

Nitrogen Dynamics in Two Created Riparian Wetlands over Space and Time

Blanca Bernal¹; Christopher J. Anderson²; and William J. Mitsch³

Abstract: Temporal and spatial variations of nitrogen (N) soil storages and fluxes were examined at two 1-ha created riverine wetlands in the U.S. Midwest. Soil N content (total N, organic-N, NO₃-N, and NH₄-N), N accumulation rates, and soil C:N ratios were compared between the two wetlands constructed 15 years earlier (one was planted and the other left to naturally colonize). Differences in wetland soil N content and accumulation were also examined in relation to proximity of river input and relative to a range of topographic features related to wetland water depth. The planted and naturally colonized wetlands showed similar rates of N accumulation. However, differences were detected related to the content of mineralized forms of N (NO₃-N and NH₄-N) that may relate to the history of vegetation communities at these wetlands. Significant spatial variation of N accumulation was detected within the wetlands, with the highest rates found in the deeper open water communities compared to shallow emergent marsh/edge vegetation communities (23.5 ± 2.0 versus 17.3 ± 1.3/18.2 ± 1.4 g N m⁻² y⁻¹). Nitrogen budgets comparing two ages of the wetlands illustrate higher N accumulation rates (by 19%), higher N reduction in the surface water (47 to 52% reduction), and increased denitrification rates (by 13%) from year 10 to year 15. We also found out that nitrogen accumulation in the soil was 7.1 to 7.5% higher than were denitrification rates in these young wetlands. We discuss the importance of efforts such as this to support better understanding of N pathways in both created and natural wetlands, while providing critical data needed to improve modelling efforts and assess the long-term effectiveness of wetlands for improving water quality. DOI: 10.1061/(ASCE)HE.1943-5584.0001397. © 2016 American Society of Civil Engineers.

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Introduction

The role of riparian wetlands for downstream water quality has continued to be an important topic for natural resource policy and management in the United States. It is understood that wetlands are important landscape features (Mitsch and Gosselink 2015), and they have been purposely restored, created, and preserved to improve and sustain downstream water quality (Kadlec and Wallace 2008). Although evidence clearly shows a benefit to water quality, research has increasingly focused on better understanding the magnitude to which adjacent riparian and headwater wetlands improve downstream water quality (U.S. EPA 2015). This has become necessary in order to evaluate the appropriateness of regulatory measures and the effectiveness of various management practices. More specifically, there has been recent interest regarding the appropriateness of adjacent and headwater wetlands for regulatory jurisdiction as part of Section 404 of the Clean Water Act. In early 2015, the U.S. Environmental Protection Agency and the U.S. Army Corps of Engineers released new rules that revise

the regulatory language related to wetlands and the administrative definition of what constitutes “waters of the United States” (Copeland 2015). Over the past 15 years, other substantial regulatory changes have occurred following three Supreme Court decisions (van der Valk and Pederson 2003; Leibowitz et al. 2008; Mitsch and Gosselink 2015). At the crux of these changes has been the uncertain role and relative importance of adjacent and geographically isolated wetlands for downstream water quality (Leibowitz et al. 2008).

Wetlands certainly improve water quality through various physical, chemical, and biological processes; but often the extent and importance of these pathways are uncertain or unknown. Research, monitoring, and modeling all play a critical role for understanding wetlands and their capability to minimize water pollutants such as excessive nutrients, which continue to be problematic. For example, nitrogen (N) has been a persistent problem since the mid-twentieth century due to the intensive use of fertilizer for crop production and fossil fuel combustion (Mitsch et al. 2001, 2005a; Galloway et al. 2003; Scavia et al. 2007). Excess nitrogen is often transported as nitrate via drainage tiles and ditches to rivers and streams, contaminating drinking waters and resulting in “nitrate advisories” when levels exceed the EPA’s acceptable 10 ppb in drinking water (Mitsch and Day 2006). Nitrate can result in the development of methemoglobinemia (also known as “blue baby syndrome” as it can cause newborns fatalities) and stomach cancer (Mitsch et al. 2001; Hanley and Spash 2003). Besides the risk to public health, excess N poses an ecological threat by being transported downstream to coastal waters where nitrogen is more likely the limiting nutrient. The Gulf of Mexico receives the waters from the Mississippi River Basin (MRB), coming all the way from the Midwest U.S. “corn belt.” The loss of natural wetlands in the MRB due to conversion and artificial drainage of the landscape in the

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last century (about 30 million ha; Mitsch and Day 2006) and the disconnection of the river from its floodplain and deltaic plain (Dahl 1990; Mitsch et al. 2001, 2005a) have exacerbated the amount of nitrate that reaches the Gulf of Mexico. Nitrate-nitrogen is a macronutrient that is naturally limited, and therefore, when excess amounts reach the waters of the Gulf, it prompts eutrophication and algal blooms that result in hypoxia (dissolved oxygen levels below 2 mg L^{-1} ; NRC 2000; Mitsch et al. 2001). Low oxygen levels are fatal for marine life, and thereby the hypoxic area that develops every summer in the Gulf of Mexico has become known as “the dead zone” (Rabalais et al. 2002; Rabalais 2011).

Natural, created, and restored wetlands are effective N sinks and have been promoted as a way to reduce downstream eutrophication (Mitsch et al. 2001, 2005a; Scavia et al. 2007). Wetlands have been demonstrated to be effective at reducing N loads from runoff and improving water quality through denitrification and N uptake (Kadlec and Wallace 2008; Fink and Mitsch 2004; Mitsch and Day 2006; Noe et al. 2013). Denitrification occurs in wetlands thanks to the presence of standing water creating anaerobic conditions in the soil, and to the availability of abundant organic matter. Though this process, microbes remove nitrate by transforming it into gas (N_2 and, to a lesser extent, N_2O ; Mitsch and Gosselink 2015). Nitrogen uptake in wetlands occurs mainly through plant biomass, transforming mineral N to organic nitrogen that can stay stored in the wetland as standing vegetation or as soil organic matter (Mitsch and Gosselink 2015).

