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ECOSYSTEMS

Re-flooding a Historically Drained Wetland Leads to Rapid Sediment Phosphorus Release

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ABSTRACT

Wetland restoration provides many benefits, but re-flooding historically drained land can have unintended negative consequences, including phosphorus (P) release from sediments. To investigate the effects of re-flooding on P cycling, this study monitored a restoration in Michigan that back-flooded old drainage ditches and re-flooded former wetland soils. Immediately after re-flooding, previously exposed sediments released substantial amounts of P to the water column. Soluble reactive phosphorus (SRP) concentrations in re-flooded areas were as high as 750 μg P l^{-1}. At peak P concentrations, there were about 20 times more SRP and 14 times more total P in the surface water than in the much smaller flooded area that existed before re-flooding. Prolific growth of filamentous algae and duckweed was observed in subsequent summers. Sedimental analyses suggest that most of the P released originated from iron-bound fractions. The highest SRP concentrations occurred during the first year when surface water dissolved oxygen was low (<5.5 mg l^{-1}). Similarly low oxygen in the second year after flooding was not associated with such high P concentrations. After 1 year postflooding, SRP concentrations remained below 50 μg P l^{-1} (but still high enough to produce eutrophic conditions) until the end of sampling about 15 months after re-flooding. When re-flooding historically drained soils, managers should consider the potential for sediment P release to jeopardize restoration goals and therefore should incorporate longer term monitoring of water quality into restoration plans. Knowledge of sediment P amounts and forms can indicate the potential for P release to overlying water.

Key words: wetland; phosphorus; internal eutrophication; iron; sediment; restoration.

INTRODUCTION

Wetlands were systematically drained and ''reclaimed'' for agricultural use throughout the United States, particularly in the late nineteenth and early to mid twentieth centuries (Dahl 1990). In recent decades, the valuable services provided by wetlands have increasingly been recognized, and managers have begun restoring wetland hydrology to historically drained areas (Zedler 2003). Wetland

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restoration provides a number of benefits, but re-flooding historically drained and cultivated land can have unintended negative consequences. Often, drained wetland sediments release nutrients upon re-flooding, as has been shown both in lab-scale reflooding experiments (Olila and others 1997; Young and Ross 2001; Lucassen and others 2005; Aldous and others 2007; Loeb and others 2008; Schönbrunner and others 2012) and in monitoring whole ecosystem re-flooding (Newman and Pietro 2001; Coveney and others 2002; Duff and others 2009; Niedermeier and Robinson 2009; Ardón and others 2010a; Wong and others 2011). Nutrient release upon re-flooding can inhibit the restored wetland's ability to support biodiversity and to improve water quality, which are two common goals of wetland restoration (Hansson and others 2005; Verhoeven and others 2006).

Although studies show that sediments often initially release phosphorus (P) when re-flooded after long-term drainage, there is some contradicting evidence that drying improves P-retention capacity upon re-flooding (Mitchell and Baldwin 1998; Baldwin and others 2000; Smolders and others 2006). Sediment biogeochemistry determines P flux rates between sediments and surface waters (that is, standing water above the sediment surface). In sediments, P exists in many forms including covalently bound in organic matter, sorbed as the inorganic phosphate (PO_4^{3-}) ion to inorganic metal oxides (particularly poorly crystalline iron and aluminum oxides), and coprecipitated with carbonates (Boström and others 1988; Reddy and DeLaune 2008). Thus, multiple processes potentially control the flux of P between sediments and surface waters including the balance between biotic uptake-microbial mineralization, sorption–desorption, and mineral coprecipitation–dissolution processes. High rates of P release upon re-flooding are often attributed to the legacy of fertilizer application, but P release has been shown to occur even in dried and re-flooded sediments without a history of high P loading (Aldous and others 2005; MacDonald and others 2012).

The persistence of sediment P release after initial re-flooding and the ultimate fate of the released P are difficult to predict. Released P can either be exported to downstream ecosystems or remain in the wetland, where it may cycle internally and eventually become buried in sediments or assimilated into biomass. Few studies have monitored re-flooded agricultural soils both during the initial flooding event and for significant periods of time after re-flooding, but studies of wetlands that were re-flooded in the past have shown that sediments can continue to release P for at least 5–10 years after re-flooding (Montgomery and Eames 2008; Duff and others 2009; Steinman and Ogdahl 2011). Theoretically, P release rates should decline over time as sediment P pools are depleted and the sediment comes into equilibrium with its overlying water, assuming the absence of new P loading, but in some cases, this can take a very long time (10–100 years, Hamilton 2012).

Despite the potential for sediment nutrient release to limit the success of (sometimes costly) wetland restoration and management, few studies have comprehensively studied ecosystem response to re-flooding in actual ecosystems both before and after the re-flooding event. To investigate the immediate and longer-term (2 years) effects of re-flooding a historically drained wetland on P cycling, we monitored a wetland restoration project that entailed construction of a control structure to back-flood old drainage ditches and the surrounding wetland soils. We sampled surface waters and sediments to investigate biogeochemical changes during the summer before re-flooding and for over 2 years (25 months) after the re-flooding event.

METHODS

Site Description and Geomorphology

We studied a fen wetland located in the Fort Custer Training Center (FCTC), a military training area in southwestern Michigan, USA. Standing water in the wetland covers 9 ha when fully flooded to the sill level of the dam at the outflow (Figure 1). The historically drained Fort Custer wetland was re-flooded in September 2008 to improve biodiversity, provide opportunities for research and education, and provide floodwater protection (Langerveld 2009). Ditches draining the wetland appear in aerial photographs from the late 1930s, indicating drainage for at least 75 years, and perhaps as many as 100 years (Langerveld 2009). During this time, the wetland flooded periodically for shorter periods of time because of beaver activity, most recently in the late 1990s, when the full extent of the wetland was flooded (Langerveld 2009). For about a decade before re-flooding, only the drainage ditches and a portion of the outflow pond were inundated. Curlytop knotweed (*Polygonum lapathifolium*), stinging nettle (*Urtica dioica*), and reed canarygrass (*Phalaris arundinaceae*) were the dominant plants in unflooded wetland areas, and duckweed and filamentous algae were present in stagnant drainage ditches. The sediment surface in most areas was exposed to the atmosphere (that

