

# Hydrologic Connectivity to Streams Increases Nitrogen and Phosphorus Inputs and Cycling in Soils of Created and Natural Floodplain Wetlands

Kristin L. Wolf, Gregory B. Noe,\* and Changwoo Ahn

Greater connectivity to stream surface water may result in greater inputs of allochthonous nutrients that could stimulate internal nitrogen (N) and phosphorus (P) cycling in natural, restored, and created riparian wetlands. This study investigated the effects of hydrologic connectivity to stream water on soil nutrient fluxes in plots ( $n = 20$ ) located among four created and two natural freshwater wetlands of varying hydrology in the Piedmont physiographic province of Virginia. Surface water was slightly deeper; hydrologic inputs of sediment, sediment-N, and ammonium were greater; and soil net ammonification, N mineralization, and N turnover were greater in plots with stream water classified as their primary water source compared with plots with precipitation or groundwater as their primary water source. Soil water-filled pore space, inputs of nitrate, and soil net nitrification, P mineralization, and denitrification enzyme activity (DEA) were similar among plots. Soil ammonification, N mineralization, and N turnover rates increased with the loading rate of ammonium to the soil surface. Phosphorus mineralization and ammonification also increased with sedimentation and sediment-N loading rate. Nitrification flux and DEA were positively associated in these wetlands. In conclusion, hydrologic connectivity to stream water increased allochthonous inputs that stimulated soil N and P cycling and that likely led to greater retention of sediment and nutrients in created and natural wetlands. Our findings suggest that wetland creation and restoration projects should be designed to allow connectivity with stream water if the goal is to optimize the function of water quality improvement in a watershed.

**H**YDROLOGY IS the essential determinant of structure and function in wetland ecosystems, including biogeochemical cycling. The connection of a wetland to the surrounding hydrologic system influences the timing, amplitude, duration, and frequency of flooding, which in turn create the biogeochemical character of the wetland by influencing the aerobic/anaerobic biogeochemistry of the soil and the importation, cycling, and storage of nutrients (Tockner et al., 1999; Hein et al., 1999; Craft and Casey, 2000; Noe et al., 2013a). Wetlands hydrologically connected to streams retain nitrogen (N) and phosphorus (P) and thereby provide a watershed with the vital ecosystem service of water quality improvement (Verhoeven et al., 2006). Consequently, wetland creation and restoration have often been proposed as a management option to remove nutrients at the landscape scale (Lowrance et al., 1997; Mitsch and Day, 2006). Improvement of water quality is a particularly important function for wetlands in the Chesapeake Bay watershed, where elevated loading of N, P, and sediment is a primary cause of its impaired status (Lowrance et al., 1997). If wetlands are isolated from the surrounding hydroscape, they cannot effectively intercept and filter incoming water, and the opportunity for flood protection and improved water quality in the watershed of the wetland is missed (Hey et al., 2012).

Nutrient cycling in wetlands is controlled by hydrology and hydrologic inputs. The flood-pulse dynamic that characterizes hydrologically connected wetlands provides an important source of allochthonous material that may stimulate N and P cycles by contributing additional mineralizable organic substrate (Junk et al., 1989; Odum et al., 1995; Nahlik and Mitsch, 2008; Noe et al., 2013a),  $\text{NO}_3^-$  for denitrification (Pinay et al., 2002; Richardson et al., 2004; Forshay and Stanley, 2005), and carbon to enhance the metabolism of heterotrophic bacteria (Robertson et al., 1999). The lack of hydrologic connectivity to stream water in natural wetlands has been shown to limit nutrient accumulation associated with sedimentation (Craft and Casey, 2000; Noe

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**Abbreviations:** CF, Clifton Farm; BFP, Manassas National Battlefield Park; BR, Bull Run Mitigation Bank; BSR, Banshee Reeks Nature Preserve; DEA, denitrification enzyme assay; LC, Loudoun County Mitigation Bank; NF, North Fork Wetlands Bank; PGW, precipitation or groundwater as the primary water source; SRP, soluble reactive phosphorus; SW, stream water as the primary water source; WFPS, water-filled pore space.

and Hupp, 2005) and to limit primary production (Bayley and Guimond, 2009). The closed material cycles of hydrologically disconnected wetlands, created or natural, thus lowers nutrient inputs and the sequestration/removal potential of the wetland in general (Hopkinson, 1992).

Furthermore, hydrologic connectivity influences soil water content and reduction-oxidation (redox), which affects nutrient cycling in wetland soils. Fluctuating redox conditions are required for microbes to cycle N completely from its organic to inorganic form through N mineralization (comprised of ammonification and nitrification processes) and for its removal as N<sub>2</sub> gas by denitrification (Reddy and Patrick, 1984; Hefting et al., 2004). Although ammonification can occur under aerobic and anaerobic conditions (it is expedited under the former) (Ponnamperuma, 1972; Brinson et al., 1981), nitrification and denitrification require aerobic and anaerobic conditions, respectively (Ponnamperuma, 1972). A fluctuating hydroperiod with a subsequent fluctuation in redox potential favors nitrification during drawdown periods, which produces nitrate (NO<sub>3</sub><sup>-</sup>) that can later be denitrified to N<sub>2</sub> during flooded periods (Reddy and Patrick, 1984). Net P mineralization in most wetland soils is the result of microbial degradation of organic P and microbial reduction of metal complexes that desorb orthophosphate, and both processes are strongly regulated by fluctuating redox (Reddy and DeLaune, 2008; Noe et al., 2013a).

Despite the essential value of hydrologic connectivity in transporting energy and materials and in providing essential ecosystem services, this characteristic is often overlooked when ecosystems are anthropogenically altered (Pringle, 2003). This is often a design flaw in the creation and restoration of wetland systems, notably in cases where a mitigation wetland is not the hydrologic equivalent of the replaced natural wetland at the landscape scale (Bedford, 1996). Wetland mitigation often results in a net loss of wetland function when improperly performed (Kentula et al., 1992; Zedler, 1996; Hoeltje and Cole, 2007). Many created wetlands incorporate a surface-driven, epiaurated “perching design” that relies on the compaction of clayey substrate and periphery berms to control the presence of water at or near the surface for a required number of days during the growing season (Whittecar and Daniels, 1999). The result is a wetland that is disconnected from groundwater exchange, has limited surface water exchange, has muted temporal variability in water stage, and is generally far wetter than natural wetlands (Brooks et al., 2005; Hoeltje and Cole, 2007). The lack of hydrologic connectivity and flood pulses in created and restored wetlands has also been shown to depress denitrification (Hunter and Faulkner, 2001; Hernandez and Mitsch, 2007), which limits the ability of the wetland to remove excess N from the landscape.

