Nutrient Interception by a Riparian Forest Receiving Inputs from Adjacent Cropland

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ABSTRACT

To investigate the ability of riparian forest to intercept nutrients leaving adjacent cropland, we examined changes in the chemistry of groundwater flowing from a corn (*Zea mays* L.) field through a riparian forest. This study provided a comparison to previous studies of a different forest. We sampled groundwater from a transect of wells, and used a Br⁻ tracer to confirm that groundwater moved laterally along the transect through the forest. As groundwater flowed through the forest, NO_3^- concentrations decreased from about 8 mg/L at the edge of the corn field to <0.4 mg/L halfway through the forest. Dissolved organic N and NH_4^+ increased by less than 0.1 mg/L, and dissolved organic C did not change with distance. Sulfate remained constant with distance until midway through the forest, where it began to increase. Chloride concentration rose until midway through the forest, then fell. Values of pH increased from under 5 at the edge of the corn field to over 7 at the stream bank, perhaps as a result of the NO_3^- consumption. Most of the change in NO_3^- occurred abruptly at the edge of a floodplain within the forest. There the water table was closest to the surface and soil Eh below the water table was less than −90 mV. Such strongly reducing conditions may have promoted denitrification in the floodplain. In contrast, soil Eh on the adjacent hill slope was above 500 mV, too high to support denitrification. There were only slight seasonal changes in groundwater chemistry. We also studied the net annual accretion of sediment in the riparian forest by measuring changes in the elevation of the soil surface. There was little or no accretion in the forest, but along a path of overland storm flow there was net erosion. Thus, nutrient retention by this forest, in contrast with the forest we previously studied, was entirely a below ground process. Functional differences within sections of this forest and among different riparian forests suggest a need for research on the factors that control nutrient retention.

S EVERAL STUDIES have shown that riparian forests can take up N, P, and sediment from discharges leaving adjacent croplands (e.g., Lowrance et al., 1984a,b; Peterjohn and Correll, 1984, 1986; Jacobs and Gilliam, 1985; Pinay and Decamps, 1988; Correll and Weller, 1989; Haycock and Pinay, 1993). This finding has important implications for land use management. Removal of NO_3^- reduces the hazards of high NO_3^- concentrations in drinking water. Nitrate removal also raises the pH of the groundwater, thus reducing acidification of downstream aquatic ecosystems. Removal of N and P also reduces eutrophication of aquatic ecosystems. Removal of sediment from surface runoff reduces turbidity and sediment deposition downstream.

Strong effects of riparian forests on agricultural runoff have consistently been reported for coastal plain forests with shallow underlying aquicludes. The aquicludes force cropland groundwater flowing toward the stream to move laterally through the near-surface layers of the riparian forest. Such forests can remove large percentages of materials they receive from the croplands. Moreover, the removals can take place over narrow widths of forest.

For example, riparian forests can retain 70 to 90% of the total N inputs from adjacent croplands (Lowrance et al., 1984b; Peterjohn and Correll, 1984), and most of the NO_3^- removal occurs within 20 m of the forest–field boundary (Peterjohn and Correll, 1984; Jacobs and Gilliam, 1985).

Many questions remain about the function of riparian forests. For example, how similar are the functions of different forests? How much forest is necessary to remove nutrients effectively? Does nutrient removal occur evenly throughout the forest or are there particular areas within the forest where nutrient removal is greatest? Is nutrient removal constant throughout the year?

In this study we examined the changes in the chemistry of groundwater flowing out of a corn field and through a riparian forest, and we measured the net sediment accretion in the forest. Our goals were to test the generality of our previous findings at a similar forest (Peterjohn and Correll, 1984, 1986; Correll and Weller, 1989) and to examine the spatial and seasonal variability of groundwater chemistry and sediment accretion in the forest.

