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RESEARCH ARTICLE

USDA Conservation Practices Increase Carbon Storage and Water Quality Improvement Functions: An Example from Ohio

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Abstract

We compared potential denitrification and phosphorus (P) sorption in restored depressional wetlands, restored riparian buffers, and natural riparian buffers of central Ohio to determine to what extent systems restored under the U.S. Department of Agriculture's Wetland Reserve Program (WRP) and Conservation Reserve Program (CRP) provide water quality improvement benefits, and to determine which practice is more effective at nutrient retention. We also measured soil nutrient pools (organic C, N, and P) to evaluate the potential for long-term C sequestration and nutrient accumulation. Depressional wetland soils sorbed twice as much P as riparian soils, but had significantly lower denitrification rates. Phosphorus sorption and denitrification were similar between the restored and natural riparian buffers, although all Natural Resources Conservation Service (NRCS) practices had higher denitrification than agricultural soils. Pools of organic C (2570–3320 g/m^2), total N (216–243 g/m^2), and total P (60–71 g/m^2) were comparable among all three NRCS practices but were greater than nearby agricultural fields and less than natural wetlands in the region. Overall, restored wetlands and restored and natural riparian buffers provide ecosystem services to the landscape that were lost during the conversion to agriculture, but the delivery of services differs among conservation practices, with greater N removal by riparian buffers and greater P removal by wetlands, attributed to differences in landscape position and mineral soil composition. At the landscape, and even global level, wetland and riparian restoration in agricultural landscapes will reintroduce multiple ecosystem services (e.g. C sequestration, water quality improvement, and others) and should be considered in management plans.

Key words: Conservation Reserve Program, denitrification, P sorption, riparian buffers, Wetland Reserve Program, wetlands.

Introduction

The Glaciated Interior Plains (GIP) of the Midwestern United States, stretching from Ohio to Minnesota, is one of the most productive agricultural regions on earth, producing more than half of the U.S. corn crop each year (Power et al. 1998). Conversion to agriculture resulted in extensive losses of wetlands and riparian zones in the region, as well as the associated ecosystem services, such as flood protection and water storage, carbon (C) sequestration, and water quality improvement (Zedler 2003; Fennessy & Craft 2011). In the United States, approximately 90 million hectares of wetlands have been drained since European settlement and converted to agriculture and urban land uses (Dahl 2000) and the loss of wetlands continues nationwide (Dahl 2011).

To maintain or enhance the environmental quality of agricultural lands, the U.S. Department of Agriculture (USDA) has established conservation programs to promote the restoration, creation, and enhancement of wetland and riparian ecosystems, such as the Wetland Reserve Program (WRP) and the Conservation Reserve Program (CRP) (Gleason et al. 2008, 2011; Fennessy & Craft 2011). These programs are managed by the USDA Natural Resources Conservation Service (NRCS) and assist landowners in restoring wetlands and riparian buffers in agricultural landscapes (Duffy & Kahara 2011; Faulkner et al. 2011; Fennessy & Craft 2011; Gleason et al. 2011). One of the ultimate goals of the restoration and conservation practices is to reintroduce ecosystem services to these highly altered landscapes (Gleason et al. 2008, 2011).

The loss of wetlands and riparian buffers and the increase in agricultural expansion throughout the GIP have contributed to greater sediment and nutrient loads throughout the region (Goolsby et al. 1999; Zedler 2003). Restoring wetlands and riparian buffers, particularly in agricultural landscapes such as the Midwest, can reintroduce ecosystem services at the

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landscape level and provide water quality improvement benefits across multiple spatial scales (Zedler 2003; Kovacic et al. 2006).

Wetlands and riparian buffers are effective in storing C and removing excess nutrients, particularly nitrogen (N) and phosphorus (P), from surface and ground water (Woltemade 2000; Bruland et al. 2003; Kovacic et al. 2006). Denitrification is the primary mechanism by which N is removed from the landscape and varies based on factors such as temperature, pH, oxygen availability, organic C quantity and quality, NO_3-N availability, soil texture, and soil moisture (Ullah & Faulkner 2006; Orr et al. 2007; Racchetti et al. 2011). Phosphorus retention by wetlands and riparian buffers also contributes to improving water quality and occurs through sorption, burial, and uptake by plants (Bruland et al. 2003; Fennessy et al. 2008).

Wetlands also sequester C, with estimated soil pools in the conterminous United States of 19.6 Pg (Bridgham et al. 2006). In the 16 million hectares of wetland area in the Prairie Pothole Region, Euliss et al. (2006) estimated C storage of 10.1 Mg/ha. They further estimated that if all cropped agricultural land in the region were restored to wetlands, 67 Tg of additional C would be sequestered in the following 5–10 years. While restoring agricultural lands to wetlands can improve overall C storage, research suggests that soil C pools are slow to develop following restoration (Hogan et al. 2004; Ballantine & Schneider 2009).