This study investigated the impact of hydrology and of hydrologic inputs of nutrients and sediment on rates of nutrient cycling among and within several created and natural floodplain wetlands. We first compare the hydrology, inputs of sediment and nutrients, and nutrient cycling rates of wetland plots with stream water classified as the primary water source with those with precipitation or groundwater as their primary water source. We then relate hydrology and sediment and nutrient inputs to nutrient cycling rates to identify the specific mechanisms determining nutrient biogeochemistry in these wetlands. The measured nutrient cycling rates include soil ammonification

and nitrification (net N mineralization), P mineralization, and denitrification potential. We hypothesized that the metrics describing gradients of greater hydrologic connectivity to stream water would be positively associated with each of the measured nutrient cycling rates in the wetlands by providing a subsidy of allochthonous material to the wetland.

## Materials and Methods

### Site Descriptions

#### General Setting

Four created and two natural floodplain riparian wetlands were chosen for study. These wetlands represent a range of hydrogeomorphic settings and hydrologic connectivity to surrounding hydrospheres that is typical of the Piedmont in the mid-Atlantic region of the United States (authors, personal observation). All study sites were nontidal freshwater wetlands located in northern Virginia (mean annual precipitation, 109 cm; mean temperature, min. 7°C/max. 18°C) (Fig. 1). The created mitigation wetland banks were constructed on old farm fields and cattle pasturelands and were created to mitigate for various local construction projects that affected a mixture of bottomland forested floodplain, shrub/scrub, and emergent wetlands and open water ponds. The natural wetlands are located on floodplains with varying water sources and hydrologic connectivity. The vegetation of all plots was mostly herbaceous with nearby interspersed young (created wetlands) or mature trees (natural wetland at Manassas National Battlefield Park), with the exception of the natural wetland at Banshee Reeks Nature Preserve, which was mature bottomland forest with little understory. In addition to sampling hydrologic variation among the six wetlands, multiple plots were selected within each wetland to represent the range of hydrology within each site. There were a total of 16 1-m<sup>2</sup> plots in four created wetlands and four plots in two natural wetlands (20 plots in all). Thus, sampled locations included variation in hydrology among and within wetlands. The primary water source for each wetland was classified a priori as stream (referred to as SW plots) or not stream (precipitation, groundwater, or local surface runoff; referred to as PGW plots) based on created wetland design and field evaluation of hydrogeomorphology.

#### Created Wetlands

Loudoun County Mitigation Bank (LC) is a 12.9-ha wetland and upland buffer complex that was constructed in the summer of 2006 in Loudoun County, Virginia (39°1'58.98" N; 77°36'26.10" W). The site is located on the floodplain of Big Branch Creek and Goose Creek. The wetland contains two contiguous areas (Cell 1 and Cell 2) that are separated by a central berm. Four sampling plots were located in Cell 2, which receives flow from a tributary of Goose Creek through a head race attached to a cross vane structure with flow regulated by a culvert structure and gate valve. Two sampling plots were located in Cell 1, which receives water from Cell 2 through a central ditch. Loudoun County Mitigation Bank also receives limited surface water runoff from an upland housing development and forested buffer and from minor groundwater inputs from toe-slope intercept seepage. All plots were classified as SW with stream water as the primary water source.

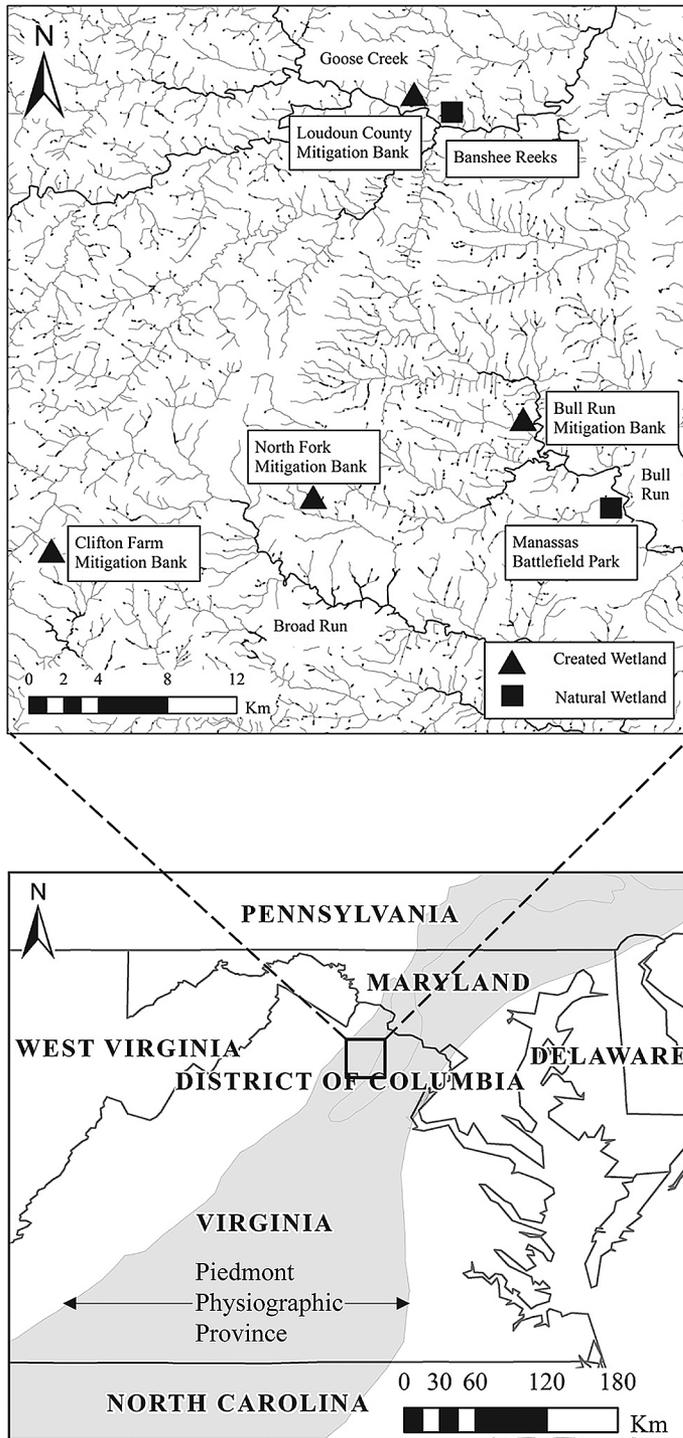


Fig. 1. Site map of wetland locations in the Virginia Piedmont.

Clifton Farm (CF) is a 0.9-ha mitigation wetland that was constructed in 2005 in Fauquier County, Virginia ( $38^{\circ}46'38.75''$  N;  $77^{\circ}47'40.61''$  W). The site receives groundwater from a small upland reservoir and surface water runoff but has no stream connection. Two sampling plots were located in the central area of the wetland. All plots were classified as PGW with stream water not being the primary water source.

Bull Run Mitigation Bank (BR) is a 20.2-ha wetland and upland buffer complex that was constructed in 2002 in Prince William County, Virginia ( $38^{\circ}51'12.74''$  N;  $77^{\circ}32'58.52''$  W).

