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Denitrification in a Restored Riparian Forest Wetland

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ABSTRACT

Groundwater nitrate moving from upland areas toward streams can be removed by denitrification in mature riparian forests, but denitrification in restored riparian forests has not been quantified. We determined denitrification rates in a restored riparian wetland below a liquid manure application site. A riparian forest buffer consisting of hardwoods along the stream and pines above the hardwoods was established according to USDA specifications. Denitrification was measured monthly using the acetylene inhibition technique on intact soil cores for 2 mo before manure application began and for 24 mo after manure application. Groundwater movement of NO_3^--N and total Kjeldahl N were estimated biweekly. Average annual denitrification rate was 68 kg N_2O-N ha^{-1} yr^{-1}. Denitrification was significantly higher in a grassed area than in either of the forested areas. Denitrification did not differ significantly between the hardwood and pine areas. Denitrification was greater than a conservative estimate of groundwater input of total N. Denitrification rates were higher in April and May 1992 and 1993, after manure application to the upland began, compared with April and May 1991, before manure application began. These results indicate that a riparian wetland, which has not undergone hydrologic modifications, can have denitrification rates comparable to mature riparian forests. Higher denitrification rates in an adjacent grassed wetland and lack of differences in denitrification in hardwood and pine zones indicates that the high denitrification rates were due to factors other than the reforestation itself. Compared with groundwater inputs of N, denitrification was an important sink for N moving from the upland management system.

NATURAL RIPARIAN FOREST WETLANDS are of considerable interest in managing nonpoint-source pollution because of their well-documented ability to remove nonpoint-source pollutants such as NO_3^- and sediment (Jacobs and Gilliam, 1985; Lowrance et al., 1984; Peterjohn and Correll, 1984). Although guidelines for wetlands management generally preclude intentional routing of nonpoint pollutants into natural wetlands, naturally occurring wetlands in agricultural landscapes will receive nonpoint-source pollutants such as N in groundwater flow and surface runoff and can act as significant N filters.

The accumulated research on nonpoint pollution control by riparian forest buffers (RFBs) has led to development of standards for riparian forest buffers for agriculture (Lowrance, 1991; Welsch, 1991). These standards recommend a three zone RFB. Zone 3 is a herbaceous buffer strip adjacent to row-crop fields. Zone 3 is used for sediment trapping and for converting concentrated flow into sheet flow. Zone 2 is a managed forest zone extending from Zone 3 to within 5 to 15 m of the stream channel system. Zone 2 is used for sediment trapping and biological removal of nutrients through vegetation uptake, denitrification, and other microbial processes. Zone 1 is a narrow band of permanent woody vegetation, which, at a minimum, encompasses the entire stream channel system. Zone 1 is used for stream bank stabilization, nutrient removal and sequestering, and providing control of the chemical, physical, and biological environment of the aquatic ecosystem. This RFB standard is intended primarily for low order streams in humid and subhumid regions. Zones 1 and 2 will often be wetlands, especially in humid regions. Zone 3 may also be a wetland.

Although mature RFBs are clearly effective, the water quality impacts of restored RFBs have not previously been studied. Research on the processes that control the water quality impacts of restored RFBs is necessary to understand how restored wetlands can be selfsustaining through time and to understand how they can be integrated as buffer systems into field and landscape scale management systems (Nat. Res. Council, 1993; Vellidis et al., 1993b).

Denitrification, the biological reduction of NO_3^- to N_2, N_2O, or other N gases, is an important potential mechanism for removing NO_3^- from soils and groundwater in RFB. Denitrification in wetland soils has been measured in a limited number of studies and has been shown to be important in wetlands receiving N subsidies and relatively unimportant in N limited wetlands (Lowrance et al., 1984; Hendrickson, 1981; Hemond, 1983; Groffman and Tiedje, 1989; Struwe and Kjoller, 1989; Korselman et al., 1989; Zak and Grigal, 1991). Freshwater wetlands receiving N subsidies from agricultural drainage and runoff or septic tank drainage can have significantly enhanced denitrification or NO_3^- removal rates (Hendrickson, 1981; Lowrance et al., 1984; Groffman et

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Abbreviations: RFB, riparian forest buffer; WFPS, water-filled pore space.

al., 1992; Haycock and Pinay, 1993). Denitrification in surface soils of riparian forests of the southeastern coastal plain has been shown to be NO_3^--limited and generally responded positively to additions of NO_3^- alone or C + NO_3^-, but not to C additions alone (Ambus and Lowrance, 1991).