Wetlands and riparian buffers are being restored throughout the United States through the WRP and CRP (Faulkner et al. 2011; Fennessy & Craft 2011; Gleason et al. 2011). A few of the many conservation practices developed by the NRCS and promoted under these programs include wetland creation (establishing wetlands on non-hydric soils), wetland enhancement (modifying the hydroperiod of an existing wetland using water control structures), wetland restoration (establishment of wetlands on hydric soils), and riparian forest buffer establishment (establishment of a riparian forest buffer along streams where former riparian wetlands existed or where they currently exist but in a degraded state). Collectively, these practices are referred to as *wetland conservation practices*. The purpose of these practices is to restore, enhance, and conserve valuable ecosystem services such as water storage, water quality improvement, wildlife habitat, and biodiversity support (Fennessy & Craft 2011). In aggregate, NRCS has successfully established conservation practices on the ground; however, the subsequent ecosystem services have not been systematically evaluated, and the relative effectiveness of restored wetlands and riparian buffers in removing N and P, thus benefitting downstream water quality, has not been documented, especially in the Midwest.

We addressed this knowledge gap by measuring two key water quality improvement functions in soils (denitrification, P sorption) and pools of soil organic C, N, and P in wetlands and riparian buffers restored or conserved under three different USDA conservation practices: restored wetlands (restored depressional wetlands on former croplands), restored riparian forest buffer strips (buffer strips on former croplands; denoted here as restored riparian), and natural riparian buffers (buffers adjacent to agricultural fields, but conserved through WRP/CRP). The riparian buffer strip projects were paired in that, wherever a buffer strip was restored, an existing forested buffer strip adjacent to a stream was conserved along with it (natural riparian buffers). Our primary objective was to quantify the success of these conservation practices in restoring the provision of selected ecosystem services (denitrification, P sorption, and C storage) on agricultural lands, and to determine which of these three systems provide the greatest levels of water quality improvement and C storage. Due to differences in hydrogeomorphic class (Brinson 1993), we predicted that riparian systems would have greater N-processing capacity (denitrification) and restored depressional wetlands would hold more C and P. In this analysis, we also evaluated how the study sites compared in their ability to provide these services relative to the agricultural soils from which they were restored.

Methods

Site Description

We sampled six restored depressional wetlands, five restored riparian buffers, and five natural riparian buffers in central Ohio (Fig. 1). Sites were selected from a USDA database of landowners enrolled in the WRP and CRP programs in this portion of the GIP. We only sampled sites after receiving permission from landowners. Natural depressional wetlands are not included in the WRP/CRP database and were therefore not sampled in this study. We also sampled five agricultural fields that were adjacent to the conservation practices to serve as a control in establishing gains in the delivery of ecosystem services. These sites were restored under the WRP and CRP between 2003 and 2007. Natural riparian buffers were second-growth forested systems that had not been recently farmed. Soils from all restored and natural sites were clay loams or silt loams, with soil orders being Alfisols, Mollisols, or Inceptisols (Table 1).

Restored wetlands supported emergent vegetation and were dominated by *Typha latifolia* and *T. angustifolia* L., *Carex* sp., *Phalaris arundinacea* L., and *Potamogeton* sp. Restored riparian sites consisted of grass cover and were planted or seeded with species of *Quercus*, *Acer*, and *Populus*. Natural riparian sites were forested and dominated by *Acer saccharinum* L., *A. saccharum* Marsh., *Plantanus occidentalis* L., *Aesculus* sp., and *Fraxinus pennsylvanica* Marsh.

Soil Sampling

Five soils cores (10 cm diameter × 5 cm deep) were collected for denitrification and phosphorus sorption index (PSI) during late June or early July in 2010. An additional five, 15-cm deep soil cores (10-cm diameter) were collected from each site during June–July 2010 using a butyrate tube for determination of organic C, total N, and total P pools. Cores were sectioned into 0–5 and 5–15 cm depth increments so that

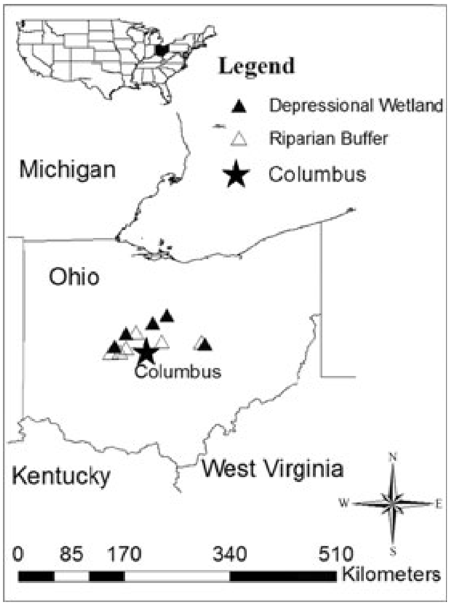

Figure 1. Map of study sites. Each riparian site consisted of a restored and natural riparian buffer.

surface soil properties could be compared to soil processes (PSI, denitrification). Cores (0–5 cm) were also collected from adjacent agricultural fields at five sites for denitrification assays to provide information on what denitrification rates might have been prior to restoration. Following collection, soils were placed on ice in sealed bags and transported to the laboratory.

Soil Processes

PSI was determined by adding 25 mL of a solution of 130 mg P/L to 5 g of field-moist soil, shaking for 24 hours at 120 rpm, and analyzing the supernatant for P using the ascorbic acid method (Bache & Williams 1971). The index ($X/(\log C)^{-1}$) is calculated by dividing the amount of P sorbed (X) by the log of the inorganic P concentration after the 24-hour incubation (log C), and corrected for the volume of the supernatant. Standard curves were prepared by serial dilution of the 130 mg P/L solution. All standard curves had r^2 values of ≥ 0.99. Because the primary objective was to quantify the services provided by different conservation practices, PSI was not measured in agricultural soils. Further, agricultural lands are more likely to be a source, rather than a sink, for P.