The site receives water from Bull Run from a culvert structure that routes water via a central ditch through the wetland and from overbank flow from Bull Run through a crevasse that formed through the berm at the corner of the site. The wetland receives limited surface water runoff from uplands and negligible groundwater. Four sampling plots were located along a transect oriented from adjacent to the crevasse to the interior of the wetland. All plots were classified as SW with stream water as the primary water source.

North Fork Wetlands Bank (NF) is a 50.6-ha wetland that was constructed in 1999 in Prince William County, Virginia ( $38^{\circ}49'31.53''$  N;  $77^{\circ}40'9.17''$  W). With the exception of minor contributions from toe-slope intercept seepage, the site is disconnected from the groundwater by an underlying clay liner and was sampled in two hydrologically distinct areas. Two sampling plots were located in the main pod area, which is fed predominantly by a tributary of the North Fork of Broad Run that is controlled by an artificial dam; these plots are classified as SW with stream water as the primary water source. Two sampling plots were located in the vernal pool area, which is located in the southwest quadrant of the wetland and fed solely by precipitation; these plots are classified as PGW with stream water not being the primary water source.

The LC, BR, and NF wetlands contain at least a 0.3-m low permeability subsoil layer covered with 0.2 m of commercially available topsoil with low organic content. This design creates a perched, surface-driven water table close to the soil surface and limits groundwater exchange in the wetland (Ahn and Peralta, 2009).

#### Natural Wetlands

Manassas National Battlefield Park (BFP), established in 1940, is a 2000-ha site with areas of natural wetland coverage located in Prince William County, Virginia ( $38^{\circ}49'24.98''$  N;  $77^{\circ}30'28.30''$  W). This floodplain site is connected to Bull Run by a culvert on its eastern end and receives limited upland surface water runoff. Two sampling plots in a low elevation area of herbaceous wetland within a matrix of forested floodplain were selected for study. All plots were classified as SW with stream water as the primary water source.

Banshee Reeks Nature Preserve (BSR), established 1999, is a 290-ha site with areas of natural seep and riparian wetlands located in Loudoun County, Virginia ( $39^{\circ}1'16.44''$  N;  $77^{\circ}35'49.10''$  W). Two sampling plots were located in floodplain wetlands that occasionally receive water from groundwater springs, surface water runoff, and rare overbank flooding from Goose Creek. All plots were classified as PGW with stream water not being the primary water source.

## Measurements

### Hydrology and Hydrologic Inputs

Monthly measurements of hydrology, hydrologic inputs, and biogeochemical flux rates in each plot occurred over a 2-d period during the second week of every month from July 2008 to July 2009. The depth of standing surface water, when present, was measured each month within each plot. Soil redox potential was measured each month by inserting a RE 300 ExStik ORP meter (Extech Instruments Corp.) to a depth of approximately 3 cm into

the soil between the initial core and resin core sampling locations for measuring mineralization (see below). Redox potential was recorded after drift was sufficiently stabilized ( $\sim 1$  min). Ceramic sedimentation tiles ( $20 \times 20$  cm) were installed monthly at a flat location in each plot. Deposited sediment (excluding coarse woody debris and litter fall free of mineral sediment) was harvested from the tiles after each collection period, oven dried, and weighed to determine monthly accumulation. Samples were ground, subsampled, and analyzed for total N concentration on an elemental analyzer (Nelson and Sommers, 1996). Total P content of the sediment was not measured due to logistical constraints. Cumulative annual sediment and N deposition rates were calculated for each plot.

Soil moisture content was determined for each initial core and resin core by removing a  $\sim 40$  g dry-weight equivalent subsample of homogenized soil, recording initial field-moist weight, and drying at  $60^\circ\text{C}$  until a constant weight was achieved. Soil water-filled pore space (WFPS), the proportion of total soil pore space occupied by water, was calculated as:

$$\text{WFPS} = \frac{\text{BD}(\text{GWC}/1.0 \text{ g mL}^{-1})}{(1 - \text{BD}/2.65 \text{ g cm}^{-3})}$$

Gravimetric water content was calculated as mass of water divided by mass of dry soil. Bulk density was determined for each core by weighing the entire field-moist core of soil, converting to dry weight based its gravimetric water content, and dividing by the total volume of the soil in the core ( $238.9 \text{ cm}^3$ ). The particle density of soil solids was assumed to be  $2.65 \text{ g cm}^{-3}$ .

### Nitrogen and Phosphorus Mineralization

Soil net N and P mineralization was measured in situ using a modification of the DiStefano and Gholz (1986) resin core technique for use in wetlands that was developed by Noe (2011). The modified design includes three mixed-bed ion-exchange resin bags located above and below soil incubating inside a 7.8-mm-diameter core tube. The two inner resin bags adjacent to the soil capture  $\text{NH}_4^+$ ,  $\text{NO}_3^-$ , and soluble reactive P (SRP) mineralized and transported out of the soil during incubation, and the two outer resin bags prevent external inorganic nutrient transport into the modified resin core. The two middle bags serve as quality-control checks on the function of the inner and outer resin bags, optimally capturing no inorganic nutrients during incubation. The sampling, extraction, and analysis of soil cores and resin beads followed the same methodology as Noe (2011) with slight changes that were detailed in Wolf et al. (2011a).

Areal net mineralization rates ( $M$ ) were determined by comparing the amount of inorganic nutrients in the modified resin cores after the month-long field incubation (soil and two inner bags) to initial soil cores collected at the start of the incubation:

$$M = \frac{S_r + R_u + R_l - S_i}{AT}$$

where  $S_r$ ,  $R_u$ ,  $R_l$ , and  $S_i$  are the quantity of nutrient (mol) in the soil of the modified resin core after incubation ( $S_r$ ), resin bag immediately above the resin core soil after incubation ( $R_u$ ), resin bag immediately below the resin core soil after incubation

( $R_l$ ), and initial soil core ( $S_i$ ), representing the net production of inorganic nutrient. Due to variation in soil bulk density among plots, mineralization rates are expressed on an area basis ( $\text{mol m}^{-2} \text{ d}^{-1}$ ).  $A$  is the area of the soil core ( $\text{m}^2$ ), and  $T$  is the duration of the incubation (d). The production of SRP,  $\text{NH}_4^+$ , and  $\text{NO}_3^-$  was used to estimate fluxes of net P mineralization, net ammonification, and net nitrification, respectively. Net N mineralization is the sum of ammonification and nitrification, and percent nitrification is calculated as nitrification divided by net N mineralization. Cumulative annual mineralization is calculated as the sum of mineralization in individual incubations, including negative mineralization rates (immobilization). Nitrogen turnover rates are calculated as the mineralization flux divided by the standing stock of TN in the soil ( $\text{mol mol}^{-1} \text{ d}^{-1}$ , or  $\text{d}^{-1}$ ). Soil TP was not measured, preventing calculation of P turnover rates.

