Redistribution of Wetland Soil Phosphorus Ten Years after the Conclusion of Nutrient Loading

Erin M. Bostic
Soil and Water Science Dep.
Univ. of Florida
Gainesville, FL 32611

John R. White*
Wetland & Aquatic Biogeochemistry Lab.
Dep. of Oceanography and Coastal Sci.
Louisiana State Univ.
Baton Rouge, LA 70803

Ron Corstanje
National Soil Resources Inst.
Cranfield Univ.
Cranfield, Bedfordshire, MK43 0AL, UK

K. Ramesh Reddy
Soil and Water Science Dep.
Univ. of Florida
Gainesville, FL 32611

There is considerable concern about ecological recovery in wetlands that have been enriched with P; however, there are few long-term studies tracking the distribution of the soil P after the termination of P loading. The Blue Cypress Marsh Conservation Area in Florida contains areas with elevated soil P levels from historical loading. The local spatial variation of soil P was determined in a 750- by 150-m area proximal to the historic surface water inflow point and a second area of the same size located in the center of the marsh with no record of historical P impacts. The average soil total P was estimated at 847 mg kg^{-1} in the P-enriched area and 643 mg kg^{-1} in the marsh interior (unenriched). When compared with previously determined historical data, it was estimated that soil P has decreased in the P-enriched area by about 61%. Meanwhile, there was an increase of 82% in P within the unenriched area during this same time period. These results suggest that P has been mobilized from the impacted areas toward the unimpacted marsh along the water flow patterns in the wetland. These observations have implications for restoration of high-P wetland systems. Remobilization from P-impacted to unimpacted areas can expand the area of increased nutrients despite termination of P inputs into the marsh. If remobilization predominates over burial, then it is also unlikely that the overall nutrient status of the system will return to pre-impact levels within a reasonable management time frame (<25 yr).

Abbreviations: BCMCA, Blue Cypress Marsh Conservation Area; MBP, microbial biomass phosphorus; SRP, soluble reactive phosphorus; TP, total phosphorus.

Nutrient enrichment from agricultural and municipal runoff can result in fundamental changes to soil biogeochemistry, water quality, and composition of plant communities in wetland systems (DeBusk and Reddy, 1998; Newman et al., 1997; Reddy et al., 1998). Nutrient loading to wetland systems can result in the formation of soil nutrient gradients emanating from the inflow points and extending outward based on hydrologic flow patterns and surface topography (Amador et al., 1991; Bridgham and Richardson, 1993; Reddy et al., 1993; Newman et al., 1997; Noe et al., 2001). Little is known about the long-term fate and mobility of P in the soil once nutrient loading has been terminated. Unlike N, P does not have a significant gaseous removal mechanism and therefore remains in the system to which it is added.

The prevailing management approach to the restoration of nutrient-impacted wetlands has been to reduce the nutrient load to the wetland with little adjustment to the overall surface water flow patterns within the wetland. As a result of these mitigation efforts, low-P surface water is often introduced into areas with elevated soil P. The expectation is that, with time, the excess P will be buried deeper in the soil as a result of organic matter accretion in wetland systems. There is a growing body of literature, however, pertaining to both lake sediments (Reddy et al., 2007) and wetland soils (White et al., 2004; Corstanje and Reddy, 2004), that demonstrates that exposing soils and sediments with elevated P concentrations to surface water with appreciably lower P results in the release of the soil P to the water column (Malecki et al., 2004). Phosphorus remobilization in this wetland is often caused by enhanced microbial activity in the impacted area, changes in the predominant plant commu-

Soil Sci. Soc. Am. J. 74:1808–1815
Published online 4 Aug. 2010
doi:10.2136/sssaj2008.0316
Received 2 Oct. 2008.
*Corresponding author (jrwhite@lsu.edu).
© Soil Science Society of America, 5585 Guilford Rd., Madison WI 53711 USA
All rights reserved. No part of this periodical may be reproduced or transmitted in any form or by any means, electronic or mechanical, including photocopying, recording, or any information storage and retrieval system, without permission in writing from the publisher. Permission for printing and for reprinting the material contained herein has been obtained by the publisher.

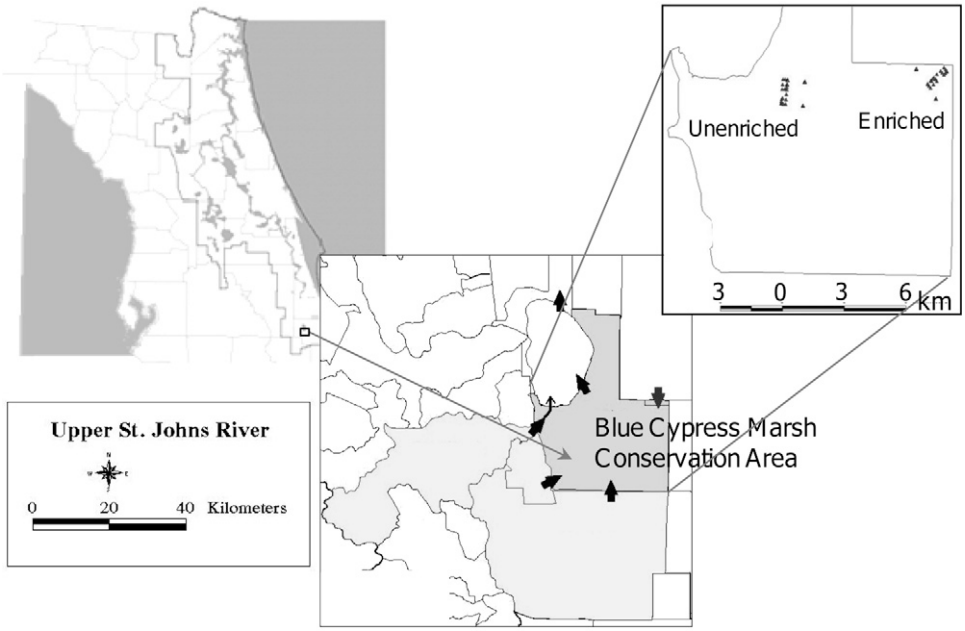

Fig. 1. Blue Cypress March Conservation Area (BCMCA) located in the headwater of the Upper St. Johns River, beginning in the east-central area of Florida and flowing north to the city of Jacksonville. The middle insert shows the agricultural discharge points into the BCMCA and subsequent outflow into the lake (after Prenger and Reddy, 2004). The rightmost insert shows the sampling locations.