This study was designed to examine the role of denitrification in a restored riparian forest wetland in the southeastern U.S. coastal plain. It is part of a larger study and demonstration project on the fate of nutrients and pesticides in this restored RFB (Vellidis et al., 1993b). The restored wetland is adjacent to a liquid dairy manure application area used for year-round forage production (Williams et al., 1991; Hubbard et al., 1991; Vellidis et al., 1993a). Here we report on denitrification for a 2-mo period before manure application began in the adjacent uplands and for the first 2 yr after manure application began. The denitrification rates are compared with estimates of N movement in shallow groundwater in the top 1 m of soil. The primary purpose of the denitrification study was to determine the annual rates of N removal through denitrification in the entire wetland. Secondary objectives were to determine: (i) whether denitrification differed in different parts of the wetland; (ii) whether denitrification would increase after beginning of the upslope liquid manure application; and (iii) whether denitrification differed in four different soil depths.

MATERIALS AND METHODS

Study Site

The study site (referred to as the *dairy wetland*) is a riparian zone along a first-order stream channel. The wetland soil is Alapaha sandy loam (loamy, siliceous, thermic Arenic Plinthic Paleaquult), a deep, poorly drained soil commonly found along drainageways (Calhoun, 1983). The wetland has a total area of about 0.7 ha and receives the surface runoff and shallow groundwater from about 2.4 ha of upland soils including both conventionally fertilized bermudagrass [*Cynodon dactylon* (L.) Pers.] pasture and the liquid manure application system (Fig. 1). The dairy wetland was forested until 1985, when the mature merchantable timber was cut and stumps were removed. The area was converted into a wet pasture with a mixture of grasses and rushes (*Juncus* sp.). The area was grazed from 1986 to 1990 and in 1990 was dominated by *Juncus* sp. apparently because of preferential cattle grazing of the grasses. No lasting hydrologic modifications, such as ditching or subsurface drainage, were done in the dairy wetland to convert it to a pasture. Therefore, restoration of the wetland involved revegetation of the area with native woody species but did not require hydrologic restoration. The wetland restoration was implemented as a nonpoint-source pollution control demonstration project (Clean Water Act Section 319) and was done in a manner similar to what would be done in an actual farm operation. The shape of the restored wetland and the pattern of planted trees was dictated by both the extent of wetland soils and by the presence of the upslope center pivot irrigation system (Fig. 1 and 2). The wetland restoration took in the entire area delineated as wetland soils, except for the portion under the center pivot (Vellidis et al., 1993b).

In January to March 1991, 0.54 ha of the total wetland area was replanted to correspond to Zones 1 and 2 of the RFB specification (Fig. 2). Zone 1 was planted with a mixture of 1-yr-old native hardwood saplings—yellow poplar (*Liriodendron tulipifera* L.), swamp blackgum [*Nyssa sylvatica* var *biflora* (Walt.) Sarg.], and green ash [*Fraxinus pennsylvanica* (Borkh.) Sarg.]. Zone 2 was planted with slash pine (*Pinus elliottii* Engelm.). The remainder of the wetland area (0.16 ha) was not planted to trees because of possible obstruction of the overhang of the center pivot system used to apply liquid manure in the uplands (Fig. 2). This area was occupied by native wetland grasses, primarily *Paspalum* sp. In February 1992 this area was planted to marsh cordgrass [*Spartina patens* (Aiton) Muhl.] but by the end of 1992, the *Paspalum* sp. had become reestablished with very low survival by *S. patens*.

The liquid manure applied in the upland area is about 80% NH_4^+-N, 10% organic N, and 10% NO_3^--N. The liquid manure averages about 200 mg total N L^{-1} (Williams et al., 1991). The field above the dairy wetland (Fig. 1) was in nonirrigated hay production until June 1991. The field above the dairy wetland received about 700 kg N ha^{-1} yr^{-1} from liquid manure beginning in June 1991 as part of an experiment on using liquid manure for year-round forage production (Vellidis et al., 1993a). This N input was contained in about 50 cm yr^{-1} of irrigation water. About 73% of the applied N was removed in forage crop production from the uplands (J. Davis, unpublished data). Denitrification in the uplands removed over 100 kg N ha^{-1} yr^{-1} from the uplands (R. Lowrance, unpublished data). We estimate that the residual N available for transport from the uplands was <100 kg N ha^{-1} yr^{-1}.

After operation of the center pivot for upland liquid manure application began, several unplanned applications of liquid manure occurred in the grassed area above the reforestation. Estimates of these unplanned and accidental inputs based on records of the center pivot operation and concentrations of N in the liquid manure were about 50 kg N ha^{-1} from June 1991 to March 1992. After March 1992 there was no waste application to the grass area. The reforested area received no direct waste application and the small amounts of waste in spray drift were not thought to be significant.

