m}^{-2} \text{ yr}^{-1}$ in 2006 (Jan – Dec). The conventional treatment wetland vegetation configurations (cattail, cattail/SAV, and torpedograss) reduced inflow TP levels from 8.6 to outflow concentrations of $7.6 - 7.8 \text{ mg L}^{-1}$, providing an average mass P removal rate of $10 \text{ g P m}^{-2} \text{ yr}^{-1}$. The pasture grass (paragrass) wetland performed slightly better (mean outflow TP of 7.3 mg L^{-1}), and the water hyacinth wetland provided the most effective P removal (mean outflow TP of 5.6 mg L^{-1}). Average P removal in the water hyacinth wetland was relatively high, averaging $25 \text{ g P m}^{-2} \text{ yr}^{-1}$ (35% on a percentage basis).

The objective of Task 5 was to review current hydrologic and P models for adaptation to the Okeechobee Drainage Basin and use these models to simulate P retention capacity (Task 5)

Modeling P reduction in a hypothetical isolated wetland landscape provided insight into the effectiveness of various wetland configurations to treat P impacted groundwater. In a single isolated wetland system, reduction of P in groundwater discharge along an outflow boundary was about 20%. Reduction of P in the landscape increased to 28% for simulations including two isolated wetlands. An increase in wetland overlap corresponded to a decrease in P reduction. The model suggests under restored conditions (non-ditched), spatially distributed isolated wetlands in a landscape may help reduce P loading.

The passive nutrient flux meter (PNFM) was utilized to measure groundwater and phosphate flux from the isolated wetlands and along a transect of a ditch draining the isolated wetlands. With the estimated phosphate flux from isolated wetlands and the drainage ditch, and knowledge of the surface water-groundwater interaction for these wetland systems, phosphate mass loads can be scaled up from single isolated wetlands to the basin-wide phosphate mass load. Preliminary analyses indicate that there is the possibility of reducing at least one to two metric tons of phosphorus per year from entering Lake Okeechobee by increasing the hydrologic retention in isolated wetlands.

The objective of Task 6 was to communicate the findings and experiences gained during this project to dairy farmers and beef cattle ranchers through extension media

Research results are being collected, consolidated and synthesized for extension activities. To date extension activities have supported landowner awareness in participating in research efforts. Past activities include: information transfer from this project to the Environmental Service Pilot Project; provided technical information and expertise into a PBS documentary to highlight water quality issues and the use BMPs within Okeechobee watershed as it relates to Everglades restoration; and continued collaboration between this project and the CSREES grant entitled: Wetland Enhancement Decision-Making Tools/Training for Landowners and Technical Service Provider.

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1.0 Introduction

Wetlands are known to accrete nutrients and other contaminants. Natural and constructed wetlands are often managed to help improve water quality. The extent of management required depends upon the nutrient/contaminant retention capacity required of the wetland, contaminant load to the wetland, and the desired wetland effluent quality. Constructed wetlands are typically used as buffers to retain nutrients and other contaminants; they are usually managed to optimize retention processes. Small-scale wetlands can be managed effectively by altering hydraulic loading or integrating them with conventional treatment systems, while large-scale wetlands can be managed by controlling nutrient/contaminant loads (Reddy et al. 1995; Reddy et al. 1999).

Phosphorus (P) retention by wetland soils includes surface adsorption on minerals, precipitation, microbial immobilization, plant uptake, and these processes may be combined into two distinct P retention pathways: sorption and precipitation with mineral soil components, and long term P retention by peat accumulation/accretion (Figure 1). Phosphorus sorption in soils is defined as the removal of phosphate from soil solution to the solid phase, which includes both adsorption and precipitation processes. Phosphorus immobilization through microbial and plant uptake are also significant pathways for P removal; however, when plants and microbes die off, the P contained in cellular tissue may either recycle within the wetland, or may be buried with refractory organic compounds. Accumulation and accretion of organic matter has been reported as the long term sink for phosphorus in wetlands. Wetland soils tend to accumulate organic matter due to the increased production of detrital material from biota and the low rates of decomposition in waterlogged conditions. Soil accretion rates can vary depending on loading rates and hydraulic influences. For example, accretion rates in productive natural systems such as the Everglades have been reported as high as one centimeter or more per year. With time, productive wetland systems will accumulate organic matter (which ultimately forms peat) that has different physical and biological characteristics than the underlying soil. Eventually, this new material settles and compacts to form new soil with perhaps different P sorption characteristics than the original soil. As the wetland ages, steady accumulation of organic matter can potentially decrease the efficiency of the wetland to assimilate additional P and alter the hydraulic flow paths, as organic accretion is seldom uniform throughout space. These conditions can result in elevated effluent P concentrations. However, management of newly accreted material by consolidation or removal can improve the overall P retention capacity of the wetland.

Small isolated wetlands are a common feature throughout the Okeechobee basin. They are also dominant feature on land areas used for cow-calf operations, especially in the four priority basins (S-191, S-65D, S-154, and S-65E), which are those sub-basins that have contributed disproportionate amounts of P entering Lake Okeechobee. These small-scale wetlands may provide a significant storage and assimilative capacity for phosphorus (P) runoff within the landscape. Historically, many of these wetlands were truly isolated and only linked to surface water courses during periods of heavy rainfall. Under present conditions, most of these wetlands have been connected and partially

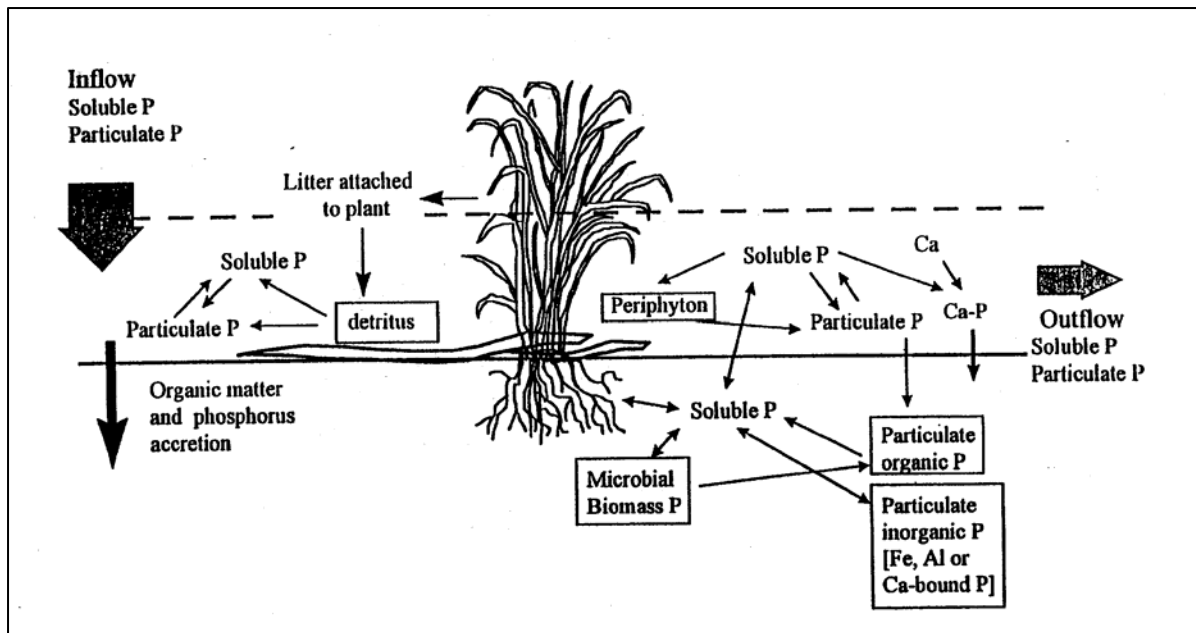


Figure 1. Schematic showing phosphorus cycling in isolated wetlands.

drained through a network of ditches and canals such that they convey surface water from wetland to the drainage network and ultimately to Lake Okeechobee. About 45% of wetlands north of the lake have been ditched.

At present, the role of isolated wetlands in P storage and assimilation capacity has only briefly been addressed within the Okeechobee basin. In addition, factors that influence storage and assimilative capacity of isolated wetlands based on land use, vegetative type, hydrologic connectivity and soil has not been addressed at a large spatial scale. Isolated wetlands cover approximately 16.6% of the landscape and understanding their role in P storage is critical to long-term efforts to reduce P loading to Lake Okeechobee. In addition, the National Wetland Inventory estimates that wetlands cover about 18% of the four priority basins and 59% of those are historically isolated. We hypothesize that isolated wetlands in the four priority basins have a significant capacity to store and assimilate P. We also hypothesize that the P stored in these systems can be stabilized by hydrologic manipulation of wetland water levels and addition of chemical amendments.

Studies conducted within the past two decades have also implicated dairy farms as a prominent source of the P that enters Lake Okeechobee through the Taylor Creek – Nubbin Slough and Kissimmee River watersheds. In the 1980's as many as 38,000 dairy cows were maintained in a 4,000 square mile area north of the lake. At that time, the P loading (from manure, fertilizers and runoff) from dairies in these watersheds was estimated to be 264 kg P ha^{-1} . Due to poor P retention by sandy pasture soils, much of the P ultimately is exported downstream. Considerable work was performed in the late 1980's and early 1990's to reduce P inputs from dairies to the lake. A number of dairies were closed, and the remaining dairies used "best management practices" (BMPs) as well

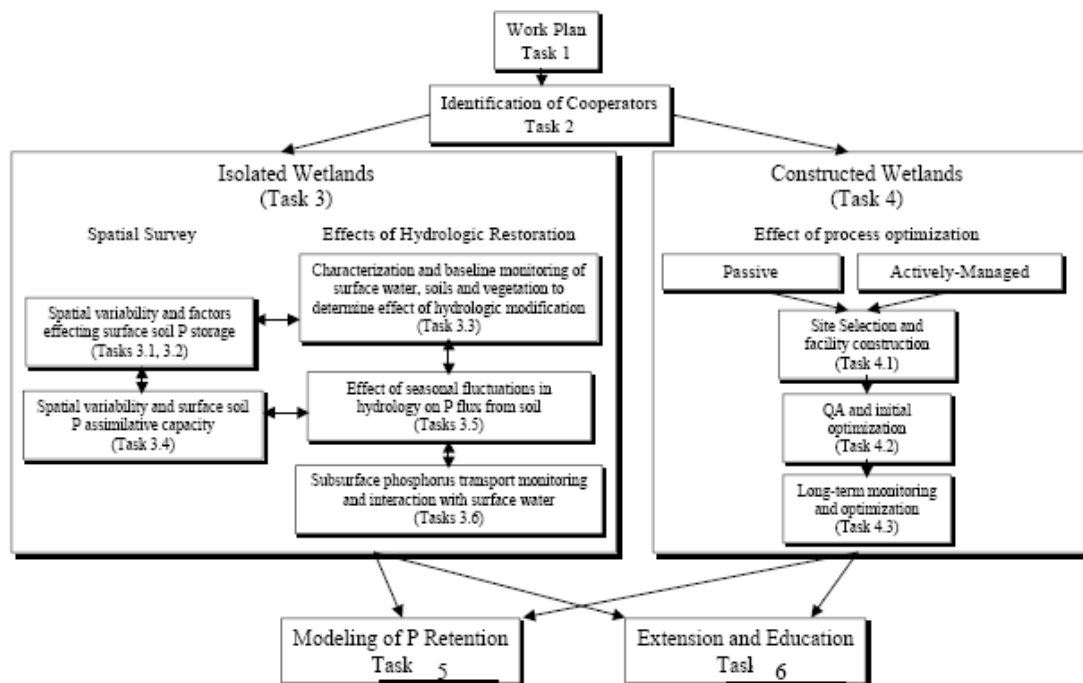
as retrofits to better control and mitigate the export of P from farms. These retrofits typically consisted of improvements to lagoon systems, as well as the use of spray irrigation systems for growing hay and forage crops. At that time, projections of 60 ha of reservoir and 135 ha of spray fields were thought to be required for a 1,000 cow dairy. Despite these efforts, P loads to Lake Okeechobee during the past decade have exceeded desired levels and continued improvements are needed to further reduce P runoff from dairies. The purpose of this part of the project is to evaluate the effectiveness of constructed wetlands to retain incoming P using wetland wet cropping systems for P removal. The wet cropping system would be a two-stage system, utilizing a vegetative crop with known animal feed value on the front end, and a passive or managed periphyton wetland on the back end for final polishing for incoming waters.

Project Objectives

The objectives of this interdisciplinary project are:

- Quantify P storage in historically isolated wetland soils at the landscape-scale. (Task 3.1)
- Determine soil characteristics and processes responsible for effective P storage in historically isolated wetland soils. (Task 3.2)
- Determine efficacy of hydrologically restored isolated wetlands and constructed wetlands to store and retain P from cow-calf and dairy operations, respectively. (Tasks 3.3-3.5)
- Measure P assimilation rates and equilibrium P concentrations of wetland soils. (Task 3.4)
- Quantify the effect of fluctuations in hydrology on P flux. (Task 3.5)
- Optimize on farm constructed wetlands to increase P removal performance using pilot scale studies. (Task 4)
- Review current hydrologic and P models for adaptation to the Basin and use these models to simulate P retention capacity. (Tasks 3.6 and 5)
- Communicate the findings and experiences gained during this project to dairy farmers and beef cattle ranchers through extension media. (Task 6)

This project represents a five year joint effort among several agencies including agencies including: the University of Florida –Institute of Food and Agricultural Sciences (UF/IFAS), FDACS, SFWMD, FLDEP, DB Environmental Lab Inc., and dairy farmers and ranchers. Specific research and monitoring tasks are described separately for each of these systems.



2.0 Identification of cooperators

Within the individual tasks we identified cooperating landowners. Most of our collaboration with landowners occurred during tasks 3.1, 3.2, 3.3, and 4. The table below lists landowners that we cooperated with during this project without whose help many of these task could not be completed.

Table 2. Landowners that were collaborated with during project.

Task	Cooperators
3.1	Frieda Wise; Williamson Cattle Company; Hanyes & Susan Williams; MW Wherrell; Edwin Walpole; William and Linda Waggener; Underhill Farms; Tripe Horse & Cattle Co; tiitf/State of Florida; The Okeechobee Partnership; Taylor Creek Ranch; Tampa Farm Service, INC; Statewide Financial Inc; SFWMD; Wm;; Allen Jr & Nickleen Smith; Susan Smith Ranch LLC; Sacramento Farms; K. Rucks; Rucks Dairy; Roy Hancock; Robert and Martha Joiner; Richard Smith; Richard J Hales; Reedy Creek Estates; Ram Jairam Trustee; Prescott, Seller; Prairie View Land Co; Pelaez & Sons Inc; Larry Overton; N. Ephraim; Gerald Newcomer; McArthur Farms, Inc.; Marion Wagner, Trustee; M Cross Ranch; Lykes Brothers; Woody Larson; Louis D Nekoliczak Trustee; LOR Inc; Lois K. Johnson; Live Oak Trust; Little Lakes Hunting Lodge; D. Alan and Karen Lewis ; Lazy O Ranch; Larson Dixie Ranch; Larson Dairy Inc.; Elsie M Lanier; Kirton Dudley R Trustee; Keith C. Wold LLC and Elaine J; JW Sod Inc; John Edward III Trustee; James Fraser; Harvey Cattle Co. Inc.; Hamrick & Sons Inc; Joseph and Wight Hall; Grassy Island Ranch; Glendoria Sutton; Fred and Sue Diamond; James Jr Fowler ; Four K Ranch; Flying G Dairy; Marshall Evans; Robt Edward Duncan; Dry Lake Dairy; Debra Seree, Trustee; Davie Dairy; Darrell W. Bowers; Danny Fairclaw; Pete & Susanne Clemons; Claudio & Yvonne Alvarez; Callaway Land & Cattle Co.; Butler Oaks Dairy; Beaty D S Trustee; Elwyn & Patricia Bass; Bass Ranch; B & E Ranch and Grove LC; Robt Arnold; Anthony & Garrity Armante
3.2	Woody Larson; Larry Overton; Jim Fraser c/o Norman Ephraim; Beaty D S Trustee; Glendoria Sutton; Louis Larson; Robert Arnold; William Smith; Kirton Dudley R Trustee; B & E Ranch and Grove LC; Dry Lake Dairy; Hamrick and Sons; Gerald Newcomer; Seller Prescott; Danny Fairclaw ; MacArthur Dairy; Tampa Farm Services; Norman Ephraim; Keith Rucks
3.3	Beaty D S Trustee and Larson Dixie Ranch (Woody Larson)
4	Larson Dairy

3.0 Evaluation of isolated wetlands for phosphorus attenuation

3.1 Synoptic survey of ambient phosphorus storage in isolated wetlands

A synoptic survey of soil P storage in isolated wetlands was undertaken in the four priority basins between May and November, 2003. Details of this study are included in the 2004 annual report. The 2005 report presents a summary of the main findings. As a recap, some of the key findings included:

Key Findings

- 12,000 ha of isolated wetlands exist in the four priority basins.
- About half of these wetlands are drained.
- Isolated wetland soils contained more P than surrounding upland soils suggesting that wetland soils could store more P than surrounding soils.
- Wetlands located in dairies contained more P than wetlands in improved pasture, which contained more P than wetland soils in unimproved pastures/rangelands.
- It was predicted using ancillary spatial data that about 290 kg P ha⁻¹ was stored in surface soils (0-10 cm) of all historically isolated wetlands in the four priority basins.

3.2 Intensive assessment of isolated wetlands in selected dairy and cow-calf operations sites.

Introduction

In the four priority basins about 48% of the agricultural land is improved pasture, 7% is dairy and 6% is unimproved. Within the synoptic survey (Task 3.1) 72% of the 118 wetlands sampled were in improved pasture, 18% in dairy and about 10% in unimproved pastures. For this task (Task 3.2), we sampled 20 isolated wetlands within dairy, improved pasture in cow-calf operations and unimproved rangeland located in the four priority basins for an intensive assessment of their soil characteristics. These candidate sites were identified using data from the synoptic survey (Task 3.1). Sites were selected based on land use, total P concentration in soil, soil organic matter, and soil metal content. On that basis, we sampled four historically isolated wetlands in dairies, 12 in improved pasture and four in unimproved/rangeland areas.

The objective of this task was to report on labile and non-labile pools of P in isolated wetland soils from the different land uses to quantify soil P availability and its stability. An additional task, which will be described later, was to intensively assess the P flux rates from wetland and uplands soils collected from Larson and Beaty ranches.

Materials and Methods

All 20 sites were sampled for soil and water during October and November, 2005. Similar to the synoptic survey, we also adopted a stratified random sampling approach, as work carried out for Task 3.1 found that there was a P gradient within historically isolated wetlands, with wetland center/open marsh soils storing more P than wetland edge/shallow marsh soils, which contained greater amounts of P than surrounding upland soils.

When wetland sites were flooded to 30 cm, we sampled site waters. Site waters were sampled and prepared for analyses that included soluble reactive P, total dissolved P, and total P. During site sampling, we also recorded site water physical chemistry using a YSI meter. Wetland water temperature, specific conductance, dissolved oxygen content and oxygen reduction potential were measured.

For the soils, we collected three soil samples from wetland centers and three soil samples from wetland edges. Soils were sampled to a depth of 30 cm. Depth increments were 0-10 cm and 10-30 cm; however, this varied somewhat, depending on soil horizon. In some instances, three depth increment samples were collected per one soil core. All soil samples were measured for: soil pH; water content; bulk density; loss on ignition (LOI); total P; inorganic P; total carbon and total nitrogen; water extractable P; P sorption capacity; acid extractable P, Ca, Mg, Al, and Fe and; ammonium oxalate extractable P, Fe, and Al. All soil samples were collected during October and November, 2005.

Twelve of the sites sampled underwent a more intensive assessment. Wetland soils from four dairy, four improved and four unimproved sites underwent soil P fractionation to determine biologically labile and stable fractions of P in surface and subsurface soils. Analyses enabled P bound to Al and Fe ($\text{NaHCO}_3\text{-P}_i$), P bound to Ca/Mg (HCl-P_i), labile organic P (Labile org. P), microbial biomass P (MBP), fulvic acid bound P (FAP), humic acid bound P (HAP), and residual non-reactive P (Res P) to be determined.

Results and Discussion

We found that dairy wetland waters tended to have greater temperatures, higher specific conductance (> dissolved solids), higher pH and lower oxidation reduction potential (ORP) than wetlands within improved pasture and unimproved/rangelands (Table 3). This suggests that dairy wetland waters were more impacted in terms of water quality parameters measured than improved and unimproved/rangelands.

Table 3. Wetland water site characteristics by major land use. Site water chemistry data was collected during October and November 2005. Values represent mean $\pm$ one standard error (SE).

Land-use	Temperature -----°C----	Specific Conductance --mS cm ⁻¹ ---	Dissolved oxygen -mg L ⁻¹ --	pH	ORP -----mV-----
Dairy	25 $\pm$ 1.7	1.35 $\pm$ 1.2	2.1 $\pm$ 2	7.2 $\pm$ 0.5	-236 $\pm$ 139
Improved	24 $\pm$ 0.9	0.08 $\pm$ 0.1	2.1 $\pm$ 2	4.9 $\pm$ 0.8	-46 $\pm$ 28
Unimproved	21 $\pm$ 0.1	0.05 $\pm$ 0.0	2.8 $\pm$ 0	4.0 $\pm$ 0.1	3 $\pm$ 3

We also collected water samples for total P (TP), total dissolved P (TDP), and soluble reactive P (SRP). All sites were not sampled, as all sites did not have open water. Where open water was present, P concentrations were greatest in dairy wetlands, which were greater than wetlands within improved pasture, which in turn, were greater than the P concentrations in wetland waters of unimproved pasture (Table 4).

Table 4. Phosphorus concentrations (total P, total dissolved P, and soluble reactive P) in waters collected from wetlands within dairy, improved and unimproved land uses. Samples were collected during October and November, 2005. Values represent mean $\pm$ one standard error (SE).

Land use	Sites	Total P	TDP	SRP
-----mg L ⁻¹ -----				
Dairy	4	2.52 $\pm$ 0.83	1.92 $\pm$ 0.8	1.96 $\pm$ 0.7
Improved	11	0.29 $\pm$ 0.05	0.21 $\pm$ 0.04	0.16 $\pm$ 0.03
Unimproved	1	0.04 $\pm$ 0.00	0.027 $\pm$ 0.003	0.016 $\pm$ 0.004

Wetland soils in dairy pasture tended to have greater soil pH, Ca, and Mg concentrations than wetland soils collected from improved and unimproved pasture wetlands (Tables 5 and 6, which is probably due to increased inputs of food and fertilizer in dairy operations. Wetland soils in improved and unimproved pasture had similar soil Mg and Ca concentration; however, wetland soils in improved pasture had greater soil pH than soil collected from wetlands in unimproved pasture (Tables 5 and 6). Dairy wetland soils also had the greatest soil organic matter content ($P < 0.05$; $n = 256$). This is surprising, as with increased cattle impacts, one expects the loss of soil organic matter due to cattle trampling. Data may suggest that rather than direct cattle impact, nutrient loading from surrounding upland areas may be the source of P.

When a t-test was undertaken to compare surface and subsurface soils, it was found that surface soils had greatest concentrations of most soil parameters measured (water content, organic matter, total N, total C, Ca, Mg, Fe) ($P < 0.01$); however, surface and subsurface soils had similar soil pH and Al content. Further, surface soils had much lower bulk density than underlying subsurface soils (t-ratio = 8.12; $P < 0.001$) which is probably due to its slightly higher organic matter and soil water content.

Table 5. Physicochemical characteristics of wetland soils collected from dairy, improved and unimproved pasture. Per parameter, the first column represent mean and the second column represents one standard error (SE).

Landuse	Depth	n	pH		Bulk Density		Water content		Organic matter		Total Nitrogen		Total Carbon	
			Mean	SE	---g cm ⁻³ ---		-----%-----							
Dairy	0-10	24	6.1	0.2	1.00	0.1	56	4	38	6	1.24	0.18	19.00	2.90
Dairy	10-30	27	5.9	0.2	1.25	0.1	38	5	27	6	0.80	0.19	14.90	3.63
Improved	0-10	68	5.1	0.1	1.01	0.0	46	2	22	2	0.73	0.07	10.56	1.08
Improved	10-30	90	5.0	0.1	1.35	0.0	29	2	12	2	0.33	0.05	5.64	0.91
Unimproved	0-10	24	4.3	0.1	0.78	0.1	44	4	30	4	0.88	0.11	15.05	2.18
Unimproved	10-30	26	4.4	0.1	1.32	0.1	23	3	11	3	0.28	0.08	5.55	1.68

Table 6. Surface and subsurface metal characteristics of soils collected from wetlands within dairy, improved and unimproved landuses. Per parameter values in first column are means and values in second columns are one standard error (SE). Soils were sampled during October and November, 2005.

Landuse	Depth	n			-----1 M HCl-----					-0.2 M Ammonium oxalate-				
			Ca		Mg	Fe	Al			Fe		Al		
			Mean	SE										
			-----mg kg ⁻¹ -----											
Dairy	0-10	24	4715	1036	828	161	296	48	4027	978	609	88	4184	1088
Dairy	10-30	27	1141	216	232	47	130	31	3947	1631	607	179	4042	1450
Improved	0-10	68	1741	174	248	21	473	45	2272	537	663	62	1798	392
Improved	10-30	90	644	106	83	17	293	53	1914	378	431	59	1576	282
Unimproved	0-10	24	1189	225	277	49	468	57	1831	494	707	91	2196	596
Unimproved	10-30	26	403	123	142	55	176	30	1764	526	230	41	1748	521

The total phosphorus concentration in dairy soils (both surface and subsurface; 0-10 cm and 10-30 cm) was nearly three times greater than the concentrations found in improved and unimproved pasture wetland soils ($P < 0.001$; $n = 255$) suggesting that dairy wetland soils were highly P impacted. Wetland soils in improved pasture had similar total P concentration to unimproved pasture wetland soils (Table 7) with concentrations being medium to low ($< 600 \text{ mg P kg}^{-1}$). Dairy wetland soils also had greatest concentrations of soil total inorganic P, water extractable P and the greatest P sorption capacity, as measured by a single incubation with $1000 \text{ mg P kg}^{-1}$ of soil ($P < 0.001$; $n = 257$; Table). This suggests that although dairy wetland soils contained large amounts of available P they still had the capacity to sorb P. From the literature, it suggests that as P load is increased, P retention also increases (Reddy et al., 1999; Dunne et al., 2006). Further, increased concentrations of Ca and Mg in dairy wetland soils may contribute to increased P sorption capacity of dairy soils relative to improved and unimproved pasture soils, which had less Ca and Mg. The iron content of wetland soils was similar between landuse; however, the aluminum content was greatest in dairy wetland soils relative to improved and unimproved ($P < 0.001$; $n = 204$). Again, this could contribute to dairy wetland soils having greater P sorption capacity relative to wetland soils from the two other landuses. Depending on land use, inorganic P was 60%, 14%, and 12% of soil total P content (dairy, improved, unimproved, respectively) suggesting that dairy soils were more impacted with inorganic P fractions; thus soil P content being potentially more available to wetland waters. The water extractable P concentrations, which is the P that is directly available to low P waters (soils are extracted with DDI water that contains little P; $< 0.01 \text{ mg P L}^{-1}$) was a small portion of soil total P. Dairy soils contained about 2.4 % of soils total P as WEP, whereas improved and unimproved pasture wetland soils contained less than 1.5% of soil total P as WEP.

Table 7. Phosphorus characteristics from wetlands within dairy, improved and unimproved land uses. Per parameter, values in first column are means and values in second columns are standard error (SE). Soils were sampled during October and November, 2005.

Parameter	Dairy				Improved				Unimproved			
	0-10 cm		10-30 cm		0-10 cm		10-30 cm		0-10 cm		10-30 cm	
	Mean	SE										
	-----mg kg ⁻¹ -----											
Total P	1253	230	394	67	354	34	147	18	315	33	107	23
Inorganic P	717	181	245	80	44	4	18	2	37	4	10	2
P sorption capacity	512	70	402	68	271	32	222	29	303	68	209	52
Water Ext. P	22	5	8	2	1	0	1	0	2	1	1	0
NaHCO ₃ Pi	164	39	85	28	13	2	8	1	7	1	4	0
HCl-Pi	433	141	78	26	9	2	3	1	20	10	2	1
Labile org. P	29	4	24	11	17	2	11	3	20	4	10	3
MBP	75	21	19	8	41	11	10	4	30	5	6	2
FAP	151	26	41	12	77	12	29	10	81	22	19	6
HAP	52	9	18	5	24	4	10	3	32	9	12	4
Res P	62	11	16	5	31	4	10	3	26	7	8	1

Water Ext. P = water extractable P; NaHCO_3 Pi = Fe and Al bound P; HCl-Pi = Ca and Mg bound P; Labile MBP = microbial biomass P; FAP = fulvic acid bound P; HAP = humic acid bound P; Res P = residual bound P.

Soil phosphorus concentrations also changed with soil depth and it was found that surface soils contained greatest concentrations of water extractable P, P sorption capacity, total P, and total inorganic P, relative to underlying soils ($P < 0.001$; Table 7).

Fractionating wetland soils into inorganic and organic P fractions suggested that land use had an impact on the availability of P in soil. For example, wetland soils from dairy sites stored 51% of soil P in inorganic fractions (Al/Fe bound P + Ca/Mg bound P) relative to improved and unimproved pasture wetland soils that stored about 17% in inorganic fractions (Figure 2). Improved and unimproved pasture wetland soils contained larger fractions of P as available organic P fractions. These include both labile organic P and microbial biomass P. For these fractions to become available to the water column, they need to be mineralized to inorganic fractions. Slowly available organic P fractions such as fulvic acid bound P and humic bound P (humic being more slowly available relative to fulvic) and residual P (recalcitrant organic P fraction or mineralized inorganic P that is not available) was nearly three times greater in improved and unimproved pasture wetland soils relative to dairy wetland soils (Figure 2). This suggests that a larger portion of soil P is unavailable in improved and unimproved wetland pasture soils to a soil depth of about 30 cm than dairy wetland soils.

To determine relationships between soil characteristics that were intensively assessed, a correlation matrix was constructed (Table 8). Many soil physical and chemical characteristics and soil P fractions had relationships with each other. Some of the best relationships were between soil water content and soil organic matter, total N and total C; total inorganic P and total P; total inorganic P and Fe/Al bound P, and Ca/Mg bound P; total P and Ca/Mg bound P; P sorption capacity and Al content; water extractable P and Al/Fe bound P; Ca and Mg were related to residual bound P; Al content with fulvic and humic acid bound P; and labile organic P with fulvic acid bound P. Using some of these initial relationships, regression analyses were determined. For example, water content accounted for between 52 and 68% of the variability in slowly to unavailable fractions P (Figure 3). Knowing this information is useful, as knowing water content alone, could provide information on relatively slowly to unavailable P forms in wetland soils. In addition, the aluminum content (as extracted with 0.2 M ammonium oxalate) of wetland soils accounted for 79% of the variability in P sorption capacity (Figure 4). Using this relationship, we scaled up to a larger landscape-scale, taking into account the 118 wetlands that were previously sampled in Task 3.1. In Task 3.1, soil Al content was measured; however, sorption capacity of these wetland soils was not. Therefore, using the relationship shown in Figure 4, it was predicted that the sorption capacity in the 118 wetland soils sampled in Task 3.1 ranges between 83 and 127 kg P ha⁻¹ depending on land use (Table 9).

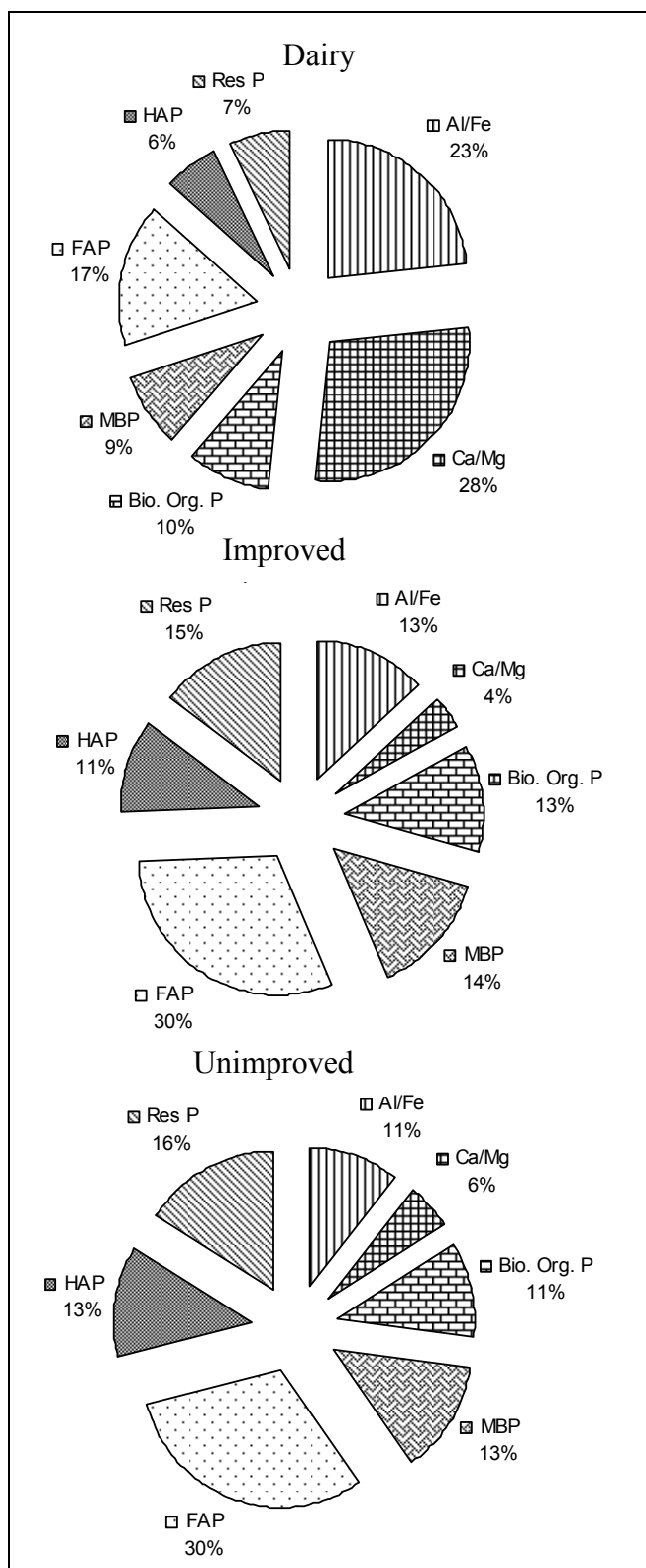


Figure 2. Inorganic and organic phosphorus fractions of surface (0-10 cm) and subsurface soils (10-30 cm) collected from dairy, improved, and unimproved pasture wetland soil. See Table 8 for faction explanations.

Table 8. Correlation matrix of soil characteristics that were measured on the wetland soils collected from dairy, improved and unimproved pastures. Values are Pearson product moment correlations, which indicate a linear relationship between two characteristics.

									1 M HCl		Ammonium oxalate		NaHCO ₃	HCl	Labile	MBP	FAP	HAP
	WC	TPi	LOI	TP	PSI	WEP	TN	TC	Ca	Mg	Fe _{ox}	Al _{ox}	Pi	Pi	org. P			
TPi	0.32																	
LOI	0.95	0.22																
TP	0.59	0.92	0.52															
PSI	0.65	0.34	0.61	0.49														
WEP	0.11	0.79	0.00	0.64	0.07													
TN	0.94	0.33	0.95	0.62	0.58	0.08												
TC	0.94	0.27	0.97	0.54	0.66	0.04	0.97											
Ca _{HCl}	0.31	0.67	0.22	0.70	0.12	0.73	0.35	0.21										
Mg _{HCl}	0.65	0.70	0.57	0.76	0.42	0.65	0.66	0.63	0.74									
Fe _{ox}	0.52	-0.17	0.53	0.08	0.26	-0.41	0.59	0.52	0.12	0.17								
Al _{ox}	0.66	0.47	0.65	0.59	0.84	0.12	0.62	0.72	0.05	0.49	0.13							
NaHCO ₃ Pi	0.04	0.90	-0.06	0.74	0.18	0.90	0.04	-0.02	0.73	0.60	-0.42	0.26						
HClPi	0.45	0.95	0.36	0.93	0.51	0.66	0.44	0.42	0.53	0.72	-0.12	0.66	0.79					
Labile org. P	0.50	0.09	0.52	0.35	0.56	-0.28	0.57	0.49	0.17	0.20	0.63	0.46	-0.09	0.17				
MBP	0.32	0.19	0.17	0.25	-0.02	0.17	0.29	0.19	0.38	0.40	0.29	-0.09	0.08	0.23	0.10			
FAP	0.78	0.56	0.73	0.77	0.67	0.19	0.80	0.75	0.50	0.74	0.46	0.70	0.32	0.68	0.71	0.40		
HAP	0.81	0.55	0.72	0.73	0.74	0.23	0.79	0.78	0.38	0.75	0.40	0.77	0.30	0.70	0.57	0.45	0.90	
Res P	0.62	0.53	0.55	0.68	0.34	0.48	0.62	0.53	0.76	0.78	0.34	0.28	0.44	0.51	0.44	0.52	0.73	0.67

WC = soil water content; LOI = loss on ignition; TP = total phosphorus; PSI = phosphorus sorption capacity; WEP = water extractable P; TN = total nitrogen; TC = total carbon; Ca = calcium; Mg = magnesium; Fe_{ox} = iron extracted with ammonium oxalate; Al_{ox} = aluminum extracted with ammonium oxalate; NaHCO₃ Pi = Fe and Al bound P; HCl Pi = Ca and Ma bound P; Labile org. P = labile organic bound P; MBP = microbial biomass P; FAP = fulvic acid bound P; HAP = humic acid bound P.

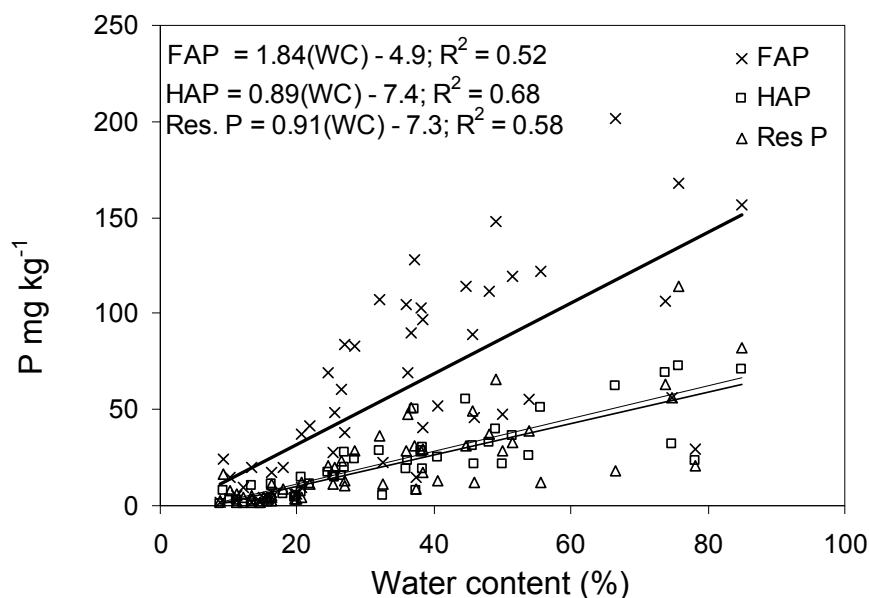


Figure 3. Relationship between soil water content (WC) and organic P fractions. FAP = fulvic acid bound P, which is P that is moderately resistant; HAP = humic acid bound P, which is highly resistant; Res P = residual bound P, which is crystalline or in an organic form that is unavailable.

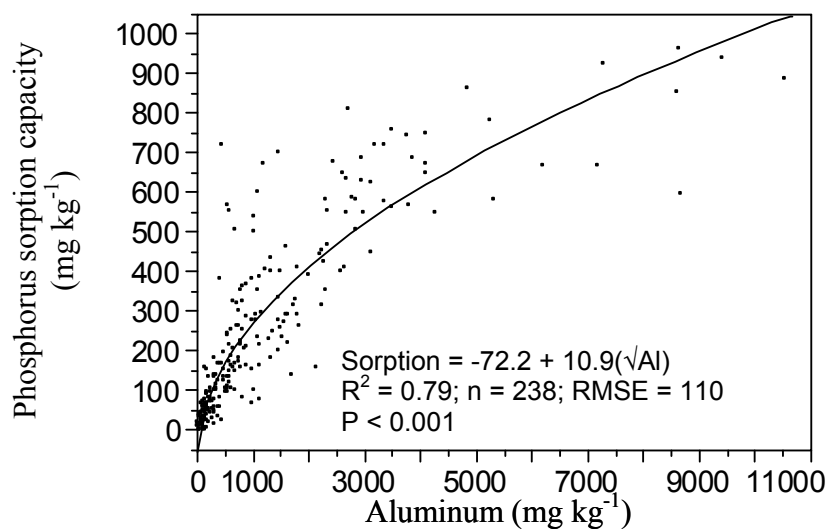


Figure 4. Relationship between wetland soil aluminum content as extracted with 0.2 M ammonium oxalate and phosphorus sorption capacity as determined by incubating soil with a $1000 \text{ mg P kg}^{-1}$ of soil.

Table 9. Wetland soil characteristics within dairy, improved and unimproved land uses.

Based upon the Al content of soils collected in Task 3.1, we predicted the sorption capacity (using relationship generated in Task 3.2; see Figure 4) of wetland soils collected from the 118 wetlands during Task 3.1. This enabled us to determine sorption capacity of wetland soils at a larger landscape-scale.

Characteristic		Land Use					
		Dairy		Improved		Unimproved	
		---n = 18---		---n = 130---		---n = 22--	
Bulk Density g cm ⁻³	Measured	0.596	0.07	0.819	0.03	0.757	0.08
Total P mg kg ⁻¹	Measured	1060	285	385	31	358	62
Total P kg ha ⁻¹	Measured	461	120	246	17	212	36
Al _{ox} mg kg ⁻¹	Measured	703	240	501	41	663	96
PSI mg kg ⁻¹	Predicted	174	37	151	9	189	23
PSI g m ⁻²	Predicted	8	1	11	1	13	2
PSI kg ha ⁻¹	Predicted	83	14	105	5	127	20

Intensive Assessment of Phosphorus Flux

Introduction

The internal supply of nutrients in soils can cause a flux of nutrients from soil to water and this can be in excess of external nutrient loading (Fisher and Reddy, 2001). For example, Fisher and Reddy (2001) suggested wetland soils that were historically loaded with phosphorus (P) had elevated total P concentrations between 990 and 1550 mg P kg⁻¹ and had a maximum P flux of about 6.5 mg P m⁻² d⁻¹. Further, Pant and Reddy (2003) indicate that P flux from soils during a 28 day flooding was greatest from severely P impacted soils. They observed greatest P flux from soils collected from active dairy pastures in the Okeechobee Basin; flux ranged between 30 and 45 mg P m⁻² d⁻¹ compared to forage grazing pasture and unimpacted forested soils, which fluxed less than 10 mg P m⁻² d⁻¹.

An additional task in Task 3.2 was to intensively assess the P flux from wetland and grazing pasture upland soils along a hydrological gradient. Typically, land use data at large spatial scales are used to estimate the loads being lost from different land uses within a watershed. However, the internal cycling of nutrients within different land uses and/or within different landscape units such as wetlands and uplands are often, not accounted for.

Materials and Methods

Site Description

All soils were collected from natural historically isolated emergent marsh wetlands in cow-calf grazing pastures in South Florida. Two wetlands, which were near each other were located on the Beaty Ranch (N 027° 24.665', W 080° 56.940') and two were located on the Larson Dixie Ranch (N 027° 20.966', W 080° 56.465') (Figure 5).

Each wetland was drained by a ditch, which during high water events transported water off site by discharging to the ditch. The Larson Dixie isolated wetlands were about 2 hectares in area, whereas the isolated wetlands at Beaty were slightly smaller and had an average area of 1.3 ha. Vegetative communities at all sites were concentric to the wetland center deep marsh zone.

During field sampling, we used vegetative communities and basin morphology to stratify the wetland and surrounding pasture upland into four zones. These zones were similar to the zones identified by Dunne et al. (2007), which were similar in vegetation community structure to van der Valk (1989; and 2005). We categorized the wetland center as “deep marsh.” These were areas that had open water present during wetland flooding. Vegetation species that typically occurred in and around this zone included *Pontedaria cordata* var. *lancifolia* (Muhl.) Torr., *Bacopa monnieri* (L.) Pennell, *Panicum hemitomon* Schult., *Polygonum* sp., and *Ludwigia repens* Forst. The second zone was a transition zone, which was delineated as “shallow marsh.” There was evidence of flooding, but this zone was often dry during field sampling. The outer extent of this zone was demarcated

by a concentric ring of *Juncus effuses* L. Other species present in this zone were *Eleocharis baldwinii* (Torr) Chapm, *Paspalum acuminatum* Raddi, and *Hydrochloa caroliniensis* Beauv. The third zone was similar to a “wet meadow” zone where *Paspalum acuminatum* Raddi was very common. Also, there were many species of sedges and grasses that were not easily identified, as this zone was heavily grazed by cattle, particularly at the Larson Dixie site. The final zone identified was the surrounding grazing “pasture upland,” which was dominated by *Paspalum notatum* Flugge.

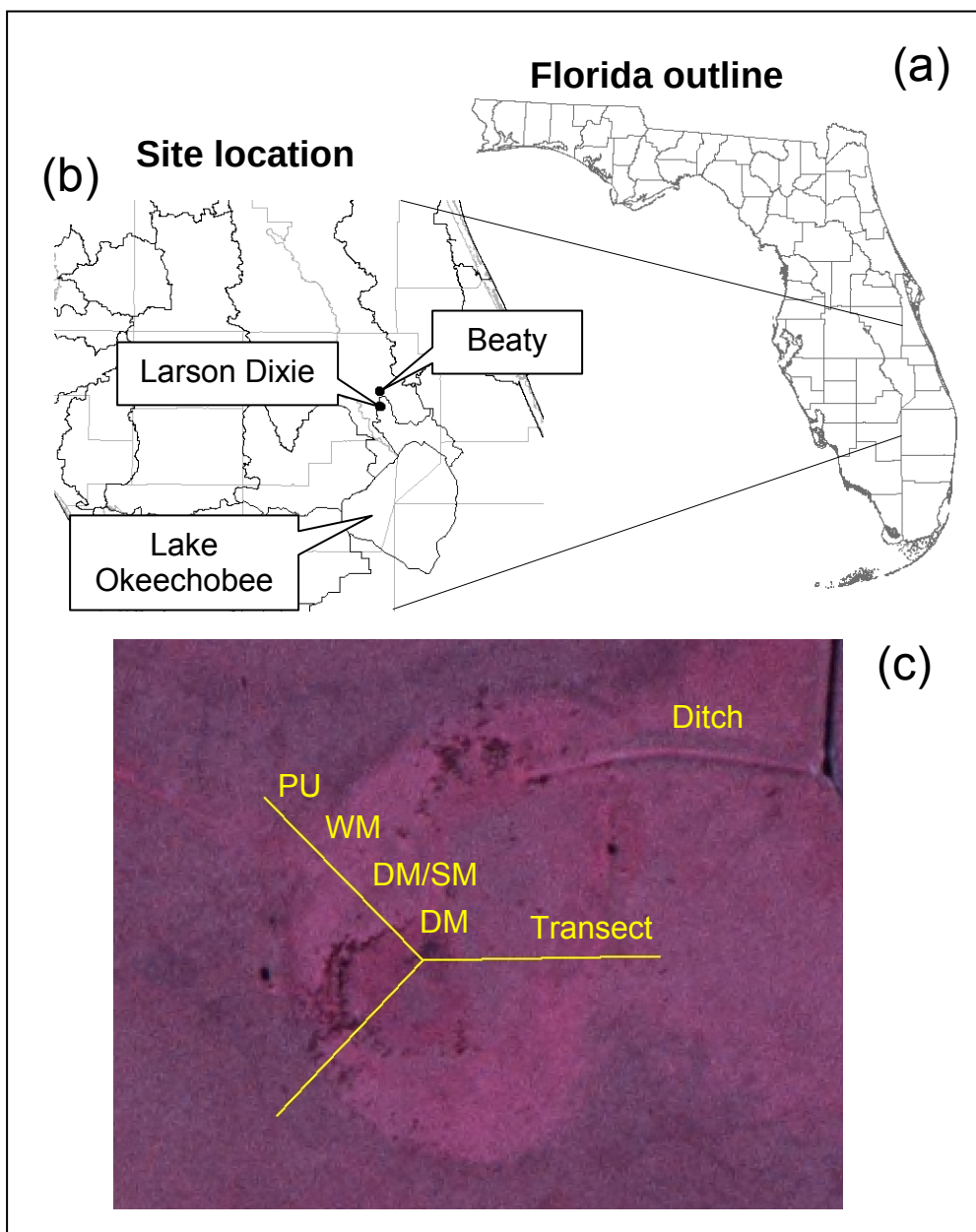


Figure 5. Location of Larson Dixie and Beaty sampling sites (a, and b). (c) Remotely sensed image of wetland at Larson Dixie showing the three sampling

transects. Along the NW-SE transect pasture upland (PU), wet marsh (WM), deep marsh/shallow marsh (DM/SM), and deep marsh (DM) show approximate locations of soil sampling locations. Similar sampling was done on the other two transects.

Lewis et al. (2001) indicated that soils at Larson Dixie Ranch site were classified as Siliceous, hyperthermic Spodic, Psammaquents (Basinger series) and Beaty Ranch site soils were Sandy, siliceous, hyperthermic Typic humaquepts (Placid series). Both soils are deep, poorly drained and formed from sandy marine sediments.

Wetland hydroperiod was determined as outlined by Dunne et al. (2007). Hydroperiod for the different soil sample locations along transects were estimated using elevation recorded at the sampling location, relative to the elevation of the groundwater well located in the wetland deep marsh zone.

Soil Sampling

To characterize site soils, soils were collected in October, 2005. Intact soil cores (0-10 cm) were taken along transects that extended from wetland deep marsh to the surrounding grazed pasture upland. At each site there were three transects extending from the deep marsh center (Figure 5). Each transect was about 100 meters in length. Along each transect, four intact soil cores were taken. Intact soil cores were taken in the deep marsh, shallow marsh, wet meadow, and pasture upland. To take intact soil cores, polycarbonate tubes (10 cm internal diameter x 0.3 cm wall depth x 15 cm in length) were sharpened at one end and hammered down to a soil depth of 15 cm. Tubes with soil in them were then extracted from soil substrate. Soils were extruded from tubes and sectioned at 10 cm depth. After that, soils were placed into a pre-labeled zip lock bag and put on ice in a cooler. Soils were transported back to laboratory, where they were stored at 4°C until being prepared for analyses.

In addition to collecting soils for characterization, intact soil cores (0-10 cm) were also collected for P flux studies. Soils were collected at four different times during 2005 and 2006 (October 7th, 2005; February 2nd, 2006; September 20th, 2006; and December 12th, 2006). During each sampling, intact soil cores (0-10 cm) were collected similarly to the methods already described. However, rather than extruding soils from tubes, the bottom and top of the soil core tubes were capped. During each sampling event, 12 intact soil cores (soil core collected in each zone along each of the three transects) were collected from each wetland. Tubes with intact soil cores were placed on ice, transported back to laboratory and stored at 4°C for one week until laboratory experiments were conducted.