Potential denitrification was measured using the acetylene-inhibition method (Tiedje 1994). Twenty-five grams dry weight equivalent of field-moist soil was added to 125-mL Wheaton bottles (Wheaton, Millville, NJ, U.S.A.) with screw caps equipped with gray-butyl septa. Fifty millilters of deionized water was added to each bottle. Water was amended with chloramphenicol (0.21mM) to inhibit microbial growth, and glucose (5.6mM) and potassium nitrate (8.4mM) to remove potential substrate and nutrient limitations. Because these systems are located in agricultural landscapes with potentially high NO_3-N loading (Hanson et al. 1994; Woltemade 2000; Fennessy & Craft 2011), we used higher than standard concentrations of both glucose (1mM) and potassium nitrate (1mM). Bottles were then flushed with ultra-high purity He gas for 5 minutes and 10% of the headspace was replaced with acetylene to block the reduction of N_2O to N_2. Incubations were conducted for 2 hours at 22°C. Headspace samples (10 mL) were collected after 30, 60, 90, and 120 minutes and stored

Table 1. NRCS practices, program codes, and soil series and taxonomy of restored wetlands and restored and natural riparian buffers. *Conserved riparian buffers are natural systems and therefore not assigned a NRCS Practice code.

NRCS Practice	*NRCS Program*	*Soil Series*	*Soil Taxonomy*
Restored wetlands	CRP	Brookston fine loam	Mixed, superactive, mesic, typic, argiaquolls
	CRP	Pewamo silty clay loam	Mixed, active, mesic, typic argiaquolls
	WRP	Millgrove silt loam	Mixed, superactive, mesic, typic argiaquolls
	WRP	Pewamo silty clay loam	Mixed, active, mesic, typic argiaquolls
	WRP	Pewamo silty clay loam	Mixed, active, mesic, typic argiaquolls
	CRP	Brownsville channery silt loam	Skeletal, mixed, active, mesic, typic dystrudepts
Restored riparian buffers	CRP	Miamian silt loam	Mixed, active, mesic, oxyaquic hapludalfs
	CRP	Sloan silt loam	Mixed, superactive, mesic, fluvaquentic endoaquolls
	CRP	Bennington silt loam	Illitic, mesic, aeric epiaqualfs
	CRP	Crosby-Lewisburg silt loam	Mixed, active, mesic, aeric, epiaqualfs
	CRP	Coshocton silt loam	Mixed, active, mesic, aquultic hapludalfs
Natural riparian buffers*	CRP	Miamian silt loam	Mixed, active, mesic, oxyaquic hapludalfs
	CRP	Sloan silt loam	Mixed, superactive, mesic, fluvaquentic endoaquolls
	CRP	Bennington silt loam	Illitic, mesic, aeric epiaqualfs
	CRP	Crosby-Lewisburg silt loam	Mixed, active, mesic, aeric, epiaqualfs
	CRP	Coshocton silt loam	Mixed, active, mesic, aquultic hapludalfs
Agricultural field	—	Crosby-Lewisburg silt loam	Mixed, active, mesic, aeric, epiaqualfs
	—	Millgrove silt loam	Mixed, superactive, mesic, typic argiaquolls
	—	Brookston silt clay loam	Mixed, superactive, mesic, typic, argiaquolls
	—	Pewamo silty clay loam	Mixed, active, mesic, typic argiaquolls
	—	Miamian silt loam	Mixed, active, mesic, oxyaquic hapludalfs

Table 2. Mean (± 1 SE) of surface soil properties (0–5 cm) separated by NRCS practice and adjacent agricultural lands. Different letters indicate significant differences between NRCS practice across both depths based on Tukey's HSD test ($\alpha = 0.05$).

	pH	*Bulk Density (g/cm³)*	*Organic C (%)*	*Total N (%)*	*Total P (μg/g)*	*Sand (%)*	*Silt (%)*	*Clay (%)*
Restored wetland	6.7 ± 0.1^a	1.01 ± 0.08^a	3.4 ± 0.5^{ab}	0.29 ± 0.05^a	683 ± 61^a	65 ± 9^a	24 ± 9^a	5 ± 1^a
Restored riparian	7.2 ± 0.2^a	1.12 ± 0.03^a	3.2 ± 0.3^{ab}	0.28 ± 0.02^a	739 ± 47^a	75 ± 6^a	14 ± 6^a	4 ± 1^a
Natural riparian	7.4 ± 0.1^a	1.01 ± 0.05^a	4.6 ± 0.4^a	0.34 ± 0.02^a	747 ± 52^a	77 ± 5^a	11 ± 4^a	3 ± 1^a
Agricultural field*	6.8 ± 0.2^a	0.96 ± 0.13^a	2.6 ± 0.3^b	0.25 ± 0.02^a	664 ± 70^a	—	—	—

*Enough soil from agricultural fields to conduct particle size analyses was not present.

in 10-mL evacuated Wheaton vials equipped with aluminum crimp tops and gray-butyl septa. After each sample collection, 10 mL of acetylene and He (1:9 ratio) was added to each bottle to maintain pressure. Samples were analyzed for N_2O using a gas chromatograph (Greenhouse Gas GC, Model 8610C, SRI Instruments, Menlo Park, CA, U.S.A. or Shimadzu Model 2014, Columbia, MD, U.S.A.) equipped with an electron capture detector. Potential denitrification was calculated by regression of N_2O accumulation against time. A 5-g field-moist sample was weighed into an aluminum weigh boat and dried at to a constant weight to determine the soil moisture content. Denitrification rates are expressed on a dry weight basis.

