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In-Stream Wetland Mitigation of Nitrogen Contamination in a USA Coastal Plain Stream

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ABSTRACT

Nonpoint source N from riverine origin is a major water quality problem throughout the world. Nitrogen removal from a contaminated (6.6 mg L^{-1}, NO_3–N) stream was evaluated in this study using an in-stream wetland (ISW). The ISW was established at the exit of a 425-ha USDA Water Quality Demonstration watershed in the Coastal Plain of North Carolina. It ranged in depth from about 0.2 to 2 m, and it was <1% (3.3 ha) the size of the watershed. The ISW dramatically lowered mean stream NO_3–N from 6.6 to 2.0 mg L^{-1}. Nitrate–N mass removal was highly correlated to inflow NO_3–N (r = 0.93) in the warmer months when biological processes were more active. Ammonia–N mass removal was opposite that of NO_3–N. It was highly correlated to inflow NH_3–N (r = 0.81) during the cooler months. Removal of both NO_3–N and total-N (NO_3–N + TKN) were positively correlated to temperature with r values of 0.77 and 0.62, respectively. Total annual N removal for the ISW was approximately 3 kg ha^{-1} d^{-1}, which was about 37% of the inflow N. The ISWs appear to be very good landscape features for mitigating excess nonpoint source N in the southeastern Coastal Plain of the USA. As such, they are a good complement to other best management practices for improved water quality.

NONPOINT source N is a major pollutant of USA waters (Baker, 1992). It kills or displaces economically and environmentally important fish and shellfish by stimulating aquatic flora whose respiration can induce anoxic and hypoxic conditions (Babalais et al., 1996; Mazouni et al., 1996). Unfortunately, the excess N loading of streams and lakes is widespread. Nonpoint source N dominates the riverine pollutant fluxes to the coasts in 14 regions of North America, South America, Europe, and Africa (Howarth et al., 1996). Nitrogen-impacted waters in the eastern USA include the Chesapeake Bay, the Gulf of Mexico, and numerous nutrient sensitive rivers. Thus, major efforts are needed to reduce nonpoint source N.

Substantial efforts have been made to reduce nonpoint source N from its numerous sources (USDA, 1995 a, b). Closer conformance of N fertilization to projected crop needs has been a long-term target for high value as well as row crops. Better management of septic systems has also been addressed. Additionally, management of N from confined animal and bird production operations has been at the center of many state initiatives (Steele, 1995). These initiatives generally focus on N management plans and manure application protocols that limit the loss of N. However, these efforts often neither address the problems of mitigating excessive N in areas that are already contaminated nor address the inherent loss of N in native and agricultural ecosystems (leaching, volatilization, immobilization, etc.).

One of the ways to address these N problems is the use of wetland ecosystems. Wetlands are one of the most active ecosystems for transforming and assimilating contaminants from water (Howarth et al., 1996). Their natural occurrence in the landscape has long provided for assimilation and transformation of excessive nutrients. Additionally, wetlands have been constructed for the specific purpose of wastewater renovation and storm water filtration (Kadlec and Knight, 1996). Their reduced redox conditions and ample C make them ideal for the removal of N via denitrification. Thus, when the landscape offers the opportunity for construction or reestablishment of an ISW, it may offer the opportunity for N load reduction (Mitsch and Cronk, 1992).

Such a wetland establishment opportunity existed on a contaminated stream in the USDA Water Quality Demonstration project in the Herrings Marsh Run watershed in Duplin County, NC. The Herrings Marsh Run had both a headwater subwatershed with a relatively clean stream and another subwatershed with a

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Abbreviations: ISW, in-stream wetlands; LSD, least significant difference; WSS, water sampling station.

contaminated tributary. Stream NO_3–N in the contaminated tributary was about 7 mg L^{-1} (Stone et al., 1995). Potential for establishment of a wetland site existed at the lower reach of the contaminated tributary of the Herrings Marsh Run. The objective of this study was to determine if an in-stream wetland would reduce the stream N discharge from a N-overloaded watershed in the eastern Coastal Plain of the USA.

MATERIALS AND METHODS

Background

The Herrings Marsh Run is a 2044-ha watershed that is typical of the eastern Coastal Plain. It has both intensive crop and animal production. The watershed and its agricultural practices were investigated from 1990 to 1998 as a USDA Water Quality Demonstration project (Stone et al., 1995). It is not densely populated, but it contains large numbers of livestock. The Herrings Marsh Run contained a 425-ha watershed that had been overloaded with N, and the stream originating in this watershed contained excessive N. This stream exited the watershed at a small wetland landscape area. We hypothesized that repair of a breached dam (breached 30 yr ago) at the wetland area would create an ISW that would improve stream water quality by lowering the NO_3–N concentration. We made plans to repair the dam by use of a water control structure developed by the USDA Natural Resources Conservation Service, but the structure was going to be expensive. Thus, when beavers (*Castor canadensis*) began to fill the breach in April 1994, we decided to stabilize the sidewalls of the beaver dam to restrain stream flow and establish an in-stream wetland. The dam has been stable for normal storm events. Even tropical storm Bertha in 1996 caused minimal damage that was repaired within 60 d by the beavers, and no alteration of sampling routines was required.

For topographic survey of the ISW, we used a Lietz SDR electronic field notebook (Overland Park, KS).[1] The survey consisted of multiple transects across its width for determination of location and water depth. Data from the survey were used to calculate the storage volume and surface area which were 29 000 m^3 and 3.3 ha, respectively. The ISW depth at the beaver dam was about 2 m, but much of the side area was <0.3 m in depth (Fig. 1). It was approximately 500 m long and 55 m wide, and emergent aquatic weeds occupied approximately 40% of its surface area. The ISW perimeter (another 40% of the area) was dominated by trees (swamp tupelo [*Nyssa biflora*], red maple [*Acer rubrum*], and black willow [*Salix nigra*]). The remaining area was occupied by downed tree boles, isolated individuals of various species, and open water.