Cumulative annual  $\text{NH}_4^+$ ,  $\text{NO}_3^-$ , and SRP loading rates to the soil surface were quantified using the accumulation of inorganic nutrients on the upper outer resin bag of each modified resin core at the end of incubation. The upper outer resin bag was horizontal and flush with the soil surface, capturing inputs of inorganic nutrients to the soil surface (Yavitt and Wright, 1996) from all sources, including floodwater, precipitation, and atmospheric deposition. Cumulative annual loading rate of each inorganic nutrient is calculated as the sum of the individual monthly incubations.

### Denitrification Enzyme Assay

Monthly denitrification potential was determined in a soil subsample from each initial soil core collected for the mineralization measurements using the denitrification enzyme assay (DEA) procedure (Smith and Tiedje, 1979; Tiedje et al., 1989; Groffman et al., 1999) with a  $1 \text{ mmol L}^{-1}$  glucose,  $1 \text{ mmol L}^{-1}$   $\text{KNO}_3$ , and  $1 \text{ g L}^{-1}$  chloramphenicol amendment in  $25 \text{ mL}$  of deionized water. Slurries of  $25 \text{ g}$  of homogenized field-moist soil were assayed in duplicate in the laboratory at ambient temperature. Gas samples were stored in freshly evacuated  $2\text{-mL}$  glass vial containers (Tyco Healthcare Group LP) until they could be analyzed for  $\text{N}_2\text{O}$  on a gas chromatograph (Shimadzu 8A; Shimadzu Scientific Instruments, Inc.) with electron capture detection, generally within 3 d of sampling. Further methodological details can be found in Wolf et al. (2011a).

### Statistical Analysis

The biogeochemical and hydrologic measurements (cumulative annual rates of mineralization, DEA, sedimentation, and inorganic nutrient inputs to the soil surface and annual average surface water depth, soil redox, and soil WFPS) were compared between SW plots located in wetlands with stream water as the primary water source ( $n = 14$ ) to PGW plots located in wetlands with precipitation or groundwater as the primary water source (not stream water;  $n = 6$ ) using nonparametric Mann-Whitney tests. Correlations between soil biogeochemistry measurements and hydrologic measurements were performed using nonparametric Spearman correlation tests. All statistical tests were performed using SPSS version 13 (SPSS, 2004), and tests were considered significant at  $\alpha = 0.10$ .

## Results

### Hydrology and Hydrologic Inputs

Surface water depth was shallow in all wetland plots (generally <10 cm) during the monthly measurements. Plots in wetlands with stream water classified as their primary water source (SW plots) had slightly, but significantly, deeper average annual surface water depth than wetland plots with either precipitation or groundwater (not stream water) as their primary water source (PGW plots) (Table 1). Variation in surface water depth over the year in PGW wetlands was small compared with SW wetlands and generally tracked regional monthly precipitation (data not shown). The SW wetlands had brief peaks and more temporal variability in surface water depth than the hydrologically buffered PGW wetlands, likely due to stream connectivity and the flashiness of stream discharge. Although the quantity, timing, and duration of stream–wetland connectivity events were not quantified, we observed evidence of stream connectivity (debris and bent vegetation along flowpaths and sedimentation) in the SW plots several times during the study.

Annual inputs of ammonium, sediment, and sediment-N to the soil surface were much greater in SW wetlands: ammonium inputs were 2.4 times greater, sediment inputs were 11 times greater, and N sedimentation inputs were 8 times greater in SW plots compared with PGW plots (Table 1). Soils in SW plots also were more reducing than in PGW plots. Plots had similar nitrate input to the soil surface and soil WFPS regardless of water source. Input of SRP to the soil surface was 2.4 times greater in PGW plots compared with SW plots.

### Nutrient Flux Rates

Annual inputs of N from sedimentation averaged among all plots was  $697 \pm 189 \text{ mmol N m}^{-2} \text{ yr}^{-1}$  (mean  $\pm$  SE;  $9.76 \pm 2.65 \text{ g N m}^{-2} \text{ yr}^{-1}$ ). Inputs of ammonium, nitrate, and SRP were on average  $15.3 \pm 1.7 \text{ mmol N m}^{-2} \text{ yr}^{-1}$ ,  $14.3 \pm 2.5 \text{ mmol N m}^{-2} \text{ yr}^{-1}$ , and  $5.30 \pm 1.02 \text{ mmol P m}^{-2} \text{ yr}^{-1}$ , respectively. The

average cumulative net annual soil ammonification rate was  $327 \pm 42 \text{ mmol N m}^{-2} \text{ yr}^{-1}$  (prorated daily mean based on number of incubation days), and the average net nitrification rate was  $84 \pm 19 \text{ mmol N m}^{-2} \text{ yr}^{-1}$ , for a percent nitrification of  $27 \pm 10\%$ . Average net N mineralization rate (the sum of ammonification and nitrification) was  $410 \pm 43 \text{ mmol N m}^{-2} \text{ yr}^{-1}$ , and the average N turnover rate was  $0.055 \pm 0.006 \text{ yr}^{-1}$ , or 18.1 years. Average net P mineralization rate was  $4.44 \pm 0.48 \text{ mmol P m}^{-2} \text{ yr}^{-1}$ . Average DEA was  $2294 \pm 215 \text{ mmol N m}^{-2} \text{ yr}^{-1}$ .

The soil nutrient cycling rates were largely uncorrelated with each other. Independently calculated biogeochemical rates were statistically unrelated, with the exception of DEA rates with nitrification and percent nitrification (Table 2). However, ammonification and N mineralization and N turnover were correlated, as were nitrification and N mineralization and nitrification and percent nitrification, but in each of these correlations the cycling rates were not independent of each other.

### The Effects of Hydrology and Hydrologic Inputs on Nutrient Fluxes

Soil net ammonification, N mineralization, and N turnover rates were significantly greater in SW plots compared with PGW plots (Table 1). These soil inorganic N production rates were two to three times greater in SW plots. In contrast, net nitrification, percent nitrification, P mineralization, and DEA were similar among plots regardless of water source.