Soil Properties

Soils were dried, ground, and passed through a 2-mm mesh sieve. Organic C and total N were determined using a Perkin-Elmer 2400 CHN Analyzer (Perkin-Elmer, Waltham, MA, U.S.A.). An in-house soil standard was analyzed after every 10 samples and resulted in mean (± standard deviation) organic C and total N concentrations of 5.9 ± 0.62% C and 0.35 ± 0.02% N. Carbonates were removed by placing subsamples in a desiccator with a Petri dish of concentrated HCl for 24 hours (Hedges & Stern 1984). Total P was determined using the ascorbic acid method following a nitric-perchloric digestion (Kuo 1996). Standards were digested and analyzed concurrently with all samples (National Institute of Standards and Technology, Estuarine Sediment, 1646a) with a mean recovery of 86%. Concentrations were determined using standard curves generated from a serial dilution of a KH_2PO_4 solution and were analyzed before and after sample analysis. All total P standard curves had r^2 values of ≥0.99. Bulk density was determined by dividing the total dry weight of each soil sample by the volume of the core (Blake & Hartge 1986). Particle size was determined for the conservation practice soils using the hydrometer method (Gee & Bauder 1986). All results are expressed on a dry weight basis. Organic C, total N, and total P pools were calculated using the concentration, sampling depth, and bulk density measurements. pH of field-moist soil was measured with a 1:1 soil:water ratio (Thomas 1996) using a Fisher Scientific pH probe (Thermo Fisher Scientific, Waltham, MA, U.S.A.).

Statistics

Differences in surface soil denitrification (log transformed), PSI, and soil properties (bulk density, pH, soil organic C, total N and P concentrations, % sand, % silt, and % clay) were tested using a one-way analysis of variance (ANOVA) with land use (natural riparian, restored riparian, restored wetland, and agricultural field) as the main factor (IBM SPSS, Armonk, NY, U.S.A.). To account for potential differences in the nine different soil types, separate one-way ANOVAs were conducted on soil processes and properties with soil series as the main factor. Organic C and N pools in the top 15 cm from the three conservation practices and top 10 cm from natural wetlands in the region (Fennessy et al. 2008) were analyzed using one-way ANOVA. Replicate data were not available for total P pools from natural wetlands, so a one-way ANOVA was conducted using only conserved and restored riparian buffers and restored wetlands. Significant differences between factors in all ANOVAs were determined using Tukey's honestly significantly difference (HSD) test ($\alpha = 0.05$). Pearson's correlation analyses were used to test for associations between surface soil (0–5 cm) properties (pH, bulk density, organic C, total N, and total P) and processes (PSI and natural log-transformed denitrification). All analyses were conducted at $\alpha = 0.05$.

Results

Soil Properties

Except for organic C, soil properties did not differ significantly among conservation practices and agricultural fields in surface soils (0–5 cm) ($p > 0.05$) (Table 2). Mean surface soil organic C was significantly greater ($p = 0.009$) in natural riparian buffers (4.6 ± 0.35%) than in agricultural soils (2.6 ± 0.30%; Table 2). Mean total P was significantly greatest ($p < 0.05$) in Brookston (973 ± 55 μg/g) and Sloan (920 ± 43 μg/g) and lowest in Coshocton (402 ± 2 μg/g) and Pewamo (487 ± 82 μg/g). No other soil properties differed by soil series ($p > 0.05$).

Mean organic C, total N, and total P pools in the top 15 cm ranged from 2570 to 3320 g organic C/m², 216 to 243 g N/m², and 60 to 71 g P/m², respectively (Fig. 2). Organic C and total N and P pools (0–15 cm) were comparable among soils from all three conservations practices ($p < 0.05$). However, the natural wetlands measured by Fennessy et al. (2008) (0–10 cm) had the largest C and N pools (7968 and 588 g/m², respectively).

Soil Processes

In contrast to soil properties, which did not differ much among conservation practices, water quality improvement functions