Experimental Core Study

The cap covering the top of each core tube was removed. The cap on the bottom of each core tube was sealed to the core tube with a metal clamp, to prevent air and water

leakage. Each soil core tube was then wrapped in aluminium foil to prevent light shining on soil portion (0-10 cm). Soil core tubes were placed at random into a glass aquarium. Soils in core tubes were initially wetted with site water until soils were surface saturated. Once saturated, soils were flooded to a water depth of 15 cm with Kissimmee River water. To maintain an isothermal temperature and prevent water leakage from core tubes, water was poured into the aquarium until the depth of water in tubes was similar to the depth of water in the aquarium. Temperature loggers iButton® (Dallas Semiconductor Corp., Dallas, Tx) were placed into aquariums to record temperature during the experiments. The overlying floodwater in each tube was sampled on days 0, 1, 2, 5, and 7. Ten mL of floodwater was sampled from each core tube using a syringe and then filtered through a 0.45 µm disc filter. Filtered water was analyzed for soluble reactive P (SRP) using an automated ascorbic acid method (Mehod 365.1; USEPA, 1993).

Phosphorus flux was determined as outlined by Pant Reddy (2003). Briefly, phosphorus flux ($\text{mg m}^{-2} \text{d}^{-1}$) was determined as the linear slope between cumulative P storage in the overlying water (mg m^{-2}) and days.

Physicochemical Characteristics of Soils

The following soil physical and chemical parameters were measured: pH, water content, bulk density, total phosphorus, total nitrogen and total carbon, organic matter (as measured by LOI), inorganic P as extracted with 1 M HCl, and water extractable phosphorus. Soil pH was measured in a 1:2 soil to water ratio. A known mass of wet soil was dried for 72 hours at 70°C and the net percentage difference between wet and dry weights was quantified as the soil water content (Gardner, 1986). Soil bulk density was determined using a simplified coring method similar to Blake and Hartage (1986). Soil total phosphorus content was determined on 0.5 g of finely ground dry soil that was combusted at 550°C in a muffle furnace for four hours. Ash was then dissolved in 6 M HCl (Andersen, 1976) and digestate analyzed for P using the automated ascorbic acid method. Depending on soil textural components, 5-60 mg of dried finely ground soil was analyzed for total carbon (TC) and total nitrogen (TN) by dry combustion using a C-N-S analyzer (Carlo Erba Model NA-1500). To determine inorganic P, soils (0.5 g of finely ground and sieved to 2 mm) were extracted with 25 mL of 1 M HCl. Samples were then centrifuged for 10 minutes at 6000 rpm and filtered through 0.45 µm filter paper using a vacuum filtration system. Soluble reactive P was analyzed using an automated ascorbic acid method as previously mentioned.

Statistical Analyses

Data distributions were tested for normality. If data was not normally distributed prior statistical analyses, it was log transformed to approximate normality. Statistical analyses were conducted on transformed data and significant differences were determined at the $P < 0.05$, 0.01, and 0.001 level. Using t-tests and an analysis of variance (ANOVA), we compared soil physicochemical characteristics and phosphorus flux between sites (Larson Dixie and Beaty), wetland (Beaty North [BN], Beaty South [BS], Larson East [LE] and

Larson West [LW]), zone (deep marsh, shallow marsh, wet meadow and pasture upland) and their interactions. Phosphorus flux was also compared with time of sampling to investigate whether there were any temporal differences in phosphorus fluxes from soils.

Pearson product moment linear correlation coefficients were used to determine whether soil characteristics were related to hydroperiod and phosphorus flux. Further, linear coefficients were also used to investigate whether there was any association between phosphorus flux from the different zonal soils (deep marsh, shallow marsh, wet meadow, and upland pasture), soil physicochemical characteristics, and hydroperiod.

Results and Discussion

Soil Physicochemical Characteristics

In general, Beaty soils had greater water content, and pH and lower bulk density than soils collected from Larson Dixie sites ($P < 0.05$; Table 10) suggesting that these wetland are wetter than Larson Dixie wetlands. Beaty soils also had the greatest total nitrogen content ($P = 0.003$; $n = 40$). When concentrations (mg kg^{-1}) and storages (g m^{-2}) of total phosphorus, 1 M HCl phosphorus, water extractable phosphorus and total carbon were compared between sites (Larson Dixie and Beaty), concentrations and storages were similar between sites. However, Beaty South and Larson West had the greatest and least amounts of soil organic matter, respectively ($P < 0.05$). Similar (but not significant) patterns were observed in soil water content and soil bulk density.

In general, deep marsh soils had lower bulk density, and greater concentrations (mg kg^{-1}) of total nitrogen, total carbon, water extractable phosphorus, and total phosphorus than upland soils ($P < 0.05$; $n = 40$). Although there were no significant differences between zonal nutrient (N, P, and C) storages (g m^{-2}) there was a trend that wetland soils (deep marsh, shallow marsh, and wet meadow) had higher nutrient storages relative to upland soils (Table 10). Deep marsh soils had greater water content and organic matter content than all other zonal soils ($P < 0.05$). Soils collected from the wet meadow zone at Larson Dixie had greatest soil bulk density; whereas, soils collected from Beaty shallow marsh zones had the lowest (Table 10; $P < 0.05$). Beaty South shallow marsh soils had the greatest soil organic matter content; whereas Larson East wet meadow soils had least amounts of organic matter ($P < 0.05$; Table 10).

Phosphorus Flux from Wetland and Upland Soils

During phosphorus flux studies, it was observed that flux was greatest from Larson Dixie soils ($P < 0.001$; $n = 190$; Figure 7; Table 11). Soils collected from Larson West fluxed greatest amounts of phosphorus ($P < 0.05$; $n = 46$) again illustrating that this site was the most P impacted relative to the other three sites. Larson East and Beaty South soils fluxed similar amounts of phosphorus. In general, soils collected from the different zones fluxed similar amounts of phosphorus (Figures 6 and 7). However, at Larson West there was a trend suggesting that pasture uplands soils fluxed higher amounts of phosphorus. The trend of drier soils such as uplands soils fluxing higher amounts of P

relative to wetland soils is important from a management perspective. With hydrologically restoring wetland areas, i.e. increasing wetland footprints in the landscape, upland soils adjacent to wetlands will become flooded or at least wetter than they are at present. This may cause a loss of P from soil to water.

Table 10. Soil physicochemical characteristics of the different wetland zones and surrounding pasture uplands at Larson Dixie ranch (LW = Larson West; LE = Larson East) and Beaty ranch (BN = Beaty North; BS = Beaty South). Values represent a mean value $\pm$ one standard error.

Site	Zone	pH	Water content	Organic Matter	Bulk Density	Total N	Total C	Total P	1 M HCl P	Water Ext. P
			-----%-----		-----g cm ⁻³ -----		-----g m ⁻² -----			
BN	Deep marsh	4.55 $\pm$ 0.8	80.8 $\pm$ 3	38.8 $\pm$ 9	0.344 $\pm$ 0.03	378 $\pm$ 38	6239 $\pm$ 761	18.50 $\pm$ 2.7	2.48 $\pm$ 0.7	0.042 $\pm$ 0.00
BN	Shallow marsh	4.83 $\pm$ 0.9	56.8 $\pm$ 9	16.9 $\pm$ 5	0.840 $\pm$ 0.21	437 $\pm$ 37	5818 $\pm$ 546	21.43 $\pm$ 2.0	3.30 $\pm$ 0.4	0.154 $\pm$ 0.07
BN	Wet meadow	4.70 -	50.4 -	19.5 -	0.646 -	370 $\pm$	6336 -	26.11 -	4.43 -	0.010 -
BN	Pasture upland	5.13 $\pm$ 1.0	35.0 $\pm$ 3	12.6 $\pm$ 0	0.650 $\pm$ 0.15	241 $\pm$ 64	3703 $\pm$ 863	11.18 $\pm$ 2.4	1.78 $\pm$ 0.7	0.041 $\pm$ 0.03
BS	Deep marsh	4.64 $\pm$ 0.7	73.8 $\pm$ 3	31.5 $\pm$ 6	0.329 $\pm$ 0.04	393 $\pm$ 130	6085 $\pm$ 2111	13.68 $\pm$ 3.0	2.68 $\pm$ 0.6	0.139 $\pm$ 0.03
BS	Shallow marsh	4.94 $\pm$ 0.7	80.2 $\pm$ 0	63.2 $\pm$ 17	0.278 $\pm$ 0.02	517 $\pm$ 108	7861 $\pm$ 1914	23.15 $\pm$ 5.2	4.03 $\pm$ 1.0	0.068 $\pm$ 0.02
BS	Wet meadow	4.83 $\pm$ 1.2	57.1 $\pm$ 13	26.5 $\pm$ 5	0.649 $\pm$ 0.43	453 $\pm$ 219	6583 $\pm$ 3110	20.19 $\pm$ 10.1	3.67 $\pm$ 2.0	0.083 $\pm$ 0.05
BS	Pasture upland	5.05 $\pm$ 0.2	32.7 $\pm$ 5	13.1 $\pm$ 4	0.866 $\pm$ 0.08	371 $\pm$ 19	5090 $\pm$ 871	18.11 $\pm$ 5.0	2.77 $\pm$ 0.6	0.082 $\pm$ 0.05
LE	Deep marsh	4.05 $\pm$ 0.0	57.6 $\pm$ 2	40.7 $\pm$ 8	0.560 $\pm$ 0.08	686 $\pm$ 35	8751 $\pm$ 575	24.90 $\pm$ 3.3	4.19 $\pm$ 0.1	0.081 $\pm$ 0.02
LE	Shallow marsh	4.40 $\pm$ 0.2	42.4 $\pm$ 4	21.4 $\pm$ 2	0.786 $\pm$ 0.10	626 $\pm$ 78	7823 $\pm$ 1186	27.58 $\pm$ 3.8	4.33 $\pm$ 0.4	0.093 $\pm$ 0.00
LE	Wet meadow	4.78 $\pm$ 0.4	26.9 $\pm$ 3	10.8 $\pm$ 2	0.945 $\pm$ 0.09	406 $\pm$ 42	4406 $\pm$ 426	14.58 $\pm$ 2.3	2.19 $\pm$ 0.6	0.096 $\pm$ 0.03
LE	Pasture upland	4.97 $\pm$ 0.7	32.9 $\pm$ 7	17.1 $\pm$ 6	0.803 $\pm$ 0.10	471 $\pm$ 79	5591 $\pm$ 1483	26.34 $\pm$ 9.2	5.85 $\pm$ 2.2	0.146 $\pm$ 0.10
LW	Deep marsh	4.43 $\pm$ 0.4	45.0 $\pm$ 6	21.0 $\pm$ 3	0.759 $\pm$ 0.06	637 $\pm$ 68	8148 $\pm$ 963	32.52 $\pm$ 3.1	4.53 $\pm$ 0.7	0.145 $\pm$ 0.02
LW	Shallow marsh	4.55 $\pm$ 0.5	26.3 $\pm$ 6	13.6 $\pm$ 4	0.961 $\pm$ 0.06	552 $\pm$ 101	6044 $\pm$ 1582	20.44 $\pm$ 5.6	2.90 $\pm$ 1.0	0.254 $\pm$ 0.08
LW	Wet meadow	4.21 $\pm$ 0.2	33.4 $\pm$ 11	14.8 $\pm$ 6	1.013 $\pm$ 0.02	618 $\pm$ 221	6926 $\pm$ 3252	25.59 $\pm$ 11.1	5.60 $\pm$ 1.4	0.243 $\pm$ 0.17
LW	Pasture upland	3.59 $\pm$ 0.1	42.4 $\pm$ 2	20.1 $\pm$ 3	0.666 $\pm$ 0.01	442 $\pm$ 57	6387 $\pm$ 923	19.19 $\pm$ 1.5	3.20 $\pm$ 0.2	0.032 $\pm$ 0.02

Table 11. Phosphorus flux from wetland and pasture upland soils collected from Beaty North (BN), Beaty South (BS), Larson East (LE) and Larson West (LW).

Core study	Site	-----Deep marsh-----				-----Shallow marsh-----				-----Wet meadow-----				-----Pasture upland-----			
		---mg m ⁻² d ⁻¹ ---		Goodness of fit		---mg m ⁻² d ⁻¹ ---		Goodness of fit		---mg m ⁻² d ⁻¹ ---		Goodness of fit		---mg m ⁻² d ⁻¹ ---		Goodness of fit	
				R ²				R ²				R ²				R ²	
October 2005	BN	-0.94	± 0.8	0.869		-1.47	± 0.7	0.867		-1.98	± 0.4	0.880		0.84	± 4.1	0.886	
	BS	0.62	± 0.6	0.664		3.88	± 2.2	0.904		3.53	± 2.1	0.951		0.68	± 4.7	0.388	
	LE	3.12	± 4.4	0.906		18.71	± 16.8	0.964		3.09	± 5.0	0.466		-6.33	± 30.5	0.712	
	LW	3.30	± 2.8	0.855		11.60	± 2.3	0.887		18.25	± 5.1	0.850		15.03	± 7.8	0.767	
February 2006	BN	-1.74	± 2.6	0.370		0.50	± 1.0	0.638		-0.52	± 0.3	0.545		1.50	± 3.5	0.504	
	BS	7.17	± 1.4	0.932		6.94	± 2.9	0.995		5.06	± 2.9	0.969		2.41	± 1.6	0.688	
	LE	4.40	± 3.2	0.963		21.03	± 8.9	0.941		5.63	± 2.6	0.736		16.36	± 13.3	0.927	
	LW	11.39	± 8.5	0.928		13.24	± 4.9	0.993		6.99	± 3.9	0.848		11.41	± 5.8	0.877	
July 2006	BN	2.58	± 2.4	0.753		1.80	± 3.0	0.740		0.53	± 11.9	0.991		-4.33	± 13.0	0.861	
	BS	11.37	± 10.5	0.675		14.11	± 4.1	0.988		10.93	± 7.2	0.934		6.30	± 4.9	0.788	
	LE	9.14	± 8.8	0.863		22.71	± 7.2	0.984		10.86	± 17.1	0.951		48.55	± 41.0	0.907	
	LW	9.76	± 2.5	0.857		18.38	± 9.5	0.952		30.13	± 16.2	0.930		36.09	± 22.9	0.938	
December 2006	BN	13.18	± 14.5	0.918		6.89	± 7.5	0.916		0.56	± 3.8	0.622		9.91	± 1.4	0.882	
	BS	11.55	± 22.5	0.742		5.48	± 10.4	0.747		5.67	± 4.3	0.672		2.92	± 5.8	0.329	
	LE	18.18	± 5.1	0.965		12.54	± 5.3	0.857		12.07	± 1.3	0.995		14.13	± 4.9	0.984	
	LW	15.93	± 6.5	0.994		17.08	± 3.1	0.989		15.87	± 2.5	0.971		19.82	± 3.8	0.966	

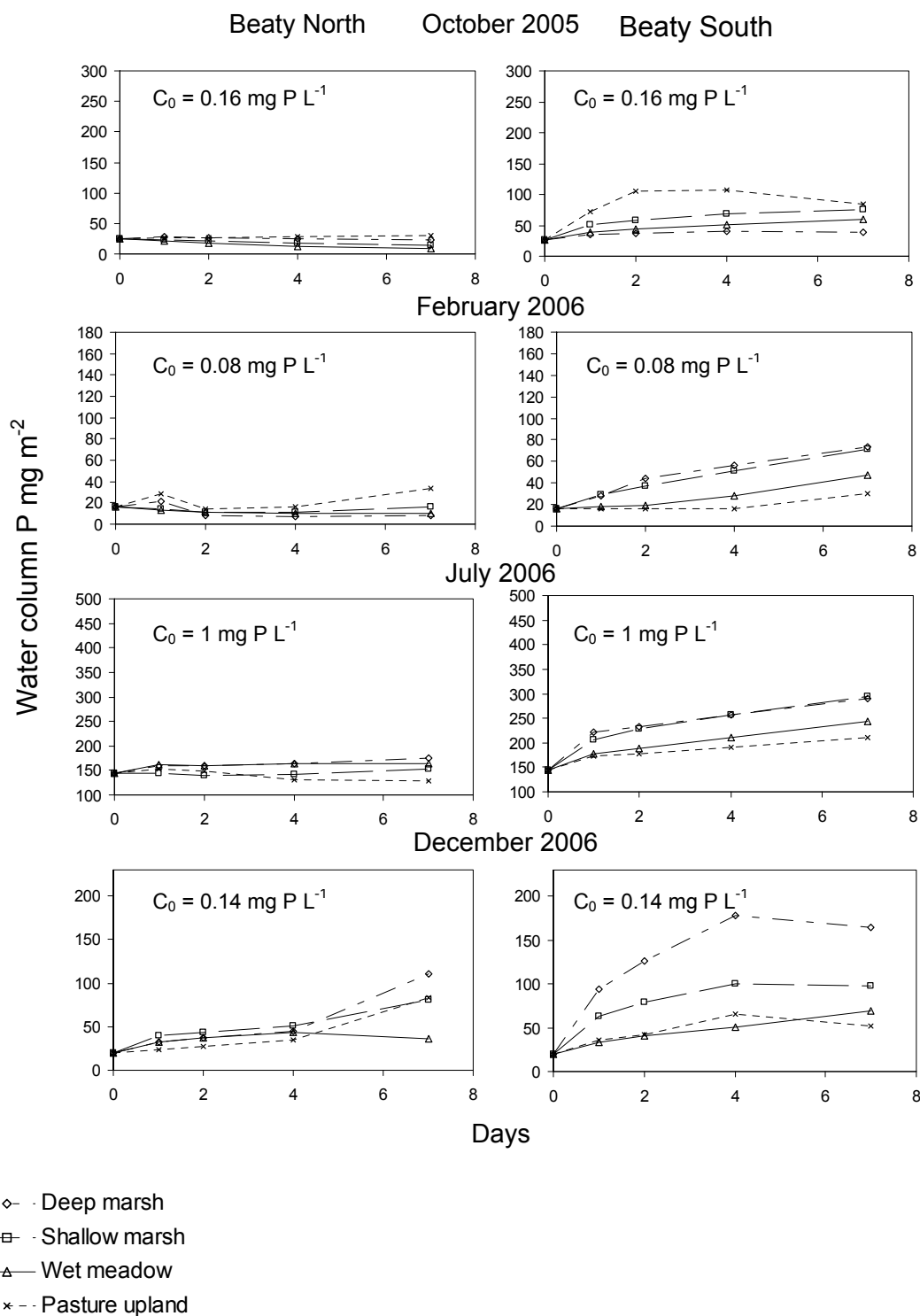


Figure 6. Phosphorus flux from soil cores collected from Beaty North and Beaty South. Flux studies were undertaken in October 2005, March 2006, July 2006 and December 2006. Soils were flooded with Kissimmee River water for seven days.

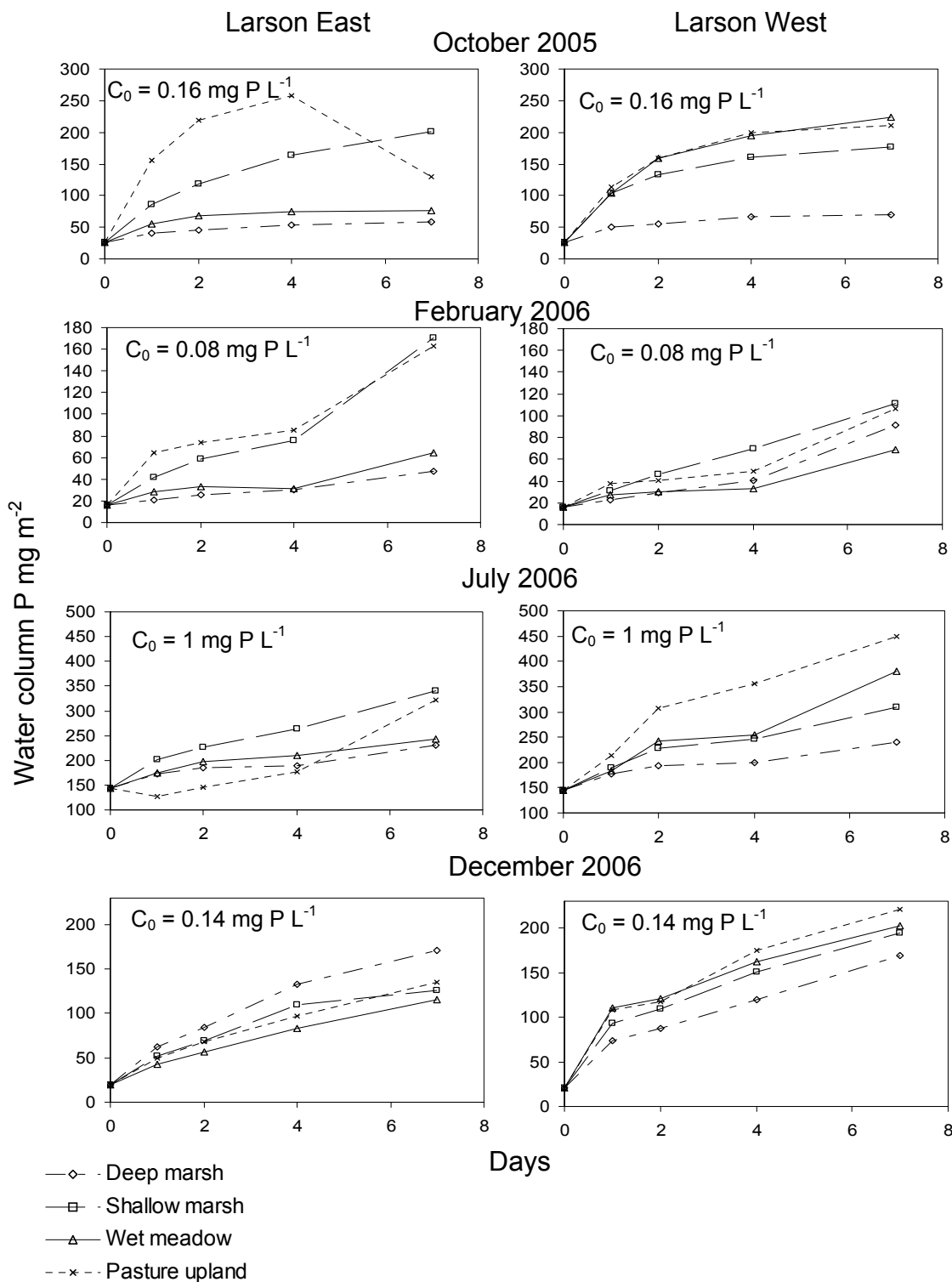


Figure 7. Phosphorus flux from soil cores collected from Larson East and Larson West. Flux studies were undertaken in October 2005, March 2006, July 2006 and December 2006. Soils were flooded with Kissimmee River water for seven days.

There was temporal variability in phosphorus flux. During October 2005, soils collected from Beaty sites fluxed less phosphorus than soils collected from Larson ($P < 0.001$; $n = 46$; Table 11). In fact, during October 2005 and February 2006 Beaty North soils (deep marsh, shallow marsh, and wet meadow) tended to retain phosphorus, whereas the uplands soils tended to flux phosphorus to overlying water (Figure 6; Table 11). Phosphorus flux rates from all sites and zones were greatest during July and December 2006 ($P < 0.01$; $n = 190$; Figures 6 and 7; Table 11). This may be because all wetland sites were dry (no open water in wetlands) since April 2006 suggesting that antecedent hydrological conditions may play an important role in P dynamics (Dunne et al., 2006). Phosphorus flux rates during the four flux studies were independent of the initial phosphorus concentration in overlying water, which indicated that other factors may be controlling this initial flux during a seven day period. Factors include antecedent hydrological conditions, temperature, soil porewater P content, the P gradient between soil porewater P and overlying wetland water, P bound to organic or soil mineral portions.

Relationships between Soil Physicochemical Characteristics, Hydroperiod and Flux

Phosphorus flux was weakly (but significant [$P < 0.05$]) associated with hydroperiod and soil water content, suggesting that as hydroperiod and water content decreased, phosphorus flux from soils that were flooded for seven days increased (Table 12). This implies that as one increases wetland hydroperiod the loss of P from soil to water decreases, which is cohesive to restoring wetland soils to store and retain water and P.

Linear correlations suggest that during the seven days of flooding, phosphorus fluxed most from soils that were initially the driest (low water content; pasture upland soils; Table 10) and fluxed least from deep marsh soils, which had the greatest water content (Table 10) prior flooding ($P < 0.01$; $n = 38$; Table 12). Again this is important for water quality management because it implies that the drier uplands soils if flooded for periods of seven days can flux most P. An opposite significant trend was observed for soil bulk density suggesting that as soil bulk density increased from wetland soil to upland soil, phosphorus flux during the seven day flooding also increased.

Table 12. Linear correlation coefficients (Pearson-product moment) between phosphorus flux and the different soil physicochemical characteristics of wetland zones and surrounding upland pasture soils. Correlations are determined on the relationships between the initial soil characteristics measured and phosphorus flux during the first phosphorus flux study in October 2005. (n = 45).

Soil physicochemical characteristics	Zone			
	Deep marsh	Shallow marsh	Wet meadow	Pasture upland
	-----Phosphorus flux-----			
Hydroperiod (Days)	0.490	-0.227	-0.473	-0.573
pH	-0.262	-0.161	-0.033	0.035
Water content (%)	0.185	-0.402	-0.591	-0.644*
Bulk density (g cm ⁻³)	-0.023	0.413	0.323	0.666*
Organic matter (%)	0.401	-0.380	-0.231	-0.086
Water ext. P (g m ⁻²)	-0.089	0.526	0.333	0.450
1 M HCl P (g m ⁻²)	0.568	0.396	-0.042	0.622
Total P (g m ⁻²)	0.376	0.077	-0.158	0.420
Total nitrogen (g m ⁻²)	0.644*	0.263	0.300	0.490
Total carbon (g m ⁻²)	0.486	-0.112	-0.057	0.306

* = significance at the $P < 0.05$ level.

Water ext. P = Water extractable phosphorus.

Total P = Total phosphorus.

3.3 Site characterization, instrumentation, and baseline water quality, soils, and vegetation monitoring

3.3.1 Wetland-Groundwater Interactions

Introduction

There are several important reasons to understand and quantify the frequency, magnitude, and duration of wetland surface water and local groundwater interaction. The first is to determine how wetland ecosystems may respond to managed groundwater drawdown. Currently, no protection policy exists for wetlands whose ecosystem functions are unfavorably affected by local or regional groundwater management. Second, wetland wetting and drying periods are related to exchange of atmospheric gases, some of which have implications for global climate change (Romanowicz et al., 1994). Finally, wetland-groundwater interaction is an important component of wetland water retention and chemical treatment potential in managed (e.g. treatment wetlands) and natural systems. It is generally expected that wetlands act as a chemical sink in a landscape because (1) they are often located in relatively low topographic positions where landscape water may collect and (2) they often develop physical, chemical, and biological mechanisms for storing and transforming nutrients and other chemicals (Dortch, 1996; Reddy et al., 1999; Price and Waddington, 2000).

The occurrence and magnitude of surface water-groundwater exchange in these wetlands has not been reported in the literature. Understanding the relationship between wetland surface water and groundwater is paramount in defining the role and function of these wetlands from an ecosystem and water quality perspective. This research will focus on characterization, quantification, and prediction of wetland-groundwater exchange in this region. Results from this study have implications for nutrient loads to Lake Okeechobee via groundwater.

Several authors have investigated characterization and quantification of wetland-groundwater exchange in other types of ecosystems (Devito et al., 1997; Hayashi et al., 1998a,b; Sun et al., 1998; Wise et al., 2000; and Parsons et al. 2003). Devito et al. (1997) devised several field experiments to determine the absence or presence of groundwater exchange with prairie pothole wetlands or depression-focused recharge. From a water balance perspective, Hayashi et al. (1998a) quantified wetland-groundwater exchange of northern prairies to understand ecosystem-scale implications of salinization in pothole wetlands. The authors concluded that groundwater outflow was significant, accounting for 75 % of water leaving the study system. Furthermore, through chloride and bromide field tracer experiments (Hayashi et al. 1998b; Parsons et al. 2004), this groundwater was found to flow laterally into the upland where it was assumed to be transpired by plants. The geology of the prairie pothole region (stratified silty sediments) is quite dissimilar from the fractured limestone overlain by coarse sandy sediments found throughout the Florida study sites in this research. These investigators assessed groundwater exchange with surface water by interpreting averaged hydraulic head gradients and soil saturated hydraulic conductivity estimates from slug tests in a Darcy's Law format. However,

these authors focused on quantifying the total groundwater exchange relative to the other water budget components.

Motz et al. (1998) estimated vertical leakage from 11 karst lakes to a deep semi-confined aquifer system (Floridan aquifer) in north-central Florida. In addition to deep aquifer recharge estimates, these investigators also used a water budget approach to estimate a vertical conductance, K_v/b [T^{-1}], which is a measure of media conductance (vertical saturated hydraulic conductivity, K_v [LT^{-1}]) and depth to aquifer, b [L] (approximately 40 m), for each lake. These vertical conductance values are the inverse of the hydraulic resistivity, R [T], as determined from the “residual” groundwater term in the water budget. Estimates of R from these 11 lakes ranged from 365 to 9264 days. Reported estimates of hydraulic resistivity from two additional lakes in north-central Florida also fall in this range, with a mean R of 3710 days (Motz et al., 2001; Watson et al., 2001).

This current research also used a water budget approach to estimate R and groundwater recharge. However, the Lake Okeechobee isolated wetlands are quite different from the lakes studied by Motz (1998), Motz et al. (2001) and Watson et al. (2001), as they are seasonally inundated and primarily interact with local groundwater, rather than a deep confined aquifer. Additionally, the scale of the wetlands in this research (approximately 100-m diameter) was much different than the lakes described above (1 to 3 km diameter).

Hydraulic resistivity was determined from a best fit to the observed groundwater outflow, as calculated from a constrained water budget. Literature values of R for wetlands of comparable scale to those of this study have only been reported by Wise et al. (2000). These investigators developed a Wetland-Aquifer Interaction Test (WAIT) method to investigate hydrologic exchange between wetland surface water and groundwater and reported on a 14-day field trial in a 70-m diameter wetland. The WAIT method involves actively pumping surface water out of a wetland and then fitting a model to the observed water level in the recovery phase. The authors reported good correlation between measured hydraulic soil properties and wetland R . In the present study, water was not actively pumped out of the wetlands, rather natural drawdown periods (typical durations of 14 to 22 days) were observed where the wetland water level naturally declined under the combined influence of evapotranspiration and groundwater recharge. These drawdown periods were considered passive WAITs. Compared to an active pump test, the methodology in this study may be advantageous because it represents a more passive, and perhaps more natural means of quantifying wetland surface water export through groundwater.

Based on the findings of these previous studies, it was expected that flow between wetland surface water and local groundwater in historically-isolated wetlands in the Okeechobee basin could be described in terms of hydraulic resistivity. In this task, approximately 1000 days of daily water budget data were used to estimate groundwater-wetland exchange with the following specific objectives:

1. Evaluate wetland-groundwater exchange using field data in a water budget framework.

2. Passively measure R values for comparison to those measured both actively and passively in other systems.
3. Evaluate the predictive capability of Darcy's Law using daily site-specific data.

The results from this study have implications for water quality monitoring and management strategies aimed to reduce solute export (especially P) from the Lake Okeechobee, FL catchment area to Lake Okeechobee.

Methods and Materials

Table 13 shows wetland areas and some site specific information. Wetland surface water elevation was monitored for three years at the four study wetlands. Internally-logging pressure transducers (Mini-troll STP, In-situ, Inc.) were deployed in 0.032-m diameter fully-screened PVC wells in the deepest location of each wetland. These pressure transducers exhibited 0.01% accuracy over a 0-34 kPa range. Wetland monitoring well installation depth was between 2.5 and 3.0 m bgs with an additional 1.5 m of well screen above the wetland ground surface. The above-ground portions of the monitoring wells were housed inside a larger diameter PVC casing (0.051-m diameter) to provide structural protection against interactions with cattle. Wetland water surface elevations were recorded in half-hourly intervals and were subsequently averaged on daily time intervals. The monitoring records for wetland water surface elevations were approximately 1000 days for each wetland (Table 14).

Upland water level monitoring wells were similarly constructed, with screened intervals ranging between 1.5 and 2 m bgs. Seven upland wells were equipped with data-logging transducers such that at least one upland well was paired with each wetland well. Data logging in upland water level monitoring wells was synchronized with wetland surface water monitoring data logging and subsequently averaged over the same 24-hour periods. The monitoring record duration for the seven upland wells ranged from 77 to 433 days (Table 15).

Table 13. Wetland area, elevation difference between the lowest point in the wetland to the highest point in the ditch channel (relative ditch elevation), and the topographic range of the four study wetlands.

Wetland ID	Footprint area (ha)	Relative ditch elevation (m)	Topographic wetland range (m)
LW1	2.6	0.30	0.75
LW2	2.1	0.30	0.80
BW1	1.8	0.50	0.76
BW2	1.6	0.40	0.74

Table 14. Monitoring periods of wetland water surface and upland water table elevations for Dixie-Larson wetlands LW1, which was the westerly wetland and LW2, which was the easterly wetland, and Pete Beaty wetlands (BW1 [North] and BW2 [South]).

Location	Monitoring Period	Net Number days
-----Wetland surface water wells-----		
LW1	7/2/2003 to 3/10/2006	962
LW2	12/18/2003 to 3/10/2006	814
BW1	7/2/2003 to 04/29/2006	1011
BW2	7/2/2003 to 04/29/2006	1033
-----Upland water level monitoring wells-----		
LW1 MW7	3/09/2004 to 3/10/2006	234
LW2 MW3	08/07/2004 to 3/10/2006	206
LW2 MW4	3/09/2004 to 3/10/2006	222
BW1 MW2	7/03/2003 to 3/10/2006	433
BW1 MW5	4/27/2005 to 11/12/2005	77
BW2 MW2	3/09/2004 to 4/04/2005	145
BW2 MW3	4/05/2005 to 3/10/2006	110

Isolated wetlands in this region are generally less than 150 m in diameter and exhibit an oval to circular geometry. It has been thought that these wetlands have been formed from sink-hole geologic processes, with a typical depth of one meter. The topography of the four study wetlands was mapped using a line-of-sight laser level in combination with a hand-held Garmin GPS. Surface maps were created from these data using Ordinary Kriging interpolation. Stage-volume and stage-area relationships were determined using these topographic maps. In addition, the elevation difference between lowest point in the wetland and the maximum ditch bottom elevation was determined. These parameters and relationships were important for estimating the water budget at the study sites.

Water Budget-Based Groundwater Outflow

Elevations of wetland surface water, H^{wet} [L], and upland water level, H^{up} [L], were translated to the same datum using the bathymetric survey to determine hydraulic gradients. These data indicated that gradients were negative (groundwater flow away from the wetland) 53% of the monitoring duration, and positive (flow towards the wetland) only 8% of the time. Because of the relatively high percentage of days corresponding to outflow from wetlands to groundwater, the emphasis here is on groundwater outflow from the wetlands.

Observed groundwater flow was calculated on a daily time step using a water budget approach, bounded by the wetland surface water. To focus on only groundwater outflow,

the daily water budget was calculated to only include days when the following three conditions were met:

1. wetland-groundwater hydraulic gradient diverged from the wetland
2. wetland surface water elevation was between the ditch outflow elevation and the wetland bottom (i.e., days with surface water outflow from the wetland were excluded)
3. no rainfall occurred.

Thus, the wetland water budget for these days included only flow out to groundwater and evapotranspiration, ET [LT^{-1}]. In this study, the Penman-Monteith methodology was used to calculate ET (Allen et al., 1998). The groundwater outflow, GW [LT^{-1}], was thus determined from the water budget as:

$$GW = -\Delta H^{\text{wet}} - ET \quad (\text{Equation 1})$$

where ΔH^{wet} [LT^{-1}] is the daily change in wetland water surface elevation. This constrained water budget provides a record of passively monitored interaction of wetland aquifer dynamics.

The number of days used to estimate surface water outflow to groundwater is shown in Table 15 for each paired upland-wetland monitoring record. An example set of paired wetland-upland well hydrographs are shown in Figure 8.

Table 15. Best-fit hydraulic resistivity determined the total monitoring record of each wetland-well pair, R^{total} , and corresponding RMSE.

Wetland-upland well pair	R^{total} (days)	RMSE (cm d^{-1})	Bias (cm d^{-1})	Days
LW1 – LW1MW7	62.8	0.75	-0.15	181
LW2 – LW2MW4	72.6	0.77	-0.04	176
LW2 – LW2MW3	61.2	0.73	-0.10	94
BW1 – BW1MW2	58.9	0.61	-0.02	349
BW1 – BW1MW5	54.5	0.70	-0.14	54
BW2 – BW2MW2	48.5	0.78	-0.12	111
BW2 – BW2MW3	28.9	0.88	-0.10	84

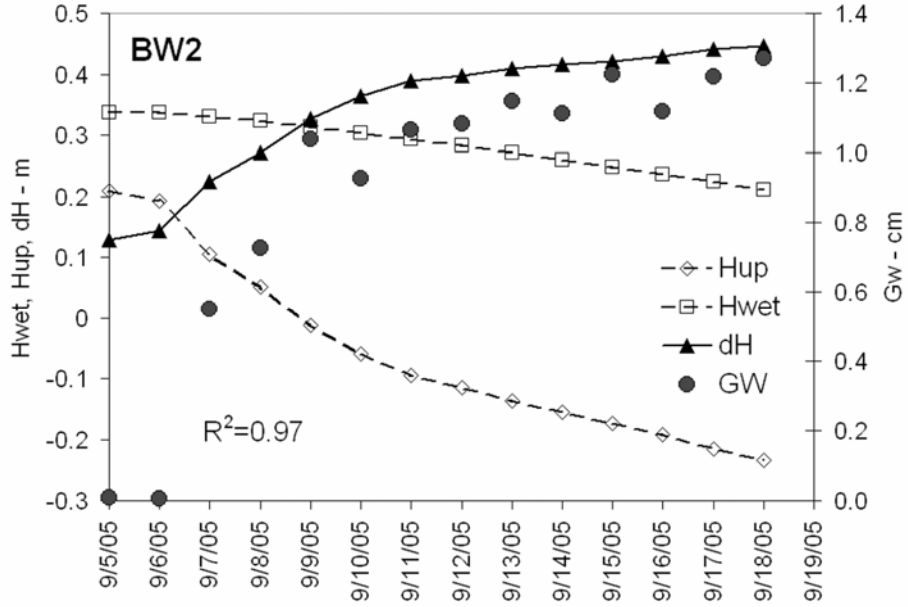


Figure 8. Typical drawdown event used to evaluate correlation between GW and hydraulic gradient between groundwater and wetland (dH). For these data from BW2-MW2, GW and dH are highly correlated ($r = 0.98$).

Groundwater Outflow Model

The wetland-groundwater exchange can also be expressed using a form of Darcy's law:

$$GW = -\frac{1}{R}(dH) \quad (\text{Equation 2})$$

where dH is the wetland-groundwater hydraulic head difference. The estimated GW from Equation 1 and dH were compared on a daily time step for each wetland-well pair to determine R by linear regression. Hydraulic resistivity does not have a true physical meaning, but might be related to a dynamic zone or annulus in the ground around the wetland through which water is moving. Two temporal scales were used to determine hydraulic resistivity: an event based approach (R^{event}), and an approach using the total monitoring record of each wetland-well pair (R^{total}). Drawdown events were defined as periods when the following conditions were met:

- Water in the wetland
- Absence of surface water outflow from the wetland
- Insignificant rainfall.

During these passive drawdown events the groundwater and surface water elevations manifested decreasing trends over the entire length of the subset record (between 14 and 22 days, Table 16). The beginning of a drawdown event was defined when dH was near zero (usually within 2 days of significant rainfall), while the end was defined when a significant uniform drop in surface water occurred (more than 5 cm) across all wetland-

well pairs. Seven to nine events were recorded for each wetland during the observation period (Table 16).

Table 16. Summary of event-based regression between observed groundwater outflow from the four study wetlands and the wetland-groundwater hydraulic gradient. Mean values for each wetland include hydraulic resistivity, R^{event} , correlation coefficient, r , and coefficient of variation, CV.

Wetland	Mean R^{event} (days)	CV	Mean r	Number events	Mean days/event
LW1	42	0.49	0.71	7	14
LW2	72	0.33	0.73	9	22
BW1	54	0.32	0.74	9	20
BW2	40	0.45	0.83	7	17

The R^{event} values were used to identify spatial and/or temporal trends (i.e. season and initial wetland water stage). However, R^{total} values were considered as the best estimate of hydraulic resistivity for each wetland-well pair. For each wetland-well pair, data were sectioned into multiple calibration and validation sets to determine model prediction capability as a function of calibration record length. Root mean square error and bias were the objective functions used to evaluate the prediction performance of groundwater outflow during the validation period.

Meteorological Data

Average daily wind speed, daily air temperature limits, net solar radiation, and average relative humidity were recorded from continuously logging weather station instruments (Campbell Scientific, Inc.) and served as input parameters in the calculation of evapotranspiration using the Penman-Monteith methodology (Allen et al., 1998). It is possible that this methodology of estimating ET might introduce a bias into the water budget; however, the magnitude of ET values fell in a range typical of other studies in the south Florida area. Because of periodic instrument malfunctions, approximately 50% of the input values originated from onsite instrumentation, while the remaining input data were recorded from the South Florida Water Management District (SFWMD) weather station L001 located approximately 14 km south of the study sites (SFWMD, 2007). Data between on-site and SFWMD inputs were highly correlated, with correlation coefficients for average air temperature, wind speed, solar radiation, and relative humidity 0.85, 0.70, 0.72, and 0.81 respectively (N=202 days). A global sensitivity analysis was conducted to determine the error between on-site and SFWMD measured meteorological input parameters for estimating daily ET values. Using the one-at-a-time method (Monod et al., 2006), less than 1% error in ET was observed for any given meteorological input parameter.

Slug tests

Slug-out tests were conducted in seven monitoring wells at the Dixie-Larson wetlands: six in upland monitoring wells and one in the monitoring well at the lowest bathymetric point of LW2. Slug test data were analyzed to find saturated hydraulic conductivity

(K^{slug}) using the Hvorslev method (Fetter, 1994). The value of K^{slug} in the wetland center was significantly different from those in upland soils (0.67 m d^{-1} in the wetland and a mean value of 1.36 m d^{-1} in the upland soils).

Results

Groundwater Outflow Rates

Groundwater outflow from each wetland was determined using Equation 1. Mean and standard deviation for all observed drawdown events for each wetland are reported in Table 17. Mean GW values were not considerably different between wetlands or ranches. The mean GW value for all wetlands of $0.68 \pm 0.25 \text{ cm d}^{-1}$ was slightly higher than the recharge rates of $0.44 \pm 0.41 \text{ cm d}^{-1}$ at 11 karst lakes in central Florida reported by Motz (1998). The daily GW values were multiplied by hydroperiod to determine annual recharge rates of 1.2 to 2.2 m yr^{-1} from these isolated wetlands.

Table 17. Wetland hydroperiod (days), mean $\pm$ standard deviation of groundwater recharge rates (cm d^{-1}), and mean $\pm$ standard deviation of annual groundwater recharge rates (m yr^{-1}) based on hydroperiod.

Wetland ID	Hydroperiod (days)	Groundwater outflow (cm d^{-1})	Groundwater outflow per hydroperiod (m yr^{-1})
LW1	256	0.7 ± 0.2	1.8 ± 0.4
LW2	219	0.5 ± 0.2	1.2 ± 0.4
BW1	302	0.7 ± 0.4	2.2 ± 0.8
BW2	233	0.8 ± 0.2	1.9 ± 0.4

Hydraulic Resistivity

Using Equation 2, the drawdown events showed linear dependence between dH and GW with good correlation (Table 16). The average R^{event} values from each wetland-well pair were between 42 and 72 days (Table 16). Some variability was found in R^{event} (CV between 0.32 and 0.49), but this parameter did not exhibit trends with initial wetland water elevation (i.e. changing spatial scale of wetland hydraulic resistivity), or season (hydraulic changes due to plants, cattle traffic, etc.). This variability might be related to non-equilibrium of water flow, where R is sampled more frequently than it may be observed as steady-state.

Values for R^{total} for each wetland-well pair and the corresponding root mean square error (RMSE) are reported in Table 18. Observed GW values are compared in Figure 9 to calculated values determined using the best fit value for R^{total} in Equation 2.

Based on observations from intact soil cores, a value of 0.9 m was used to represent a depth to soil confining unit (h). This estimate of h and the mean R^{total} (54 days) from all

wetlands were used to back-calculate a saturated hydraulic conductivity (K^{bc}) value of 0.02 m d^{-1} . This value was compared to K_s^{slug} , which ranged from $0.6\text{--}2.4 \text{ m d}^{-1}$ with a mean and standard deviation of $1.3 \pm 0.6 \text{ m d}^{-1}$. The K^{bc} estimate is approximately 64 times less than the mean K_s^{slug} value and 33 times less than the K_s^{slug} value from in the wetland proper, which is likely most representative of the conductivity measured by the passive drawdown. These ratios are similar to those found by Wise et al. (2000), who reported K^{bc} between 12 and 70 times less than K^{slug} .

Model Calibration/Validation

The predictive capability of the passive-drawdown method for determining R to approximate actual GW values on a daily time step was evaluated by defining calibration and validation periods for each upland-wetland monitoring well pair. Calibration periods were selected to calculate R from Equation 2 using data subsets ranging from 10% to nearly all of the data (90%). The wetland-averaged GW RMSE values for these validation periods were all in the range between 0.57 and 0.90 cm d^{-1} , which is comparable to the range of RMSE values based on best-fits to the complete data sets (0.61 to 0.88 cm d^{-1} , Table 18). Thus, the RMSE for the predicted GW did not significantly improve with longer calibration period, with a mean reduction of 16% when using 90% rather than 10% of the total data for each wetland-well pair.

Table 18. Best-fit hydraulic resistivity determined the total monitoring record of each wetland-well pair, R^{total} , and corresponding RMSE.

Wetland-upland well pair	R^{total} (days)	RMSE (cm d^{-1})	Bias (cm d^{-1})	Days
LW1 – LW1MW7	62.8	0.75	-0.15	181
LW2 – LW2MW4	72.6	0.77	-0.04	176
LW2 – LW2MW3	61.2	0.73	-0.10	94
BW1 – BW1MW2	58.9	0.61	-0.02	349
BW1 – BW1MW5	54.5	0.70	-0.14	54
BW2 – BW2MW2	48.5	0.78	-0.12	111
BW2 – BW2MW3	28.9	0.88	-0.10	84

Discussion and Conclusions

Groundwater Outflow

The isolated wetlands in this study discharge to surficial groundwater, where pasture vegetation-driven ET cycles water to the atmosphere, lowering upland water tables and thus enhancing recharge from surface water. These processes (controlling hydraulic gradients between wetland and upland) and the soil hydraulic properties (represented by R in this study) resulted in higher groundwater recharge rates compared to those reported by Motz (1998) for 11 karst lakes in central Florida. However, considering the shorter hydroperiod at the wetlands in this study (less frequent occurrence of standing water in

the wetland), the annual recharge (1.2 to 2.2 m yr^{-1}) was found to be similar to values from the 11 karst lakes (Motz, 1998) (average $1.6 \text{ m} \pm 1.5 \text{ m yr}^{-1}$). Even though the wetlands in this study exhibited lower R values (suggesting greater recharge rates) compared to values from those lakes (55 ± 14 days versus 2380 ± 2683 days respectively), annual cumulative groundwater recharge depth was comparable because of hydrologic differences between systems.

The wetlands studied here were all head-of-ditch, with no ditch inflows, thus inflows to these wetlands are driven only by rainfall and runoff mechanisms. Average yearly rainfall in Okeechobee County is between 1.3 and 1.6 m (Lewis et al., 2001). These inflows are distributed to surface water storage, ET, ditch flow out of the wetland, and groundwater recharge. While a complete water budget was not within the scope of this study, the average annual GW recharge values found here for these isolated wetlands is significant compared to the average annual rainfall total. Thus, groundwater outflow from these wetlands is an important water flow path, with important implications for water quality management practices aimed to reduce solute export from these catchments (particularly nutrients that are related to eutrophication of receiving water bodies).

Hydraulic Resistivity

Differences in R between wetlands at the Okeechobee research sites might be explained by differences in the depth to a confining unit, site-specific saturated hydraulic conductivity, or land management practices. Processes that could lead to variability in estimated R^{event} values for a given wetland include errors in water budget components, spatial variability of soil hydraulic properties, cycles of vegetative growth and die-back, and cattle stocking rates and associated hydraulic impacts (e.g., compaction, physical mixing). Motz et al. (1998) found some variability in R (coefficient of variation between 0.09 and 0.56), but attributed it to errors in water budget components. Identifying and quantifying the mechanisms responsible for temporally variable R values was outside the scope of this study, but further research could quantify impacts of cattle and plant growth cycles on wetland hydraulic properties related to groundwater recharge.

The hydraulic resistivity at the study sites reflects groundwater recharge to a relatively shallow surficial aquifer through primarily sandy soil, and was thus found to be appropriately different than the R values reported by Motz (1998), Motz et al. (2001), and Watson et al. (2001), where groundwater recharge traversed an average of 40 m of clayey confining material. Also, a considerable difference in R was found between the study wetlands (R between 28 and 73 d) and that was reported by Wise et al. (2000) ($R = 8 \text{ d}$). The larger R values at the wetlands evaluated here can be attributed to the lower saturated hydraulic conductivity ($K^{bc} = 0.02 \text{ m d}^{-1}$) of the wetland sub-soils, compared to the peat soil ($K = 0.083 \text{ m d}^{-1}$) studied by Wise et al. (2000).

Comprehensive site characterization that included measurement of R , which may be estimated from a monitoring record during conditions describing Equation 1, could relate land management practices to long-term prediction of groundwater recharge. In addition,

a database of R values could be developed to identify sensitive areas of local groundwater recharge. Wetlands might also be ranked by R in terms of potential for hydraulic and chemical treatment potential, where higher R values might suggest longer residence time and perhaps finer underlying soil materials.

Model Performance

Predicting wetland surface water outflow through groundwater on a daily time-step was not particularly notable, as the RMSE resulting from the linear regression of R^{total} (which represents the best model fit for the entire data set) for any well pair approached the value of the mean groundwater recharge. However, it is important to note that the cumulative comparison of modeled and estimated groundwater outflow from all four wetlands was quite close, which illustrates that the model may be used to describe longer-term time periods (i.e. weeks, months, and years).

