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ORIGINAL PAPER

Phosphorus retention of forested and emergent marsh depressional wetlands in differing land uses in Florida, USA

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Abstract The translocation of phosphorus (P) from terrestrial landscapes to aquatic bodies is of concern due to the impact of elevated P on aquatic system functioning and integrity. Due to their common location in depressions within landscapes, wetlands, including so-called geographically isolated wetlands (GIWs), receive and process entrained P. The ability of depressional wetlands, or GIWs, to sequester P may vary by wetland type or by land use modality. In this study we quantified three measures of P sorption capacities for two common GIW types (i.e., emergent marsh and forested wetlands) in two different land use modalities (i.e., agricultural and least impacted land uses) across 55 sites in Florida, USA. The equilibrium P concentration (EPC_0) averaged 6.42 ± 5.18 mg P L^{-1} (standard deviation reported throughout); and ranged from 0.01–27.18 mg P L^{-1}; there were no differences between GIW type or land use modality, nor interaction effects. Significant differences in phosphorus buffering capacity (PBC) were found between GIW types and land use, but no interaction effects. Forested GIWs [average 306.64 ± 229.63 (mg P kg^{-1}) (µg P $L^{-1})^{-1}$], and GIWs in agricultural settings [average 269.95 ± 236.87 (mg P kg^{-1}) (µg P $L^{-1})^{-1}$] had the highest PBC values. The maximum sorption capacity (S_{max}) was found to only differ by type, with forested wetlands (1274.5 ± 1315.7 mg P kg^{-1}) having over three times the capacity of emergent GIWs (417.5 ± 534.6 mg P kg^{-1}). Classification trees suggested GIW soil parameters of bulk density, organic content, and concentrations of total P, H_2O-extractable P, and HCl-extractable P were important to classifying GIW P-sorption metrics. We conclude that GIWs have high potential to retain P, but that the entrained P may be remobilized to the wetland water column depending on storm and groundwater input P concentrations. The relative hydrologic dis-connectivity of GIWs from other aquatic systems may provide sufficient retention time to retain elevated P within these systems, thereby providing an ecosystem service to downstream waters.

Keywords Equilibrium phosphorus concentration · Phosphorus buffering capacity · Phosphorus sorption maximum · S_{max} · Connectivity · Agriculture · Palustrine wetland

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Introduction

Phosphorus (P) is a limiting nutrient important to primary productivity in both terrestrial and aquatic systems, yet this nutrient also has deleterious impacts on aquatic ecosystem eutrophication (Correll 1998; Reddy et al. 1999; Smith et al. 1999). Efforts to decrease the effects of excessive P loading on aquatic systems have continued through the development of best management practices for application (e.g., Sharpley et al. 2000; Kleinman et al. 2011), decreasing point-source pollution (Carey and Migliaccio 2009), and through the use of existing natural and built "green infrastructure", such as wetlands (e.g., Ewel and Odum 1984; Kadlec and Wallace 2008; Piechnik et al. 2012). Wetlands retain phosphorus through both short-term and long-term processes (Reddy and DeLaune 2008). Short-term processes include incorporation into periphyton and vegetative material and slow mineralization and cycling in detrital matter (Cheesman et al. 2010). Long-term processes include soil accretion and both chemical and physical sorption kinetics (Craft and Casey 2000; Dunne et al. 2006).

Because of their widespread spatial distribution (e.g., Tiner 2003; Bowen et al. 2010; Lane et al. 2012) and frequent co-location with phosphorus-intensive activities (i.e., agricultural activities; Johnston 2013; Wright and Wimberly 2013), extant geospatially or so-called "geographically isolated wetlands" (GIWs) may serve as local or watershed-scale sinks for phosphorus-entrained runoff (e.g., Zhang et al. 2009). A GIW can be defined as a wetland system, typically depressional, with no obvious hydrologic surface connection to downstream systems. While the term GIW has served to describe these systems for over a decade (e.g., Tiner 2003), it is worth noting that these wetlands exist along a continuum of connectivity rather than a binary condition, and the use of the term GIW is retained as a convention while acknowledging that "…most, if not all, wetlands scientists would agree that there is no such thing as an isolated wetland from an ecological standpoint" (Tiner 2003, p. 494). Iconic GIW systems include cypress domes, woodland seasonal and California vernal pools, Carolina bays, and prairie potholes (Tiner 2003). They are frequently unaffected by seasonal or annual flooding from streams and rivers, and may also be defined as "non-adjacent" (i.e., to downstream waters) wetland systems external to floodplains (Mushet et al. 2015).

Phosphorus retention in GIWs has been examined in parts of the Southeastern U.S., especially in the Lake Okeechobee, Florida Watershed (e.g., Dunne et al. 2006, 2010; Bhadha and Jawitz 2010; Min et al. 2010; see also Cohen et al. 2007). Dunne et al. (2007) reported that restoring 5–20 % of the GIW area in priority basins draining to Lake Okeechobee could increase P storage in GIWs by up to 13 kg P ha^{-1}, mostly through increased soil organic matter (OM) with its concomitant P in wetland soils. Cheesman et al. (2010) found that GIWs stored greater amounts of total phosphorus (TP) than the uplands in which they were bedded (236 vs. 114 kg ha^{-1}), mostly in the OM, which is found in greater quantities in GIWs than in upland soils. However, because of P concentration gradients internal to GIWs, they could serve as P sources, rather than sinks, depending on soil characteristics, hydroperiod and P concentration in inflowing waters, amongst other variables (Hoffmann et al. 2009). For instance, Bhadha et al. (2011) reported that internal loading (i.e., P in the soils within the GIW) accounted for 18 % of the P "entering" two studied historically isolated wetlands in southern Florida; a shallow ditch draining these wetlands, which eventually connected to Lake Okeechobee, accounted for 49 % of the P outflow, thus creating a potential P source. Bhadha et al. (2011) also found infiltration to the ground accounted for 14 % of the P loss from the GIW, suggesting that near-surface flow gradients are important to landscape-level P dynamics.

P retention is a dynamic process, and these processes can be characterized through the calculation of multiple metrics, including the equilibrium P concentration (EPC_0), P buffering capacity (PBC), and the maximum P sorption capacity (S_{max}). The EPC_0 is a measure of the "P in solution that is in equilibrium with P in the solid phase" (Belmont et al. 2009, p. 988). Water entering a GIW with P in solution that is below the EPC_0 will cause the release of P bound to wetland soils (see Bhadha et al. 2011), whereas water with P in solution $>EPC_0$ will be sorbed onto available sorption sites until a capacity is reached (Reddy and DeLaune 2008). The slope of the EPC_0 line is described as the P buffering capacity (Rayment and Lyons 2011), a measure of the ability of a soil to effectively sorb additional P. Loading of P into systems may decrease the ability of the wetland to sorb P, resulting in an increase in the EPC_0. However, the capacity of soils to sorb P is finite and may be measured by fitting an adsorption model using the Langmuir equation to calculate S_{max}, the maximum sorption capacity of the system (Reddy and DeLaune 2008). All three of these parameters would be expected to vary based on land use surrounding GIWs and the vegetative structure of the systems. In this study land use was a proxy for expected anthropogenic inputs affecting GIWs. The GIWs located within agricultural settings would be expected to have greater inputs of sediments (Wardrop and Brooks 1998; Skagen et al. 2008), higher nutrient concentrations (Cheesman et al. 2010), and potentially altered hydrology (Kleinman et al. 2011; Babbar-Sebens et al. 2013), all of which may affect P retention processes (Hoffmann et al. 2009). Wetland structure, or dominant vegetation, would be expected to differ based on local-scale hydrodynamics and soil characteristics. For instance, forested wetlands would be expected to be more prevalent in relatively drier GIWs than herbaceous systems (Hofstetter and Sonenshein 1990; Sharitz and Gresham 1998), and with increased hydroperiod comes biogeochemical processes affecting P sorption, as well as other characteristics (e.g., increased OM) that also affect P retention processes (Johnston 1991; Hoffmann et al. 2009).