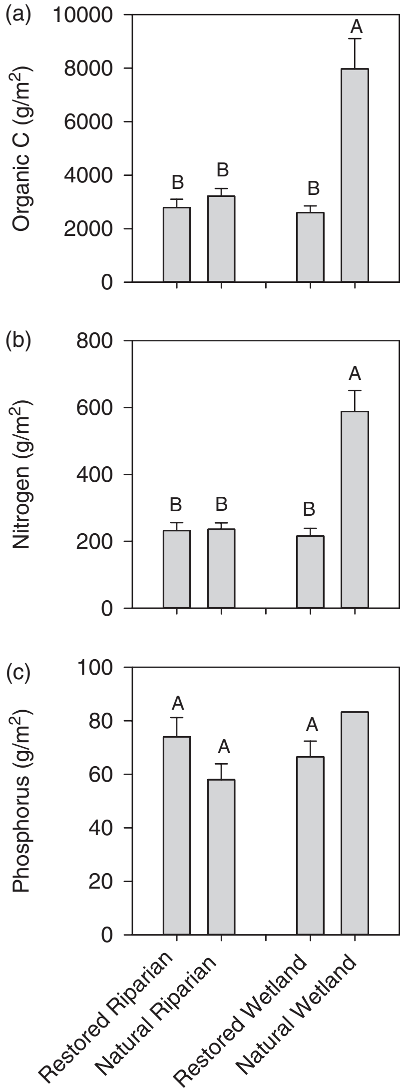

Figure 2. (a) Organic C, (b) total N, and (c) total P pools in top 15 cm of soil. Different letters indicate significant differences based on Tukey's HSD test ($\alpha = 0.05$). Conservation practices (0–15 cm) and natural wetland data (0–10 cm; Fennessy et al. 2008) were integrated to a 15-cm depth. C, N, and P pools from this study were calculated by multiplying the concentrations by the bulk density and 15-cm sampling depth. Data from natural wetlands were calculated by multiplying concentrations by bulk density and 10-cm sampling depth. Replicate values for P pools in natural wetlands were not available and therefore not included in the statistical analysis.

exhibited pronounced differences among the three practices. As predicted, mean (± standard error) PSI was significantly greater ($p < 0.001$) in the restored wetlands (40.3 ± 3.1 mg P/100 g soil) than in natural riparian (18.7 ± 1.2 mg P/100 g soil) and restored riparian (18.9 ± 1.1 mg P/100 g soil) soils (Fig. 3a) and it was significantly but not strongly correlated with organic C across the three practices ($r^2 = 0.10$, $p = 0.02$) (Fig. 4). Mean PSI was significantly greater in Brookston (51.3 ± 5.2 mg P/100 g soil), Pewamo (38.5 ± 4.0 mg P/100 g soil), and Millgrove (29.0 ± 2.4 mg P/100 g soil) relative to the remaining soil series (12.5–21.9 mg P/100 g soil).

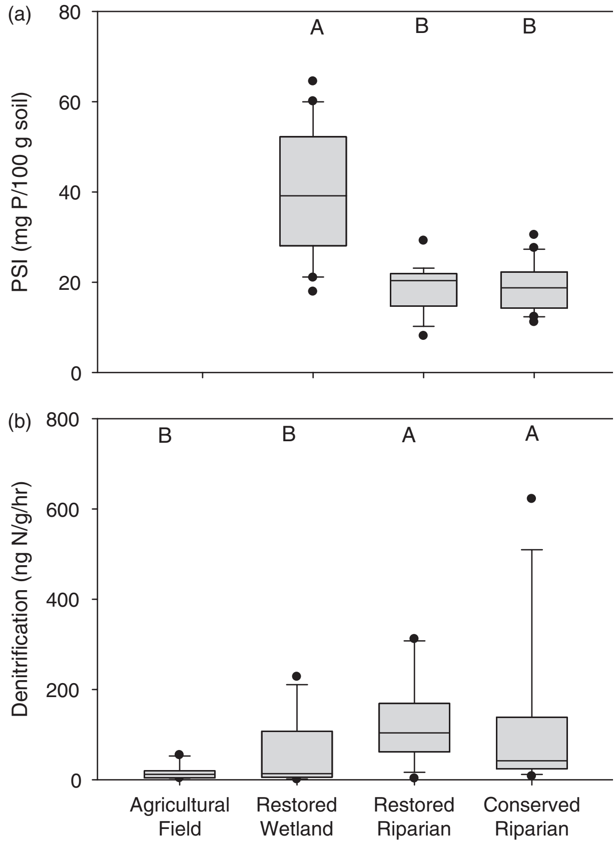

Figure 3. PSI in top 5 cm (a) and potential denitrification (b) from each NRCS practice. Denitrification rates were natural log-transformed prior to analysis Different letters indicate significant differences based on Tukey's HSD test ($\alpha = 0.05$).

In contrast to PSI, denitrification rates were significantly greater ($p < 0.001$) in natural (34.7 ± 9.6 ng N g^{-1} $hour^{-1}$) and restored riparian buffers (42.4 ± 6.9 ng N g^{-1} $hour^{-1}$) relative to restored depressional wetlands (12.3 ± 4.5 ng N g^{-1} $hour^{-1}$) and agricultural fields (5.3 ± 1.7 ng N g^{-1} $hour^{-1}$) (Fig. 3b). Denitrification rates (natural log transformed) were significantly correlated with soil pH ($r^2 = 0.17$, $p < 0.01$), organic C ($r^2 = 0.10$, $p = 0.011$), and C:N ($r^2 = 0.09$, $p = 0.021$) (Fig. 5). When analyzed by soil series, denitrification was significantly greatest in Bennington (235 ± 70 ng N g^{-1} $hour^{-1}$) and Sloan (170 ± 80 ng N g^{-1} $hour^{-1}$), and lowest in Millgrove (14.1 ± 5.7 ng N g^{-1} $hour^{-1}$).

Discussion

Our findings show that restoring depressional wetlands and riparian buffers in agricultural landscapes can enhance C sequestration and water quality improvement functions. Further, at the landscape level, aggregate water quality benefits may be substantial when considering the extensive restoration acreage throughout the region, although these services differ

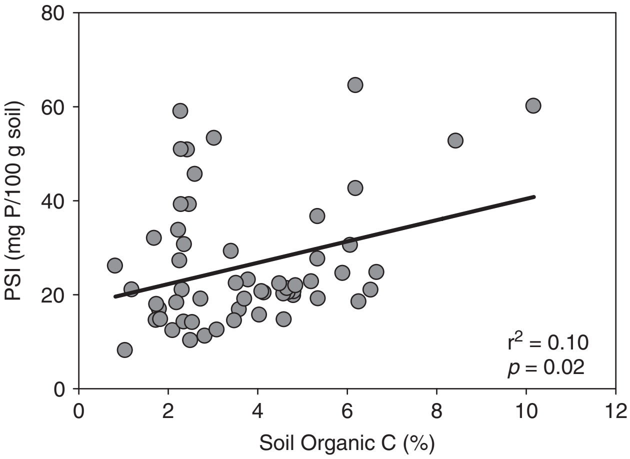

Figure 4. Linear relationship between PSI and soil organic C ($n = 58$). Regression was performed using organic C data from the 0- to 5-cm depth increment.