nity composition, and seasonal changes in the redox conditions of the soil (Richardson, 1985; Weyers and Suberkropp, 1996; Grace et al., 2008). If this occurs in a wetland with an established nutrient gradient oriented along surface water flow paths, then a portion of the P from the more enriched soils will be transported in the surface water to areas previously not impacted, despite the termination of external P loading to the system. This redistribution of soil P, referred to as the *internal load*, can therefore maintain the wetland in a eutrophic state despite efforts to stop the external load from the surrounding watershed.

The Blue Cypress Marsh Conservation Area (BCMCA) is a freshwater marsh that has experienced nutrient enrichment due to inflows from agricultural lands along its periphery. The overall spatial distribution of soil P in BCMCA has recently been documented in a study by Grunwald et al. (2006), showing areas with elevated soil P close to these inflows. In these areas, numerous studies have identified changes in the plant community structure (Prenger and Reddy, 2004), soil biology (Ipsilantis and Sylvia, 2007), and soil biogeochemical cycling (Corstanje and Reddy, 2004) similar to those described in the well-studied northern Everglades system (DeBusk and Reddy, 1998; Reddy et al., 1998; White and Reddy, 2000). In the early 1990s, these nutrient inputs were diverted away from the BCMCA. Previous studies have documented P release from P-enriched soils (Corstanje and Reddy, 2004; Bostic and White, 2007). What is currently unknown is whether there is an actual loss of P from the impacted areas and whether this leads to an increase in soil P further downstream (i.e., how the spatial distribution of P has changed since curtailing P inputs).

Whereas the study by Grunwald et al. (2006) documented the coarse-scale distribution of soil P in this system, most studies of the processes in this system occurred at a considerably finer scale (e.g., Corstanje and Reddy, 2004: three cores across 1–2 m^2). This is mainly due to resource limitation because large spatial sampling efforts are expensive, time consuming, and sample intensive and detailed process measurements on many samples is often not a viable option. Wetlands are locally heterogeneous systems, however, as is the case in the BCMCA, in which a rich mixture of slough communities, standing aquatic macrophytes, tree islands, willow communities, and wet prairies are present. Sampling and monitoring for changes soil P and associated changes to soil biogeochemical cycling must therefore be sufficiently robust to capture this local heterogeneity, yet remain cost effective for process measurements. In this study, we used a targeted sampling approach in which the localized spatial heterogeneity of soil P was captured.

The specific objectives of this research were therefore to: (i) determine the local spatial distribution of P in an area historically nutrient enriched and in an unenriched area located downstream along surface flow patterns at some distance from the P-impacted area; (ii) determine if there is any other evidence of nutrient movement in either area; and (iii) compare recent values with historical values for each area.

MATERIALS AND METHODS

Study Area

The BCMCA is located in east-central Florida at the headwaters of the St. Johns River basin, which runs northwards to Jacksonville (Fig. 1). This 8000-ha freshwater marsh was modified by the construction of levees that completely surround the marsh. A breach in the levee directed P-enriched nutrient runoff into the northeast (here referred to as the P-enriched) area of the marsh during the 1970s and '80s. The breach was sealed in the early 1990s as part of the St. Johns River restoration project to reduce P loading to the wetland and subsequently to the St. Johns River basin. The historical loading led to the formation of a local P-enriched area adjacent to the levee breach, with the highest concentrations closest to the surface water inflow point as documented by Grunwald et al. (2006).

Since the early 1990s, precipitation has been the primary input of water to the wetland. The typical hydrologic pattern observed for the BCMCA therefore responds closely to precipitation and reflects a typical Florida climate of distinct dry and wet periods in a year, where the winter months are generally the colder and drier months. Water levels vary from between 30 and 50 cm of standing water to complete drawdown and exposure of the soil. During the period in which P loading took place, the vegetative species (*Cladium* spp. and *Panicum* spp.) shifted to a dominance of *Typha latifolia* L. and *Salix caroliniana* Michx. within the enriched areas of the marsh with time (Corstanje and Reddy, 2004), a transformation in vegetation communities that is similar to that which has been described in

the northern Everglades (Wu et al., 1997). The interior of the marsh is characterized by a mosaic of sawgrass (*Cladium jamaicense* Crantz) and maidencane (*Panicum hemitomon* Schult.) flats.

Experimental Design

Sample collection occurred in September 2000 in two areas, one established in the P-enriched area (Fig. 2a) and one in the unenriched area (Fig. 2b). The sites are located at about 10 km from each other and are connected hydrologically as water flows north toward Blue Cypress Lake. These sites were selected in close proximity (within 100–200 m) to sites used in previous studies and referenced here (Brenner and Schelske, 1995; Olila et al., 1995; D'Angelo et al., 1999).

Sampling points were located on two rectangular grids (150 by 250 m) established in the enriched and unenriched areas. A number of samples were located at distances of 1.5, 15, 75, and 750 m apart; this type of nested design ensures that appropriate spatial scales were sampled for the ensuing geostatistical analysis. The grid in the P-enriched area was angled such that it originated from the inflow point at the levee and extended toward the interior of the marsh along the surface water flow path. The unenriched transect was oriented in a north–south direction, oriented parallel to the surface water flow directed by the topographic gradient (Prenger and Reddy, 2004).