To better understand the contribution and controls of GIWs to P dynamics throughout Florida, we quantified EPC_0, PBC, and S_{max} from GIW soils across two different land-use modalities and wetland vegetation types, and analyzed soil parameters to ascertain controls on P metrics and to classify GIWs. Understanding the potential influence of GIWs on nutrient dynamics can help to determine the impact or effect of GIWs on downstream waters. While no direct linkages were made in this study, the outputs from this study, and similar analyses, may be useful to model the movement of water and nutrients from landscapes through GIW elements to downstream systems.

Methods

Sampling sites and land use modalities

To quantify P sorption capacities, 55 palustrine forested (PFO, $n = 27$) and palustrine emergent marsh (PEM, $n = 28$) depressional GIWs with no surface water inputs or outputs (such as flowing water systems with bed and bank features connecting them to other systems) other than surface runoff from across Florida were sampled in agricultural (AG, $n = 32$) and "reference" (REF, $n = 23$) land-use modalities in the summer of 2005 (Fig. 1). In some cases, agricultural sites (in particular) may have ditches leading into them from agricultural fields to increase the movement of water off the landscape. The average size was 0.78 ± 0.53 ha (standard deviation reported throughout). The AG sites were identified based on the prevalence of cattle grazing and row crops within approximately 300 m of the site, with sites typically embedded within one or both of those land-use modalities. The REF sites were located within local, state, and national parks and forests, with no obvious signs of human alterations or disturbances. However, it is recognized that these binary classifications do not fully capture the range of land use intensity, resilience or recovery. They are meant to grossly categorize the landscape to allow comparisons across the natural variability to be found in the study area.

Sampling and chemical analyses

Sites were sampled a single time. A single soil core was collected from the surface at the approximate center of each wetland using a 7.5 by 10-cm circular stainless steel hand-powered coring device. The collected soil cores were homogenized and frozen at −20 °C until analyzed within 24 months of collection. For P retention analyses, a subsample was extracted from the preserved and homogenized material and air-dried. Following Rayment and Lyons (2011), P retention metrics were determined by diluting laboratory-grade PO_4–P stock solution to create solutions with initial concentrations of 5, 10, 50, 100, and 150 mg PO_4–P L^{-1}. A 5-g dry soil sample was added to 50 mL solution along with approximately 0.10 mL chloroform ($CHCl_3$) to suppress any biological activity. The solution was then shaken for 17 h at 25 °C. The concentration of PO_4–P remaining in solution after 17 h was analyzed following U.S. EPA Method 365.3 (U.S Environmental Protection Agency 1979). The amount sorbed versus the amount remaining in solution was used to calculate the descriptive P measures of EPC_0 and PBC following Rayment and Lyons (2011). The S_{max} (P sorption maxima, mg P kg^{-1}) was calculated following the nonlinear, least-square fitting spreadsheet Langmuir equation algorithms provided by Bolster and Hornberger (2007). The Langmuir equation describes adsorption of P onto soil surfaces, with assumption that soil particles have a finite capacity to absorb available P (Reddy and DeLaune 2008). The maximum value of the P-sorption capacity is considered the value along the vertical axis of the asymptote of the calculated sorption capacity. The Langmuir adsorption model parameter S_{max} was calculated by finding the least-squares fit of C/S plotted against C:

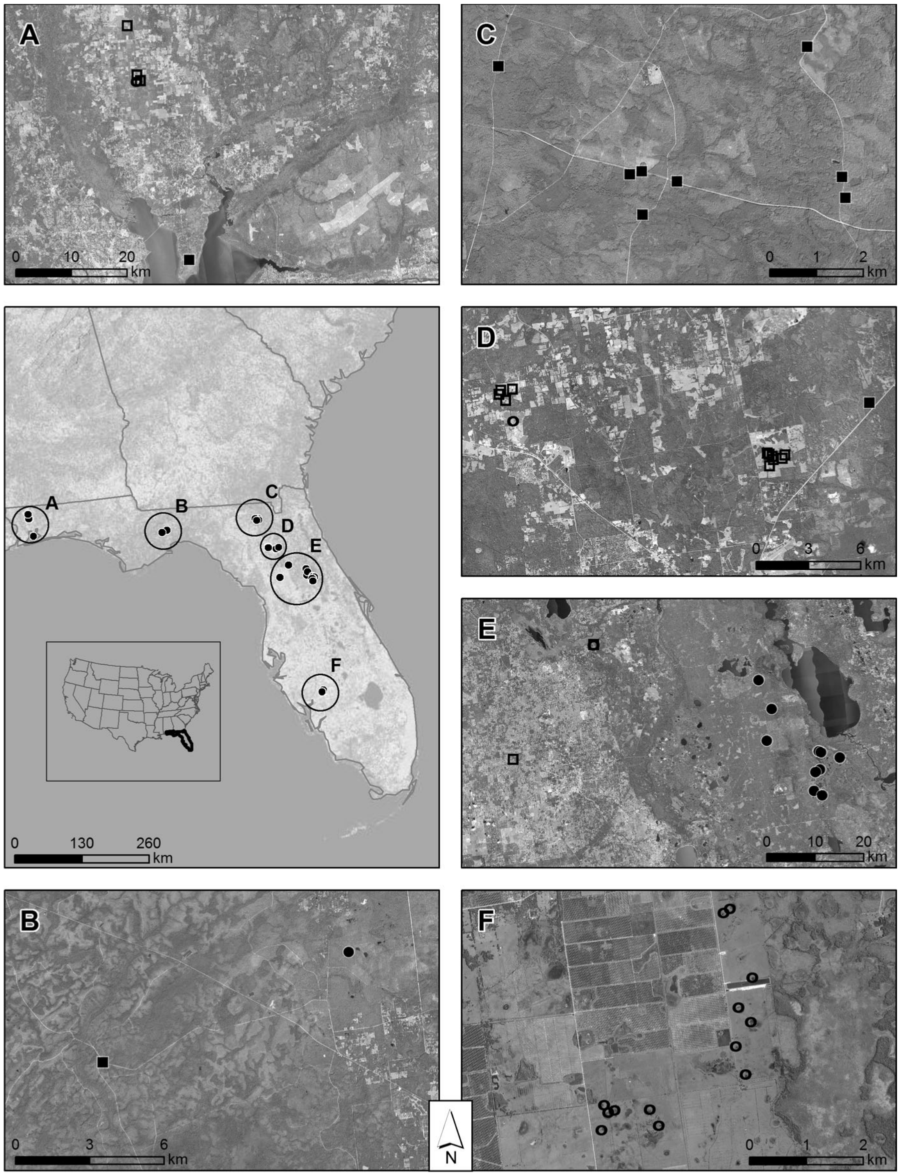

Fig. 1 Site locations sampled across Florida in 2005 ($n = 55$). *Inset* figures (**a**–**f**) identify the wetland types and landscape setting of the sites. In the *inset* figures, emergent marsh wetlands ($n = 28$) are denoted by circles, with *hollow circles* representing sites in agricultural landscapes ($n = 16$), with the balance in reference/least impacted settings ($n = 12$). Palustrine forested wetlands ($n = 27$) are denoted by *squares*, again with hollow squares representing agricultural sites ($n = 16$) and *solid squares* reference/least impacted sites ($n = 11$)

$$C/S = 1/(kS_{max}) + C/S_{max} \quad (1)$$

where C is soil P concentration (mg L^{-1}), S is the P in absorbed phase (mg kg^{-1}), and k is the Langmuir sorption coefficient (Reddy and DeLaune 2008).

Soil parameters were analyzed following standard methods from a separate portion of the homogenized subsample, including bulk density (g mL^{-1}; ASTM-C 29/C 29 M-07, ASTM 2009), H_2O-extractable P (mg kg^{-1}) and HCl-extractable P (mg kg^{-1}; U.S. EPA method 365.3—U.S. EPA 1979), nitrite-nitrate as N (mg kg^{-1}; U.S. EPA method 353.2—U.S. EPA 1979), organic content (i.e., percent organic matter, %OM; Nelson and Sommers 1982), pH (American

Public Health Association (APHA) 1995), TP (mg kg^{-1}; U.S. EPA method 365.3—U.S. EPA 1979), specific conductance (μs cm^{-1}; American Public Health Association (APHA) 1995), total carbon (TC, mg kg^{-1}; U.S. EPA method 9060, U.S. EPA 2004), and total nitrogen (TN, mg kg^{-1}; Plumb 1981).

Statistical analyses

We tested for main and interaction effects of wetland type (i.e., PEM vs. PFO) and land use (i.e., AG vs. REF) on our characterizations of interest with a fixed-effects ANOVA in SAS (SAS Institute, Cary, NC, version 9.2). Values were log, arcsine square root, or square root transformed, which improved their approximation of a normal distribution for the parametric ANOVA analyses. Linear correlations were calculated between square root-transformed EPC_0 and PBC, and log-transformed S_{max} and the transformed, if necessary, soil characteristics. We also grew conditional inference (CI) trees to explore controls on P retention dynamics and to classify sites based on P metrics using the package "Party" in R (Hothorn et al. 2006). Much like classification and regression trees [e.g., "CART" (Breiman et al. 1984)], CI trees are tree-based non-parametric regression algorithms that recursively partition the dataset exploring all bifurcation permutations to identify covariates that optimize the best split between nodes. Splits occur when conditions are met [i.e., the minimum criterion for the hypothesis testing independence between input variables and the response is exceeded; see Hothorn et al. (2006) for additional information]. We set a minimum split criterion of 0.05 and an initial minimum group membership of seven, (decreased to four where necessary based on preliminary results), used 9999 Monte Carlo runs to ascertain the significance of the splits, and used non-transformed data in our CI tree development.

Results

The EPC_0 averaged 6.42 mg P L^{-1} ± 5.18; range 0.01–27.18 mg P L^{-1} (see Table 1), and there were no differences between the main effects of type ($F = 3.74$, $p = 0.0586$) or land use ($F = 0.26$, $p = 0.6146$), or the interaction effects of type and land use ($F = 2.28$, $p = 0.1375$). For PBC, there were significant results from the main effects of type ($F = 15.91$, $p = 0.0002$) and land use ($F = 8.67$, $p = 0.0049$), but no interaction effects ($F = 0.83$, $p = 0.3657$). The PFO wetlands had a significantly ($p = 0.0002$) higher PBC than PEM wetlands, with PFO wetlands averaging (306.64 ± 229.63 (mg P kg^{-1}) (μg P L^{-1})$^{-1}$), range (9.52–927.76 (mg P kg^{-1}) (μg P L^{-1})$^{-1}$). The PBC values in PEM wetlands averaged (117.71 ± 128.23 (mg P kg^{-1}) (μg P L^{-1})$^{-1}$), range (4.99–555.78 (mg P kg^{-1}) (μg P L^{-1})$^{-1}$). In addition, AG wetlands (269.95 ± 236.87 (mg P kg^{-1}) (μg P L^{-1})$^{-1}$) had a significantly greater PBC than REF wetlands (127.68 ± 115.88 (mg P kg^{-1}) (μg P L^{-1})$^{-1}$). A Langmuir equation fit was only found for 39 sites (Bolster and Hornberger 2007). The main effect of type on S_{max} was significant ($F = 12.09$, $p < 0.0014$), and PFO GIWs averaged 1274.5 ± 1315.7 mg P kg^{-1}, which was significantly greater than PEM GIWs (417.5 ± 534.6 mg P kg^{-1}). No significant effects of land use ($F = 2.74$, $p = 0.1066$) or interactions ($F = 0.47$, $p = 0.4970$) were found for S_{max}.

There were significant main effects (Table 2) of type and land use for bulk density (type), HCl-extractable P (type and land use), percent OM (type), TP (type and land use), specific conductance (type and land use), TC (land use), and TN (type). Interaction effects were found for measured soil values of H_2O-extractable P, nitrite–nitrate as N, and pH. PEMs in AG settings had significantly greater H_2O-extractable P than PEMs in REF settings ($p = 0.0003$). The AG PFO was significantly higher in H_2O-extractable P than REF PEM, and there was also a significant difference between REF PEM and REF PFO, with REF PFO having greater H_2O-extractable P than REF PEM. Significant differences were found in nitrite-nitrate as N between AG PEM and AG PFO, with AG PEM being significantly greater, and AG PFO was also significantly greater than both REF PFO and REF PEM. Lastly, significant interactions were found for all measures of pH, excepting that AG PEM and AG PFO did not differ.