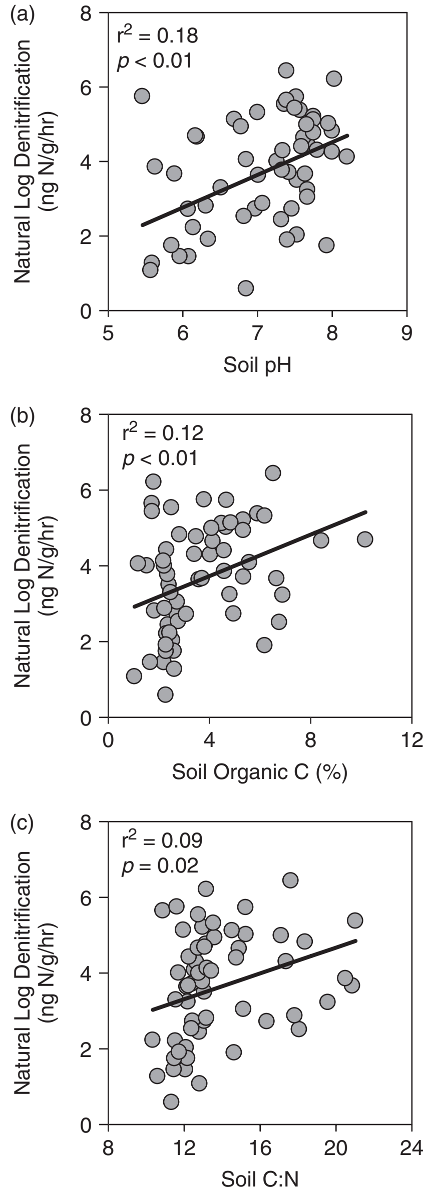

Figure 5. Linear relationships between natural log-transformed denitrification and pH (a), soil organic C (b), and soil C:N ratio (c) ($n = 58$).

between wetlands and riparian buffers. This is particularly important considering that as global population and agricultural production increase around the world, global N and P fertilizer use are predicted to triple to 236 and 83.7×10^6 MT, respectively, by 2050 (Tilman et al. 2001).

Many studies have shown that P sorption in soils can potentially reduce P loading to adjacent waters (Bridgham et al. 2001; Bruland et al. 2003; Hogan et al. 2004; Bruland & Richardson 2006). In this study, PSI was two times greater in restored depressional wetlands than in the restored and natural riparian buffers and organic C explained between 32% (wetland) and 58% (natural riparian) of the variation in PSI. Similarly, Bruland and Richardson (2006) found that PSI (15–184 mg P/100g soil) in depressional wetlands in Minnesota was positively correlated to soil organic matter and extractable Ca. Axt and Walbridge (1999) also found a positive relationship between soil organic matter and P sorption ($r^2 = 0.78$) and suggested that Al–organic matter complexes could play a significant role in P sorption by allowing Al to sorb more P. This is particularly important for riparian areas with their strong connection between uplands and P-limited freshwater aquatic ecosystems, which are susceptible to P-induced eutrophication (Carpenter et al. 1998). The highest PSI values were measured in the soil series found in the depressional wetlands (i.e. Brookston, Pewamo, and Millgrove), which consisted of poorly drained aquic Mollisols.

Although restored wetlands exhibited the greatest PSI, riparian areas were the most active sites of denitrification, which coincided with the Sloan, Bennington, and Miamian soil series. The depressional wetlands sampled were flooded throughout the year, whereas the fluvaquentic, aeric, and oxic soils of the riparian buffers had a more pulsed hydrology, which can lead to higher denitrification rates (Hernandez & Mitsch 2007). Racchetti et al. (2011) found that denitrification rates in river-connected wetlands were two orders of magnitude greater than in isolated wetlands in the Po River Plain of northern Italy. The pulsed hydrology likely contributed to comparable denitrification rates between natural and restored riparian buffers in this study, suggesting that the hydrologic regime was successful in reestablishing N removal via denitrification within 5 years following restoration.

All NRCS practices had higher denitrification rates and soil organic C relative to agricultural soils, indicating that the lost water quality functions were regained through conservation practices and accumulation of soil C, which increased NO_3-N removal from the landscape. Slowed decomposition in flooded depressional wetlands and greater allocthonous inputs in riparian systems can increase soil C storage, thereby increasing denitrification. Our findings are consistent with results from studies in the United States (Bruland et al. 2003), along the Mediterranean coast of Spain (Comín et al. 2001), and other regions around the world (Erwin 2009), where restoring wetlands and riparian buffers can reintroduce water quality improvement functions.