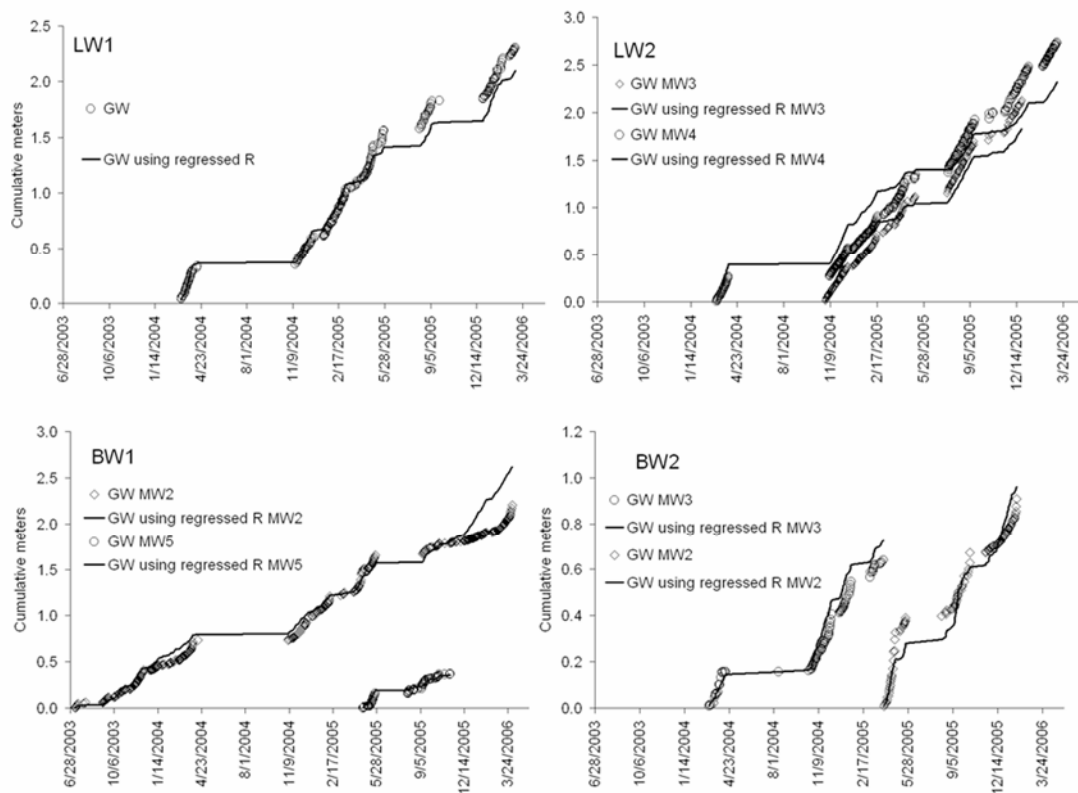


Figure 9. Cumulative water-budget based groundwater recharge, GW , compared to modeled values using R^{total} .

The bias for the entire record of each wetland well pair was small (between -0.02 and -0.15 cm d^{-1}), but suggests that the linear regression model slightly under-predicted observed groundwater outflow. Furthermore, when evaluating the distribution of daily observed and modeled groundwater recharge rates for each well pair, one-sided t-test

analysis indicated that the hypothesis that the two distributions were not similar could not be rejected ($P < 0.01$).

Predicted groundwater recharge, as also noted by Wise et al. (2000), was quite sensitive to the R parameter. For example, comparing modeled and observed groundwater recharge from well pair BW1-MW5 using $R^{\text{total}} = 55$ d resulted in a RMSE of 4 cm. Considering that $CV = 0.32$ was found for R^{event} for this wetland, estimates one standard deviation greater and less than the mean ($R = 68$ and 40 d, respectively) were also used to predict groundwater recharge over the same period, with corresponding RMSE values of 13 and 18 cm, respectively. Thus, because of the variability of GW estimates resulting from R^{event} , a period of at least several weeks may be required (under conditions describing Equation 1) to develop a predictive model of GW .

The approach described here was used to model groundwater outflow from the wetlands at times when no surface water outflow occurs. This model could be extended to all times when water is in the wetland, which would improve confidence in estimates of groundwater outflow from wetland surface water that are currently elusive, leaving surface water outflow through ditches as the only undetermined outflow. Such a water budget would address the relative importance of these different system outflows. Further characterization of other regional managed depressional wetlands using the methodology included in this research may ultimately be used to evaluate important solute treatment performance scenarios.

3.3.2 Quantifying hydrologic pathways using a water budget approach

Introduction

For this task, a water budget was estimated for four isolated wetlands (two wetland at Beaty and two wetlands at Larson ranch, respectively) to determine the quantities of inflow and outflow from pertinent processes. The conceptual hydrologic model for these wetlands will be better understood by determining water budget. Water budgets have been used in hydrologic assessments of wetlands for several reasons: (1) Assessment of water resources (Makhlouf and Michel, 1994); (2) Identification of water quality and quantity impacts to wetland ecosystems (Price and Waddington, 2000), (3) and determining links between wetlands, uplands, and receiving water bodies (Drexler et al., 1999 and Hayashi et al, 1998a). Using these estimates of inflows and outflows, an analysis of hydraulic residence time will be done to begin to evaluate the overall potential. This component will involve inflow estimates resulting from surface runoff (overland flow to the wetland), precipitation on the wetland water surface, and groundwater inflow. The outflows of these wetlands are from *ET*, groundwater outflow, and surface water outflow through the ditch network. The dynamics and quantities of inflows and outflows of managed wetlands in the Basin have not been reported in the literature, and will be presented herein.

Results from the water budget hold information describing the temporally- and spatially-variable extent of the surrounding uplands that are involved in the treatment process, or hydrologic zone of influence. The idea of a temporal and spatial component non-point source inflow has been described in river systems and termed variable source zones (Ward, 1984). These zones are controlled by groundwater proximity, topography, and rainfall rate (Dunne and Black, 1970; Montgomery and Dietrich, 1995; Frankenberger et al., 1999; and Lyon et al., 2004).

The objectives of this task were to:

1. Estimate the inflows and outflows of each wetland to build a conceptual hydrologic model
2. Identify the wetland hydrologic zone of influence within the landscape
3. Determine the distribution of nominal residence times to interpret the hydrologic component associated with contaminant treatment potential.

This research will provide a conceptual hydrologic model for isolated wetlands in the Basin. It also has implications related to hydrologic management of wetlands for landscape-scale chemical treatment potential and nutrient loading to Lake Okeechobee.

Water Budget

Detailed topographic surveys were conducted to determine the bathymetry of the four study wetlands at Larson Dixie and Beaty ranches. The bounds for these surveys were typically 10-20 m into the upland (as delineated by a vegetative border of *Serenoa repens*). A line-of-sight laser level was used in combination with a hand-held Garmin GPS to collect point elevation data. Ordinary Kriging interpolation of those elevation points, using ARCGIS (ESRI ArcMap 9.1),

yielded a continuous surface (average root mean square error, RMSE = 0.175 m). Wetland footprint area and relative ditch elevation (elevation difference between lowest point in the wetland and the maximum ditch bottom elevation) were determined from this survey. For this study, it is important to note that the critical flow depth (h^d) (elevation difference between the bottom of the wetlands and the highest elevation of the ditch) for wetlands (LW1, LW2, BW1, and BW2) was 0.3, 0.3, 0.5, and 0.4 m respectively. The relative areas of these wetlands were quite different and resulted in volume estimates between wetlands (while changes in depth were similar) (Figure 10). The differences in area are critical in understanding the differences in flow rates of each component between wetlands.

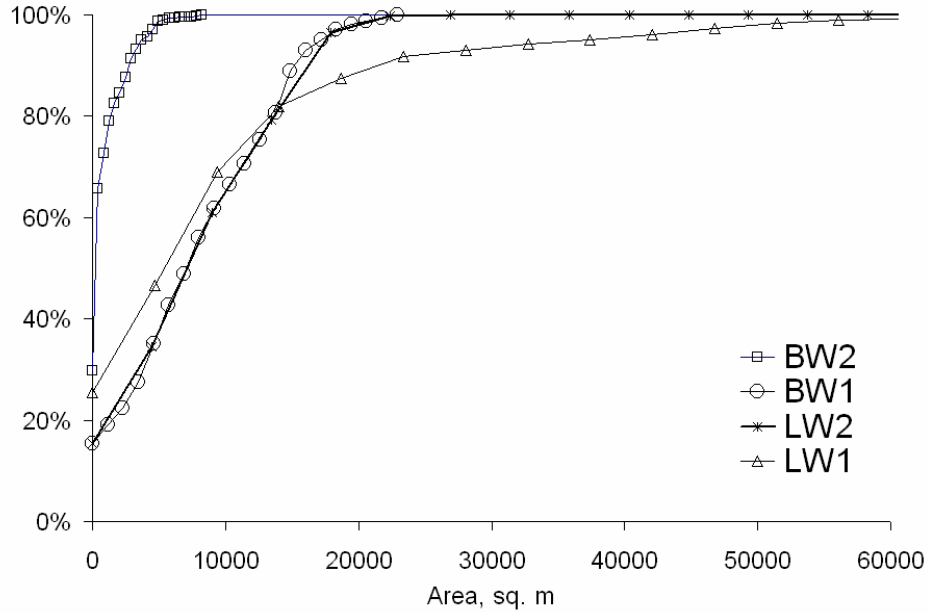


Figure 10. Cumulative frequency distribution of wetland areas.

Water budget data for this study was calculated over approximately 640 days between May 2004 and March 2006. The water budget for these isolated wetlands was defined as:

$$dH^{wet} = P - ET - D + S \pm GW \quad (\text{Equation 1})$$

where S is overland flow, D is surface water outflow (through ditches), dH^{wet} is the change in wetland surface water storage, ET is evapotranspiration, and GW is groundwater inflow or outflow. Rainfall and dH^{wet} were calculated from direct measurement, while ET was estimated using an empirical function using meteorological data. A combination of direct measurement and linear regression modeling was done to determine the remaining residual term(s) in the water budget: groundwater flow, ditch flow, and overland flow. It was necessary to incorporate linear regression modeling of wetland-groundwater flow because not all processes were directly measured, leaving sometimes two or three unknown water budget components, depending on the hydrologic conditions. Equation 1 may be reduced to fewer terms when the wetland stage is lower than h^d for ditch flow and/or when rainfall and overland flow do not occur. For the simplest case, no rainfall, runoff, or ditch flow, Equation 1 can be reduced to:

$$dH^{wet} = P - ET - GW^{out} \quad (\text{Equation 2})$$

In this case, the GW^{out} term is the only unknown, as ET was assumed to be a known component. When rainfall occurred, but no ditch flow occurred, Equation 1 reduced to:

$$dH^{wet} = P - ET + S \pm GW^{mod} \quad (\text{Equation 3})$$

Here, the rainfall, ET , and GW^{mod} (modeled groundwater inflow/outflow from linear regression) were known. This left only the overland flow (S) as the residual term. When ditch flow occurred without rainfall and runoff, Equation 1 reduced to:

$$dH^{wet} = P - ET - D \pm GW^{mod} \quad (\text{Equation 4})$$

Ditch flow (D) is left as the residual term. During these periods, the ditch flow was linearly regressed to Manning's equation and will be explained further below in the groundwater section of this chapter. When all water budget components occurred, such as in Equation 1, the overland flow was left as the residual term and modeled groundwater inflow/outflow and ditch flow (D^{mod}) were used as known components:

$$dH^{wet} = P - ET + S \pm GW^{mod} - D^{mod} \quad (\text{Equation 5})$$

To evaluate the degree of error in the estimated water budget, the measured daily wetland stage was compared to the wetland stage as determined by the sum of the estimated water budget components. Figure 11 shows an example of the daily measured wetland stage (storage) and the water budget-estimated storage. Detailed methodologies for estimating and/or measuring each water budget component will be described later in this section. Calculating the volume-based water budget was done by first calculating the depth-based water budget for each day and comparing estimated wetland water stage against the measured daily values. Then, the depth values for each water budget component were multiplied by the day-specific wetland area using information from the stage-area relationships to generate volume-based estimates. These volume-based estimates assumed that the wetland porosity was equal to one, meaning that the volume of wetland surface water occupied by vegetation was negligible. In reality, this is probably not the case, as Kadlec and Knight (1994) report porosity values of treatment wetlands between 0.7 and 0.9.

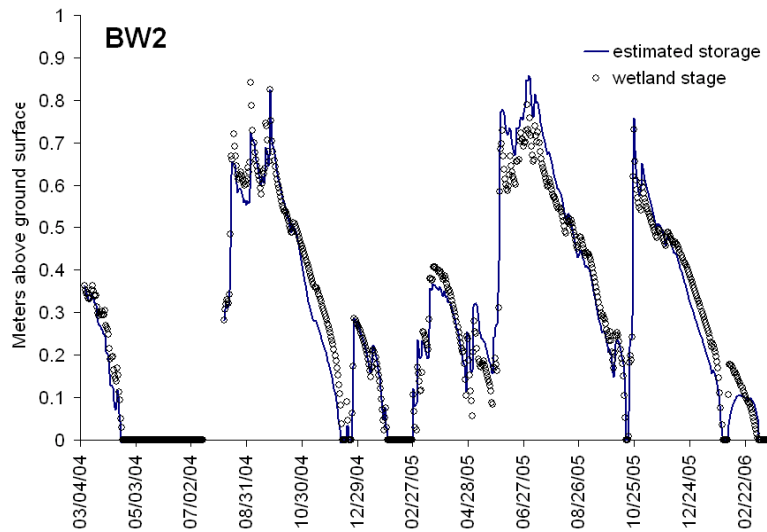


Figure 11. Comparison of daily wetland stage (storage) and water budget-estimated storage for Beaty wetland 2.

Precipitation and Evapotranspiration

Precipitation (P) was measured onsite at two locations using data logging tipping buckets (Onset Communications Corp. model RG3-M) and the Bassett monitoring station maintained by the South Florida Water Management District (SFWMD) (located on the Beaty Ranch).

Approximately half of the rainfall data came from onsite measurements, with the remaining half from the Bassett rain gage (SFWMD).

The Penman-Monteith method was used to estimate evapotranspiration (ET) from the wetlands. Approximately half of the meteorological data necessary for this was collected onsite with Campbell Scientific instrumentation, while the remaining half was collected at the SFOO1 station, also maintained by the SFWMD (located at a distance of approximately 14 km). See Task 3.3.1 for further details.

Groundwater Outflow

Groundwater inflow and outflow in water budget analyses have been the most difficult to quantify because of its complex behavior (Drexler et al., 1999; Hunt et al., 1996). Groundwater outflow (GW^{out}) between wetland surface water and upland groundwater was determined from a constrained water budget that excludes ditch flow, overland flow, and rainfall. Data logging pressure transducers (Minitroll 500, In-Situ Inc.) were deployed in fully-screened 3.18-cm diameter well casings to measure water levels in the upland and wetland surface water. At each wetland, one well was located at the lowest topographic position in the wetland and at least one other was located at the wetland fringe where wetland vegetation transitions to *Serenoa Repens* (at approximately 50 m from the wetland center). These measurements of hydraulic head in the wetland surface water and upland groundwater will serve to estimate the groundwater inflow and outflow between wetland and upland. Linear regression modeling was used to approximate groundwater inflow and outflow when overland flow and/or ditch flow occurred. It was important to develop a means to directly determine as well as estimate groundwater flow so that

overland flow and ditch flow would be left as the residual terms in the water budget (for which no measurements were collected).

This study incorporates the data analysis done in Task 3.3.1 to develop observed outflow estimates, as well as linear regression modeled approximations of groundwater outflow. Observed groundwater outflow was derived from a constrained daily water budget that excluded rainfall, runoff, and ditch flow (wetland water stage was below h^d). Of the total number of days when groundwater outflow occurred, a wetland-averaged percentage of 34 was directly determined from the constrained water budget. Linear regression of Darcy's law and these observed outflow data was used to determine a wetland-specific hydraulic resistivity and approximate the observed groundwater outflows. A reasonably good fit (r between 0.71 and 0.85) resulted from the linear regression of groundwater outflow (from the water budget) and Darcy's law (Task 3.3.1). For times when groundwater outflow conditions did not meet the criteria for the constrained water budget, Darcy's law was used to approximate groundwater outflow and included the fit hydraulic resistivity and upland-wetland pressure head difference.

Groundwater Inflow

The occurrence of groundwater inflow, as determined by the days for which upland water elevation was greater than wetland surface water elevation, was only 8% of the time that standing water was in the wetland. The constrained water budget described in the "Groundwater outflow" section was not usually viable for direct observation of groundwater inflow, because hydraulic gradients that resulted in groundwater flow toward the wetland were controlled by rainfall, which in turn added overland flow to the water budget equation as a second unknown. With relatively fewer periods of time when groundwater inflow occurred, it was assumed that the inflow dynamics could be approximated by Darcy's law and the fit hydraulic resistivity determined from the groundwater outflow data.

Surface Water Outflow

Low topographic gradients and vegetation in ditches often contribute to flooded conditions, which limit the design of surface water outflow control structures. The occurrence of surface water outflow through these ditches is principally controlled by hydrologic conditions that nudge the wetland stage above the lowest elevation of the ditch bottom (between 0.2 and 0.5 m above the lowest elevation in the wetland bathymetry). Once surface water in the wetland has risen above h^d , the magnitude of flow is conceptualized using elements of open-channel flow described by Manning's equation: roughness, slope, and the geometry of the ditch.

Long-throated weirs were installed in the outflow ditches of the four wetlands, using flow specifications best suited to low-flow, submerged conditions. Water surface elevations measured in the stilling well and behind the weir (approximately 2 m) did not exhibit adequate head-loss to qualify a flow rate. However, observations of surface outflow events through ditches (unidirectional grass matting and physical observations) were noted for all wetlands.

Because of the inadequate head-loss across the flumes, another methodology was developed to estimate ditch flow. This methodology involved another form of the daily water budget determined by the following constraints:

1. Water surface elevation was above h^d
2. Rainfall/runoff did not occur
3. Groundwater inflow/outflow, derived from linear regression modeling, was considered a known variable.

Table 30 reports the percent of ditch flow record for which ditch flow was the residual term in the constrained water budget. The wetland-averaged percent of days for which ditch flow occurred (ditch flow was the residual term in the water budget) was 45. The other 55 % of the ditch flow record that included rainfall/runoff (a second unknown) was estimated using Manning's equation. Manning's roughness coefficient of 0.03 (typical value for lightly vegetated, un-maintained channel) was used for all wetlands to reflect the grassy conditions of the ditches. Linear regression of the daily ditch outflow rates (resulting from the constrained water budget as described above) and Manning's equation was done to fit the slope of the ditch to best approximate ditch flow. Once direct measurements and linearly regressed estimates of groundwater and ditch flow were established, the water budget had only overland flow as the remaining term.

Overland Flow

One of the most common methods for determining runoff is using water control structures (Branfireun and Roulet, 1998; Metcalfe and Buttle, 1999). In these examples, runoff would be comparable to overland flow at the Larson and Beaty study sites. When water control structures are not feasible to record overland flow, empirical functions that hinge on simple rainfall-dependent relationships have been used (Prescott and Tsanis, 1997). Some investigations of water budgets exclude the overland flow component because it is difficult to measure or infrequent enough to be considered insignificant (Drexler et al., 1998). However, in this study, overland flow was considered to be important because of shallow water table conditions that reduce soil-water storage. In addition, while soils in this region are primarily sand, drainage is poor (Soil Conservation Service classification A/D or B/D). Capece et al. (1988) reported saturated hydraulic conductivities between 1.5 and 15 cm h⁻¹ in the basin, while values between 6 and 24 cm d⁻¹ were reported at Larson and Beaty study sites (Task 3.3.1). Because the overland flow at these wetlands entered the wetlands from around the perimeter, inflow water control structures were not viable. Overland flow was considered to be the only residual term, once all other components were either measured or estimated through regression modeling (depending on the hydrologic conditions).

Hydraulic Residence Time

An important hydraulic parameter related to treatment wetland performance (usually in constructed wetlands) is the hydraulic or *nominal* residence time (*HRT*):

$$HRT = \frac{V}{Q} \quad \text{(Equation 6)}$$

where V is the wetland volume [L^3] and Q is usually defined as the average inflow and outflow rate [L^3T^{-1}]. Paired with reaction and uptake coefficients, the HRT of a treatment wetland may be incorporated into a continuously-stirred tank reactor or tanks-in-series modeling approach to evaluate treatment performance. This type of analysis is typically applied to steady-state flow conditions with controlled inflow and outflow to evaluate wetland hydraulic efficiency. These study wetlands do not exhibit these characteristics, as the wetland volume is transient and inflows and outflows are not only transient, but result from different hydrologic processes, which exhibit unique rates and dynamics.

In this research, a methodology was implemented to compare HRT of traditional treatment wetlands to transient, dynamic wetlands in general by assuming a pseudo-steady-state flow condition. Still applying Equation 6, the rates of water budget components were calculated on a daily time step and assumed at steady-state for each day. The Q term in Equation 6 was defined as the sum of all outflows (groundwater outflow, ET , and ditch flow out when it occurred). Following this methodology, a distribution of flow rates was generated and calculation of HRT was done using the resulting distribution. Wetland volume was determined from the daily stage-volume relationship and combined with daily total Q to ultimately create a distribution of HRT s based on data from the entire monitoring record of each wetland.

Hydroperiod

Hydroperiod may be expressed on either a depth or volume basis. Dunne et al. (2007) reported hydroperiods in terms of depth for these study wetlands during the same time period. In this work, it was important to describe the hydroperiod on a volume basis, because of the implications for wetland treatment efficiency and HRT s. Using the time series wetland water stage data from each wetland (described in the “Water Budget Estimation” section), the hydroperiod was calculated for each wetland. To better compare hydroperiods between the study wetlands, results from the volume-based analysis of hydroperiod distribution were scaled to their corresponding average wetland volume (AWV).

Results and Discussion

Hydroperiod

The depth-based hydroperiod for BW1, BW2, LW1, and LW2 was reported as 302, 233, 256, and 219 respectively (Dunne et al., 2007). Results from the volume-based hydroperiod analysis showed that there was a difference in AWV distribution between more-intensely ditched wetlands (LW1 [westerly wetland], LW2 [easterly wetland], and BW2 [northerly wetland]) and the less-intensely ditched BW1 wetland [southerly wetland]. The majority of the wetland hydrologic dynamics occur within two AWV s, consisting of approximately 85% of the variability of surface water occurrence. At LW1, LW2, and BW2, 73% of the variability was described at or below one AWV , whereas 60 % of the variability was described at or below one AWV for BW1. Similar volume-based hydroperiod distributions were observed between the study wetlands except BW1, which exhibited a consistently wetter condition (Figure 12). The most obvious explanation for differences in AWV distributions is because each wetland had a specific h^d that defined the dynamics between more rapid surface water outflow and longer groundwater and ET outflows.

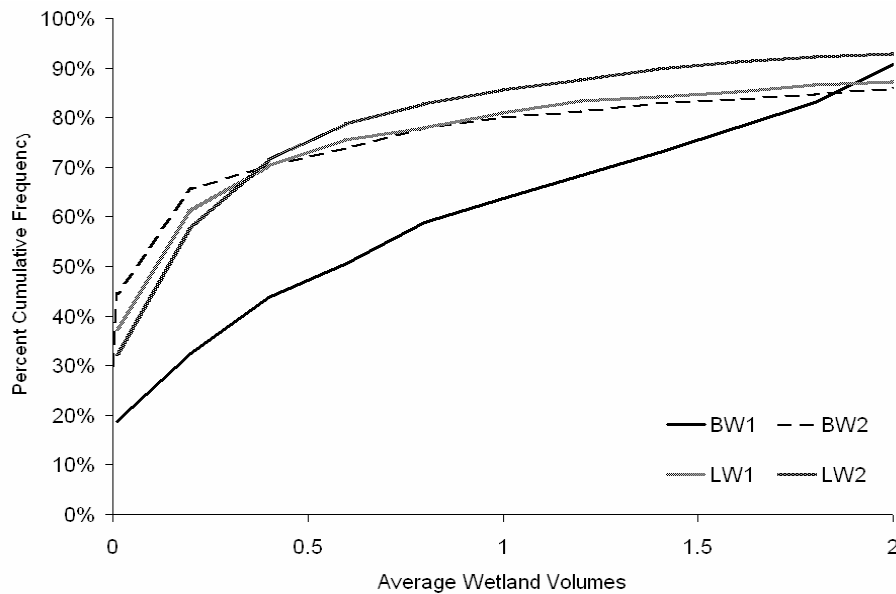


Figure 12. Cumulative distribution functions of scaled volumes for the study wetlands.

While the h^d of the wetlands affected the distribution of $AWVs$, wetland bathymetry was also an important aspect of the behavior of wetland hydroperiod. For example, at a value of one AWV , LW1, LW2, and BW2 exhibit similar distributions. However, the h^d at BW2 was 10 cm higher than that at LW1 and LW2, suggesting that reduced ditch flow out of BW2 should skew the distribution of $AWVs$ toward relatively wetter conditions. This is not the case, suggesting that wetland bathymetry may also be important in describing the variability in AWV distribution.

Water Budget

In this study, one objective was to evaluate the potential treatment of these wetlands based on hydrology. As presented previously, a depth-based water budget was completed to develop a close fit between the observed daily change in wetland surface water and the sum of inflow and outflow components in Equation 1. Results from this type of analysis were scaled by the corresponding daily values of wetland surface water area to yield a volume balance. The volume-based water budget was used exclusively in this paper to allow more perspective into the potential management of these wetlands, especially for their use as natural treatment wetlands. The absolute values of flow rates among wetlands were quite different. For example, LW1 wetland exhibited consistently large values for each water budget component, relative to the other study wetlands. The differences in the absolute values between wetland-specific flow rates (Table 19) are due to the differences in the stage-area relationship and corresponding volume-weighted hydrograph (Figure 13) as well as the h^d , which constrains the occurrence of ditch flow.

Table 19. Geometric mean flow rate of each water budget component and their associated geometric standard deviations

Site	Rain	ET	Runoff	GW^{out}	GW^{in}	Ditch flow
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LW1 [$\text{m}^3 \text{d}^{-1}$]	39 ± 10	26 ± 4	164 ± 7	88 ± 3	69 ± 4	226 ± 6
LW2 [$\text{m}^3 \text{d}^{-1}$]	16 ± 13	12 ± 4	42 ± 11	30 ± 4	5 ± 3	120 ± 5
BW1 [$\text{m}^3 \text{d}^{-1}$]	18 ± 8	15 ± 2	48 ± 22	22 ± 2	11 ± 4	52 ± 7
BW2 [$\text{m}^3 \text{d}^{-1}$]	1 ± 18	1 ± 5	3 ± 15	1 ± 4	1 ± 6	19 ± 4

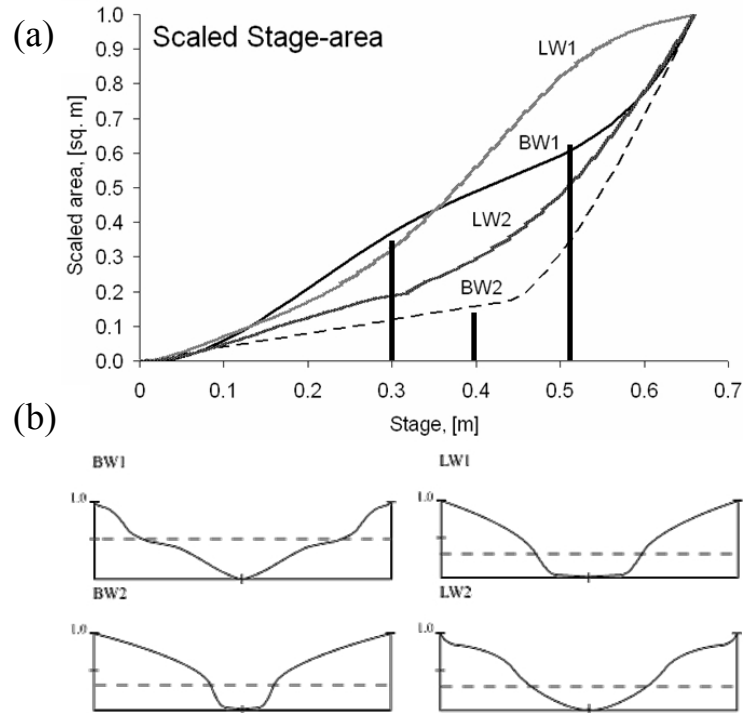


Figure 13. Stage-area relationship for study wetlands (a) as well as their corresponding generalized bathymetric cross-sections (b). The solid vertical lines represent the locations of the critical ditch elevation.

The distributions of inflow and outflow rates for each wetland were log-normal, and thus the geometric mean and corresponding geometric standard deviation were used (Table 19). Trends in relative inflow and outflow rates were similar for all wetlands, with runoff (overland flow) and ditch flow exhibiting the greatest values. Overland flow and ditch flow are important components in the conceptual model of these wetlands as a means to improve water quality discharged to Lake Okeechobee. Overland flow is particularly important because it has the potential to transport nutrients from mineralized, decomposing plants and surface-applied fertilizers. Relatively higher ditch flow rates were expected because a critical volume (corresponding to h^d) of water must be present before water will discharge from wetlands to ditch. Conceptually, these large outflow rates from the ditch are not only important in terms of quantity, but also because of the connectivity between wetland surface water (and associated water quality) and receiving water bodies. Cumulative frequency plots in Figure 14 show the distributions of normalized ditch flow (m d^{-1}), which was scaled to the maximum ditch flow rate so that all wetlands could be displayed reasonably well on the same plot. Also shown in Figure 14 are the cumulative frequency distributions of water surface elevation in the ditches of the study wetlands. These were also included to identify the occurrence of backwater influence in the ditches. All wetlands exhibit some frequency of standing water in the ditch in the absence of ditch flow, ranging from 2 to 15% of the time that ditch flow occurs.

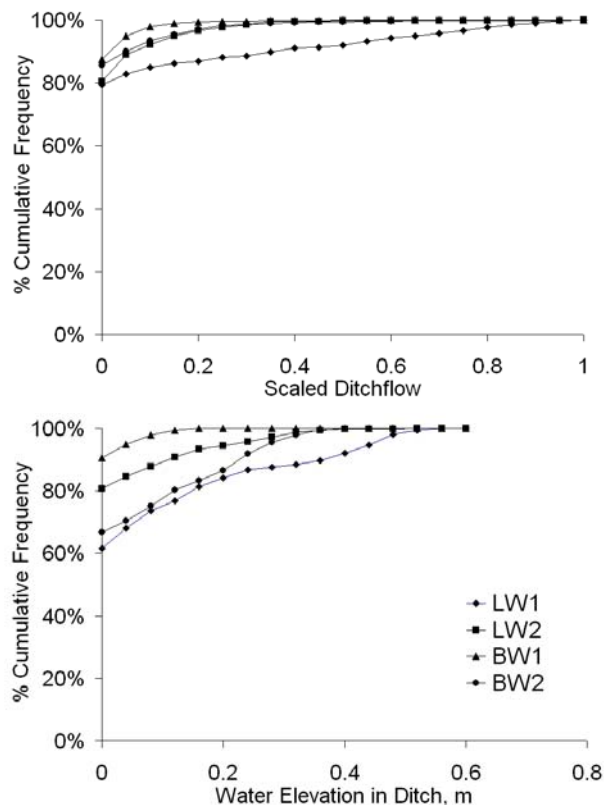


Figure 14. Percent cumulative frequency plots of scaled ditch flow and water surface elevation for the study wetlands.

Resulting groundwater inflow and outflow rates were higher than expected, being equal to or greater than *ET* rates. This contributes significant information to the overall conceptual model of wetland hydrology, as wetland surface water interacts with groundwater in the surrounding upland landscape at significant rates. Intrinsic to this interaction is the water quality aspect and wetland treatment efficiency that is linked to these outflow pathways.

In this study, it was important to characterize the wetland inflow and outflow rates of each study wetland to identify the role of each hydrologic component related to water and chemical transport. The total volume of water associated with each water budget component is also important to quantify because it offers information toward development of a conceptual hydrologic model. An important discovery related to the hydrologic behavior of these wetlands is the equally large sources of overland flow and rainfall. Rainfall would be expected to dilute chemicals that enter the wetland from outside sources, while overland flow may transport chemicals from the surrounding pasture upland areas. Therefore, the conceptual model of a flow-through wetland does not necessarily apply to these wetlands, as so little groundwater enters the wetlands.

For ease of inter-wetland comparison, the percentage of total wetland inflow or outflow was reported. This was taken from cumulative flow rates at the end of each wetland's monitoring record (Table 20).

Table 20. Percentage of the total volume of inflow or outflow of each water budget component with estimates of fraction of contributing upland area to wetland area (F^{cont}).

	Rain	Runoff	GW^{in}	ET	GW^{out}	Ditch	α	F^{cont}
LW1	36	59	4	19	27	54	6.0	0.92 ± 1.3
LW2	42	57	1	17	26	57	3.7	1.17 ± 1.5
BW1	50	48	2	30	53	17	3.7	1.03 ± 1.4
BW2	53	45	2	19	11	70	3.6	0.77 ± 1.2

Wetlands LW1 (west) and LW2 (east) (less than 1 km apart on the same ranch) exhibited similar contributions of inflows and outflows, even though the inflow and outflow rates were higher at LW1, with ditch flow comprising approximately 50% of the total outflow. In addition, approximately 30% of the total outflow was attributed to groundwater outflow, with *ET* making up the remaining 20%. It is important to note that the groundwater component was significant, considering the hydraulic management to drain the surface water via ditches. Differences in bathymetry between LW1 and LW2 were not significant enough to independently govern the relative contributions of inflows and outflows. Corresponding h^d for these two wetlands was also similar, which probably contributed to their similar hydrologic dynamics by constraining the occurrence of inflows and outflows to a similar relative degree.

The fraction of rainfall, runoff, and groundwater to the total inflow at the Beaty ranch wetlands (BW1 and BW2) were similar to those at Larson ranch (LW1 and LW2) (Table 20). This implies that the physical properties, such as soil hydraulic properties related to groundwater inflow and infiltration mechanics related to overland flow, that govern each inflow do not vary significantly enough to create large differences in relative fractions of inflow to the wetlands. Any subtle differences in inflow fractions between wetlands may be attributed to differences in

wetland bathymetry as well as errors in the estimates themselves. The more pronounced differences in fractions of outflow from the BW1 and BW2 wetlands indicated that probably both h^d and wetland bathymetry had a significant role in determining their cumulative volume.

Ditch outflow from BW1 only made up 17% of the total outflow, compared to 70% at BW2. BW1 had the highest h^d , which resulted in relatively low ditch flow and exaggerated ET and groundwater outflow compared to the other study wetlands. One possible explanation for the relatively large volume of ditch outflow is that the wetland bathymetry exhibits relatively small surface area below h^d and very large surface area above it. It was important to observe that in BW1 (the wetland least impacted by ditching) groundwater outflow made up 53% of the total outflow. Nutrients transported through groundwater will require a longer time to reach a receiving water body and may be sequestered by plants or immobilized by sorption to soil and organic matter, or precipitated through metal complexation processes.

Wetland hydrologic dynamics in a time-series format offer information that is also important for creating a conceptual hydrologic model as well as determining consequences of hydraulic management related to nutrient treatment efficiency. To identify the dynamic behavior of the wetlands in the basin, two examples of hydraulic regimes, BW1 representing relatively low impact from ditching compared to LW1 which was more extensively ditched, were evaluated using cumulative $AWVs$ (Figure 15).

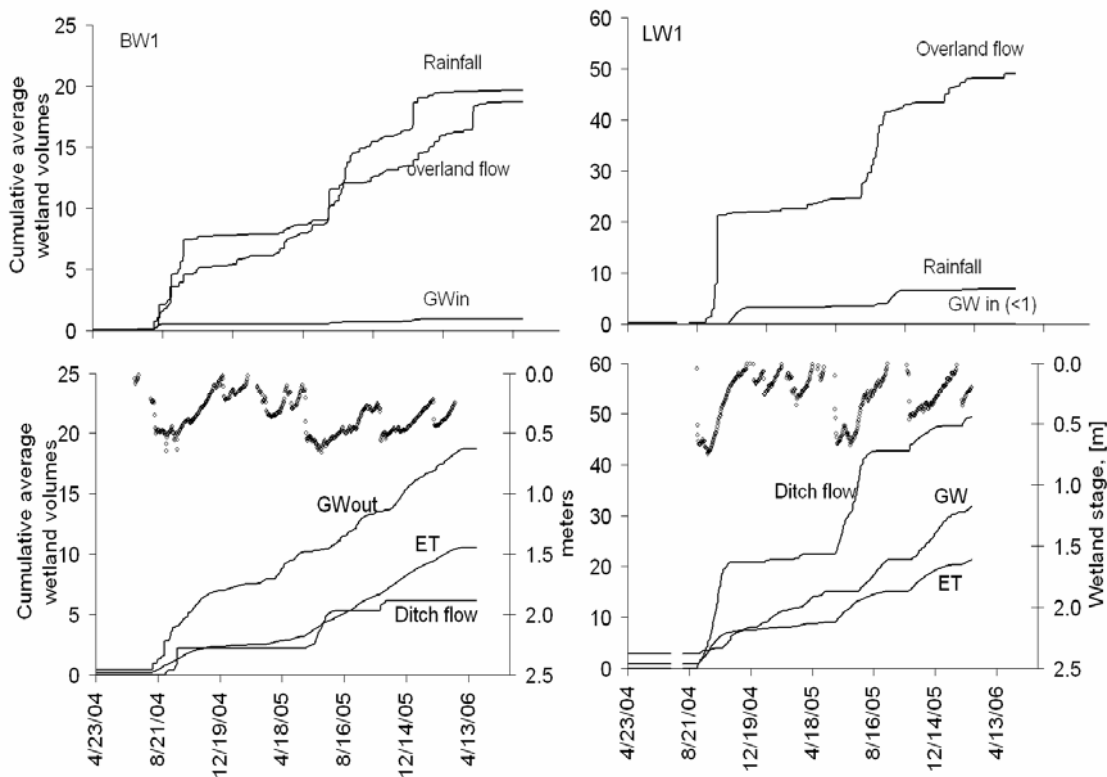


Figure 15. Cumulative inflow and outflow volumes for BW1 wetland and LW1 wetland

Overland flow to the wetland demonstrated a high intensity low duration behavior, corresponding with large increases in wetland surface water storage. Rainfall on the wetland water surface was more frequent than overland flow, but exhibits lower inflow rates. Evapotranspiration exhibited significantly smaller flow rates than the other hydrologic components, partially because the wetland area scaling wetland volume was at or below h^d 65% of the monitoring record (with standing water in the wetland) and partially because the ET rate itself was relatively lower than the other outflow rates. Groundwater inflow only occurred under very constrained conditions, where the upland water table elevation was greater than the wetland surface water elevation (a wetland-averaged 24% of the time standing water occurs in the wetland). It only appeared to occur under relatively wet conditions and was limited by the rate by which water was evapotranspired from the surrounding upland plants that induced divergent outflow from the wetland surface water via groundwater. Unlike overland flow, high ditch flow rates from the wetlands did not necessarily coincide with large increases in wetland volume. This is probably because rainfall/overland inflow events often occurred during times when the wetland contained little or no water, resulting in a large pulse of overland flow to the wetland that, in turn, filled the wetland instead of flowing out of the ditch. Once h^d was exceeded, the duration of significant ditch outflow occurred over considerably long periods of time (in the order of weeks).

In addition to hydrologic dynamics in the time-series data, the frequency of each inflow and outflow was also reported in Table 21. The frequency of ditch flow at BW1 was less often compared to LW1, occurring only 9% of the hydroperiod versus 20% at LW1 (Table 21).

Table 21. Percentage of total number of days that each component occurred over the monitoring record. Also, the Predicted Ditch column represents the percentage of time that predicted ditch flow was calculated versus observed from the water budget.

	Rain	Runoff	GW^{in}	ET	GW^{out}	Ditch	Predicted Ditch
LW1	37	27	32	100	69	20	56
LW2	39	22	28	100	62	21	51
BW1	34	27	8	100	87	9	59
BW2	37	15	27	100	65	15	61

Nonetheless, ditch flow in general only occurs over very short time intervals, but the amount of water generated during these outflow events is incredibly significant (Table 22).

Table 22. Percentage of the total volume of inflow or outflow of each water budget component with estimates of fraction of contributing upland area to wetland area (F^{cont}).

	Rain	Runoff	GW^{in}	ET	GW^{out}	Ditch	α	F^{cont}
LW1	36	59	4	19	27	54	6.0	0.92 ± 1.3
LW2	42	57	1	17	26	57	3.7	1.17 ± 1.5
BW1	50	48	2	30	53	17	3.7	1.03 ± 1.4

BW2	53	45	2	19	11	70	3.6	0.77 ± 1.2
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This short duration, high energy outflow may be closely associated with distressing climatic conditions, such as high intensity and duration rainfall (e.g. tropical storms). Similarly, runoff to the wetlands was also less-frequent and much more sporadic than rainfall, occurring 15 and 27% of the hydroperiod at BW1 and LW1 wetlands respectively. Parallel with ditch flow, the total volume of runoff was very important in the overall water budget, even though the duration was short. It is worth mentioning that the duration of groundwater outflow was considerably longer than the ditch flow (87% of the hydroperiod at BW1 and 69% of the hydroperiod at LW1), but accounted for less volume (Table 22).

Upland Area Contribution

As overland flow was found to play a critical role in the wetland water budget, it was important to identify the extent to which the wetland and upland are connected. The contributing upland area associated with overland flow was estimated for the study wetlands. In the water budget estimation, the volume of overland flow was scaled to the wetland area, so it was necessary to re-scale those values to an analogous upland area. Once those values were re-scaled, a daily value for contributing area was calculated. To re-scale the overland flow volumes to the upland, a correlation between precipitation and observed overland flow (water budget based) was developed to independently calculate a depth of overland flow (d^{over}):

$$d^{over} = average(P_{t=i}, P_{t=(i-1)}) \cdot \alpha \quad (\text{Equation 7})$$

where $average(P_{t=i}, P_{t=(i-1)})$ is the average value for rainfall for the i^{th} time step and the previous time step and α is a unit-less runoff coefficient. This approximation is related to the rational method, but it incorporates the idea of routed overland flow from earlier rainfall events. The modeled overland flow captured between 50 and 60% of the variability of the observed overland flow. The population of fraction of contributing area to wetland area (F^{cont}) exhibited a log-normal distribution. Geometric mean and standard deviation values F^{cont} and α for each wetland are reported in Table 22. From this table, values of F^{cont} appear to be, on average, similar to wetland area; however, the geometric nature of the distribution is worth analyzing in terms of daily values of F^{cont} , as seen in the cumulative density function(cdf) plot (Figure 16). Data suggests that between 45 and 60% of the time, the value of F^{cont} is less than one, indicating that the upland contributing area is less than the wetland area. It is worth noting that the frequency of overland flow for the entire monitoring record was between 22 and 33% of the time. This fact alone is an important finding, as this flow path contributes at least 50% of the inflow to these wetlands. Another important point is that the value of F^{cont} is highly variable for the remaining days when overland flow occurred, reaching a value greater than 10 (Figure 16). Six snapshots in time of LW1 bathymetry and contributing upland area is shown in Figure 17 to better visualize the wetland area – overland flow relationship. This figure illustrates that greater rainfall amounts correspond to the

relative contributing area, where greater rainfall depth corresponds to a larger upland contributing area.

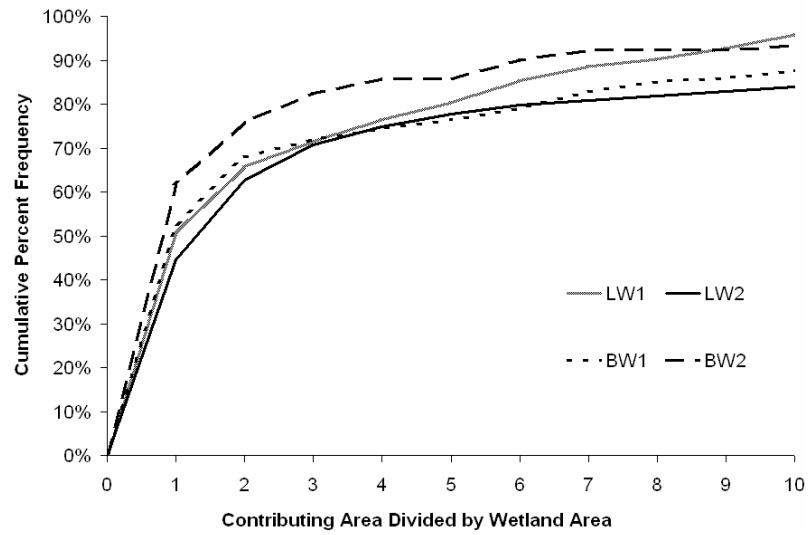


Figure 16. Cumulative density functions for the fraction of contributing upland area to wetland area for the study wetlands.

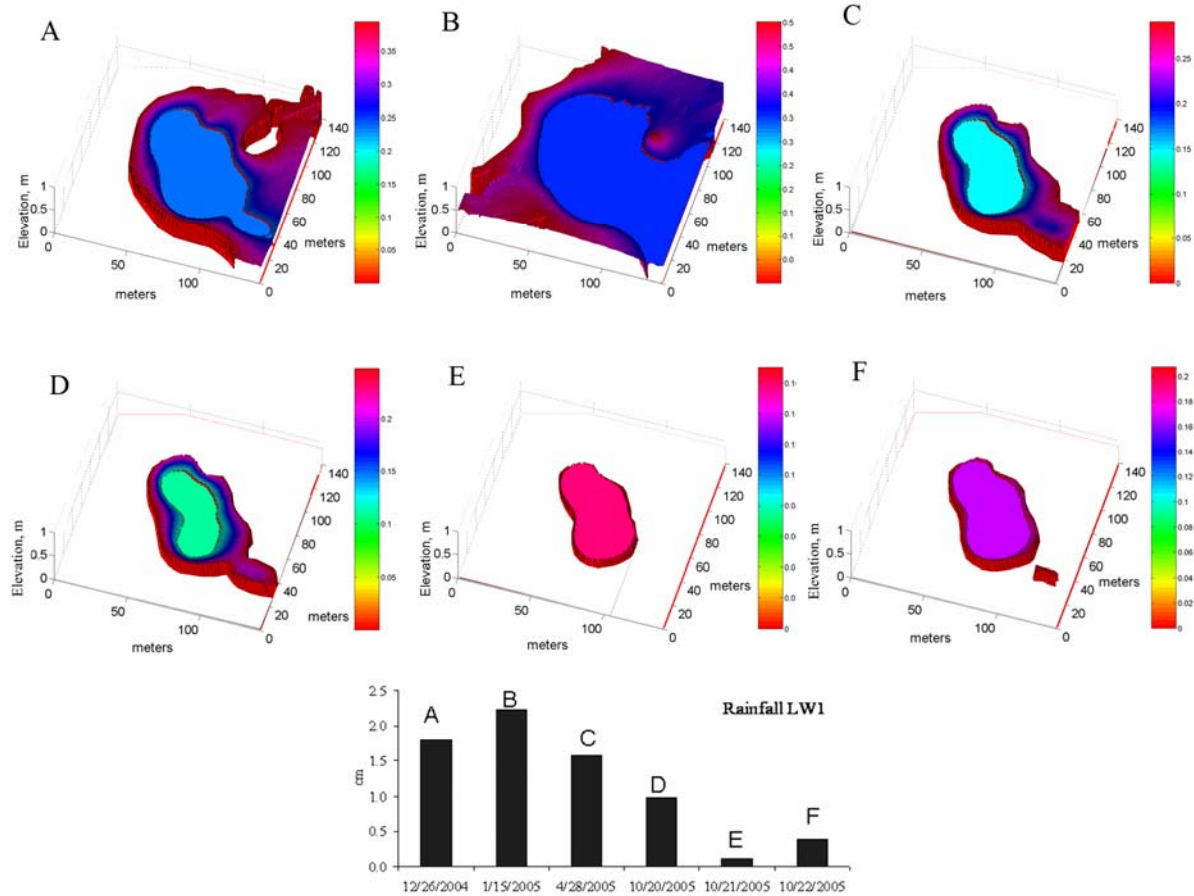


Figure 17. Wetland LW1 contributing area and corresponding rainfall for six days.

Hydraulic Residence Time

In this study, the average of the inflows and outflows were not used to represent the flow rate in the *HRT* equation. The study wetlands did not exhibit steady-state flow rates through the wetland and were constrained by the sum of the outflows. Therefore, the residence time was determined by the sum of the outflows (ditch flow, evapotranspiration, and groundwater outflow). The volume term in the hydraulic residence time concept integrated the inflows from multiple sources (rain, overland flow, and groundwater inflow). A distribution of *HRTs* was calculated based on the daily wetland volume and sum of daily outflows. The distribution of hydraulic residence time for LW1 and LW2 wetlands were similar, with less than 30-day residence times occurring 55% of the time (Figure 18). The cumulative *HRT* distribution for BW1 was shifted, denoting generally longer residence times. At BW1, *HRTs* of less than 30 days occurred only 40% of the time. A skewed *HRT* distribution was noted for BW2, resulting in shorter overall residence times. This relatively dryer behavior at BW2 was also seen in trends in hydroperiod, but the *HRT* analysis offers a quantitative comparison of hydraulic performance between wetlands.

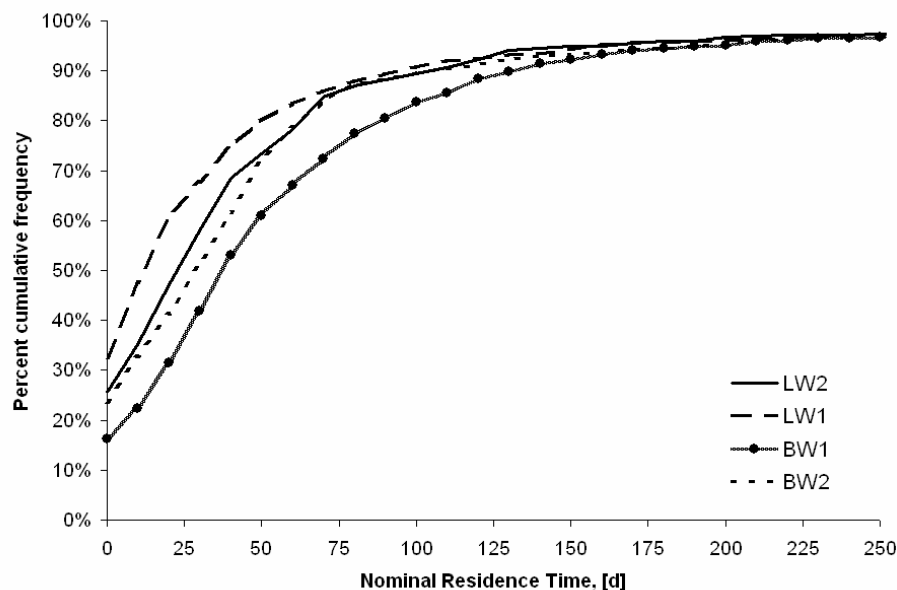


Figure 18. Cumulative distribution function of hydraulic residence times for the study wetland.

Conclusions

Overland flow and ditch flow are two of the most important components in the hydrologic regime of the isolated wetlands for several reasons. While their relative frequency was low compared to the total inflow and outflow, the flow rates and total volumes associated with these components are consistently greater than the other flow rates. Also, nutrient loads associated with overland flow may be inflated from the particulate and labile forms of P that collect on the surface of upland soils from decomposing plant material and cattle manure. These loads may be transported from considerable distances in the catchment. Finally, the ditch flow integrates the mixing processes from inflows and outflows and creates a rapid conduit of flow to receiving water bodies over considerable periods of time. The climatic conditions associated with the monitoring record in this study were atypical, as four named hurricanes passed within kilometers of the field sites. The resulting water budget was clearly not representative of long-term meteorological averages, but may instead represent the worst-case scenario of hydraulic loading, which ultimately impacts Lake Okeechobee.