Field Sampling

Surficial (0–10-cm) soil samples were taken at each station using a 10-cm i.d. Al push-core. If detrital material, composed of readily identifiable plant leaves and stems was present, it was removed from the soil surface by hand before coring. A total of 82 soil samples was collected, stored on ice, and refrigerated on return to the laboratory. Six locations were replicated with three cores taken within 1.4 m.

Laboratory Analysis

Gravimetric moisture content (70°C until constant weight) and dry-weight bulk density (Blake and Hartge, 1986) were determined for all samples. The pH was determined on soil slurries (1:1 soil/water). Total N and total C were determined on dried, ground samples using a Carlos Erba NA-1500 CNS Analyzer (Haak-Buchler Instruments, Saddlebrook, NJ). Total P (TP) was determined using the ignition method followed by acid digestion (Andersen, 1976) and determined colorimetrically (USEPA, 1993). Acid-extractable Ca, Mg, Fe, and Al concentrations were determined from 0.5 g of oven-dried soil treated with 25 mL of 1.0 mol L^{-1} HCl and placed on a reciprocal shaker for 3 h. The supernatant was filtered through 0.45-μm membrane filters and analyzed by inductively coupled Ar plasma spectrometry (Method 200.7, USEPA, 1993; Malecki-Brown et al., 2007). Microbial biomass P (MBP) was determined as the difference in TP content between 0.5 g dry-weight equivalent chloroform-fumigated and unfumigated samples after a 3-h extraction with 25 mL of 0.5 mol L^{-1} $NaHCO_3$ (Olsen's reagent; Allen et al., 1974; Brookes et al., 1982, 1984; Hedley and Stewart, 1982). Field-moist soil subsamples were subjected to a sequential fractionation after Reddy et al. (1998). Total inorganic P (TPi) was represented by a summation of KCl-, NaOH- and HCl-extractable soluble reactive P (SRP). Total organic P was represented by TP − TPi.

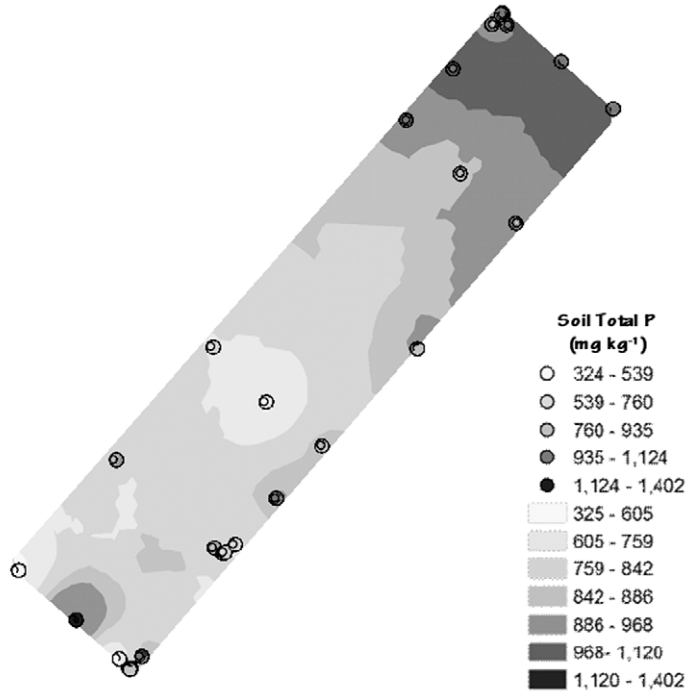

Fig. 2. Spatial distribution of soil total P concentrations along the P-enriched (northeast) area of the Blue Cypress Marsh Conservation Area. The interpolated total P values are mapped and the observed values at the sampling points are represented by the overlaid dots.

Statistical Analysis

There are existing geostatistical models that describe the spatial variation of soil properties across the entire area (Grunwald et al., 2006) and it was this information that we used to determine the local spatial variability of soil P forms in the two areas. The advantage of using this approach is that it allowed us to characterize local patterns within these sites. The geostatistical models obtained by Grunwald et al. (2006) described a coarser scale of variation, so we had to construct a new set of models that described variation at a finer scale. The spatially dependent variance in the data was explored using the semivariance $\gamma(\mathbf{h})$, which can be calculated as

$$\gamma(\mathbf{h})=\frac{1}{2}\mathrm{E}\left\{\left[Z(\mathbf{x})-Z(\mathbf{x}+\mathbf{h})\right]^2\right\} \qquad [1]$$

which under the assumption of second-order stationarity (Matheron, 1971, 1973), is a function of the lag vector **h** only and describes the spatial dependence of the random function $Z(\mathbf{x})$, assumed to be realized in the data at a set of sampling sites $\mathbf{x}_1, \mathbf{x}_2, \ldots, \mathbf{x}_i$. The autovariograms were then obtained using the standard estimator (Webster and Oliver, 2001):

$$\hat{\gamma}(\mathbf{h})=\frac{1}{2N(\mathbf{h})}\sum_{i=1}^{N(\mathbf{h})}\left[z(\mathbf{x}_i)-z(\mathbf{x}_i+\mathbf{h})\right]^2 \qquad [2]$$

where z is the observed random function.

All the data from these two areas were used to construct the experimental autovariogram, assuming that the random function $Z(\mathbf{x})$ is realized throughout the area. There are no clear discontinuities throughout the BCMCA and variograms for the same soil properties as those measured in this study have been produced previously for all of the BCMCA (Grunwald et al., 2006). We therefore had some evidence to

Table 1. Characterization of soil pH, dry-weight bulk density, total C, total N, and total P in the P-enriched and unenriched areas. Data shown are means ±1 standard error in parentheses.

Area	Depth	pH	Bulk density	Total C	Total N	Total P
	cm		g cm^{-3}	g kg^{-1}		mg kg^{-1}
Enriched	0–10	5.8 (0.43)	0.066 (0.019)	467 (12)	29.8 (6)	847 (212)
Unenriched	0–10	5.5 (0.13)	0.067 (0.014)	465 (6.3)	27.8 (2)	643 (77)

support the assumption that observations from the P-enriched and unenriched areas are an expression of the same random function.