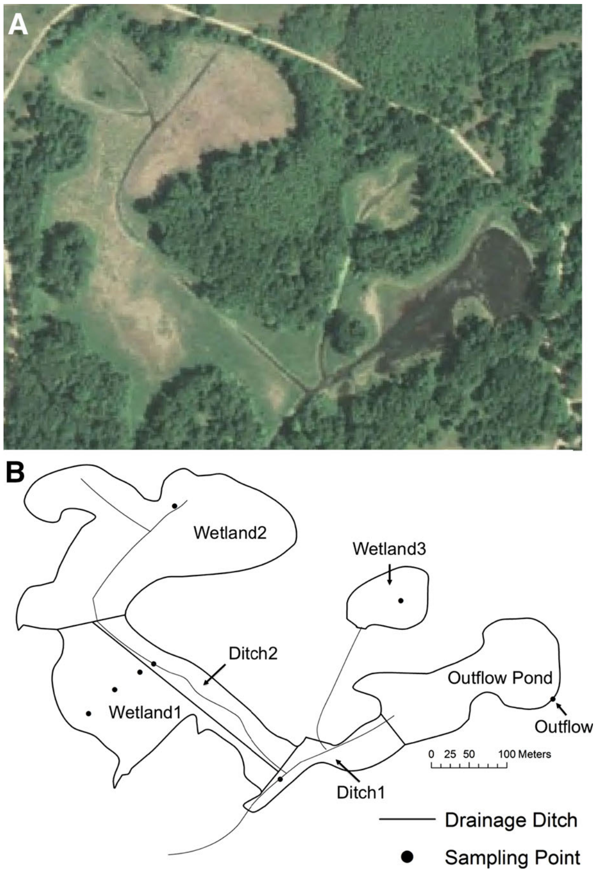

Figure 1. **A** Aerial image of the Fort Custer Training Center restored wetland taken before re-flooding (July 31, 2006, USDA-FSA Aerial Photography Field Office) with drainage ditches visible. Drainage ditches appear in aerial photographs dating as far back as the 1930s. **B** Outline of Fort Custer Wetland divided into sampling zones, with points showing locations of repeated sampling for water chemistry.

is, was not under standing water), but the water table was close enough to the surface to keep the soil moist, and to support wetland vegetation and prevent shrub and tree encroachment. Soils in the wetland are greater than 90% Houghton muck, a euic, mesic Typic Haplosaprist (Soil Survey Staff 2012). The months before the re-flooding event were relatively hot and dry, and so even some areas in drainage ditches that were usually inundated were exposed by late summer 2008.

A partnership among FCTC, Ducks Unlimited, the U.S. Fish & Wildlife Service, and other organizations restored this wetland area by back-flooding drainage ditches. In July 2008, a water control structure (Agri-Drain, Adair, IA) and emergency spillway were installed at the outflow (Figure 1), establishing the maximum water level at 266.4 m above sea level. Water levels remained close to their original level (≈265.4 m) until September 2008, when an extreme precipitation event (239-mm rain in 3 days: lter.kbs.msu.edu) associated with the remnants of Hurricane Ike flooded the wetland to its full capacity. After this re-flooding event, the wetland remained at or near its full flooded capacity throughout our sampling period.

The wetland drains a relatively small area (≈100 ha), most of which is within the FCTC. Even when fully flooded, no single, channelized inflow feeds the wetland. Drainage Ditch 1 (Figure 1) drains a larger area before entering the study wetland, but was rarely, if ever, flowing measurably. Based on dissolved magnesium (Mg^{2+}) concentrations in surface waters, which are a reliable indicator of groundwater inputs in this landscape (Whitmire and Hamilton 2005), Ditch 1 is evidently fed partially by groundwater in its upstream reach, but short-circuits most of the wetland as it flows into the Outflow Pond. Groundwater represented more than half of hydrologic inputs to Ditch 1 and the Outflow Pond (with the remainder attributable to direct capture and overland flow of precipitation), whereas the newly re-flooded wetland areas contained much lower Mg^{2+} concentrations, indicating a lower influence of groundwater and a low rate of mixing between the inflowing Ditch 1 water and the newly re-flooded areas. During the study period, one storm water drain directed the runoff from an unpaved parking area into the wetland.

To understand the relationships between the wetted area and the volume, we measured depths at marked waypoints with high spatial resolution on August 10 and 12, 2009 and October 20, 2010. We used measurements of depth together with wetted area from aerial photos to estimate contours at 0.2-m intervals, from which we estimated the volume of the fully flooded wetland using the truncated cone method (Wetzel and Likens 2000). Subsequently, we divided the wetland into zones corresponding to water sampling points (Figure 1) and estimated the volume within each zone using the same method.

Water Chemistry

We monitored surface and pore water chemistry during the summer before and for over 2 years after re-flooding at the inflow ditch (Ditch 1), the main drainage ditch (Ditch 2), three areas that were initially dry, but later flooded (Wetlands 1–3), and at the wetland's outflow (Figure 1). Before the re-flooding event, we sampled surface waters in drainage ditches and the outflow at least twice each, and also obtained one sample from Wetland 3, in which there was a small amount of water (<20 cm deep and a very small proportion of the zone's area).

After re-flooding in September 2008, we monitored water chemistry approximately monthly until January 2011. One of the zones, Wetland 1, contained three repeatedly sampled points along a transect running perpendicular to Ditch 2 (Figure 1). At each point for each sampling event, we measured surface-water temperature, dissolved oxygen (DO), pH, and specific conductivity (corrected to 25°C) using a Hydrolab multisonde, collected surface-water samples, and sampled sediment pore water from a known depth range (7–12 cm) using a plastic syringe and tubing connected to a screened filter at the end of a drive-point sampler. We filtered the pore water samples through a 0.45-μm cellulose–acetate filter with a glass fiber prefilter (Steriltech), taking care to minimize oxygenation, and added reagents in the field for analysis of dissolved (filterable) reduced iron (Fe(II)) and free hydrogen sulfide (H_2S). We measured discharge at the outflow using a Marsh-McBirney flow meter, recorded water depth at a standard location, and sampled surface water at the outflow for soluble reactive phosphorus (SRP), total dissolved P (TDP), and total P (TP).

For analyses of dissolved ions and nutrients, surface waters were filtered (0.45 μm polyethersulfone filter, Pall Supor). We measured SRP concentrations in surface and pore water using the molybdate blue colorimetric method (Murphy and Riley 1962), nitrate-N (NO_3^--N), sulfate (SO_4^{2-}), calcium (Ca^{2+}), and Mg^{2+} ions using membrane-suppression ion chromatography, and total ammonium nitrogen (NH_4^+-N) using the indophenol blue method (Grasshoff and others 1983; Aminot and others 1997). We measured TDP and TP after persulfate digestion of filtered and unfiltered samples, respectively, followed by colorimetric SRP analysis. We estimated the concentration of dissolved organic P (DOP) in a water sample by subtracting its SRP from TDP. Dissolved organic carbon (DOC) was measured in filtered samples using a Shimadzu TOC 5000 analyzer. We measured Fe(II) using reaction with ferrozine based on a colorimetric method modified from Lovley and Phillips (1987) and Stookey (1970), in which the sample was added to a solution of 50 mM HEPES buffer containing ferrozine (1 g l^{-1}). We measured H_2S using the methylene blue spectrophotometric method (Golterman and Clymo 1969).