To better understand the role of created and natural wetlands for addressing excess nutrients on the landscape, models are increasingly being employed. With advancements in microcomputing, there have been substantial developments in the capabilities of ecological models. Physical-based wetland models are becoming better able to represent and synthesize the interactions of soil, hydrology, and vegetation while predicting how changes in the environment or management regime may alter various wetland functions (Mitsch et al. 1982, 1998; Mitsch and Reeder 1991; Wang and Mitsch 2000; Rousseau et al. 2004; Cui et al. 2005; Zhang and Mitsch 2005; Li et al. 2010; Sharifi et al. 2013; Marois and Mitsch 2016). In lieu of standard water quality monitoring, modeling provides several benefits including lower cost and labor. Models can also improve our understanding of the complex biogeochemical processes that constitute wetland environments and ultimately be used to help design constructed wetlands for optimal treatment capacity. Models have been being developed for a wide range of treatment applications including agricultural N runoff (Dørge 1994; Spieles and Mitsch 1999), municipal wastewater (Merlin et al. 2002; Dušek et al. 2008), and stormwater (Walker 2001; Wong et al. 2006).

With all the possible advantages that models provide and their increasing sophistication, they are ultimately only as good as the information used to develop them. The challenge of developing effective wetland models continues to be the availability of reliable and comprehensive wetland data. One example of long-term monitoring and research effort ideally suited for development of ecological models is the Olentangy River Wetland Research Park (ORWRP) on the campus of Ohio State University in Columbus, Ohio, U.S. (Mitsch et al. 2012, see site description in the following). The ORWRP includes two nearly identical 1-ha constructed wetlands that were completed in 1994 (Fig. 1). Initially, the only difference between these wetlands was that one was planted with common macrophytes while the other was left to naturally colonize as part of a large-scale experiment in ecological self-design (Mitsch et al. 1998, 2012). Since construction, these wetlands have been managed identically including the amount of water pumped into them from the adjacent Olentangy River.

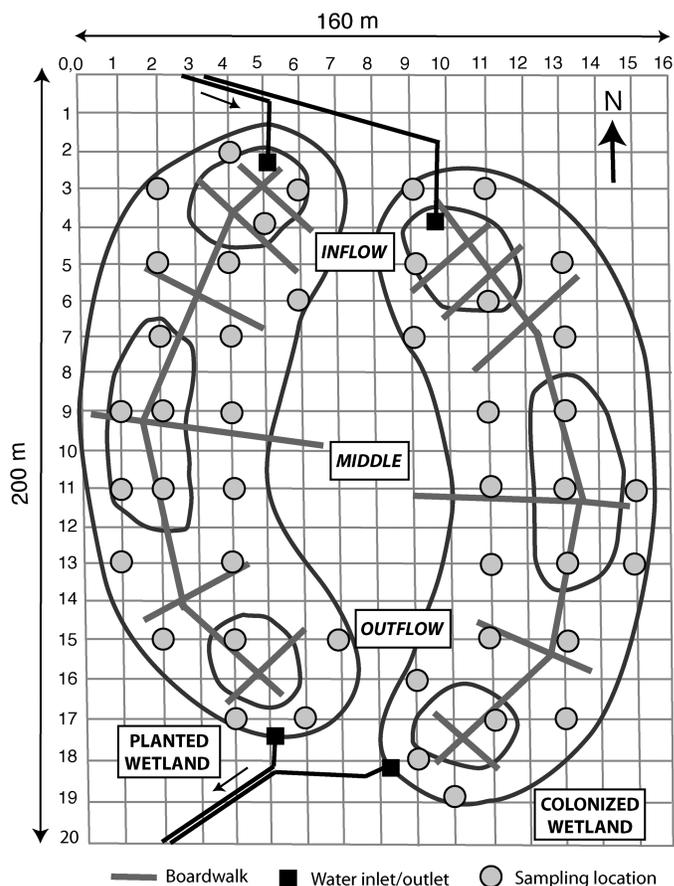


Fig. 1. Schematic of the two created experimental wetlands at the Olentangy River Wetland Research Park in central Ohio showing soil sampling locations within the 10-m grid, water inflows and outflows, direction of flow, and boardwalks; the wetland sections (inflow, middle, and outflow) and the three major open water zones (enclosed areas within each wetland) are also indicated (from Bernal and Mitsch 2013, reprinted by permission, ASA, CSSA, SSSA)

Over the 15 years since its construction, the wetlands have been regularly monitored for an array of functional measures such as daily water quality improvement, hourly water budget, annual changes in vegetative communities, and marsh productivity (Mitsch et al. 2012, 2015). Furthermore, the wetlands have been utilized for multiple experimental research projects. Some of the pertinent research related to wetland carbon and nutrient cycling has included examinations on sediment accretion (Harter and Mitsch 2003; Mitsch et al. 2014a), organic matter accrual (Anderson and Mitsch 2006; Bernal and Mitsch 2013), denitrification (Hernandez and Mitsch 2007; Batson et al. 2012; Song et al. 2014), methane emission (Altor and Mitsch 2006; Nahlik and Mitsch 2010; Sha et al. 2011), and particulate organic carbon (Waletzko and Mitsch 2013). The combined monitoring and research at the ORWRP is well suited to reveal important temporal and spatial variations and supports (and has supported) a wide array of modeling efforts (Wang and Mitsch 2000; Zhang and Mitsch 2005; Mitsch et al. 2013). The objective of this paper is to emphasize the value of long-term, intensive monitoring toward further understanding of wetland processes. Long-term, historic data collected from two created riparian wetlands were used to examine:

1. Changes in N (accumulation rates, content, dissolved N) based on vegetation history, proximity to inflow, and topographic communities.

- Potential changes in measured N pools and processes 10 and 15 years after construction.
- The relative importance of organic matter accumulation, denitrification, and other processes for N reduction by created wetlands.