METHODS

We studied a riparian forest located on a hill slope and a floodplain adjacent to cropland on the Delmarva peninsula near Centreville, MD (39° 8′ N, 75° 56′ W). The forest is deciduous, dominated by beech (*Fagus grandifolia* Ehrh.) and tulip poplar (*Liriodendron tulipifera* L.) on the hill slope and by sycamore (*Platanus occidentalis* L.) and box elder (*Acer negundo* L.) on the floodplain near the stream. The stream is the northern branch of Southeast Creek, a third-order tributary of the Chester River. The cropland was planted to corn during the study but has been rotated to soybean [*Glycine max* (L.) Merr.]. The corn was grown by no-till methods. Surface sediments in this coastal plain region are part of the Pensauken formation, which forms the Columbia aquifer, which is perched on the less permeable sediments of the Chesapeake Group throughout the region (Bachman and Wilson, 1984). The Pensauken sediments are feldspathic sands, which become gravelly at the bottom of the formation just above the clayey sediments of the Chesapeake Group at about 10 m above sea level in our study area (Bachman and Wilson, 1984). The thickness of the underlying clayey sediments is variable. At our study site we verified that the clay extended at least 0.5 m below the Pensauken sediments. Deeper drillings near our site penetrated 2 to 4 m into the clay (Bachman and Wilson, 1984).

In July 1990, we dug a transect of groundwater sampling wells using a 5.5-cm diam. bucket auger. The wells were lined with a 5-cm diam. PVC pipe perforated from the surface down to the Chesapeake clay at a depth of 1.2 to 5.3 m (Fig. 1). Initially, we installed three replicate wells at each of four distances from the stream (Fig. 2). Later, six additional wells were added in the middle of the transect (Fig. 2) to examine in detail the region where NO_3^- concentration changed rapidly with distance from the corn field. The wells were pumped out several times after installation to remove loose particulate matter.
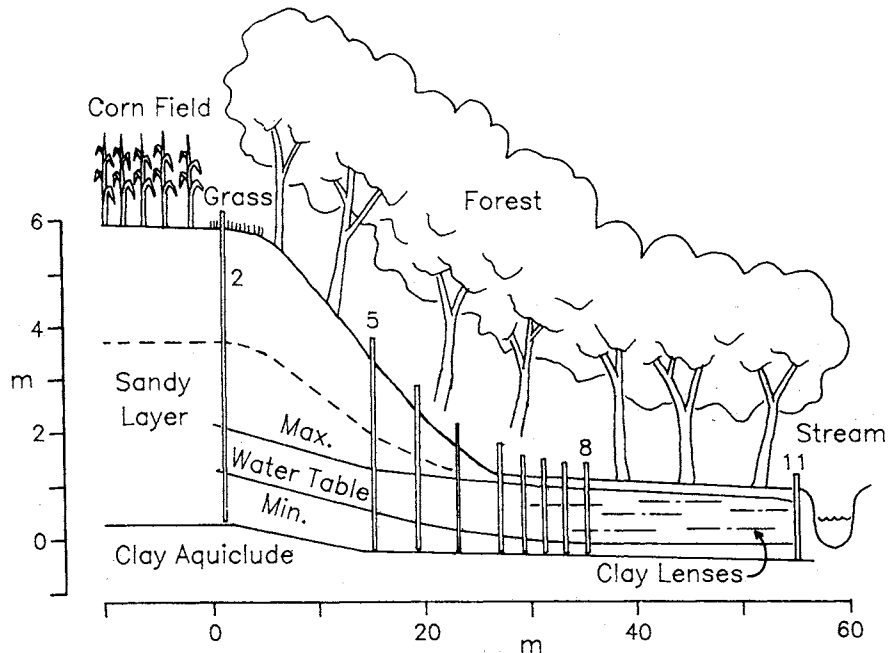
We sampled the wells about once a month from July 1990

Fig. 1. Cross-section of study site showing wells along the center of the transect, sediment layers, and minimum and maximum water table elevations.
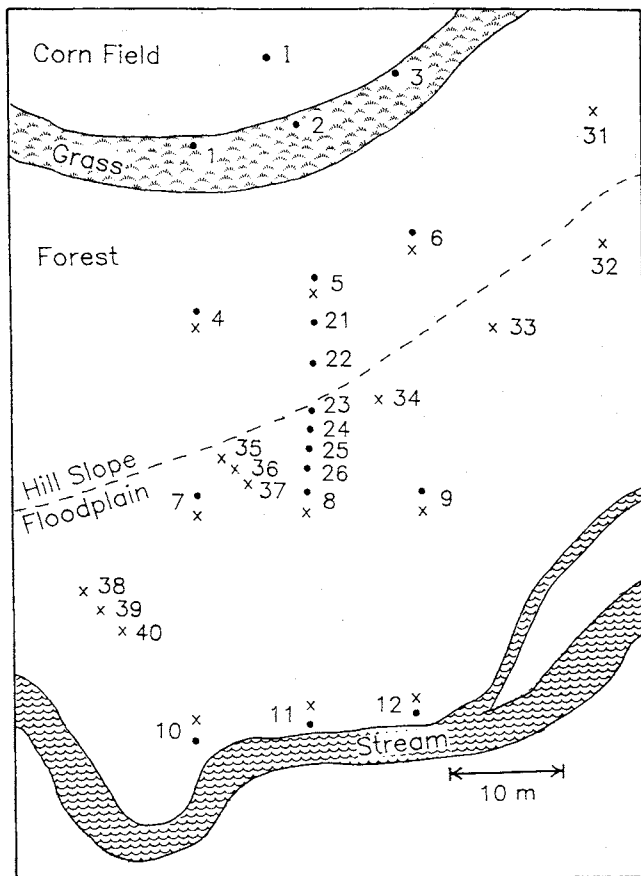