[1] Mention of a trademark, proprietary product, or vendor is for information only and does not constitute a guarantee or warranty of the product by the U.S. Department of Agriculture and does not imply its approval to the exclusion of other products or vendors that may also be suitable.

Data Collection

The ISW was located <30 m upstream of a U.S. Geological Survey water sampling station (WSS). Water samples for chemical analyses as well as flow measurements had been obtained at WSS since 1990 as part of the USDA Water Quality Demonstration project on the entire Herrings Marsh Run. This time span of sampling at WSS provided data for pre- and post-ISW conditions. Initially, samples were obtained hourly and composited. After 1993, the water samples were obtained every 2 h, combined into 3.5-d composites, and collected weekly. Flow measurements were recorded by the U.S. Geological Survey for WSS and transmitted electronically. The gaging station measured flow at 15-min intervals using automated water level recorders. A water level recorder was also installed in 1994 just above the dam.

Initially, the ISW had a well-defined inlet. It was monitored by use of a water sampling station established in October 1993. Stream samples were obtained using ISCO 2700, 3700 (Medina, NY) or American Sigma (Lincoln, NE) samplers with the same protocol as WSS, but flow was determined discreetly rather than continuously using a current meter (Scientific Instruments Model 1205 Price-type current meter, Milwaukee, WI). For flow determination, the width of the stream was divided into equal partial sections, and the mean velocity of each partial section was measured using the six-tenths method (Buchanan and Somers, 1969). In the winter of 1994, the beavers raised the height of their dam, and the inlet sampling station was flooded. In the larger configuration, the ISW inlet was about 100 m upstream, and it had two inlet streams

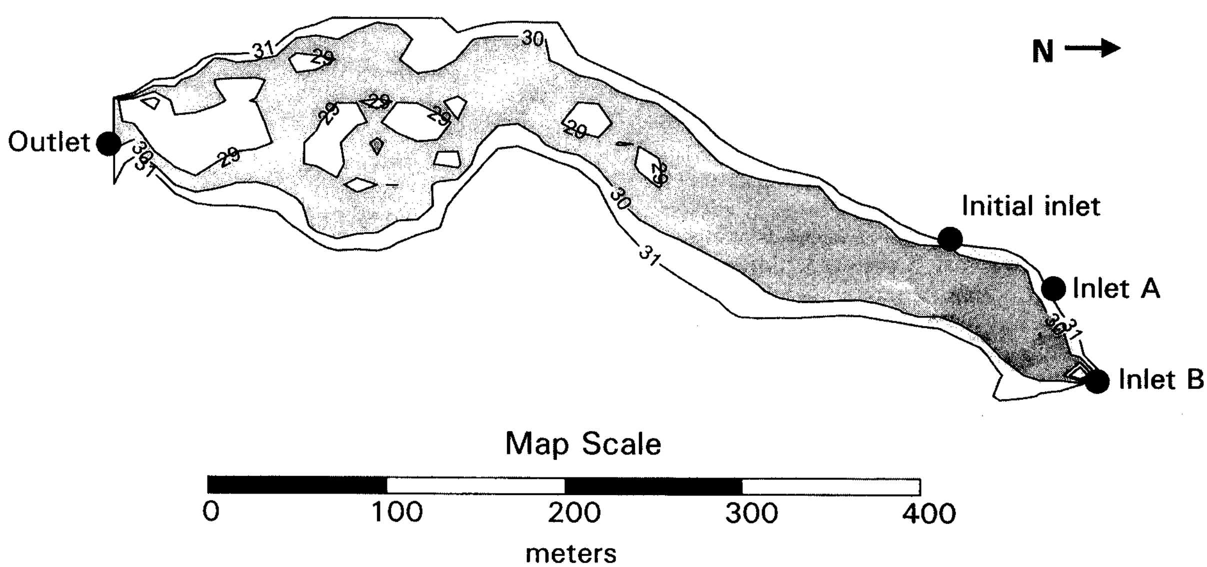

Fig. 1. In-stream wetland contour plot (meters above sea level).

that were within 30 m of each other. This altered configuration changed the NO_3–N at the original inlet but not the total Kjeldahl nitrogen (TKN). Since the beaver alteration of boundaries was of unknown duration, TKN values were obtained at the initial inlet sampling station, but NO_3 and NH_3 values were obtained at the two new inlet streams by use of weekly grab samples from April 1994 until March 1995, when stream samplers were installed on the two inlets. Concentration values from the streams were flow adjusted to calculate ISW nutrient inputs. The streams were generally about 60:40 in flow, but the specific proportions were used in each measured increment. Quarterly mean values for the NO_3/chloride ratios were calculated from the ratio of each 3.5-d sample.

The spatial distribution of dissolved oxygen and redox potential in the ISW water column (dissolved oxygen and redox potential) was measured by use of a Sonde 803PS multi-parameter water quality probe (Solomat, Norwalk, CT). Calibration of redox potential electrode was accomplished by use of both pH 4 and pH 7 buffer solutions saturated with quinhydrone. Calibration of the dissolved oxygen electrode was accomplished with both a 100% DO saturation (electrode head loosely wrapping a moist paper towel) and a 0% DO saturation (6% sodium sulfite solution). The sampling distribution consisted of 10 equal sections perpendicular to the stream flow within the ISW. Within each section, measurements were taken from four locations across the width of the wetland. At each location, measurements were taken from the bottom (0.3–1.9 m) and surface depths (0.15 m). Data were pooled for depth and location for evaluation of these parameters vs. distance from inlet to outlet. Samples for N and Cl analyses were collected from the water surface at these same locations. Samples were collected manually in glass bottles and handled as the other stream samples.