The HRT distributions for these wetlands appear to be quite long, since the hydroperiod of these wetlands was between 219 and 305 days. More work is needed to evaluate the overall treatment efficiency of these wetlands. A first-order uptake model paired with residence time information may shed further light on the treatment efficiency of these wetlands. Furthermore, a larger system domain might be more appropriate for estimation or modeling nutrient treatment efficiency, because wet and dry cycles influence the availability and production of labile P within, as well as on top of wetland and upland soil.

The hydrologic dynamics of the study wetlands appear to offer some encouragement toward their use as treatment wetlands. However, further hydrologic management of the ditch outflow may increase the *HRT* and thereby promote water retention and subsequent nutrient uptake. The BW1 wetland exhibited hydrologic dynamics that may most closely reflect those of a hydrologically restored wetland, because its h^d was the highest of all study wetlands. However, during extreme rainfall events, which occur relatively often in Florida in the form of tropical storms and hurricanes, surface water can flood entire pastures, creating a highly connected and hydrologically un-manageable conditions. Secondary treatment mechanisms, such as retention basins down-stream of these wetlands, may be essential to reduce nutrient export to Lake Okeechobee for all scenarios of hydrologic conditions.

3.3.4 and 3.3.5 Phosphorus storages in biomass, litter, and soil

These tasks contained two main studies: the first investigated the P storage in biomass, litter and soil at the field-scale (in four of the wetlands at Larson and Beaty ranches) through time; the second study investigated phosphorus leaching and decomposition from plant litter material at the laboratory- and field-scale.

The specific objectives of the first task was to: (i) quantify P storages in ecosystem compartments (plant biomass, litter, and soil) of historically-isolated wetlands and surrounding improved pasture uplands, (ii) determine if a P storage gradient existed with landscape position, and (iii) evaluate the potential to increase wetland ecosystem P storage if these wetlands are hydrologically restored. Wetlands were categorized into two zones: wetland centers (deep marsh zones dominated by emergent marsh vegetation and open water), and wetland edges (shallow marsh/wet grassland zones dominated by species such as *Juncus effusus* L.). The surrounding pasture uplands were dominated by *Paspalum notatum* Flugge. It was hypothesized that a P storage gradient could be measured from wetland to surrounding upland, with P standing stocks greatest in wetland centers relative to wetland edges and surrounding pasture uplands.

Materials and methods

Study sites

Within the four priority basins of Lake Okeechobee Basin, four historically hydrologically isolated emergent marsh wetlands were studied. These wetlands were located in improved pastures of cow-calf operations (Figure 19) and the wetlands were ditched. Two wetlands were located at the Larson Dixie Ranch (N 27° 24.665', W 80° 56.940'), while the other two were located at the Beaty Ranch (N 27° 20.966', W 80° 56.465'). Ranch managers reported that typical cattle stocking rates were one cow per hectare at Larson Dixie and about 0.5 cows per hectare at Beaty, with similar amounts of inorganic fertilizer (224 kg of ammonium sulfate ha⁻¹, which contained no P) applied at both sites. Wetlands at each site were similar in size and had similar vegetation communities. Soils at Larson Dixie were classified as Siliceous, hyperthermic Spodic, Psammaquents (Basinger series), whereas soils at Beaty wetlands were classified as Sandy, siliceous, hyperthermic Typic, humaquepts (Placid series) (Lewis et al., 2001).

Site hydrology

Wetland surface water depth was monitored for three years at the four study wetlands. An internally-logging pressure transducer (Mini-troll STP, In-situ, Inc.) was deployed in a 0.032-meter diameter fully screened PVC well in the deepest location of each wetland.

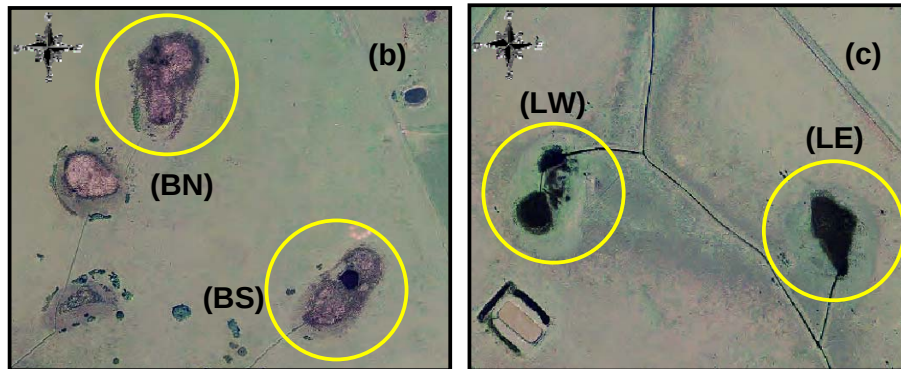
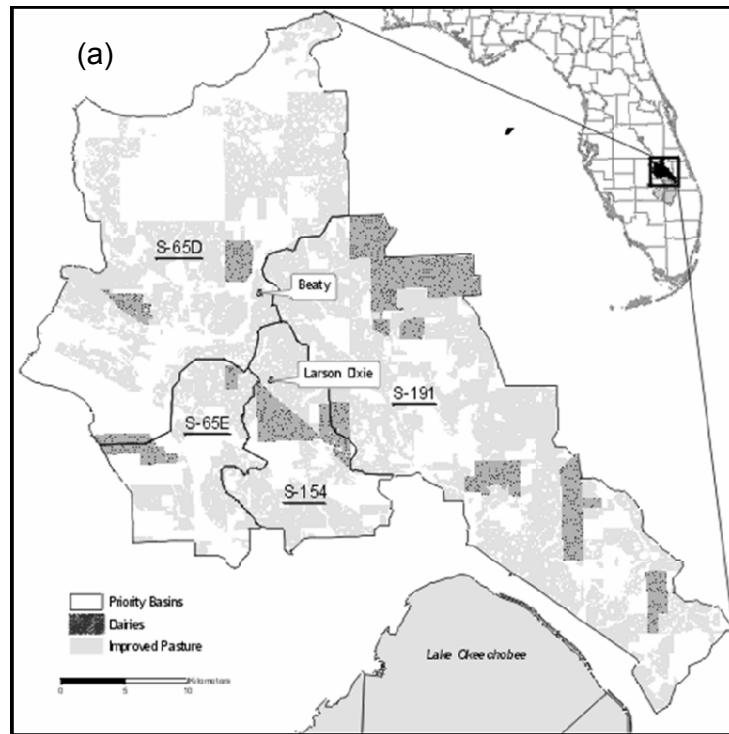


Figure 19. (a) Location of Larson Dixie and Beaty ranches in the four priority basins of Okeechobee Basin. Circles indicate historically isolated emergent marsh wetlands at Beaty Ranch (b) and Larson Dixie ranch (c). BN = Beaty North, BS = Beaty South, LW = Larson West, and LE = Larson East.

Monitoring well installation depth was between 2.5 and 3.0 meters. Wetland water surface depth was recorded continuously, at half-hour intervals. The period of record for wetland water depth was similar at the four sites, except for LE, which was monitored for five months less than the other sites (Table 23). Wetland hydroperiod was recorded as the number of days per year that wetlands had water above the ground surface at well locations.

Table 23. Monitoring period of wetland water levels for Larson Dixie East (LE) and west (LW) and Beaty north (BN) and south (BS). Net number of days represents the total number of days recorded unaffected by equipment failure.

Wetland Site	Monitoring Period		Net number days	Percent of total record
	----Start----	----End----		
LE	12/18/2003	3/10/2006	814	79
LW	7/2/2003	3/10/2006	962	93
BN	7/2/2003	04/29/2006	1011	98
BS	7/2/2003	04/29/2006	1033	100

Site sampling

Wetlands were sampled three times (November 2004, March 2005, and July 2005) using a stratified random sampling design. The November 2004 sampling, which we called the “wet period”, followed a very busy Atlantic hurricane season (August 1st until November 30th) during which our sites were hit by four hurricanes (Charley, Frances, Ivan, and Jeanne). The deep marsh, shallow marsh/wet grassland, and surrounding pasture upland zones were identified using hydrological indicators and vegetation type, similar to the criteria adopted by Van der Valk (1989). Deep marsh zones were dominated by open water; vegetation species included *Pontedaria cordata* var. *lancifolia* (Muhl.) Torr., *Bacopa monnieri* (L.) Pennell, *Panicum hemitomon* Schult., *Polygonum* sp., and *Ludwigia repens* Forst. Shallow marsh/wet grassland zones had evidence of inundation, but were periodically dry during sampling. Plant species included *Juncus effuses* L., *Eleocharis baldwinii* (Torr) Chapm, *Paspalum acuminatum* Raddi, and *Hydrochloa caroliniensis* Beauv. The pasture upland was predominantly dry and dominated by *Paspalum notatum* Flugge.

Sample analyses

Above-ground biomass, litter, below-ground biomass and soil samples were collected from a 1 x 1 m quadrat at each location within each sampling zone. Above-ground plant biomass was sorted by species and then into live and dead fractions. All above-ground plant biomass, below ground biomass and litter samples were oven dried. Samples were

then weighed, ground and sieved. Ground and sieved samples were stored for analyses. Soils characteristics determined included soil water content, bulk density, pH, inorganic P (1 M HCl extractable). To help understand how labile or recalcitrant P stores in soil were, we determined inorganic and organic P fractions in wetland surface soils (0-10 cm). Soil P was fractionated using the scheme devised by Ivanoff et al. (1998) and later described in detail by Pant and Reddy (2001). All plant biomass, litter and soil samples were analyzed for total phosphorus (TP), total carbon (TC), and total nitrogen (TN).

Data and statistical analyses

Data distributions were tested for normality. If data was not normally distributed prior statistical analyses, it was log transformed. Statistical analyses were conducted on transformed data and significant differences were determined at the $P < 0.05$ level or lower. Using an analysis of variance (ANOVA), we compared: hydroperiod between deep marsh, shallow marsh and upland areas; nutrient concentrations in above-ground plant biomass; nutrient concentrations of above-ground plant biomass; and P storages in ecosystem compartments. Nitrogen (N) to P ratios were calculated as the reciprocal of the linear slope between N and P concentrations of above-ground plant biomass, plant litter, and soil. We used ANOVA to determine whether the linear regression between N and P concentrations were significant at the $P < 0.05$, and 0.01 levels. Further, we used linear regressions and ANOVA to determine significant relationships between soil total P and soil organic matter as measured by loss on ignition (LOI).

Results

Wetland hydrology

Wetland hydroperiod based on the water levels recorded in the well at the deepest portion of the wetland was 241, 262, 247, and 315 days for LE, LW, BS, and BN respectively. Average wetland hydroperiods were progressively longer for lower-elevation wetland zones, such that deep marshes had longer hydroperiods than surrounding shallow marsh/wet grassland zones, which in turn, had longer hydroperiods than surrounding pasture uplands (Figure 20). The Beaty North deep marsh zone had a significantly longer hydroperiod relative to Larson Dixie deep marsh zones ($P < 0.05$; Figure 20). Beaty shallow marsh/wet grassland zones had longer hydroperiods than similar zones at Larson Dixie ($P < 0.05$). Figure 20 also shows that the hydroperiod of LW deep marsh zones were similar to shallow marsh/wet grassland zones of all the other sites. Upland zones had the shortest hydroperiods (~ 20 days) ($P < 0.05$).

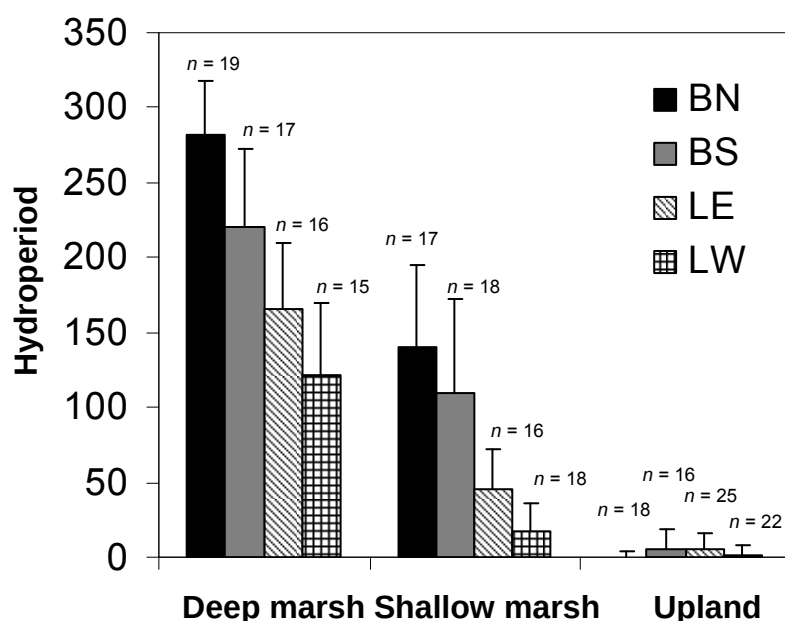


Figure 20. Hydroperiod of wetland zones and surrounding uplands of Beaty and Larson Dixie sites that were monitored during 2004 and 2005. BN = Beaty North, BS = Beaty South, LE = Larson East, and LW = Larson West.

Phosphorus concentrations in above-ground biomass

Live plant material had significantly greater P concentrations compared to dead plant material ($P < 0.05$; Table 24). Phosphorus and nitrogen concentrations also varied between plant species. *Juncus effusus*, *Panicum hemitomon*, and *Andropogon glomeratus* tended to have lower P concentrations relative to other species such as *Paspalum accuminatum*, *Althernanthera philoxeroides*, and *Polygonum* sp. (Table 24).

During November 2004 and March 2005, P concentrations in above-ground plant biomass collected in deep marsh zones was significantly greater than the P concentrations in above-ground plant biomass in shallow marsh/wet grassland areas, which in turn, was significantly greater than the P concentrations in above-ground plant biomass in pasture uplands. During July 2005 there were significantly lower P concentrations in above-ground plant biomass at both Beaty wetlands relative to Larson Dixie wetlands ($P < 0.01$).

Biomass and litter characteristics

During November 2004, above-ground plant biomass (g m^{-2}) was similar between wetland and upland zones and similar between the four different sites (Table 24). However, during March and July (2005) above-ground plant biomass in Beaty wetlands was greater than the above-ground plant biomass in Larson Dixie wetlands ($P < 0.05$). At

Larson Dixie the pasture uplands had greater above-ground plant biomass than wetland areas ($P < 0.05$); whereas at Beaty, wetlands had significantly greater above-ground plant biomass than their surrounding uplands.

At Beaty sites, plant litter collected in November 2004 was greatest in deep marsh areas and least amounts of litter occurred in pasture uplands ($P < 0.05$). However, this was not the case during the other two sampling events. At Larson Dixie, wetland zones contained least amounts of plant litter compared to their surrounding pasture uplands during March and July (2005), whereas in November 2004, they contained similar amounts.

In general, below-ground plant biomass (g m^{-2}) was much greater than above-ground plant biomass (Table 25; $P < 0.01$). Below-ground plant biomass was greater at Beaty than it was at Larson Dixie ($P = 0.046$). The relative partitioning between above- and below-ground plant biomass was greatest at Larson Dixie sites, where below-ground plant biomass was about eleven times greater than above-ground biomass; while at Beaty, it was eight times greater. Table 24 also illustrates that below-ground plant biomass was greatest prior the wet period (November, 2004), whereas after it, below-ground plant biomass decreased.

Nutrient ratios of ecosystem components

The nitrogen to P ratio was lower in above-ground plant biomass relative to litter and soil (Figure 21). The N and P concentrations in above-ground plant biomass in wetland zones were greater than the N and P concentrations in above-ground plant biomass in uplands ($P < 0.05$). There was no relationship between N and P concentrations of below-ground plant biomass tissue.

Table 24. Phosphorus, nitrogen and carbon concentrations of live and dead vegetation in the different landscape zones. Values are means $\pm$ one standard error.

Species	Zone	Life stage	-----%-----					
			Phosphorus		Nitrogen		Carbon	
<i>Alternanthera philoxeroides</i>	Deep marsh	Live	0.45	± 0.02	2.97	± 0.16	38.72	± 0.7
<i>Eleocharis baldwinii</i>	Deep marsh	Live	0.22	± 0.08	2.3	± 0.69	42.08	± 1
<i>Ludwigia repens</i>	Deep marsh	Live	0.185	± 0.01	1.3	± 0.17	42.38	± 1.1
<i>Panicum hemitomom</i>	Deep marsh	Dead	0.098	± 0.01	1.39	± 0.06	44.19	± 0.3
<i>Panicum hemitomom</i>	Deep marsh	Live	0.2	± 0.01	1.6	± 0.10	42.98	± 0.4
<i>Paspalum accuminatum</i>	Deep marsh	Dead	0.287	± 0.03	2.26	± 0.10	34.41	± 1.7
<i>Paspalum accuminatum</i>	Deep marsh	Live	0.338	± 0.01	2.58	± 0.10	39.47	± 0.6
<i>Polygonum</i> sp.	Deep marsh	Dead	0.2	± 0.01	1.89	± 0.10	42.5	± 0.7
<i>Polygonum</i> sp.	Deep marsh	Live	0.306	± 0.02	1.91	± 0.12	41.89	± 0.6
<i>Pontedaria cordata</i>	Deep marsh	Dead	0.143	± 0.01	1.74	± 0.15	42.58	± 0.7
<i>Pontedaria cordata</i>	Deep marsh	Live	0.297	± 0.02	1.95	± 0.11	39.98	± 0.5
<i>Juncus effusus</i>	Shallow marsh	Dead	0.077	± 0.01	1.14	± 0.04	44.22	± 0.2
<i>Juncus effusus</i>	Shallow marsh	Live	0.143	± 0.01	1.51	± 0.07	45.41	± 0.9
Other	Shallow marsh	Dead	0.164	± 0.01	1.69	± 0.08	43.49	± 0.2
Other	Shallow marsh	Live	0.236	± 0.01	1.83	± 0.07	42.01	± 0.4
<i>Andropogon glomeratus</i>	Upland	Dead	0.06	± 0.02	0.88	± 0.19	44.26	± 0.4
<i>Andropogon glomeratus</i>	Upland	Live	0.111	± 0.04	1.23	± 0.31	44.31	± 0.1
<i>Paspalum notatum</i>	Upland	Dead	0.111	± 0.01	1.34	± 0.04	42.77	± 0.2
<i>Paspalum notatum</i>	Upland	Live	0.19	± 0.00	1.68	± 0.03	42.48	± 0.1

Table 25. Above-ground biomass, litter, and below-ground biomass of sampled wetland zones (deep marsh and shallow marsh/wet grassland) and surrounding pasture uplands of Beaty and Larson Dixie sites. BN = Beaty North, BS = Beaty South, LE = Larson East, and LW = Larson West.

Date	Zone	BN		BS		LE		LW	
-----g m ⁻² -----									
-----Above-ground biomass-----									
11/1/2004	Deep marsh	502	± 76	629	± 137	495	± 158	346	± 108
	Shallow marsh	235	± 81	455	± 85	207	± 76	162	± 24
	Pasture upland	242	± 50	264	± 39	277	± 48	248	± 66
3/1/2005	Deep marsh	292	± 97	379	± 150	76	± 38	52	± 14
	Shallow marsh	704	± 172	497	± 82	37	± 8	27	± 13
	Pasture upland	295	± 45	193	± 55	213	± 49	137	± 33
7/1/2005	Deep marsh	341	± 56	306	± 40	76	± 50	42	± 28
	Shallow marsh	533	± 136	402	± 69	59	± 29	6	± 2
	Pasture upland	244	± 35	265	± 62	192	± 20	233	± 48
-----Litter-----									
11/1/2004	Deep marsh	181	± 55	323	± 79	329	± 169	369	± 142
	Shallow marsh	225	± 87	238	± 50	206	± 56	198	± 35
	Pasture upland	70	± 16	156	± 27	190	± 23	184	± 38
3/1/2005	Deep marsh	204	± 62	301	± 88	549	± 89	352	± 56
	Shallow marsh	128	± 34	250	± 41	45	± 13	32	± 11
	Pasture upland	192	± 39	250	± 33	260	± 47	253	± 64
7/1/2005	Deep marsh	586	± 141	186	± 19	53	± 12	177	± 62
	Shallow marsh	248	± 143	207	± 59	14	± 10	24	± 15
	Pasture upland	101	± 33	252	± 91	196	± 108	198	± 67
-----Below-ground biomass-----									
11/1/2004	Deep marsh	862	± 319	977	± 253	216	± 69	643	± 452
	Shallow marsh	968	± 232	643	± 154	928	± 222	1213	± 396
	Pasture upland	1336	± 251	1641	± 166	1521	± 186	2441	± 784
3/1/2005	Deep marsh	2204	± 254	1791	± 413	1072	± 412	683	± 227
	Shallow marsh	2178	± 428	2133	± 352	1381	± 228	663	± 164
	Pasture upland	1860	± 482	1732	± 50	2070	± 297	890	± 116
7/1/2005	Deep marsh	2464	± 625	1662	± 470	622	± 320	313	± 134
	Shallow marsh	1121	± 525	1872	± 175	954	± 294	1792	± 249
	Pasture upland	1934	± 266	1423	± 193	3595	± 885	2048	± 289

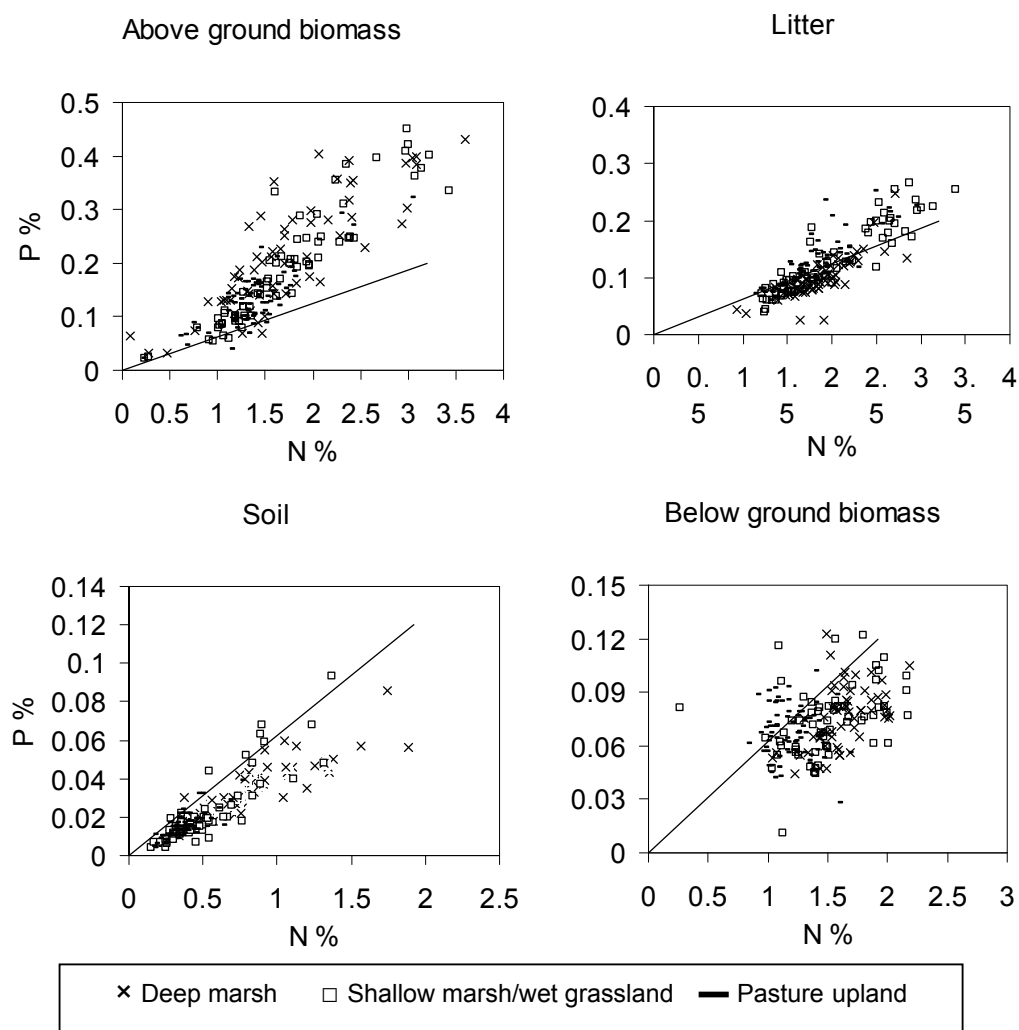


Figure 21. Graph of significant linear relationships between nitrogen (N) and phosphorus (P) concentrations in (a) above-ground plant biomass, (b) litter, (c) soil, and (d) below-ground plant biomass. Diagonal line in each graph represents an N:P ratio of 16. The N:P values in each graph are the reciprocal of the significant linear regression relationship between N and P concentrations.

Phosphorus storage

Phosphorus storage in surface soils (0-10 cm) was greatest (more than 87%; $P < 0.001$) relative to the sum of all other ecosystem compartments (above-ground plant biomass, plant litter, and below-ground plant biomass) (Figure 22). Wetland soils tended to have greater P stores than surrounding upland soils. Larson Dixie deep marsh soils contained more P than Beaty wetland deep marsh soils ($P = 0.004$). Shallow marsh/wet grassland and upland soils stored similar amounts of P across and between sites; however, at BN, shallow marsh soils stored more P than deep marsh and upland soils.

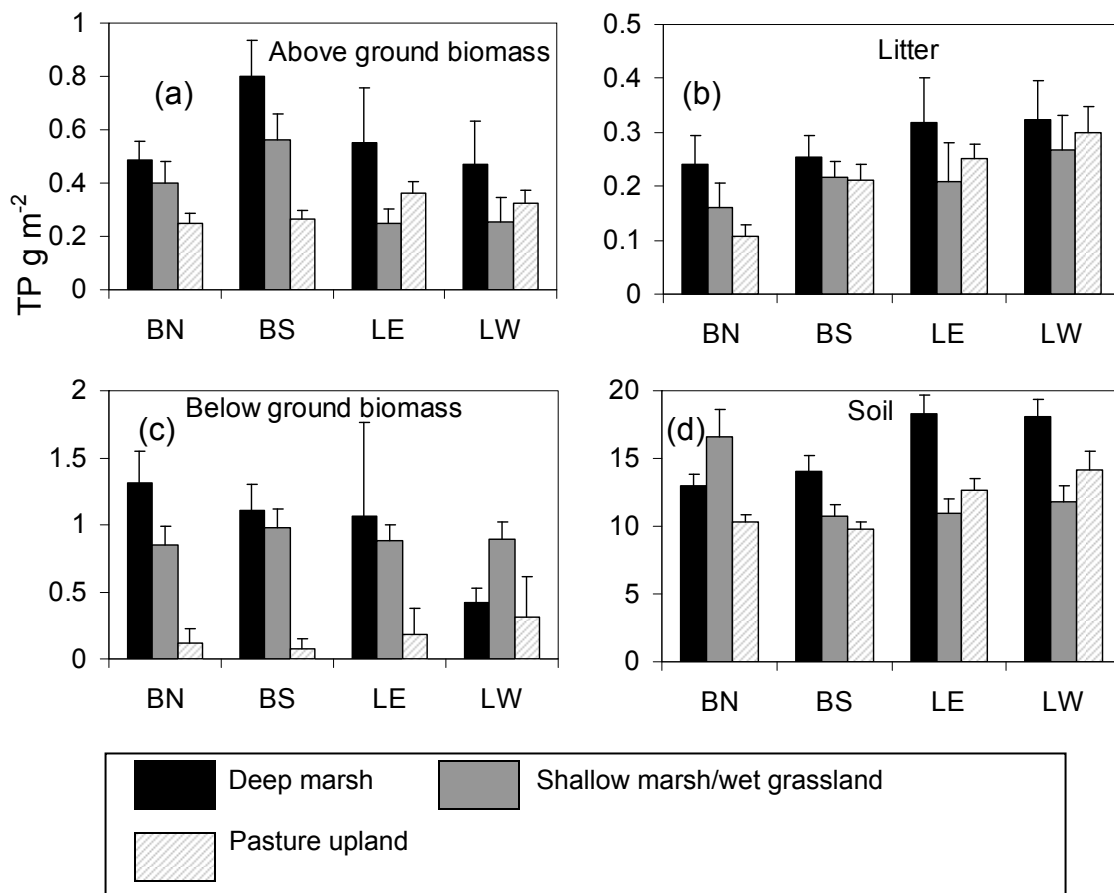


Figure 22. Zonal phosphorus storage in standing stocks of above-ground plant biomass (a) plant litter (b), below-ground plant biomass (c) and soil (d) of wetlands and surrounding uplands at Beaty and Larson Dixie sites. Sites sampled three times between November 2004 and July 2005. Error bars represent one standard error. ($n = 18$).

After soils, P storage was greatest in below-ground plant biomass, which was greater than above-ground plant biomass, and least in plant litter. In general, P storage in above-ground biomass was greatest in wetlands relative to their surrounding uplands. At Beaty,

there was a similar P storage gradient in plant litter, with litter collected from deep marsh zones containing significantly greater amounts of P relative to shallow marsh/wet grassland zones, which had greater P storages relative to uplands. At Larson Dixie, this P storage gradient in plant litter did not exist (Figure 22). There was greater P storage in litter collected from uplands relative to shallow marsh/wet grassland zones.

Below-ground biomass of Beaty wetlands stored more P, relative to below-ground biomass in their surrounding uplands (Figure 22). At LE, the typical trend of greater P storage in wetland zones relative to uplands existed, with a lesser P storage gradient at LW.

Relationships between soil phosphorus storage and soil chemical characteristics

Between sampling periods, soil characteristics were relatively uniform; therefore, we present results by zone and site (Table 26). Typically, deep marsh soils of Beaty sites were wetter, had greater amounts of organic matter, were more acidic and had lower bulk density than soils at Larson Dixie ($P < 0.05$). Similar patterns existed for shallow marsh/wet grassland soils, with Beaty shallow marsh/wet grassland soils being significantly wetter and having lower bulk densities than similar soils at Larson Dixie. Larson Dixie deep marsh soils had greater storages of total nitrogen (TN) and inorganic P (as extracted with 1 M HCl) than Beaty ($P < 0.05$). Similarly, upland soils at Larson Dixie contained greatest amounts of inorganic P relative to upland soils at Beaty (Table 26). We found that deep marsh soils had greater organic matter content than shallow marsh/wet grassland soils or surrounding upland soils ($P < 0.001$). There were significant relationships between soil total P and soil organic matter for wetland soils, and upland soils (Figure 23); however, the linear relationship between upland soil total P and soil organic matter although significant, was not as good, compared to wetland soils.

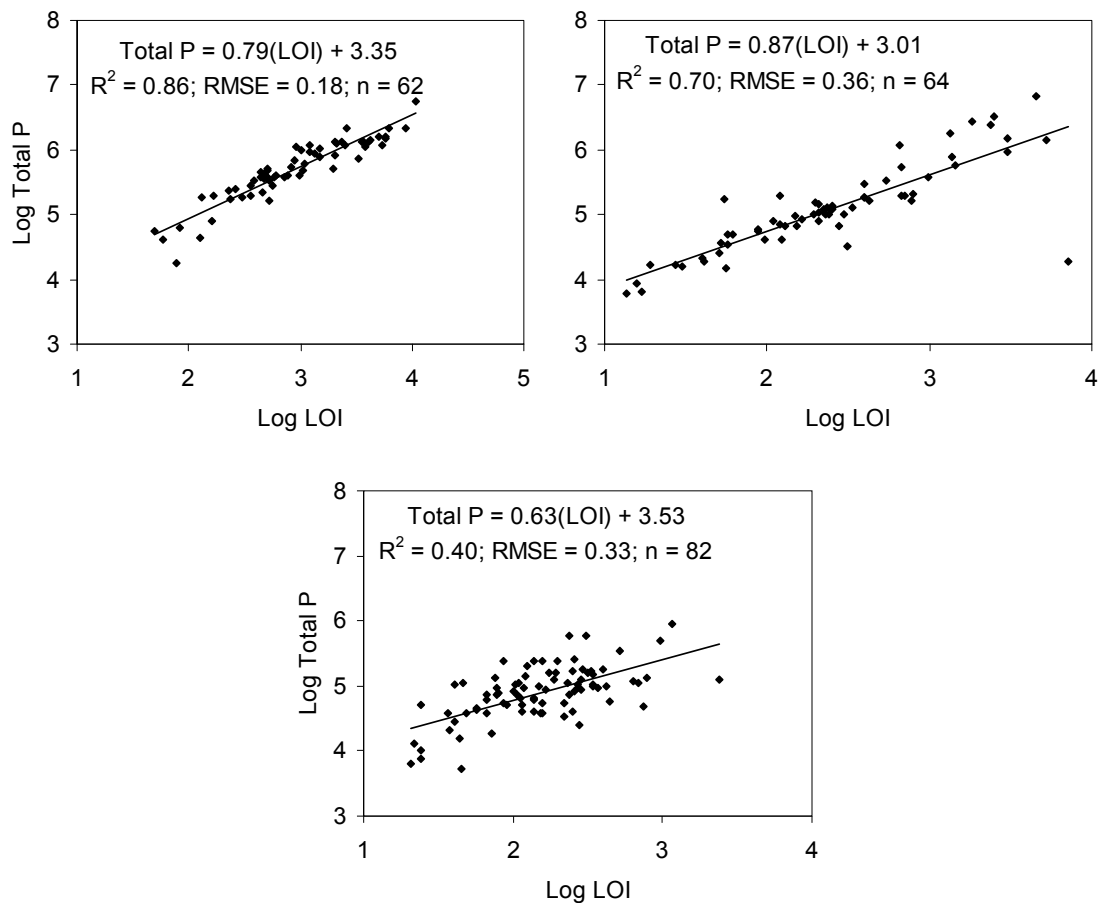


Figure 23. Relationship between soil total phosphorus (mg kg^{-1}) and soil organic matter, as measured by loss on ignition (LOI) %. Values are represented as log scale. All relationships between total P and LOI are significant at the $P < 0.001$ level.

There were no significant differences between inorganic and organic P fractions of wetland surface soils at the different sites (Figure 24). We found total inorganic P fractions and labile organic P fractions were about 8% - 12% of soil total P. The greatest organic P fractions in wetland surface soils were microbial biomass P (MBP) and fulvic acid bound P (FAP). Microbial biomass P was about 21% of soil total P; whereas FAP was greater, being 33% of soil total P. On average, unavailable organic P fractions (humic acid bound P) accounted for 14% of soil total P, whereas highly recalcitrant organic P or P associated with minerals (Res-P) represented 12% of soil total P. We found that deep marsh soils had greater organic matter content than shallow marsh/wet grassland soils or surrounding upland soils ($P < 0.001$). There were significant relationships between soil total P and soil organic matter for wetland soils, and

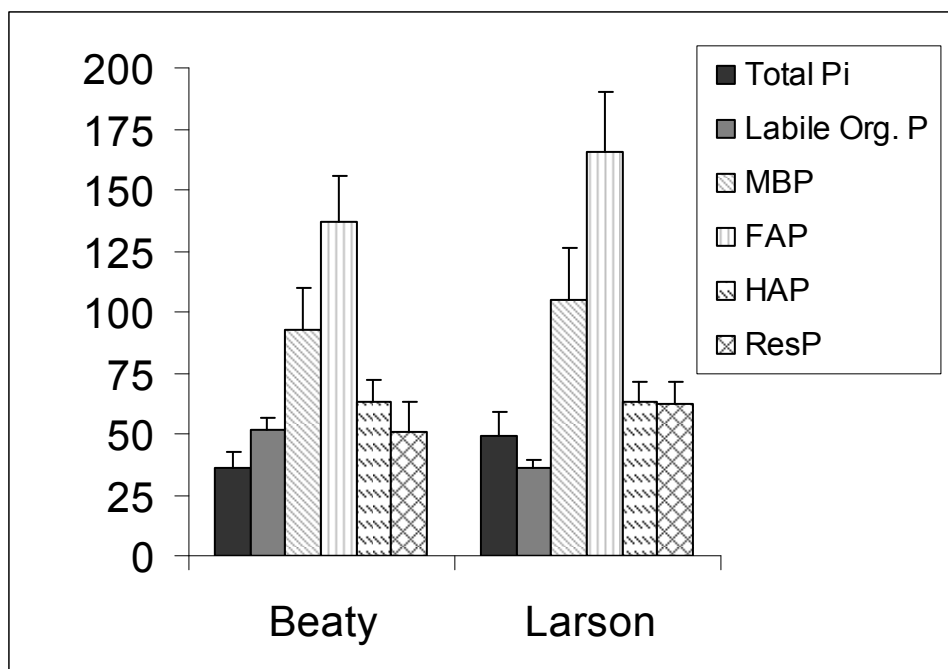


Figure 24. Phosphorus fractions in wetland surface soils (0-10 cm) of Larson Dixie and Beaty wetlands. Total Pi = total inorganic P (SRP in nonfumigated soil extracted with NaHCO_3 + SRP in HCl extract); Labile Org. P = labile organic P (TP in nonfumigated soil extracted with NaHCO_3 - SRP in nonfumigated soil extracted with NaHCO_3); MBP = microbial biomass P (TP in fumigated extract - TP in nonfumigated extract); FAP = fulvic acid bound P (TP in pretreated NaOH extract); Humic acid bound P (TP in NaOH extract - TP in pretreated NaOH extract); and ResP = residual P.

Discussion

Site hydrology

In general, Beaty wetlands had longer hydroperiods than Larson Dixie wetlands ($P < 0.05$). The average hydroperiod of the deep marsh zone at LW was more similar to other wetland shallow marsh/wet grassland zones, suggesting LW was not as wet as other sites. The ground surface elevation for most of the wetland area at both Larson Dixie and Beaty wetlands was below ditch ground surface elevation, suggesting that a sufficient depth of water must accumulate in the wetland before there is any discharge to the ditch. At each wetland, the minimum wetland water depth required for ditch export of wetland surface

Table 26. Soil physicochemical characteristics of wetland zones and surrounding pasture upland soils collected during 2004 and 2005. BN = Beaty North, BS = Beaty South, LE = Larson East, and LW = Larson West. Values are means $\pm$ one standard error ($n = 19$).

Zone	Site	Water Content ----%----	Bulk Density -----g cm ⁻³ ----	pH	LOI ---%---	TN -----g m ⁻² -----	TC -----g m ⁻² -----	1 M HCl P
Deep marsh	BN	62 $\pm$ 3	0.424 $\pm$ 0.04	4.9 $\pm$ 0.1	25 $\pm$ 3	304 $\pm$ 11	4365 $\pm$ 209	1.87 $\pm$ 0.1
	BS	58 $\pm$ 4	0.463 $\pm$ 0.05	5.0 $\pm$ 0.1	28 $\pm$ 3	357 $\pm$ 26	5257 $\pm$ 437	2.28 $\pm$ 0.2
	LE	49 $\pm$ 2	0.602 $\pm$ 0.03	5.3 $\pm$ 0.0	18 $\pm$ 2	429 $\pm$ 29	5453 $\pm$ 410	2.98 $\pm$ 0.2
	LW	45 $\pm$ 2	0.660 $\pm$ 0.03	5.4 $\pm$ 0.1	15 $\pm$ 2	409 $\pm$ 26	4974 $\pm$ 411	2.96 $\pm$ 0.2
Shallow marsh/ wet grassland	BN	48 $\pm$ 3	0.628 $\pm$ 0.05	4.9 $\pm$ 0.1	17 $\pm$ 3	323 $\pm$ 19	4442 $\pm$ 360	2.98 $\pm$ 0.4
	BS	45 $\pm$ 3	0.666 $\pm$ 0.06	5.1 $\pm$ 0.1	16 $\pm$ 2	319 $\pm$ 20	4359 $\pm$ 328	1.95 $\pm$ 0.1
	LE	34 $\pm$ 2	0.836 $\pm$ 0.03	5.3 $\pm$ 0.1	9 $\pm$ 1	321 $\pm$ 23	3503 $\pm$ 335	1.60 $\pm$ 0.2
	LW	30 $\pm$ 2	0.867 $\pm$ 0.02	5.2 $\pm$ 0.0	9 $\pm$ 1	332 $\pm$ 23	3560 $\pm$ 327	3.03 $\pm$ 0.4
Pasture upland	BN	28 $\pm$ 1	0.804 $\pm$ 0.02	4.8 $\pm$ 0.1	12 $\pm$ 1	291 $\pm$ 10	4660 $\pm$ 268	2.14 $\pm$ 0.2
	BS	27 $\pm$ 2	0.841 $\pm$ 0.03	5.1 $\pm$ 0.1	9 $\pm$ 1	272 $\pm$ 13	3427 $\pm$ 267	1.93 $\pm$ 0.1
	LE	29 $\pm$ 2	0.844 $\pm$ 0.02	5.4 $\pm$ 0.0	8 $\pm$ 1	273 $\pm$ 15	2954 $\pm$ 211	3.20 $\pm$ 0.5
	LW	30 $\pm$ 2	0.828 $\pm$ 0.02	5.1 $\pm$ 0.1	10 $\pm$ 1	316 $\pm$ 17	3526 $\pm$ 241	5.04 $\pm$ 1.0

water was between 30 and 50 cm. Therefore, restoring hydrology by ditch management (blocking ditches to hold water within the wetland) may only be effective at wetland water depths greater than these.

Phosphorus concentrations in above-ground plant biomass

Above-ground plant biomass tended to have greater tissue P concentrations than upland vegetation. Boyd (1978) suggests similar ranges for plant tissue P ranging between 0.1 and 0.4 % of dry weight. This study investigated a range of different wetland plant types including submersed, floating and emergent herbaceous vegetation. McJannet et al. (1995) report somewhat similar tissue P concentrations ranging between 0.13 and 1.07 % dry weight. They also indicated total above-ground biomass ($\text{g dry weight m}^{-2}$) did not vary with P concentrations in plant tissue. Knowing tissue P concentrations in above-ground biomass could be useful for measuring the effectiveness of restoration (Whigham et al., 2002), as increases in tissue P concentration may indicate a greater potential for plant P storage.

N:P ratios

Above-ground plant biomass in wetlands and uplands were N limited, as both above-ground plant biomass and plant litter had lower N:P ratios than 16 (Figure 21). Koerselman and Meuleman (1996) determined that vegetation with an N:P ratio > 16 were P limited, while a N:P ratio less than 14, suggests vegetation was N limited. In a study similar to ours, Gathumbi et al. (2005) also suggested above-ground plant biomass of wetlands in subtropical improved and unimproved pastures of south Florida were N limited. Nitrogen to P ratios have implications for hydrologic restoration, as at the vegetation community level, if N is limiting, P uptake by vegetation may be increased by adding N. This may have positive short-term effects on water quality by increasing plant P uptake. However, P storage by vegetation is regarded as short-term storage, with substantial amounts of P being released at the end of the growing season (Richardson and Marshall, 1986) and/or during winter dry periods (Reddy and DeBusk, 1991).

Typically, marshes have lower N:P ratios in live plant biomass relative to live plant biomass in bogs, fens and swamps (Bedford et al., 1999). Bedford et al. (1999) investigated nutrient availability in temperate US wetlands and suggested plant litter had higher N:P ratios relative to live biomass. Plant litter P concentrations ranged between 0.004 and 0.64 %. Similarly, we found higher N:P ratios in litter relative to above-ground biomass.

Walbridge (1991) reported that the N:P ratios of organic soils they collected decreased with increasing P availability. Nitrogen to P ratios for organic soils in their study ranged between 17 and 57. Our study indicated that N:P ratios of historically-isolated wetland soils were on the low end of this range. This may be because wetlands on more mineral soils tend to have lower N:P ratios than those on more organic soils (Bedford et al., 1999). Bedford et al. (1999) found temperate marsh soils to have a N:P ratio of 9, whereas bogs and fens had slightly higher N:P ratios of 24. We hypothesize that

hydrologic restored wetlands should have greater N:P ratios in surface soils relative to un-restored wetland soils, as P availability in soil may decrease. Therefore, knowing N:P ratio of soils may be useful, to help evaluate the effect of hydrologic restoration on P storage.

Surprisingly, we found few relationships between the N:P ratios of the different ecosystem compartments. For example, there was no correlation between the N:P ratio in soil relative to the N:P ratio in plant litter. Klopatek (1978) suggested that attempts to relate levels of nutrients in soil and water to emergent macrophytes can result in weak correlations, due to environmental factors including seasonal changes in concentrations, nutrient regimes, and loss of nutrients to the water column.

Plant biomass

After the wet period (November 2004), there was little difference in above-ground plant biomass between wetlands and uplands (Table 25). During the study, biomass ranged between 6 and 533 g m⁻². For emergent marsh tidal wetlands in the mid-Atlantic coastal region (USA), standing stock biomass is least in species such as *Pontedaria* sp. (682 g dry wt. m⁻²), and greatest in species such as *Typha* sp. (1,215 g dry wt. m⁻²) and *Zizania* sp. (1,218 g dry wt. m⁻²) (Whigham et al., 1978).

In our study, above-ground plant biomass was greater in Beaty wetlands relative to their surrounding uplands; however, during dry periods (March 2005 and July 2005) the reverse was true at Larson Dixie. We hypothesize that these reverse trends exist at Larson Dixie due to cattle grazing, as we observed (but did not measure) a high intensity of cattle grazing in shallow marsh/wet grassland areas at Larson Dixie. Therefore, it may be possible that cattle grazing reduced above-ground biomass at Larson Dixie, thereby limiting source material for litter production.

Relative to above-ground biomass, below-ground biomass can be similar to or greater than above-ground biomass (Van der Valk and Davis, 1978). Below-ground plant biomass in this study was much greater than above-ground plant biomass, with the magnitude of difference being greater at Larson Dixie wetlands relative to Beaty, which was probably magnified by cattle grazing effects at Larson Dixie.

For Iowa prairie marshes and other freshwater wetlands mostly in northern wetlands of USA, researchers report below-ground plant biomass values can range between 130 and nearly 2,000 g dry wt. m⁻² (Bernard and Gorham, 1978; and Van der Valk and Davis, 1978). Bernard and Gorham (1978) report a below-ground plant biomass production rate for sedge dominated wetlands of about 200 g m⁻² yr⁻¹. Relative to these studies, our measurements suggest emergent marsh wetlands in improved grazing pastures of south Florida have relatively high below-ground plant biomass.

Plant biomass and litter phosphorus storage

Phosphorus storage in above-ground plant biomass was greatest in wetland areas relative to uplands suggesting that there was a P storage gradient. Similarly, Whigham et al. (2002) observed P storage gradients in restored wetlands of Maryland's (USA) coastal plain. Phosphorus storages in above-ground plant biomass was similar ($0.2 - 0.5 \text{ g m}^{-2}$) and they observed greater P storage in temporary and emergent/seasonally inundated areas relative to permanently flooded areas.

Plant litter P stores in our study were about half of what Kadlec and Knight (1996) report for an enriched peatland (1.25 g P m^{-2}). In enriched wetlands relative to unenriched wetlands, increases in plant litter P concentrations can increase decomposition rates (Qualls and Richardson, 2000). These findings have implications for hydrologic restoration, as with restoration, deep marsh zones would potentially increase in area. Above-ground plant biomass and plant litter collected from deep marsh zones had greatest stores of P, which implies if these deep marsh zones increased in area, so would P storage, thereby increasing decomposition rates. Decomposition rates affect organic matter accretion, which provides long-term storage of P (Pant and Reddy, 2001).

Phosphorus storage in below-ground plant biomass was greatest for wetlands relative to uplands, but not always, with P storage in below-ground biomass being variable through time. Prentki et al. (1978) in their studies of lakeshore marshes dominated by *Typha latifolia* in Wisconsin USA determined maximum P standing stocks of 4.3 g m^{-2} . With the onset of the growing period, nearly a third of total P stored in below-ground biomass was translocated to above-ground parts, while with the onset of plant senescence; P was transported back to below-ground portions. Translocating P from above-ground to below-ground tissue after a growing cycle can range between 20 and 75% of P stored in above-ground biomass (Prentki et al., 1978; Wetzel, 2001; Reddy et al., 1999).

Little if no information exists on the impact of hydrologic restoration on below-ground plant biomass P storage and the translocating of P from below to above biomass and visa versa. We hypothesize that hydrological restoration may increase P storage in below-ground plant biomass by increasing wetland area, as there were greater P stores in below-ground plant biomass collected from most wetland zones.

Soil phosphorus storage

Our findings, in addition to others, suggest soil is the most important ecosystem storage compartment for long-term P storage (Reddy et al., 1999; and Bridgham et al., 2001). In the past, wetland mesocosm mass balance studies reported soils stored about 68% of total ecosystem P stores (Dolan et al., 1981). Further, at Houghton Lake fen (Michigan, USA) soil (peat to a depth of 20 cm) stored greater than 97% of P, relative to above-ground biomass and plant litter (Richardson et al., 1978). In general, our study found that wetland surface soils (0-10 cm) contained more P than surrounding upland soils, implying greater P storage as a function of landscape position. Organic matter seemed to control total P content of wetland soils. Axt and Walbridge (1999) reported good relationships between P sorption of palustrine forested wetland surface soils (0-15cm)

and soil organic matter ($R^2 = 0.78$). They also investigated a P gradient and reported forested wetland soils had higher P sorption capacities than streambank and surrounding upland soils. Larson Dixie upland soils had greater amounts of inorganic P (extracted with 1 M HCl) relative to soil total P (24-29%), in comparison to Beaty upland soils (20-21%), which suggests Larson Dixie soils were impacted by greater amounts of nutrients, possibly a result of greater grazing pressure.

Microbial biomass P and FAP were slightly greater in Larson Dixie wetland soils relative to Beaty. Microbial biomass P represents a bioavailable fraction of soil organic P, whereas fulvic acid bound P, which is plant derived, is moderately available (Reddy et al., 1999).

Both total inorganic P and labile organic P were less than 13% of soil total P, which is high relative to other studies. Reddy et al. (1998) reported labile inorganic P fractions ranged between 1 and 4% of soil total P, for surface soils of different hydrological units in Water Conservation Areas of Florida's Everglades. Walbridge (1991) found only a small part of soil total P being in an extractable form (~ 3%) in organic soils along a landscape continuum. Although labile soil P fractions may only represent a small fraction of soil total P, they can be responsible for most of the P flux from soil to overlying water (Pant and Reddy, 2001). Labile soil P fractions such as water soluble P can help explain a large amount of the variability in P flux from soil to water; however, other factors including soil physicochemical characteristics and P concentration in overlying water affect P flux from soil to overlying water (Pant and Reddy, 2003).

In general, our study suggests that most P in emergent marsh wetland soils was stored in organic forms and about 58% of total P may be slowly available or recalcitrant; thereby being relatively unavailable to overlying wetland waters. These findings are somewhat similar to Reddy et al. (1998). They found 37-70% of P was stored in organic forms for surface soils (0-10 cm) collected from different hydrological units of the Everglades. In other wetland systems such as natural and recently restored peatlands, 67% of soil total P can be present in recalcitrant forms such as humic acid bound P and residual P (Graham et al., 2005). Phosphorus forms stored in the more recalcitrant forms such as residual P can represent long-term storage; however, the proportion of residual P stored can be affected by variations in hydrology, with dry conditions having the potential to decrease residual P proportions (Reddy et al., 1998).