Sample bulk density and longitude were significantly correlated with all three P retention metrics (Table 3), though the direction of the relationships differed between EPC_0 (positive) and S_{max} and PBC (negative). The P measures (i.e., H_2O-extractble P, HCl-extractable P, and TP) were significantly correlated with the P-retention metrics (i.e., H_2O-extractble P with EPC_0; HCl-extractable P and TP with PBC and

Table 1 Equilibrium P concentration (EPC_0), P buffering capacity (PBC), and maximum P-sorption capacity (S_{max}) characteristics of GIWs sampled in this study

Site name	Type	Land use	EPC_0 (mg P L^{-1})	PBC (mg P kg^{-1}) (μg P L^{-1})$^{-1}$	S_{max} (mg P kg^{-1})
Site.1	PEM	AG	1294.98	145.49	230.5
Site.2	PEM	AG	174.31	155.78	494.4
Site.3	PEM	AG	9510.65	13.33	
Site.4	PEM	AG	7644.75	69.02	51.3
Site.5	PEM	AG	8951.52	67.32	53.7
Site.6	PEM	AG	10100.57	159.73	132.9
Site.7	PEM	AG	11318.81	86.97	
Site.8	PEM	AG	6648.48	158.35	
Site.9	PEM	AG	6420.59	150.57	1031.1
Site.10	PEM	AG	9557.19	51.84	
Site.11	PEM	AG	10947.21	112.02	55.6
Site.12	PEM	AG	9361.50	70.26	54.5
Site.13	PEM	AG	7824.37	118.78	
Site.14	PEM	AG	27180.57	555.78	1155.8
Site.15	PEM	AG	14592.39	4.99	
Site.16	PEM	AG	4872.16	382.32	1708.0
Site.17	PEM	REF	6488.72	24.86	
Site.18	PEM	REF	10250.20	31.66	54.7
Site.19	PEM	REF	6547.12	11.50	66.3
Site.20	PEM	REF	9912.56	17.48	51.1
Site.21	PEM	REF	4333.36	19.97	
Site.22	PEM	REF	8693.92	105.35	89.0
Site.23	PEM	REF	441.84	323.99	1008.5
Site.24	PEM	REF	8512.53	256.27	
Site.25	PEM	REF	5.74	146.74	1313.1
Site.26	PEM	REF	9664.58	17.56	93.1
Site.27	PEM	REF	9364.84	17.31	185.3
Site.28	PEM	REF	478.08	20.49	102.6
Site.29	PFO	AG	8348.39	279.20	270.8
Site.30	PFO	AG	10545.89	43.21	
Site.31	PFO	AG	478.32	140.75	207.0
Site.32	PFO	AG	2632.71	491.52	1092.9
Site.33	PFO	AG	19709.08	200.78	
Site.34	PFO	AG	9751.94	536.75	
Site.35	PFO	AG	2466.12	311.73	740.1
Site.36	PFO	AG	9409.28	428.75	
Site.37	PFO	AG	5605.51	9.52	
Site.38	PFO	AG	3315.56	927.76	3056.7
Site.39	PFO	AG	485.90	540.66	1252.0
Site.40	PFO	AG	3248.74	511.55	2851.3
Site.41	PFO	AG	1104.62	782.32	3215.8
Site.42	PFO	AG	95.70	540.96	1635.0
Site.43	PFO	AG	225.48	324.99	952.5
Site.44	PFO	AG	5.09	265.22	1361.7

Table 1 continued

Site name	Type	Land use	EPC_0 (mg P L^{-1})	PBC (mg P kg^{-1}) (μg P L^{-1})$^{-1}$	S_{max} (mg P kg^{-1})
Site.45	PFO	REF	4949.18	255.36	5115.0
Site.46	PFO	REF	6683.14	278.96	763.4
Site.47	PFO	REF	11941.34	221.44	107.0
Site.48	PFO	REF	8054.28	85.96	
Site.49	PFO	REF	1825.97	284.46	680.8
Site.50	PFO	REF	2361.38	315.53	815.2
Site.51	PFO	REF	6246.36	74.10	93.9
Site.52	PFO	REF	2478.82	74.95	333.3
Site.53	PFO	REF	1683.63	78.35	118.6
Site.54	PFO	REF	2315.64	257.18	827.0
Site.55	PFO	REF	6171.92	17.25	

Sites were determined to be either palustrine emergent marsh (*PEM*) or palustrine forested (*PFO*) wetlands based on predominant vegetation type. Land use was determined to be either agriculturally (*AG*) or reference condition (*REF*) based on visual assessment of the area immediately surrounding the site (i.e., within an approximate visually estimated buffer area of ~300 m). The data are sorted based on type, then land use

S_{max}), and the following were significantly correlated with PBC and S_{max}: latitude, organic content, specific conductance, and TN. Nitrite–nitrate as N was significantly linearly correlated with PBC.

Following the ANOVA results, CI trees were explored for EPC_0 (all sites, $n = 55$); PBC [PEM ($n = 28$) and PFO ($n = 27$); AG ($n = 32$), and REF ($n = 23$)]; S_{max} [PFO ($n = 20$) and PEM ($n = 19$)]; data voids were filled using average values for seven sites for pH, three sites for specific conductance, two sites for TC, and one site for TP and TN. Four groups were identified for EPC_0, split first on values for soil H_2O-extractable P, followed by bulk density and HCl-extractable P (Fig. 2). The PBC CI trees split into binary groups on TP (PBC–PEM; Fig. 3a), HCl-extractable P (PBC–PFO; Fig. 3b), HCl-extractable P again (PBC–AG; Fig. 3c), and organic content (PBC–REF; Fig. 3d). A single CI tree with a bifurcation at TP > 170 mg kg^{-1} was grown for PEM values of S_{max}, and that was only grown if the minimum group size was relaxed to ≤4 (Fig. 4). No tree was successfully grown for S_{max} PFO.

Discussion

Equilibrium phosphorus concentration

The EPC_0 is a measure of the ability of a wetland system to retain P, and equates to the point at which the P adsorbed is in balance with that in the pore water (Reddy and DeLaune 2008). Water entering the wetland system with *p* values <EPC_0 (e.g., direct precipitation or runoff without entrained P) would cause the release of P bound to the soils, thereby increasing P concentration in the water column until equilibrium is reached. Higher EPC_0 values thus imply decreased P sorption ability and increased potential P release, as well as suggesting many of the P-sorption sites within the soil matrix have been utilized by system P. Continued precipitation within the wetland basin of high EPC_0 sites resulting in overland flows or connection between GIWs and downstream waters (e.g., Leibowitz and Vining 2003; Pomeroy et al. 2014; Golden et al. 2015) would then result in movement of the P from the wetland "hot-spot" to a receiving water, with potentially deleterious effects of increased eutrophication.