Air temperatures were the mean of weather stations within 30 miles of the ISW at Clinton, Mount Olive, and Warsaw, NC. Water temperature was not taken during the study period. However, they were taken via automated sensors in 1997 and 1998. The correlation of water temperature and the daily mean air temperature was $r = 0.94$ (Water temp °C = 0.74 air temp °C + 5.4). Monthly mass inflow and discharges were calculated from the monthly means of the 3 yr of the study to give equal weight to each year.

Laboratory Analyses

All water samples were transported on ice to the USDA-ARS, Coastal Plains Research Center in Florence, SC. They were analyzed using a TRAACS 800 Auto-Analyzer (Bran + Luebbe, Buffalo Grove, IL) for NO_3–N, NH_3–N, TKN, and chloride using EPA Methods 353.2, 350.1, 351.2, and 325.2, respectively (EPA, 1983). Total Kjeldahl N analyses were done on one sample per week. Precision and accuracy of the chemical analyses were determined with certified quality control samples (Spex Industries, Inc, Edison, NJ). Coefficients of variation for NO_3–N, NH_3–N, and TKN were 5, 7, and 6%, respectively, for a 5-yr period. Five percent of all samples were analyzed in duplicate with a coefficient of variation for NO_3–N, NH_3–N, and TKN, of 4, 8, and 4%, respectively. Samples were spiked weekly to determine percent recovery of the analyte. The percent recovery for NO_3–N, NH_3–N, and TKN was 103, 101, and 103%, respectively. Data were compared using paired *t*-test, Wilcoxon sign test, and regression analyses SAS (SAS Institute, 1990).

Table 1. Monthly means of nitrate–N, ammonia–N, and TKN concentrations for the inlet and outlet of an in-stream wetland in the southeastern Coastal Plain during a 3-yr study.

	Nitrate–N		Ammonia–N		TKN	
	Mean	SD	Mean	SD	Mean	SD
	mg L^{-1}					
Inlet	6.6	1.2	0.5	0.3	1.6	0.4
Outlet	2.0	1.4	0.7	0.7	1.5	0.5
	level of significance					
Wilcoxon Sign Test	0.01		0.09		0.72	
Paired *t*-test	0.01		0.10		0.72	

SD = standard deviation of 36 monthly means.

RESULTS AND DISCUSSION

During the initial years (1990–1993) of the Water Quality Demonstration project, we estimated that 85 kg ha^{-1} yr^{-1} excess N was being added to the watershed above the ISW (Stone et al., 1995). To lower this excess, NRCS worked with farmers to make repairs to animal waste lagoons and implement on-farm nutrient plans in the watershed. However, these improvements did not totally eliminate the problems of excess N applications or continued leaching from previously N-overloaded fields (Stone et al., 1998). Thus, the stream water quality impact of excess N remained evident in the high NO_3–N concentrations at the watershed outlet (WSS) prior to the establishment of the ISW (Fig. 2).

After establishment of the ISW, there was little

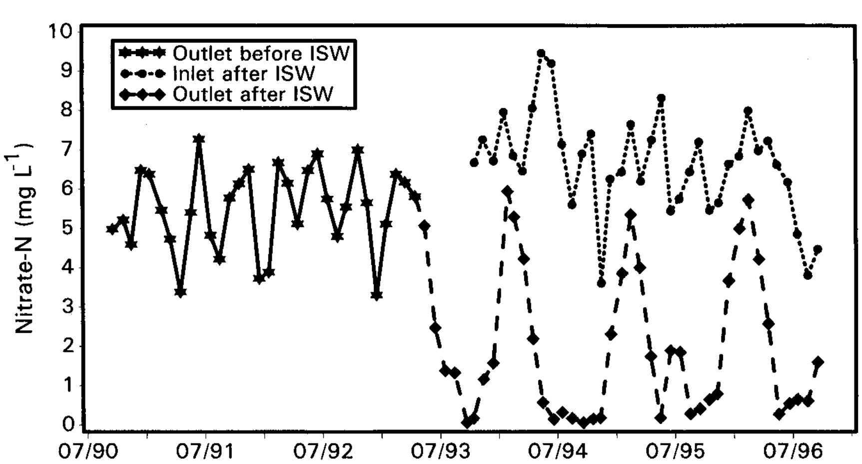

Fig. 2. Monthly means of NO_3–N concentrations for a Coastal Plain stream before and after the establishment of the in-stream wetland.

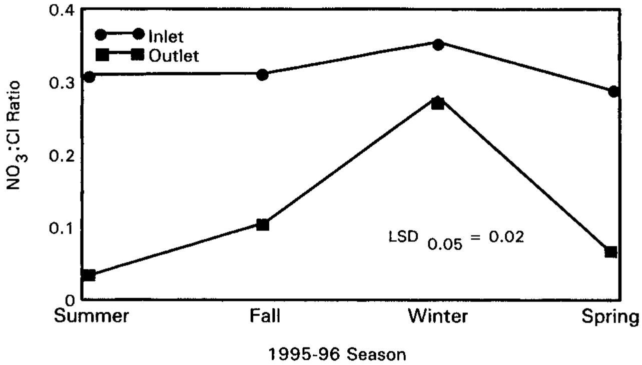

Fig. 3. Nitrate–N/chloride ratio at the inlet and outlet of the in-stream wetland (ratios are means of individual samples for 3 mo).