In the case where there are strong trends or gradients of soil P, then the assumption with regard to the mean will not be met and detrending or a method based on the residual maximum likelihood (Marchant and Lark, 2007) would have to be applied. In our case, it is clear that the gradients that we described were local (Grunwald et al., 2006), and it was our expectation that these would have little effect on the spatial estimation process. We constrained the lag vector, **h,** to a maximum of 800 m, just over the maximum distance between sample points within each of the areas. For this analysis, soil values were normalized using the total sample standard deviation.

Model variograms were fitted to the experimental variograms of TP by maximum likelihood (Lark, 2000) in the geoR package in R (Ribeiro and Diggle, 2001). The exponential model provided the most parsimonious fit according to the computed Akaike's information criterion (Akaike, 1973). This is expressed as

$$\gamma(\mathbf{h})=c_0+c_1\left[1-\exp\left(-\frac{\mathbf{h}}{r}\right)\right] \qquad [3]$$

with nugget c_0, sill c_1, and range r. The nugget estimates the semivariance with no observed spatial dependence, the sill describes the total estimated semivariance, and the range defines the spatial extent of the model. The estimated model parameters for TP were c_0 = 0.4082, c_0 = 1.0759, and r = 100 m. A map of the interpolated values was generated using kriging in the ArcMap environment (ArcMap version 8.3, ESRI, Redlands, CA).

As we also conducted a simple comparison of means tests (Student's *t*-test) between the P-enriched and unenriched areas at an α level of 0.05. There is an inherent assumption that the treatments (sites) are independent. We believe that this assumption does hold because the largest observed range of autocorrelation reported by Grunwald et al. (2006) is smaller than the distance between the two areas. The variance observed in the data is reported as ± standard error.

RESULTS

The dry-weight bulk density values were generally low and not significantly different between sites, with a mean of 0.066 g cm^{-3} for the P-enriched site and 0.067 g cm^{-3} for the unenriched site (Table 1). There was also no significant difference between sites for total C (467 and 465 g kg^{-1}) or total N (29.8 and 27.8 g kg^{-1}) for the P-enriched and unenriched sites, respectively.

Total P was significantly different, with a higher mean value of 847 mg kg^{-1} in the P-enriched site vs. 643 mg kg^{-1} for the unenriched site (Table 1). The MBP was not significantly different, with a mean of 151 mg kg^{-1} for the P-enriched site and 181 mg kg^{-1} for the unenriched site. When we consider MBP as a percentage of total P, however, which is a measure of P limitation, we find that there is a significant difference of 17.9% for the P-enriched site and 28.1% for the unenriched site. This higher percentage of microbial biomass to total P in the unimpacted area is consistent with greater nutrient limitation (Reddy et al., 1999).

The majority of P present is organic (Table 2) and we observed significant differences in the organic P content in the soil layer, with mean 0- to 10-cm soil organic P concentrations in the unenriched and P-enriched sites of 522 (±55.7) and 699 (±170) mg kg^{-1}, respectively, representing approximately 81.1 and 82.2%, respectively, of the soil layer TP. The $NaHCO_3$–extractable SRP from 0 to 10 cm demonstrated no significant concentration variation between areas. The P-enriched and unenriched areas contained 0.34 (±0.04) and 0.39 (±0.04) mg $NaHCO_3$–extractable SRP L^{-1}, respectively. The quantities of Fe, Al, and Ca in the soil and detrital layers were similar quantities in the samples obtained from both areas (Table 2).

Local Spatial Patterns

A distinct P gradient was observed in the P-enriched site of the marsh (Fig. 2). There was an overall estimated mean difference of 311 mg TP kg^{-1} in the 0- to 10-cm soil layer along the length of this gradient, with larger concentrations proximal to the historical inflow point than farther from the inflow point. The highest TP levels, expressed as the fourth quartile of the interpolated TP values (868–1402 mg kg^{-1}), covered about 22% of the P-enriched site values above the interpolated TP median of 837 mg kg^{-1}. These values where primarily located close to the inflow point.

We also observed a gradient in soil P in the unenriched site, from south to north (Fig. 3). The highest values, represented by the fourth quartile, 700 to 757 mg kg^{-1}, covered an estimated 15% of the area. These values were primarily located on the southern end of the sampling grid, at the upper end of the flow direction. It should be noted that both areas show some localized heterogeneity, with large TP observations (points in Fig. 2 and 3) close to smaller values, although the estimated nugget variance

Table 2. Mean concentrations (n = 36) of total inorganic P, total organic P, $NaHCO_3$–extractable soluble reactive P (SRP), microbial biomass P, and HCl-extractable Al, Ca, and Fe within the P-enriched and unenriched areas. Data shown are mean concentrations ±1 standard error in parentheses.

Area	Depth	SRP	Total inorganic P	Total organic P	Microbial biomass P	Al	Ca	Fe
	cm	mg L^{-1}	mg kg^{-1}				g kg^{-1}	
Enriched	0–10	0.34 (0.04)	151 (97)	699 (170)	151 (93)	277 (98)	16 (2)	0.97 (0.34)
Unenriched	0–10	0.39 (0.04)	121 (27)	512 (56)	181 (45)	217 (39)	11.3 (3)	1.19 (0.77)

comprises only about 38% of the total estimated variance for this data. Post plots (not shown) of extractable SRP in both areas closely resembled the patterns observed for TP.