Materials and Methods

Site Description

Two symmetrical 1-ha wetlands were created in 1994 at the Olentangy River Wetland Research Park (40° 01' N, 82° 01' W) in central Ohio. Since their creation, these wetlands receive water pumped from the third-order Olentangy River, following the hydrologic pulses that a riverine wetland would naturally have at this site (Mitsch et al. 2012, 2014b). The introduction of water allowed hydric wetland soils to develop quickly over the non-hydric parent material (Ross silt loams; NRCS 2014), and sediment has been accreting in these two wetlands ever since creation (Harter and Mitsch 2003; Anderson et al. 2005). The western basin was planted in 1994 with 13 native species macrophytes while the eastern basin was left unplanted to be colonized naturally (Mitsch et al. 1998, 2012, 2014a). Over time, this planting experiment has resulted in differences of aboveground biomass and net primary productivity and diversity of vegetation communities (Mitsch et al. 2012, 2014b). At the time of this study, 15 years after wetland creation, both wetlands had converged in number of species (101 in the planted wetland and 97 in the colonized wetland), representing a total of 22 vegetation communities (Mitsch et al. 2012, 2014b). The successful wetland hydrology, vegetation establishment, and soil development have allowed these created wetlands to function as sinks of nutrients (nitrogen and phosphorus; Mitsch et al. 2012, 2014b), and to provide the ecosystem services that characterize natural riverine wetlands. More information about how these wetlands developed and changed 3, 10, 15, and 20 years after creation can be found in Mitsch et al. (1998, 2005b, 2012, 2014a).

Soil Sampling and Samples Preparation

As described in Bernal and Mitsch (2013), 44 soil cores were extracted 15 years after the wetlands were created, following a 10-m grid spatial pattern (Fig. 1) that was used as a reference in 1993 before the wetlands were flooded, one year after flooding

(1995), and 10 years after flooding, in 2004 (Anderson et al. 2005; Anderson and Mitsch 2006). Sampling points were evenly distributed in the inflow, middle, and outflow sections of each wetland. These points also represented three distinct generic communities (Bernal and Mitsch 2013): open water (permanently flooded communities with no emergent vegetation), emergent (permanently flooded communities with emergent vegetation), and edge community (shallow, intermittently flooded communities with emergent vegetation). As a reference nonwetland site, two extra cores were collected in the forested upland area between both wetland basins. All the soil samples were oven-dried until constant weight was reached (60°C for the wetland soils, 105°C for the upland ones; Grossman and Reinsch 2002; Bernal and Mitsch 2012), weighed to determine bulk density (Mg m^{-3}), ground to a 2-mm particle size, and homogenized for laboratory analyses.

Soil Nitrogen Analysis

Total N in the soil was measured in a Nitrogen Combustion Analyzer (Elementar America's VarioMax, Hanau, Germany). The soil N concentration (g N kg^{-1}) and pool (kg N m^{-2}) of each core were calculated following the equations described in Bernal and Mitsch (2008, 2012):

$$\text{N concentration (g N kg}^{-1}\text{)} = 10 \times \text{TN}(\%) \quad (1)$$

$$\text{TN}(\text{g}) = W(\text{g}) \times \text{TN}(\%) \times 10^{-2} \quad (2)$$

$$\text{N pool}(\text{kg N m}^{-2}\text{)} = 10^{-3} \times \text{TN}(\text{g}) \times A^{-1}(\text{m}^2) \quad (3)$$

TN = total soil N; W = dry weight of soil in g; and A = area of the sediment sampler in m^2 .

Inorganic soil N content ($\mu\text{g NO}_3\text{-N g}^{-1}$ and $\mu\text{g NH}_4\text{-N g}^{-1}$) was determined with a 2 M KCl extraction and a flow injection analysis (Mulvaney 1996). Organic soil N content was calculated by subtracting inorganic N from total N (Bernal and Mitsch 2013). From the C concentration (g C kg^{-1}) measured in these same soil samples and reported in Bernal and Mitsch (2013), soil C:N ratios of each wetland, section, and community were determined by dividing their total carbon by their total nitrogen.

We determined the total soil N and sediment pools of the accretion layer (i.e., layer of hydric soil developed over the nonhydric parent material; Table 1) in each point of the grid and calculated

Table 1. Summary of Physiochemical Conditions, Pools, and Rates of the Planted and Naturally Colonized Wetlands and Their Wetland Communities (Average of Both Wetlands) at the Olentangy River Wetland Research Park, 15 Years after Their Creation

Parameter	Planted wetland ^a (n = 17)	Colonized wetland ^a (n = 19)	Wetland community		
			Open water (n = 12)	Emergent (n = 15)	Edge (n = 9)
Physiochemistry					
Bulk density, Mg m^{-3}	0.63 ± 0.02	0.71 ± 0.04	0.59 ± 0.04	0.67 ± 0.03	0.75 ± 0.06
Hydric soil depth, cm	14.6 ± 1.3	13.2 ± 1.2	17.9 ± 1.7	12.0 ± 0.8	11.7 ± 0.8
Total N content, g N kg^{-1}	3.1 ± 0.1 ^A	3.2 ± 0.1 ^B	3.1 ± 0.2	3.2 ± 0.3	3.2 ± 0.3
NH ₄ -N content, $\mu\text{g g}^{-1}$	45.1 ± 3.9	37.0 ± 5.1	60.2 ± 7.0 ^A	31.2 ± 4.3 ^B	33.9 ± 5.1 ^B
NO ₃ -N content, $\mu\text{g g}^{-1}$	2.1 ± 0.2 ^A	3.0 ± 0.2 ^B	2.4 ± 0.3	2.6 ± 0.3	2.7 ± 0.5
Organic N: Total N, %	98.5 ± 0.1 ^C	98.8 ± 0.1 ^D	98.1 ± 0.3 ^A	98.9 ± 0.1 ^B	98.9 ± 0.2 ^B
C: N ratio	11.6 ± 0.3 ^C	13.1 ± 0.8 ^D	11.5 ± 0.4 ^A	11.9 ± 0.6 ^A	14.4 ± 1.3 ^B
Pools and rates					
Soil N pool, $\text{kg N m}^{-2\text{b}}$	0.27 ± 0.02	0.31 ± 0.02	0.35 ± 0.03 ^A	0.26 ± 0.02 ^B	0.27 ± 0.02 ^B
Sedimentation rate, $\text{kg m}^{-2} \text{y}^{-1}$	6.0 ± 0.4	5.9 ± 0.4	6.8 ± 0.5	5.3 ± 0.3	5.9 ± 0.6
N accumulation rate, $\text{g m}^{-2} \text{y}^{-1}$	18.0 ± 1.47	21.0 ± 1.38	23.5 ± 2.00 ^A	17.3 ± 1.33 ^B	18.2 ± 1.38 ^B