Fig. 2. Map of study site showing groundwater sampling wells (●) and deposition posts (x). Posts 31–40 are distributed along the path of overland storm-water flow. Point *I* in the cornfield marks an auger hole where NaBr tracer was added.

to August 1991 by first pumping the wells dry, then sampling the water that immediately refilled them. Usually we mixed together equal volumes of sample from the three replicate wells to produce one composite sample for each distance from the stream. Just prior to pumping, the water level in the wells was measured by lowering a float on a measuring tape.

The samples were filtered through 0.45-μm membrane filters and analyzed for dissolved N, P, and organic C species. Total P was digested to phosphate with perchloric acid (King, 1932). Phosphate in the digestate as well as in the undigested samples was analyzed by reaction with stannous chloride and ammonium molybdate (APHA, 1976). Dissolved organic P (DOP) was calculated by difference. Total Kjeldhal N was digested with sulfuric acid, Hengar granules, and hydrogen peroxide (Martin, 1972). The resultant NH_4^+ was distilled and analyzed by Nesslerization (APHA, 1976). In the undigested samples, NH_4^+ was oxidized to NO_2^- with alkaline hypochlorite (Strickland and Parsons, 1972), NO_3^- was reduced to NO_2^- with Cd amalgam, and NO_2^- was analyzed by reaction with sulfanilamide (APHA, 1976). When NO_3^- concentrations exceeded 1 mg/L, NO_3^- was analyzed with a Dionex Series 4000i ion chromatograph (Dionex Corporation, Smyrna, GA). We present data on the sum of NO_2^- and NO_3^- concentrations. Dissolved organic N (DON) was calculated by difference. Dissolved organic C (DOC) was analyzed by drying samples at 60 °C, followed by reaction with potassium dichromate in 67% sulfuric acid at 100 °C for 3 h (Maciolek, 1962) with $HgSO_4$ added to complex halides (Dobbs and Williams, 1963). Organic C was calculated from the amount of unreacted dichromate measured colorimetrically (Maciolek, 1962; Gaudy and Ramanathan, 1964). The pH was measured with a Ross electrode after the sample equilibrated with air. Conductivity was measured with a Yellow Springs Instruments Model 32 conductivity meter (Yellow Springs Instruments, Yellow Springs, OH), and SO_4^{2-} and Cl^- were analyzed with the Dionex ion chromatograph.

The path of groundwater flow was traced by adding NaBr to groundwater and observing the appearance of elevated Br^- concentrations in wells downslope. Bromide was analyzed by ion chromatography. We introduced 1.5 L of 8 *M* NaBr into a hole augered in the corn field and observed its subsequent appearance in Well 2 at the edge of the field (Fig. 2). Then we added 1.5 L of Br^- solution to Well 2, and observed its

appearance in Wells 4 and 6 farther down the transect. Finally we added 1.5 L of Br⁻ solution to Wells 4 to 6 and observed its appearance in Wells 7 and 8. In this way, we verified that groundwater moves through the forest toward the stream.

We used slug tests (Bouwer and Rice, 1976; Bouwer 1989) to measure hydraulic conductivity in the saturated zone of the Columbia aquifer and in the underlying clay. For these tests, four piezometers extending down into the Columbia aquifer and two into the underlying clay (Fig. 2) were installed in holes dug with a 5.5-cm bucket auger. The holes were lined with 5-cm diam. PVC pipes perforated along the bottom 20 cm. Sand was packed around the end of each piezometer and a bentonite seal was packed on top of the sand.