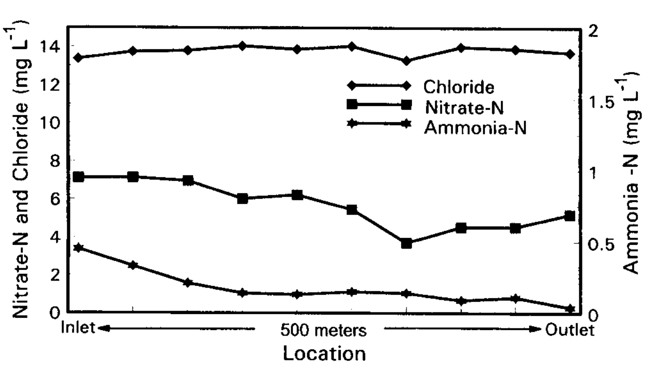

Fig. 4. Nitrate–N, NH_3–N, and chloride concentrations in the in-stream wetland during February 1996 ($LSD_{0.05}$ values for NO_3–N, NH_3–N, and chloride were 1.3, 1.4, and 0.2, respectively).

change in the input from the stream. The 3-yr mean of NO_3–N concentration in stream water entering the wetland (6.6 ± 1.2 mg L^{-1}) was about the same as it was at WSS during the pre-wetland time frame (Fig. 2 and Table 1). However, the stream water exiting the ISW at WSS was dramatically lower in NO_3–N. The 3-yr mean for the outlet was 2.0 ± 1.4 mg L^{-1}. This 70% reduction in mean concentrations was different at the P >[z] 0.01 level by the paired t-test and the Wilcoxon sign test (Table 1). The removal of NO_3–N by the ISW was most effective during the warmer months;

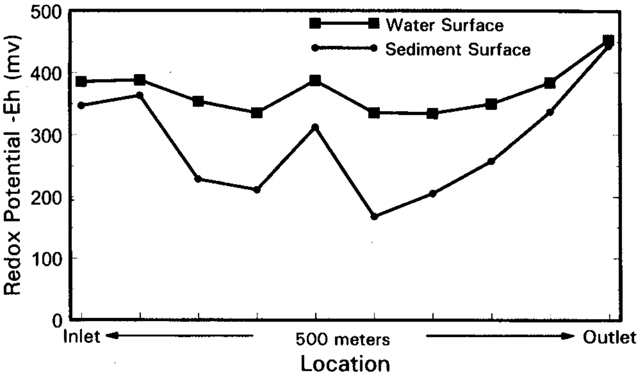

Fig. 5. Redox potential near the water and sediment surfaces in the in-stream wetland during February 1996 (Eh adjusted to pH 7 and hydrogen electrode; dissolved oxygen < 50% saturation; $LSD_{0.05}$ values for redox potential at water and sediment surfaces were 44 and 78, respectively).

concentrations were typically below 1 mg L^{-1} (Fig. 2). Removal was least effective during the cooler months when concentrations of NO_3–N were often as high as 5 mg L^{-1}. Nevertheless, monthly NO_3–N means at the outlet were always lower than at the inlet.

This large reduction of NO_3–N during the warmer months was likely related to the biological processes such as denitrification, microbial assimilation, and plant uptake (Kadlec and Knight, 1996; Stober et al., 1997). The removal of NO_3–N by biological processes can be checked by comparison of the ratio of the conservative anion (chloride) and the nonconservative anion (NO_3). Little biological removal would have been indicated by small changes in the ratio of NO_3/chloride. For instance, if there were only dilution or concentration changes associated with rainfall or evaporation, the ratio of the two anions would be the same at the inlet and outlet. The opposite condition, high biological removal, would have been indicated by large changes in the ratio. Such large changes occurred. The ratio at the inlet was relatively stable (about 0.30); conversely, the outlet ratio varied greatly during the year (Fig. 3). In the summer of 1995, the ratio at the outlet was about 0.05, but it dramatically increased during the first week of Decem-

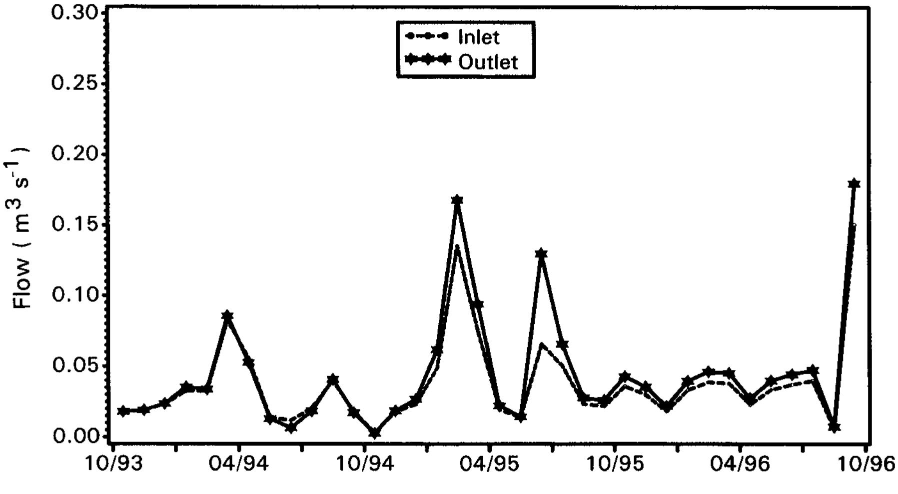

Fig. 6. Monthly means of daily stream flow for the in-stream wetland.