Sediment Chemistry

We collected sediments at least once each year for biogeochemical analysis. To investigate sediment P-binding fractions, we sampled sediments immediately before (September 3, 2008) and immediately after (September 17, 2008) the flooding event. On these dates, sediment samples were taken throughout the wetland (Ditches 1–2 and Wetlands 1–3) for sequential P extraction and analysis of organic matter by loss on ignition, total P (sedTP), and total iron (TFe). To measure a broader range of characteristics, in 2009 and 2010, sediments were collected from a subset of sites (Wetland 1 and Wetland 3) and analyzed for organic matter, sedTP and TFe, oxalate-extractable iron (Ox-Fe, an indicator of poorly crystalline iron oxides), total solid carbonates ($CaCO_3$), total aluminum (TAl), and acid volatile sulfide (AVS, an approximate indicator of iron bound to sulfide, and thus unavailable to sorb PO_4^{3-}).

Triplicate sub-subsamples (~1 g d.w.) of dried and ground sediment were combusted for analysis of organic matter as loss on ignition (550°C). We extracted combusted samples for 10 min in boiling 1 M hydrochloric acid (HCl) for measurement of sedTP, TFe, and TAl (Andersen 1976). To measure Ox-Fe, we extracted approximately 0.4 g moist sediment in a 0.2 M acid ammonium oxalate solution for 4 h in darkness (Walbridge and others 1991). We measured $CaCO_3$ in triplicate sub-samples of dried and ground sediment (0.1–1 g d.w.) by acidifying in a sealed container and measuring carbon dioxide produced using a pressure transducer. We measured iron in Ox-Fe and TFe extracts using flame atomic absorption spectrophotometry.

We froze approximately 100 g of moist sediment for analysis of AVS following US EPA Method 821-R-91-100, acidifying sediment samples with HCl to convert AVS to hydrogen sulfide, which was then trapped as S^{2-} in an alkaline solution (0.5 M NaOH) and measured colorimetrically (Allen and others 1991). Analytic sulfide standards were prepared from a stock solution standardized versus thiosulfate.

We used a sequential extraction procedure (Paludan and Jensen 1995) to measure operationally defined P-binding fractions in sediments collected immediately before and immediately after the flooding. We collected sediments at sampling points in Ditch 2, Wetland 1 (3 samples, a–c), and Wetland 2 (2 samples, a–b). Sediments from Ditch 2 were separated into two samples—one from the overlying loose sediment (''floc'' layer) and one representing the top 10 cm of the more consolidated sediment below. All other sediment samples were taken from the top 10 (±3) cm of sediment and homogenized in a glove bag with an anoxic atmosphere before sequential P extraction.

We sequentially extracted P-binding fractions (Paludan and Jensen 1995) in triplicate samples of wet sediment (0.5–5 g d.w.). The first step used deoxygenated deionized water to extract loosely bound P (H_2O–P). Next, bicarbonate-buffered dithionite (0.11 M) extracted phosphate ($PO_4{}^{3-}$) bound to redox-sensitive oxidized iron minerals (BD–P) and some nonreactive (mostly organic) P (BD–DNRP) (Reitzel and others 2006). The third step used sodium hydroxide (0.1 M NaOH) to extract $PO_4{}^{3-}$ bound to redox-insensitive aluminum and iron oxides that undergo anionic exchange with hydroxide (NaOH–SRP) and nonreactive organic and inorganic P (pyro- and polyphosphates). Nonreactive P extracted by NaOH was acidified to separate out precipitating humic-acid-associated P (HA–P) from other nonreactive P molecules (NaOH–DNRP). The remaining P-binding fractions, carbonate-associated P and residual P, were estimated by subtracting the sum of all other fractions from the independently measured sedTP (HCl + Res–P). We expect carbonate-associated P to be of minimal importance at this site due to low total calcium carbonate concentrations in the sediments (<1 mg $CaCO_3$ g^{-1} d.w.). We did not detect any nonreactive H_2O–P, and so the results are not reported for that fraction.

We did not measure sedTP in the Ditch 2 post-flood floc layer sample because of an insufficient amount of material, and so the sedTP value from flocculent Ditch 2 sediment before flooding was used to estimate HCl + Res–P. Two samples taken approximately 1 m apart were collected at Wetland 2a in the preflood sampling, and P fraction data from these two were averaged to produce a single value before statistical analysis.

Intact Core Experiment

To investigate how oxygen in overlying water may control sediment P release in the absence of photosynthesis, we collected intact sediment cores and measured rates of P release to surface waters in unaerated surface-water treatments and in continuously aerated controls. We collected 12 intact cores (diameter = 4.5 cm, sediment depth = 15 cm, and water column depth = 15 cm) from each of two locations in June (Wetland 1) and July (Wetland 2) 2009. We gently aerated six cores from each site with an aquarium bubbler to maintain 90–100% equilibrium with atmospheric oxygen (''High DO'') and left the remaining six from each site unaerated (''Low DO'': $<25\%$ saturation, mean 9%). The surface water in unaerated cores was carefully mixed before sampling to produce a uniform water column, remove concentration gradients that may have formed in between mixing events, and to mimic mixing created by bubblers in the aerated treatments. Approximately twice a week for 50 days, we sampled surface water in each core for SRP. The total volume of water sampled was replaced with low-P water (~2 µg P l^{-1}) sampled from the outflow pond. Cores were stored in the dark at room temperature, and little-to-no algal growth was observed during the experiment, although nonphotosynthetic microbial growth and P uptake would have continued. We used surface-water SRP concentrations to estimate sediment P release rates (mg P m^{-2} day^{-1}) using the first 14 and 21 days of sampling from Wetland 1 and Wetland 3 cores, respectively.

Calculations and Statistics

To estimate the amount of P exported from the wetland, we interpolated between export rates for sampling dates (g P day^{-1}). Although we do not have discharge measurements between September 3, 2008 (immediately before re-flooding) and February 25, 2009 (163 days after re-flooding), discharge and water levels during this time period in nearby Fair Lake and Crooked Lake (S. Hamilton, unpublished data) and Augusta Creek (water.usgs.gov) support interpolation of the time series. The gradual rise and decline in water levels are attributed to the importance of groundwater flow paths in this glacial landscape, where soils are generally very permeable.

To estimate the amounts (by mass) of SRP, DOP, and TP in the wetland surface water, we scaled up point measurements of water chemistry (in mass/volume) to the entire zone within which each sampling point was located by multiplying the measurement for a given day by the estimated zone volume. To estimate the bulk mass of P in sediment P-binding fractions that could be exchanged with surface waters, we used an estimated organic soil bulk density of 0.5 g cm^{-3} and made the assumption that the top 5 cm of sediment was most likely to exchange solutes with surface waters.

All statistical analyses were carried out in R version 2.13.2 (R Development Core Team 2011). Unless stated otherwise, variables were natural-log transformed before analysis to more closely meet linear model assumptions of normal distributions and equal variance. To examine the effects of re-flooding on sediment P-binding fractions, we conducted pairwise comparisons between each fraction that we measured in sediment samples ($N = 7$) collected just before and shortly after flooding ($\alpha = 0.05$). To test if P release rates and maximum P release were significantly different between aerated and nonaerated treatments in the sediment core experiment, we ran linear models on untransformed data with treatment as a fixed factor for each site ($\alpha = 0.05$).