DISCUSSION

Traditional sampling of a wetland involves obtaining a number of replicate samples at the desired number of sites, in which the sites represent treatments and the difference between sites is then reported. This approach ignores spatial variation in that this variation is lumped in with the treatment effect. The alternative is to execute a spatial survey of the area, which often requires a considerably larger sampling effort and cannot be as targeted because a minimum of coverage is required to effectively describe the spatial variation across the area. We used a novel approach to sampling that builds on a previous geostatistical survey. Geostatistical methods are generally more robust for interpolation but require a fairly large sample size (>100 samples; Webster and Oliver, 2001). They also assume that the observed variation is the expression of the same random function. In our case, we were able to combine the observations from the two sites to generate a model of the spatial variability because we knew from a previous survey that these observations belonged to the same random function (Grunwald et al., 2006). This allowed us to target two areas efficiently, and the spatial variation of key properties could then be described at two fairly disparate scales: a finer scale (within site) and a coarser scale (between sites).

At the finer scale, we were able to observe that spatial patterns of soil P in both areas can be described as gradients. At the enriched site, the higher concentrations were nearer to the inflow point and these decreased farther into the marsh along what we suspect is the predominant flow path into the marsh interior. We also observed a gradient at the unenriched site, with higher concentrations along the southern edge and decreasing in a northerly direction. We are less certain as to what might be causing this gradient. At this site, the lake is to the north and we suspect that the predominant water flow path is south to north. This area of the marsh is also shallow, however, consisting of *Panicum* spp. flats interspersed with clumps of *Cladium* spp., so sheet flow toward the lake can be obstructed when water levels are low. An airboat trail runs close, about 10 to 20 m south of the southern end of this transect, which is part of a network of trails that connect entry points into the BCMCA along the levees to the lake (personal observation). It may be the case that these trails form a hydrologic shunt through the marsh toward the lake and are therefore the source of P at the southern end of the unenriched site. The coarse-scale map of P distribution (see Fig. 6 in Grunwald et al., 2006) depicts a gradient in soil P from south to north; however, the local concentrations we found at the southern end are substantially larger than those observed on the coarse-scale map for this general area. This would seem to suggest that the trails, rather than sheet flow movement, are conduits for the P-laden water from the impacted areas. Although we cannot identify the exact hydrologic pathway, the enrichment observed at the unimpacted site would suggest that there is some remobilization of P toward the unenriched site.

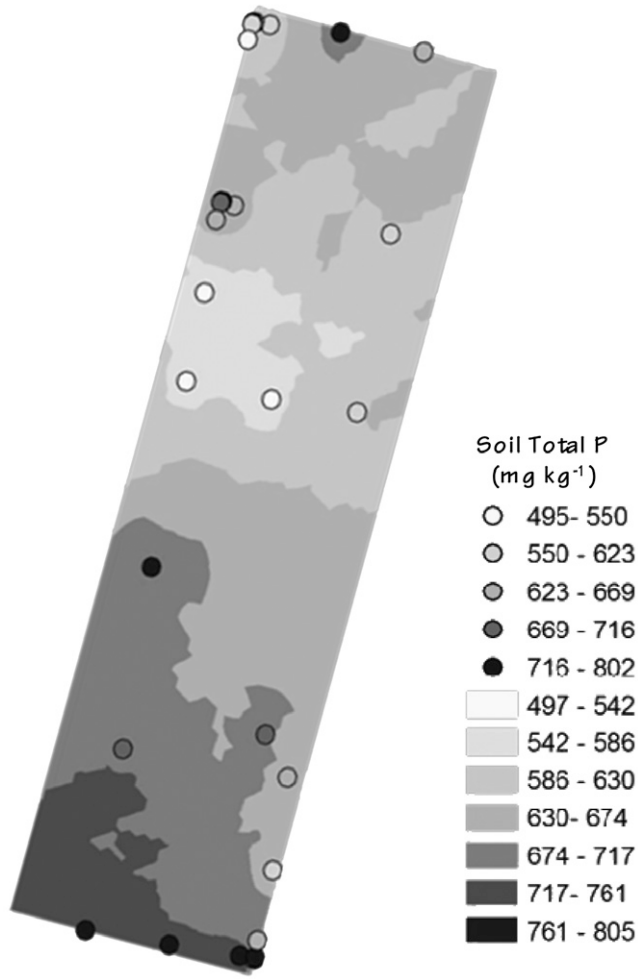

Fig. 3. Spatial distribution of soil total P concentrations along the unenriched areas (northwest) of the Blue Cypress Marsh Conservation Area. The interpolated total P values are mapped and the observed values at the sampling points are represented by the overlaid dots.

In these oligotrophic systems, microbial communities are often a response variable to changing levels of P availability (Corstanje et al., 2007), with increasing bioavailability of P resulting in increased levels of MBP (Reddy et al., 1999). Given the sampling design, we were able to construct a set of general linear models for each area, with MBP as the response variable, with sampling coordinates and a number of key physiochemical variables: TP, total N, total C, pH, and SRP as predictors. In the case of the P-enriched area, the model performance was poor ($R^2 = 0.34$; Fig. 4) and TP the only significant predictor ($P = 0.02$). This would suggest that the variation in MBP in this area is spatially heterogeneous, with no linear or gradient trend. These same variables were generally effective predictors at describing the variation in MBP in the unenriched area ($R^2 = 0.76$; Fig. 4), however, and the variation in MBP had a general south to north trend ($P = 0.02$), suggesting a gradient in MBP that may be responding to P inputs.