Note: A and B indicate significant differences at $\alpha \leq 0.05$; C and D indicate significant differences at $\alpha \leq 0.10$.

^aWeighted averages based on corresponding surface area of open water and vegetated communities (emergent and edge).

^bSoil N pools calculated for the average hydric soil depth in each wetland.

their sediment and nitrogen accumulation rates by dividing these pools by the 15 years since the wetland was created. Rates were estimated for each community, section, and wetland. Total wetland averages were weighted based on the surface area of each community (Bernal and Mitsch 2013).

Nitrogen Budget

We posit that soil N in these wetlands had mainly two important sources, (1) the N entering the wetland from the Olentangy River (organic and inorganic N), and (2) the organic N already contained in the vegetation growing in the wetland basins. Nitrogen contained in the water would enter as suspended particles that settle on the soil. Some N also occurs in dissolved forms N within the water column and soil pore space, eventually interacting with the vegetation and the microbial communities. Aboveground plant matter in the wetlands decays at the end of season and contributes to the organic N soil pool. To synthesize these processes, we analyzed annual data compiled since wetland creation on hydrology, water quality, and aboveground net primary productivity (ANPP) to create a simple budget of soil N for the years 10 (2004) and 15 (2009) of these wetlands published by Mitsch et al. (2012, 2014b). We used estimates by Stefanik (2012) on N content of aboveground biomass. We used soil N accumulation rates for year 10 (2004) from Anderson and Mitsch (2006) and soil N data obtained in this study for year 15 (2009) to estimate N accumulation in the wetland soil. We used reported denitrification rates measured in year 10 by Hernandez and Mitsch (2007) and in year 15 by Song et al. (2014) and Batson et al. (2012) to complete the budget.

Data Analyses

Data were organized to compare N dynamics between wetlands (planted wetland and naturally colonized wetland); however, wetlands data were pooled to examine potential differences in community (open water, emergent, and edge). Statistical analyses were performed with IBM SPSS Statistics version 19.0 for Macintosh. In every data set we did normality checks (with Q-Q plot, Kolmogorov–Smirnov test, and Shapiro–Wilk test) and tested the homogeneity of variances with Levene Statistic. With a statistical analysis of variance (ANOVA) and Tukey HSD we tested the effect of section and community (independent variables) in the soil N pool as well as in the soil N accumulation rates (dependent variables), and determined the significance of differences in N accumulation between the two wetlands. Nitrogen and carbon content in the soil and in the water was examined with Pearson product moment correlations to assess the potential relationship between carbon and nitrogen soil pools, and between dissolved carbon and nitrogen water concentrations. In all statistical analyses, significant differences were determined at $P < 0.05$.

Results

Soil and Water Nitrogen Content

Total soil nitrogen content was significantly different in both wetlands ($P < 0.05$), even though their weighted average was similar ($3.1 \pm 0.1 \text{ mg N g}^{-1}$ in the planted wetland and $3.2 \pm 0.1 \text{ g N kg}^{-1}$ in the naturally colonized wetland; Table 1). It increased significantly ($P < 0.01$) from the inflow to outflow in both wetlands [Fig. 2(a)], but no differences were found when comparing between communities (Table 1). The total N pool was similar in both

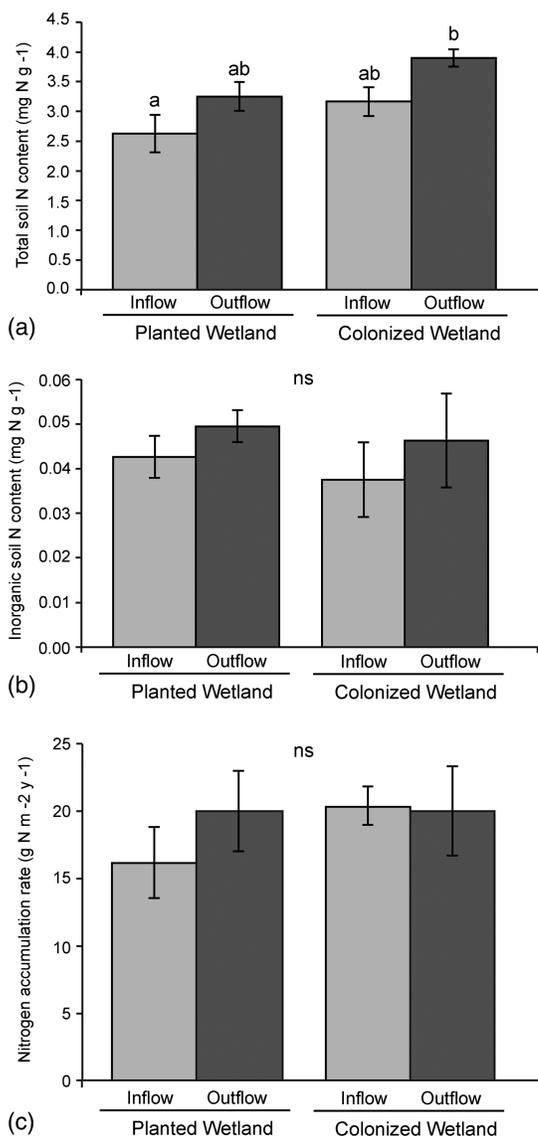


Fig. 2. Comparison of (a) total soil nitrogen content (mg N g soil^{-1}); (b) total soil inorganic nitrogen content (mg N g soil^{-1}); (c) soil nitrogen accumulation rate ($\text{g N m}^{-2} \text{ y}^{-1}$) inflows versus outflows for the planted and naturally colonizing experimental wetlands at the Olentangy River Wetland Research Park; bars represent SE and different letters indicate significant differences at $\alpha < 0.01$