Hydrology Measurements

The dairy wetland is used for a detailed water quality study and is instrumented to provide samples for measurements of groundwater quality and depth and surface runoff quantity and quality. The methodology and instrumentation for the water quality study have been described in detail elsewhere (Vellidis et al., 1993b). Only groundwater N input data are reported here.

A network of 63 shallow groundwater wells was installed on an approximately 10-m grid. The wells were constructed using a 1-m section (belowground) of Triloc[1] well screen (0.25-mm screen) attached to a 50-cm riser of schedule 40 PVC pipe (aboveground). The borehole around the screened portion of the well was backfilled with coarse washed sand and the top of the well was sealed using bentonite clay. Wells were sampled biweekly beginning in January 1991 and continuing through August 1994. Water table depth was measured with an electronic water level indicator. Water samples were collected using a peristaltic pump. About 3 L of water was pumped from the well to purge it and then a 750 mL sample was collected in a nalgene sample bottle. Samples were analyzed for dissolved NO_3^- + NO_2^-, and total Kjeldahl N (in an acid digest) on a Lachat QuikChem Flow Injection Analyzer using methods 10-107-04-1-A (NO_3^--N) and 10-107-06-2-D (total Kjeldahl N).

Shallow groundwater inputs of N to the dairy wetland were

[1] Mention of a product or product name does not imply endorsement by USDA.

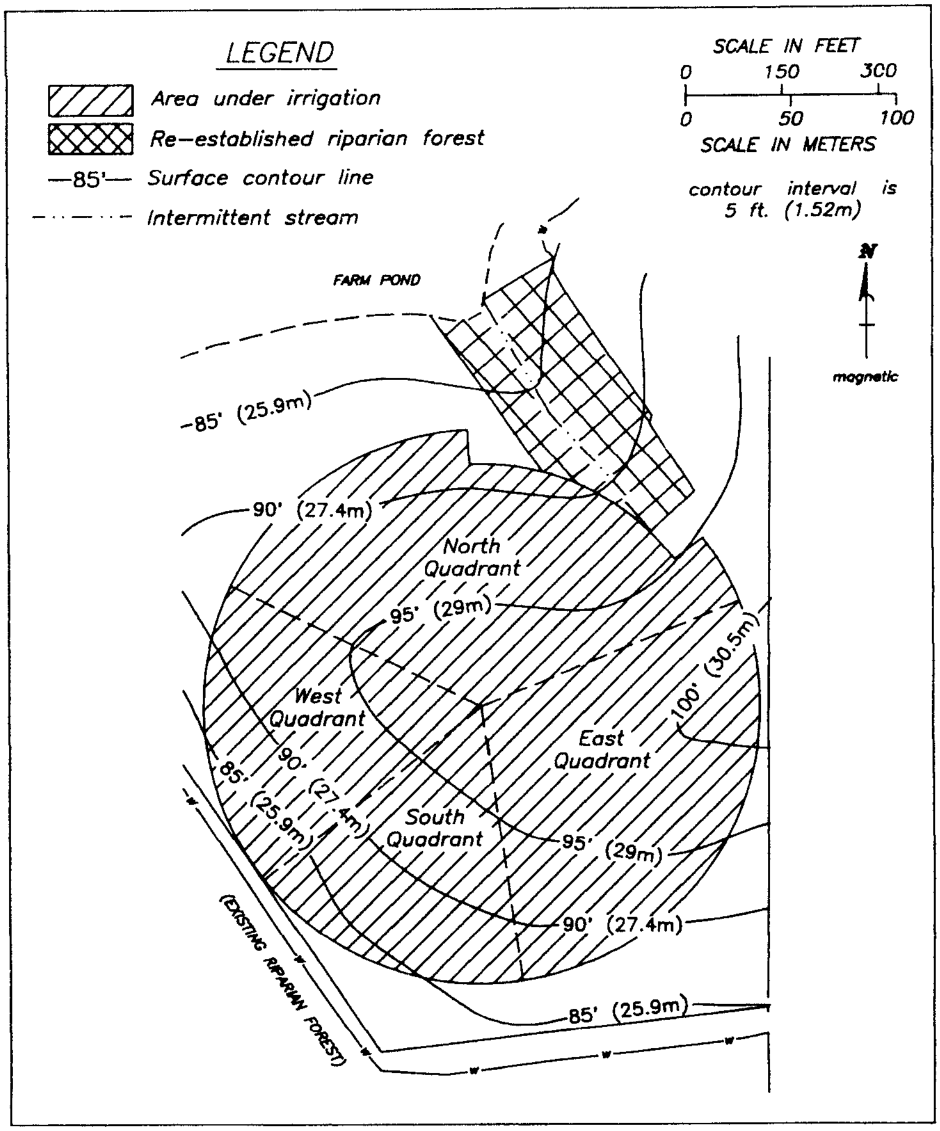

Fig. 1. Map of field site showing the liquid manure application area and the dairy wetland.