RESULTS

Geomorphology and Flooding

The re-flooding in September 2008 rapidly and drastically changed conditions in the Wetland zones, which make up the majority of the ecosystem's area and had been exposed soils dominated by grasses and forbs. After the wetland was fully flooded, most of its area consisted of newly re-flooded sediments and was shallow, with an area-weighted mean depth of 46 cm (Table 1).

Discharge from the outflow was the highest each year in mid-winter and early spring, declining to zero by late summer/early fall (Figure 2). The highest recorded discharge rate (73 l s^{-1}) was in late February 2009 after re-flooding. The wetland was flushed at the highest rate at this time, with a residence time for the entire wetland of 4.7 days, although much water probably moved preferentially through Ditch 1 and the Outflow Pond, in which case the residence time would have been only 2.7 days.

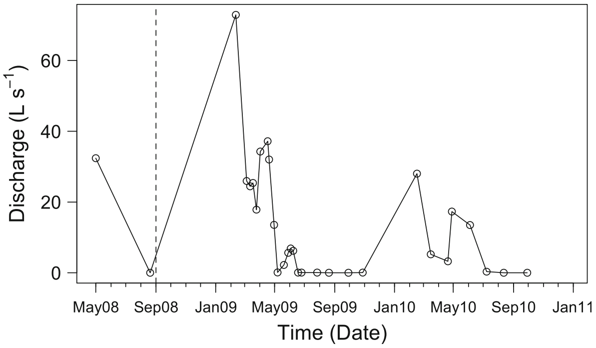

Figure 2. Discharge measured at the outflow of the Fort Custer Wetland restoration site over time, before and after the wetland was re-flooded for restoration (re-flooding timing shown by the *vertical dashed line*).

Using discharge measurements and wetland volume to estimate residence times produced wide-ranging estimates because of highly variable discharge rates measured at the outflow. In late winter and spring months (February through mid-May), residence times ranged from 4.6 to 104 days for the entire wetland's volume and from 2.7 to 61 days for the more rapidly flushed portion (Ditch 1 + the Outflow Pond). The shortest residence times occurred in mid-winter and early spring, and the longest residence times occurred in summer and early fall, when little or no water was flowing out of the wetland.

Sediment Phosphorus Release

Surface-water P concentrations were high throughout the re-flooded wetland after the rapid re-flooding (Figure 3). In the newly re-flooded areas, SRP concentrations in the days after re-flooding were as high as 750 µg P l^{-1} (at one of the three sampling locations within the Wetland 1 zone). Concentrations of SRP and TP remained high during winter of 2008–2009 (Wetland Zones mean SRP ± SE = 323 ± 118 µg P l^{-1}). In April 2009, SRP concentrations throughout the wetland declined rapidly to less than 20 µg P l^{-1}. These

Table 1. Area, Volume, Area-weighted Mean Depth, and Percent Groundwater Influence Measured Within Each Zone of the Fort Custer Wetland

Zone	Area (m^2)	Volume (m^3)	Mean depth (cm)	Groundwater influence (%)
Ditch 1	5,603	3,597	65	64
Ditch 2	6,580	3,394	52	43
Outflow pond	16,569	13,623	83	59
Wetland 1	19,471	3,101	25	35
Wetland 2	34,277	3,364	36	29
Wetland 3	4,940	2,000	41	45
Entire wetland	87,440	29,079	46[a]	51[b]

Percent groundwater influence was calculated from dissolved magnesium (Mg^{2+}) concentrations (Whitmire and Hamilton 2005) using a mixing model assuming groundwater and precipitation represent the only inputs of dissolved Mg^{2+} to the wetland waters, and assuming Mg^{2+} concentrations of 0.04 mg l^{-1} for precipitation (1979–2002 mean from NADP/NTN 2003) and 25 mg l^{-1} for groundwater (Kalamazoo County mean). Evaporative concentration was not accounted for.

[a] *Area-weighted average.*

[b] *Volume-weighted average.*

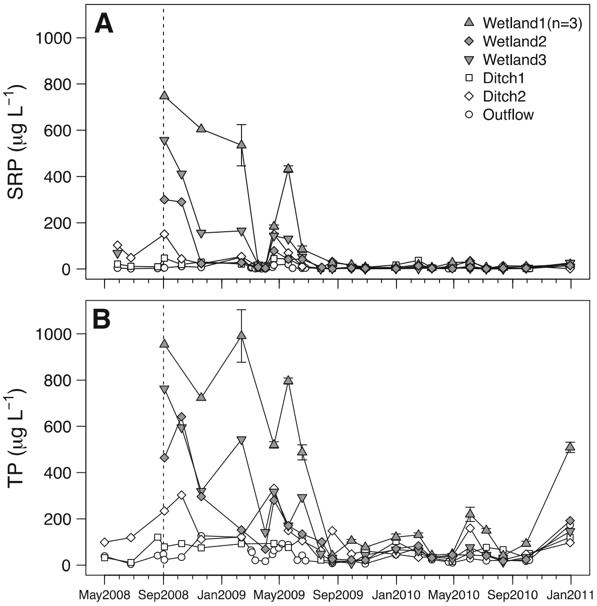

Figure 3. Soluble reactive phosphate (SRP) and total phosphorus (TP) measured in surface water through time in the re-flooded Fort Custer Wetland. *Dashed vertical line* represents heavy rain events (September 15–17, 2008) that re-flooded historically drained areas at locations labeled Wetland 1–3. Before September 15, 2008, the soil surface of Wetlands 1–3 was exposed to air. After re-flooding, these areas became covered by water 0.1–1 m deep (area-weighted average depth = 0.46 m) that persisted throughout the rest of the study period. Values for Wetland 1 represent the averages (with standard error bars) of three sampling locations along a transect within this zone. See map (Figure 1) for water sampling locations.

lower SRP concentrations coincided with a large bloom of filamentous algae. The highest SRP values (>45 μg P l^{-1}) were only observed during the first year following re-flooding, and only when surface-water DO was <5.5 mg l^{-1} (Figure 4). Similarly low DO concentrations in the second year were not associated with such high P concentrations, although filamentous algae and duckweed blooms were still prevalent. Surface-water TP concentrations remained high but variable through September 2009 (range, 9–1,537 μg P l^{-1}), after which concentrations were lower, but still mostly in the range that would be considered eutrophic for surface waters (range: 7–577 μg P l^{-1}; mean: 77 μg P l^{-1}, Figure 3).

The high P concentrations reflect large postflooding increases in surface water P mass. Immediately after re-flooding, the wetland's surface waters contained about 20 times more SRP and 14 times more TP than before re-flooding. Before re-flooding (summer 2008), standing water in inundated areas contained approximately 233, 204, and 503 g SRP, DOP, and TP, respectively. Immediately after flooding (September 17, 2008), the surface waters contained about 5,200 g of SRP and 7,400 g TP, and undetectable (within measurement and calculation error) DOP.