We used platinum electrodes (Faulkner et al., 1991) to measure Eh in the soil below the water table at four distances along transect (Fig. 2). The electrodes consisted of 1-mm diam. platinum wire soldered to insulated 2-mm diam. single-stranded copper wire. The solder joint was covered by epoxy leaving 8 mm of the platinum wire exposed. The Eh was measured with an Orion model 407 ion meter (Orion Research, Cambridge, MA) as mV potential relative to a calomel reference electrode and converted to mV relative to hydrogen by adding 250 mV (assuming groundwater temperature averaged about 13 °C). The electrodes were calibrated with saturated quinhydrone solution buffered at pH 7 (40.8 mV at 25 °C) and pH 4 (218 mV). Each electrode was buried below the water table in a hole dug with a screw auger 1 m from a groundwater well. For Eh readings in the field, the calomel reference electrode was lowered into the water in the nearby well.

We measured the net accretion (or erosion) of soil in the transect using a grid of vertical posts projecting about 30 cm above the soil surface. Posts were placed 2 m from each well and additional posts were placed in a path of overland flow through the study site (Fig. 1). Initially, a metal washer loosely fitted over each post was placed on the soil surface, and the distance from the top of the post to the washer was measured. After a year, we measured the distances from the top of the post to the litter layer, the soil surface, and the washer. If deposition occurred, the washer would be buried by the deposited material. If erosion occurred, the washer would be further below the top of the post after a year. If both erosion and deposition occurred, the washer would be farther below the top of the post and buried.

RESULTS

The chemistry of groundwater differed dramatically among wells at different distances from the corn field, but did not change very much seasonally. Nitrate concentration showed the greatest spatial variation, dropping from about 8 mg/L at the edge of the corn field to less than 0.4 mg/L halfway through the forest (Fig. 3). Most of the change in NO_3^- concentration occurred 25 to 35 m from the edge of the corn field. In contrast, DON and NH_4^+ concentrations increased by less than 0.1 mg/L with increasing distance from the corn field, and DOC did not change with distance from the corn field. Sulfate remained constant with distance from the corn field until midway through the forest, where it increased in conjunction with the decrease in NO_3^-. Values of pH increased from under 5 at the edge of the corn field to over 7 at the stream bank. Groundwater near the stream was unique. It had high DOP and phosphate concentrations, high conductivity, and low Cl⁻ concentration compared with groundwater from other locations along the transect. Soil Eh below the water table also changed markedly along the transect, averaging about 210 mV at the edge of the corn field, about 570 mV on the hill slope, and −95 to −260 mV on the floodplain (Table 1).

Groundwater chemistry varied little through time. However, NO_3^- concentration at the edge of the corn field rose slightly in the winter and spring, whereas NO_3^- concentrations 15 m into the forest declined (Fig. 4). Groundwater DOC and DON concentrations declined from late July to late October. Dissolved organic C concentrations rose again the following May through July, suggesting a summer maximum and winter minimum in concentration, but DON concentrations did not rise in the late spring. Ammonium and phosphate concentrations 35 m into the forest rose during June through August. Of course, with only 1 yr of data we cannot conclude that any of these temporal changes are part of a repeating seasonal cycle.

Interpreting the spatial variation in groundwater chemistry requires knowledge of the groundwater hydrology. Observations of Br⁻ movement verified that groundwater moved laterally from the cropland through the forest to the stream. Lateral movement is induced by the relatively impermeable clay underlying the much more permeable surface sediments. Hydraulic conductivity within the clay ranged from 1.7×10^{-4} to 2.3×10^{-4} cm/h, but in the surface sediments it was more than a thousand-fold greater (0.7–18 cm/h). The hydraulic characteristics of our site are typical for the Columbia aquifer, although the hydraulic conductivity of the surface sediments at our site is at the low end of the range for region (Bachman and Wilson, 1984).

Water table elevation declined toward the stream, suggesting groundwater flow nearly parallel to the transect and perpendicular to the stream. Water table elevation varied seasonally with the cycle of evapotranspiration (Fig. 5). The seasonal variation in water table elevation resulted in variation in groundwater flow. We used water table elevation and hydraulic conductivity data and methods of Todd (1964) to estimate groundwater flow between Wells 5 and 8, where most of the NO_3^- decline took place. Flow through a 1 m wide by 20 m long path ranged from 1.2 L/h in fall to 2.6 L/h in spring. These rates of groundwater flow and the difference in NO_3^- concentrations between Wells 5 and imply removal of NO_3^--N at a rate of 0.4 mg/m² per h in fall and 0.9 mg/m² per h in spring.