estimated for comparison to the denitrification estimates. Groundwater inputs were estimated by applying average bi-weekly concentrations for the wells at the perimeter (upslope border) of the dairy wetland to Darcian flows calculated through the saturated thickness of the top 1 m of soil. Saturated hydraulic conductivity was estimated to be 6 cm h^{-1} (Hubbard et al., 1985). Hydraulic gradient was calculated using water table elevations of the perimeter wells and the downslope wells near the stream.

Denitrification and Soil Variables

Soil samples were taken monthly for measurement of denitrification, soil inorganic N, and soil water content using procedures similar to those described in Lowrance and Smittle (1988) and Lowrance (1992). Sampling began in April 1991 so there were two sample dates before waste application to the uplands began in June 1991. Intact soil cores (2.5 cm diam.) were taken at 24 sites in the wetland by randomly choosing a groundwater well as a reference point and then sampling within 1 m of the well. The location of the sample was recorded and the sample sites were later mapped. The intact soil cores were taken at 6-cm increments to 24 cm and placed in 60-mL plastic syringes for incubation (Lowrance and Smittle, 1988). The incubation syringes were returned to the laboratory immediately after sampling and the head space on each sample was adjusted to 30 mL. A serum stopper was placed on the tip of the incubation syringe and 3 mL of acetylene was added to the head space after removal of 3 mL of air. The cores were incubated at 25°C for 5 h with 5 mL head space gas samples taken at 1 and 5 h. All incubations were done on the same day the cores were taken. Incubations were done at 25°C to approximate the mean annual soil temperature in the top 24 cm of soil. Soil temperatures were monitored at a weather station 0.5 km from the research site. All denitrification values reported here were corrected for actual soil temperature based on a Q_{10} of 2.

Analysis for N_2O concentration in the head space was done using the electron capture detector of a Hewlett Packard 5840A gas chromatograph (through August 1992) and a Varian 3600 gas chromatograph (September 1992–May 1993). The two gas chromatographs were operated under identical conditions using a 5-m Poropak Q column with a detector temperature of 350°C,

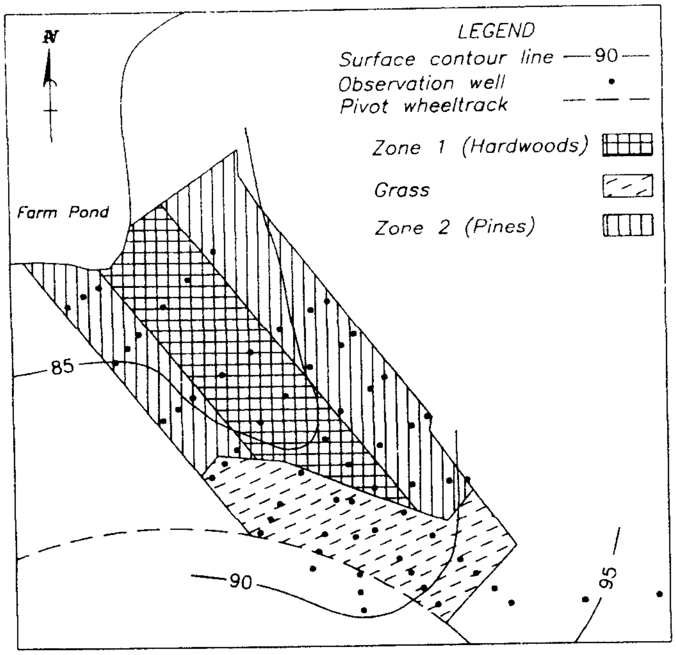

Fig. 2. Map of dairy wetland showing Zone 1, Zone 2, the grassed area, and the location of shallow groundwater wells.

oven and injector temperatures of 70°C, and AR/CH_4 (95%: 5%) as a carrier at a rate of 20 mL min^{-1}. All results were corrected for the solubility of N_2O in water based on values given in Moraghan and Buresh (1977). Time 2 (5 h) samples were corrected for the 5 mL removed from the head space in the Time 1 (1 h) sample.