Table 2. Monthly means of nitrate–N for inflow, removal, and discharge for the in-stream wetland during the 3-yr study.

Month†	Inflow	SD‡	Removal	SD	Discharge	SD
	kg d^{-1}					
1	22.4	4.2	4.1	2.2	18.3	2.0
2	44.3	39.1	6.3	6.5	38.0	32.6
3	35.5	11.3	9.1	3.1	26.4	8.9
4	21.7	15.1	15.2	12.1	6.5	3.3
5	13.4	5.1	12.8	4.8	0.6	0.4
6	19.8	10.0	12.2	5.4	7.5	11.4
7	18.3	6.7	13.8	1.7	4.5	5.1
8	11.5	8.7	10.9	8.6	0.5	0.2
9	27.3	26.8	17.2	10.1	10.1	16.7
10	9.5	7.1	8.6	6.0	0.9	1.2
11	10.9	4.3	9.1	3.0	1.8	1.3
12	12.1	1.6	6.9	3.3	5.2	1.8
Mean	20.6	16.6	10.6	6.5	10.0	14.9

† Monthly means are based on the 3 mo/year, and the overall mean is based on 36 monthly means. January is month #1.
‡ SD = standard deviation.

ber to 0.18 and continued to rise to >0.30 in February 1996. In March the inlet ratio began to decline, and by April it was <0.10. These ratio differences between the inlet and outlet clearly illustrate the seasonal variation in biological impact on NO_3–N removal within the ISW. These findings are consistent with the general variation in biological processes with seasonal temperature changes, and they are consistent with the specific variation of denitrification rate in Coastal Plain riparian soils reported by Pavel et al. (1996).

Dissimilar to NO_3–N, the mean concentration of NH_3–N for the 3-yr study was increased from the inlet to the outlet ($P >$ [z] 0.10 by the paired t-test and the Wilcoxon sign test, Table 1). However, this increase was only 0.2 mg L^{-1} and not continuous for the year. The NH_3–N increases were likely related to detrital mineralization under low oxygen conditions that limited the conversion of NH_3 to NO_3 in the ISW. In the winter when NO_3–N removal was lowest, the outlet NH_3–N concentrations were less than the inlet concentrations. The TKN concentrations of the inlet and outlet were not significantly different by the paired t-test and the Wilcoxon sign test ($P >$ [z] 0.72, Table 1).

Ammonia–N, NO_3–N, and chloride concentrations at various locations within the ISW in February 1996 are shown in Fig. 4. Flow on the sampling day was 0.037 m^3 s^{-1}, and the temperature was 13.7°C. The N, particularly NH_3–N, was removed and transformed as water flowed through the ISW. Ammonia–N entering the wetland was 0.46 mg L^{-1}. It was rapidly lowered to 0.28 mg L^{-1} and exited the wetland at 0.06 mg L^{-1}. Nitrate–N was only reduced from about 7 to 5 mg L^{-1} as the water moved through the wetland. This was likely related to several factors such as lower plant growth, microbial respiration, and C, as well as more oxidized conditions during the cooler months. Concentrations of chloride both entering and leaving the ISW were about 14 mg L^{-1}. Dissolved oxygen was generally <50% saturation throughout the wetland, and redox potential was in a range (<325 mv) that would indicate denitrification near the sediment surface but not near the surface (Fig. 5).

Even though the NO_3–N and NH_3–N concentrations were at times dramatically lowered, determination of the mass removal required flow measurement. Mean daily flows at the outlet ranged from 0.0003 to 1.66 m^3 s^{-1}. Monthly mean flow at the outlet ranged from 0.003 to 0.18 m^3 s^{-1} (Fig. 6). These flow rates resulted in mean monthly residence times in the ISW of 111 to 1 d.

The mean monthly mass inflow, removal, and discharge of NO_3–N for the ISW were 20.6, 10.6, and 10.0 kg d^{-1}, respectively (Table 2). The mean annual removal on an area basis for the 3.3-ha ISW was 3.2 kg N ha^{-1} d^{-1}, which is a fairly conservative rate for a treatment wetland (Hunt et al., 1995). The annual percentage reduction was 51 for NO_3–N. Loading was highest in the winter months (>22 kg d^{-1}) when NO_3–N removal was the lowest (≤9.1 kg d^{-1}). Consequently, NO_3–N discharges were higher during January through March (≥18.3 kg d^{-1}), but during the remainder of the year they were ≤10.1. The effect of temperature can be seen by the correlation of NO_3–N removal with mean monthly minimum air temperature (r = 0.77, P < 0.01, $y = 0.37T_{min} + 6.80$, Fig. 7).

The variation in N removal with temperature contributed to the low correlation of retention time to N removal. However, during the warmer months (April-

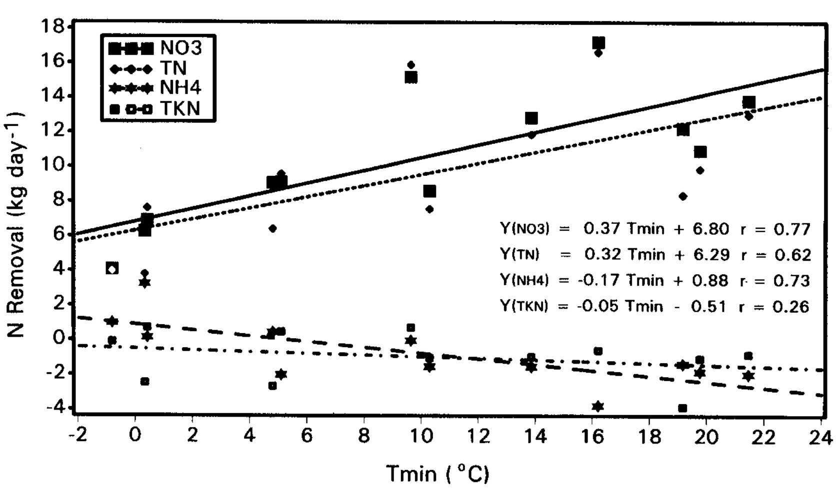

Fig. 7. Mean monthly N component removal vs. minimum monthly air temperature for the in-stream wetland.