Annual soil deposition or erosion, measured by elevation changes at posts (Fig. 2), was highly variable (Table 2). Erosion predominated along the overland flow channel with net erosion of 1 to 4 mm in most locations and net erosion of 10 to 30 mm in one area (posts 35–37, Fig. 2). Burial of the washers along the overland flow path indicated that, although there was net erosion, both erosion and deposition occurred at the same locations. Outside of the overland flow path, there was neither erosion nor deposition in the forest near the cropland, and 1 to 5 mm of erosion nearer the stream at Posts 7 to 12. However, one location near Well 10, at the edge of the overland flow path, showed 18 mm of deposition. Litter layouts of up to 60 mm built-up throughout the site over the year. At one location (Post 21), the post snagged litter carried in overland flow and built up an artifically thick layer 190 mm deep.

DISCUSSION

Others have found, as we have, that NO_3^- concentrations in groundwater are much lower in riparian forests than in adjacent croplands (Lowrance et al., 1984a,b;

Peterjohn and Correll, 1984, 1986; Jacobs and Gilliam, 1985; Correll and Weller, 1989; Weil et al., 1990; Pinay and Decamps, 1988; Haycock and Pinay, 1993). Such concentration differences could arise if groundwater in the forest originates from infiltration of rainwater or emergence of NO_3^--poor groundwater from an underlying aquifer. However, in our study, and those cited above,

an aquiclude forces the lateral flow of groundwater and blocks emergence of deep groundwater. Moreover, infiltration of rainwater in our forest evidently cannot account for the change in NO_3^- concentration, because corresponding change in Cl^- concentration did not occur (Fig. 3). Thus, we conclude that the decline in NO_3^- concentration with distance from the cropland must be
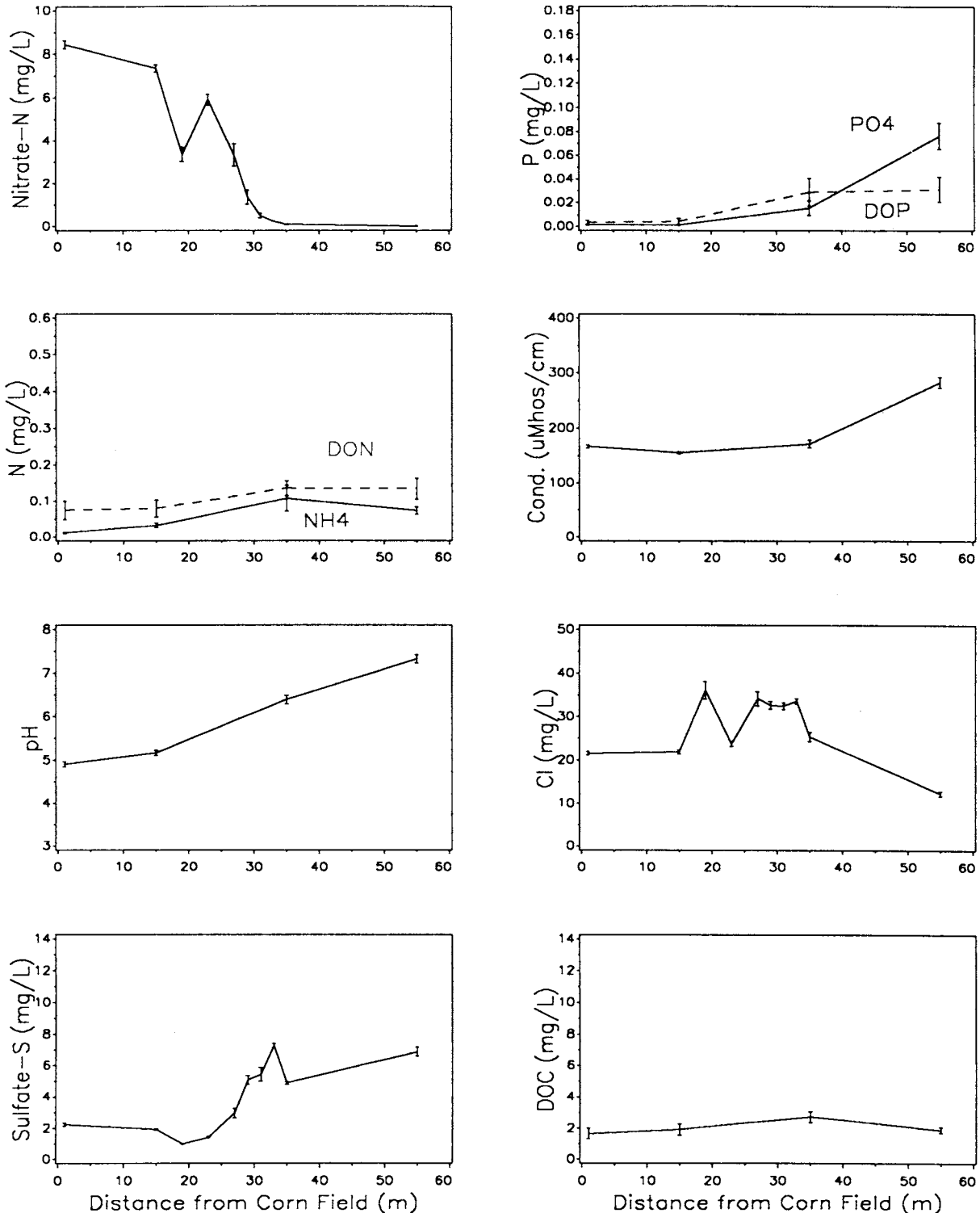