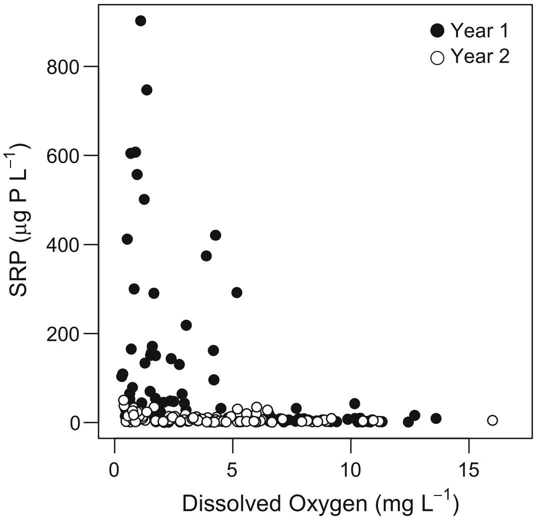

Figure 4. Soluble reactive phosphorus (SRP) and DO measured in surface waters of a re-flooded historically drained wetland in the first year following re-flooding (September 2008–2009) and the second year following re-flooding (September 2009–January 2011).

The newly flooded sediments represent the most probable source of the P increase. The amount of SRP in the inflowing drainage ditch (Ditch 1) only increased four fold (compared to the 20-fold increase in other zones), indicating that P entering the wetland from overland flow, precipitation, and the drainage ditch inflow can only explain a fraction of the observed P increase at the wetland's center (Wetland 1). Assuming the entire increase in P concentration immediately after flooding was due to sediment release, sediments had released on average 382 g SRP and 534 g TP day^{-1}, or 6.5 mg SRP m^{-2} day^{-1} and 9.1 mg TP m^{-2} day^{-1}, in response to flooding (sum of Wetland 1–3 Areas, September 3–17, 2008).

Phosphorus Export

Phosphorus export via the outflow stream was the highest in the months after initial flooding because of higher discharge rates and P concentrations during this time. We estimate that during our study period after re-flooding, the Fort Custer Wetland exported a total of 110 kg P, 40% of which (≈44 kg P) was as SRP. The majority (72%) of P export during our study occurred in the first

5 months following re-flooding (September 3, 2008–February 25, 2009).

Nitrogen and Dissolved Organic Carbon

Nitrogen in surface waters exhibited a starkly different pattern than P. Nitrate concentrations ranged from 0 to 0.3 mg NO_3^--N l^{-1} throughout the wetland during the study period, except in one instance in January 2010 when higher (1.5 mg NO_3^--N l^{-1}) concentrations were observed in Wetland 1. Ammonium concentrations were less than 200 µg NH_4^+-N l^{-1} throughout the first year following re-flooding, except for a few episodically higher (200–400 µg NH_4^+-N l^{-1}) concentrations measured immediately after flooding and in April 2009. During the second year after re-flooding, however, higher ammonium concentrations ($>$200 µg NH_4^+-N l^{-1}) were observed in spring and fall 2010, and in winter 2010–2011. Taken together as molar N:P ratios ((NO_3^- + NH_4^+)/TDP), the nutrient patterns we observed reveal that during our study period, average N:P ratios were spatially variable (Tables 2, 3), but N:P ratios were lower than the Redfield ratio (16:1) in most sampling sites during most of our study, except in a few zones immediately following re-flooding (Outflow, Ditch 1), throughout the wetland during winter 2010, and in a few zones during winter 2011 (Kinsman-Costello 2012).

Dissolved organic carbon (DOC) concentrations in surface waters were more variable among sites before re-flooding and during the first year after re-flooding than in the second year (Kinsman-Costello 2012). During the first year, DOC concentrations ranged from 8.7 to 77 mg l^{-1} with a mean of 33 mg l^{-1}. In the second year, the range was smaller, from 11 to 37 mg l^{-1} with a mean of 18 mg l^{-1}. Concentrations of DOC at all sampling locations peaked in early spring of 2009, and declined through the following fall and winter.

Pore Water Chemistry

In general, dissolved P and N in sediment pore waters did not display seasonal patterns. Average pore water SRP and NH_4^+-N concentrations after re-flooding—630 µg P l^{-1} and 3,900 µg NH_4^+-N l^{-1}, respectively—were higher than surface-water concentrations ($P < 0.0001$), suggesting a strong diffusion gradient from the sediment pore water environment to surface waters. Wetland 1, the zone in which the highest surface-water SRP concentrations were observed, contained the highest pore water SRP concentrations

Table 2. Surface-Water Chemistry at the Fort Custer Wetland Restoration Site

	SRP (µg l^{-1})	DOP (µg l^{-1})	TP (µg l^{-1})	NH_4^+ (µg N l^{-1})	NO_3^- (µg N l^{-1})	N:P[a]	SO_4^{2-} (mg l^{-1})	Mg^{2+} (mg l^{-1})	DOC (mg l^{-1})	Sp. Cond. (µS cm^{-1})
Ditch 1	15 ± 3 (24)	13 ± 2 (18)	61 ± 7 (22)	64 ± 24 (24)	67 ± 42 (21)	6.4	2.6 ± 0.8 (21)	25 ± 3 (16)	22 ± 2 (19)	441 ± 13 (24)
Ditch 2	29 ± 9 (23)	16 ± 4 (17)	110 ± 20 (20)	43 ± 13 (23)	33 ± 15 (22)	5.7	3.9 ± 2.3 (22)	20 ± 2 (16)	25 ± 3 (17)	325 ± 18 (23)
Wetland 1	108 ± 28 (55)	33 ± 6 (43)	299 ± 47 (49)	158 ± 34 (55)	147 ± 71 (54)	2.4	1.2 ± 0.5 (54)	10 ± 1 (39)	35 ± 3 (42)	280 ± 8 (57)
Wetland 2	40 ± 18 (22)	32 ± 8 (17)	139 ± 35 (21)	50 ± 23 (22)	21 ± 71 (20)	1.7	1.3 ± 0.4 (20)	13 ± 1 (15)	27 ± 2 (17)	240 ± 20 (22)
Wetland 3	78 ± 31 (22)	19 ± 4 (17)	179 ± 47 (21)	35 ± 10 (22)	23 ± 10 (21)	2.3	4.3 ± 0.4 (21)	14 ± 1 (16)	21 ± 2 (16)	305 ± 13 (22)
Outflow pond	8 ± 2 (48)	15 ± 1 (30)	49 ± 6 (45)	42 ± 13 (48)	15 ± 6 (45)	5.4	2.0 ± 0.4 (45)	15 ± 1 (30)	22 ± 1 (37)	375 ± 7 (56)

Values are means ± standard errors (n) of approximately monthly measurements made after re-flooding (means of all measurements taken between September 17, 2008 and January 13, 2010). SRP = soluble reactive phosphorus; DOP = dissolved organic P; TP = total P; DOC = dissolved organic carbon; Sp. Cond. = specific conductance (25°C).