November), the mass removals of NO_3–N were highly correlated to the mass inputs of NO_3–N (r = 0.93, P < 0.01, $y = 0.80\ NO_{3\ inflow} - 0.60$, Fig. 8). The considerable capacity of the ISW to remove large amounts of NO_3–N during the warm months is demonstrated by the high removal of the NO_3–N when NO_3–N inputs ranged from 1 to 130 kg N d^{-1}. At some higher loading rate, the ISW would not have continued to remove a similar portion of the added NO_3–N. However, within the observed range and the warm months, the ISW was capable of removing a similar portion of the load, and it provided a significant buffer to NO_3–N movement from the watershed. Conversely, during the cooler months (December-March), the correlation for NO_3–N removal vs. input was low (r = 0.44). As a result, the ISW provided a relatively small buffer for NO_3–N movement from the watershed during this period.

This small buffer capacity was true even though the redox values in the stream water near the bottom of the wetland were in the denitrifying range during the cooler periods (<325 mv). The denitrification zone of the wetland was probably closer to the ISW bottom. Under these conditions, denitrification would be less efficient because NO_3–N diffusion into the ISW sediment surface may have been required (Stober et al., 1997; Reddy et al., 1978). Additionally, the removal of NO_3–N by other biological processes such as plant uptake would have been lower. For instance, 2.0 kg ha^{-1} d^{-1} of N were accumulated by rush (*Juncus effusus*), bulrushes (*Scirpus americanus*, *Scirpus cyperinus*, and *Scirpus validus*), bur-reed (*Sparganium americanum*), and cattails (*Typha angustifolia* and *Typha latifolia*) during about 150 d growth in a nearby constructed wetland that received swine wastewater at 3 to 10 kg ha^{-1} d^{-1} during 1993 and 1994 (Hunt et al., 1998). Swamp tupelo can also accumulate about 0.6 kg ha^{-1} d^{-1} of N during a similar growing period (Fail et al., 1986). Assuming that the ISW was about equally occupied with these plant communities, it is reasonable to estimate that plants were removing about 1.3 kg ha^{-1} d^{-1} of N in the ISW during the 150 d of highest plant growth. However, these plants accumulated very little N during the remainder of the year.

Table 3. Monthly means of ammonia–N for inflow, removal, and discharge for the in-stream wetland during the 3-yr study.

Month†	Inflow	SD‡	Removal	SD	Discharge	SD
	kg d^{-1}					
1	2.4	0.5	1.0	0.9	1.4	0.6
2	5.4	5.6	3.2	4.1	2.2	1.7
3	2.4	0.4	0.4	0.6	2.0	1.0
4	1.7	1.6	−0.0	2.3	1.8	1.0
5	0.8	0.2	−1.5	3.2	2.4	3.0
6	2.1	1.5	−1.4	1.4	3.5	2.8
7	1.8	1.3	−2.0	0.9	3.8	1.7
8	0.5	0.4	−1.8	1.3	2.3	1.6
9	0.9	1.3	−3.8	5.5	4.7	6.8
10	0.5	0.4	−1.5	2.6	2.0	2.7
11	0.7	0.4	−2.0	1.8	2.7	1.5
12	0.7	0.4	0.1	0.3	0.6	0.5
Mean	1.7	2.0	−0.8	2.8	2.4	2.4

† Monthly means and SD are based on the 3 mo/year, and the overall mean is based on 36 monthly means. January is month #1.
‡ SD = standard deviation.

Ammonia–N was different from NO_3–N in most aspects of mass balance. The mean monthly mass inflow, removal, and discharge of NH_3–N for the ISW were 1.7, −0.8, and 2.4 kg d^{-1}, respectively (Table 3). The months of best NH_3–N mass removal were opposite those of NO_3–N. During the four cooler months, NH_3–N removal was positive; but during the eight warmer months, the removal was negative. Monthly NH_3–N removal ranged from 3.2 to −3.8 kg N d^{-1} in February and September, respectively. On an annual basis, the removal of NH_3–N was negatively correlated to the mean monthly minimum air temperature (r = 0.73, P < 0.01, $Y = -0.17T_{min} + 0.88$, Fig. 7). Removal of NH_3–N was not well correlated to input during the warm months (r = 0.17). However, during the four cooler months, the removal of NH_3–N was positively correlated to NH_3–N input (r = 0.81, F < 0.01, $y = 0.70\ NH_{4\,inflow} - 0.80$, Fig. 9). Several factors may have been involved. In the warmer seasons, ammonification of organic N likely increased, nitrification was likely limited by low oxygen, and mammals may have added organic N. During the cooler months when oxygen solubility was higher, respiration and ammonification were lower; during this period, nitrification was probably sufficient to remove the NH_3–N. Since NH_3–N was a significant fraction of TKN,