After incubation, the length (L) and mass (G) of each soil core was determined. A 12.00-g subsample was then analyzed for NH_4^+-N and NO_3^--N following extraction with 40 mL of a 2 *M* KCl solution (Keeney and Nelson, 1982). Extracts were analyzed colorimetrically for NH_4^+-N and NO_3^--N using Lachat Methods No. 12-107-06-2-A and 12-107-04-1-B, respectively. Gravimetric soil water content (θ_m) was determined by drying the remaining field moist soil at 105°C for 3 d and reweighing. Bulk density (P_B) was determined for each soil core based on the total mass of dried soil (including the portion removed for KCl extraction) and the volume of the core ($V = \pi r^2$ L). Total porosity (TP) was determined as

$$TP = (1 - P_B/P_p)$$

where P_p = particle density, assumed to be 2.65 Mg/m^3.

Percent water filled pore space (%WFPS) was calculated as

$$\%\ WFPS = [\theta_m G/(TP\ V)] \times 100$$

Each 6-cm core sample was used to calculate a daily rate (g N_2O-N ha^{-1} d^{-1}). Calculations of total denitrification loss (kg N_2O-N ha^{-1} yr^{-1}) were based on summing the g N_2O-N ha^{-1} d^{-1} rates for all cores of a site. This calculation assumes that all gases produced in the top 24 cm would eventually leave the soil surface and be lost to the atmosphere. These daily rates for each 24-cm sample were multiplied by 365 to convert each summed daily rate into an annual rate. The means of these annual rates (288 per year based on 12 sampling dates and 24 sites per sampling date) were calculated for the 2 full years of waste application (June 1991–May 1992 and June 1992–May 1993). A total of 1152 core rates (288 summed rates for the entire 24-cm soil layer) were used to calculate each mean annual rate.

Denitrification, soil inorganic N, soil water content, and water-filled pore space data were not normally distributed. Therefore, data were analyzed using the NPAR1WAY Procedure of the Statistical Analysis System (SAS Inst., 1989) using the Kruskal-Wallis test. NPAR1WAY is a nonparametric procedure for testing whether the distribution of a variable has the same location parameter across different groups. The Kruskal-Wallis procedure tests the null hypothesis that the groups are not different from each other by testing whether the rank sums are significantly different based on a chi-square distribution (Sokal and Rohlf, 1981). In testing for differences among locations, years and depths using the NPAR1WAY procedure, the denitrification rate, soil inorganic N, soil water, or WFPS for each core was used as an individual observation. For these statistical analyses, the denitrification rate was expressed in two different ways—on an area basis (g N_2O-N ha^{-1} d^{-1}) and on a dry weight basis (μg N_2O-N kg dry $soil^{-1}$ d^{-1}).

RESULTS AND DISCUSSION

Annual Denitrification and Groundwater Nitrate

Denitrification rate, based on means of the summed 6-cm cores was 68 kg N_2O-N ha^{-1} yr^{-1} (SE = 7.10). Denitrification in the first year of waste application was 80.4 kg N_2O-N ha^{-1} yr^{-1} (SE = 12.4). Denitrification for the second year was 56.3 kg N_2O-N ha^{-1} yr^{-1} (SE = 6.9). The years were not significantly different.

The denitrification loss of N was substantial relative to the subsurface inputs of NO_3^--N and TKN. Denitrification was greater than the total calculated input of N (total N = NO_3^--N + TKN) from shallow groundwater moving in the top meter of soil and subsoil. The total N input was about 76% of the measured denitrification rate. The total N moving into the dairy wetland in shallow groundwater was about 51 kg N ha^{-1} yr^{-1}. The calculations of shallow groundwater movement rates, based on strictly Darcian flow, are an underestimate of total groundwater movement because preferential flow in macropores was not included (Blume et al., 1987; Hubbard and Sheridan, 1989). Other N inputs to the dairy wetland in precipitation, surface runoff, deeper groundwater movement, spray drift, and accidental waste application also occurred. In addition, it is not known what contribution of mineralized N was made by stored soil organic matter and roots derived from the forest vegetation removed in 1985. The N balance of the dairy wetland is not reflected by a simple comparison of denitrification and shallow groundwater. The amount of denitrification relative to the shallow groundwater input is an indication that the measured denitrification was substantial relative to subsurface N inputs.