Typically, wetland soils are characterized by slow turnover of organic matter, due to flooding (Fisher, and Reddy, 2001), as flooding slows decomposition. Under hydrologic restored conditions, duration and frequency of wetland flooding would increase; therefore, decomposition of organic material and release of organic bound nutrients would also decrease (McLatchey and Reddy, 1998). Further, flooding would promote organic matter accretion rates, which typically is the controlling factor for long-term P storage (Richardson and Marshall, 1986).

Phosphorus storage and hydrologic restoration

In the absence of any post hydrologic wetland restoration data, we present a hypothetical scenario of the potential effect restoration may have at Larson Dixie and Beaty wetlands (Table 27). We increased wetland area by 5%, 10% and 20% and determined increased total P storage capacity (P stored in plant biomass, plant litter and soil), as the difference between present ecosystem P storages of wetlands relative to uplands. We multiplied this by the increase in wetland area at each individual site. Relative to present ecosystem total P stores in wetlands, increased total P storage capacity ranged between 0.5 and 9%, with no P storage being potentially gained at LW.

Table 27. Potential areal P storage capacity of wetlands. Values represent means $\pm$ one standard error.

Site	Area	Wetland		Upland		Additional P storage capacity with increased wetland area¶					
	-ha-	Total P storage†				5%		10%		20%	
		-----kg ha ⁻¹ -----									
Beaty North	1.5	161	10	118	± 5	3.3	± 0.9	6.3	± 1.5	13.3	± 3.0
Beaty South	1.1	142	14	114	± 4	1.7	± 0.7	3.0	± 1.0	6.3	± 2.3
Larson East	1.1	160	15	148	± 17	0.7	± 0.9	1.7	± 2.0	2.7	± 4.3
Larson West	2.2	162	22	167	± 27	-0.7	± 0.7	-1.3	± 1.3	-2.7	± 2.7

† Total P storage is the sum of P storages in all ecosystem compartments (above-ground plant biomass, litter, below-ground plant biomass and soil (0-10 cm)).

¶ P storage capacity with increased wetland area was calculated as the difference between total P storage of wetland relative to upland, multiplied by a 5, 10, and 20% increase in wetland area.

Conclusions

During this study, wetland hydroperiods recorded in the deepest location of the wetland suggested that hydroperiods were somewhat similar between wetlands. Most of the P stored within historically isolated wetlands is stored in soil relative to other ecosystem compartments such as plant biomass, and plant litter. In general, a P storage gradient existed between wetland zones and surrounding uplands, as wetlands tended to store more P than surrounding uplands.

Above-ground wetland plant biomass had greater tissue P concentration than surrounding upland plant biomass, which suggests that hydrologic restoration (by increasing wetland area) could increase P storage in above-ground plant biomass. Above-ground plant biomass was N limited; therefore, adding N to facilitate P uptake by plant biomass, may also be beneficial for increased P storage. This increase should be considered short-term.

Wetland soils contained more organic matter than surrounding upland soils and there was a good relationship between wetland soil total P and soil organic matter. The greater amounts of plant biomass in wetland zones and the subsequent accumulation of plant litter is critical for long-term P storage. Soil organic matter accumulation in our studied

wetlands may be impacted by cattle. Wetland restoration efforts need to take into account ecosystem disturbance by cattle (physical trampling, grazing, and nutrient impacts) as we think this has implications for restoration. Increasing wetland area by 20% could potentially increase wetland P storage by about 13 kg P ha⁻¹; however this may be site-specific.

Leaching of Nutrients from Wetland Plant Litter

Introduction

The second study within these set of tasks investigated the leaching of nutrients from plant litter and decomposition. Leaching studies were conducted in the laboratory, while decomposition studies were undertaken in the field. We will describe the leaching study first and the decomposition study second.

Leaching is a natural process where mass is lost from senesced vegetation due to the release of soluble organic and inorganic compounds from the plant biomass (Robertson, 1988). The leaching process is the first of three phases of decomposition, and is subsequently followed by microbial mineralization and physical and biological fragmentation (Valiela, 1985; Webster and Benfield, 1986). There is evidence that leaching is correlated with rainfall when standing dead vegetation is attached to the plant (Taylor et al, 1989; Qiu, 2005). The most rapid period of leaching usually lasts from a few days to a few weeks (Davis et al., 2003) depending on water availability, and is not mediated by microbial processes. The greatest leaching rates usually occur during the first rainfall or the first hours after emersion in the water column (Tope, 2003; Qiu, 2005). Leaching processes can result in a large nutrient flux of P, N, and C from plant biomass into the water column during fall and spring as a result of vegetative senescence (Mitsch et al., 1989). This is the primary reason why leaching has recently become one of the primary concerns relating to water quality in agricultural areas (Kuusemets and Mander, 2002). Short-term leaching can lead to a significant release of P into the environment. Anywhere from a 20-50% loss of the total P in the plant biomass can be released from vegetation in a few hours and upwards of 80% of the total P can be released during the first 2 months of mineralization (Simpson et al., 1978; Webster and Benfield, 1986; Tope, 2003; Qiu, 2005; Reddy et al., 2005).

The main objective of this study within these set of tasks was to quantify the amount and rate of P loss from senesced vegetation under aerobic and anaerobic water column conditions, and in response to nitrogen enrichment. The species that releases the lowest amount of P in this study relative to its P assimilation rate could be used as information for a potential BMP to increase on site P storage in historically isolated wetlands. In addition, knowing species specific P leaching characteristics, provides managers and landowners with guidance on beneficial and problematic plants when trying to retain P on pastures and in wetlands.

Materials and Methods

Recently senesced (standing dead) vegetation was harvested from Larson Dixie ranch in November 2004. Dominant species in the wetland center, edge, and upland were collected. The senesced vegetation was air dried. Air dried litter material was placed in a covered plastic containers and filled with 200 ml of treatment water. A hypodermic needle was inserted into the container head space through a rubber septum in the top of

the container as a pressure relief vent. A longer needle was pushed through the same rubber septum and below the water surface to bubble atmospheric gas (aerobic conditions) or nitrogen gas (anaerobic conditions). Water treatments consisted of low P water collected from a cypress slough near the research site, which is defined as “Site Water” in this chapter (SRP concentration of 0.09 ± 0.00 mg/L, a DOP concentration of 0.09 ± 0.01 mg/L, and a TKN concentration of 1.22 ± 0.09 mg/L). Other water treatments included site water diluted 50% with DI water (SRP concentration of 0.04 ± 0.00 mg/L, a DOP concentration of 0.08 ± 0.01 , and a TKN concentration of 1.15 ± 0.18), and site water spiked with 3 mL of 1000 mg/L NO_3 to double the N concentration to 0.41 mg/L (SRP concentration of 0.09 ± 0.00 mg/L, a DOP concentration of 0.09 ± 0.00 mg/L, and a TKN concentration of 2.80 ± 0.14 mg/L).

Flux containers were covered and kept in darkness to prevent algae growth. The timescale of sampling was after 2 hrs, 24 hrs, 3 days, 7 days, and 17 days, with an initial characterization of site water before the experiment began. At each sampling period 20 ml of water was removed from the flux container and analyzed for SRP. At time zero and after 17 days, an additional 40 mL of water was removed and analyzed for Dissolved Organic Phosphorus (DOP) and Total Kjeldahl Nitrogen (TKN).

The mean P flux from each species were compared using Tukey-Kramer HSD (honestly significant difference) to determine significant differences between the cumulative P flux among species. Regression analyses and one way analysis of variance (ANOVAS) were used to determine which substrate quality parameter was the best predictor of short term P release using the R^2 and p-values.

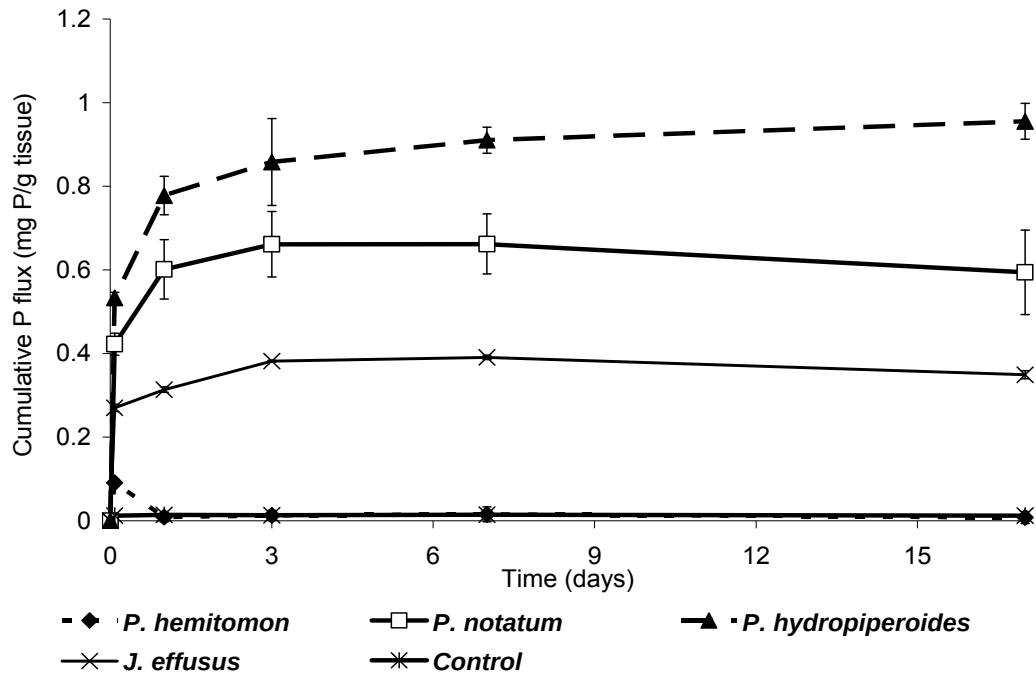
Results

Cumulative P flux was averaged and ranged from of 0.01 ± 0.01 mg P/g tissue during a period of 17 days (*P. hemitomom*) to 0.96 ± 0.17 mg P /g tissue (*P. hydropiperoides*) under aerobic conditions (Figure 25). Cumulative P flux from *P. notatum* and *J. effusus* fell between this range. Leaching was greater under aerobic conditions in each species except *J. effusus*, which had the greatest P flux under anaerobic conditions. The only statistically significant differences observed between aerobic and anaerobic conditions during 17 days was in *P. notatum* and *J. effusus* where P fluxes were greater under anaerobic conditions (Table 28).

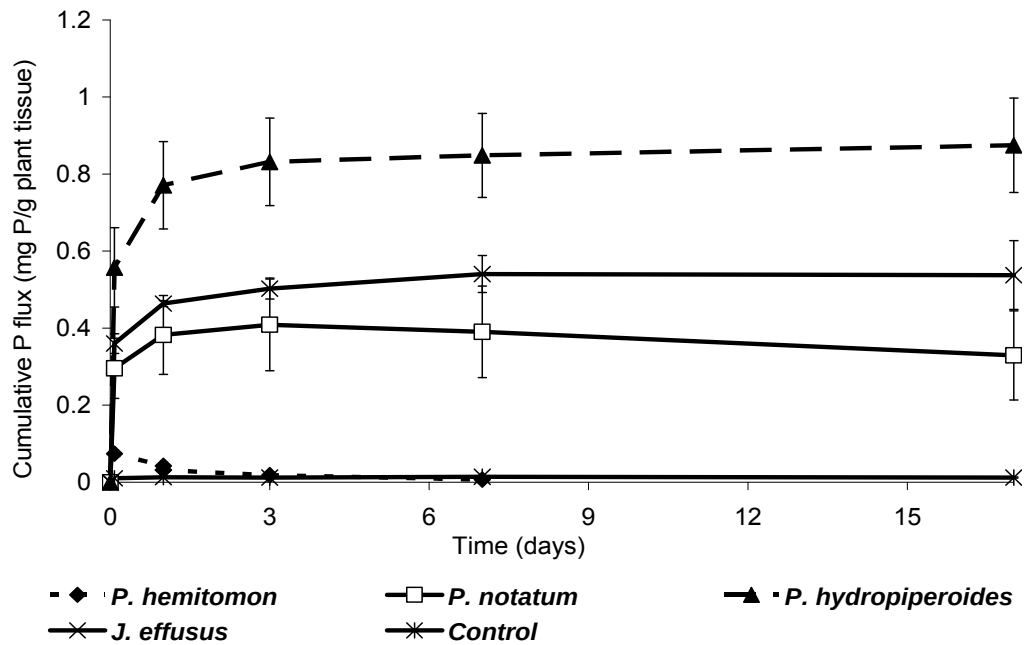
Significant differences in flux rate among individual species under aerobic conditions (Fig. 25) occurred within the first 2 hours. At this sampling period, P flux from *P. notatum* and *P. hydropiperoides* were significantly higher than the other two species and *P. hemitomom* had a significantly lower P release compared to all other species. During the study, each species had a significantly different flux rate.

Phosphorus flux rates were significantly different only 2 hrs after water was added to flux containers (Fig. 25). At this sampling period *P. hydropiperoides* again had a significantly

greater P flux, while *P. hemitomom* still had the least. At day 17, there were significant differences among each species.



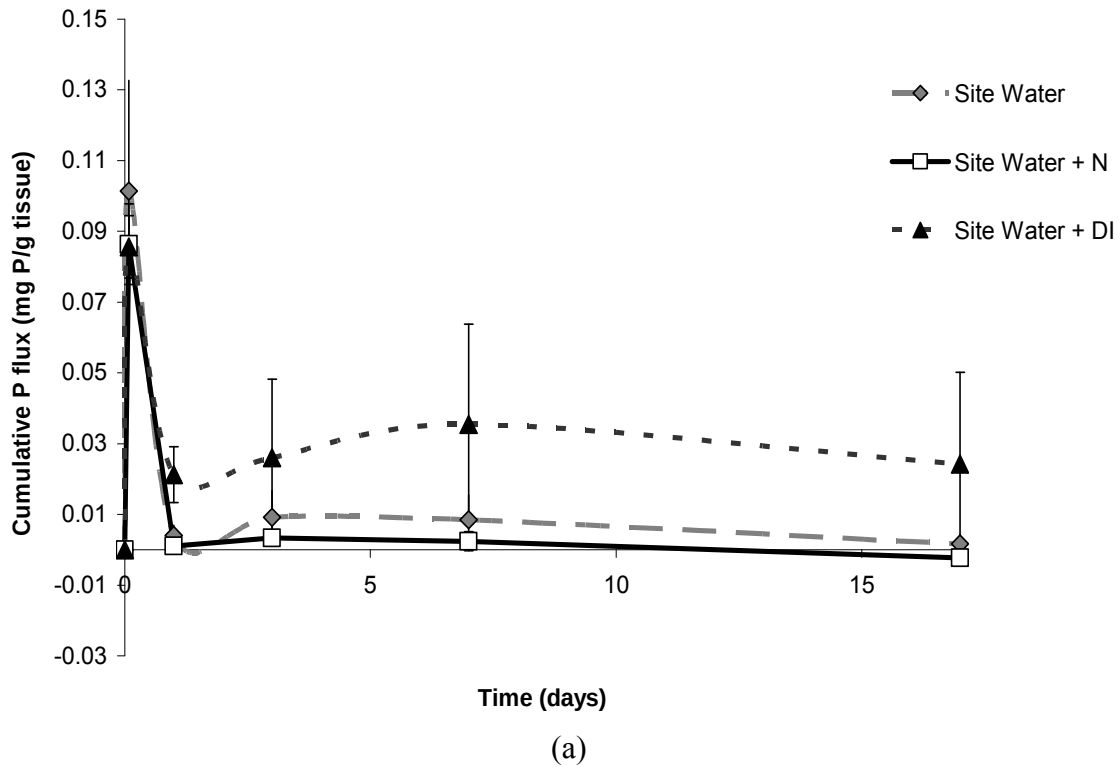
(a)



(b)

Figure 25. Phosphorus leaching rates of four senesced species averaged across all three water treatments (a) aerobic and (b) anaerobic conditions. Values represent mean ($\pm$ standard deviation).

The species with the highest P flux under aerobic and anaerobic conditions was *P. hydropiperoides* (Fig. 25). The P flux of this species continuously increased throughout the experiment, with the highest cumulative flux of 1.00 ± 0.2 mg P/g plant tissue observed under aerobic conditions in the site water + N treatment, and 0.98 ± 0.2 mg P/g under anaerobic conditions in the site water + DI treatment after 17 days (Fig. 27). The lowest 17 day P flux for this species was seen in the site water + DI treatment under aerobic conditions (0.94 ± 0.15 mg P/g) and site water + N treatment under anaerobic conditions (0.74 ± 0.08 mg P/g).



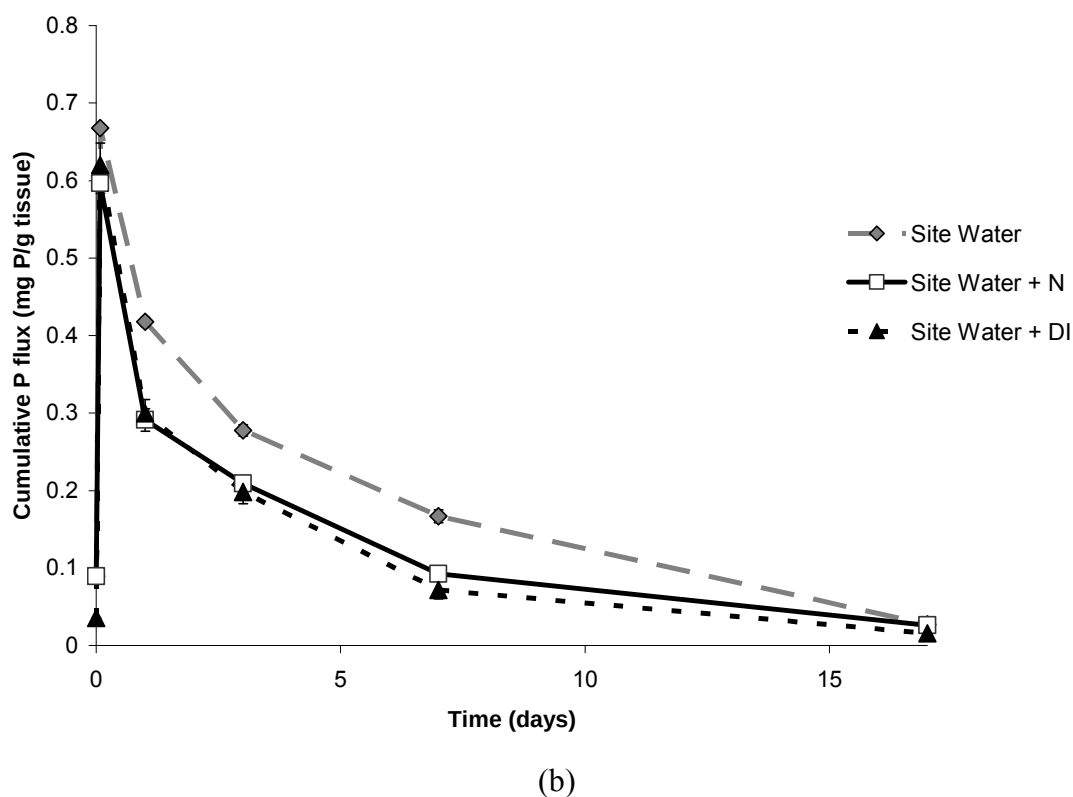
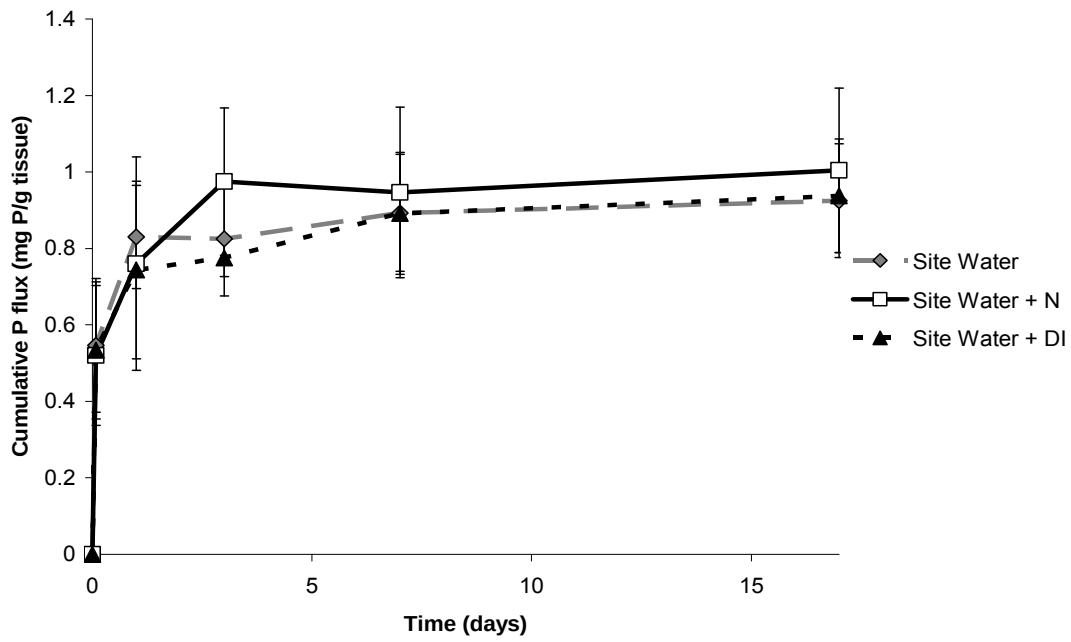
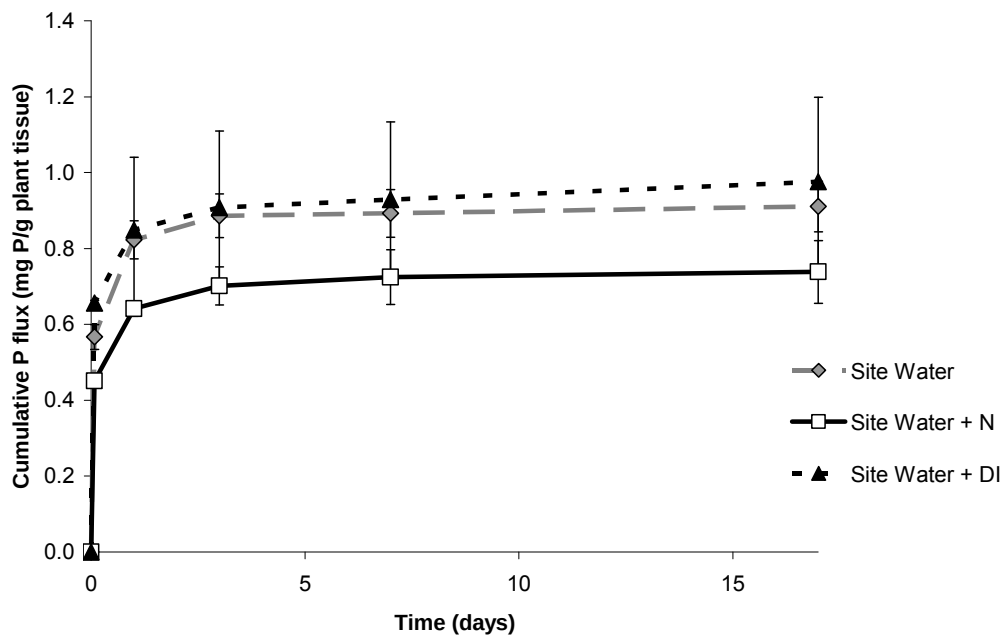


Figure 26. Litter phosphorus release rate for *P. hemitomom* litter under (a) aerobic and (b) anaerobic conditions . Values represent mean (± 1 standard deviation)

The remaining two species, *P. notatum* and *J. effusus* released a moderate amount compared to *P. hydropiperoides*. There were not any significant differences between treatments in *P. notatum* under aerobic conditions; however, the site water + N treatment had a significantly lower P flux for this species under anaerobic conditions compared to the other two treatments over the course of 17 days (Fig. 28). *Juncus effusus* however, had no significant differences in P flux between treatments under aerobic or anaerobic conditions (Fig. 29), although P flux was significantly higher under anaerobic conditions compared to aerobic conditions for this species (Table 28).



(a)



(b)

Figure 27. Litter phosphorus release rate for *P. hydropiperoides* litter under a) aerobic and b) anaerobic conditions. Values represent mean (± 1 standard deviation).

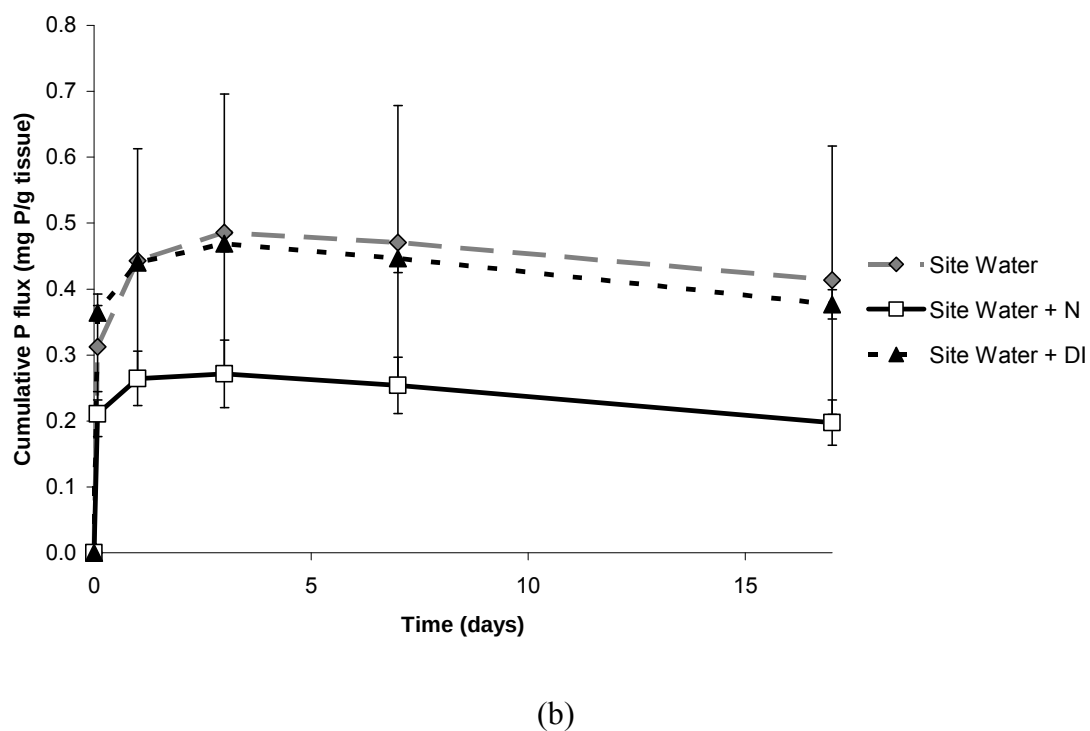
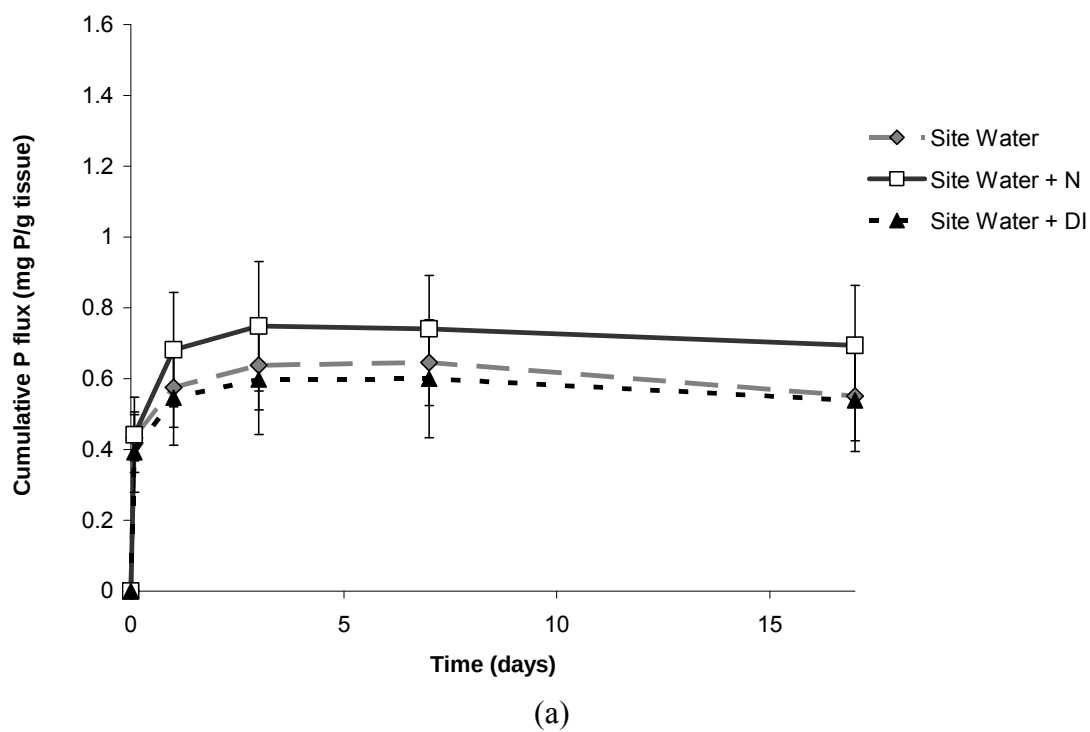
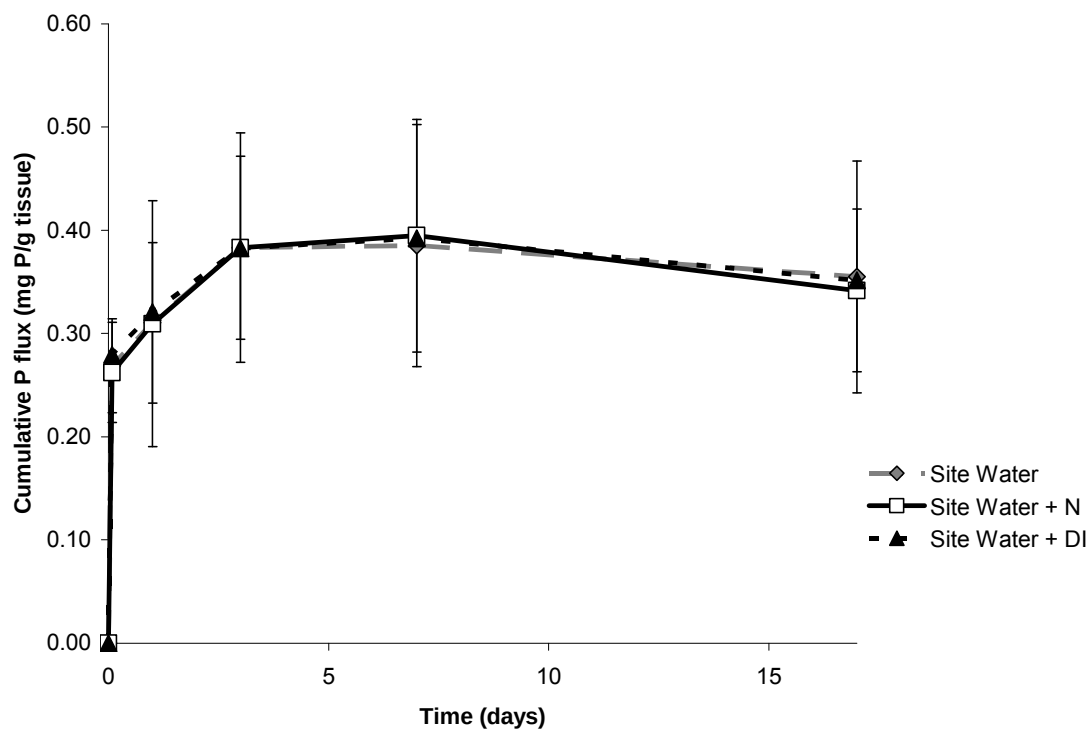
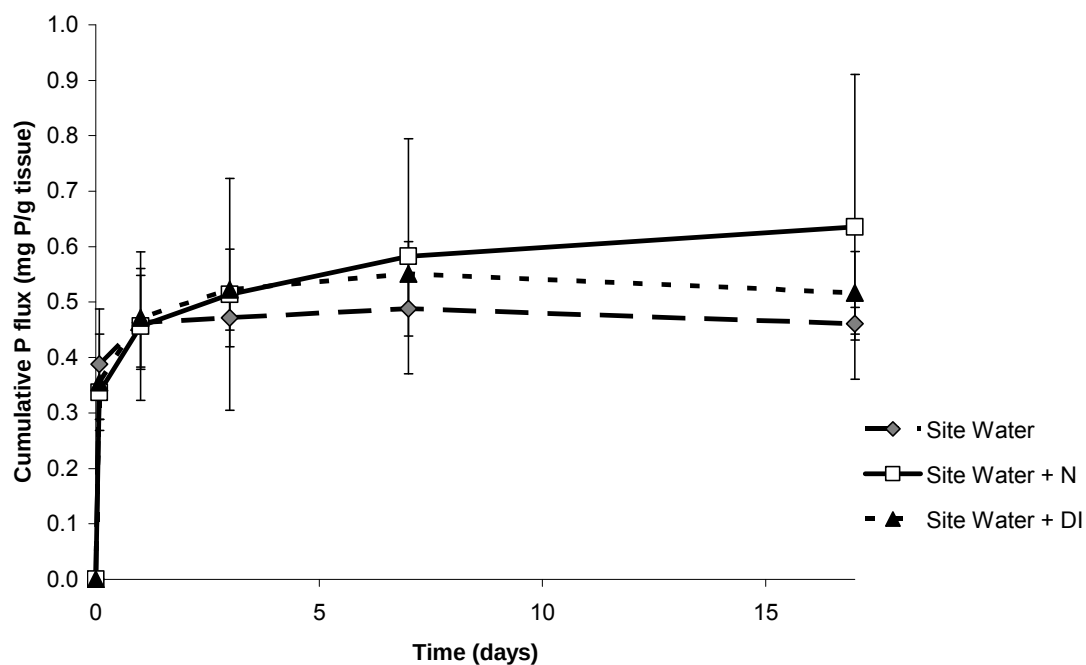


Figure 28. Phosphorus release rate for *P. notatum* litter under a) aerobic and b) anaerobic conditions. Values represent mean (± 1 standard deviation).



(a)



(b)

Figure 29. Phosphorus release rate for *J. effusus* litter under a) aerobic and b) anaerobic conditions. Values represent mean (± 1 standard deviation).

The water treatment or redox condition did not appear to have an effect on P released from senesced vegetation during a 17 day period (Table 28).

Table 28. Cumulative P release (mg P/g litter) and percent tissue P released from four species over a 17 day period, under aerobic and anaerobic conditions.

Species	Treatment	17 Day Total Cumulative Flux (mg P/g tissue)					
		Aerobic Condition		% P Released	Anaerobic Condition		%P Released
<i>P. hemitomon</i>	Site Water	0.002 (a)	± 0.003	0.80	0.007 (a)	± 0.001	3.55
	Site Water + N	-0.002 (a)	± 0.001	-1.08	0.003 (a)	± 0.004	1.28
	Site Water + DI	0.024 (a)	± 0.026	11.61	0.011 (a)	± 0.006	5.45
<i>P. notatum</i>	Site Water	0.551 (b)	± 0.126	66.37	0.414 (c)	± 0.203	49.86
	Site Water + N	0.694 (b)	± 0.170	83.58	0.197 (d)	± 0.034	23.78
	Site Water + DI	0.539 (b)	± 0.144	64.94	0.377 (d)	± 0.022	45.41
<i>P. hydropiperoides</i>	Site Water	0.925 (e)	± 0.149	72.34	0.911 (e)	± 0.066	71.22
	Site Water + N	1.004 (e)	± 0.215	78.55	0.738 (f)	± 0.082	57.74
	Site Water + DI	0.937 (e)	± 0.149	73.32	0.976 (e)	± 0.223	76.32
<i>J. effusus</i>	Site Water	0.355 (g)	± 0.112	44.16	0.461 (h)	± 0.030	57.33
	Site Water + N	0.342 (g)	± 0.079	42.52	0.636 (h)	± 0.275	79.09
	Site Water + DI	0.351 (g)	± 0.055	43.72	0.517 (h)	± 0.074	64.32

*Values represent mean (± 1 standard deviation). Negative values indicate P removed from the water column due to plant or microbial uptake. Lowercase letters indicate significant differences over the 17 day study between species, treatment, and redox condition with a p-value of 0.05.

Over the span of 17 days, SRP concentration significantly increased from the initial 0.09 mg/L in the site water flux containers of every species except *P. hemitomon*.

Approximately 96% of the P in the water column at the end of the study was SRP, a readily labile form, while the remaining P fraction was DOP (Table 29).

Table 29. Mean concentrations (± 1 standard deviation) of water column P for various species under aerobic and anaerobic conditions after 17 days.

Species	Treatment	SRP (mg/L)		DOP (mg/L)	
		Aerobic	Anaerobic	Aerobic	Anaerobic
<i>P. hemitomon</i>	Site Water	0.03 $\pm$ 0.01	0.03 $\pm$ 0.01	0.06 $\pm$ 0.02	0.09 $\pm$ 0.02
<i>P. notatum</i>		2.72 $\pm$ 0.67	2.07 $\pm$ 1.09	-0.22 $\pm$ 0.06	0.15 $\pm$ 1.07
<i>P. hydropiperoides</i>		4.99 $\pm$ 0.84	4.84 $\pm$ 0.37	0.18 $\pm$ 1.00	-0.03 $\pm$ 0.33
<i>J. effusus</i>		1.87 $\pm$ 0.65	2.37 $\pm$ 0.25	0.05 $\pm$ 0.53	0.13 $\pm$ 0.33
<i>P. hemitomon</i>	Site Water + N	0.02 $\pm$ 0.00	0.03 $\pm$ 0.00	0.06 $\pm$ 0.03	0.07 $\pm$ 0.01
<i>P. notatum</i>		3.57 $\pm$ 0.88	0.92 $\pm$ 0.16	0.23 $\pm$ 1.91	0.18 $\pm$ 0.13
<i>P. hydropiperoides</i>		5.51 $\pm$ 1.07	3.97 $\pm$ 0.34	0.05 $\pm$ 1.11	0.04 $\pm$ 0.64
<i>J. effusus</i>		1.77 $\pm$ 0.34	3.61 $\pm$ 1.65	0.06 $\pm$ 0.31	0.14 $\pm$ 1.56
<i>P. hemitomon</i>	Site Water + DI	0.11 $\pm$ 0.00	0.02 $\pm$ 0.00	0.09 $\pm$ 0.16	0.06 $\pm$ 0.01
<i>P. notatum</i>		2.68 $\pm$ 0.74	1.74 $\pm$ 0.17	0.12 $\pm$ 0.70	0.23 $\pm$ 0.13
<i>P. hydropiperoides</i>		5.13 $\pm$ 0.73	5.17 $\pm$ 1.27	0.23 $\pm$ 0.57	0.11 $\pm$ 1.29
<i>J. effusus</i>		1.77 $\pm$ 0.27	2.67 $\pm$ 0.43	0.09 $\pm$ 0.28	0.24 $\pm$ 0.20

*Negative DOP values are due to high standard errors and represent virtually no measurable DOP in the water column.

In general TKN concentrations were higher under anaerobic conditions compared to aerobic conditions (Table 30). *Polygonum hydropiperoides* was the only species where TKN concentrations were higher under aerobic conditions (in site water and site water + N treatment).

Table 30. Mean concentrations (± 1 standard deviation) of water column TKN for various species under aerobic and anaerobic conditions after 17 days.

Species	Treatment	TKN (mg/L)	
		Aerobic	Anaerobic
<i>P. hemitomon</i>	Site Water	3.61 $\pm$ 2.42	6.21 $\pm$ 0.24
<i>P. notatum</i>		5.53 $\pm$ 0.32	6.31 $\pm$ 0.68
<i>P. hydropiperoides</i>		7.34 $\pm$ 3.54	6.08 $\pm$ 1.69
<i>J. effusus</i>		3.46 $\pm$ 0.58	6.47 $\pm$ 2.13
<i>P. hemitomon</i>	Site Water + N	4.41 $\pm$ 1.19	6.62 $\pm$ 1.54
<i>P. notatum</i>		5.15 $\pm$ 1.08	6.49 $\pm$ 0.60
<i>P. hydropiperoides</i>		6.02 $\pm$ 1.28	5.4 $\pm$ 1.55
<i>J. effusus</i>		3.81 $\pm$ 0.18	6.31 $\pm$ 1.50
<i>P. hemitomon</i>	Site Water + DI	7.07 $\pm$ 3.48	6.47 $\pm$ 1.54
<i>P. notatum</i>		6.02 $\pm$ 1.94	9.47 $\pm$ 5.17
<i>P. hydropiperoides</i>		3.53 $\pm$ 0.38	5.13 $\pm$ 2.28
<i>J. effusus</i>		2.97 $\pm$ 0.73	6.78 $\pm$ 1.04

Regressions were made between the cumulative P flux on day 17 and initial tissue concentration of N, P, and C and various ratios of these parameters. It was determined that a relationship existed between P contained in the tissue and the amount of P that tissue will release to the water column (Fig. 30).

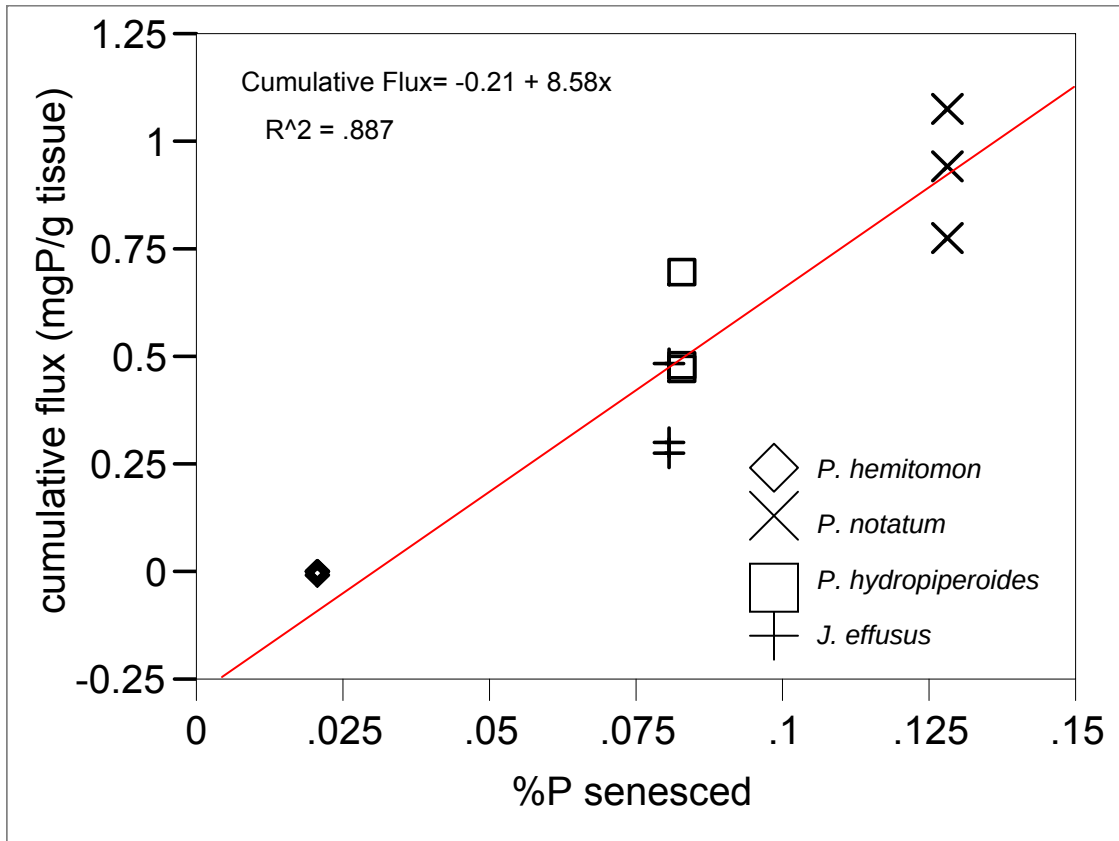


Figure 30. Bivariate fit of cumulative P flux by initial senesced tissue P content. Correlation is for site water treatment under aerobic conditions on day 17.

Initial substrate quality parameters of the live tissue characterized previous, were also correlated with the cumulative P release from each senesced species of vegetation. If changes in P mass can be predicted with confidence using a live tissue parameter, the application of the data may be more useful since live vegetation is more abundant. The strongest correlation was between P flux and % NDF or the labile carbon compounds associated with live tissue with an R² of .883 (Table 31).

Table 31. P-Values associated with initial nutrient parameters of live tissue to estimate the best predictor of P flux for site water under aerobic conditions on Day 17.

Substrate Quality Parameter	P-value	R ²
N:P	0.769	0.01
% Residual Fiber	0.097	0.25
% ADF	0.013	0.48
% C	0.008	0.53
% P	0.001	0.66
C:P	0.001	0.70
% N	<.001	0.83
C:N	<.0001	0.83

% SADF	<.0001	0.86
% NDF	<.0001	0.88

Discussion

Findings suggest the potential for vary rapid and significant amounts of tissue P release from plant litter upon exposure to rainfall or inundation, and the amount of P released from the senesced tissue was primarily dependant on species type. Approximately 70% of the P in the tissue of *P. hydropiperoides* was released on average after 17 days, while *P. notatum* and *J. effusus* both released 55% of the P that was initially associated with the biomass, and *P. hemitomon* released only 4% of the P associated with its biomass. The P flux values closely follow the amount of P each species originally contained in biomass, which helps explain why the cumulative P flux from *P. hydropiperoides* was greater than the cumulative flux for other species.

Results suggest that a potentially large amount of P can be released from the senesced tissue shortly after vegetative senescence. High quantities of P fluxing into the water column after senescence has the potential to increase P content in overlying water, which during high water events is discharged from wetland to ditch.

For each species, with the exception of *P. hydropiperoides*, there was a negative P flux observed in either aerobic or anaerobic conditions during the last 7-14 days of the study. A decrease in the cumulative P flux can only be explained by microbial uptake of P from the water column or P uptake by litter. Near day 7, a bacterial growth occurred around the needles bubbling gases into the flux containers. It is likely that the negative slopes depicted in the graphics above were due to microbial nutrient uptake.

Looking at the cumulative flux curves for each species, it is estimated that P leaching took place for the first 7 days of this study. It was suspected that the water spiked with N may increase P release during mineralization because all species initially had a high C:N, and would have been N limited under aerobic conditions. A nitrogen addition would increase the N concentration, as well as the amount of nutrients available to microbes, and therefore stimulating mineralization, C breakdown and P release.

The high initial C:N and low P concentrations in *P. hemitomon* is likely the reason why this species had the lowest change in mass after 17 days, while *P. hydropiperoides* underwent the highest mass change because it's initial C:N was the lowest.

The importance of Table 29 is to demonstrate that an extremely large proportion of P in the water column was SRP. This is an extremely labile form of P that is readily bioavailable to overlying water.

Results also show that approximately 90% of the P flux from litter after vegetative senescence can be predicted using the initial P tissue concentration of the senesced tissue; therefore, P fluxes are likely to vary depending on the species type. It is likely that short-

term P fluxes from senesced tissue can be predicted with a high level of confidence for many different species found on historically isolated wetlands in the Okeechobee Basin, not just the four species used in this study. Therefore, the release and transport of P may be managed using short-term P release rates predicted by an initial substrate quality parameter in live or senesced tissue. Future studies may indicate that *P. hemitomom* should be promoted along with hydrological restoration because this species may reduce P transport and improve water quality.

Plant Litter Decomposition

Introduction

The second part of this study was to investigate vegetation decomposition at the field-scale. Vegetation organic matter is made up primarily of carbon, and is a driver of many wetland ecosystem processes. Organic carbon can function as a nutrient source for microbes, can adsorb toxic compounds or nutrients, and can provide an exchange capacity for cations (Cotrufo, 2006). The breakdown of organic matter or microbial mineralization of nutrients is known as decomposition. Plant tissue substrate quality partly determines decomposition rates (Melillo, 1982; Berg, 1998; Villar, 2001). Substrate quality is determined partly by the recalcitrance or lability of the fiber fractions within plant tissue. In addition, the amount of available nutrients such as N and P relative to the available carbon content is also a component of substrate quality. Substrate quality is an important parameter to consider when estimating decomposition rates of different plant species because it can either enhance or inhibit microbial colonization of the litter.

In most cases nutrient enrichment increases decomposition because N and P are often limiting factors controlling microbial growth (Qualls, 1984; Taylor et al., 1989; Corstanje et al., 2006). Others have found that nutrient enrichment has no effect or can cause a lower rate of decomposition due to a possible C limitation (Berg et al., 1998; Carreiro et al., 2000; Villar et al., 2001). It may also be that the influence of nutrient content is variable during different stages of decomposition.

When living plants are present in a wetland, they are likely accumulating P and incorporating it into their biomass; however, when the plant senesces and litter enters the water column, litter may become a P source (Moore and Reddy, 1994). Species type is likely to be an important factor when estimating long-term P release since some species accumulate more nutrients than others (Hobbie, 1992; Knops, et al., 2002). McJanet et al. (1995) reported significant differences in the N and P content of 41 different wetland plant species with N concentrations ranging from 0.25-2.1% and P concentrations of 0.13-1.1%. Age of the senesced material may also play a role in the rate of decomposition.

Although P assimilation rate by plants is a critical factor in assessing potential efficacy of wetland P storage, if plants senesce *in-situ*, the overall effectiveness of a wetland to immobilize and store P is also dependant on the release rate of P during litter decomposition. The relationship between decomposition rates and substrate quality, as well as substrate quality and plant species suggest that the type of plant species present in

a wetland is also a critical factor regulating litter mineralization rates and long-term P storage. A better understanding of the relationship between tissue substrate quality and P litter mineralization will assist in evaluating the implications of vegetative community change in response to hydrologic restoration on downstream P load to Lake Okeechobee.

The objectives of this study were to measure plant litter chemical characteristics and quantify their decomposition rates along a hydrologic gradient from historically isolated wetlands extending up into grazed pasture upland.

Materials and Methods

Field Methods

Litter decomposition rates were determined by measuring percent mass loss of standing dead vegetation collected from wetland center, wetland edge, and upland communities. Recently senesced standing dead biomass was collected from *P. hemitomon*, *P. hydropiperoides* (representative of the dominant vegetation in the wetland center), *J. effusus* (representative of the dominant edge species), and *P. notatum* (representative of the dominant upland species) during November 2004.

The biomass was air dried and air dried biomass was placed in litterbags. Litterbags were deployed along a hydrological gradient from the center of the wetland to the upland. Litterbags were harvested on June 1st, 2005, July 27th, 2005, December 8th, 2005, and April 5th, 2005, representing 2, 4, 8, and 12 months of exposure.

Laboratory Methods

After litterbags were collected from the field and brought back to the laboratory, litterbag contents were dried. After drying, litter was weighed to determine mass loss after each exposure time. Samples were then ground and sieved and analyzed for total N and total C and total P. An Ankom 200 Fiber Analyzer was used to quantify the neutral detergent fiber (NDF) which is the starches, sugars and labile components associated with the plant tissue, the acid detergent fiber (ADF) which is the hemicellulose fiber fraction, the strong acid detergent fiber (SADF) which is the cellulose fiber fraction, and the residual fiber or “lignin” percentages of the litter. Fiber analysis was only conducted on litter collected 12 months after deployment.