The coordinates can be viewed as proxies for effects of the nutrient gradient that are not described by the other predictor variables, such as changes in nutrient cycling rates or responses to

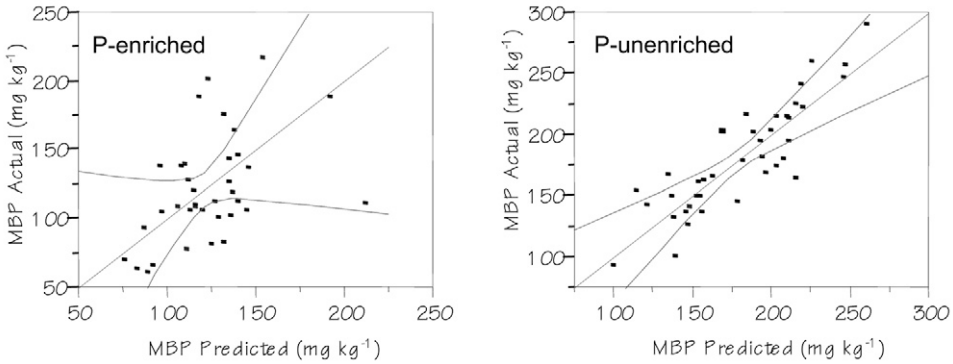

Fig. 4. Performance of predictive models of microbial biomass P (MBP), in which the geographic coordinates, total C, total N, total P, pH, and soluble reactive P were introduced as predictors. Performance statistics were $P = 0.146$, $R^2 = 0.34$, and RMSE = 36.49 for the P-enriched area and $P < 0.0001$, $R^2 = 0.76$, and RMSE = 24.45 for the unenriched area. The solid line represents the 1:1 observed to predicted line; the dashed lines represent the 95% confidence around this line.

increased levels of nutrients by the vegetation. If we assume (based on Reddy et al. [1999] and our previous research, e.g., Corstanje et al., 2007) that microbial communities are responsive to eutrophication (indicators), then where there is an overall change in MBP that can be ascribed above and beyond the change in TP; this change suggests that the soil biological system at this site is responding to increases in P availability. We found this for the unenriched area. At the enriched site, the model did not explain much of the variation observed in MBP, indicating that the site has reached a level of eutrophication where increased levels of P are no longer causing significant changes in this system. If the unenriched area is not currently being enriched, then we would expect almost a similar lack of response to these variables as for the enriched site. Because the coordinates are important predictors, however, this suggests directional P loading from the enriched site to the unenriched site.

Table 3. Historical and current mean (±1 standard error in parentheses) total P concentration values for the northeast (P-enriched) and northwest (unenriched) areas of the Blue Cypress Marsh Conservation Area.

Year sampled and study	Soil depth interval	Total P	
		Enriched	Unenriched
	cm	mg kg^{-1}	
1992			
Brenner and Schelske (1995)†	0–2	2340	445
	2–4	2265	430
	4–6	2150	370
	6–8	2075	310
	8–10	2110	280
1995			
Olila et al. (1995)‡	0–4	2092	628
	8–12	1865	455
1999			
D'Angelo et al. (1999)§	0–10	754 (116)	530 (209)
2000			
This study¶	0–10	847 (280)	643 (77)

† n = 2 cores, 0–10-cm avg. 2188 and 367 mg kg^{-1} for P-enriched and unenriched areas, respectively.

‡ n = 2 cores, 4–8-cm data missing.

§ n = 3 cores.

¶ n = 40 cores.

Several previous studies have documented soil TP concentrations (Table 3) in the P-enriched and unenriched areas of the BCMCA with time (Brenner and Schelske, 1995; Olila et al., 1995; D'Angelo et al., 1999). These earlier studies showed appreciably greater concentrations of P in the P-enriched area than the values obtained for this study. The first sampling occurred in 1992, shortly after the termination of the nutrient inflow, with a mean TP concentration of 2188 mg kg^{-1} in the P-enriched soil layer (0–10 cm), which was six times greater than those found in the unenriched area (367 mg kg^{-1}; Brenner and Schelske, 1995). Subsequent sampling events, in 1995 and 1998, recorded declining TP concentrations within the P-enriched area (Olila et al., 1995; D'Angelo et al., 1999). The data from this study show a significant reduction in TP concentrations to 847 ± 212 mg kg^{-1} within the P-enriched area. These studies indicate a 61% reduction in mean TP values in the 0- to 10-cm soil layer in 8 yr.

These lower concentrations have coincided with a noticeable increase in TP concentrations within the unenriched area. Brenner and Schelske (1995) documented mean TP concentrations for the 0- to 10-cm soil layer of 353 mg kg^{-1}. Those concentrations have shown an increase to 643 mg TP kg^{-1} in 8 yr, equating to an 82% increase in soil TP concentrations. The comparison of the TP data observed in this study with those collected in 1992, 1995, and 1998 indicate a general trend of increasing TP concentrations in the previously unenriched areas and a decrease in the impacted regions of the marsh (Table 3).

These historical comparisons depend on legacy data, which were collected, analyzed, and interpreted by others for reasons unrelated to this study, using a number of different sampling and analytical techniques. The effect of these different approaches is illustrated by the different sampling depth intervals, presented in Table 3, which makes a rigorous comparison difficult. There are, however, substantial further repercussions for the comparisons described above. It is not always clear, as was the case in this study, where the precise locations of the historical sampling sites were. We are confident that these were within 100 to 200 m of the current sampling scheme, but as we have shown, there can be considerable local spatial variability. A second problem is one of sample support. Support, in sampling and monitoring, is defined as the volume or area on which the measurements are made. For example, the averages reported for this study are that of an area covered by a grid; in the other studies, they can represent an average of a number cores sampled within 1 to 2 m of each other (e.g., three samples in D'Angelo et al., 1999), thereby complicating the comparison. Nonetheless, while these are common problems with using legacy

data, the overall magnitude of the change with time would seem to suggest that the P levels at these sites are changing.

Phosphorus mobilization can occur when conditions in the wetland favor aerobic conditions, facilitating the rapid decomposition of accumulated organic matter (DeBusk and Reddy, 1998). It has been demonstrated that wetland soils undergo fluctuations in hydrologic periods, alternating aerobic and aerobic conditions to experience a flux of previous bound P to the overlying water column after reintroduction of flooded conditions (White et al., 2004, 2006). Bostic and White (2007) have shown that readily available P concentrations in the water column can increase to >1 mg L^{-1} directly after flooding, with P release continuing for at least 30 d. Corstanje and Reddy (2004) conducted a similar hydrologic experiment in the BCMCA and found that soils from the P-enriched areas could release P at a concentration up to 10 times greater than unenriched cores. Clearly, the dry season can lead to significant P release, which can then be mobilized and transported into previously unenriched areas of the marsh following the return of surface water flows.