The denitrification rates measured here were high relative to those measured in other studies, likely due to enrichment from the upland manure application system. Groffman (1994) summarized the published estimates of annual denitrification losses from wetlands. The published rates based on the acetylene block technique range from <0.1 kg N ha^{-1} yr^{-1} for unenriched sphagnum bogs and red maple swamps in the north-central USA and southern Canada (Urban et al., 1988; Merril and Zak, 1992) to 36 kg N ha^{-1} yr^{-1} for enriched red maple

Table 1. Denitrification, soil N, soil water, and water-filled pore space in Zone 1 (hardwoods) and Zone 2 (pines) of the riparian forest buffer and in the grassed area above the reforestation. Means followed by standard errors. Groups are different at the 0.01 level based on the Kruskal-Wallis test (except soil NO_3^--N).

Location	Denitrification	NO_3^--N	NH_4^+-N	Soil water	WFPS
	g N_2O-N ha^{-1} d^{-1}	mg kg^{-1}		kg kg^{-1}	%
Zone 1 (hardwoods)	42.5 (8.9)	0.80 (0.09)	8.13 (0.48)	0.27 (0.006)	70.3 (0.8)
Zone 2 (pines)	30.4 (3.6)	0.62 (0.03)	6.18 (0.17)	0.21 (0.003)	62.8 (0.6)
Grass	68.8 (7.8)	0.63 (0.04)	6.34 (0.24)	0.21 (0.003)	68.7 (0.7)

swamps in Rhode Island (Hansen et al., 1994). Riparian forests within a few kilometers of the dairy wetland site in this study had denitrification rates of 31 kg N ha^{-1} yr^{-1} in the top 10 cm of soil when there was no enrichment due to animal waste (Lowrance et al., 1984). Annual rates reported in this study (2-yr mean of 68 kg N ha^{-1} yr^{-1}) exceed any of the published rates. In an unpublished study also conducted a few kilometers from the dairy wetland (Hendrickson, 1981), a mature hardwood forest site receiving runoff from a swine parlor had a denitrification rate of 198 kg N ha^{-1} yr^{-1} in the top 10 cm of soil. Of the limited number of published studies, a number have reported substantial denitrification responses to N subsidies in both forested and nonforested wetlands (Dierberg and Brezonik, 1983; Broderick et al., 1988; Cooper, 1990; Hanson et al., 1994).

Spatial Effects

Denitrification was higher in Zone 1 than in Zone 2, but the differences were not significant based on the Kruskal-Wallis test (Table 1). Soil water, water-filled pore space, and soil NH_4^+-N levels were significantly higher in Zone 1 than in Zone 2. Given the young age of the trees during the period of study, it is unlikely that the vegetation differences would lead to differences in denitrification rates. Significantly higher levels of soil water and NH_4^+-N did not lead to significant differences in denitrification.

The grassed area above the reforestation had significantly higher denitrification rates than either of the forest zones (Table 1). Although denitrification was highest in the grassed area, the soil NH_4^+-N, soil water, and water-filled pore space were significantly higher in Zone 1 of the forest, based on the Kruskal-Wallis test (Table 1). The present study does not allow us to determine the reason for significantly higher denitrification rates in the grassed area. At least two factors could be involved. This area received about 50 kg N ha^{-1} of direct waste application. This waste application, mostly NH_4^+, was not reflected in the soil NH_4^+ levels (Table 1). It is also possible that the C from the grass biomass is a more readily available energy source for denitrifiers that the other herbaceous weeds and young trees in Zones 1 and 2. Groffman et al. (1991) hypothesized that C derived from roots of different grass species may differ in availability to denitrifiers. In comparing the grassed area to Zones 1 and 2 of the forest, it is possible that both the mass and availability of root derived C differs.

Temporal Effects

Based on groupings of individual core rates, denitrification was lower (by ≈3-fold) in April and May 1991 (before waste application began) than in the same months for 1992 and 1993 (Table 2). The April and May 1991 rates reflected the N subsidy conditions of a typical wet pasture that would receive some N in subsurface flow from upland areas. The April and May rates for 1992 and 1993 reflected the direct and indirect manure subsidy. Soil NH_4^+-N was higher in April and May of both 1992 and 1993 than in 1991 (Table 2). Soil NO_3^--N levels were higher in April and May 1993, only. The %WFPS was significantly higher in 1991 (Table 2). Average soil temperature, measured at the nearby weather station, was also highest in 1991. The March to May precipitation was higher in 1991 than 1992 or 1993 with a total of 48.3 cm in 1991 compared with 16.8 and 18.2 cm, respectively, in 1992 and 1993. The long-term average precipitation for March to May is 30.9 cm (Batten, 1980). These data suggest that the denitrification increases in 1992 and 1993 were caused by the initiation of waste application and that the increase shows that denitrification in the restored RFB responded to increased N loads from the upland. The differences between 1991 and 1992/1993 were apparently caused by the N subsidy in waste application, because soil water and temperature levels were lower in April and May 1992 and 1993 while soil inorganic N levels showed an opposite pattern. Mean values for %WFPS for all 3 yr were above 65%. At %WFPS above 60%, denitrification should proceed at high rates based on availability of C and N (Linn and

Table 2. Denitrification, soil N, and water-filled pore space for April and May samplings in 1991 (before waste application), 1992, and 1993. Means followed by standard error (except soil temperature). Groups (years) are different at the 0.01 level based on Kruskal-Wallis test. Soil temperature are means of the means of maximum and minimum 10-cm temperatures.