Results

There were no significant differences in decomposition rates among the three wetlands when all species were combined. Overall, results indicate a slightly slower decomposition rate in Larson South (LS) compared to Larson East (LE) which had the highest percent mass loss of the three wetlands (Fig. 31).

Litter decomposition in the four hydrologic zones (center, edge, transitional zone, and upland) did not vary significantly during the study. The only significant difference between litter mass loss and hydrologic zone occurred after 2 months when the wetland center litterbags had a higher percent mass loss compared to the other three zones (Fig. 32). Decomposition rates over time were best modeled by an exponential decay curve, with similar rates between zones. The litter decomposition rates in the four hydrological zones was opposite to that hypothesized and it is believed to be in response to conditions occurring at the sites during the first two months. During April 1, 2005 and June 1, 2005, little or no rainfall occurred and no standing water was present in the wetland. If moisture was limiting decomposition during this period, results suggest that wetland centers may have had more optimal conditions to promote decomposition, while other zones may have been moisture limited. After this initial 2 month period, rain frequency increased and moisture limitation was less likely a limiting factor.

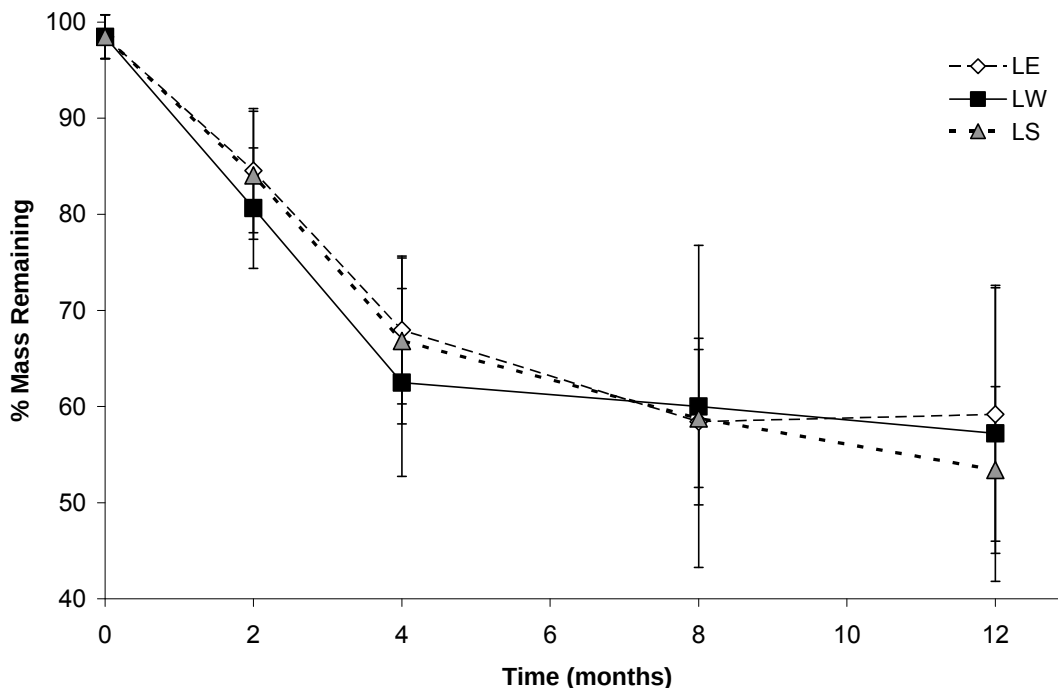


Figure 31. Litter decomposition of all species in each hydrologic zone among the three different wetlands, Larson East (LE), Larson West (LW), and Larson South (LS). Values represent mean $\pm$ 1 standard deviation.

Paspalum notatum had a significantly higher decomposition rate during the 12 months compared to the three other species of senesced vegetation. During the first 2 months, there were not any significant differences among the decomposition rates of the four species. However, *P. notatum* had a significantly greater percent mass loss compared to the three other species during the 4 and 12 month sampling periods (Fig. 33). The greatest amount of decomposition was observed in *P. notatum* litter collected from the upland with roughly a 62% loss in mass after 12 months. The least amount of

decomposition was observed in *P. hydropiperoides* (a 32 % loss after 12 months in the edge) (Table 32). An average of 43 % of the original litter mass from each species within the 4 hydrological zones had decomposed after 12 months. In addition, the highest rate of decomposition was observed during the first 4 months where approximately 34% of the original biomass had decomposed, while roughly only an additional 9% of the remaining biomass was lost during the next 8 months (Fig. 33). There have been other studies that observed a lower rate of decomposition after the first 4 months of deployment, which may indicate that most of the labile components associated with the biomass of the senesced tissue have been lost by this time (Boyd, 1971; Berg et al., 1998; Villar et al., 2001).

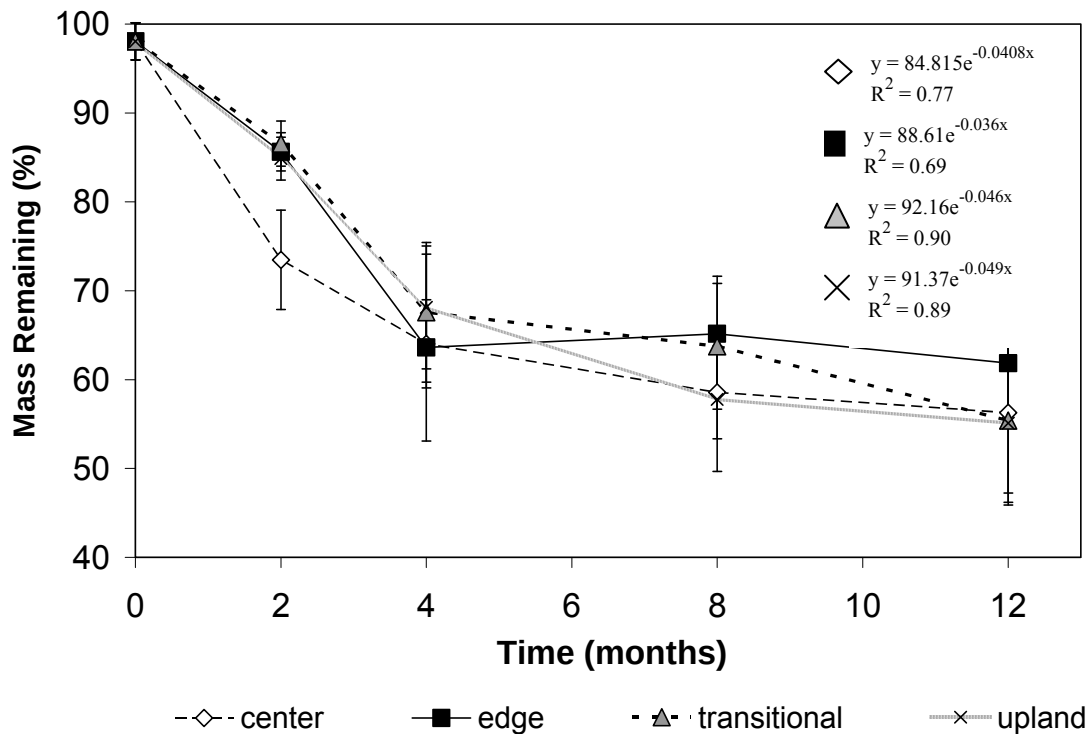


Figure 32. Decomposition in the four hydrological zones over a 12 month period. Values represent mean $\pm$ 1 standard deviation.

Out of all the initial substrate quality parameters characterized (outlined previous) there were no parameters in either the live or senesced tissue that could predict percent mass remaining among the four different species after 12 months in the field. The initial substrate quality parameter that best predicted the mass remaining after 12 months was carbon content. Although the relationship was significant, it only explained 20% of the variability in decomposition rate.

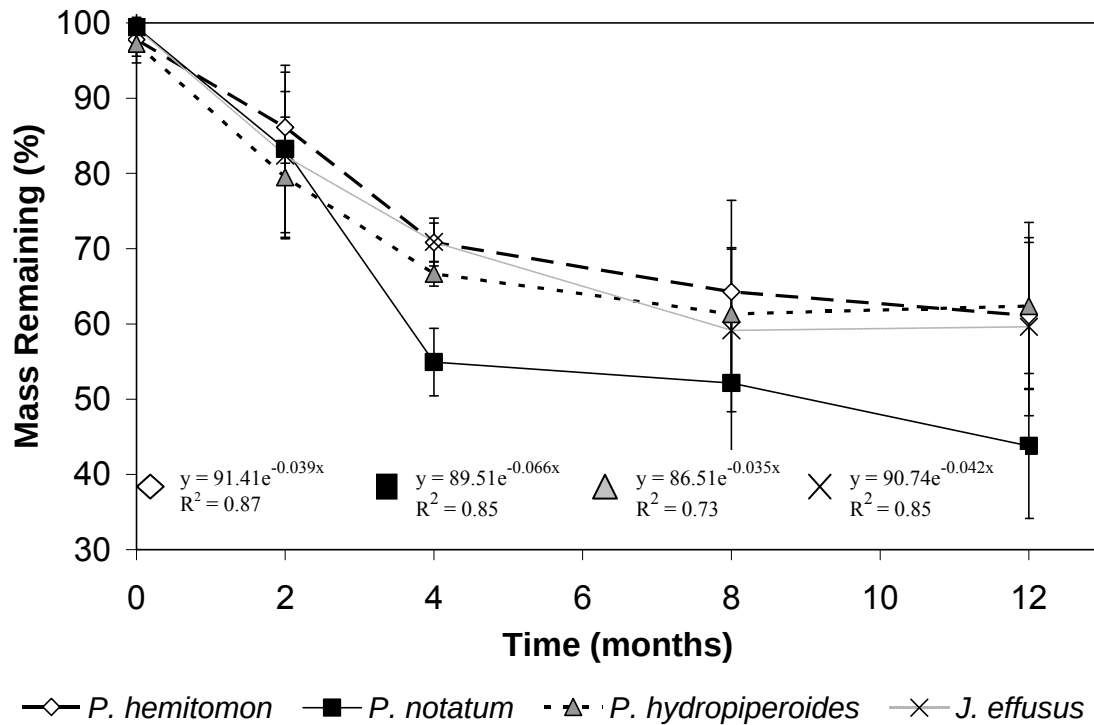


Figure 33. Average litter decomposition of each species over a 12 month period. Values represent mean $\pm$ 1 standard deviation.

During the course of this study, litterbags deployed in the upland had a significantly lower P content than litterbags deployed in the edge and transitional zone. Bags in the edge and transitional zone had a significantly higher P content after 8 months compared to the center, and after 12 months litterbags deployed in the transitional and edge had a significantly higher P content compared to the upland (Fig. 34).

Table 32. Species decomposition in each zone of litterbag deployment after 12 months. Values represent mean ($\pm$ 1 standard deviation).

Species	Zone	Mass Loss (%)
<i>P. hemitomon</i>	center	42 $\pm$ 7
	edge	36 $\pm$ 15
	transitional	41 $\pm$ 7
	upland	36 $\pm$ 8
<i>P. notatum</i>	center	58 $\pm$ 4
	edge	48 $\pm$ 9
	transitional	58 $\pm$ 5
	upland	62 $\pm$ 13
<i>P. hydropiperoides</i>	center	36 $\pm$ 16
	edge	32 $\pm$ 13
	transitional	43 $\pm$ 6
	upland	39 $\pm$ 4

<i>J. effuses</i>	center	42 ± 19
	edge	36 ± 10
	transitional	42 ± 10
	upland	42 ± 5

When evaluating P mass remaining in litterbags, some species of litter (*P. hemitomon*) acted as a P sink throughout the course of this study, while *P. hydropiperoides* and *J. effusus* were a P source for the first 2-4 months (Fig. 35). During the first 4 months, *P. hydropiperoides* released approximately 50% of the initial P contained in the biomass, while *J. effusus* released approximately 30% of its original P mass during the first 2 months in the field. *Panicum hemitomon* had a significantly lower P mass compared to the other three species surveyed during the 12 months. After 2 months in the field, *P. hydropiperoides* lost the greatest amount of P compared to the three other species, while *P. hemitomon* had the lowest P mass (Fig. 35).

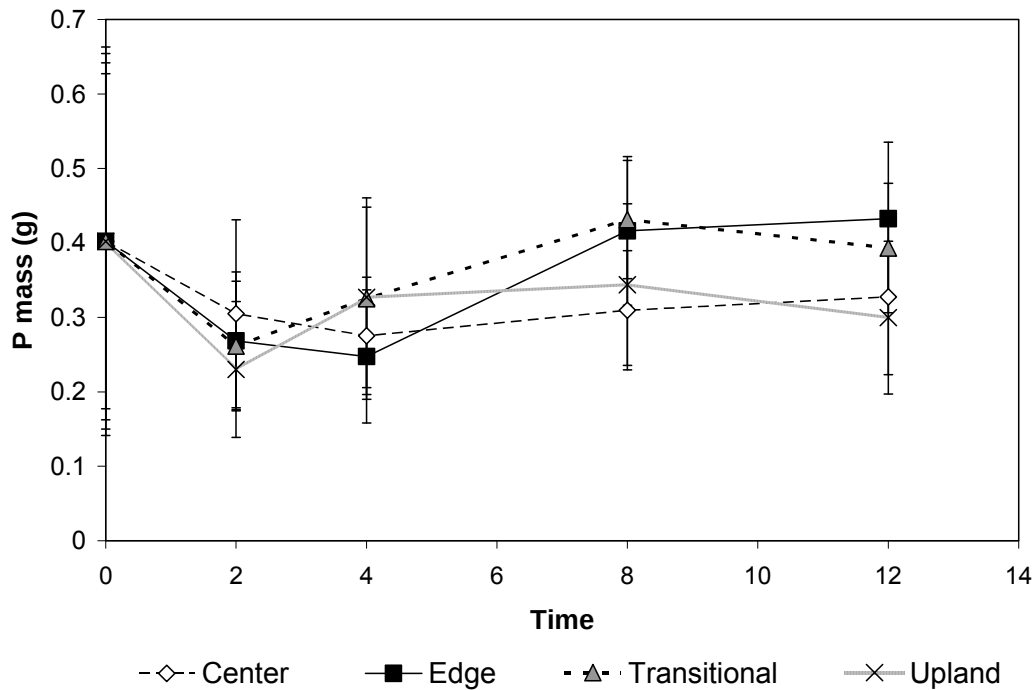


Figure 34. Change in litter P mass in the 4 hydrological zones over a 12 month period. Values represent mean $\pm$ 1 standard deviation.

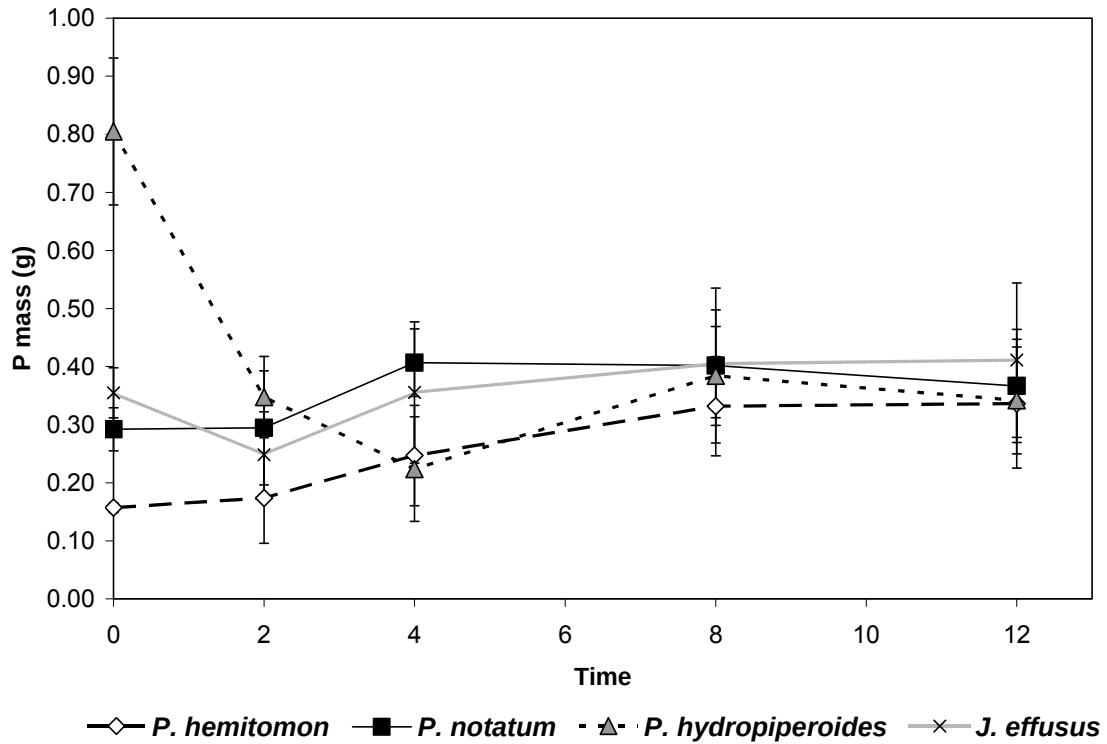


Figure 35. Change in P mass of the 4 dominant species over time. Values represent mean $\pm$ 1 standard deviation.

There was no clear relationship between initial substrate quality parameters in the senesced tissue and the mass of P remaining after the litter was in the field for 12 months. There were stronger relationships observed between initial substrate quality parameters and P content after 2 months exposure (Table 33).

Table 33. Significance and R^2 values of relationship between remaining litter P content and initial senesced substrate quality characteristics after 2 months.

Substrate Quality Parameter	All Zones		Center		Edge		Transitional		Upland	
	p-value	R^2	p-value	R^2	p-value	R^2	p-value	R^2	p-value	R^2
% C	.3724	.008	.8723	.001	.8123	.003	.7045	.007	.0901	.125
% SADF	.0603	.037	.6213	.009	.2305	.065	.5480	.017	.0726	.139
% ADF	.0003	.132	.6115	.012	.0119	.255	.0784	.134	.0003	.459
% Residual Fiber	.0001	.143	.6285	.011	.0087	.274	.0589	.153	<.0001	.505
C:N	<.0001	.167	.4838	.022	.0018	.365	.0409	.177	.0003	.453
%N	<.0001	.179	.5129	.020	.0014	.379	.0305	.195	<.0001	.523
N:P	<.0001	.280	.6747	.008	<.0001	.635	.0002	.481	<.0001	.782
% NDF	<.0001	.281	.5634	.016	<.0001	.598	.0013	.383	<.0001	.836
%P	<.0001	.287	.5750	.014	<.0001	.610	.0009	.403	<.0001	.859
C:P	<.0001	.324	.5924	.015	<.0001	.742	<.0001	.531	<.0001	.901

Initial substrate quality parameters of the live tissue characterized previous, was also correlated with changes in P mass in the senesced litter. If changes in the P mass can be predicted with confidence using a live tissue parameter, the application of the data may be more useful since live vegetation is more abundant than senesced material in the field. Like the senesced tissue parameters, there was not a live tissue parameter that could be used to predict the P mass after 12 months in the field. After only 2 months however, there were some significant relationships between several parameters in the live tissue (Table 34). Taking hydrologic zone into account, the initial C:N of the senesced tissue had the strongest negative relationship with P content in edge ($R^2 = 0.781$), transitional ($R^2 = 0.551$) and upland zones ($R^2 = 0.788$) (Fig. 36).

Table 34. Significance and R^2 values of relationship between the P content in the litter collected after 2 months from the center, edge, and transitional zones and initial live substrate quality characteristics after 2 months.

Substrate Quality Parameter	P-value	R^2
% NDF	0.807	0.00
N:P	0.7061	0.00
% ADF	0.7004	0.00
% SADF	0.6764	0.00

% Residual Fiber	0.5006	0.00
% C	<.0001	0.34
% P	<.0001	0.50
C:P	<.0001	0.53
% N	<.0001	0.67
C:N	<.0001	0.68

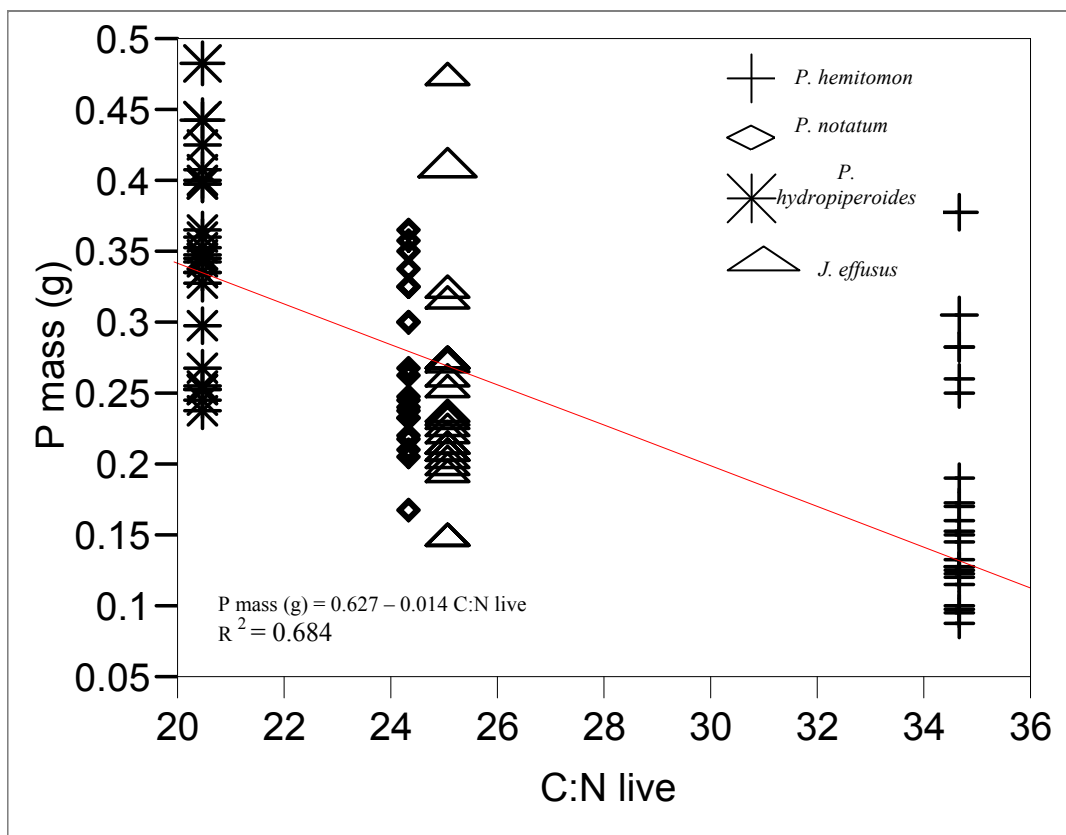


Figure 36. Correlation between initial C:N in live tissue and P mass after 2 months from litterbags collected in upland, transitional and edge hydrologic zones. Center litterbag data was not used as it showed no significant relationship.

The C:N ratio in species generally decreased during the first eight months of exposure. However, between 8 and 12 months in the field, C:N ratio in the litter increased in the transitional and upland hydrologic zones. Overall, the upland litter had significantly higher C:N compared to other zones (Fig. 37). During the first 2 months after deployment, litter in the center had a significantly lower C: N compared to litter in the other zones, while litter in upland zones had the highest C: N. After 4 months exposure, litter in uplands continued to have a significantly higher C:N compared to litter in center and after 12 months, litter in uplands had significantly higher C: N compared to the three other zones. A slight increase in the C:N in the upland (0-2 and 8-12 months sampling periods) and transitional zones (8-12 month periods) may suggest leaching or microbial mineralization during the dry months of the year (Dec. – June) instead of N accumulation relative to C losses seen in the wetland edge and center (Fig. 38).

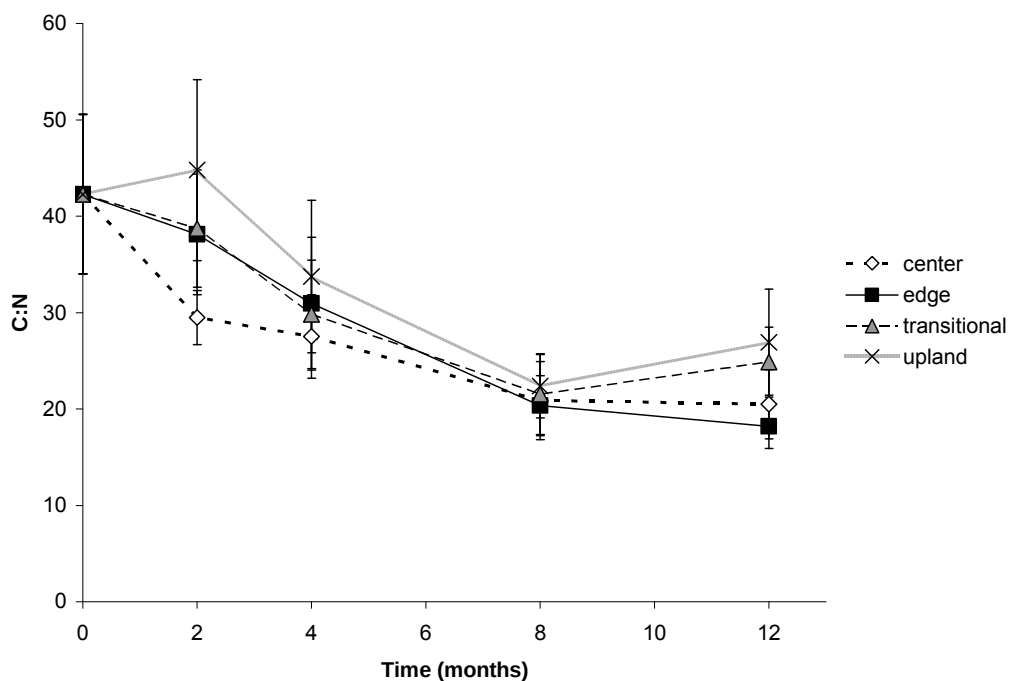


Figure 37. Litter C:N (all species combined) among different wetland hydrologic zones. Values represent mean $\pm$ 1 standard deviation.

Panicum hemitomon had a slightly higher C:N compared to the three other species surveyed prior to litterbag deployment.(Fig. 37). In addition this species also had one of the lowest amounts of decomposition over the 12 months of exposure. After 8 months exposure, changes in C: N in each species appeared to stabilize, and after 12 months C:N of each species converged around 15-25. This ratio is just below the value (25) often times considered N limiting under aerobic conditions. After 2 months in the field *P. hemitomon* and *J. effusus* had significantly higher C/N ratios compared to *P. notatum* and *P. hydropiperoides*. *Polygonum hydropiperoides* had the lowest C: N after 2 and 4 months, while *P. notatum* had the lowest C:N after 8 and 12 months. In addition, the average N content of the litter (by %) in this study doubled after 1 year in the field.

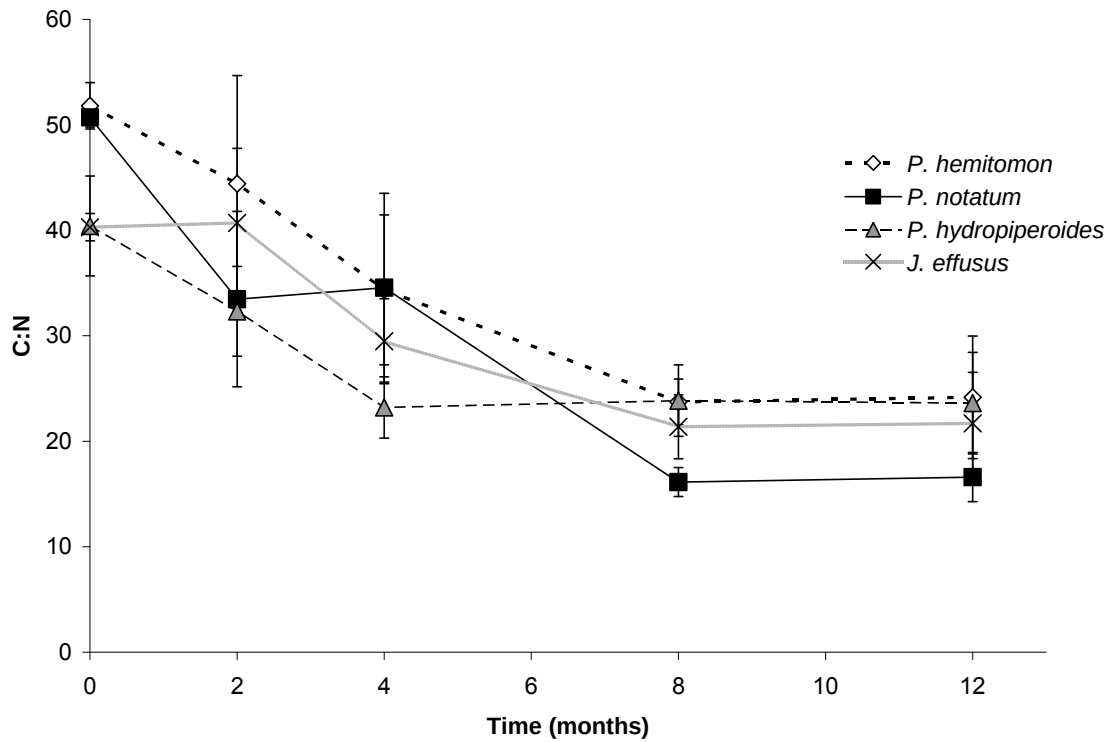


Figure 38. Change in litter C:N among species over time. Values represent mean $\pm$ 1 standard deviation.

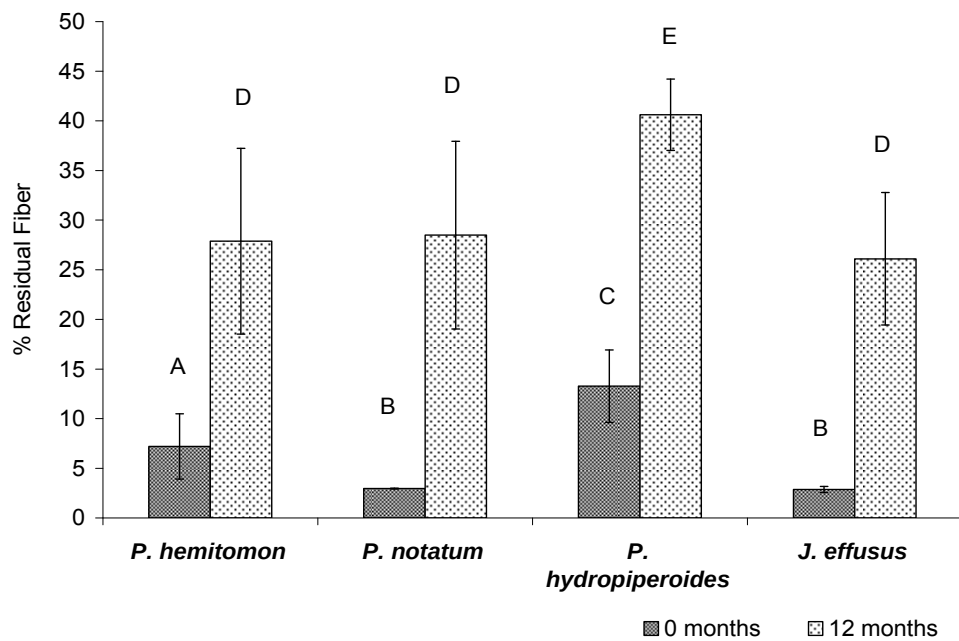


Figure 39. Residual fiber content of initial and 12 month exposed litter among four species tested. Values represent mean ($\pm$ 1 standard deviation). Letters

indicate significant differences between species and sampling times (Tukey HSD $\alpha = 0.05$).

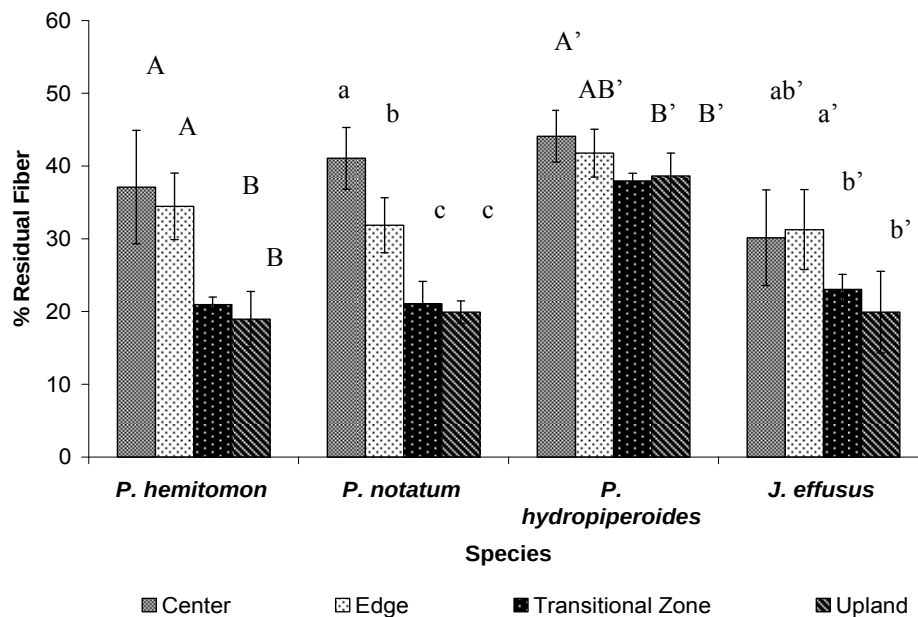


Figure 40. Residual fiber content of species in each hydrologic zone after 12 months. Values represent mean (± 1 standard deviation). Letters of similar character indicate significant differences between hydrologic zone within each species (Tukey HSD $\alpha = 0.05$).

The initial residual fiber content was significantly lower in every species compared to litter fiber content after 12 months exposure (Fig. 39). After 12 months, *P. hydropiperoides* had the highest residual fiber fraction among the three species surveyed in upland, edge, and transitional hydrologic zones. In addition, *P. hydropiperoides* had a significantly higher residual fiber fraction compared to *J. effusus* and *P. hemitomom* in the center zone. There were also significant differences in residual fiber content within each species among the 4 hydrologic zones. Residual fiber content in bags deployed in the upland and transitional hydrologic zones were generally significantly lower in residual fiber content than litterbags located in the center and edge hydrologic zone (Fig. 40).

Discussion

An average of 43% of the original litter had decomposed after 12 months exposure. The rate of decomposition in this study was lower than the 60% loss in litter mass observed within 4 months of a litterbag study of *J. effusus* (Boyd, 1971). Lower decomposition may be attributed to low N content of the litter. All species were N limited initially under aerobic conditions for the first 4-8 months, and higher N content could have possibly enhanced decomposition rates (Qualls, 1984; Taylor et al., 1989; Corstanje et al., 2006).

Regardless of the zone of deployment (center, edge, transitional, or upland), *P. notatum* decomposed faster than *P. hemitomom*, *P. hydropiperoides*, and *J. effusus*, and lost an average of 56% of the original biomass over the 12 month period (Fig. 33). Results from initial characterization of litter quality indicate that *P. notatum* also had the lowest initial amount of residual fiber. A low percentage of residual fiber in the biomass of *P. notatum* could have been responsible for a higher degree of decomposition compared to other species. Lower litter recalcitrance can increase the likelihood of microbial colonization since carbon compounds are more bioavailable and material can be broken down more easily.

Results presented earlier indicate that *P. hydropiperoides* had the greatest initial NDF (non-detergent fiber) fraction as well as the most residual fiber compared to the 3 other species of vegetation. Sugars, starches and other components that are easily broken down make up the NDF fraction. It is likely that the leaves of this species are relatively labile and contain the majority of the NDF fraction, while the woody stems of the plant contain most of the residual fiber. The high NDF fraction as well as the significantly high N and P content of this species is a possible reason why *P. hydropiperoides* had the highest degree of decomposition for the first 2 months (Fig. 33). After this fraction was consumed by microbes, the decomposition rate decreased because the remaining biomass was primarily composed of residual fiber. After 12 months exposure, *P. hydropiperoides* still had the highest amount of residual fiber compared to the other species in each zone of litterbag deployment.

Panicum hemitomom had a relatively high percentage of residual fiber initially in the senesced tissue and the lowest N and P content, which may explain why this species had the lowest amount of decomposition primarily during the first 2 months. Slow decomposition of species with a low nutrient content could be supporting evidence that nutrients control decomposition until more than 30% of the original mass is lost, similar to findings by Lewis (2005).

It was surprising that decomposition rate was not more significantly influenced by hydrological zones of the wetland. It was originally thought that increased oxygen availability in the upland would promote decomposition since oxygen is the preferred electron acceptor and yields the most energy during microbial catabolism. Results from this experiment suggest that species type is what primarily determines short-term P release, while the location may influence decomposition and P release over much longer time periods.

Presence of cattle is an environmental factor on the research site that may significantly impact decomposition rate of litter, especially in the center where soil is primarily organic and more influence by trampling. Cattle may actually encourage decomposition in the center by stepping on litterbags, pushing them underneath the soil surface, and breaking the litter into smaller fragments. After the litterbags had been deployed in the field for 12 months the bags located in the center were buried up to 30 cm beneath the soil surface. While litterbags located in the 3 other hydrological zones were covered with a mat of vegetation, they were not pushed underneath the soil surface by cattle, indicating

that the bags in the center may have had environmental factors which could influence mass loss that were not an issue in other zones (Fig. 41). It is likely that mechanical fragmentation was not seen in the edge because the soil was much firmer in this zone compared to the center.

Results presented in Fig. 40 indicate that the residual fiber content after 12 months was greatest in the center compared to other hydrologic zones. This may suggest that litter collected from the center after 12 months was less decomposed compared to the litter in the upland, but greater mass was lost in the center from mechanical fragmentation and exiting the litterbag as a result of cattle trampling. The lower decomposition of litter in the wetland edge compared to the wetland center may also indicate that cattle have a greater effect on litter decomposition in the wetland center than biogeochemical processes taking place. In the future it may be beneficial to repeat this study with cattle exclosures to quantify cattle impacts on decomposition and P release. Many different confounding physical and environmental factors present in the field may have been the reason why a clear relationship between initial substrate quality parameters and the percent mass remaining or P mass in the litter after 12 months was not seen.



Figure 41. Litterbags collected after 12 months from A) wetland center, which were approximately 20-30 cm underneath the soil surface and B) wetland edge, which were on top of the soil surface.

The C:N ratio decreased in each species throughout the course of the entire study (Fig. 38). It is thought that N was assimilated by microbes from the water column, or sorbed to the surface of the litter as particles from manure or urine of cattle in the upland and transitional zone where standing water was rarely present. Villar et al. (2001) reported that the N content of litter increased 7 times after 2 years in the field while the N concentration doubled over 1 year for each of the species in this study.

Litter often alternates between releasing and absorbing nutrients as it decomposes (Jordan et al., 1989). Some species of litter in this study, *P. hemitomom* and *P. notatum*, increased in P concentration and content throughout the course of this study, while *P. hydropiperoides* and *J. effusus* released P during the first 2-4 months (Fig. 35). It was surprising not to see a decline in the P mass in the litter throughout the course of the study due to mineralization processes; however similar results of an increase in the P

mass have been reported from past decomposition studies (Villar et al., 2001; Davis et al., 2003). The increase in P content throughout the study was likely caused by soil contamination but could also be due to microbial assemblages adhering to the litter or manure particles.

Findings from the laboratory leaching study provided an estimate of short-term P release and P release rates were well correlated with initial P content. However when P release was evaluated at the field scale and over longer periods no strong relationship between an initial substrate quality parameter and the P release was evident. It is likely that the presence of uncontrolled environmental factors such as cattle, cycles of wetting and drying, changing moisture content, and various temperatures in different hydrological zones could have had a significant impact on P content in the litter after 12 months. There was a strong relationship between initial C:P in the senesced tissue and P mass of litter, as well as the initial C:N in the live tissue and the P mass in the litter exposed to upland, transitional and edge hydrologic regimes after 2 months. However, after this moderate time frame, any relationship between initial substrate quality parameters and P content was marginal. These results suggest that P release could not be reasonably predicted beyond a period of 2 months.

Differences in P release among species during the first 4 months of this study suggest that species composition of a wetland may significantly influence P storage capacity in litter and soils. During the first 4 months after senescence, a marsh dominated by *P. hemitomon* would have significantly lower P release rates than a marsh dominated by *P. hydropiperoides*. The large amount of P released by *P. hydropiperoides* after senescing could indicate that much of the P in this species was associated with the NDF fiber fraction and therefore relatively labile compared to the P contained in *P. hemitomon* which may be associated with residual fiber fraction and remain incorporated in the litter for a longer period of time. Because wetland species often have different hydrologic tolerances, changes in hydrologic regime of a wetland may result in changes in species dominance within the wetland. The relationship between species specific P release rates and factors that may result in shifts in species composition could have significant effect on water quality of the isolated wetland and potentially influence efforts to address P loading to Lake Okeechobee.

3.4 Phosphorus assimilation and equilibrium phosphorus concentrations of wetland soils

Introduction

The objective of this study was to determine the areal P release and retention rates of isolated wetland soils and determine the water column equilibrium P concentration (EPC_w) at which soils will not retain or release P. Values such as these are helpful to water and land managers to determine appropriate concentrations for water column waters such that there is no net release and/or retention of P from soils or sediments.

Materials and Methods

Thirty six intact isolated wetland soil cores (0-30 cm) were taken using PVC tubes. Sites selected included Larson Dixie, Beaty and Paleaz ranches. These sites were selected as they represent a range of (1) soil characteristics and (2) P loading. Water for core study was collected from the Kissimmee River. Water was spiked to various concentrations: 0.3, 0.5, 1.0, and 5 mg P L⁻¹ and cores were flooded to a depth of about 20 cm with P spiked water. All soil/water cores were incubated in an insulated water bath at approximately 22° C. Cores were flooded for three runs, each run lasting 21 days.

Twenty (20) mL water samples were removed at days 1, 3, 7, 14, and 21 days from the centre of each water column during the incubation period. Core studies were undertaken three times, which we call runs. The purpose of this was to determine if release and retention decreased with time. Water samples were filtered through a 0.45 µm membrane filter and analyzed for soluble reactive P (SRP) at each day of sampling. Total phosphorus, total dissolved P (TDP), pH, dissolved oxygen and electrical conductivity (EC) were also sampled intermittently. Phosphorus release and retention rates were determined as outlined by Equation 1.

$$P_{\text{flux}} = \{(C_0 * V_0) - (C_t * V_t) / A\} / \text{days}$$

Where C_0 = concentration at time 0, mg L⁻¹; V_0 = volume of water at time 0, L; C_t = concentration at time t, mg L⁻¹; V_t = volume of water at time 0, L; A = surface area of cores, m²; and days = number of days incubated.

Phosphorus flux (at respective P concentrations; 0.3, 0.5, 1, and 5 mg P L⁻¹) during the 21 day flooding period was plotted against the concentration of soluble reactive P in the water column. The equilibrium phosphorus concentration of the water column (EPC_w) was the x axis intercept, when P flux was 0 (meaning the concentration at which point there is no release or retention of P).

Results and Discussion

Water column concentrations of SRP increased at similar rates during runs and between wetland soils collected from Beaty and Peleaz. Concentrations increased similarly at all P spike concentrations (Figure 42). However, the increase in water column SRP concentrations from soils collected at Larson Dixie increased the greatest at P spike concentrations (0.3, 0.5, and 1 mg P L⁻¹). In soil-water cores that were spiked at about 5 mg SRP L⁻¹ concentrations changed little between the first and final day. Further, concentrations were similar between the three runs. At this concentration level, there was little difference between sites.

Using the changing phosphorus concentrations with time in the water column of cores, phosphorus flux from soil to water was determined. Soils all released P, under all P spike concentrations (Table 35). Larson Dixie soils fluxed greatest amounts of P under most P spike concentrations suggesting that these soils were more P impacted relative to the other two wetland site soils. Beaty and Paleaz soils fluxed similar amounts indicating that these soils were similarly impacted by historical P loads, as they responded similar to overlying water column P concentrations.

Typically, as P concentrations in overlying water increase, the P retention by underlying soils also increases (Reddy et al., 1999; Dunne et al., 2006). However, we observed the reverse; for example at a P spike of 5 mg P L⁻¹ P flux, rather than P retention was greatest. All soils fluxed P to overlying water rather than retained P from overlying water. Between runs there was generally no difference between P flux rates; however as runs increased P (time) flux decreased from Larson Dixie wetland soils (Figure 43). Typically, less P is fluxed as flooding cycles (runs) increase. This is due to a decrease in the pool of solubilised P, stored in wetland soils (Pant and Reddy, 2002).

The main objective of this task was to determine the EPC_w of wetland soils sampled. Depending on site, it was found that EPC_w varied between 1.2 and 4.6 mg SRP L⁻¹. Larson Dixie soils had the greatest EPC_w value suggesting that it was most P impacted relative to the other sites. For soils to retain P rather than flux P, the P concentrations in overlying wetland water would have to be greater than the EPC_w values shown in Table 36.

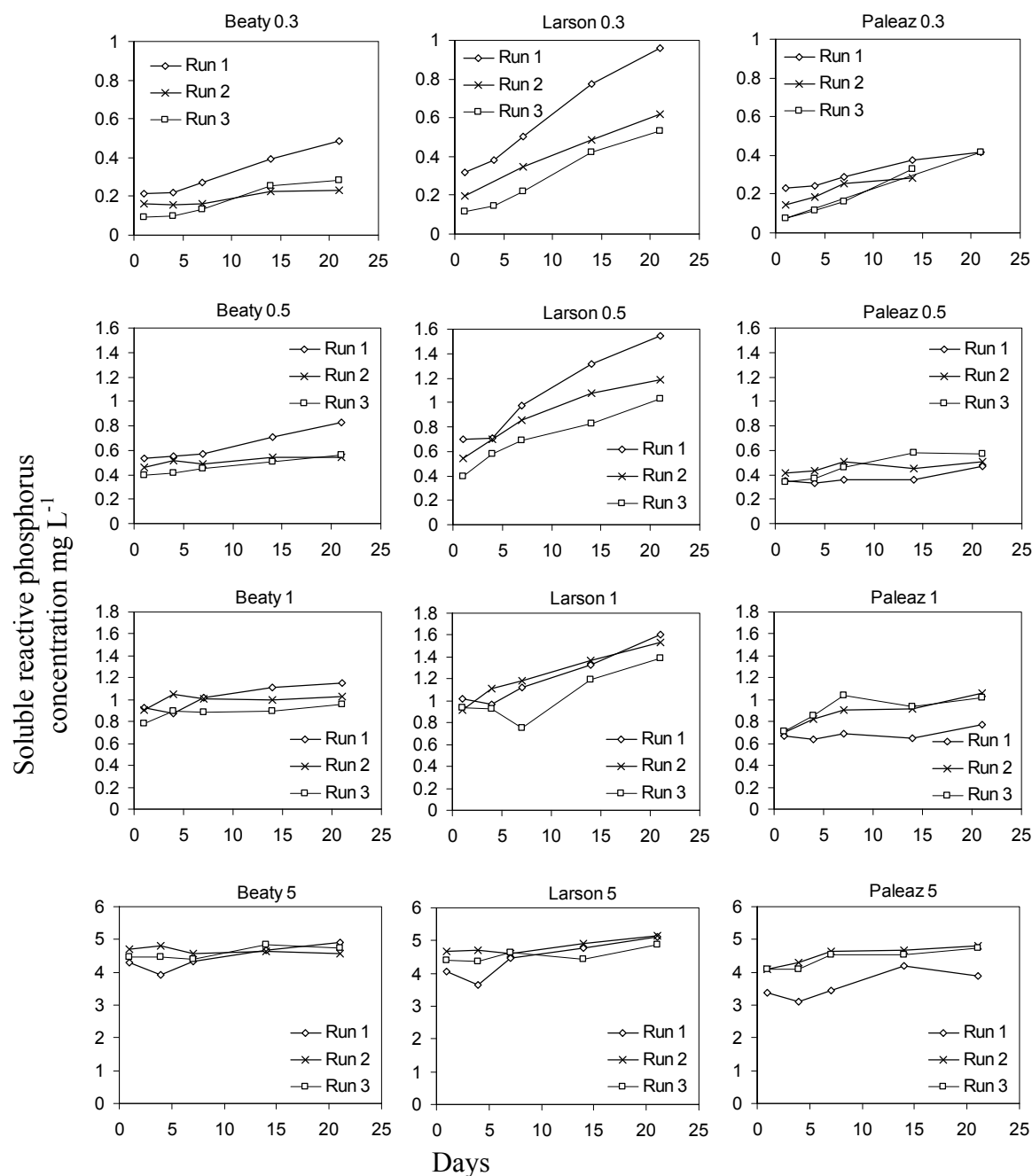


Figure 42. Phosphorus concentration change in water column of intact soil-water cores collected from Beaty, Larson Dixie and Paleaz ranches. Intact cores were incubated under flooded conditions for 21 days and were initially spiked at various P concentrations (0.3, 0.5, 1, and 5 mg SRP L⁻¹). Intact soil cores were flooded for three runs, each lasting 21 days.

Table 35. Phosphorus flux from wetland soils that had overlying water spiked at various concentrations of phosphorus. Flux values represent mean values for the three 21 day runs ($\pm$ one standard error).

Site	P spike	Flux	R ²
	mg L ⁻¹	---mg m ⁻² d ⁻¹ ----	
Beaty	0.3	2.185 $\pm$ 0.40	0.823
Beaty	0.5	1.882 $\pm$ 0.73	0.890
Beaty	1	1.197 $\pm$ 0.52	0.512
Beaty	5	4.634 $\pm$ 1.47	0.710
Larson Dixie	0.3	5.531 $\pm$ 0.47	0.983
Larson Dixie	0.5	6.797 $\pm$ 0.84	0.931
Larson Dixie	1	6.344 $\pm$ 0.79	0.913
Larson Dixie	5	7.058 $\pm$ 1.92	0.732
Paleaz	0.3	2.491 $\pm$ 0.26	0.896
Paleaz	0.5	0.906 $\pm$ 0.73	0.683
Paleaz	1	1.141 $\pm$ 0.99	0.639
Paleaz	5	6.997 $\pm$ 1.17	0.474

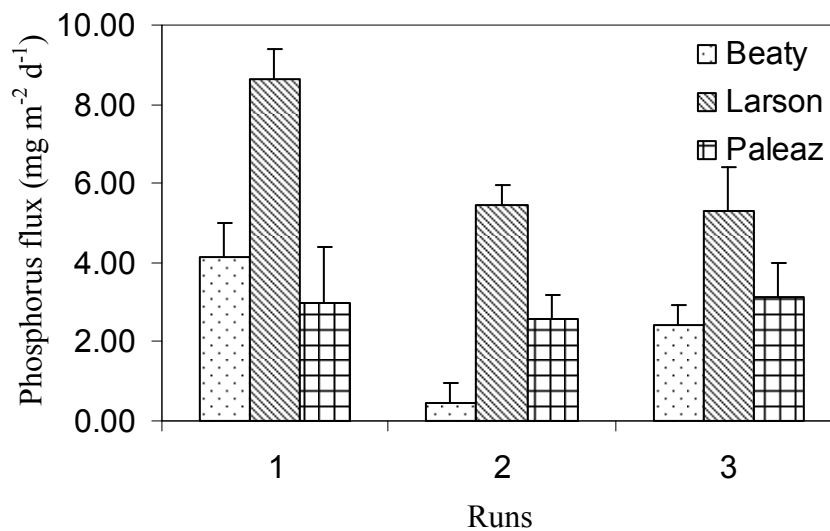


Figure 43. Phosphorus flux ($\text{mg m}^{-2} \text{d}^{-1}$) from soil to overlying water during the three 21 day runs.