We found comparable soil organic C and total N pools in the top 15 cm in restored and natural riparian buffers, suggesting that restored riparian buffers achieve comparable nutrient pools to their natural counterparts within 5 years following restoration. Natural depressional wetlands in Ohio measured by Fennessy et al. (2008) had organic C and total N and P pools (0–10 cm) that were 1.2 (P) to 3 (organic C and total N) times greater than the 5-year-old restored wetlands. Carbon storage in natural wetland soils was even greater considering that Fennessy et al. (2008) measured the top 10 cm of soil, whereas we measured the top 15 cm. Using a 55-year chronosequence of restored wetlands in New York, Ballantine and Schneider (2009) found less soil organic matter in restored wetlands (20%) than in natural wetlands (46.4%). Conversely, we measured similar soil organic C and total N and P pools in natural and restored riparian buffers. Gift et al. (2010) also found comparable soil organic matter concentrations in surface soils (0–10 cm) of degraded, restored (10–12 years old), and natural riparian areas in Baltimore, Maryland. Development of soil organic matter can be highly variable between systems with different hydrogeomorphic settings and soil type, which influences the reestablishment of denitrification and P sorption. In this study, soil organic C did not significantly differ by soil series, indicating that differences were likely due to conservation practice. It is beneficial to both conserve and restore riparian buffers and depressional wetlands, although to maximize C storage on the landscape, it is better to conserve these systems and prevent loss and degradation.

In conclusion, increased restoration of wetlands and riparian buffers in the agricultural landscape of the GIP has the potential to restore water quality improvement through nutrient removal, particularly N, although the decision of which type of habitat to restore will ultimately depend on the desired outcome (greater N removal by riparian buffers vs. greater P removal by wetlands). Regardless of which conservation practice is implemented, all practices resulted in an increase in water quality benefits and organic C sequestration versus maintaining the land in agriculture. More research is needed to evaluate whether these patterns are consistent between riparian areas in different landscapes and across multiple geographic regions.

Implications for Practice

- Hydrologic regimes and soil type influence the degree of nutrient removal and C sequestration. Restoring systems with hydrologic connectivity and higher soil organic C will better improve local water quality than more isolated systems.
- The efficacy of these nutrient removal and C sequestration will depend on the age of the system and the degree of continued disturbance. Benefits from conservation practices may not be noticeable for several years, although may be more rapid in intermittently flooded systems.
- Conservation practices put into place for other reasons (e.g. wildlife habitat, erosion control, and flood protection) will also have the additional benefit of providing water quality improvement functions and increasing carbon storage on the landscape. As agriculture expands globally, wetland and riparian restoration may help offset increased nutrient loads and soil C losses.

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LITERATURE CITED

Axt, J., and M. Walbridge. 1999. Phosphate removal capacity of palustrine forested wetlands and adjacent uplands in Virginia. Soil Science Society of America Journal **63:**1019–1031.

Bache, B., and E. Williams. 1971. A phosphate sorption index for soils. Journal of Soil Science **22:**289–301.

Ballantine, K., and R. Schneider. 2009. Fifty-five years of soil development in restored freshwater depressional wetlands. Ecological Applications **19:**1467–1480.

Blake, G., and K. Hartge. 1986. Bulk density. Pages 363–375 in A. Klute, editor. Methods of soil analysis. Part 1. Physical and mineralogical methods. Agronomy Monograph No. 9. 2nd edition. American Society of Agronomy, Madison, Wisconsin.

Bridgham, S., C. Johnston, J. Schubauer-Berigan, and P. Weighampel. 2001. Phosphorus sorption dynamics in soils and coupling with surface and pore water in riverine wetlands. Soil Science Society of America Journal **65:**577–588.

Bridgham, S., J. Megonigal, J. Keller, N. Bliss, and C. Trettin. 2006. The carbon balance of North American wetlands. Wetlands **26:**889–916.

Brinson, M. 1993. A hydrogeomorphic classification for wetlands. Wetlands Research Technical Report WRP-DE-4, U.S. Army Corps of Engineers Waterways Experiment Station, Vicksburg, Mississippi.

Bruland, G., and C. Richardson. 2006. An assessment of the phosphorus retention capacity of wetlands in the Painter Creek watershed, Minnesota, USA. Water, Air, and Soil Pollution **171:**169–184.

Bruland, G., M. Hanchey, and C. Richardson. 2003. Effects of agriculture and wetland restoration on hydrology, soils, and water quality of a Carolina bay complex. Wetlands Ecology and Management **11:**141–156.

Carpenter, S., N. Caraco, D. Correll, R. Howarth, A. Sharpley, and V. Smith. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecological Applications **8:**559–568.

Comín, F., J. Romero, O. Hernández, and M. Menéndez. 2001. Restoration of wetlands from abandoned rice fields for nutrient removal, and biological community and landscape diversity. Restoration Ecology **9:**201–208.

Dahl, T. 2000. Status and trends of wetlands in the conterminous United States 1986 to 1997. U.S. Department of the Interior; Fish and Wildlife Service. Washington, D.C.: 82.

Dahl, T. 2011. Status and trends of wetlands in the conterminous United States 2004 to 2009. U.S. Department of the Interior; Fish and Wildlife Service. Washington, D.C.: 110.

Duffy, W., and S. Kahara. 2011. Wetland ecosystem services in California's Central Valley and implications for the Wetland Reserve Program. Ecological Applications **21:**18–30.

Erwin, K. 2009. Wetlands and global climate change: the role of wetland restoration in a changing world. Wetlands Ecology and Management **17:**71–84.