Hydrologic patterns in the BCMCA primarily reflect precipitation, observing the distinct wet and dry seasons typical of the Florida climate. The result of these two seasons is the periodic drawdown of surface waters and the dominance of aerobic conditions in the soil. The BCMCA experiences surface water sheet flow in a distinct southeast to northwest flow pattern (Fig. 1; Prenger and Reddy, 2004). Surface water flows from the enriched areas toward the unenriched before discharge into Blue Cypress Lake. These flow patterns, coupled with seasonal drawdown and reflooding events and driven by precipitation patterns, can drive P release from the enriched areas into the unenriched area. Therefore, it is reasonable to expect that the continual redistribution of P throughout the wetland, perpetuated by hydrologic fluctuations, would cause the nutrient gradient to vanish with time, leading to an eventual elevation of the overall background P concentrations to some higher-than-baseline concentration across the marsh. This mechanism of P redistribution reaches beyond this single freshwater marsh and suggests potential consequences for other wetlands that have historically received high P inflows, such as the Florida Everglades. The redistribution of P will lead to decreased soil P concentrations around the initial P-enriched area but will increase the overall baseline of soil P in the wetland. These findings bring into question the ability for a wetland to naturally recover to preenrichment conditions through the burial of P, in particular, for wetland systems with a well-defined wet and dry season, which can lead to the oxidation and release of organically bound P. To prevent the redistribution of soil P during restoration, one potential solution is to hydrologically isolate the P-enriched area by a series of berms until such time as the high-P peat is buried by a decade or more of new accreted organic matter or, where feasible, treat the soil with chemical amendments that will prevent P release (Malecki-Brown et al., 2007, 2009).

CONCLUSIONS

Historical runoff from agricultural fields has led to a wetland soil P gradient with decreasing concentration with increasing distance from the input in the BCMCA. Ten years after of the end of P loading, the total soil P was still elevated proximal to the former inflow point. A comparison of current total P concentrations with those determined in previous studies showed a 70% decline in TP concentrations within this P-enriched area. When comparing these to samples taken from formerly unenriched areas of the marsh, we find that the average P concentration has increased 82%. This study would seem to suggest that this decrease in the P-enriched area and the increase in the previously unenriched areas of the marsh may be linked. During dry periods, significant amounts of P are released from these organic soils (Bostic and White, 2007). Therefore, the decade since termination of the P inflow and the reestablishment of a naturally fluctuating hydrology has led to a decrease in P in the enriched areas and an increase in the previously unenriched area. The implication of these observations is that, for the near future, the overall marsh may stabilize at an elevated soil P level instead of following a swift trajectory to the previous unenriched state. It is clear that this can be only inferred from this field study, as it is based in part on legacy data and that further, controlled studies will be needed to establish the magnitude of this remobilization process.

If the high-P soil can be hydrologically isolated from the surface water, it would prevent the spread of soil P to the unimpacted portions of the wetland. This strategy might also be used in other systems, including the Florida Everglades, to prevent the excess P in the enriched portions of the wetland system from spreading to unenriched portions of the wetland, thereby increasing the likelihood of successful restoration.

ACKNOWLEDGMENTS

We appreciated the sampling assistance provided by Dr Joe Prenger and others and the analytical assistance provided by Ms. Yu Wang. This research was supported by the USEPA Grant no. R-827641-01 and with assistance in kind by the St. Johns River Water Management District.

REFERENCES

Akaike, H. 1973. Information theory and an extension of the maximum likelihood principle. p. 267–281. *In* B.N. Petrov and F. Csaki (ed.) Proc. Int. Symp. on Inf. Theory, 2nd, Tsahkadsor, Armenia. 2–8 Sept. 1971. Akadémiai Kiadó, Budapest.

Allen, S.E., H.M. Grimshaw, J.A. Parkinson, and C. Quarmly. 1974. Chemical analysis of ecological materials. Blackwell Sci. Publ., Oxford, UK.

Amador, J.A., G.H. Richany, and R.D. Jones. 1991. Factors affecting phosphate uptake by peat soils of the Florida Everglades. Soil Sci. 153:463–470.

Andersen, J.M. 1976. An ignition method for determination of total phosphorus in lake sediments. Water Res. 10:329–331.

Blake, G.R., and K.H. Hartge. 1986. Bulk density. p. 363–375. *In* A. Klute (ed.) Methods of soil analysis. Part 1. Agron. Monogr. 9. ASA and SSSA, Madison, WI.

Bostic, E.M., and J.R. White. 2007. Soil phosphorus and vegetation influence on wetland phosphorus release after simulated drought. Soil Sci. Soc. Am. J. 71:238–244.

Brenner, M., and C.L. Schelske. 1995. Historical Sedimentation and Nutrient storage rates in the Blue Cypress Marsh Conservation Area. Final Rep. for St. Johns River Water Manage. District. Spec. Publ. SJ95–SP3. Fisheries and Aquatic Sciences, Univ. of Florida, Gainesville.

Bridgham, S.D., and C.J. Richardson. 1993. Hydrology and nutrient gradients in North Carolina peatlands. Wetlands 13:207–218.

Brookes, P.C., D.S. Powlson, and D.S. Jenkinson. 1982. Measurement of microbial biomass phosphorus in soil. Soil Biol. Biochem. 14:319–329.

Brookes, P.C., D.S. Powlson, and D.S. Jenkinson. 1984. Phosphorus in the soil microbial biomass. Soil Biol. Biochem. 16:169–175.

Corstanje, R., and K.R. Reddy. 2004. Response of biogeochemical indicators to a drawdown and subsequent flood. J. Environ. Qual. 33:2357–2366.

Corstanje, R., K.R. Reddy, J.P. Prenger, S. Newman, and A.V. Ogram. 2007. Soil microbial eco-physiological response to nutrient enrichment. Ecol. Indic. 7:277–289.