Soil net nutrient mineralization rates were correlated with many of the metrics of hydrologic inputs (Table 2). The strongest correlations for soil ammonification, N mineralization, and N turnover rates were with the rate of ammonium loading to the soil surface, increasing with greater ammonium input to the plot (Fig. 2A,C,E). However, the amount of ammonia produced in the soil greatly exceeded the amount input through ammonium loading to the soil surface (Fig. 2A). Soil ammonification rate also was positively correlated with sedimentation rate and with N

**Table 1. Comparisons of average hydrologic and hydrologic input measurements and annual soil nutrient cycling rates in wetlands with stream water as the primary water source vs. groundwater or precipitation.**

Measurement†	Stream water source (n = 14)		Groundwater or precipitation source (n = 6)		Mann-Whitney test	
	Mean	SE	Mean	SE	Z	P value‡
Surface water depth, cm	4.0	0.5	1.1	0.4	-3.300	<b>0.001</b>
Sedimentation, g m <sup>-2</sup> yr <sup>-1</sup>	2532	946	230	23	-2.639	<b>0.008</b>
N sedimentation, mmol N m <sup>-2</sup> yr <sup>-1</sup>	944	243	121	22	-2.639	<b>0.008</b>
SRP loading, mmol P m <sup>-2</sup> yr <sup>-1</sup>	3.74	0.41	8.95	2.91	-1.897	<b>0.058</b>
NO <sub>3</sub> <sup>-</sup> loading, mmol N m <sup>-2</sup> yr <sup>-1</sup>	14.2	2.2	14.7	6.9	-0.742	0.458
NH <sub>4</sub> <sup>+</sup> loading, mmol N m <sup>-2</sup> yr <sup>-1</sup>	18.6	1.6	7.6	1.5	-3.217	<b>0.001</b>
Soil WFPS, %	104	4	102	5	-0.330	0.741
Soil redox, mV	221	14	281	16	-2.227	<b>0.026</b>
Ammonification, mmol N m <sup>-2</sup> yr <sup>-1</sup>	414	34	124	58	-3.134	<b>0.002</b>
Nitrification, mmol N m <sup>-2</sup> yr <sup>-1</sup>	81	22	90	36	-0.082	0.934
N mineralization, mmol N m <sup>-2</sup> yr <sup>-1</sup>	495	37	214	65	-2.804	<b>0.005</b>
N turnover, yr <sup>-1</sup>	0.066	0.006	0.030	0.007	-2.969	<b>0.003</b>
% nitrification	15	4	54	30	-1.237	0.216
P mineralization, mmol P m <sup>-2</sup> yr <sup>-1</sup>	4.97	0.51	3.21	0.95	-1.567	0.117
DEA, mmol N m <sup>-2</sup> yr <sup>-1</sup>	2444	286	1945	236	-1.072	0.284

† DEA, denitrification enzyme assay; SRP, soluble reactive phosphorus; WFPS, water-filled pore space.

‡ Significant P values are highlighted in bold.

inputs from sedimentation (Table 2). Ammonification rate was similar to and increased with N sedimentation rate for the plots with relatively little N sedimentation, but N inputs at plots with relatively high N sedimentation rates exceeded in situ soil net ammonification (Fig. 3A). As a result of the positive relationship between sediment inputs and ammonification, percent nitrification was negatively correlated with sedimentation and N-sedimentation rates. In addition, ammonification rate and N turnover rate increased with the average depth of surface water. Soil P mineralization rate also was positively correlated with sedimentation and N-sedimentation rates (Table 2; Fig. 3B). However, P mineralization was most strongly correlated with the rate of nitrate loading to the soil surface, increasing with nitrate inputs (Fig. 2B).

Soil nitrification and DEA rates were only significantly correlated with soil WFPS (Table 2). Nitrification (Fig. 2D), percent nitrification, and DEA decreased in wetter soils (Fig. 2F). Although nitrification rate decreased linearly with greater soil WFPS (Fig. 2D), DEA peaked at around 80% WFPS and decreased in wetter soils (Fig. 2F). In addition, percent nitrification was positively correlated with soil redox, mostly due to soil ammonification being negatively correlated with redox, and with SRP loading to the soil surface (Table 2).

## Discussion

### Hydrology Characterization

Wetland hydrologic connectivity describes the degree to which a wetland is connected to the surrounding hydrologic system. Hopkinson (1992) uses the term to differentiate between hydrologically “open” systems, in which material and nutrient inputs largely exceed ecosystem standing stock and community need, and “closed” systems, in which the reverse is true. For example, hydrologic connectivity to streams enhances sediment and nutrient deposition in connected riparian floodplain wetlands compared with disconnected riparian floodplain wetlands (Noe and Hupp, 2005) or closed, depressional wetlands (Craft and Casey, 2000). Additionally, Richardson et al. (2004) found that hydrologic connectivity led to the delivery of river  $\text{NO}_3^-$  to C-rich floodplain sediments that stimulated N removal along the Upper Mississippi River.

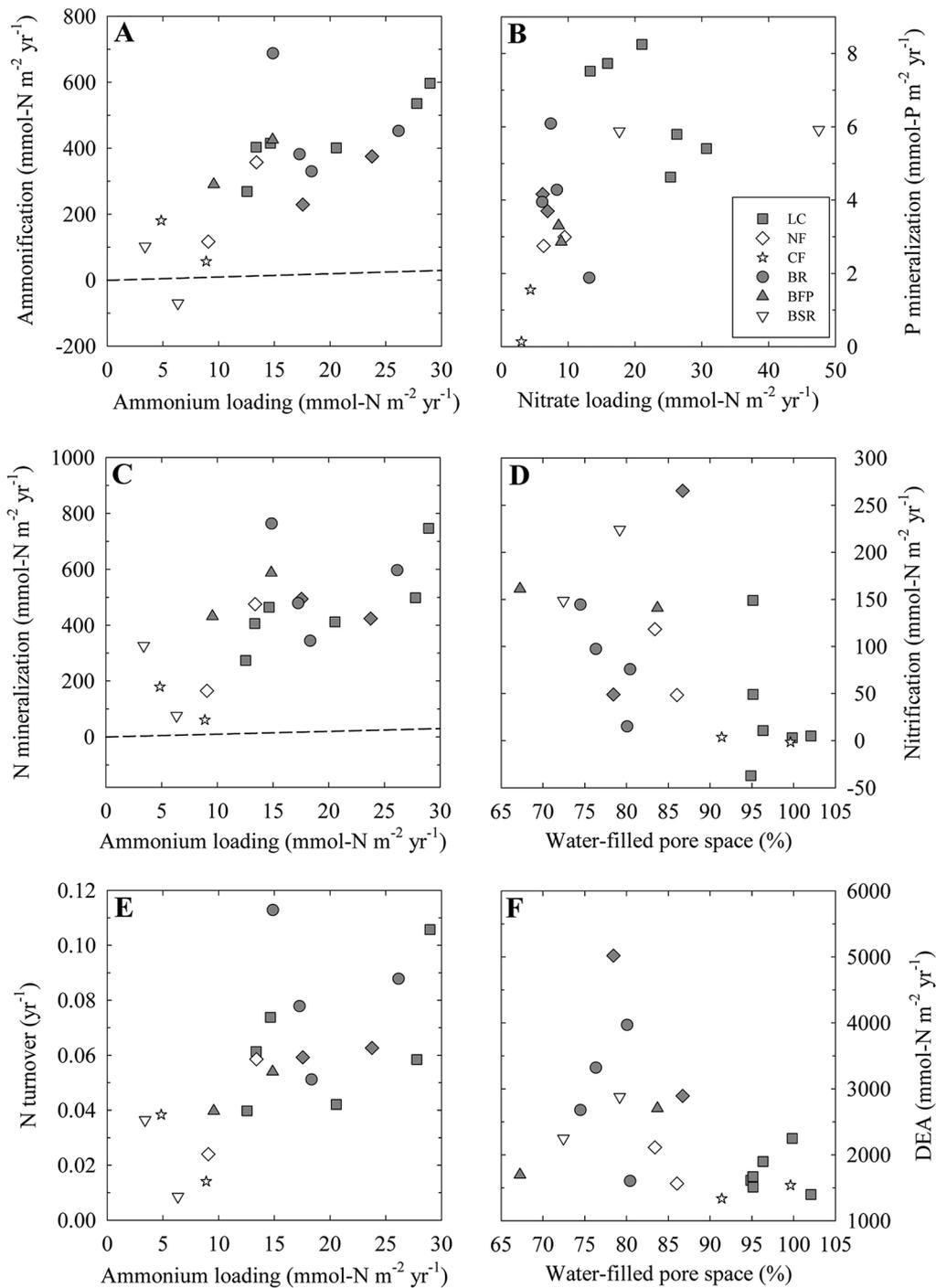
The wetlands in this study represent a wide range of hydrologic regimes that, when paired with associated hydrologic input measurements, indicate a gradient of hydrologic openness. Some wetland plots demonstrated muted variation in surface water depth, no evidence of surface water flowpaths associated with stream connectivity, and negligible amounts of sedimentation and inorganic N loading throughout the course of the study and can be considered hydrologically