Our EPC_0 result, which averaged 6.42 mg P L^{-1} ± 5.18 (range 0.01–27.18 mg P L^{-1}), was higher than many wetland studies, though within ranges expected in wetland systems. For instance, Dunne et al. (2006) reported EPC for pore-water (EPC_w, akin to EPC_0) for soluble reactive *p* values in historically GIWs ranged from 0.12 to 1.3 mg P L^{-1} during a 4-week laboratory study, while Pant and Reddy (2003) reported EPC_w values from 1.3 to 3.4 mg P L^{-1}; Young and Ross (2001) reported values from 0.02 to 7.2 mg P L^{-1}, and Lyons et al. (1998) reported mean EPC_0 values from different soil

Table 2 Means and standard deviations (SD) for soil parameters sampled

Parameter	Type				Land use			
	PEM		PFO		AG		REF	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD
Bulk density (dry, g mL^{-1})	1.038	0.250	0.668	0.255	–	–	–	–
HCL extractable P (mg kg^{-1})	65.582	139.262	259.296	303.206	250.813	299.710	35.274	32.638
Organic content (%)	11.5	13.9	32.1	15.1	–	–	–	–
Total (as P, mg kg^{-1})	112.025	129.707	612.769	1057.033	506.313	975.859	130.305	138.549
Specific conductance (µs cm^{-1})	56.370	39.074	126.840	61.673	107.807	63.855	64.333	49.729
Total Carbon (mg kg^{-1})	–	–	–	–	279064.516	193517.258	394590.900	245409.400
Total Nitrogen (mg kg^{-1})	1955.500	2031.774	5973.846	3182.707	–	–	–	–

H_2O-extractable P (mg kg^{-1})	AG land use		REF land use	
	Mean	SD	Mean	SD
PEM	12.443[a]	8.230	2.567[b]	2.089
PFO	26.763[a]	23.825	14.891[a]	8.820

Nitrite–nitrate as N (mgkg^{-1})	AG		REF	
	Mean	SD	Mean	SD
PEM	1.469[a]	0.484	1.396[a]	0.371
PFO	6.769[b]	6.728	1.318[a]	0.160

pH	AG		REF	
	Mean	SD	Mean	SD
PEM	4.736[a]	0.558	4.133[b]	1.199
PFO	4.900[a]	0.764	3.367[c]	0.132

Null values (–) indicate no significant differences by either type or land use (see Table 1 for abbreviations). Significant interaction effects were found for H_2O-extractable P, nitrite–nitrate as N, and pH, and the means by interaction and significant differences are indicated by superscript letters. Similar letters were not significantly different at $p < 0.05$

types in riparian forests (i.e., poorly drained, somewhat poorly drained, and moderately well-drained) that ranged from 0.52–3.90 mg P L^{-1}.

Interestingly, our mean EPC_0 study result of 6.42 mg P L^{-1}did not differ by type, land use, or interaction effects. Considering the P loading that occurs in agricultural landscapes (Kröger et al. 2013), it was anticipated that EPC_0 would differ by at least land-use modality. While the highest EPC_0 value was found in an AG setting (27.2 mg P L^{-1}), the lowest value was also found in an AG setting (<0.01 mg P L^{-1}). P is a conservative nutrient that frequently cycles within the depressional wetland basin, with the possible exception of when connected to other systems (see Leibowitz and Vining 2003; Bhadha et al. 2011). It may be that there is substantial production in the surrounding wetland basin such that relatively high amounts of P are translocated to the wetland depression on the landscapes regardless of the land use. Bhadha et al. (2011) found that atmospheric deposition into their GIWs was ~2 % of the total P load, but perhaps the cumulative load over time would eventually become deposited in these depressional features on the landscape. However, we found significantly different soil TP values between AG and REF settings (see Table 2), which suggests that land use has

Table 3 Linear correlations between P retention metrics and site characteristics

	$(EPC)^{-1/2}$	$(PBC)^{-1/2}$	Log (S_{max})	Lat.	Long.	Size (ha)	Bulk density
$(PBC)^{-1/2}$	−0.286*						
Log (S_{max})	−0.460**	0.829***					
Latitude	−0.240	0.317*	0.466**				
Longitude	0.354**	−0.279*	−0.397*	−0.562***			
Size (ha)	0.004	0.064	−0.002	0.280*	−0.197		
Bulk density	0.397*	−0.659***	−0.847***	−0.418**	0.420**	−0.206	
Log (H_2O-extractable P)	0.542***	0.143	−0.006	0.018	0.229	0.064	−0.221
Log (HCl-extractable P)	−0.207	0.710***	0.618***	0.341**	−0.278*	0.142	−0.743***
Log (nitrite–nitrate as N)	0.073	0.291*	0.290	0.212	−0.043	−0.201	−0.211
Arc sine $(OM)^{-1/2}$	−0.221	0.544***	0.639***	0.512***	−0.245	0.236	−0.893***
Log (pH)	−0.067	0.221	0.099	−0.372**	0.192	−0.322*	−0.160
Log (TP)	−0.021	0.629***	0.579**	0.252	−0.048	0.072	−0.693***
Log (specific conductance)	−0.087	0.542***	0.404*	0.200	−0.008	0.024	−0.729***
Log (TC)	−0.039	0.075	−0.064	0.204	0.149	0.251	−0.322*
Log (TN)	−0.041	0.457**	0.428**	0.541***	−0.308*	0.316*	−0.886***

	Log (H_2O-extractable P)	Log (HCl-extractable P)	Log (nitrite–nitrate as N)	Arc sine $(OM)^{-1/2}$	Log (pH)	Log (specific conductance)	Log (TC)	Log (TN)
$(PBC)^{-1/2}$								
Log (S_{max})								
Latitude								
Longitude								
Size (ha)								
Bulk density								
Log (H_2O-extractable P)								
Log (HCl-extractable P)	0.482**							
Log (nitrite–nitrate as N)	0.413**	0.534***						
Arc sine $(OM)^{-1/2}$	0.316*	0.586***	0.200					
Log (pH)	0.079	0.391**	0.464**	−0.173				
Log (TP)	0.573***	0.794***	0.570***	0.639***	0.257			
Log (specific conductance)	0.598***	0.755***	0.434**	0.658***	0.249	0.814***		
Log (TC)	0.146	0.111	−0.202	0.429**	−0.295*	0.214	0.293*	
Log (TN)	0.326*	0.508***	0.195	0.655***	−0.148	0.592***	0.487**	0.441**

The significance of the correlation (Pearson's *r*) is indicated: * <0.05; ** <0.01, *** <0.0001. Units for the parameters, where appropriate, are given in Tables 1 and 2

an impact on the concentration of P in a given wetland, as found by Cheesman et al. (2010), amongst others (e.g., Marton et al. 2013). Though we did not contrast *p* values in uplands, as noted above Cheesman et al. (2010) reported that wetlands in an Okeechobee basin (Florida) had greater storage of TP than the surrounding uplands, 236 versus 114 kg ha^{-1}, respectively. Marton et al. (2014) similarly found that depressional wetlands sampled in Ohio had greater sorption capacities than uplands, 297 and 86 mg P kg^{-1} soil,