Fig. 3. Annual mean concentrations in groundwater vs. distance from cornfield along the transect through the riparian forest. Brackets are ± standard errors of means of different dates.

Table 1. The Eh in saturated soil at different depths below the soil surface. Eh is the average (± SD) of monthly measurements from December 1991 to March 1992. Eh electrodes were 1 m from nearest well. Well positions are shown in Fig. 2.

Location	Nearest well	Depth	Eh
Edge of field		m	mV
	2	4.5	222 ± 15
	3	4.1	197 ± 10
Hill slope			
	5	2.8	560 ± 66
	6	2.9	580 ± 12
Floodplain			
	8	1.0	−132 ± 33
	9	1.2	−130 ± 36
	11	1.0	−95 ± 21
	12	1.0	−510 ± 12

due to removal of NO_3^- from groundwater flowing laterally from the cropland through the forest.

The amount of removal implied by the drop in NO_3^- concentration depends on the rate of lateral groundwater flow. Various methods have been used to estimate this rate. For example, hydrologic models (Lowrance et al., 1984a; Jacobs and Gilliam, 1985), stream hydrograph analysis (Peterjohn and Correll, 1984), and watershed budgets of Cl^- (Peterjohn and Correll, 1986). Our estimates of groundwater flow, based on methods similar to Lowrance et al. (1984a), imply removal of NO_3^--N from groundwater at rates of 0.4 to 0.9 mg/m² per h ($\approx$60 kg NO_3^--N/ha per yr) in the section of forest where removal rates were highest. In comparison, estimates for total annual N removal in two other riparian forests are 40 kg/ha (Peterjohn and Correll, 1984; 1986) and 70 kg/ha (Lowrance et al., 1984b).