**Table 2. Spearman correlations among annual soil biogeochemistry rates and hydrologic input and soil hydrology measurements ( $n = 20$  plots).**

Measurement†	Correlation	Ammonification	Nitrification	N mineralization	N turnover	% nitrification	P mineralization	DEA
Nitrification	$r_s$	-0.014						
	$P$	0.955						
N mineralization	$r_s$	<b>0.856‡</b>	<b>0.395</b>					
	$P$	<b>&lt;0.001</b>	<b>0.084</b>					
N turnover	$r_s$	<b>0.841</b>	0.179	<b>0.845</b>				
	$P$	<b>&lt;0.001</b>	0.450	<b>&lt;0.001</b>				
% nitrification	$r_s$	-0.346	<b>0.877</b>	0.048	-0.161			
	$P$	0.135	<b>&lt;0.001</b>	0.840	0.498			
P mineralization	$r_s$	0.284	-0.014	0.072	0.235	-0.158		
	$P$	0.225	0.955	0.762	0.319	0.506		
DEA	$r_s$	-0.039	<b>0.474</b>	0.177	0.195	<b>0.421</b>	0.120	
	$P$	0.870	<b>0.035</b>	0.454	0.409	<b>0.064</b>	0.613	
N sedimentation	$r_s$	<b>0.426</b>	-0.302	0.155	0.111	<b>-0.445</b>	<b>0.471</b>	-0.179
	$P$	<b>0.061</b>	0.195	0.515	0.641	<b>0.049</b>	<b>0.036</b>	0.451
Sedimentation	$r_s$	<b>0.516</b>	-0.325	0.239	0.189	<b>-0.528</b>	<b>0.468</b>	-0.195
	$P$	<b>0.020</b>	0.162	0.310	0.424	<b>0.017</b>	<b>0.038</b>	0.409
Surface water depth	$r_s$	<b>0.477</b>	-0.111	0.346	<b>0.398</b>	-0.336	0.308	-0.178
	$P$	<b>0.034</b>	0.643	0.135	<b>0.082</b>	0.147	0.187	0.452
Soil redox	$r_s$	<b>-0.492</b>	0.301	-0.266	-0.337	<b>0.436</b>	-0.323	0.167
	$P$	<b>0.028</b>	0.198	0.257	0.146	<b>0.055</b>	0.164	0.482
Soil WFPS	$r_s$	0.020	<b>-0.641</b>	-0.219	-0.069	<b>-0.668</b>	0.181	<b>-0.579</b>
	$P$	0.932	<b>0.002</b>	0.355	0.774	<b>0.001</b>	0.445	<b>0.008</b>
SRP loading	$r_s$	-0.299	0.266	-0.104	-0.298	<b>0.389</b>	-0.080	0.257
	$P$	0.200	0.257	0.663	0.202	<b>0.090</b>	0.738	0.274
$\text{NO}_3^-$ loading	$r_s$	0.174	0.023	0.011	-0.089	-0.011	<b>0.639</b>	-0.128
	$P$	0.462	0.925	0.965	0.710	0.965	<b>0.002</b>	0.591
$\text{NH}_4^+$ loading	$r_s$	<b>0.755</b>	0.057	<b>0.725</b>	<b>0.728</b>	-0.244	0.221	0.284
	$P$	<b>&lt;0.001</b>	0.811	<b>&lt;0.001</b>	<b>&lt;0.001</b>	0.301	0.349	0.225

† DEA, denitrification enzyme assay; SRP, soluble reactive phosphorus; WFPS, water-filled pore space.

‡ Significant  $r_s$  and  $P$  values are highlighted in bold.

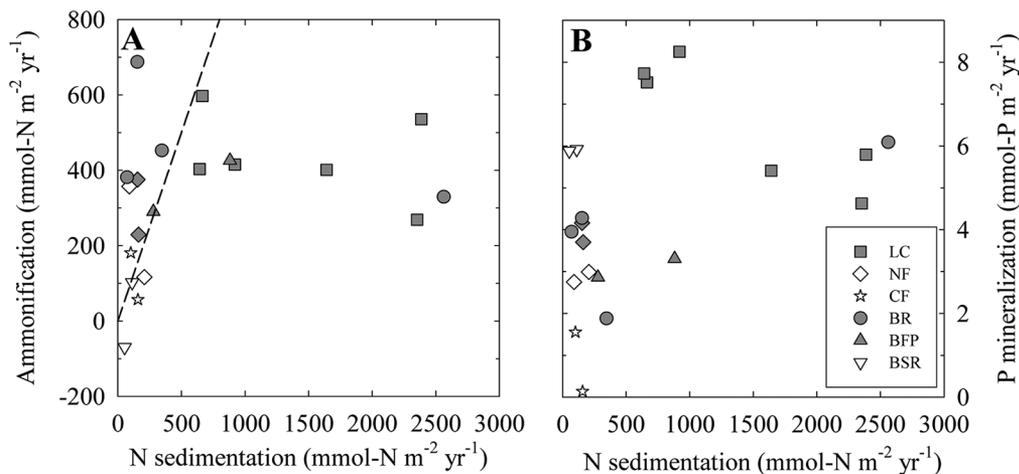


**Fig. 2.** Relationships between soil nutrient cycling rates and their most strongly correlated hydrology or hydrologic input measurement. Gray-filled symbols represent plots with stream water as their primary water source, and open symbols represent plots with groundwater or precipitation (not stream water) as their primary water source. Dashed line shows 1:1 relationship. BFP, Manassas National Battlefield Park; BR, Bull Run Mitigation Bank; BSR, Banshee Reeks Nature Preserve; CF, Clifton Farm; LC, Loudoun County Mitigation Bank; NF, North Fork Wetlands Bank.