D'Angelo, E.M., T.C. Oña, R. Corstanje, and K.R. Reddy. 1999. Selected biogeochemical properties of impacted and unimpacted marsh sites in the Upper St. Johns River basin. SJ98B368. St. Johns River Water Manage. District, Palatka, FL.

DeBusk, W.F., and K.R. Reddy. 1998. Turnover of detrital organic carbon in a nutrient-impacted Everglades marsh. Soil Sci. Soc. Am. J. 62:1460–1468.

Grace, K.A., F.E. Dierberg, T.A. DeBusk, and J.R. White. 2008. Phosphorus uptake by *Typha* leaf litter as affected by oxygen availability. Wetlands 28:817–826.

Grunwald, S., R. Corstanje, B.E. Weinrich, and K.R. Reddy. 2006. Spatial patterns of labile forms of phosphorus in a subtropical wetland. J. Environ. Qual. 35:378–389.

Hedley, M.J., and J.W.B. Stewart. 1982. Method to measure microbial biomass phosphorus in soils. Soil Biol. Biochem. 14:377–385.

Ipsilantis, I., and D.M. Sylvia. 2007. Abundance of fungi and bacteria in a nutrient-impacted Florida wetland. Appl. Soil Ecol. 35:272–280.

Lark, R.M. 2000. A comparison of some robust estimators of the variogram for use in soil survey. Eur. J. Soil Sci. 51:137–157.

Malecki, L.M., J.R. White, and K.R. Reddy. 2004. Nitrogen and phosphorus flux rates from sediment in the Lower St. Johns River estuary. J. Environ. Qual. 33:1545–1555.

Malecki-Brown, L.M., J.R. White, and K.R. Reddy. 2007. Soil biogeochemical characteristics influenced by alum application in a municipal wastewater treatment wetland. J. Environ. Qual. 36:1904–1913.

Malecki-Brown, L.M., J.R. White, and M. Sees. 2009. Alum application to improve water quality in a municipal wastewater treatment wetland. J. Environ. Qual. 38:814–821.

Marchant, B.P., and R.M. Lark. 2007. Robust estimation of the variogram by residual maximum likelihood. Geoderma 140:62–72.

Matheron, G. 1971. The theory of regionalized variables and its applications. Ctr. Géostat., Fontainebleau, France.

Matheron, G. 1973. The intrinsic random functions and their applications. Adv. Appl. Probab. 5:439–468.

Newman, S., K.R. Reddy, W.F. DeBusk, Y. Wang, G. Shih, and M.M. Fisher. 1997. Spatial distribution of soil nutrients in a northern Everglades marsh: Water Conservation Area 1. Soil Sci. Soc. Am. J. 61:1275–1283.

Noe, G.B., D.L. Childers, and R.D. Jones. 2001. Phosphorus biogeochemistry and the impact of phosphorus enrichment: Why is the Everglades so unique? Ecosystems 4:603–624.

Olila, O.G., K.R. Reddy, and L. Keenan. 1995. Labile and nonlabile pools of phosphorus in surface water and soils in the Upper St. Johns River basin. SJ94D179. St. Johns River Manage. District, Palatka, FL.

Prenger, J.P., and K.R. Reddy. 2004. Microbial enzyme activities in a freshwater marsh after cessation of nutrient loading. Soil Sci. Soc. Am. J. 68:1796–1804.

Reddy, K.R., R.D. DeLaune, W.F. DeBusk, and M.S. Koch. 1993. Long-term nutrient accumulation rates in the Everglades. Soil Sci. Soc. Am. J. 57:1147–1155.

Reddy, K.R., M.M. Fisher, Y. Wang, J.R. White, and R.T. James. 2007. Potential effects of sediment dredging on internal phosphorus loading in a shallow, subtropical lake. Lake Reservoir Manage. 23:27–38.

Reddy, K.R., Y. Wang, W.F. DeBusk, M.M. Fisher, and S. Newman. 1998. Forms of soil phosphorus in selected hydrologic units of the Florida Everglades. Soil Sci. Soc. Am. J. 62:1134–1147.

Reddy, K.R., J.R. White, A.L. Wright, and T. Chua. 1999. Influence of phosphorus loading on microbial processes in soil and water column of wetlands. p. 249–273. *In* K.R. Reddy et al. (ed.) Phosphorus biogeochemistry in subtropical ecosystems. CRC. Press, Boca Raton, FL.

Ribeiro, P.J., Jr., and P.J. Diggle. 2001. GeoR: A package for geostatistical analysis. R News 1/2:15–18.

Richardson, C.J. 1985. Mechanisms controlling phosphorus retention capacity in freshwater wetlands. Science 228:1424–1427.

USEPA. 1993. Methods of chemical analysis of water and wastes. 600/R-93/100. Environ. Monit. Support Lab., Cincinnati, OH.

Webster, R., and M.A. Oliver. 2001. Geostatistics for environmental scientists. John Wiley & Sons, New York.

Weyers, H.S., and K. Suberkropp. 1996. Fungal and bacterial production during the breakdown of yellow poplar leaves in 2 streams. J. North Am. Benthol. Soc. 15:408–420.

White, J.R., and K.R. Reddy. 2000. Influence of phosphorus loading on organic nitrogen mineralization of Everglades soils. Soil Sci. Soc. Am. J. 64:1525–1534.

White, J.R., K.R. Reddy, and M.Z. Moustafa. 2004. Influence of hydrology and vegetation on phosphorus retention in Everglades stormwater treatment wetlands. Hydrol. Processes 18:343–355.

White, J.R., K.R. Reddy, and J.M. Newman. 2006. Hydrology and vegetation effects on water quality in subtropical constructed wetlands. Soil Sci. Soc. Am. J. 70:1242–1251.

Wu, Y., F.S. Sklar, and K.R. Rutchey. 1997. Analysis and simulations of fragmentation patterns in the Everglades. Ecol. Appl. 7:268–276.