Euliss, N., R. Gleason, A. Olness, R. McDougal, H. Murkin, R. Robarts, R. Bourbonniere, and B. Warner. 2006. North American prairie wetlands are important nonforested land-based carbon storage sites. Science of the Total Environment **361:**179–188.

Faulkner, S., B. Keeland, S. Walls, and D. Telesco. 2011. Effects of conservation practices on wetland ecosystem services in the Mississippi Alluvial Valley. Ecological Applications **21:**S31–S48.

Fennessy, S., and C. Craft. 2011. Agricultural conservation practices increase wetland ecosystem services in the Glaciated Interior Plains. Ecological Applications **21:**S49–S64.

Fennessy, M., A. Rokosch, and J. Mack. 2008. Patterns of plant decomposition and nutrient cycling in natural and created wetlands. Wetlands **28:**300–310.

Gee, G., and J. Bauder. 1986. Particle-size analysis. Pages 383–411 in A. Klute, editor. Methods of soil analysis. Part 1. Physical and mineralogical methods. American Society of Agronomy, Madison, Wisconsin.

Gift, D., P. Groffman, S. Kaushal, and P. Mayer. 2010. Denitrification potential, root biomass, and organic matter in degraded and restored urban riparian zones. Restoration Ecology **18:**113–120.

Gleason, R., M. Laubhan, and N. Euliss. 2008. Ecosystem services derived from wetland conservation practices in the United States Prairie Pothole Region with an emphasis on the U.S. Department of Agriculture Conservation Reserve and Wetlands Reserve Programs. U.S. Geological Professional Paper 1745, p. 58.

Gleason, R., N. Euliss, B. Tangen, M. Laubhan, and B. Browne. 2011. USDA conservation program and practice effects on wetland ecosystem services in the Prairie Pothole Region. Ecological Applications **21:**S65–S81.

Goolsby, D., W. Battaglin, G. Lawrence, R. Artz, B. Aulenbach, R. Hooper, D. Keeney, and G. Stensland. 1999. Fluxes and sources of nutrients in the Mississippi-Atchafalaya Basin: Topic 3 Report for the Integrated Assessment on Hypoxia in the Gulf of Mexico. NOAA Coastal Ocean Program Decision Analysis Series Number 17. NOAA Coastal Ocean Office, Silver Spring, Maryland.

Hanson, G., P. Groffman, and A. Gold. 1994. Denitrification in riparian wetlands receiving high and low groundwater nitrate inputs. Journal of Environmental Quality **23:**917–922.

Hedges, J., and J. Stern. 1984. Carbon and nitrogen determinations of carbonate-containing solids. Limnology and Oceanography **29:**657–663.

Hernandez, M., and W. Mitsch. 2007. Denitrification in created riverine wetlands: influence of hydrology and season. Ecological Engineering **30:**78–88.

Hogan, D., T. Jordan, and M. Walbridge. 2004. Phosphorus retention and soil organic carbon in restored and natural freshwater wetlands. Wetlands **24:**573–585.

Kovacic, D., R. Twait, M. Wallace, and J. Bowling. 2006. Use of created wetlands to improve water quality in the Midwest-Lake Bloomington case study. Ecological Engineering **28:**258–270.

Kuo, S. 1996. Phosphorus. Pages 869–919 in D. Sparks, editor. Methods of soil analysis. Part 3. Chemical methods. Soil Science Society of America, Madison, Wisconsin.

Orr, C., E. Stanley, K. Wilson, and J. Finlay. 2007. Effects of restoration and reflooding on soil denitrification in a leveed Midwestern floodplain. Ecological Applications **17:**2365–2376.

Power, J., A. Flowerday, R. Wiese, and D. Watts. 1998. Agricultural nitrogen management to protect water quality. Bulletin 4. USDA-CREES-IDEA, Washington, D.C.

Racchetti, E., M. Bartoli, E. Soana, D. Longhi, R. Christian, M. Pinardi, and P. Viaroli. 2011. Influence of hydrological connectivity of riverine wetlands on nitrogen removal via denitrification. Biogeochemistry **103:**335–354.

Thomas, G. 1996. Soil pH and soil acidity. Pages 475–490 in D. Sparks, editor. Methods of soil analysis. Part 3. Chemical methods. Soil Science Society of America, Madison, Wisconsin.

Tiedje, J. 1994. Denitrifier enzyme activity. Pages 256–257 in R. Weaver, S. Angle, and P. Bottomley, editors. Methods of soil analysis. Part 2. Microbiological and biochemical properties. Soil Science Society of America, Madison, Wisconsin.

Tilman, D., J. Fargione, B. Wolff, C. D'Antonio, A. Dobson, R. Howarth, D. Schindler, W. Schlesinger, D. Simberloff, and D. Swackhamer. 2001. Forecasting agriculturally drive global environmental change. Science **292:**281–84.

Ullah, S., and S. Faulkner. 2006. Denitrification potential of different land-use types in an agricultural watershed, lower Mississippi valley. Ecological Engineering **28:**131–140.

Woltemade, C. 2000. Ability of restored wetlands to reduce nitrogen and phosphorus concentrations in agricultural drainage water. Journal of Soil and Water Conservation **55:**303–309.

Zedler, J. 2003. Wetlands at your service: reducing impacts of agriculture at the watershed scale. Frontiers in Ecology and the Environment **1:**65–72.