closed systems. Other wetland plots had sources of sediment and N whose location relative to stream water inputs indicate that they can be considered open systems. Wetland plots with stream water classified as their primary source (SW plots) had clearly differing hydrology, including greater variation in surface water depth associated with flow pulses in the stream channel, evidence of flowpaths from stream to wetland, and deeper water, as well as greater inputs of sediment-N and ammonium than plots in wetlands with either precipitation or groundwater as the primary water source (PGW plots). We

interpret these findings as greater loading of stream-derived sediment, particulate N, and ammonium to SW wetlands. The greater sediment inputs in stream-connected wetlands likely also led to greater P sedimentation inputs, although this was not measured. This loading from streams was in addition to loading from atmospheric deposition, groundwater discharge, and local runoff, which was generally represented by loading rates to PGW plots.

The differences in rates of N sedimentation in SW and PGW plots were similar to those measured in hydrologically



**Fig. 3.** Relationships between soil ammonification and P mineralization with N sedimentation rate. Gray-filled symbols represent plots with stream water as their primary water source, and open symbols represent plots with groundwater or precipitation (not stream water) as their primary water source. Dashed line shows 1:1 relationship. BFP, Manassas National Battlefield Park; BR, Bull Run Mitigation Bank; BSR, Banshee Reeks Nature Preserve; CF, Clifton Farm; LC, Loudoun County Mitigation Bank; NF, North Fork Wetlands Bank.

open and closed systems in other studies. The average annual N sedimentation rate in SW plots,  $13.2 \text{ g N m}^{-2} \text{ yr}^{-1}$ , is in the upper range of those measured in open and hydrologically connected floodplain systems ( $1.4\text{--}8.0 \text{ g N m}^{-2} \text{ yr}^{-1}$  [Craft and Casey, 2000] and  $4.2\text{--}13.4 \text{ g N m}^{-2} \text{ yr}^{-1}$  [Noe and Hupp, 2005]). The average annual N sedimentation for the PGW plots was  $1.7 \text{ g N m}^{-2} \text{ yr}^{-1}$ , which is in the lower range found for closed depressional wetland systems ( $1.5\text{--}5.3 \text{ g N m}^{-2} \text{ yr}^{-1}$  [Craft and Casey, 2000]) and hydrologically disconnected riparian wetlands ( $3.5\text{--}4.8 \text{ g N m}^{-2} \text{ yr}^{-1}$  [Noe and Hupp, 2005]).

Atmospheric deposition generally provides around  $1 \text{ g N m}^{-2} \text{ yr}^{-1}$  to North American wetlands, with roughly equal amounts of nitrate, ammonium, and organic N (Morris, 1991). Estimated atmospheric TN deposition for the watersheds of the wetlands in this study ranges from  $1.39$  to  $1.62 \text{ g N m}^{-2} \text{ yr}^{-1}$  (EPA Watershed Deposition Tool v1.4.14; Schwede et al., 2009). Of this TN deposition to the region, from  $0.42$  to  $0.53 \text{ g N m}^{-2} \text{ yr}^{-1}$  is in the form of total reduced N, compared with the average  $0.26 \text{ g N m}^{-2} \text{ yr}^{-1}$  of ammonium trapped by ion-exchange resin bags at the soil surface of SW plots vs.  $0.11 \text{ g N m}^{-2} \text{ yr}^{-1}$  of ammonium loading at PGW plots. Although ammonium loading rate in SW plots was lower than regional atmospheric deposition rates, likely due to postdepositional uptake or nitrification in the wetlands, the greater loading rate of ammonium in SW plots represents an increase in N loading relative to PGW plots and thus likely represents a subsidy of ammonium loading from stream water. Nitrate loading to the wetland soil surfaces did not differ among plots with differing water sources and averaged  $0.20 \text{ g N m}^{-2} \text{ yr}^{-1}$  of nitrate. In contrast to ammonium loading, phosphate loading to the soil surface was significantly greater in PGW plots ( $0.28 \text{ g P m}^{-2} \text{ yr}^{-1}$ ) than in SW plots ( $0.12 \text{ g P m}^{-2} \text{ yr}^{-1}$ ). These phosphate loading rates are much greater than mean bulk TP deposition in the Chesapeake region ( $0.04 \text{ g P m}^{-2} \text{ yr}^{-1}$  [Jordan et al., 1995]). The greater phosphate loading to PGW plots was likely due to groundwater discharge to those wetlands because bulk atmospheric P deposition is likely lower than the observed phosphate loading rates.

These findings suggest that hydrologic connectivity to streams increased N loads to wetlands. This subsidy of

allochthonous N, both dissolved and particulate, likely increases the storage of N in stream-connected wetlands. This relationship supports the idea that wetlands that are disconnected from surrounding fluvial hydrology receive fewer inputs of sediment and associated nutrients (Hopkinson, 1992; Richardson et al., 2004; Noe and Hupp, 2005). Furthermore, loading of N from stream water to wetland soils removes N from streams that would otherwise be transported downstream, and thus it decreases stream N loads.

### The Effects of Hydrology on Nutrient Cycling

In addition to the greater loading of sediment N and ammonium to the soil surface, SW plots had greater in situ soil net production of ammonium, N mineralization (ammonification plus nitrification), and N turnover. Because nitrification was on average only 15% of the inorganic N produced during mineralization in SW plots (and thus mineralization was 85% ammonium production) and nitrification rates were similar among hydrologic classes, the pattern of greater N mineralization is due mostly to the net production of ammonium. The N mineralization rates for SW plots (average,  $495 \text{ mmol N m}^{-2} \text{ yr}^{-1}$ ) is intermediate compared with other in situ soil N mineralization studies in wetlands, but N mineralization rates in PGW plots (average,  $214 \text{ mmol N m}^{-2} \text{ yr}^{-1}$ ) were less than half than the SW plots and are among the lowest rates observed in the literature (typical range from  $171$  and  $850 \text{ mmol N m}^{-2} \text{ yr}^{-1}$ , respectively [Noe et al., 2013a]). Nitrogen mineralization rates in SW and PGW plots were similar to and spanned the range of rates measured along gradients of stream hydrologic connectivity in floodplain wetlands in the U.S. mid-Atlantic Piedmont (Noe et al., 2013a). Nitrogen turnover rates in PGW plots were among the slowest measured (average N turnover time, 33 yr), whereas N turnover rates in SW plots (average N turnover time, 15 yr) were among the fastest measured (typical range from 26 and 10 yr, respectively [Noe et al., 2013a]). In the SW plots, the greater N turnover rate compared with the N mineralization rate relative to other studies is due to either the somewhat small pool of TN in the soils of this study (0.21% TN) or stimulation of microbial decomposition of organic N (Scott and Binkley, 1997;

Noe et al., 2013a,b). Net P mineralization rates did not differ between plots with different water sources (average, 4.44 mmol P m<sup>-2</sup> yr<sup>-1</sup>) and were relatively low compared with rates in other wetlands (typical range from 1.46 and 39.1 mmol P m<sup>-2</sup> yr<sup>-1</sup>, respectively [Noe et al., 2013a]).