[a] *N:P calculated as the molar ratio of within-site averages of dissolved inorganic N (NO_3^--N + NH_4^+-N) to total dissolved P.*

Table 3. Sediment Pore Water Chemistry at Sampling Sites in the Fort Custer Wetland Restoration Site

	SRP ($\mu g\ l^{-1}$)	DOP ($\mu g\ l^{-1}$)	NH_4^+ ($\mu g\ N\ l^{-1}$)	NO_3^- ($\mu g\ N\ l^{-1}$)	N:P[a]	SO_4^{2-} ($mg\ l^{-1}$)	Mg^{2+} ($mg\ l^{-1}$)	Fe(II) ($mg\ l^{-1}$)	H_2S ($\mu g\ S\ l^{-1}$)
Ditch 1	168 ± 42 (12)	452 ± 332 (6)	2304 ± 508 (10)	23 ± 7 (10)	7.9	2.0 ± 0.5 (10)	12 ± 1 (7)	26.1 ± 5.1 (10)	30 ± 22 (8)
Ditch 2	85 ± 62 (16)	290 ± 167 (10)	4297 ± 702 (14)	87 ± 74 (15)	24	4.1 ± 0.4 (15)	14 ± 0 (11)	42.8 ± 8.2 (13)	30 ± 10 (11)
Wetland 1	1233 ± 223 (53)	153 ± 85 (36)	4860 ± 484 (53)	27 ± 7 (48)	8.6	1.6 ± 0.4 (48)	9 ± 0 (36)	36.1 ± 3.6 (39)	33 ± 62 (37)
Wetland 2	151 ± 30 (18)	43 ± 17 (10)	3,874 ± 673 (18)	15 ± 5 (15)	54	0.9 ± 0.4 (15)	7 ± 1 (11)	11.7 ± 2.9 (14)	44 ± 33 (12)
Wetland 3	144 ± 37 (18)	41 ± 14 (10)	2,263 ± 384 (18)	16 ± 6 (15)	26	1.0 ± 0.3 (15)	12 ± 1 (11)	7.1 ± 0.9 (15)	43 ± 42 (12)

Values are means ± standard errors (n) of approximately monthly measurements after the re-flooding event (average of all measurements taken between September 17, 2008 and January 13, 2010). SRP = soluble reactive phosphorus; DOP = dissolved organic P.

[a] *N:P calculated as the molar ratio of within-site averages of dissolved inorganic N (NO_3^- + NH_4^+) to total dissolved P.*

(Tables 2, 3). Pore water SRP concentrations were slightly elevated compared to pore waters of other wetlands in the region sampled in late summer, although on average not significantly different ($P = 0.059$, mean ± one standard error = 366 ± 114 $\mu g\ P\ l^{-1}$, $N = 45$ sites: Kinsman-Costello 2012). Pore water dissolved Fe(II) concentrations were very high compared to regional wetlands ($P < 0.001$), with Fort Custer Wetland pore waters averaging 27 mg Fe l^{-1} and the southwest Michigan wetland pore waters containing on average 5.8 mg Fe l^{-1} ($N = 50$ sites: Kinsman-Costello 2012).

Sediment Characteristics

Compared with other sediments from the region (Kinsman-Costello 2012), the wetland sediments were high in TP (area-weighted average = 1,229 $\mu g\ P\ g^{-1}$) and TFe (area-weighted average = 11.56 mg Fe g^{-1}), but with moderate Fe:P molar ratios (area-weighted average = 6.78). Sediments were also highly organic, in the range of 10–80% organic matter, with an average of 60% (Table 4). On the other hand, sediments contained relatively low amounts of $CaCO_3$ and AVS, constituents typical of sediments in water bodies of the region with high groundwater influence (Table 4).

The relative distributions of P-binding fractions reflect the high organic matter and iron content of these sediments (Figure 5). Redox-sensitive iron-bound P (BD–SRP) represented 19% of total sediment P on average, and P associated with organic matter (NaOH–DNRP and HA–P) represented about 25% of total sediment P. Although initial flooding did not greatly change total sediment P content, relative amounts of some chemical forms of sediment P changed significantly. Phosphorus extracted by the BD solution, both nonreactive and reactive, significantly decreased after flooding, while nonreactive P extracted with NaOH and P associated with humic acids increased significantly after flooding ($P < 0.02$, Supplemental Table 1).

The ''loosely sorbed'' (H_2O–P) and redox-sensitive iron-bound (BD–SRP) fractions are expected to be the most readily released. The total mass of H_2O–SRP and BD–SRP in sediments before flooding far exceeded the observed increase in surface-water P. For example, the top 5 cm in Wetland 1 contained an average of about 2.8 kg H_2O–P and 193 kg BD–SRP before flooding, compared with an observed P increase in surface waters in that zone of about 2 kg SRP. The average declines we observed in BD–SRP and BD–DNRP, extrapolated to the entire wetland area, each represents

Table 4. Sediment Characteristics Measured in the Top 10 cm of Cores Sampled from Two Drainage Ditches (Ditches 1–2) and Three Newly Re-flooded Wetland Areas (Wetlands 1–3) of the Fort Custer Wetland

Site	Organic matter (%)	Total phosphorus ($\mu g\ P\ g^{-1}$)	Total iron ($mg\ Fe\ g^{-1}$)	TFe:P molar ratio	Ox-Fe ($mg\ Fe\ g^{-1}$)	Total Al ($mg\ Al\ g^{-1}$)	$CaCO_3$ ($mg\ g^{-1}$)	AVS ($\mu mol\ S\ g^{-1}$)
Ditch 1	13	334	10.75	17.88				
Ditch 2	45 ± 4	904 ± 163	12.47 ± 2.18	7.99 ± 1.55				
Wetland 1	65 ± 7	1,502 ± 286	18.63 ± 3.83	7.57 ± 1.71	11.31	4.79	0.68	1.31
Wetland 2	68 ± 3	1,245 ± 246	7.98 ± 0.7	4.81 ± 1.59				
Wetland 3	63 ± 3	1,489 ± 198	8.24 ± 0.46	3.17 ± 0.43	5.62	10.51	0.58	0.94
Entire wetland	60	1,229	11.56	6.78				

Sediment characteristics for the entire wetland are means weighted by the areas of the zones in which they were measured. Except for organic matter, all values are expressed as per gram dry weight. All sites were sampled in 2008 for organic matter (OM), total phosphorus (TP), and total iron (TFe) measurements. In addition, Wetland 1 and Wetland 3 were sampled in 2009 and 2010 for OM, TP, and TFe, and in 2010 only for oxalate-extractable iron (Ox-Fe), total aluminum (Total Al), total calcium carbonate ($CaCO_3$), and acid volatile sulfides (AVS)