Table 36. Equilibrium phosphorus concentrations (EPC_w) of wetland soils. Soils were incubated for 21 days under flooded conditions at various P concentrations (0, 0.5, 1, and 5) for several runs.

	Beaty		Larson Dixie		Paleaz	
n	13		9		11	
$EPC_w \text{ mg L}^{-1}$	1.89	± 0.70	4.59	± 1.20	1.22	± 0.40
$K_d \text{ L m}^{-2}$	4.60	± 5.80	15.55	± 5.10	26.73	± 4.50
R^2	0.832		0.780		0.870	

3.5 Phosphorus assimilation as influenced by seasonal fluctuations in hydrology

The details of this task were described in the Annual report submitted for 2004. In summary below here are the main objectives and findings of this task.

Objectives of this study were to (i) quantify soil P forms, (ii) determine effects of soil characteristics on P release, and (iii) determine effects of antecedent soil hydrological conditions on P dynamics during flooding.

Findings indicate that labile and inorganic P fractions increased as a percentage of total P with soil depth. Humic/fulvic acid bound P and residual P fractions accounted for the majority (> 78%) of TP in surface soils. Soils with the highest nutrient status and greatest labile fractions of P fluxed P to overlying water at greatest rates during 60 days of continuous flooding. During an additional 28 day flooding period, soils with overlying water columns spiked at 1 mg soluble reactive P (SRP) L⁻¹ retained and/or released P, whereas those spiked at 0.1 mg SRP L⁻¹ generally released P. Pre-flooded Larson Dixie columns spiked at 0.1 mg SRP L⁻¹ had lower P release rates than pre-saturated and pre-drawn down soils of the same site, suggesting that antecedent hydrological conditions had an effect on P dynamics between soil and water.

During the second 28-day flooding period, soils with overlying floodwaters initially spiked at 1 mg SRP L⁻¹ generally retained P (up to 8 mg SRP m⁻² d⁻¹), indicating that flooding period, antecedent hydrological conditions had little impact on P dynamics, with P levels in overlying waters governing P dynamics between soil and water.

The practical implications of this work indicate that increasing an isolated wetland system's flooding periods by hydrological restoration, should take into account short-term effects of site-soil P characteristics and antecedent soil hydrological conditions. If soils in isolated wetland ecosystems are saturated or water levels are below the soil surface and are then flooded due to heavy rainfall events, or during seasonal changes from a dry to a wet season, it may be necessary to store wetland waters for P retention by ditch management. Alternatively, if soils are previously flooded (going from wet season to dry season) it maybe appropriate to allow water flow from isolated wetland to ditch, by management of wetland water levels to provide additional landscape storage. Further, when soils are flooded for longer periods, concentrations in overlying water may control P dynamics rather than inherent soil characteristics and/or antecedent hydrological conditions.

5.0 Phosphorus budgets for isolated wetlands

Introduction

The objectives of this task were to develop a phosphorus budget for isolated wetlands, to determine potential export to receiving water bodies, and identify the treatment potential of these types of wetlands using a first-order uptake approach (e.g., Kadlec and Knight, 1994). This study will describe the extent to which these wetlands may currently function for retaining phosphorus and includes Task 3.6 (Subsurface P transport monitoring) and Task 5 (Modeling P storage and retention).

The current retention dynamics of these wetlands may help to determine their utility for landscape-scale P reduction, as well as their role among all other BMPs at the basin-scale to reduce P load to the lake. Finally, these results offer insights into the future role these wetlands may have in terms of retaining water and nutrients.

Treatment potential determined from the P budget will also be compared to the P transport results from the two-dimensional landscape-scale distributed hydrologic model of Perkins et al. (2005) (also included in the Annual report of 2005) which also incorporated first-order uptake processes. That study indicated that wetlands may reduce landscape-scale P loading between 19% and 28%, assuming steady-state flow-through the wetlands. However, this assumption likely does not adequately represent the hydrologic behavior of these systems.

Hydrologic Inflows/Outflows

During the two years of data collection at Larson Dixie and Beaty ranches, four significant tropical depressions affected site hydrology. While tropical disturbances are a reasonable phenomenon to include in hydrologic studies (in this part of Florida), the number of large storms that occurred during the data collection for this task was atypical. Therefore, this data may represent a “worst-case” scenario for nutrient runoff, which may be desirable in design parameters of water and P control strategies.

Hydrologic data collection and water budget component calculation is described in detail in previous tasks (3.3.1, 3.3.2, and 3.6). A brief summary of that research is presented here.

The depth-based water budget for these wetlands was defined as:

$$dH^{wet} = P - ET - D + S \pm GW \quad (\text{Equation 1})$$

where dH^{wet} is the change in wetland surface water storage, P is rainfall, ET is evapotranspiration, D is surface water outflow (through ditches), S is overland flow, and GW is groundwater inflow or outflow. Rainfall and dH^{wet} were calculated from direct measurement, while ET was estimated using an empirical function using meteorological data (Allen et al., 1998).

Before the 1950s, wetland surface water was hydraulically disconnected from receiving water bodies (Flaig and Reddy, 1995). Once cow-calf and dairy operations started, the forests and cypress domes were converted to pasture land. Along with this land use conversion, the hydraulic management of pastures emphasized drier pasture conditions. Networks of ditches (no more than one meter deep) were created to drain surface water from the pastures and maintain lower groundwater levels. These ditches are not completely effective, as the degree to which they were excavated was often more shallow than wetland. This inefficient ditching was intentionally created to yield a semi-drained hydraulic management regime, where some surface water could be used for cattle cooling ponds and drinking water, while increasing the land area available for grass. Because of the low gradients in the watershed (<1% typical slope), water control structures were difficult to implement, due to backwatering effects and inadequate hydraulic head-loss to measure flow. The ditch flow out of the wetlands was left as a residual term in the water budget equation. During times when ditch flow occurred and rainfall (and by definition overland flow as well) did not, Equation 1 was used to determine the ditch flow (as ditch flow was the residual term). Manning's surface water flow equation was calibrated using these data to fit a ditch slope that best described the ditch flow estimates from equation 1. Thus, ditch outflow became a known parameter. Ditch flow was a significant portion of the total outflows (approximately 50%).

Daily groundwater outflow from the wetland surface water to the uplands was determined as the residual term in a constrained form of equation 1, where only ET and dH^{wet} occurred. These groundwater outflow values were used in a linear regression model to determine the best fit of a hydraulic resistivity coefficient as part of a linear flux law (a form of Darcy's law). The calibrated linear flux equation was then applied to all other times in the monitoring record and assumed to be a known component. The same resistivity was also applied to groundwater inflows, assuming isotropy. Groundwater flow between the wetland and upland was assumed to occur radially, because differences in upland water table elevation around the perimeter of the wetland were not considerably different. Groundwater outflow from the wetland represented approximately 30% of the total outflow from the wetland.

Surface runoff to the wetland, or overland flow, was not explicitly measured in the field. This term was the unconditional residual term in equation 1. It was also one of the largest hydrologic inputs to the wetland, which was important in the overall conceptual model of these wetlands (approximately 50% of the total inflow to the wetland). The initial assumption was that the wetland was a flow-through wetland, receiving principally regional groundwater flow.

Phosphorus Budget

Each water budget component had an associated chemical counterpart, as described by the P budget:

$$\frac{d(V_W C_W)}{dt} = M_P + M_S - M_D - M_{ET} \pm M_{GW} \pm \varphi \quad (\text{Equation 2})$$

where V_W is the wetland water volume (L^3), C_W is the P concentration in the wetland surface water ($M L^{-3}$), M_S is the P flux in overland flow ($M T^{-1}$), M_{GW} is P flux in groundwater to and from the wetland ($M T^{-1}$), M_P is the P flux in precipitation ($M T^{-1}$), and M_D is the P flux out of the wetland via the ditch ($M T^{-1}$). Assuming the first-order uptake kinetics, φ ($M T^{-1}$) was defined as:

$$\varphi = (C_{init} - C^*) \cdot \exp\left[-\frac{\kappa}{2H^{wet}} \tau\right] + C^* \quad (\text{Equation 3})$$

where C_{init} is the initial concentration in the wetland surface water ($M L^{-3}$), C^* is the equilibrium surface water concentration ($M L^{-3}$), κ is the first-order uptake rate coefficient ($L T^{-1}$), τ is the mean hydraulic residence time (T), and H^{wet} is the wetland surface water elevation (L).

The P concentration in the wetland surface water was measured from grab samples collected approximately monthly during the study period and more intensely for the last few weeks of the study. Groundwater samples were drawn from 2-m deep, fully-screened groundwater monitoring wells along the perimeter of each wetland at a similar time interval. Total P values for ditch outflow were averaged from water quality samples from the ditch during and after runoff events.

Concentrations of TP in rainfall P and runoff were based on those reported by Hiscock et al. (2003) in the same basin. Of course some variability in overland flow concentration would be expected, depending on land use and intensity of chemical management. Hiscock et al. (2003) reported that P concentration in overland flow from high intensity dairy farms may exceed 10 mg L^{-1} ; however, for improved pastures, such as the ones being investigated in this task, a more appropriate overland flow P concentration was reported as 1.32 mg L^{-1} . This value was used exclusively for calculations in the current work, as no values were measured onsite.

One of the earliest reported wetland models was presented by Kadlec and Hammer (1998). This self-proclaimed simple model was too extensive for application to this study, implementing just less than 50 input parameters. Thus an even simpler approach was taken to characterize P retention. First-order uptake kinetics was used, as it is very simple to apply and does not require in-depth knowledge of the wetland P cycle processes and dynamics. First order kinetics, as represented by the k-C* model have been used extensively to model treatment wetland dynamics (Kadlec and Knight, 1994). The first-order approximation of P uptake has been applied with success in modeling chemical uptake in treatment wetlands (Dortch, 1996; Kadlec, 1999; Sérodes and Normand, 1999; and Wang et al., 2006).

k-C* Modeling

Using equation 3, a generalized Reduced Gradient nonlinear optimization code (Microsoft Excel Solver) was used to determine the best fit between ϕ and wetland TP concentrations while varying the uptake coefficient, κ over the entire period. A value of 0.05 mg L^{-1} was initially used to represent C^* , which was the mean P equilibrium concentration measured by Pant et al. (2002) in a newly constructed treatment wetlands in a wetland in the Okeechobee Basin that was proposed to be converted into a stormwater treatment area. However, onsite wetland water-soil P information was also incorporated into this task to evaluate a range of resulting treatment efficiencies. Onsite (Larson and Beaty ranches) soil column studies, reported that values for equilibrium P concentration (EPC_w) ranged between 0.12 and 1.3 mg L^{-1} ; however, this depended on the P concentration of overlying and antecedent hydrological conditions (depending on whether soils were previously flooded, saturated or dry) of soils (Dunne et al., 2006).

A value of 1.3 mg L^{-1} was used for C_{init} , which is a value of TP concentrations in runoff from improved pastures in the Okeechobee basin reported by Hiscock et al (2003). In reality, the concentrations associated with runoff from each wetland was probably different; however, based on the information available, the value reported by Hiscock et al. (2003) was used. The Larson ranch is more intensely managed compared to the Beaty ranch, and would therefore be expected to exhibit higher concentrations of P in overland flow. Because EPC_w values observed at the sites for saturated initial conditions were relatively close to the value assigned to P concentration in runoff (Dunne et al., 2006), the treatment effectiveness of these wetlands would likely be reduced significantly from what is reported in Table below. From the P budget dynamics in Figure 44 and 45, these wetlands may behave as sources of P, rather than sinks. The k-C* modeling effort also reflects this, as the EPC_w values of BW1, BW2, and LW2 are all greater than the average wetlands P concentration.

Because of the non-linear and transient behavior of the wetland residence time behavior, it was important to incorporate the specific wetland surface water residence times (τ) for each day that P treatment effectiveness was estimated. The values for τ were the nominal residence times (volume of wetland divided by the sum of the hydrologic outflows) derived from daily calculations using the water budget (see previous tasks).

Results and Discussion

Phosphorus Budget

Average concentrations of TP measured in surface water, groundwater, and ditch flow at each study wetland are reported in Table 37. The surface water and ditch flow concentrations were not significantly different; however, the groundwater concentrations were significantly different from both surface and ditch water TP concentrations. Results from the P budget are reported in several ways (Table 38): percent of the total TP mass input/output, to compare the relative P fluxes over the study period, mean and coefficient of variation of daily flux for each component, to identify important loading rates and their duration, and TP fluxes are scaled to the Lake Okeechobee basin to compare loading

rates from the sites to observed loads to the lake. While scaling up P loading results was not completely within the scope of this study, it was useful to identify compare linearly scaled P loading estimates from this study to larger scale TP loading estimates to the lake (derived from multiple sources within the basin (Bogges et al., 1995)).

Table 37. Water quality measurements of total P in wetland surface water, ditch water, and groundwater.

Wetland	Surface Water		Ground water		Ditch water	
	Mean [mg/L]	CV	Mean [mg/L]	CV	Mean [mg/L]	CV
LW1	1.2 (N=27)	0.4	0.6 (N=35)	0.9	0.9 (N=109)	0.5
LW2	1.0 (N=16)	0.4	0.5 (N=31)	0.7	0.8 (N=170)	0.6

Table 38. Percentage of total P contribution from each chemical component, mean TP daily flux, and TP loads scaled to the Lake Okeechobee basin. Percent efficiency is also shown as a fraction of total P outflows to the total P inflows.

	%M _P	%M _{GWin}	%M _S	%M _D	%M _{GWout}	%Uptake
LW1	0.8	1.3	97.8	43.7	46.1	10.2
LW2	1.7	0.5	97.8	76.2	18.3	5.1
BW1	2.5	2.6	94.9	9.8	7.6	82.6
BW2	0.0	0.1	99.9	43.3	14.8	42.0

Mean flux	M _P	M _{GWin}	M _S	M _D	M _{GWout}	Uptake – Release
LW1 [mg d ⁻¹]	3.0	4.0	310	138	146	284 – 251
CV	2.9	0.6	2.7	1.1	0.6	2.3 – 1.0
LW2 [mg d ⁻¹]	1.0	<1	84	65	16	266 – 99
CV	4.1	1.1	2.7	2.0	1.3	0.9 – <1
BW1 [mg d ⁻¹]	1.0	1.0	31	3	3	63 – 7
CV	2.6	1.4	1.9	1.5	1.8	0.2 - <0.1
BW2 [mg d ⁻¹]	<1	<1	15	7	2	34 – 11
CV	1.8	1.3	2.7	1.5	1.0	0.5 – <0.1

Cumulative mass of P (g) from the wetland P budgets at Larson and Beaty wetlands are shown in Figures 44 and 45 respectively. Also shown in these plots is the uptake/release variable (ϕ) from Eqn. 2, but was calculated as the residual term in the daily P budget and not by the k-C* model given in Eqn. 3. The dynamics of each component are important to identify, as well as any differences between wetlands. The temporal dynamics of the P budget were driven by hydrology, as mean P concentrations were used to represent all times of each component. For example, P mass associated with overland flow (M^{over}) exhibited periodic and relatively infrequent behavior, compared to mass of P corresponding to groundwater flow out (M^{gwout}). Mass of P associated with rainfall (M^P) was slightly more frequent, compared to M^{over} , but was also event-driven. Perhaps a more interesting observation is that ϕ is temporally variable, where a wetland might behave as a P sink and then a source, depending on the dynamics at the time.

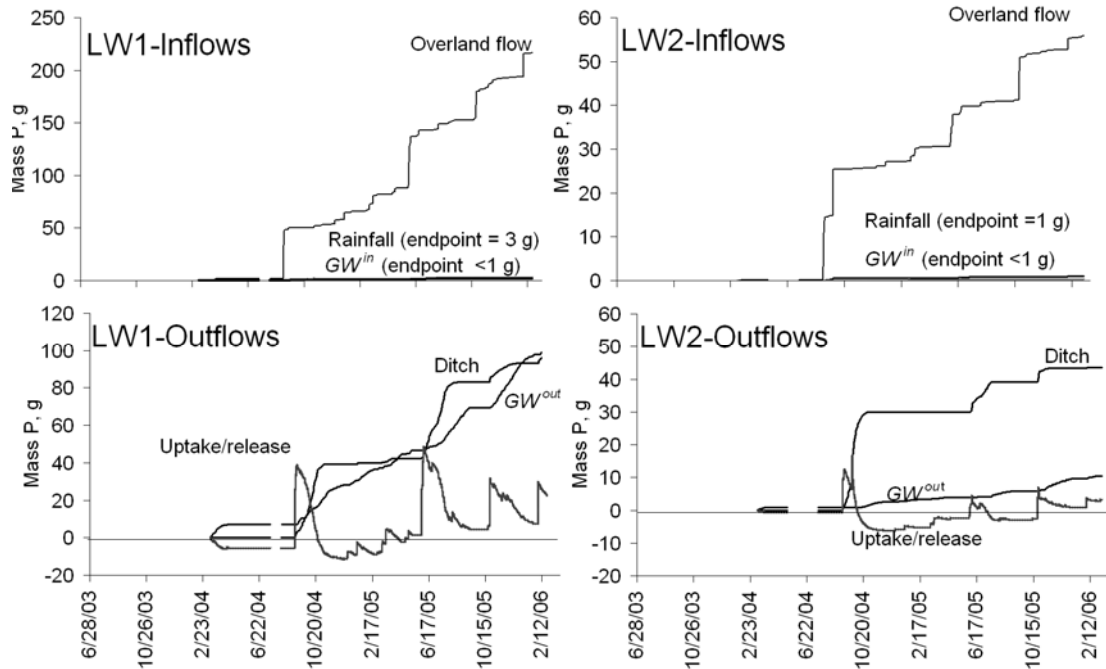


Figure 44. Cumulative mass of P with time for the Larson wetlands.

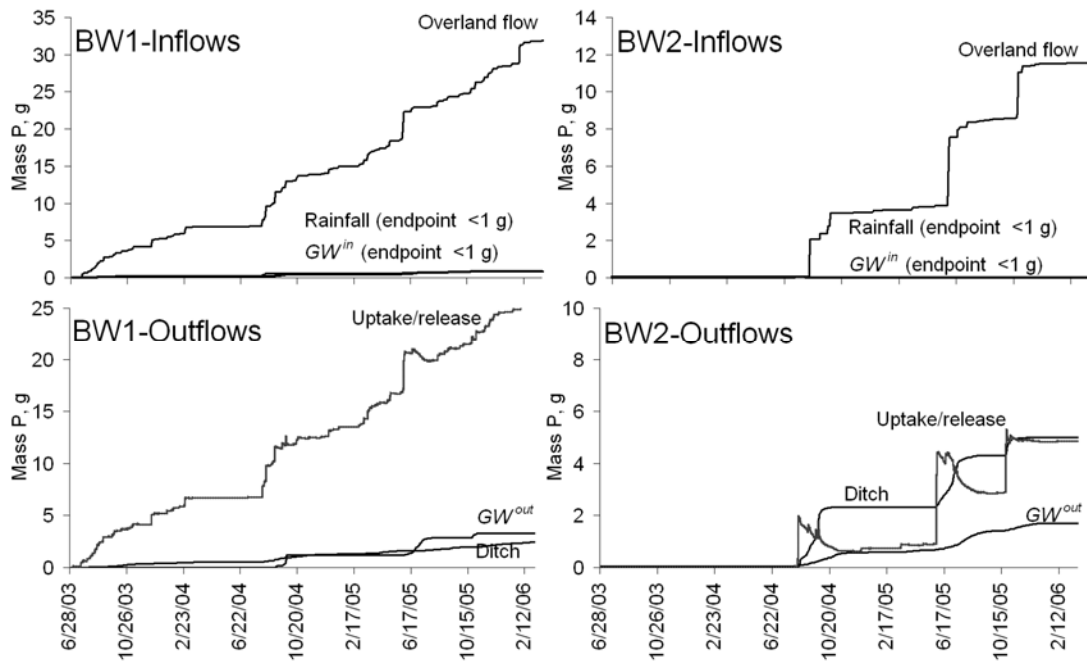


Figure 45. Cumulative mass of P with time for the Beaty wetlands.

Phosphorus temporal dynamics at LW2 appear to be quite different from the other wetlands, as the mass P associated with ditch flow (M^{ditch}) nearly parallels the M^{over} over the entire record. It is easy to see that, with the other components of the P budget do not

significantly contribute to the overall P dynamics, P that enters the wetland via overland flow exits the ditch almost immediately. This inflow/outflow dynamic appears to be damped in the other wetlands.

The amount of P input to the wetland was overwhelmingly dominated by overland flow (Table 38). While water volumes of overland flow and rainfall to the wetlands were similar (approximately 50% each) the higher P concentration associated with overland flow was the driving force of P loading to the wetland. Better estimates of P loading to the wetland via overland flow are necessary to determine the P budget. The results from this study will build the conceptual framework for such an analysis.

While groundwater outflow comprised approximately 35% of the total hydraulic outflow from the study wetlands, the principal component responsible for the majority of P export was ditch flow, which yielded between 7.6 and 43.7 % (wetland-averaged) of the total TP outflow. This finding is critical in the development of a conceptual model of the P dynamics because the treatment effectiveness is a function of not only what comes out the ditch as surface water export, but also the load to groundwater that will ultimately be conducted to the lake. The wetland BW1, which acted most like a restored wetland, was highly influenced by the amount of P load coming to the wetland associated with overland flow, but also had a considerably larger uptake compared to the other wetlands (83% of the total outflows). Total P loads associated with rainfall and groundwater inflow were not significant pathways. The opposite was true for groundwater inflow, concentrations of P in the pore water were considerable, but the occurrence of groundwater inflow was not high enough to transport comparable amounts of P to the wetlands.

Daily loading rates of P for each wetland flow component (Equation 2) were important to characterize, because the variability of the system dynamics may be inferred. These data may be useful for determining appropriate management strategies aimed at reducing P export from these fields. When implementing management strategies one needs to take account of the relatively rapid nature of ditch flow and overland flow. While these events are not as frequent as the others, their associated P flux out of the system is nearly an order of magnitude greater.

Using the product of wetland ditch outflow rates from Table 38 and the area of Okeechobee Basin corresponding to isolated wetlands (16%), an estimate of basin-scale P load was calculated. Values between 0.2 and 9.6 metric tonnes per year were estimated represented a significantly smaller estimate compared to basin-scale estimates of 415 tonnes yr⁻¹ (Boggess et al., 1995) and 582 tonnes yr⁻¹ (SFWMD, 2001). The estimate in this work assumes that all historically isolated wetlands function hydrologically similarly within the landscape and have had similar historic and current land use practices (fertilizer, cattle stocking, etc.). This estimate of basin-scale load is only pertinent to the fraction of runoff passing through depressional wetlands and out ditches. It does not incorporate P load related to regional drainage from ditch networks, land use contribution of dissimilar nature (dairies and other relatively high-intensity operations), and P load from tributary and in-stream internal loads.

The P treatment effectiveness listed in Table 39 is derived from the sum of chemical outflows divided by the sum of the chemical inflows (similar form as in equation 5). When the term “treatment” is used in this document, it is somewhat different from the strict definition of the word as commonly used in analysis of constructed wetlands. Constructed wetlands do not usually consider the export of P via groundwater discharge from the wetland, as they are designed to allow minimal groundwater export. However, in the wetlands of this study, groundwater out flow may be considerable and carry a significant P load. When the term treatment is used, it is the sum of the outflows (via ditch and groundwater) divided by the sum of the outflows (rainfall, overland flow, and groundwater inflow) (Table 39). Therefore, the treatment is a measure of not only the amount of P exiting the ditch, but also what is exiting with groundwater. The estimates of treatment would increase if groundwater outflow was not considered in the term, but would not represent the hydrology of these systems.

Table 39. Input and optimized parameters in k-C* model and associated error.

Site	C_{init}	C^*	k (m d ⁻¹)	k-C* based % treatment effectiveness	P budget based % treatment
LW1	1.3	0.05	0.003	10	10 ± 0.25
LW2	1.3	0.05	0.001	5	6 ± 0.34
BW1	1.3	0.05	0.035	92	83 ± 0.05
BW2	1.3	0.05	0.005	42	42 ± 0.30

These data suggest that these wetlands may sequester P, even though several extreme hydrologic events occurred during data collection that may have skewed results. An average of 8 % of the total P inflow was assimilated in the Larson wetlands, while an average of 63% of P entering the Beaty wetlands was retained (and assumed to be sequestered). For BW1, this might be due to the fact that ditch flow occurs less frequently, allowing P chemical transformation and sequestration more productive. Also, wetland vegetation might be more productive under the more predictable (i.e. slightly wetter hydrologic conditions) at the Beaty wetlands.

k-C Model*

The optimized values of C^* , C_{init} , and κ for each wetland, as determined from equations 2 and 3, are shown in Table 39. The model fits were evaluated based on comparisons to the time series of measured P concentrations in the wetland surface water (Figure 46). Root mean square error (RMSE) of the modeled and measured surface water concentrations for LW1, LW2, BW1 and BW2 was 0.35, 0.32, 0.05, and 0.58 mg L⁻¹ respectively. This level of error in prediction was acceptable for the objectives of this study, given that mean measured surface water concentrations were generally larger than their corresponding error (Table 39).

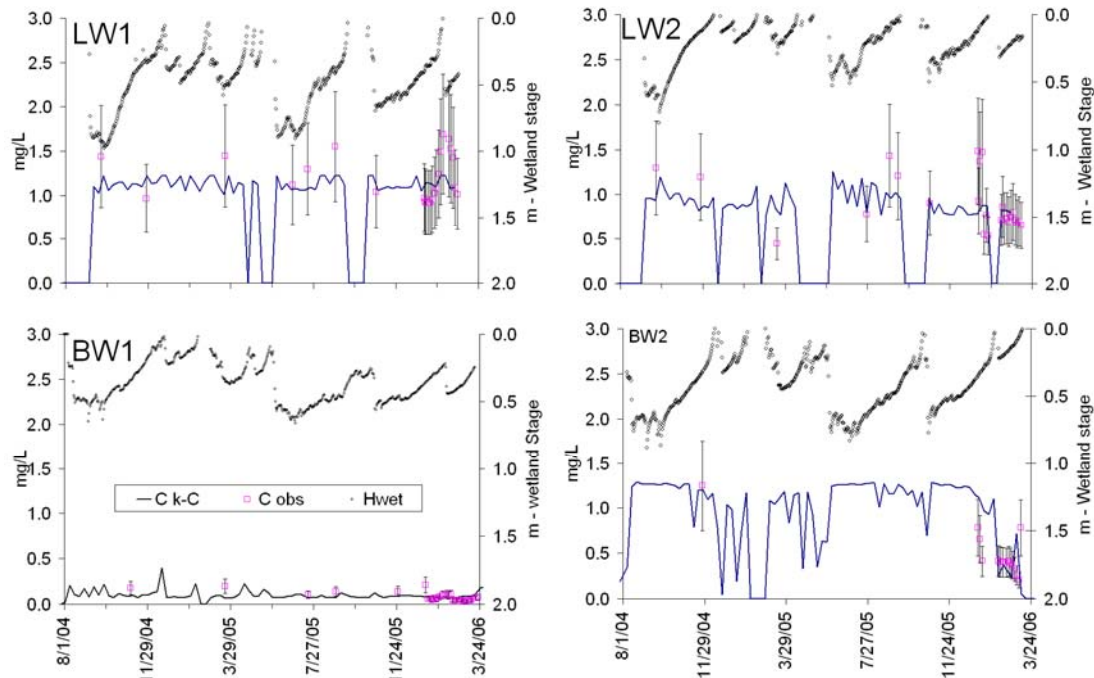


Figure 46. Results from k-C* model compared to measured P concentrations. A) Larson wetland-1. B) Larson wetland-2. C) Beaty wetland-1. D) Beaty wetland-2

Conclusions

A phosphorus budget was estimated for four isolated wetlands (two in Larson Dixie ranch and two in Beaty ranch) in the Lake Okeechobee Basin to (i) identify important P flow pathways and (ii) quantify their effect on downstream water quality. Phosphorus associated with overland flow was the dominant mass input to these systems, where P from groundwater inflow and rainfall combined, accounted for less than 10% of the total P inputs. Groundwater output of P was proportional to the P flux associated with ditch outflow in two wetlands, suggesting that groundwater outflow is an important part of the conceptual P budget model. Phosphorus export associated with ditch outflow was either comparable to or more significant than groundwater outflow for each wetland.

The effectiveness of the study wetlands to reduce P load was only significant for BW1 wetland, where ditch flow P was constrained by the infrequency of ditch water flow. All other wetlands exhibited insignificant uptake of P. A non-trivial error may be associated with these estimates due to the lack of time-intensive P sampling and further work should focus on concentrations and flow of overland flow to the wetland.

The k-C* modeling effort produced acceptable levels of error considering the somewhat infrequent measured P concentrations in the wetland surface water. The fit parameters may be further validated with future measurements of more frequent water quality data. In this study, commencement of higher frequency water sample collection coincided with dry hydrologic site conditions.

Phosphorus treatment in these wetlands was nearly what was estimated and described in detail in tasks of the annual report of 2005 using the landscape-scale pseudo-2d modeling approach, except for BW1, as findings suggested that this wetland was capable of treating future P loads to the wetlands before export. Under current managed wetland hydrology, the utility of these wetlands to retain P may not result in P load reduction, but with measures to increase retention time in the wetlands; higher treatment potential might be expected. The duration of P treatment under a modified hydraulic regime, such as blocking the export of P loading via ditches, is not known and would require a more in-depth look at the mechanisms and limits of the P cycle.

4.0 Evaluation of passive and actively managed treatment wetlands for phosphorus removal from dairy and basin-wide runoff

Introduction and Background

Wetlands currently are one of the most promising technologies for use in controlling nutrients from agricultural operations. Constructed wetlands designed to treat nutrients and other contaminants are usually managed to enhance sustainability, and to consistently achieve desired levels of outflow quality. Management scenarios can vary, depending on factors such as the type of wetland and the contaminant loading rates. The simplest wetland management is related to the control of desired hydraulic and contaminant loading rates. A slightly more complex level of management is the selection of vegetation type within the wetland, which may require planting of target species, and the control of undesirable species through the use of herbicides or water depth manipulation. Periodic sediment management is another maintenance activity that can improve wetland sustainability, in which drydown, dredging or chemical inactivation of accreted sediments is practiced. The most intensive wetland management approach entails routine harvesting of a productive macrophyte or periphyton crop. Because routine harvesting can produce enormous quantities of biomass, it is mandatory to identify an economically viable product for this material. At dairies, for example, harvested biomass may find a use as a cattle feed ingredient, substituting for green chop.

In the Okeechobee region, both on-farm wetlands and larger, regional treatment wetlands are being proposed as a method of reducing phosphorus (P) inputs to Lake Okeechobee. The purpose of this effort is to evaluate the effectiveness of constructed wetlands for P removal from both low nutrient (basin runoff) and high nutrient (lagoon wastewaters and runoff) source waters. Wetlands used for both purposes will need to be cost effective, requiring designs that maximize areal phosphorus removal rates in order to minimize wetland footprints. This report describes our “year four” progress on demonstrating and optimizing on-farm and regional treatment wetlands using both passive and active management approaches.

General Description of Work Effort

This research project was conducted in two phases. Phase I was a mesocosm-scale effort, performed west of Okeechobee, FL at Rio Ranch, a site consisting of nearly 18 acres of improved pasture. We established mesocosm wetlands that were used to treat runoff from the L-62 canal, which drains the S-154 basin. For this mesocosm study, we assessed the performance of a spectrum of wetland and ‘wet crop’ configurations. This mesocosm effort was initiated early in 2003 and completed in mid-2004 (Annual report 2005).

Phase II has been in operation since January 2005. During this phase the most promising configurations evaluated at the mesocosm scale were tested using a larger, pilot-scale facility. The facility is located at a dairy (Larson Barn #5) on Williams Road, east of the City of Okeechobee. The pilot wetlands receive water pumped from the third stage of a

three-stage treatment lagoon. The facility is comprised of 12 smaller wetlands, each approximately 20 m long by 5.5 m wide. In these pilot wetlands, we are evaluating the P removal performance of conventional wetland vegetation.

Originally, two replicate wetlands containing cattail, cattail followed by submerged aquatic vegetation [SAV], paragrass and floralta (pasture grasses), water hyacinth (floating aquatic vegetation) and rice were evaluated. The results of this comparison can be found in the 2005 Annual Report for this project (Annual report 2006). In Fall 2005, due to the inability of the floralta to thrive in the constructed wetlands, the floralta was harvested and restocked with paragrass. Early in 2006, the vegetation in the rice wetlands was mowed, re-flooded and the wetlands were replanted with torpedo grass.

Our general basis for evaluating a variety of vegetation types, including aquatic plants and agricultural crops, is as follows. Cattail and SAV represent the most common vegetation communities of the Everglades Stormwater Treatment Areas (STAs). These plant types therefore will undoubtedly play a prominent role in both on-farm as well as regional treatment wetlands (STAs) in the Lake Okeechobee watershed. For the present study, we are evaluating cattail wetlands alone, as well as cattail (front end of wetland) followed by SAV (back end of wetland). Paragrass and torpedo grass are productive grasses that grow well under wet conditions, and reportedly can be used as a cattle feed component. Therefore, shallow wetlands containing these species may provide effective P removal as well as a useful, harvestable crop. Water hyacinth has been studied extensively for use in removing P in the Lake Okeechobee watershed (DeBusk et al 1989; Hydromentia 2005), and its rapid growth rate and P uptake capabilities are well defined. In order to maintain rapid growth rates and sustain high rates of P removal, however, the vegetation must periodically be harvested to maintain a favorable standing crop for plant growth. Widespread deployment of water hyacinth technology has been limited because specialized equipment is required for harvesting, and the value of the harvested plant biomass (e.g., as compost, or a feed ingredient) usually doesn't cover the costs of harvesting and processing.

In our pilot-efforts, we are using a novel approach to deal with the water hyacinth biomass. Rather than periodically harvesting the plants from the water, we are investigating the utility of simply tilling the biomass into the soil when the standing crop becomes too dense. This is a technique that can be accomplished quickly and easily with conventional farm machinery. The soils are the ultimate storage reservoir of P in conventional wetlands, and this tilling approach is merely accelerating the rate of transferring organic matter (and associated P) into the soil.

The following sections describe our work efforts for Phase II. This report represents our research findings from January 2006 through December 2006.

Methods

Site Location and Description

The Phase II testing facility was constructed during December 2004, adjacent to one of Larson Dairy Barn #5's treatment lagoons (Figure 47). The pilot wetlands are constructed of earthen berms, and equipped with plastic liners. The finished facility consists of 12 wetlands, each approximately 20 m long by 5.5 m wide with an average surface area of 108 m² (range: 91 – 122 m²). Water from the adjacent treatment lagoon is pumped into a head tank and then gravity fed into the 12 pilot wetlands (Figure 48).



Figure 47. Location of pilot wetland facility at Larson Dairy Barn #5.

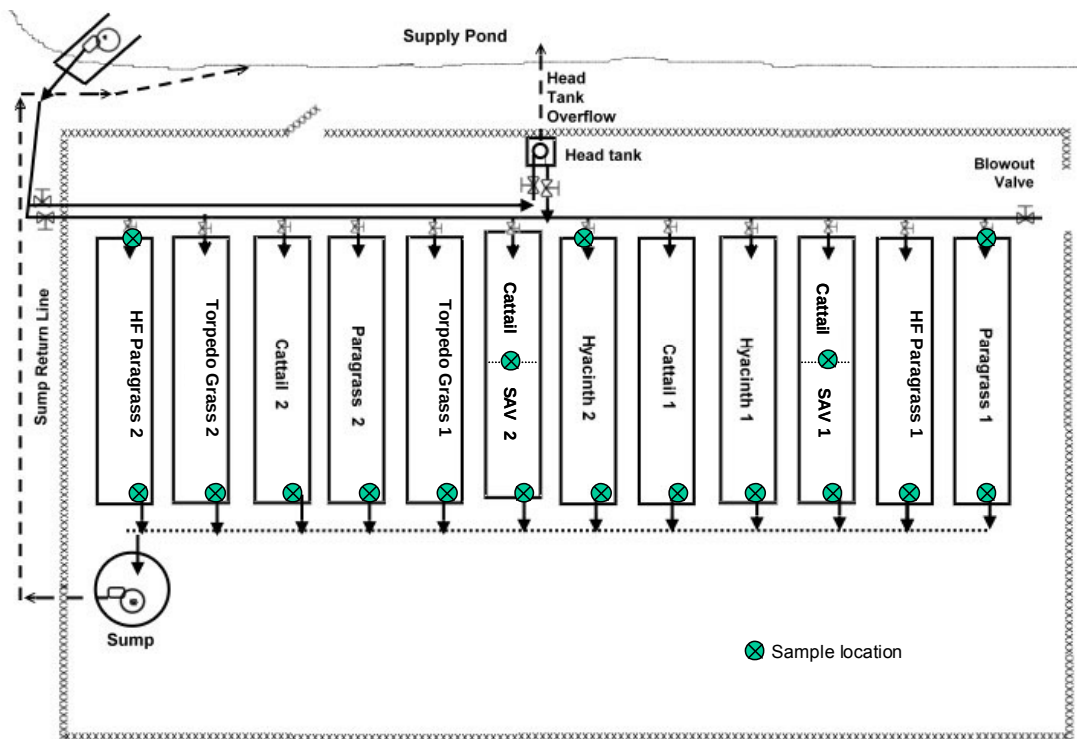


Figure 48. Schematic of the 12 pilot treatment wetlands.

During 2006, we used this facility to evaluate the P removal performance of six different wetland configurations, each operated in duplicate. Five of the configurations contain a different vegetation type, some of which require either periodic harvesting or tilling and operated at similar hydraulic loading rates (HLR). These vegetation types are cattail, cattail followed by submerged aquatic vegetation (SAV), paragrass, torpedo grass, and water hyacinth (Figure 48). The sixth configuration (Paragrass HF) has paragrass as a vegetation type but is a high flow (HF) system with an HLR approximately 3.5 times that of the other paragrass configuration.

The Cattail, Cattail/SAV, and Water Hyacinth treatments were monitored from January 2006 – December 2006. During this time these wetlands were maintained at a water depth of 40-cm. From January – October, these treatments received lagoon effluent at an average hydraulic loading rate (HLR) of 2.2 cm day⁻¹, which resulted in an average hydraulic retention time (HRT) of 17.8 days (Table 40). In order to reduce the mass P loading rate to the wetlands, on October 29, 2006 the flows were reduced five-fold, resulting in an average HLR of 0.5 cm day⁻¹ and average HRT of 89 days.

The Paragrass, Torpedo Grass, and Paragrass HF treatments were monitored from June 2006 – December 2006. During this time the water depths were maintained at 15 cm. These treatments received lagoon effluents at an average HLR of 2.6 cm day⁻¹, which resulted in an average HRT of 5.8 days. The Paragrass HF treatment had an HLR and HRT of 7.8 cm day⁻¹ and 1.9 days, respectively (Table 40). On October 29, 2006 the flows were reduced five-fold, resulting in an average HLR of 0.5 cm day⁻¹ and average HRT of 26.7 days for the Paragrass and Torpedo Grass treatments, and 1.6 cm day⁻¹ and 8.7 days, respectively, for the Paragrass HF treatment (Table 40).

Due to observed nitrogen (N) limitations of the foliage, in April 2006 Urea fertilizer (46% nitrogen 46-0-0) was broadcast at an equivalent rate of 45 kg ha⁻¹ over all of the treatment wetlands.

Table 40. Average ($\pm$ 1 S.E.) hydraulic loading rates (HLR), surface area, water depth and hydraulic retention times (HRTs) for each wetland configuration.

Treatment	Surface Area (m ²)	Water Depth (cm)	Period	HLR (cm/day)	HRT (days)
Cattail	144 $\pm$ 5	40	Jan 06 – Oct 06	2.3 $\pm$ 0.03	17.6 $\pm$ 0.3
			Nov 06-Dec 06	0.5 $\pm$ 0.01	88 $\pm$ 1.3
Cattail/SAV	117 $\pm$ 5.0	40	Jan 06 – Oct 06	2.2 $\pm$ 0.09	18.0 $\pm$ 0.7
			Nov 06-Dec 06	0.4 $\pm$ 0.02	90 $\pm$ 3.8
Water Hyacinth	155 $\pm$ 0.4	40	Jan 06 – Oct 06	2.2 $\pm$ 0.01	17.8 $\pm$ 0.1
			Nov 06-Dec	0.5 $\pm$ 0.00	89 $\pm$ 0.3

Paragrass	98 ± 4.5	15	06 Jun 06 – Oct 06	2.6 ± 0.12	5.7 ± 0.3
			Nov 06-Dec 06	0.5 ± 0.02	28 ± 1.3
Paragrass HF	101 ± 9.8	15	06 Jun 06 – Oct 06	7.8 ± 0.77	1.9 ± 0.19
			Nov 06-Dec 06	1.6 ± 0.15	10 ± 0.9
Torpedo Grass	1.2 ± 10.0	15	06 Jun 06 – Oct 06	2.6 ± 0.25	5.9 ± 0.6
			Nov 06-Dec 06	0.5 ± 0.05	29 ± 2.9

Water Quality Monitoring

Inflow and outflow water quality samples were collected at the locations shown in Figure . In general, samples were collected twice weekly and composited to form one weekly sample for analysis. Methods and sampling frequency for parameters analyzed are listed in table below. Samples for the analysis of soluble reactive P (SRP), calcium (Ca), ammonium (NH₄-N) and nitrate + nitrite (NO_x) were field-filtered at the time of collection using a 0.45 µm filter. After May 27, 2006, only weekly samples for TP, TSP and SRP were collected.

Due to a high occurrence of particles in the water samples in spring 2006 (as a result of the “high-strength” inflow waters), in May we began collecting surface water samples by using a suction pump rather than dipping the collection bottle directly into the water. While this approach did not totally eliminate particulate matter, this method did provide visibly cleaner samples.

Table 41. Water quality parameters measured during the study.

Parameter	Method	Frequency
total phosphorus (TP)	SM4500P-F	weekly
total soluble phosphorus (TSP)	SM4500P-F	weekly
soluble reactive phosphorus (SRP)	SM4500P-F	weekly
total suspended solids (TSS)*	EPA 160.2	weekly
dissolved calcium (Ca)*	EPA 215.1	every other week
Alkalinity*	EPA 310.1	every other week
chemical oxygen demand (COD)*	Hach Method 8000	monthly
total Kjeldahl nitrogen (TKN)*	EPA 351.1	monthly
nitrate + nitrite (NO _x)*	EPA 353.2	monthly
ammonia (NH ₄ -N)*	EPA 350.1	Monthly

* analysis of this parameter was terminated 5/27/06

Vegetation Management/Monitoring

Vegetation management and monitoring for this project consisted of several different approaches, depending on the vegetation type. During 2006, no vegetation harvesting or standing crop monitoring was performed in the conventional aquatic systems (cattail and cattail/SAV). The water hyacinth was tilled into the soil during late spring, after which time the floating plants were “reinoculated”. The paragrass treatments were harvested on two dates by clipping the foliage.

Water Hyacinth

The water hyacinths were tilled into the soil on May 5, 2006 using the methods previously described in the 2005 Annual Report (Annual report 2006). Due to the shallow soils and close proximity of the underground plastic liners in our wetlands, only very shallow tilling could be performed. Consequently, numerous plant fragments remained on the soil surface following tilling. The wetlands were then reflooded and 100 wet kg of water hyacinths were re-introduced to each pond.

Torpedo Grass

By February 2006, the rice treatments were no longer being sampled due to an inability to obtain clear samples in the shallow wetlands. Upon termination of the rice treatment, the plants were mowed down twice and then each wetland was flooded to a depth of about one meter to drown the remaining shoots. Due to its reported high growth rates and desirable foliage (for cows), torpedo grass was chosen as the replacement vegetation in these wetlands.

Torpedo grass was initially stocked into these mesocosms in April 2006. This initial inoculum didn't thrive, so in July 2006 several additional clumps of torpedo grass were spread throughout the wetland to encourage growth. By the end of the year, the growth still was not adequate to support a foliage harvest.

Paragrass

The high flow paragrass (Paragrass HF) wetlands were initially stocked in fall 2005 using plants retrieved from a local canal. In December 2005, these wetlands were stocked with additional foliage obtained from the adjacent “low flow” paragrass wetlands. Both the Paragrass HF and Paragrass wetlands were harvested on August 24, 2006 and October 29, 2006 using methods described in the 2005 Annual Report. Vegetation subsamples were collected for harvestable foliage dry weight biomass and tissue P analyses.

Results and Discussion

Vegetation Characteristics

Only the Paragrass and Paragrass HF wetlands were harvested in 2006, since vegetation in the torpedo grass wetlands did not exhibit significant foliage growth. The harvestable

foliage in the inflow region of the Paragrass HF treatment during August and October was 65 - 70% lower than in the Paragrass LF treatment (Figure 49). Foliage yields in the mid region of both treatment wetlands were comparable. Similar to the trend observed in the inflow region, foliage biomass in the outflow region of the Paragrass HF treatment was 57 - 63% lower than the Paragrass LF treatments for August and October.

From August through October 2006, the harvestable (foliage) biomass productivity of the Paragrass LF and Paragrass HF treatments averaged 699 and 379 $\text{g P m}^{-2} \text{yr}^{-1}$, respectively, and the P removal rate (achieved by foliage harvest) averaged 14.3 and 9.0 $\text{g P m}^{-2} \text{yr}^{-1}$, respectively (Figure 50).

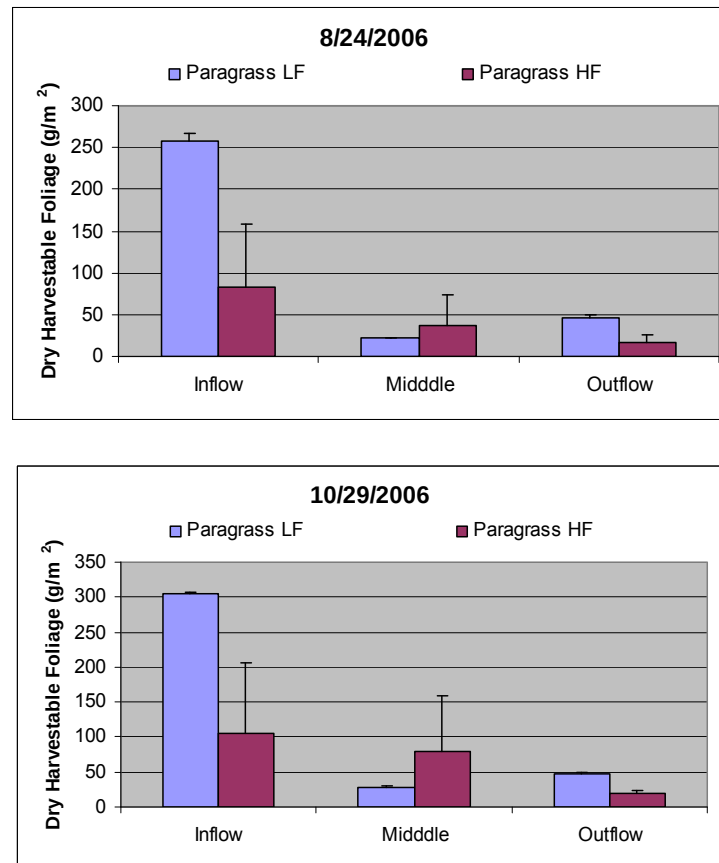


Figure 49. Harvestable foliage biomass of the Paragrass LF and Paragrass HF treatments at three locations (inflow, mid and outflow regions) within the wetlands. Measurements were performed in late August (top) and late October (bottom) 2006.

Both foliage biomass productivity and P removal rates were also calculated for the Paragrass LF systems (Figure). The average foliage productivity and P removal rates for these systems were 119 $\text{g P m}^{-2} \text{yr}^{-1}$ and 2.7 $\text{g P m}^{-2} \text{yr}^{-1}$, respectively, from September 2005 - August 2006. From August - October 2006, these rates increased approximately 80% to 669 $\text{g P m}^{-2} \text{yr}^{-1}$ and 14.3 $\text{g P m}^{-2} \text{yr}^{-1}$, respectively.

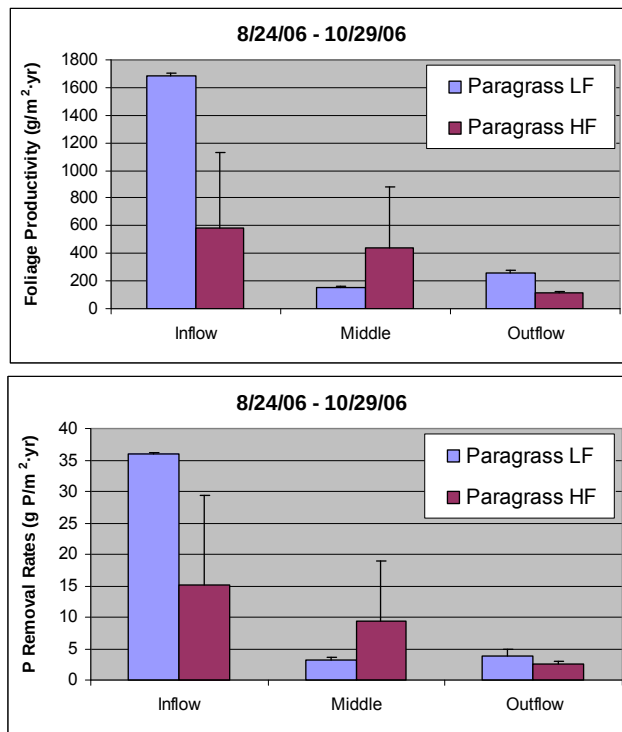
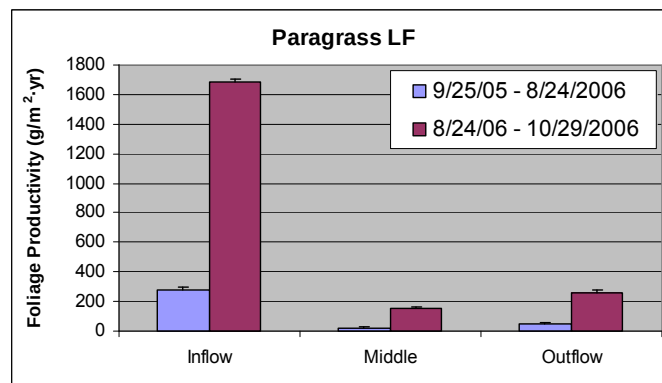


Figure 50. Foliage productivity (top) and P removal rates (bottom) of Paragrass LF and Paragrass HF treatments at three locations (inflow, mid and outflow regions) within the wetlands from August 24, 2006 to October 29, 2006.