Hydrologic connectivity did not influence soil denitrification potential or nitrification rates. Denitrification enzyme activity rates (an estimate of denitrification potential if carbon and nitrate were not limiting) for this study were similar to the range of actual denitrification fluxes measured in floodplain wetlands (Mitsch and Day, 2006). Rates of denitrification measured using DEA in floodplain soils, however, are much higher in other studies (Richardson et al., 2004; Welti et al., 2012), suggesting that actual denitrification flux is small in the created and natural floodplain wetlands of this study.

Although most studies attempt to characterize hydrologic connectivity on a categorical basis (hydrologically open vs. closed), this study also attempted to characterize hydrologic connectivity to streams using the loading rates of sediment, sediment N, and inorganic nutrients as proxies for hydrologic flux measurements to differentiate wetlands and explain nutrient fluxes. Soil ammonification rates increased with greater inputs of sediment N and ammonium in this study (Fig. 2A and 3A). This supports the idea that wetlands that are more connected to surrounding hydrology, notably in the form of a stream for this study, receive greater nutrient input from stream bank overflow. Soil ammonium production increased with TN sedimentation and ammonium loading to the soil surface likely as a result of the increased mineralizable N substrate and energy in the form of labile C. This allochthonous subsidy prevents the exhaustion of material and energy supplies that occurs with the continuous cycling of autochthonous matter in closed wetland ecosystems that favor immobilization over ammonification (Reddy and DeLaune, 2008). The greater soil ammonium production was not the result of greater loading of ammonium to the soil surface leading to ammonium transport into the incubating resin cores that was then sampled and measured as mineralization. The design of modified resin cores prevents transport of external inorganic nutrients from outside the core into the incubating soil (Noe, 2011). Instead, ammonium and sediment N loading to the soil surface being concurrent with greater soil ammonia production was likely due to long-term additions of N to the soil and stimulation of the soil microbes that mineralize N. Combined inputs of N from sedimentation and ammonium deposition were typically greater than soil ammonification rates, with the exception of plots receiving the lowest N sedimentation inputs. Furthermore, erosion in the study sites was rare (authors, personal observation). Although nutrient export from the wetland to the streams was not measured in this study, these observations suggest that wetlands with greater connectivity to stream water were more retentive of N through increased soil storage despite similarity in denitrification potential.

Soil net ammonification rate was greater in plots with deeper surface water and lower soil redox. The relationship between hydrology and net ammonium production in soil is dynamic. An increase in standing water and a subsequent decrease in redox potential creates a reducing environment that favors the persistence of ammonium as the primary N

mineralization product (Reddy and Patrick, 1984). However, organic N is more efficiently mineralized under aerobic conditions (Ponnampetuma, 1972; Brinson et al., 1981; Bridgham et al., 1998). Increased net ammonification, but not net N mineralization, with greater inundation suggests that more reducing conditions preserved mineralized ammonium and limited nitrification and, conversely, that drier conditions oxidized mineralized ammonium to nitrate but that the total amount of inorganic N produced was not influenced by water depth and oxygen availability. Ammonification patterns were similar to those of Hefting et al. (2004), who found that higher water table levels (within 10 cm of the soil surface) favored ammonification and fully inhibited nitrification. Using the same in situ modified resin core technique, Noe et al. (2013a) found lower net ammonification fluxes in nearby natural floodplain wetlands in the Virginia Piedmont, most likely due to the drier soils in that study.

Nitrification and percent nitrification decreased with soil WFPS. Decreased oxygen transport into the soil matrix of wetter, higher WFPS soils likely impeded soil nitrification. Nitrification rates in this study were less than those in a natural Piedmont floodplain with drier soils (Noe et al., 2013a) and were within range of those found in natural and restored saltwater marshes (Thompson et al., 1995) and tidal freshwater forests (Noe et al., 2013b). Because tidal wetlands tend to be highly anoxic (and thus nitrate deficient), this highlights the particularly low rate of nitrification in the created and natural wetlands in this study. Percent nitrification (of net N mineralization) for wetland plots averaged 27% and was similar to bogs and acidic fens (Bridgham et al., 1998).

Nitrification also was positively correlated with denitrification potential (DEA), which is to be expected because increases in nitrate availability should support greater activity of denitrifying microbes. Denitrification potential was negatively correlated with soil WFPS. However, the relationship was not linear, and DEA peaked around 80% WFPS. Above that level of soil wetness, DEA declined presumably because decreased oxygenation of saturated soils resulted in a decrease in long-term nitrification, which led to fewer denitrifying microbes. Below that level of soil wetness, soils may have been more oxidizing, also leading to fewer denitrifying microbes. A similar pattern has been documented in floodplain soils across Europe (Pinay et al., 2007). Considering the low proportion of ammonium production that was nitrified in the created wetlands (nitrification was about a quarter of total N mineralization), it is likely that denitrifying microbes are limited by low nitrate availability associated with low redox potential. These results support those of Richardson et al. (2004), who found that denitrification rates significantly increase with the addition of NO<sub>3</sub><sup>-</sup> to floodplain wetlands. Study results also support similar studies where denitrification was positively correlated with redox potential (Verhoeven et al., 2001) and soil moisture (Schnabel et al., 1997; Hunter and Faulkner, 2001; Groffman and Crawford, 2003; Wolf et al., 2011a,b).

## Conclusions

Wetlands must be connected to nutrient sources to reduce nutrient loading downstream. The design of created and restored wetlands sometimes prevents stream connectivity to better control the hydrology of the project or to limit nutrient

and sediment inputs, but this practice may limit the wetland's capability to improve water quality. Our findings suggest that wetland creation and restoration projects should be designed to allow connectivity with stream water if the goal is to maximize water quality improvement in a watershed. Wetlands that do not experience connectivity with stream surface water can have positive influences on water quality, such as riparian buffers that intercept N-rich groundwater flowpaths. However, stream connectivity can provide large surface water inputs of nutrients and sediment to wetlands in addition to loading from groundwater, local runoff, and precipitation. This stimulation of nutrient inputs could also be important to the development of newly created and restored wetlands, where poor soil quality can contribute to low rates of nutrient cycling and poor vegetation establishment and floristic quality. Alternatively, provision of biodiversity or wildlife habitat is sometimes the goal of wetland restoration and may be best managed by limiting nutrient and sediment inputs by limiting stream hydrologic connectivity.

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