Year	Denitrification	NO_3^--N	NH_4^+-N	WFPS	Soil temperature
	g N_2O-N ha^{-1} d^{-1}	mg kg^{-1}		%	°C
1991 (n = 182)	12.2 (1.0)	0.21 (0.04)	2.87 (0.21)	76.7 (1.19)	23.7
1992 (n = 192)	41.2 (10.9)	0.22 (0.03)	8.83 (0.43)	65.4 (1.21)	20.6
1993 (n = 192)	40.2 (10.6)	0.72 (0.08)	5.25 (0.43)	66.9 (1.21)	22.1

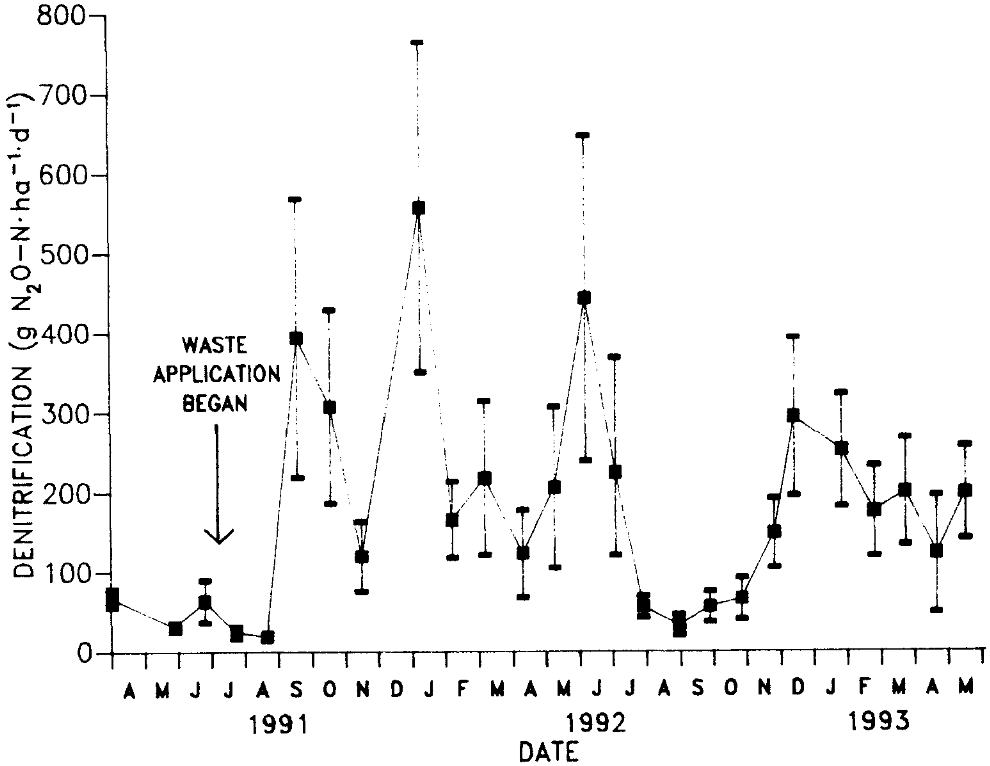

Fig. 3. Mean denitrification rates (plus and minus one standard error) for monthly samples from April 1991 through May 1993.

Doran, 1984). Comparison of two samplings in each of 1991, 1992, and 1993 is a limited test of the effects of the upslope manure application and direct and indirect nutrient inputs to the wetland.

Denitrification increased by September of 1991, about 3 mo after waste application began (Fig. 3). The maximum daily rate for any month was 558 g N_2O–N ha^{-1} d^{-1} in January 1992. The second highest daily rate occurred in June 1992 and the third highest rate was in September 1991. Although periods of low denitrification occurred, rates were generally >100 g N_2O–N ha^{-1} d^{-1}. Lowest denitrifcation rates were in Summer and Fall 1992 with rates in July through October generally <100 g N_2O–N ha^{-1} d^{-1}. Rainfall during this time period was low, especially in July and September. July rainfall was 8.02 cm and only 0.77 cm of rainfall was recorded in September 1992 before the soil sampling date. Mean WFPS for July and September samples were the lowest for any two sample dates with means of 52.2 and 48.3% WFPS, respectively.