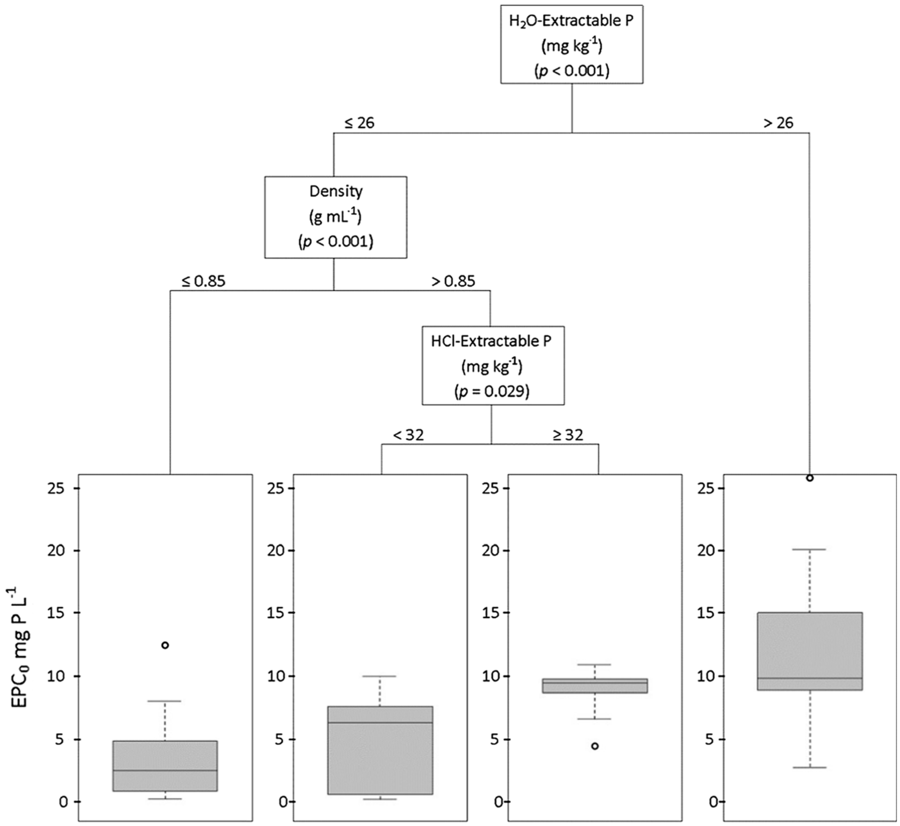

Fig. 2 Conditional inference (CI) tree grown using EPC_0 values for all sites ($n = 55$) classified the data into four groups based on H_2O-extractable P, followed by bulk density and HCl-extractable P

respectively. We sampled in the center of each wetland, which would likely be the most buffered areas with the furthest distance to sources of perturbation. Murray-Hudson et al. (2012) found that the vegetation within the center area of impacted and reference GIWs was more similar than the peripheral vegetation zones. Perhaps P retention characteristics are likewise buffered by distance such that no significant differences were found. For instance, we found similarly high OM concentration between wetland types (see Table 2) regardless of perturbations associated with different land-use modalities and OM is strongly related to P retention characteristics (see Hoffmann et al. 2009; Kröger et al. 2013).

Longitude, bulk density, and H_2O-extractable P were significantly correlated (Pearson's r) with EPC_0 (see Table 3). Longitudinal relationships may be due to the sampling of many "reference" sites on Florida's central sand ridge, and many AG sites on the relatively lower areas to the west of the ridge (Brown et al. 1990), though longitude was not correlated with OM. To further explore these relationships, and to explore relationships other than linear, we grew CI trees. The classification tree grown for EPC_0 identified four GIW groups based on splits at the concentration of H_2O-extractable P (at 26 mg L^{-1}), bulk density (at 0.85 mg L^{-1}), and HCl-extractable P (at 32 mg L^{-1}), suggesting that, statistical significance of the comparative statistical tests aside (see Table 2), antecedent P levels and soil densities are important controlling factors affecting EPC_0 values. Note that these results also suggest that our binary classifications of "reference" and "agricultural" settings, as well as "forested" and "emergent marsh" systems, belie the simple classifications of the complex ecosystems that we analyzed. Frequently, marsh systems would have substantial but unquantified forested components and vice versa, while agricultural systems may be relatively unimpacted by cattle depending upon unquantified stocking densities, flow direction, abundance of other systems, etc. For instance, Lane et al. (2012) found that approximately 50 % of the putative GIW area in an eight-state study was comprised of wetlands with multiple vegetation classes. Other researchers

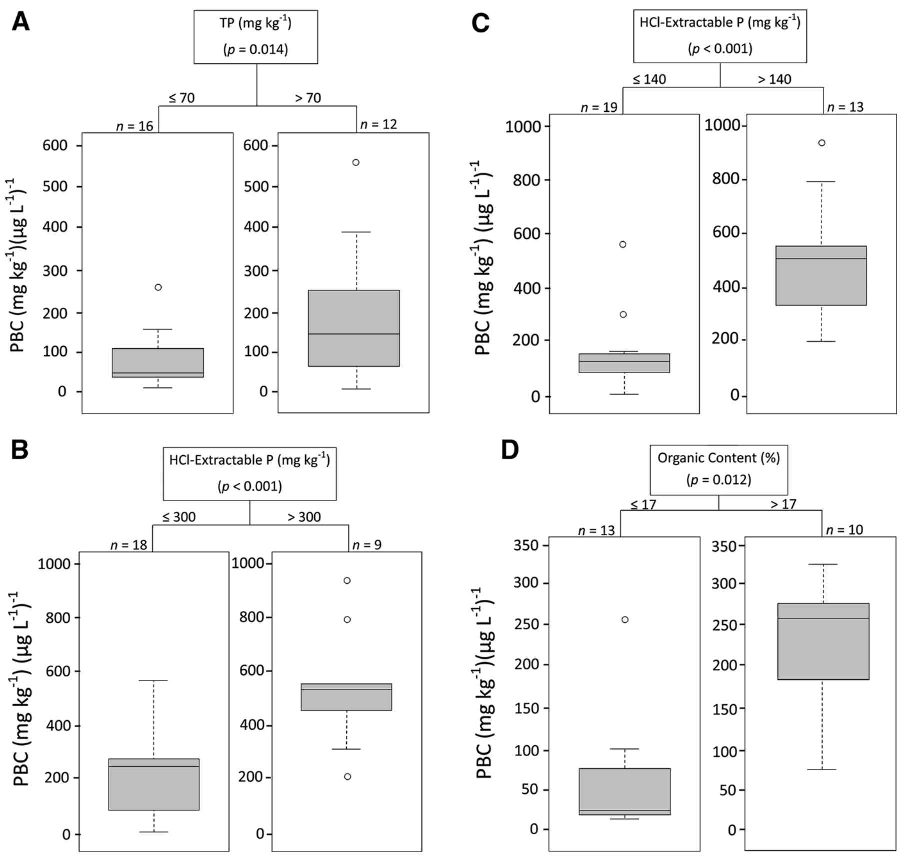

Fig. 3 **a–d** Following the ANOVA results, conditional inference trees were grown using PBC values for **a** PEM ($n = 28$), with a split based on TP, **b** PFO ($n = 27$), with a single split based on HCl-extractable P, **c** AG ($n = 32$), with a split again based on HCl-extractable P, and **d** REF ($n = 23$), with a split based on organic content