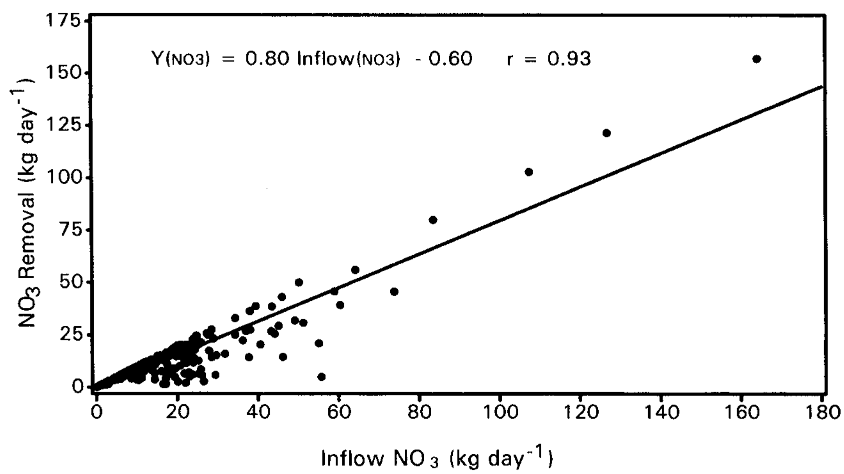

Fig. 8. Nitrate–N removal vs. mass inflow of NO_3–N for the in-stream wetland (excluding the months of December, January, February, and March).

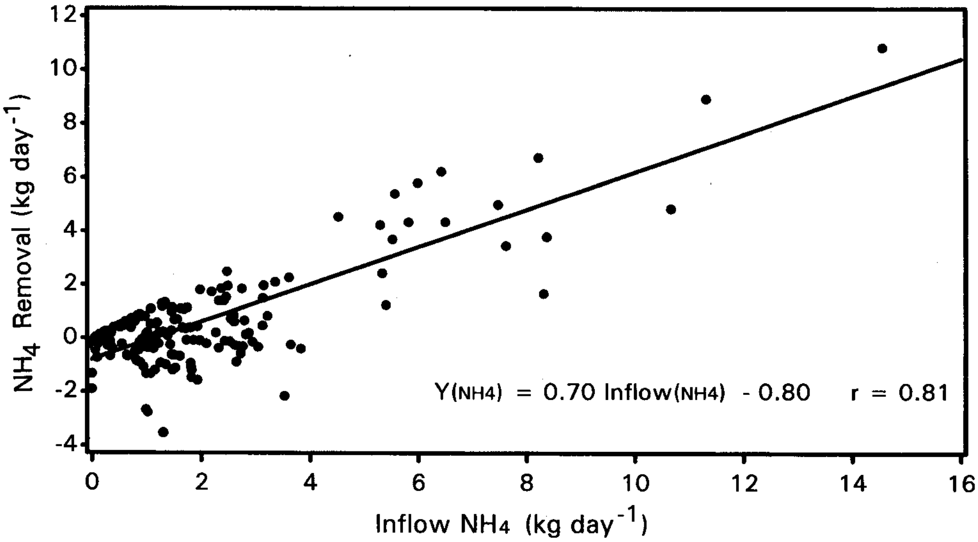

Fig. 9. Ammonia–N removal vs. mass inflow of NH_3–N for the in-stream wetland (for the months of December, January, February, and March).

it caused TKN removal to be generally negative; monthly values ranged from 0.7 to −3.9 kg N d^{-1}. However, TKN was less affected by temperature than NO_3–N and NH_3–N (Fig. 7).

The combined effects of NO_3–N and TKN removal resulted in monthly total-N (NO_3–N + TKN) removals of 3.8 to 16.6 kg N (Table 4). Since total-N was dominated by NO_3–N, the response of total N to temperature was similar to the response of NO_3–N ($y = 0.32\ TN_{inflow} + 6.29$, $r = 0.62$, Fig. 7). The mean monthly mass inflow, removal, and discharge of total N for the ISW were 25.7, 9.5, and 16.2 kg d^{-1}, respectively. Total annual N removal for the ISW was approximately 3 kg ha^{-1} d^{-1}, which was about 37% of the inflow N. Since the ISW was new, this removal process could change as it matures, but the ISW is likely to be a significant treatment site as long as the predominant N component is NO_3 and the ISW has reducing conditions and ample C.

The lack of effective NO_3–N removal during January through March is a major contributor to the N remaining in the stream. During these months, the removal is low as would be expected from the low net removal of total-N from beaver ponds in Canada (Devito et al., 1989). Nitrate–N mass passing through the ISW during these months was double the monthly mass exiting the ISW during the other 9 mo. Thus, improvement of the cooler weather performance of the ISWs would greatly enhance their already considerable importance in N scrubbing from streams. This will, of course, require considerable innovation.

Our overall findings on the performance of the ISW for removal of N are consistent with the theoretical basis for wetland restoration reported by Fleischer et al. (1991) and the projected effectiveness of Baker (1992). Thus, ISWs appear to be very good landscape features for mitigating excess nonpoint source N that moves from agricultural ecosystems in the southeastern Coastal Plain of the USA. As such, they are a good complement to other best management practices for improved water quality.

Table 4. Monthly means of total–N for inflow, removal, and discharge for the in-stream wetland during the 3-yr study.