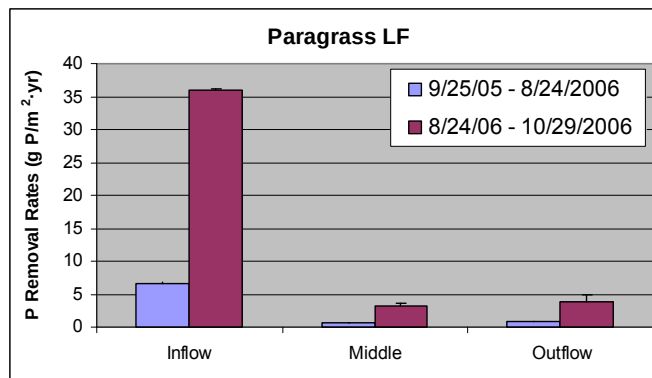


Figure 51. Foliage productivity (top) and P removal rates (bottom) of the Paragrass LF treatment at three locations (inflow, mid and outflow regions) within the wetlands during two periods in fall 2006.

Water Quality

Phosphorus

The water quality results presented herein represent data collected from January 2006 through December 2006. Drought conditions decreased the level of water in the Barn #5 tertiary lagoon that provided the inflow to the treatment wetlands. As a result, inflow TP concentrations nearly doubled from an average of 4.7 mg L^{-1} in 2005 (July – Dec) to 8.7 mg L^{-1} in 2006 (Jan – Dec).

In 2006, the cattail and cattail/SAV treatments were monitored for the entire year. During this time, inflow TP concentrations averaged $8700 \text{ } \mu\text{g L}^{-1}$ for both treatments and mean outflow TP concentrations were 7700 and $7600 \text{ } \mu\text{g L}^{-1}$, respectively (Figure). While both cattail and cattail/SAV treatments were effective at removing particulate phosphorus (PP), soluble reactive phosphorus (SRP) was often exported rather than removed (Figure and 53). As with SRP, dissolved organic phosphorus (DOP) was removed by wetlands until approximately April 2006, after which time it began to be exported, although to a lesser degree than SRP.

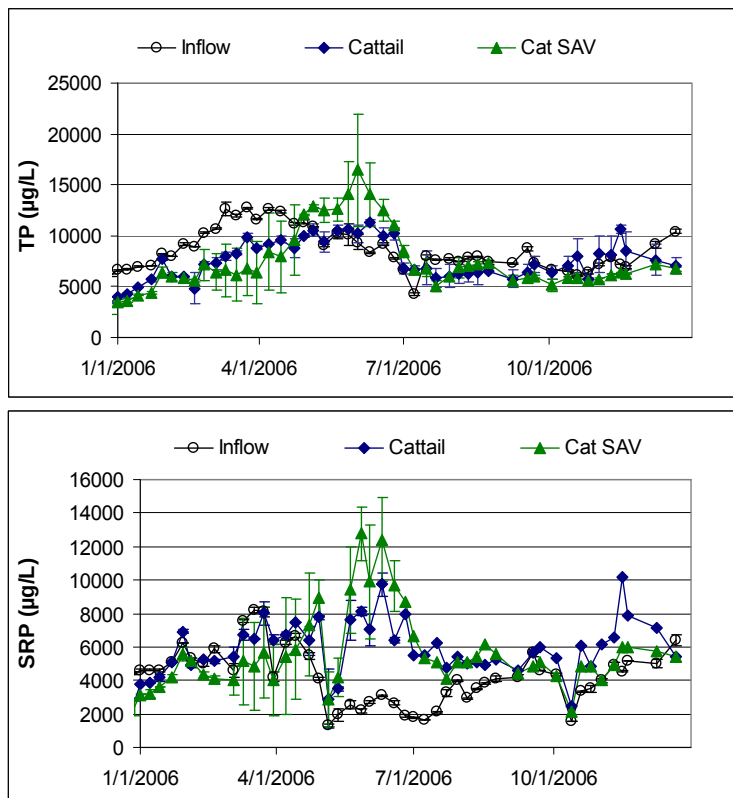
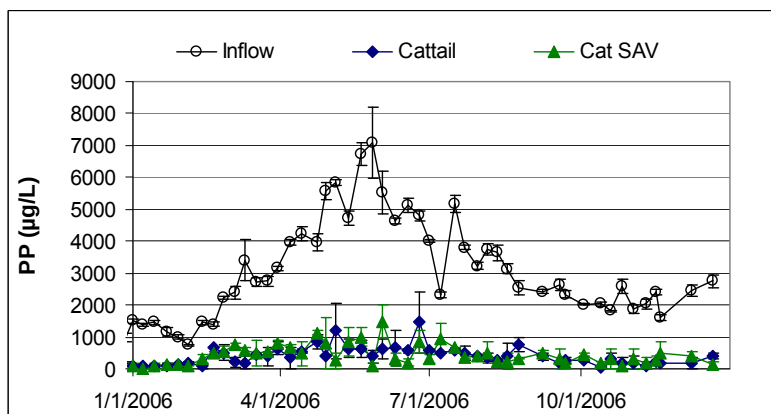


Figure 52. Mean inflow and outflow TP and SRP concentrations for cattail and cattail/SAV wetlands. Error bars represent $\pm$ one standard error (n = 2).



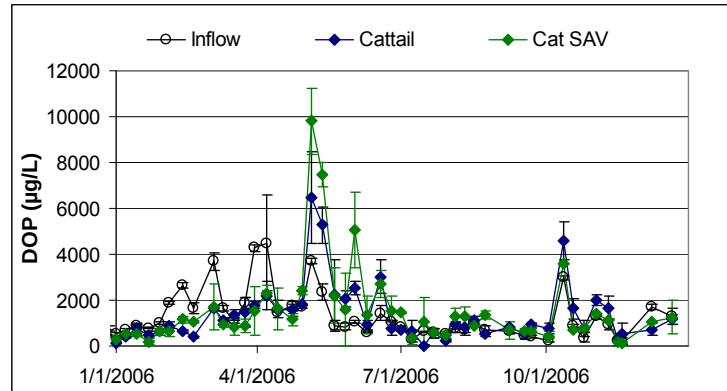
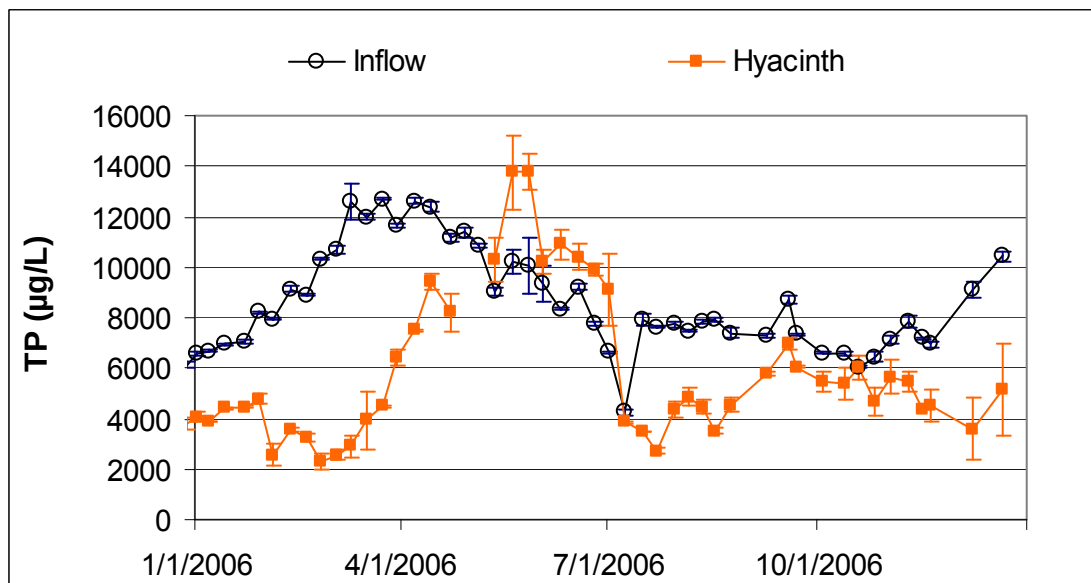


Figure 53. Mean inflow and outflow DOP and PP concentrations for cattail and cattail/SAV wetlands. Error bars represent +/- one standard error (n = 2).

The water hyacinth treatment was also monitored for the entire year (2006). This treatment provided the best overall TP removal, with average inflow and outflow TP concentrations of 8700 and 5800 $\mu\text{g L}^{-1}$, respectively. Except for the two-month period from May to July (immediately following shallow tilling of vegetation) the wetlands consistently removed TP. Coincidentally, during this time the inflow TP concentrations, which had peaked at 12,700 $\mu\text{g/L}$ on March 24th, began to decline. The export of TP following shallow tilling was primarily due to the release of SRP (Figure). A similar response to shallow tilling was observed in 2005. The water hyacinth wetlands were effective at removing PP even immediately after the tilling event (Figure). Dissolved organic P was removed up until May, at which time the wetland exported DOP until July. After July, the wetland continued to remove DOP, which was probably a result of reduced inflow DOP concentrations. Dissolved organic P removal during the 2nd half of 2006 was less than that observed at the beginning of the year (Figure).



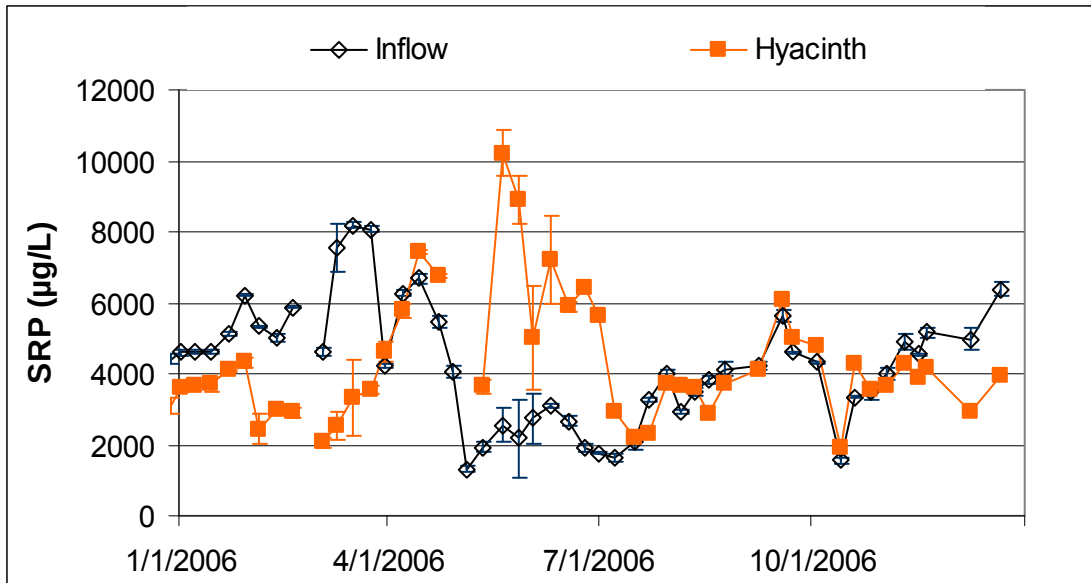
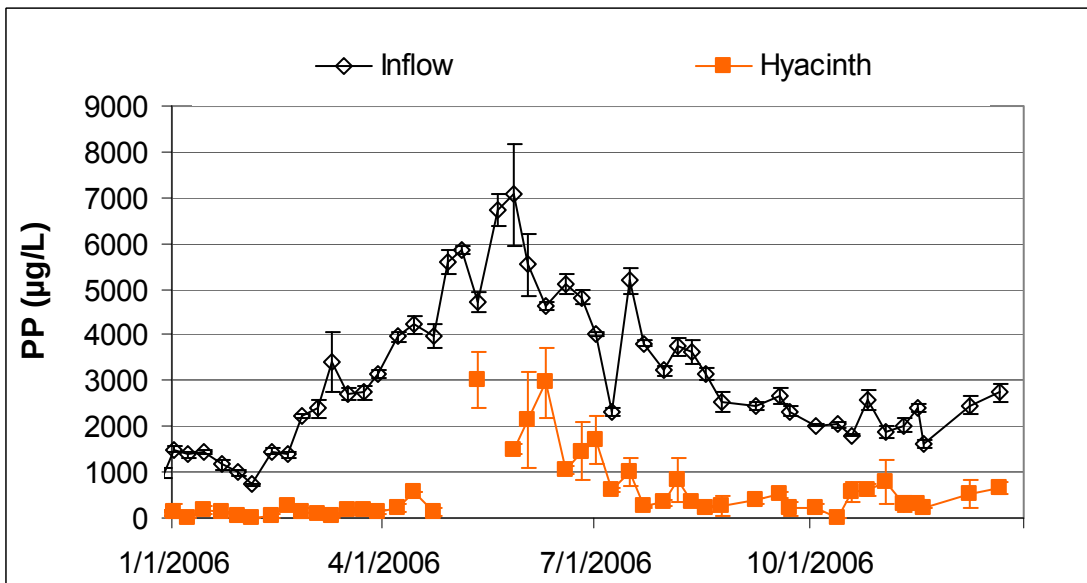


Figure 54. Mean inflow and outflow TP and SRP concentrations for the water hyacinth wetlands. The observed two-month export of TP and SRP was a result of rehydration after shallow tilling. Error bars represent +/- one standard error (n = 2).



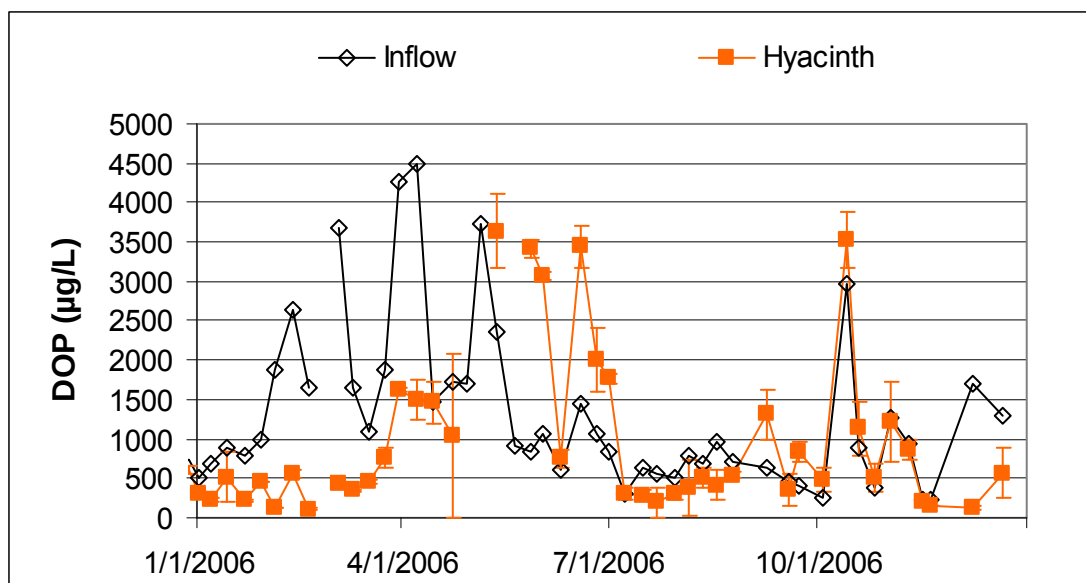
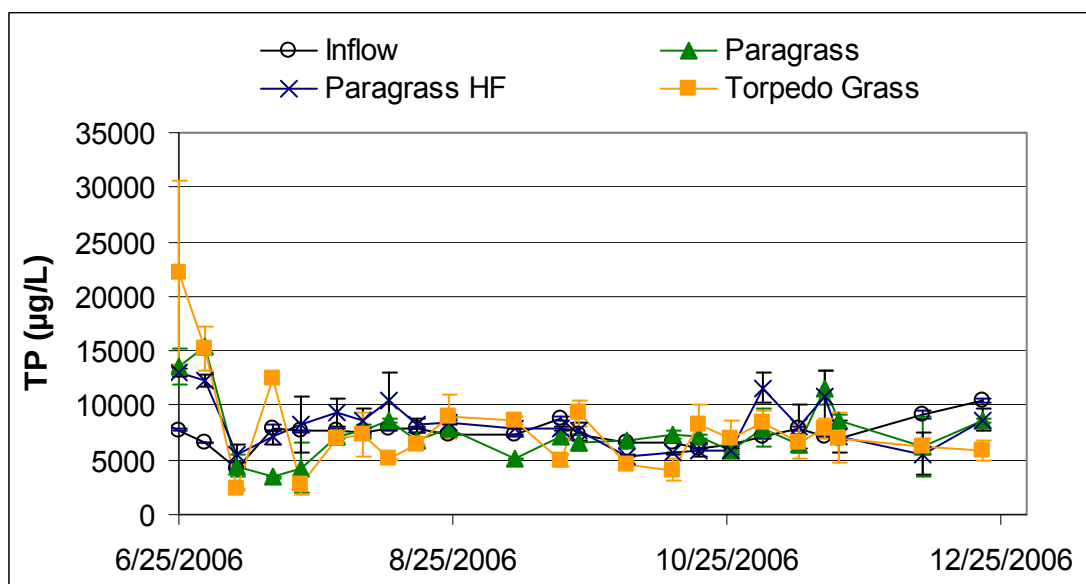


Figure 55. Mean inflow and outflow PP and DOP concentrations for the water hyacinth wetlands. The observed two-month export of DOP was a result of rehydration after shallow tilling. Error bars represent +/- one standard error (n = 2).

Water quality for the remaining treatments, including paragrass, paragrass HF and torpedo grass, was monitored from June – December 2006. During this period inflow TP concentrations averaged 7400 $\mu\text{g/L}$. Mean average outflow TP concentrations for the paragrass, paragrass HF and torpedo grass treatments were 7600, 8200 and 7800 $\mu\text{g L}^{-1}$, respectively, indicating a net export rather than removal of P (Figure). In general, these wetlands removed PP, but exported SRP and, to a lesser extent, DOP (Figure and 57).



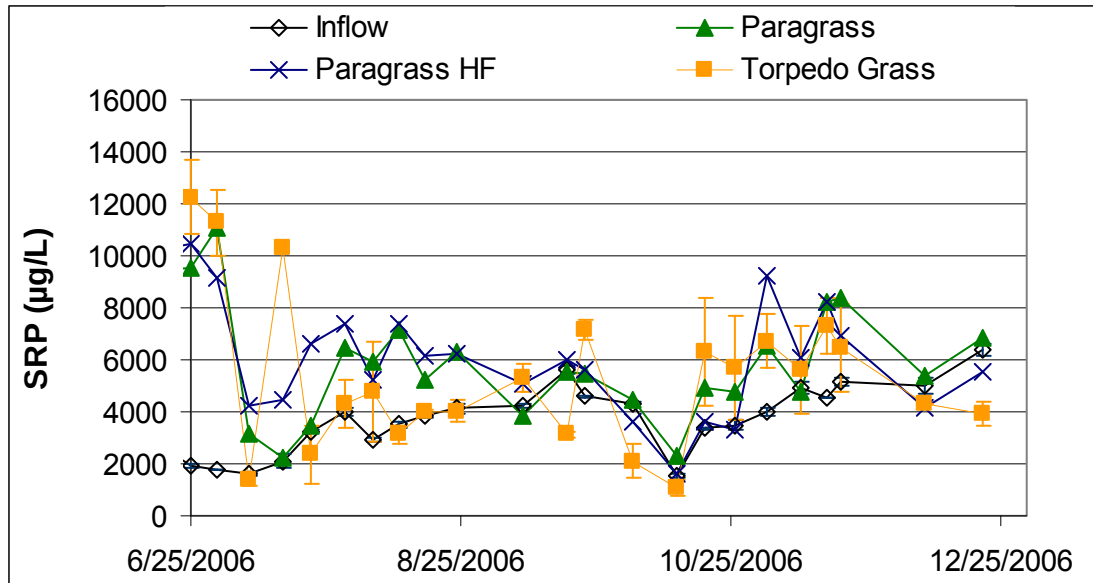
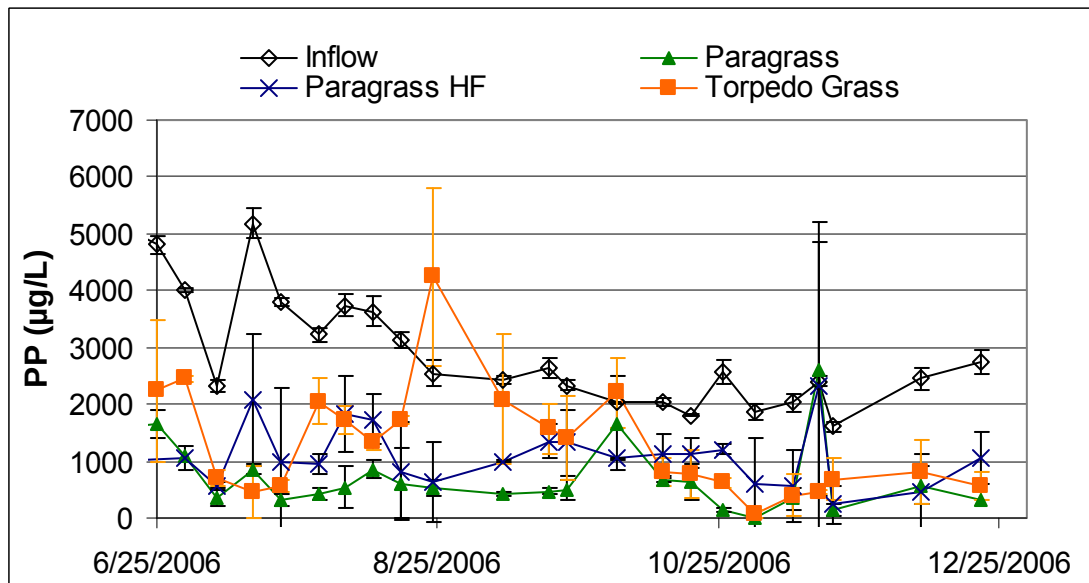


Figure 56. Mean inflow and outflow TP and SRP concentrations for paragrass, paragrass HF and torpedo grass wetlands. Error bars represent $\pm$ one standard error (n = 2).



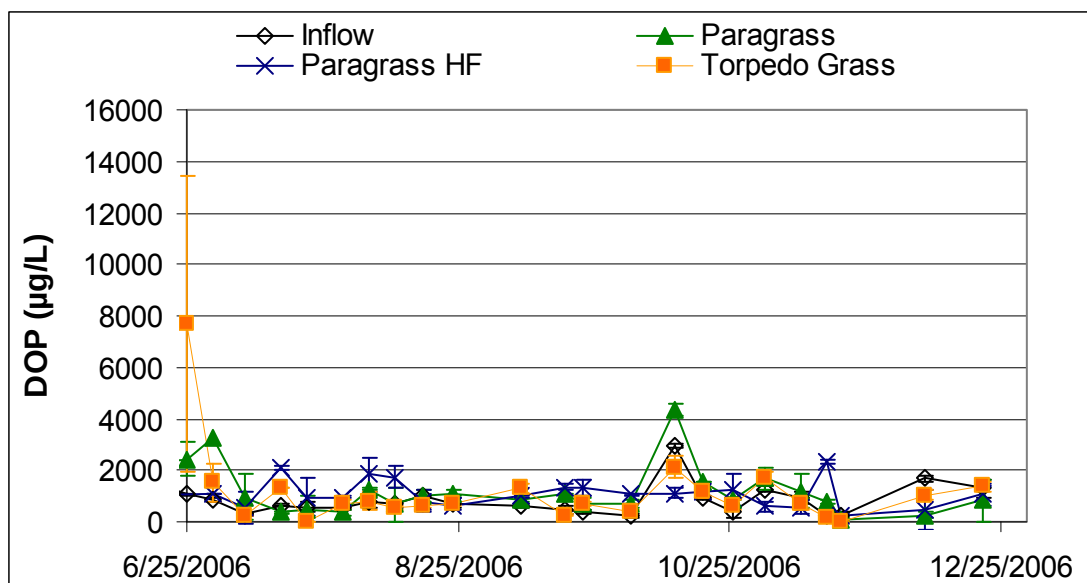
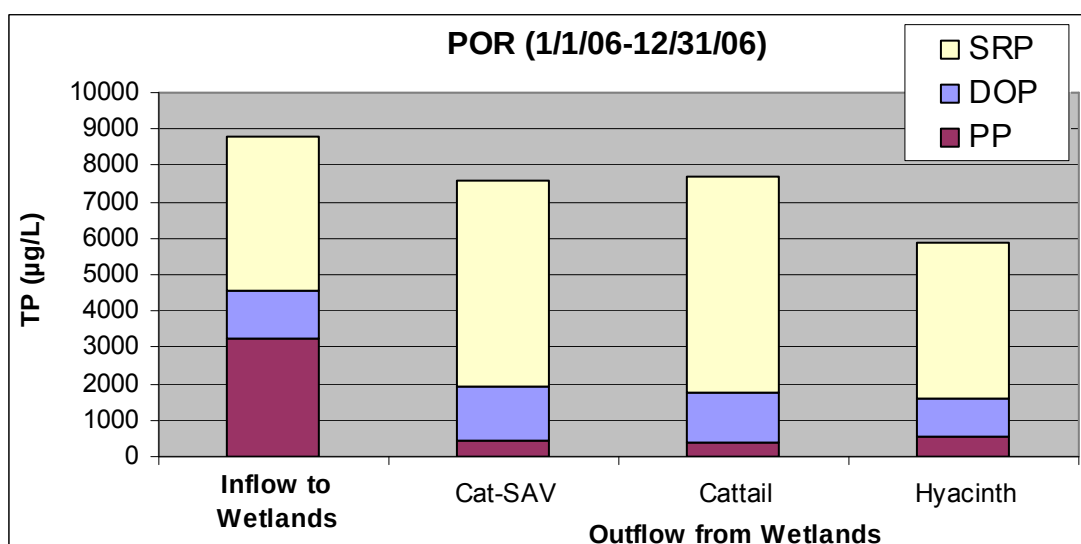


Figure 57. Mean inflow and outflow PP and DOP concentrations for paragrass, paragrass HF and torpedo grass wetlands. Error bars represent +/- one standard error (n = 2).

Figure 58 depicts the average P speciation for waters flowing into and out of the wetlands. All of the conventional wetland treatments were effective at removing particulate P. However, with the exception of the water hyacinth treatment, soluble P species (SRP and DOP) were exported from the wetlands. The pasture grass treatments, which were monitored from June – Dec 06, also removed PP and exported SRP and DOP.



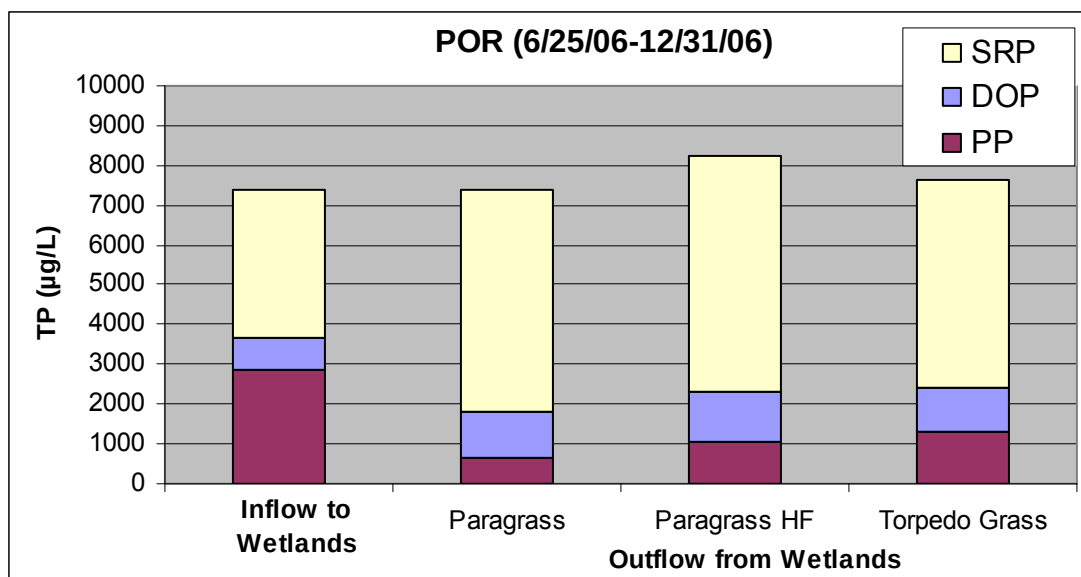
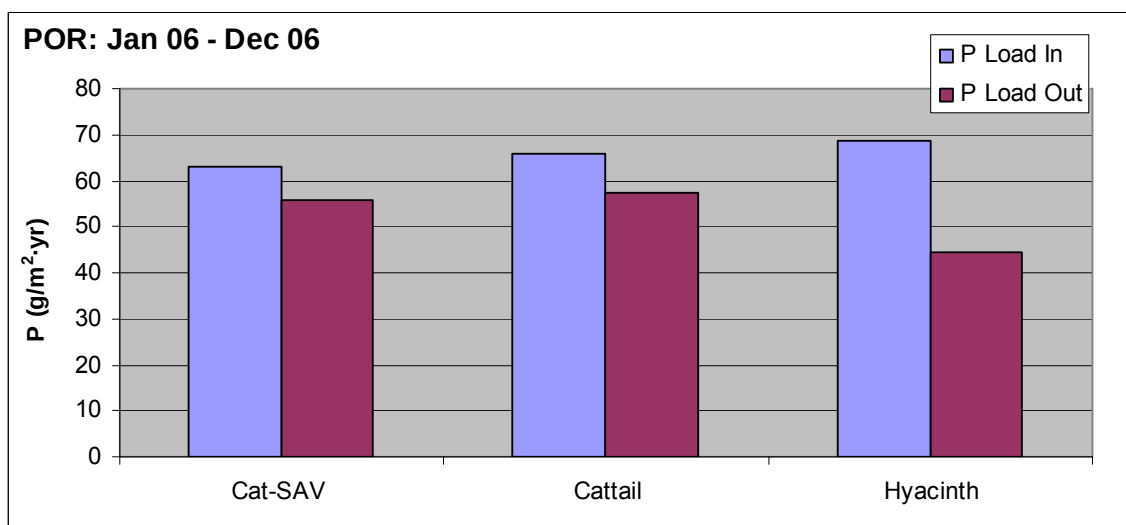


Figure 58. Mean inflow and outflow P concentrations for cattail, cattail/SAV and water hyacinth wetlands during Jan 2006 to Dec 2006 (top) and for paragrass, paragrass HF, and torpedo grass wetlands during Jun 2006 – Dec 2006.



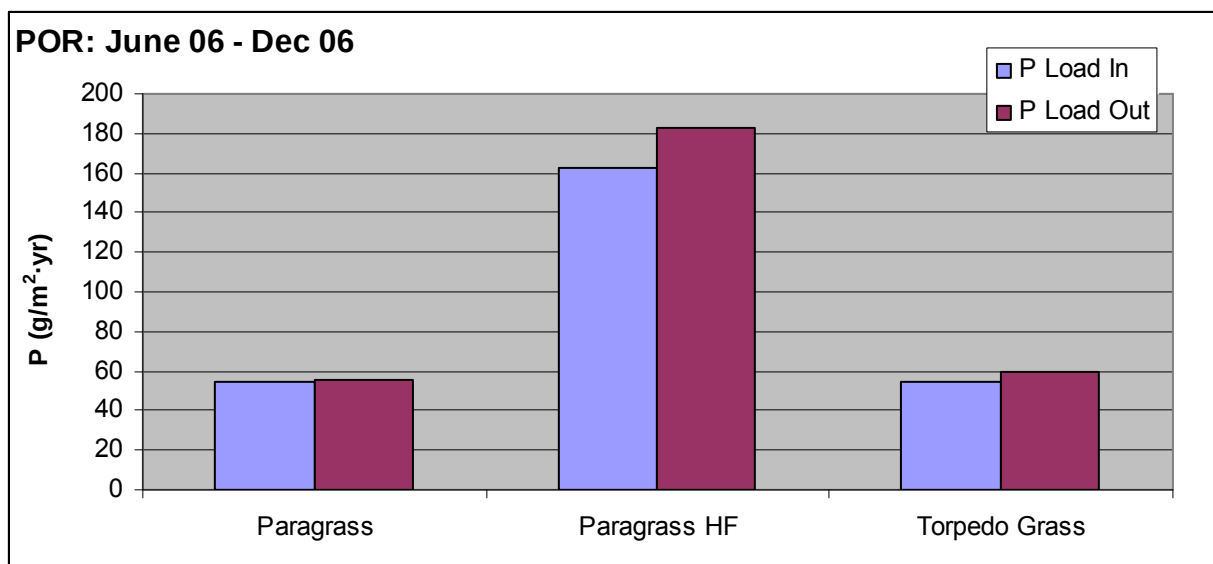


Figure 59. Inflow and outflow P loading rates for the cattail/SAV, cattail and water hyacinth wetlands during Jan 2006 – Dec 2006, and for paragrass, paragrass HF and torpedo grass wetlands from Jun 2006 – Dec 2006.

The net uptake of P exhibited by the harvested paragrass foliage (Figure 59) suggests that the Paragrass LF and Paragrass HF treatments should have exhibited a net removal of P based on inflow and outflow water quality measurements. However, the mid- and outflow regions of the wetlands, which were not covered by dense macrophyte vegetation, frequently exhibited a fluctuating standing crop of filamentous algae biomass. The senescence of this open water algae biomass probably accounted for the observed P export from the system during the second half of 2006.

Synopsis of Key Phosphorus Removal Performance Metrics

A performance summary for the six treatments is provided in Table . For these between-treatment comparisons, it is important to note the differing periods of record, and the fact that due to slightly different wetland surface areas, the average HLRs and mass P loads were not identical for all wetlands, particularly the paragrass HF wetlands. In general, however, the water hyacinth provided the most effective P removal, from both a mass removal ($24.1 \text{ g P m}^{-2} \text{ yr}^{-1}$) and mean outflow TP concentration ($5824 \text{ } \mu\text{g L}^{-1}$) basis. The requirement for a drydown period, and recovery period following tilling, clearly will slightly diminish the overall performance of the water hyacinth on an annual basis. However, periodically-tilled water hyacinth wetlands should still prove to be a highly effective P removal system, particularly if farm equipment is utilized to provide a deeper burial of the plant material than achieved in this study.

Cattail and cattail/SAV provided similar mass P removal rates of 8.8 and $7.6 \text{ g m}^{-2} \text{ yr}^{-1}$, respectively (Table 42). The remaining treatments (the pasture grasses) all exhibited a mass P export rather than removal.

Table 42. Mean operational characteristics and P removal performance of six treatment wetlands that received dairy lagoon wastewaters.

	POR (2006)	Average TP Inflow ($\mu\text{g L}^{-1}$)	Average TP Outflow ($\mu\text{g L}^{-1}$)	P Load ($\text{g P m}^{-2} \text{yr}^{-1}$)	Mass P Removal ($\text{g P m}^{-2} \text{yr}^{-1}$)*
Cattail/SAV	Jan 06 – Dec 06	8700	7600	63.2	7.6
Cattail	Jan 06 – Dec 06	8700	7700	66.0	8.77
Water Hyacinth	Jan 06 – Dec 06	8700	5800	68.8	24.1
Paragrass HF	June 06-Dec 06	7400	8200	162	-20.0
Paragrass	June 06-Dec 06	7400	7600	54.4	-1.63
Torpedo Grass	June 06-Dec 06	7400	7800	54.4	-5.61

* negative values denote P export

Other Parameters

Ammonia-N ($\text{NH}_4\text{-N}$), nitrate/nitrite-N (NO_x), total kjeldahl-N (TKN), dissolved calcium (Ca), total suspended solids (TSS), and chemical oxygen demand (COD) were monitored in the cattail, cattail/SAV and water hyacinth treatments from January – May 2006.

Total suspended solids concentrations were reduced from an average of 71 mg L^{-1} to 17, 16 and 20 mg L^{-1} for the cattail, cattail/SAV and water hyacinth treatments, respectively (Figure). Dissolved Ca levels decreased slightly through the cattail and water hyacinth wetlands (9 and 15% reduction, respectively). The cattail/SAV treatment provided the greatest dissolved Ca removal at 24%. Average inflow and outflow alkalinity concentrations were comparable for the cattail/SAV treatment (507 and $503 \text{ mg CaCO}_3 \text{ L}^{-1}$, respectively), whereas average alkalinity concentrations increased slightly in the cattail and water hyacinth treatments from $507 \text{ mg CaCO}_3 \text{ L}^{-1}$ (inflow) to 536 and $546 \text{ mg CaCO}_3 \text{ L}^{-1}$ (outflow).

After shallow tilling of the water hyacinths in May, there was no noticeable change in dissolved Ca levels. However, TSS levels increased from an average of 20 mg L^{-1} (Jan - April) before tilling to 117 mg L^{-1} for the two weeks immediately following tilling (Figure). Alkalinity levels also increased during this two-week period from $546 \text{ mg CaCO}_3 \text{ L}^{-1}$ to $981 \text{ mg CaCO}_3 \text{ L}^{-1}$ (Figure). A comparable short-term increase in TSS was observed following the 2005 shallow tilling event.

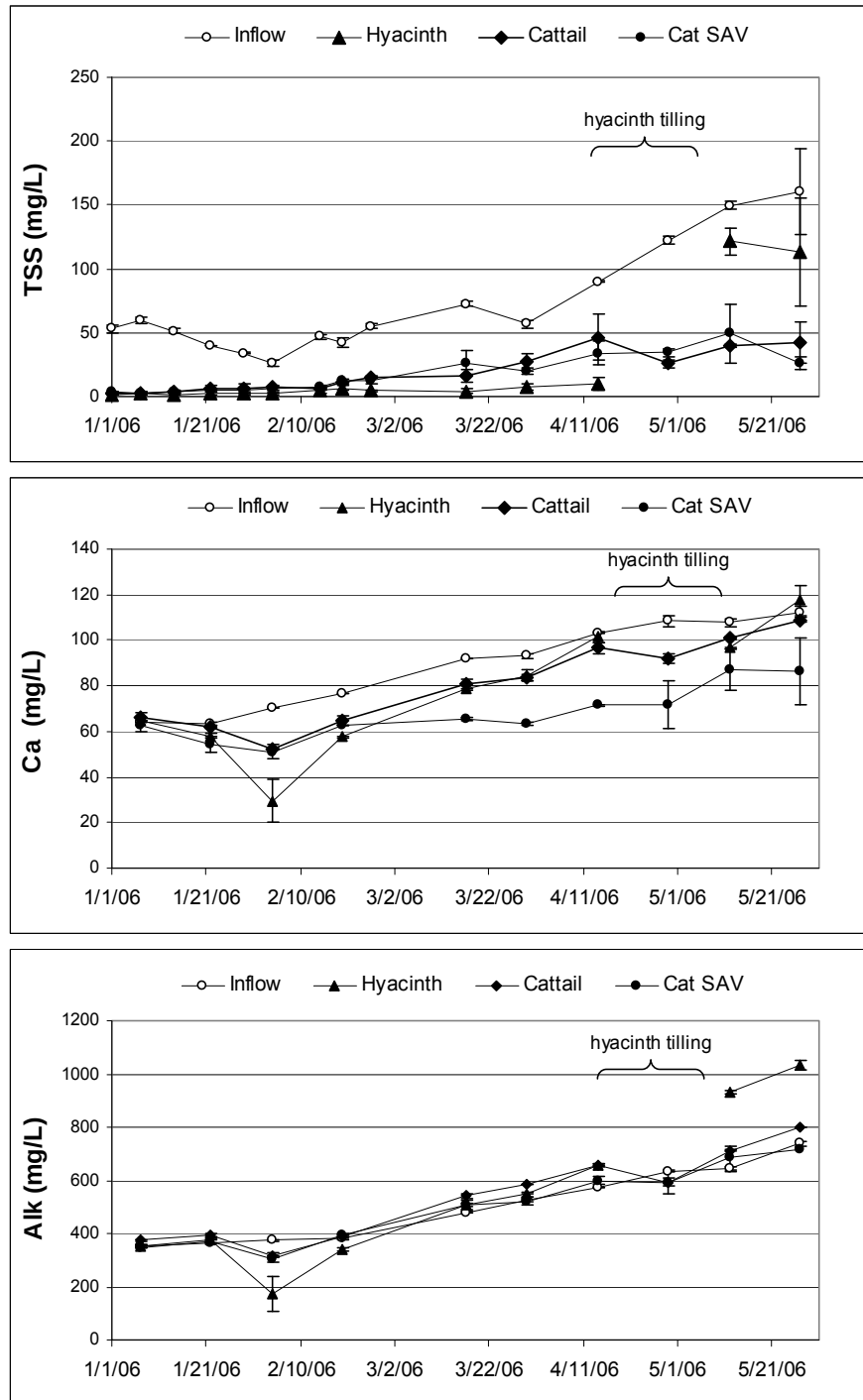
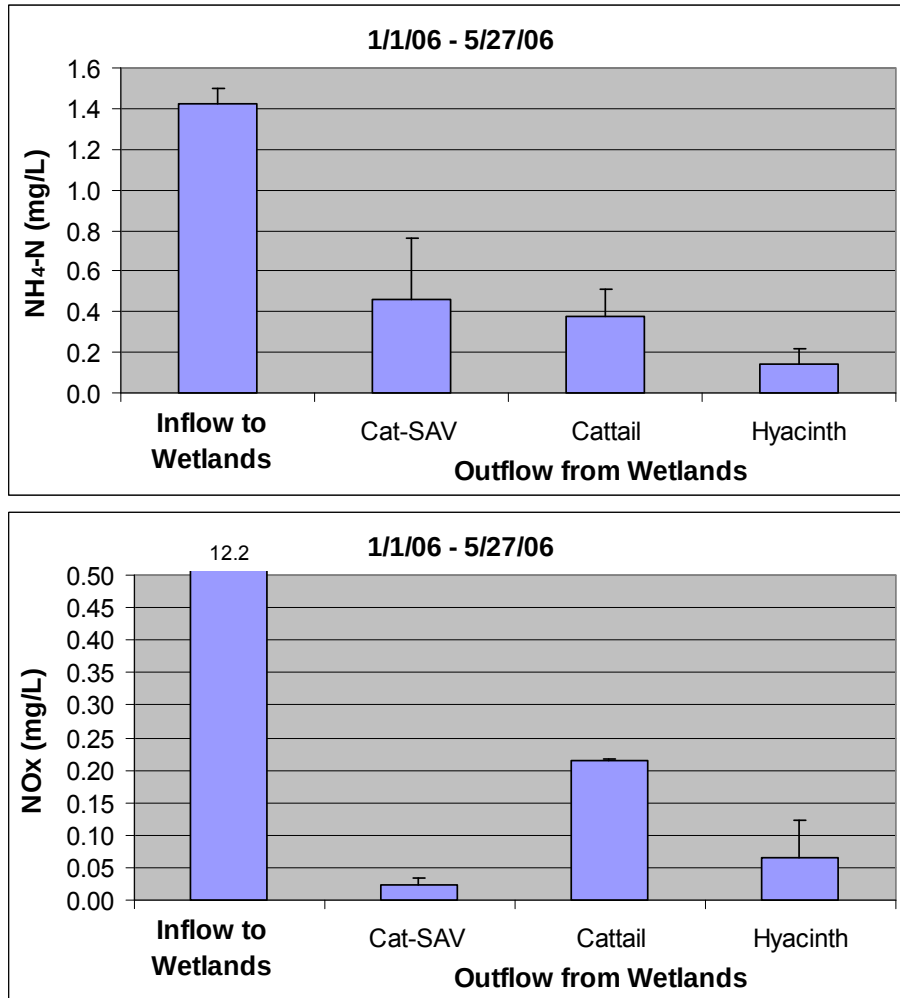


Figure 60. Mean inflow and outflow TSS, dissolved Ca, and alkalinity concentrations for cattail, cattail/SAV and water hyacinth wetlands from January – May 2006. Error bars represent ± 1 standard error (n=2).

All nitrogen species were reduced by the cattail, cattail/SAV and water hyacinth treatments (Figure 61). The inflow waters consisted of 58% organic-N, 39% NO_x, and 4% ammonia-N. The outflow of the cattail/SAV, cattail, and water hyacinth treatments

were 96, 94, and 98% organic-N, 0, 2 and 1% NO_x , and 4, 4, and 2% ammonia-N, respectively (Figure).

During this period (January – May 2006) average inflow COD concentrations were reduced from 380 mg L⁻¹ to 265, 272 and 277 mg L⁻¹ for the cattail/SAV, cattail and water hyacinth treatments, respectively (Figure).



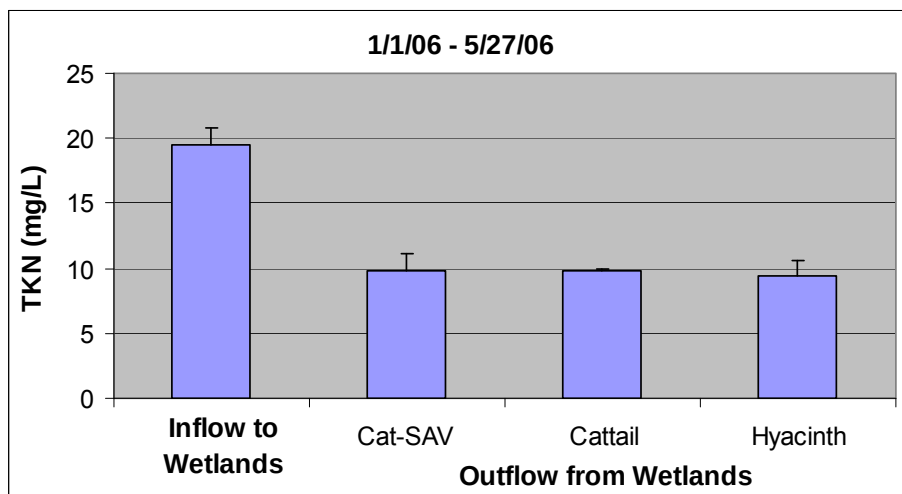


Figure 61. Mean inflow and outflow ammonia-N, nitrate+nitrite-N, and total Kjeldahl-N concentrations for cattail, cattail/SAV, and water hyacinth wetlands. Error bars represent ± 1 standard error (n=2).

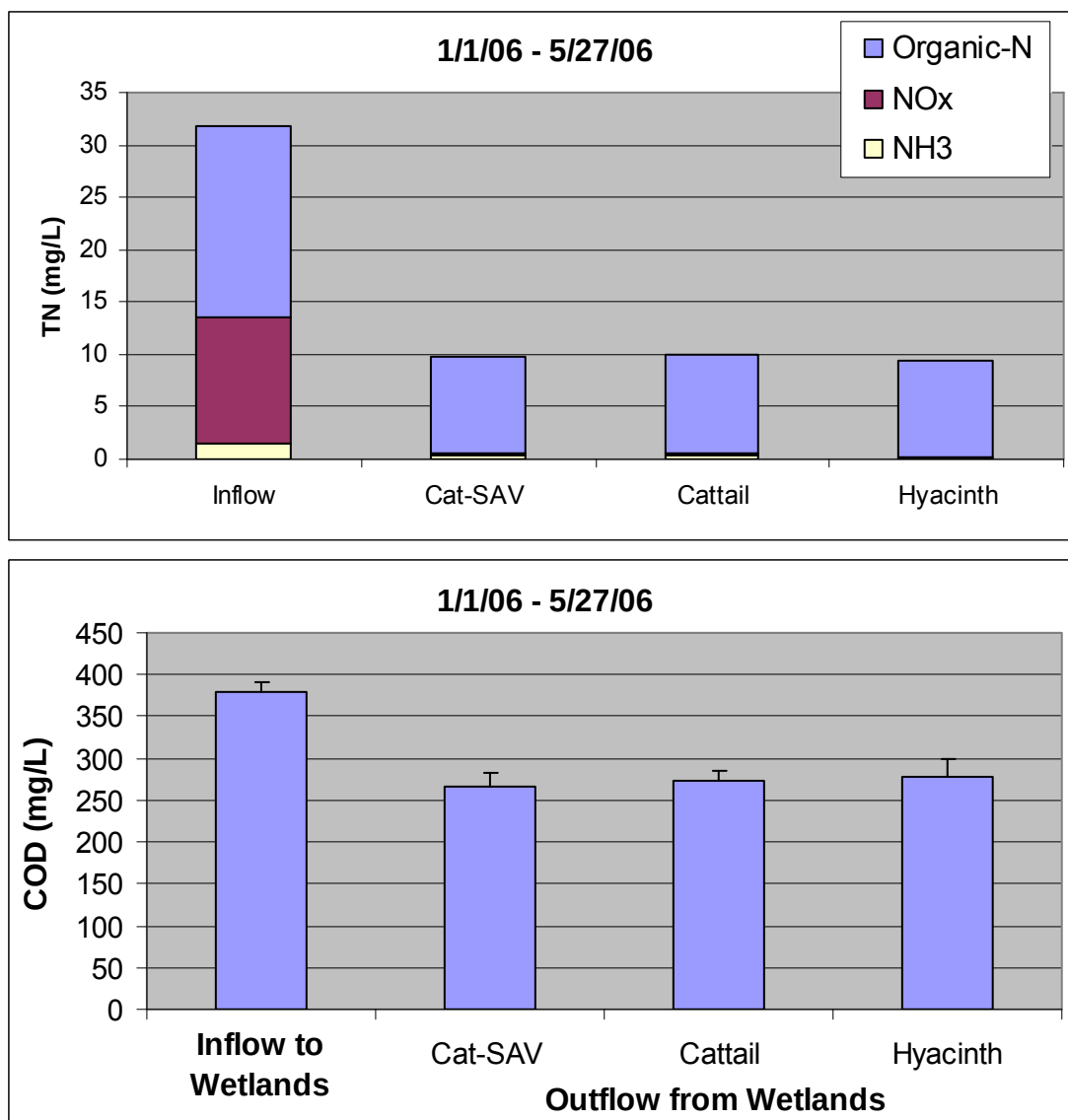


Figure 62. Mean inflow and outflow nitrogen species (top) and COD (bottom) concentrations for cattail, cattail/SAV, and water hyacinth wetlands. Error bars represent ± 1 standard error ($n = 2$).

6.0 Extension education activities

The objective of this extension plan will be to facilitate transfer of technology to a wide range of stakeholders and other end-users, including agricultural producers, public (agency) and private natural resources management and environmental monitoring organizations.

Provide expertise and information transfer between this project and the Environmental Services Project headed by World Wildlife Fund in collaboration with multiple landowners and governmental agencies.

Provide technical support and on camera expertise for an upcoming film titled Everglades: Currents of Change. This documentary will be aired in state and nationally on PBS and will highlight many aspects associated with water quality and restoration of the Everglades. Specifically this film investigates solutions to phosphorus loading and highlights many BMPs being implemented to reduce loads. General findings from this research are expected to be included in the final cut.

A factsheet based on findings of the spatial survey is being finalized and will be distributed via EDIS shortly to make all landowners, not just those participating in the study, aware of this phase of the projects findings.

Several field days are also being planned. Once sites are hydrologically restored (Task 3.3) we will begin the organization of field days to discuss implications (both positive and potentially undesirable in the form of reduced extent of upland pasture) in response to flooding, and to highlight results of the study. These workshop/field days will be coordinated with other agency staff and local commodity associations to increase awareness and hopefully increase participation in use of isolated wetlands as part of water quality related BMPs.

There has also been preliminary development of an extension focused In-Service Training event in Okeechobee that will focus on use of isolated wetlands in agricultural landscapes to improve water quality. This workshop is being planned for spring 2007.

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