wetlands, being highest in the open water ($0.31 \pm 0.02 \text{ kg N m}^{-2}$) and lowest in the emergent areas ($0.25 \pm 0.02 \text{ kg N m}^{-2}$). The total N in the soil was mostly organic ($\sim 99\%$), and $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ increased from inflow to outflow as well, although $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ contents were not significantly different between these sections [Fig. 2(b)]. Soil $\text{NO}_3\text{-N}$ was significantly higher ($P < 0.001$) in the colonized wetland, whereas soil $\text{NH}_4\text{-N}$ did not differ between wetlands. On the contrary, no differences were found on $\text{NO}_3\text{-N}$ content between wetland communities, but $\text{NH}_4\text{-N}$ was significantly higher in the open water ($P < 0.001$; Table 1). Soil $\text{NO}_3\text{-N}$ was consistently higher in every wetland community of the colonized wetland, whereas $\text{NH}_4\text{-N}$ was always higher in the planted wetland.

Soil C:N ratios were similar in both wetlands (12.4 ± 0.5 , on average), although it was consistently higher in the colonized one (Table 1). The edge communities had the highest ratio (14.4 ± 1.3),

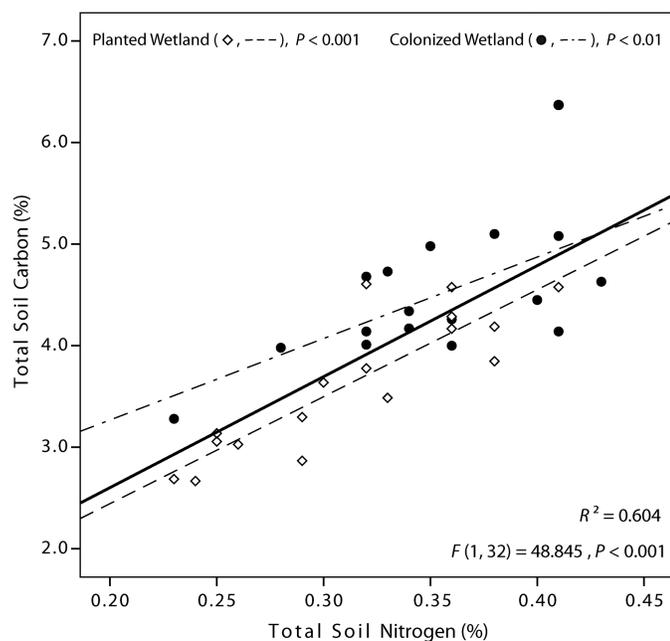


Fig. 3. Total carbon content (%) versus total nitrogen content (%) in the soil of the two created wetlands at the Olentangy River Wetland Research Park, with significance of the Pearson correlation (F (df) and P values) and linear R^2 ; solid line indicates the average trend of both wetlands (corresponding to the noted F , P , and R^2 values); individual planted and naturally colonized wetland percentages, trend, and significance of the correlation are also indicated

whereas the open water sites had the lowest (11.5 ± 0.4). The total soil N was correlated with the total soil carbon in these two wetlands ($P < 0.001$, $R^2 = 0.604$; Fig. 3). Regarding the relationship of soil carbon and nitrogen in each wetland individually, the correlation was still significant ($P < 0.001$ in the planted wetland; $P < 0.01$ in the colonized wetland) but the linear regression was stronger in planted wetland ($R^2 = 0.753$) than in the colonized one ($R^2 = 0.384$).

Based on reported data from Mitsch et al. (2012, 2014b), the total N dissolved in the water column at 15 years since wetland creation was uncorrelated to the total carbon dissolved in the water column ($P = 0.79$; $R^2 = 0.002$). Total dissolved N loads remained significantly higher ($P < 0.01$) in the inflows ($118 \text{ g N m}^{-2} \text{ y}^{-1}$, on average in the two wetlands) than in the outflows ($59 \text{ g N m}^{-2} \text{ y}^{-1}$ on average in the two wetlands, about half of the inflow concentration). Productivity (ANPP), however, increased from inflow to outflow in the planted wetland and decreased from inflow to

outflow in the colonized one being highest ($P < 0.01$) in the inflow of the colonized wetland ($832 \text{ g dry biomass m}^{-2} \text{ y}^{-1}$).

Sedimentation and Nitrogen Accumulation Rates

Both wetlands had similar sedimentation rates ($6.0 \pm 0.4 \text{ kg m}^{-2} \text{ y}^{-1}$ in the planted wetland, $5.9 \pm 0.4 \text{ kg m}^{-2} \text{ y}^{-1}$ in the colonized wetland; Table 1). Over these 15 years, the open water sites had the deepest accretion layer ($17.9 \pm 1.7 \text{ cm}$; Table 1), yielding a greater sedimentation rate than the vegetated areas ($6.6 \pm 0.5 \text{ kg m}^{-2} \text{ y}^{-1}$).

On average, nitrogen accumulation rates were similar in both wetlands ($18.01 \pm 1.47 \text{ g N m}^{-2} \text{ y}^{-1}$ in the planted wetland, $20.95 \pm 1.38 \text{ g N m}^{-2} \text{ y}^{-1}$ in the colonized wetland; Table 1). No significant differences were found between the inflow and outflow sections [Fig. 2(c)], but accumulation rates differed ($P < 0.05$) between the open water and the vegetated communities (23.45 ± 2.00 and $17.74 \pm 1.36 \text{ g N m}^{-2} \text{ y}^{-1}$, respectively; Table 1).