Depth Effects

Denitrification rates were significantly higher in the 0- to 6-cm depth than in the three lower depth increments (Table 3). There were no significant differences in either the area rate or mass rate below the 6-cm depth. On both the area basis and mass basis, there was greater total denitrification in the bottom 18 cm of soil (6–24 cm) than in the top 6-cm soil layer. This indicated that NO_3^- enriched shallow groundwater moving anywhere in the top 24 cm would be affected by denitrification. Shallow groundwater was often within the top 24 cm of soil. Stratification of core denitrification rates and denitrification potentials has been observed in all riparian wetland soils sampled in the region (Lowrance, 1992;

Table 3. Denitrification and soil N for four depths for all dairy wetland samples. Means followed by standard error in parentheses. Groups (depths) are different at the 0.0001 level based on Kruskal-Wallis test (except NO_3^-–N, which is not significantly different).

	Denitrification		Soil N	
Depth	Area rate	Mass rate	NO_3^-–N	NH_4^+–N
cm	g N_2O–N ha^{-1} d^{-1}	μg N_2O–N kg^{-1} dry soil d^{-1}	mg kg^{-1}	
0–6 (n = 621)	79 (10.8)	50 (4.7)	0.81 (0.06)	11.12 (0.45)
6–12 (n = 621)	35 (5.8)	25 (2.5)	0.62 (0.04)	5.72 (0.20)
12–18 (n = 620)	32 (4.0)	29 (3.3)	0.61 (0.05)	4.97 (0.17)
18–24 (n = 620)	31 (4.6)	28 (4.0)	0.59 (0.05)	4.62 (0.15)

Ambus and Lowrance, 1991; Hendrickson, 1981). Hendrickson (1981) found a similar degree of stratification as in this study with rates below the top 8 cm that were one-half to one-third of the surface soil rates. Stratification more extreme than found in this study was found at two less enriched sites (Ambus and Lowrance, 1991). Denitrification potentials were 8 to 14 times higher in 0- to 5-cm samples than in 5- to 10-cm samples in two areas receiving indirect inputs of N fertilizer (Ambus and Lowrance, 1991).

Soil NH_4^+-N levels were also highly stratified (Table 3). NH_4^+-N was highest in the surface soil, reflecting the input of liquid manure with high NH_4^+ from either direct waste application, drift, or rain-induced surface runoff following waste application. Surface soil NH_4^+-N levels in this study were about twice the highest levels found at two less enriched sites (Ambus and Lowrance, 1991). There were no significant differences in %WFPS or NO_3^--N among the four depths. The mean %WFPS in all depths was above 60%. The mean NO_3^--N levels were very low, possibly reflecting the high rates of denitrification found in the top 24 cm of soil.

IMPLICATIONS FOR RIPARIAN WETLAND RESTORATION

The high levels of denitrification in the dairy wetland showed that NO_3^- levels in soil and shallow groundwater could be substantially reduced by denitrification in this restored riparian wetland. The levels of denitrification were greater than a conservative estimate of shallow groundwater N input, but were not compared with total NO_3^- availability as part of this report. Overall, the results show that a riparian wetland, restored to conditions described in USDA specifications, can have a high level of NO_3^- removal via denitrification. It is likely that during the course of this study, the reforestation had very little effect on the measured denitrification rates. The grassed area above the reforested zones was a very effective NO_3^- filter. If Zone 3 (herbaceous filter strip) of the riparian forest buffer system is located in wet conditions below a liquid manure application system as was the grassed area in this study, Zone 3 would likely have similarly high denitrification rates. Alternatively, the denitrification rates reported here would indicate that a grass riparian buffer could have rates of denitrification as high or higher as a newly restored forest. The relative effects of mature forest vs. grass cannot be evaluated with these results, nor can the effects of the forest soil that was developed under the mature riparian forest, which was removed in 1985.

It appears that restoration of formerly forested wetlands that were previously converted to wet pastures can have a significant role in removing NO_3^- via denitrification. The overall water quality benefits of the forest restoration will not be realized in the short time period after forest restoration covered by this study. Regardless of whether the wetland is responding to inputs from the liquid manure system upslope, the levels of denitrification reported here represent a significant loss of N from the combined forage production–wetland system. The dairy wetland therefore was important for NO_3^- removal. Systems of this sort can be used as integral parts of a landscape management system for environmentally sound use and disposal of N from liquid manure.

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