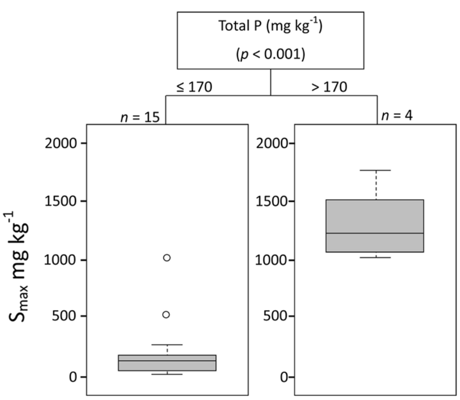

Fig. 4 Conditional inference tree grown using S_{max} values for PEM ($n = 19$) classified the sites into two groups based on TP; no tree could be grown for PFO

have found that antecedent P levels were significantly positively correlated with P retention dynamics (e.g., Reddy et al. 1998; Bridgham et al. 2001; Dunne et al. 2006), with higher antecedent levels potentially a result of higher OM levels and concomitant sorption sites within humic-Fe(Al) compounds or Ca/Mg complexes. The H_2O-extractable P represents the most easily released P to the water column in the system, and the CI tree classes with the highest EPC_0 values were those with the highest H_2O-extractable *p* values, suggesting these wetland soils will readily release P and serve as a P source to other systems (e.g., downstream waters if connected via ditches, upland areas if overland flow occurs, etc.,). All of the wetlands in the highest EPC_0 class (average 11.98 ± 7.52 mg P L^{-1}) were located in AG settings, with only two emergent marsh systems. This suggests that these systems are likely functioning as biogeochemical P hotspots, P storage areas on the landscape until events with low-concentration P waters connect these wetlands to other systems.

A subsequent split was found based on bulk density; the lowest EPC_0 values (3.23 ± 3.09 mg P L^{-1}, $n = 22$) were found in wetlands with the lowest soil bulk density, as one would surmise based on linear correlations (see Table 3). Only three of these were emergent marshes, reflecting the significant differences in bulk density between wetland types (see Table 2); the low-EPC_0 sites were more evenly split between REF ($n = 13$) and AG ($n = 9$). Though in a study of P sorption in riverine soils, Bridgham et al. (2001) did not find bulk density to be a factor in P sorption, our results suggest that bulk density has a controlling effect on EPC_0 that transcends land use, and which may be related to lower bulk density soils likely having less surface area/sorption sites and increased water-holding capacities for P sorption kinetics to occur (Kröger et al. 2013). The final and middle two CI tree splits occurred based on HCl-extractable P, a measure of the P retained in calcium and magnesium complexes. The HCl-extractable P generally comprises a higher portion of the soils in neutral to alkaline settings, and the CI class with the higher HCl-extractable P did have significantly higher pH (Wilcoxon test $Z = 2.1162, p = 0.0343$). Yu et al. (2006) found that soil pH, as well as Ca and Fe were negatively correlated with TP in surface runoff from an agricultural setting and concluded that these three parameters could control P solubility. More soluble P could contribute to higher EPC_0 values, though in this study we found lower EPC_0 in high bulk density sites with higher values of HCl-extractable P. The HCl-extractable P split appears that it could bifurcate the sites based on structure more so than chemical constituents, as all but one of the 24 sites were dominated by non-woody plants, which were found to have significantly greater HCl-extractable P (see Table 1). Eight of the ten sites with HCl-extractable $P > 32$ mg kg^{-1}, and with the second lowest average EPC_0 of 4.87 ± 3.85 mg P L^{-1}, were categorized as AG sites, while only six of the 14 higher EPC_0 (8.99 ± 1.81 mg P L^{-1}) group were AG. Though we did not otherwise quantify calcium or magnesium in our study, P complexes have been significantly positively correlated with the abundance of these elements in wetland (Novak and Watts 2006) and lake systems (Belmont et al. 2009).

Phosphorus buffering capacity

The PBC is a measure of the amount of P that a system can absorb before releasing P. Lower PBC values imply a decreased ability to "…modulate the effect of further P additions on solution concentrations of P" (Sui and Thompson 2000, p. 168). Our study found that emergent marshes, and wetlands in "reference" settings both had the lowest ability to modulate P additions, or the greatest effect of P additions would be found in those systems. Emergent marshes and reference wetlands were both sites with low organic content, and lower OM may affect the ability of the soil to buffer additional P loadings as fewer sorption sites would be available for binding P (e.g., with humic complexes; Hoffmann et al. 2009).

Linear correlations between PBC and latitude likely reflect additional forested systems sampled in the northern portion of the state, while a negative relationship with bulk density may be due to higher soil densities with increased sand concentration in the samples, though this is only speculative as it was not quantified. Sand, versus OM, would be expected to have dramatically fewer sorption sites for buffering P loading. To further explore the relationships between PBC and soil parameters across classes based on type and land use, four different CI trees were developed. Three of the four trees bifurcated based on P availability, either TP (emergent marshes) or HCl-extractable P (forested systems; AG wetlands). The REF sites were split into two groups based on the percent organic content (strongly and inversely linearly correlated with bulk density) of the soils.

Emergent marshes (Fig. 3a) were split into two PBC classes based on TP at 70 mg kg^{-1}, with a low PBC class [65.66 ± 68.29 (mg P kg^{-1}) (µg P L^{-1})$^{-1}$, $n = 16$] and a high PBC [183.10 ± 160.73 (mg P kg^{-1}) (µg P L^{-1})$^{-1}$, $n = 12$].Three REF sites were among the 12 high PEM PBC sites with >70 mg kg^{-1} TP. The high PBC sites (220.00 ± 136.32 mg kg^{-1} TP) averaged seven times higher soil TP than the low PBC class (31.04 ± 18.28 mg kg^{-1}). While possible that these three REF sites had been previously exposed to high P loading, all three were located in a national forest, so near-pristine antecedent conditions likely prevailed. However, P may have accumulated in these systems over time and not been "flushed", or intermittent overland flow may have brought substantial OM replete with P (and/or P-sorption sites), as the "high PBC" sites also averaged three times the OM content than "low PBC" sites (18.8 vs. 6.1 %). The low class of PBC emergent marshes, including those in agricultural settings, had low organic content (ranging from 3.4–12 %). OM content may contribute to the higher PBC, or resistance to change following P loading, as OM frequently has high concentrations of metallic compounds present in amorphous or poorly crystalline forms complexed with OM which can positively contribute to P sorption capacities, though OM can also compete with P for sorption sites (Bhadha and Jawitz 2010).

Forested wetlands and wetlands in agricultural land-use modalities were both split into two groups (Fig. 3b, c) based on HCl-extractable P with splits at 300 mg kg^{-1} (forested) and 140 mg kg^{-1} (AG). The mean PBC values for the "high PBC" classes for both PFO [522.14 ± 225.96 (mg P kg^{-1}) (μg P $L^{-1})^{-1}$] and AG (Fig. 3, 478.03 ± 206.26 (mg P kg^{-1}) (μg P $L^{-1})^{-1}$ are similar, though the "low PBC" classes have a greater deviance between their averages [PFO average 198.88 ± 141.25 (mg P kg^{-1}) (μg P $L^{-1})^{-1}$]; AG average 127.57 ± 124.91(mg P kg^{-1}) (μg P $L^{-1})^{-1}$. As noted earlier, HCl-extractable P is a measure of the P retained in Ca and Mg complexes. Higher HCl-extractable P concentrations suggest higher Ca or Mg concentrations, with potential for greater opportunities for P to be bound to Ca and Mg complexes in the soil matrix, thereby increasing the resilience of "high PBC" classes to P loading.