The thickness and density of the accreted hydric soil increased over time (Table 2), thereby increasing the soil nitrogen pool (almost doubling it, from 0.15 kg N m^{-2} when the wetlands were 10 years old to 0.29 kg N m^{-2} when they were 15 years old), even though the average soil nitrogen content decreased slightly (9%) between those years. Consequently, the nitrogen accumulation rate increased over time as well, from 16.4 to $19.5 \text{ g N m}^{-2} \text{ y}^{-1}$.

Discussion

Nitrogen Dynamics in Created Wetlands

Results from this study showed that N pools and accumulation in both experimental wetlands have been similar; however, there were some notable differences between them. Organic matter accumulation was clearly an important mechanism for N accumulation (most of the soil N was organic) and may explain some of the differences that were observed between wetlands. The planted wetland has a history of greater plant diversity and lower ANPP over the 15-year period. These differences may have contributed to some of the detected differences in N measures between the two wetlands. For instance, the colonized wetland has regularly maintained a higher cover of cattails (combined *Typha angustifolia*, *T. latifolia* and *T. x glauca*) than the planted wetland (Mitsch et al. 2012). Cattails, being a highly productive and competitive species, have the capacity to generate more detritus after taking up and retaining more N in their biomass than most of the other macrophyte species common to the ORWRP. In a wetland mesocosm experiment under steady flow conditions, average N-biomass content (combined aboveground and belowground) for *T. angustifolia*

Table 2. Comparison of Wetland Soil Physiochemical Conditions at 10 and 15 Years after Creation (Average of Both Wetlands), Including the Percentage Change between Both Periods, and Values of the Reference Natural Wetland and Reference Upland Adjacent to the Created Wetlands

Site	Bulk density (Mg m^{-3})	Soil accretion (cm)	Total nitrogen (g N kg^{-1} soil)	Nitrogen pool ^a (kg N m^{-2})	Nitrogen accumulation rate ($\text{g N m}^{-2} \text{ y}^{-1}$)	Reference
Created wetlands						
10 years since creation (1994–2004)	0.5	9	3.4	0.15	16.4	Anderson and Mitsch 2006
15 years since creation (1994–2009)	0.7	14	3.1	0.29	19.5	Bernal and Mitsch 2013; this study
Change (2004–2009)	+37%	+55%	−9%	+93%	+19%	Bernal and Mitsch 2013; this study
Reference natural wetland	0.8	—	2.8	0.34	7	Bernal and Mitsch 2012; this study
Reference upland	1.2	—	2.4	0.43	—	This study

^aNitrogen pool to average hydric soil depth in created wetlands, to 35 cm in reference natural wetland, and to 15 cm in reference upland.

was $28.4 \pm 3.3 \text{ g N m}^{-2}$ but only $16.3 \pm 1.3 \text{ g N m}^{-2}$ for *Schoenoplectus tabernaemontani* (another common macrophyte in the ORWRP) while controlling for N inflow (Anderson and Mitsch 2005). Species type may also explain other differences between wetlands such as C:N ratios (Windham 2001) which were significantly higher in the colonized wetland compared to the planted wetland. Differences in vegetation composition have been shown to elicit differences in related wetland soil processes such as decomposition and recalcitrance of organic matter (Enriquez et al. 1993; Windham 2001), due to the inputs of fresh labile C through below-ground root exudates and surface litter accumulation (Fontaine et al. 2003; Kuzyakov 2010; Schmidt et al. 2011).

Variation in productivity may also cause differences in physical filtration that affect sediment accretion and organic matter accumulation. Evidence from previous ORWRP research and elsewhere (Anderson and Mitsch 2006; Darke and Magonigal 2003) has shown that dense recruitment of macrophytes may result in preferential flows through a wetland and ultimately contribute to variation in sediment deposition patterns. Not surprisingly, we found significant spatial variation related to N pools and processes across the experimental wetlands. Elevation and relative water depth determine the various community types (defined by vegetation type and hydrology, i.e., open water, emergent, and edge) and are an important spatial aspect detected at the ORWRP wetlands. After 15 years, the greatest accumulation of soil N was located in the open water sections of the wetland, which were significantly higher than the emergent and edge communities. These deep-water soils are often composed of deeper sediment accumulation and composed of finer organic material resembling the formation of a gyttja layer (Wetzel 2001). This layer is likely the result of deposition of more algae and suspended plant detritus, the main source of organic matter to the soil because the water level is high enough to keep emergent vegetation to settle in this wetland area. The emergent and edge communities, on the contrary, had abundant emergent macrophytes established over these 15 years, and thus autochthonous organic matter is likely the greatest contributor to organic matter and associated N accumulation in these communities.

Soil $\text{NH}_4\text{-N}$ content in the open water zone was typically twice as high compared to the emergent and edge zone. Higher levels of $\text{NH}_4\text{-N}$ are likely due to more anaerobic soil conditions in the permanently flooded open water communities. These areas are located in the lowest elevations that stay flooded nearly continuously unlike the emergent and edge communities. Variations in flood frequency and its related effects on vegetation recruitment and other processes have been shown to contribute to spatial variation elsewhere. Bai et al. (2005) found clay content, species litter inputs, soil moisture, and water table all contributed to variation in soil organic matter and total N at wetland floodplain soils in China. Further, nitrification of ammonium was likely enhanced by macrophytes (and enhanced oxidation in the rhizosphere) in the emergent and edge communities, thereby reducing soil $\text{NH}_4\text{-N}$ concentrations compared to the open water zones. Others have found that soil N dynamics can be affected by macrophyte species type (Gumbrecht 1993). Windham and Ehrenfeld (2003) found that various N process rates were generally greater in salt marshes where *Phragmites australis* replaced *Spartina patens*. The depth and extent of roots provided by different macrophyte species can also cause variation in the vertical distribution of sediment redox levels and ultimately influence N cycling (Chen and Barko 1988).