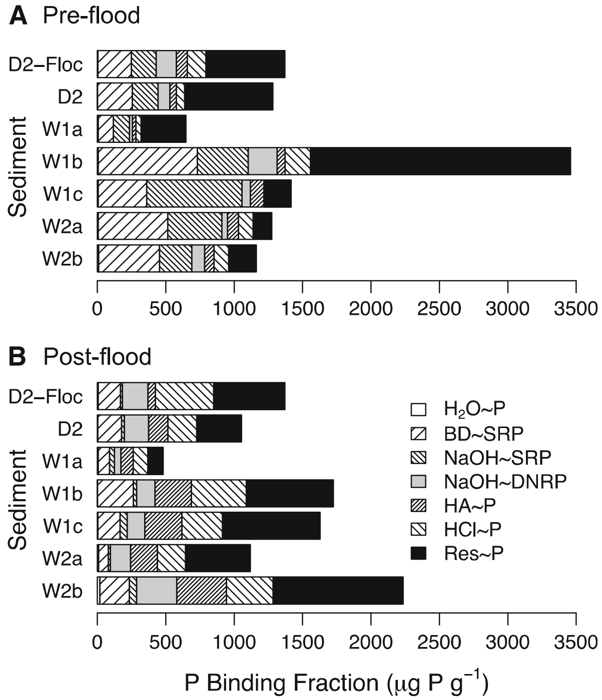

Figure 5. Comparison of sequentially extracted P-binding fractions in sediments collected immediately before (10 days) and immediately after (2 days) re-flooding the Fort Custer Wetland restoration site in $\mu g\ P\ g^{-1}$ dry weight. Locations include Ditch 2 (D2) and Wetlands 1–2 (W1 and W2, sampling points within locations denoted by *letters*). See "Methods" section for definitions of P-binding fractions.

approximately 100–300 kg P. The average increases in NaOH–DNRP and HA–P correspond to about 72–180 and 60–140 kg P, respectively. Overall, it appears that as much as 600 kg P was released from the BD–SRP and BD–DNRP fractions combined, and all or part of this was transformed to NaOH–DNRP and HA–P. Thus, the release of all of the H_2O–P and/or a small fraction of the BD–SRP from the sediments could explain the estimated total release of P into the surface waters.

Intact Core Experiment

In the experiment testing the effects of surface-water DO on sediment P release to surface waters in intact cores, sediments from Wetland 1 exhibited higher P release rates than Wetland 3 (Table 5). Wetland 1 sediments in the Low DO treatment released P at a significantly higher rate than in the High DO treatment, whereas cores from the Wetland 3 showed no significant effect of DO treatment (Table 5).

Table 5. Mean P Release Rates (Plus or Minus One Standard Error) for Intact Core Experiments from Two Locations in the Fort Custer Wetland Restoration Site in which Surface Water was Aerated (High DO) or Unaerated (Low DO)

Site	Treatment	P release rate (mg P m^{-2} day^{-1})	*P* value
Wetland 1	High DO	3.6 ± 1.3	
	Low DO	11.8 ± 2.7	**0.022**
Wetland 3	High DO	0.18 ± 0.08	
	Low DO	0.34 ± 0.15	0.3

Six replicate cores per treatment were assayed, and bold P values indicate significant differences ($\alpha = 0.05$) between High and Low DO treatments (F test) for each location.

DISCUSSION

Upon re-flooding of this historically drained wetland, we observed rapid release of inorganic P from sediments to surface waters, generating high water column P concentrations that supported the proliferation of duckweed and filamentous algae. In addition, considerable amounts of P were likely exported to downstream ecosystems in the months following re-flooding. Restoring wetland hydrology to historically drained lands may thus have unintended consequences for water quality.

Previous studies have shown that flooding historically drained soils and sediments may lead to P release, but fewer studies have provided evidence for the mechanism by which P is released in shallow water bodies. We have provided direct evidence by measuring sediment characteristics, including P-binding fractions, before and after re-flooding. During the first year of flooding, the occurrence of the highest water column P concentrations at times of low DO suggests that the release of Fe-bound P under reducing conditions was responsible for much of the increase, as discussed below. The large pool of sediment P in this fraction provides support for the plausibility of this hypothesis. Release of the Fe-bound P has often been observed when hypolimnetic waters of deep stratified lakes become depleted in oxygen (Marsden 1989), but this mechanism has not often been documented in wetland waters that are too shallow to develop persistent thermal stratification.

This wetland had been drained for most of the last century, but it had been occasionally backflooded by beaver dams as recently as the 1990s. It is likely that P accumulated in the muck soils during drained periods, although some accumulated P was presumably lost to the outflow during dammed periods. Net P accumulation may have occurred because the drained years exceeded the flooded ones. Ecosystems with longer periods of drainage might be expected to accumulate more soil P that is stored in forms that are susceptible to release upon re-flooding.

Sediment P Release Rates and Mechanisms

In both the field and laboratory studies across a diversity of environments, re-flooding historically drained sediments has been shown to lead to P release from sediments to surface waters, although reported release rates vary greatly (Table 6). The rates of sediment P release we observed (6.5 mg SRP and 9.1 mg TP m^{-2} day^{-1}) immediately following re-flooding were in the range of observed values from past studies (Table 6). Following the initial pulse of P release, we observed high surface-water P concentrations and filamentous algal blooms during the ensuing year, suggesting that sediment P release continued to occur but the released P was effectively sequestered in biomass.

Initial sediment P release was rapid (within days), and over 70% was released as SRP, much of which likely originated in Fe-bound sediment fractions. Summer flooding probably caused rapid oxygen depletion in these highly organic sediments, leading to reduction of oxidized Fe and subsequent mobilization of the formerly Fe-sorbed PO_4^{3-}. We measured high dissolved Fe(II) concentrations in pore waters, and both the loosely sorbed and redox-sensitive iron-bound inorganic P pools (H_2O–P and BD–SRP) measured in sediments before flooding were large enough to explain the observed increase in surface-water P. The rapid decline observed in sediment BD–SRP after flooding also suggests the release of Fe-bound P. Finally, the higher rates of P release in experimental intact cores with low-oxygen surface water, which we observed in the Wetland site containing the highest total sediment Fe concentration (Tables 4, 6), provide further evidence for the sediment's potential to release Fe-bound P under hypoxic conditions.