Month†	Inflow	SD‡	Removal	SD	Discharge	SD
	kg d^{-1}					
1	27.5	4.1	4.0	2.8	23.5	2.3
2	53.4	47.6	3.8	4.1	49.6	43.5
3	43.5	13.0	6.4	4.7	37.1	9.8
4	26.5	18.6	15.9	14.9	10.6	4.9
5	16.6	7.8	11.9	3.4	4.8	4.4
6	25.9	14.1	8.4	10.9	17.6	21.8
7	24.4	8.1	13.0	0.9	11.4	8.9
8	15.0	11.9	9.9	8.3	5.1	3.5
9	35.6	36.9	16.6	9.3	19.0	27.6
10	11.1	8.6	7.6	4.7	3.6	4.6
11	13.6	4.2	9.6	2.9	4.0	1.8
12	15.2	2.6	7.6	4.7	7.6	2.3
Mean	25.7	20.7	9.5	7.2	16.2	19.8

† Monthly means and SD are based on the 3 mo/year means, and the overall mean is based on 36 monthly means. January is month #1.
‡ SD = standard deviation.

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REFERENCES

Babalais, N.N., W.J. Wiseman, R.E. Turner, B.K. Sengupta, and Q. Dortch. 1996. Nutrient changes in the Mississippi River and system responses on the adjacent continental shelf. Estuaries 19:386–407.
Baker, L.A. 1992. Introduction to nonpoint source pollution in the United States and prospects for wetland use. Ecol. Eng. 1:1–26.
Buchanan, T.J., and W.P. Somers. 1969. Discharge measurements at gaging stations, Book 3 applications of hydraulics. p. 65. *In* Techniques of water resources investigations of the United States geological survey. U. S. Gov. Print. Office, Washington, DC.
Devito, K.J., P.J. Dillon, and B.D. Lazerte. 1989. Phosphorus and nitrogen retention in five Precambrian shield wetlands. Biogeochemistry 8:185–204.

Fail, J.L., Jr., M.N. Hamzah, B.L. Haines, and R.L. Todd. 1986. Above and below ground biomass, production and element accumulation in riparian forests of an agricultural watershed. p. 193–224. *In* D.L. Correll (ed.) A watershed research perspective. Smithsonian Inst., Washington, DC.

Fleischer, S., L. Stibe, and L. Leonardson. 1991. Restoration of wetlands as a means of reducing nitrogen. Ambio 20:271–272.

Howarth, R.W., G. Billen, D. Swaney, A. Townsend, N. Jaworski, K. Lajtha, J.A. Downing, R. Elmgren, N. Caraco, T. Jordan, F. Berendse, J. Freney, V. Kudeyarov, P. Murdoch, and Zhu Zhao-Liang. 1996. Regional nitrogen budgets and riverine N and P fluxes for the drainages to the North Atlantic Ocean: Natural and human influences. Biogeochemistry 35:75–139.

Hunt, P.G., K.C. Stone, F.J. Humenik, and J.M. Rice. 1995. Impact of animal waste on water quality in an eastern Coastal Plain watershed. p. 257–264. *In* K. Steele (ed.) Animal waste and the land-water interface. CRC Press, Boca Raton, FL.

Hunt, P.G., A.A. Szogi, F.J. Humenik, and J.M. Rice. 1999. Treatment of animal wastewater in constructed wetlands. *In* Jose Martinez (ed.) Management strategies for organic waste use in agriculture. Rennes, France. (In press).

Kadlec, R.H., and R.L. Knight. 1996. Treatment wetlands. Lewis Publ., Boca Raton, FL.

Mazouni, N., J.C. Gaertner, J.M. Deslouspaoli, S. Landrein, and M.G. Doedenberg. 1996. Nutrient and oxygen exchanges at the water-sediment interface in a shellfish farming lagoon. J. Exp. Marine Biol. Ecol. 205:91–113.

Mitsch, W.J., and K.K. Cronk. 1992. Creation and restoration of wetlands: Some design consideration for ecological engineering. Adv. Soil Sci. 17:217–259.

Pavel, E.W., R.B. Reneau, D.F. Berry, E.P. Smith, and S. Mostaghimi. 1996. Denitrification potential of nontidal riparian wetland soils in the Virginia Coastal Plain. Water Res. 30:2798–2804.

Reddy, K.R., W.H. Patrick, Jr, and R.E. Phillips. 1978. Role of nitrate diffusion in determining the order and fate of denitrification in flooded soil. I. Experimental results. Soil Sci. Soc. Am. J. 42: 268–272.

SAS Institute. 1990. SAS language references. Version 6. 1st ed. SAS Inst., Cary, NC.

Steele, K. (ed.). 1995. Animal waste and the land-water interface. CRC Press, Boca Raton, FL.

Stober, J.T., J.T. O'Connor, and B.J. Brazos. 1997. Winter and spring evaluations of a wetland for tertiary wastewater treatment. Water Environ. Res. 69:961–968.

Stone, K.C., P.G. Hunt, S.W. Coffey, and T.A. Matheny. 1995. Water quality status of a USDA Water Quality Demonstration project in the eastern Coastal Plain. J. Soil Water Conserv. 50:567–571.

Stone, K.C., P.G. Hunt, M.H. Johnson, and S.W. Coffey. 1998. GLEAMS simulation of ground water nitrate-N from row crop and swine wastewater spray fields in the eastern Coastal Plain. Trans. ASAE 41:51–57.

U.S. Department of Agriculture. 1995a. Clean Water-clean Environment-21st Century. Vol. II. Nutrients. ASAE, St. Joseph, MI.

U.S. Department of Agriculture. 1995b. Clean Water-clean Environment-21st Century. Vol. III. Practices, systems, and adoption. ASAE, St. Joseph, MI.

U.S. Environmental Protection Agency. 1983. Methods for chemical analysis of water and wastes. EPA-600/4-79-020. Environmental Monitoring and Support Lab, Office of Research and Development, USEPA, Cincinnati, OH.