It was noteworthy that the upland soil adjacent to our created wetlands did not develop hydric conditions and thus its bulk density was almost twice the average density measured in both wetlands ($1.22 \pm 0.08 \text{ Mg m}^{-3}$ in the upland, $0.67 \pm 0.02 \text{ Mg m}^{-3}$ on average in these wetlands). The total soil N content of the created

wetlands was about 31% higher, on average, than the content of its adjacent upland ($2.4 \pm 0.08 \text{ g N kg}^{-1}$). In the upland area, the lack of standing water maintaining anaerobic conditions kept soil $\text{NH}_4\text{-N}$ content ($9.04 \pm 1.3 \mu\text{g NH}_4\text{-N g}^{-1}$) low in comparison to the $\text{NH}_4\text{-N}$ content measured in the created wetlands ($41.1 \mu\text{g NH}_4\text{-N g}^{-1}$, on average).

Other variations in soil N were related to proximity to the wetland inflows. The N contained in soils increased toward the outflow for both wetlands, opposite to the N contained in the water. These two wetlands have been effective removers of N in the water (Mitsch et al. 2005b, 2012), mainly by means of organic accumulation but also denitrification (Hernandez and Mitsch 2007; Song et al. 2014). Hernandez and Mitsch (2007) reported that in permanently flooded areas of the wetlands (deeper emergent and open water zones) there was a trend of greater denitrification near the inflow ($>600 \mu\text{g N m}^{-2} \text{ h}^{-1}$) than in the comparable zones near the wetland outflows ($<400 \mu\text{g N m}^{-2} \text{ h}^{-1}$). This trend has been supported by results that found greater amounts of dissolved N in soils near the inflow compared to the outflow (Mitsch et al. 2012, 2014b).

Being mostly organic, part of this N is likely coming from the organic material that is entering the ecosystem in the water from the adjacent eutrophic river and then, for the most part, produced within the wetland (as plant biomass). To further explore this point we compared the C:N ratios in both wetlands' soil (Fig. 3) and water. The water carbon and nitrogen contents showed no relationship ($P = 0.79$, linear $R^2 = 0.002$). In the soil, however, the C:N ratios remain somewhat constant in both wetlands (Table 1), and soil C and N are strongly correlated in both basins [Fig. 2(a)]. The correlation is not as strong in the colonized compared to the planted wetland, probably because the range of total soil carbon in the colonized wetland was narrower and thus the values are more consistent. This C:N trend suggests that soil carbon and nitrogen are bound and change together despite the different conditions that these wetlands present. The soil C:N could have responded to differences in plant productivity between both wetlands because vegetation can alter decomposition patterns in the soil and, depending on soil organic matter recalcitrance, mineralize differently carbon and nitrogen compounds (Hobbie 2005; Castellano et al. 2015). Carbon and nitrogen can be related in the soil organic matter in the form of labile organic compounds (C:N of 20–30 or higher; Stevenson and Cole 1999; Lehmann and Kleber 2015) that remain undecomposed due to the anaerobic conditions, or as stable or resilient organic material (C:N of 10–12; Stevenson and Cole 1999). Our colonized wetland, having higher soil C:N ratios than the planted basin, could have more labile organic compounds due to its greater biomass productivity (Castellano et al. 2015).

Nitrogen Dynamics over Time

We developed N budgets for years 10 and 15 (Fig. 4). Nitrogen input as plant biomass (NPP) doubled in five years (average increase from 7.2 to $14.4 \text{ g N m}^{-2} \text{ y}^{-1}$ at wetland ages of 10 and 15 years, respectively). Consequently, nitrogen accumulation in the soil increased as well in this period of time, from 16.4 to $19.5 \text{ g N m}^{-2} \text{ y}^{-1}$. Losses through denitrification increased slightly from 2.3 to $2.6 \text{ g N m}^{-2} \text{ y}^{-1}$, suggesting that an increase in organic matter accumulation from higher NPP might have also increased the availability of carbon substrates necessary for denitrification. Total N in the surface inflow was essentially identical in years 10 and 15 ($120 \text{ g N m}^{-2} \text{ y}^{-1}$). Total N outflow, including surface flow and subsurface seepage, has been fairly constant (88 and $89 \text{ g N m}^{-2} \text{ y}^{-1}$ in year 10 and 15, respectively), but the proportion of outflowing N through seepage relative to surface outflow was

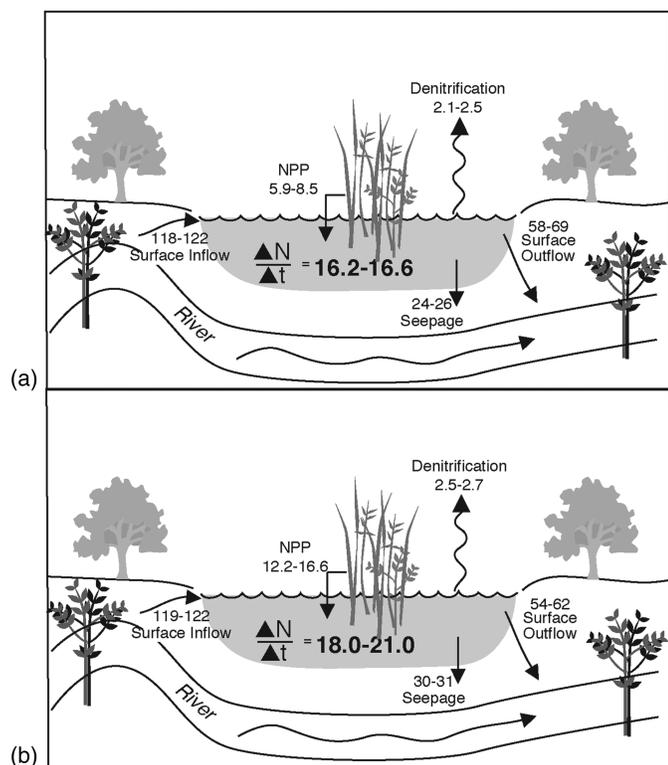


Fig. 4. Soil nitrogen budgets expressed in $\text{g N m}^{-2} \text{y}^{-1}$ for the two experimental wetlands at the Olentangy River Wetland Research Park in central Ohio for (a) 2004 when created wetlands were 10 years old; (b) 2009 when created wetlands were 15 years old; data ranges represent range of rates in two created wetlands; hydrologic fluxes include pumped surface inflow, surface outflow, and seepage (illustration adapted from Mitsch and Gosselink 2015; data from Anderson and Mitsch 2006; Mitsch et al. 2012, 2014b; Batson et al. 2012; Song et al. 2014)

higher at year 15 (35% of the total outflow) than at year 10 (28% of the total outflow).