PBC in REF sites (Fig. 3d) was found to fall into low and high classes based on percent organic content, with a bifurcation occurring at 17 % separating "low PBC" [average 54.16 ± 68.29 (mg P kg^{-1}) (μg P $L^{-1})^{-1}$] from "high PBC" [223.27 ± 54.16 (mg P kg^{-1}) (μg P $L^{-1})^{-1}$] classes. Though split based on land use, the bifurcation closely follows wetland typology, with 77 % (10 of 13) of the "low PBC" REF GIWs classed as emergent marshes (with 9 of 10 emergent marshes located on Florida's sand ridge) and 80 % (8 of 10) of the "high PBC" GIWs classed as forested wetlands (with no forested "high PBC" systems located on the Sand Ridge). The average OM content of the "high PBC" sites was 34 % (±13.2) while the "low PBC" sites averaged 8.2 % (±3.6). Forested systems, or systems with ample amounts of forested structure, in addition to being frequently found off the central sand ridge in Florida, likely have higher OM in the soil profile due to recalcitrant decomposition of woody material and leaves versus herbaceous material in emergent marshes (Ewel 1990; Kushlan 1990), though they may also have shorter hydroperiods which would increase soil oxidation over more herbaceous systems. As noted above, this higher OM content may have increased concentrations of metal (e.g., Fe, Al, Mg) complexes, which may explain the higher PBC associated with high levels of OM.

Maximum phosphorus retention

The S_{max} provides a metric for adsorption maxima, quantifying a point where all available adsorption sites in the sample have been filled. In our study, we were able to successfully fit the data to Langmuir equations in 71 % (39 of 55) of our sites. Sites without S_{max} values were fairly evenly distributed between PEM (56.25 % of the 16 sites without S_{max} values) and PFO (43.75 %), though twice as many sites were in REF land use (68.75 vs. 31.25 %). A comparison of soil characteristics between sites with and without S_{max} values found only bulk density to vary significantly (Wilcoxon $Z = 2.0262$, $p = 0.0427$), though the effect of this on fitting Langmuir equations is unknown.

In a study of riverine and depressional wetlands of the southeastern US, Cohen et al. (2007) reported single-point P sorption index (PSI) values from <0 to 990 mg P kg^{-1}, with a mean of 462.7 ± 295.2 mg P kg^{-1}. Axt and Walbridge (1999) reported PSI ranges in bottomland palustrine forested wetlands and adjacent upland soils of North Carolina's Coastal Plain and Piedmont to range from 520 to 3410 mg P kg^{-1}. Our study found S_{max} values (i.e., multi-point isotherms, which have been found to have similar results to single-point isotherms (Reddy et al. 1998) to range from 51.3 to 5115.0 mg P kg^{-1}. Forested wetlands (1274.5 ± 1315.7 mg P kg^{-1}) had a significantly higher S_{max} than emergent marshes (417.5 ± 534.6 mg kg^{-1}), though no other main or interaction effects were significant. These results differ from Cohen et al. (2007), who found significant differences between the PSI values in between "minimally impacted" and "impacted" wetland systems in two of three southeastern regions, one of which (Southern Coastal Plain, Omernik 1987) covers sites in this study. However, these values include both riverine and depressional systems, the former of which was reported by Cohen et al. (2007) to sorb 308.3 mg kg^{-1} more P than the depressional systems of the Southeastern Coastal Plain.

A CI tree was only able to be grown for PEM, frustrating our exploration of the relationships between forested wetland and P sorption. Our PEM CI tree was grown with a split between low and high maxima occurring at TP > 170 mg kg^{-1}, though the higher class only has four members. The S_{max} of the higher class averaged over seven times that of the lower (181.6 ± 256.5 vs. 1302.0 ± 294.2 mg P kg^{-1}). Linear correlations (see Table 3) identified bulk density and organic content as the two most strongly correlated variables with S_{max}; note that these two variables are also highly correlated (Pearson $r = -0.89, p < 0.0001$). As noted elsewhere, increased organic content may increase the abundance of Fe and Mg compounds with which P may create complexes, depending on additional characteristics such as redox potential and pH while bulk density may reflect a decrease in sorption sites.

Conclusion

Results of this study reinforce the findings (e.g., Dunne et al. 2006; Cohen et al. 2007; Dunne et al. 2010; Bhadha and Jawitz 2010; Min et al. 2010) that GIWs of the Florida peninsula have a high potential to retain P, but with the retention comes the possibility that the P sink may become a P source when runoff enters the GIW with values <EPC_0. For instance, Yu et al. (2006) reported P in runoff over 44 rain events across five Florida citrus grove sites to range from 0.51 to 2.64 mg P L^{-1}, sufficiently below our measured EPC_0 value to suggest that each event would cause a release of P to the water column, further increasing the P concentration. However, the majority of the P entrained in the runoff in their study, and that of P-fractionations in wetland soils by Dunne et al. (2006), was either in plant-available forms, or forms mineralized by microbial or enzymatic processes. Thus run off reaching wetlands may be in a form readily incorporated into the microbial or higher plant forms. This, however, assumes a sufficient retention period in wetlands for these processes to occur. That the EPC_0 values were rather high suggests that GIWs should not be connected to other systems via ditches or drains, which decrease retention time, as the GIWs would simply be increasing the P concentration in the water column, which would be promptly shunted downstream or out of the wetland system. Thus, it is the hydrologic dis-connectivity of the GIWs which provides a service to downstream waters by retaining P. Maintaining an adequate hydroperiod in GIWs would support the development of OM in these wetland systems, which would likely increase the P storage capacity, or at the very least, decrease the mineralization rates for OM, keeping P in OM complexes. OM accretion, though low in depressional wetlands when compared with systems adjacent to flowing water systems (Craft and Casey 2000), would provide for long-term burial of P, further supporting the maintenance of hydrology in these wetland systems, which in some cases (e.g., in Florida) have been noted to stay inundated for 64–86 % of a two to three-year study period (Min et al. 2010). Forested wetlands had higher organic content than emergent marsh systems, which was also correlated with higher P sorption in PFOs. Maintaining forested wetlands and the P sorption characteristics of these features, versus elements with decreased nutrient functioning, should be encouraged for long-term sorption of P. However, Dahl (2011) reported a net decrease in forested wetland acreage in the US 2004–2009 of over 250,000 ha. Lastly, Cohen et al. (2008) reported substantial spatial variability in soil TP and OM within four Florida cypress domes and suggested that 11–33 samples are needed to achieve site characterization within 10 % of the true mean. Additional within-site analyses and characterization of P assimilation dynamics (as well as other physical and biogeochemical processes) is thus a fruitful area of research. We conclude that wetland systems perform myriad ecosystem services and determining and calculating the net services of GIWs can facilitate how resource managers prioritize the landscape to maximize these benefits (Blackwell and Pilgrim 2011).

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