Table 6. Published P Release Rates from Re-flooded Historically Drained Sediments Measured in Situ (A) or in Intact Cores (B) Sampled from Drained or Recently Re-flooded Wetland Ecosystems

(A) Field studies

Source	Location	SRP release (mg m^{-2} day^{-1})	TP Release (mg m^{-2} day^{-1})
Ardón and others (2010a)	Timberlake Restoration Project, North Carolina, USA (440 ha)	0.03	0.15
Newman and Pietro (2001)	Everglades Nutrient Removal Project, Florida, USA (1,545 ha)	0.33–1.5	
Coveney and others (2002)	Marsh Flow-Way Demonstration Project, Florida, USA (210 ha)	10–20	
Wong and others (2011)	Williamson River Delta, Oregon, USA (2,200 ha)		10
Duff and others (2009)	Wood River Wetland, Upper Klamath Basin, Oregon, USA (1,300 ha)	19.2–72	
This study	Fort Custer Wetland, Michigan, USA (9 ha)	6.5	9.1

(B) Intact core studies

Source	Sediment source	SRP release (mg m^{-2} day^{-1})	TP release (mg m^{-2} day^{-1})
Aldous and others (2005)	Restored wetlands near Upper Klamath Lake, Oregon, USA	8.6–55	
Bostic and White (2007)	Blue Cypress Marsh Conservation Area, Florida, USA	5.74–43	8.69–26.6
Corstanje and Reddy (2004)	Blue Cypress Marsh Conservation Area, Florida, USA	0.7–109	2.8–436
Martin and others (1997)	Everglades Nutrient Removal Project Wetland, Florida, USA	1.2–6	
Pant and Reddy (2003)	Artificially drained agricultural (dairy) land slated for a constructed wetland, Florida, USA	11–22	
Zak and others (2010)	Drained fens, Germany and Poland	0–52.3	
This study	Fort Custer Wetland, Michigan, USA	0.18–11.8	

Ultimately, P release occurred because the drained sediments contained large amounts of readily mobilized P forms. Wetland sediments store large amounts of P in organic matter, which is slow to decompose under flooded, anoxic conditions. However, when wetland sediments are drained, mineralization rates increase as organic sediments are exposed to atmospheric oxygen (McLatchey and Reddy 1998), producing inorganic P. In addition to the conversion of the existing sediment organic P to the more reactive inorganic P, large amounts of P are commonly added to drained wetland soils when they are used for agriculture. The specific agricultural management history of the Fort Custer Wetland is unknown; the area was converted to a military training area in 1917, and it may have been farmed subsequent to conversion as well as before. Although it is unlikely that the site received high loads of industrial P fertilizer, it may have received P supplements through the use of manure or other amendments.

The Fe-rich sediments in this wetland likely led to even higher rates of inorganic P accumulation during drained periods than may have otherwise occurred because of the high P-sorption capacity of oxidized Fe when the soils were unsaturated. The sediment Fe:P molar ratios were lower (~7) than the suggested threshold ratios of 8–10 below which sediments are likely to release formerly Fe-bound P (Jensen and others 1992; Geurts and others 2008; Zak and others 2010). In addition, a high proportion of sediment total Fe was in the amorphous, poorly crystalline form (i.e., oxalate extractable), which has a higher P-sorption capacity than more crystalline forms of oxidized iron due to higher surface area and greater content of –OH groups for PO_4^{3-} ions to exchange with (McLaughlin and others 1981; Axt and Walbridge 1999).

Comparatively low concentrations of inorganic N and low (below Redfield) N:P ratios indicate high P availability in the wetland waters during most of our sampling periods, and suggest that aquatic primary production would have been N-limited during this time. Sediment re-flooding is less frequently associated with release of inorganic N than with release of P. In fact, the oxygen depletion associated with flooding likely enhances denitrification, leading to net retention or transformation, rather than release, of sediment N (Ardón and others 2010b). High DOC concentrations in the wetland following re-flooding may reflect leaching of the newly flooded soils, and the released DOC may have been associated with P as a component of organic molecules or sorbed to humic materials via metal oxides (Gerke 2010). Sediments in this wetland contained remarkably high humic-associated P compared with other wetlands in the area (Kinsman-Costello 2012); thus, some P release from these fractions associated with DOC is possible. However, most of the initial pulses of released P we observed in the water column were in the form of SRP rather than DOP.

The duration of P release beyond the 2 years of this study is difficult to ascertain. To roughly estimate the potential range of duration of sediment P release, we calculated the amounts of ''releasable P'' in sediments as the masses of P in loosely bound (H_2O–P) and redox-sensitive iron-bound (BD–SRP) fractions in the newly flooded sediments. We estimated the amount of time it would take to release this mass of P using the high (Wetland 1, Low DO) and low (Wetland 3, High DO) P release rates measured in the intact core experiment (Table 5). These estimates of P release duration are based on the assumptions that (1) only, and all, P in H_2O–P and BD–SRP fractions will be released from sediments, (2) P fraction pools are homogenous throughout Wetland zones, (3) P release rates measured in intact cores reflect actual release rates in the ecosystem, and 4) P release rates are not temporally variable.

Based on the above approach, the estimated duration that the wetland sediments could continue to release P into surface waters ranges from 3 to 177 years, a very large range that reflects the variable rates of P release in the core experiment. It is notable that we observed much lower water column P concentrations in the second year of inundation, although high aquatic primary production was still evident, suggesting nutrient-replete conditions that could be maintained by ongoing P release. Remote-sensing imagery available in Google Earth shows that this high aquatic production persisted in the summers of 2011 and 2012.

Fate of P Released from Sediments

Sediment-released P was ultimately returned to the sediments, assimilated into algal and vascular plant biomass, and/or exported from the ecosystem via the outflow. Differences in the distributions of P-binding fractions before and after flooding (Figure 5, Supplemental Table 1) provide some evidence that at least a portion of readily mobilized inorganic P was transformed to less-reactive forms, even in the days following re-flooding.

Export of total P via the wetland outflow was very high, and exceeded the estimate of initial P release, suggesting that sediments continued to release P after the initial release we observed. Most of the export was as organic P, although the initial P release was largely as SRP. There is uncertainty in our estimates of export because of a deficiency of discharge data early on in the study. However, concentrations of conservative ions, such as Mg^{2+}, as well as the lack of observed internal wetland flow, suggest that mixing between most of the newly re-flooded Wetland zones and Ditch 1 and the Outflow Pond was minimal. Thus, a substantial portion of sediment-released P in the wetland zones was likely not flushed from the wetland but rather was ultimately stored in sediments or continues to be internally cycled. The balance between P stored in sediments and transported to downstream ecosystems depends on the morphometry and hydrology of a particular wetland, and would thus be system-specific.

CONCLUSIONS

Shallow flooding of historically drained organic soils high in Fe caused rapid release of large amounts of inorganic P from sediments to surface waters, with resultant prolific growth of duckweed and filamentous algae. The observed increase in water column P concentrations diminished after 1 year, but in the meantime, substantial concentrations of P were exported to downstream ecosystems. High biomass of aquatic primary producers was observed for several years after re-flooding. Our observations demonstrate that when restoring wetlands by re-flooding historically drained areas, especially if sediments are high in readily mobilized P, managers should incorporate monitoring of water quality before and after re-flooding and consider the potential for sediment P release to jeopardize restoration goals and negatively affect downstream ecosystems, at least in the first few years after re-flooding. More research is needed to improve our ability to predict how restoration of hydrological regimes in particular wetland settings will affect P release and export, and how the ecological effects of substantial P releases may be managed or mitigated.

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