A 19% increase in soil N retention and a doubling of N input through plant biomass in these wetlands (with a modest increase in denitrification) suggest that their N retention will continue to increase in the foreseeable future. The unaccounted for N in our budgets was only 10.4% in year 10 and 8.5% in year 15. Several authors have pointed out that denitrification methods often underestimate gaseous nitrogen fluxes from wetlands (e.g., Groffman et al. 2006; Song et al. 2014), which could account for our unaccounted N in these budgets.

Challenges for Wetland Modeling and Application

It is widely recognized that wetlands are critical components of the landscape for clean water, flood abatement, biodiversity, and a host of other services (Mitsch et al. 2005b, 2015). However, it is becoming necessary for managers to not only demonstrate water quality improvement is provided but to quantify these benefits. Determining the extent that wetlands reduce N or other pollutants is increasingly being conducted through the application of models. One challenge of developing effective wetland models continues to be the availability of reliable monitoring data that can better account for variations over space and time. The collection of multi-parameter water and soil data, particularly over the long term, is uncommon but provides the best information for the development

and refinement of robust wetland models. Long-term monitoring and research data is an important contribution to the scientific community, as it ultimately contributes to the development of and improvement of existing predictive models of nutrient dynamics.

Models are increasingly being designed to incorporate the spatial variability that inevitably occurs (Martin and Reddy 1997; Rousseau et al. 2004; Sharifi et al. 2013). Our results and others from the ORWRP underscore the complexity of processes related to N, which varies significantly in form and concentration across gradients of topography (i.e., community type) and source (i.e., inflow versus outflow). This variation over space has been demonstrated elsewhere. For example, Brueske and Barrett (1994) described variation in sediment deposition as it related to wetland community type and proximity to inflow at the Des Plaines River Wetland Demonstration Project in Illinois, U.S. Their results showed that the influence of these wetland features were important at low hydrologic loading rate (6 cm/wk) but decreased at a higher hydrologic loading rates (50 cm/wk). These and other monitoring data can aid managers and be incorporated into efforts to model important wetland processes such as at the Des Plaines River wetlands (Christensen et al. 1994). Careful consideration is needed when using data for developing models. Likewise, suitable model application must consider the origin of the data. For example, comparing the Ohio created wetlands in this study to a natural reference riverine wetland in the region (as described in Bernal and Mitsch 2012), nitrogen accumulation rates of the created wetlands ($20 \text{ g N m}^{-2} \text{y}^{-1}$) were more than double the rates in the natural reference wetland ($7 \text{ g N m}^{-2} \text{y}^{-1}$). This difference between sites likely reflects the higher accumulation of organic and sediment accretion that has occurred for at least 15 years in the created wetlands. Despite the quick organic matter accretion of these young wetland soils, it is noteworthy that their total and inorganic N content (3.1 g N kg^{-1} and $41.05 \pm 4.45 \mu\text{g NH}_4\text{-N g}^{-1}$) appears to be trending toward those measured at the Ohio reference natural wetland (2.8 g N kg^{-1} and $34.80 \pm 5.90 \mu\text{g NH}_4\text{-N g}^{-1}$).

Our study and previous research published from these created wetlands (Mitsch et al. 1998, 2005b, 2012, 2014b) demonstrates that nitrogen cycling and other related processes have continually changed over the 15 years since construction. This type of change is rarely demonstrated because of the unusual duration of monitoring and research that is ongoing at the ORWRP. Additionally, various measures of N flux also need to consider intra-annual variability because estimates of N cycling can vary depending on season, vegetation community, and wetland hydrology. For instance, Batson et al. (2012) detected peaks in denitrification during the spring and summer months compared to autumn and winter related to soil temperature and nitrate availability. Likewise, differences between the planted and colonized wetlands at the ORWRP indicate that there are still important differences in carbon and nutrient cycling that may be traced to the historical differences in vegetation types. Seasonality, successional changes, and ecological maturity are difficult attributes to predict but are changes that will affect wetland biogeochemistry and must be considered by modelers seeking to predict future conditions.

Physically based wetland models have become increasingly sophisticated and more capable of providing reliable predictions of current and future conditions. Model development complemented by comprehensive research and monitoring has the potential to greatly improve our understanding of important wetland biogeochemical processes while identifying sensitive and critical pathways. Research and monitoring can improve modelling efforts by accounting for variations that occur over space and time. For example, ecological model performance often suffers during extreme conditions or disturbance (e.g., droughts, muskrat activity in

wetlands); however, with longer-term monitoring data these types of events are more likely to be documented and better evaluated relative to baseline conditions (Cirmo and McDonnell 1997). Similarly, wetlands and their effect on water quality represent a combination biological, chemical, and physical processes that may vary seasonally. Understanding seasonal variation is necessary when wetlands are considered as part of a policy strategy to reduce N runoff from agricultural regions (Mitsch et al. 2001). Results from the ORWRP along with other long-term wetland research and monitoring efforts (Vepraskas et al. 2004; Vymazal and Březinová 2014) provide a better understanding of important biogeochemical functions, demonstrating the spatial and temporal variability and ultimately improving the forecasting capabilities of wetland models. Support and expansion of such efforts should be promoted as part of a national strategy for improved management of water resources in the United States and worldwide.

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