How is the NO_3^- removed? It is clear that the NO_3^- is not being transformed into DON or NH^+_4 dissolved in groundwater, because their concentrations do not increase very much in conjunction with the decrease in NO_3^- concentration (Fig. 3). Nitrate may be removed both by assimilation by trees and by denitrification. Most of the change in NO_3^- occurs abruptly 25 to 35 m from the forest/field boundary (Fig. 3) where the hill slope ends and the floodplain begins (Fig. 1). In this zone, groundwater is forced closer to the soil surface by the underlying aquiclude. There, NO_3^- removal may be promoted as groundwater comes in increasing contact with tree roots, and as the organic-rich surface soils become increasingly waterlogged, producing low Eh conditions favoring denitrification. Lowrance (1992) concluded that denitrification could remove large amounts of NO_3^- from riparian forest groundwater if the water table was within 60 cm of the soil surface where denitrification rates were highest. In our forest, the water table is often near the surface in the floodplain (Fig. 5). Denitrification requires Eh levels below 300 mV (Kralova et al., 1992). Such levels were found in the floodplain but not on the hill slope (Table 1). Others have found higher dentrification potential in floodplains than in hill slope soils in both deciduous and coniferous riparian forests (Davidson and Swank, 1986; Gregory et al., 1991; Groffman et al., 1991, 1992). However, there are few estimates of actual denitrification rates in riparian forests, because of the general lack of accurate field methods (Tiedje et al., 1989; Payne, 1991). Deciduous forests are thought to have annual denitrification rates of up to 50 kg N/ha with
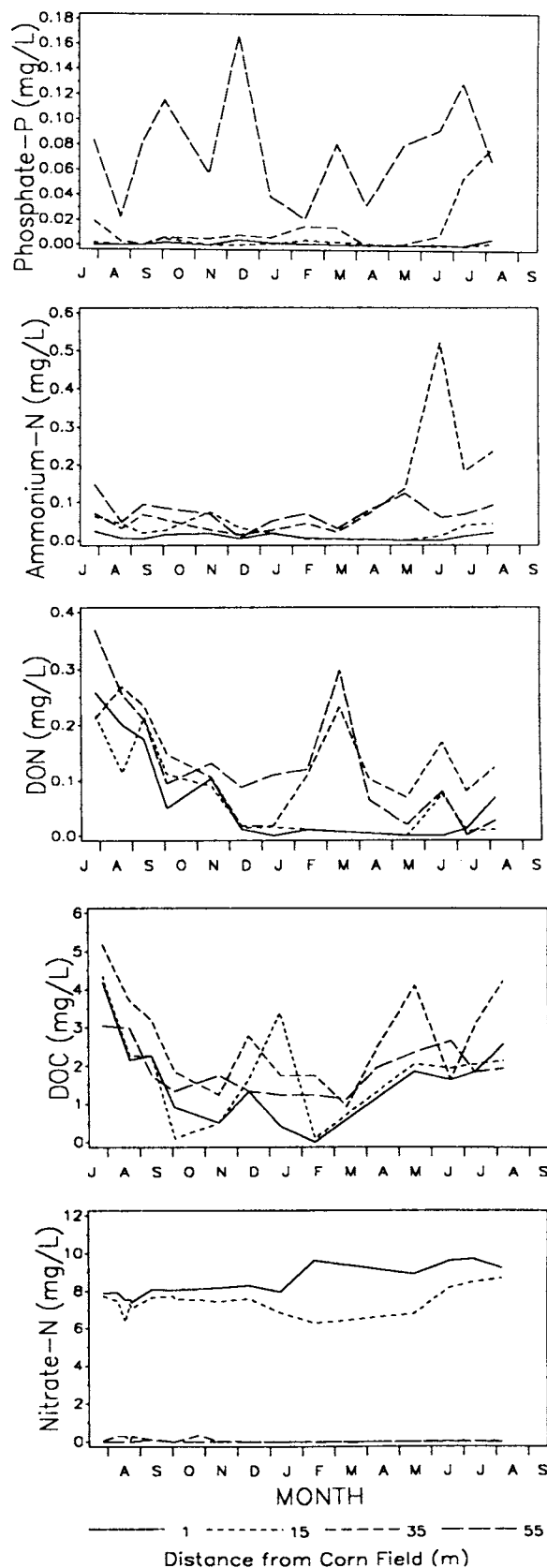

Fig. 4. Concentration versus date at different distances from the cornfield along the transect through the riparian forest.
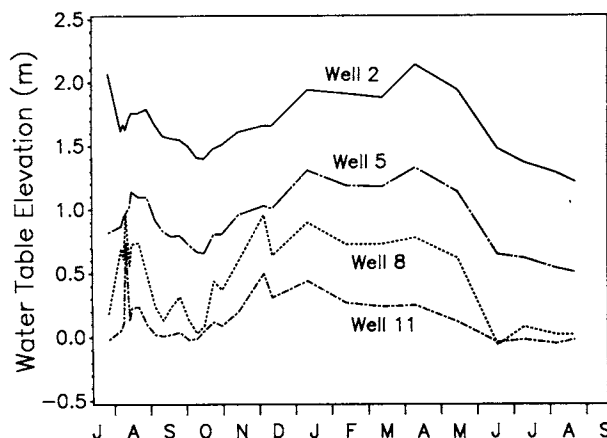
Fig. 5. Elevation of watertable (m) at center wells in the transect vs. date. The elevation scale is the same as the vertical scale in Fig. 1.

Table 2. Accretion or erosion, and litter accumulation after 1 yr measured at posts. Accretion is the build up of soil at the post. Negative accretion is erosion. Burial of the washer is the depth of the washer below the soil. Posts numbered 4–12 are located near wells of the same number (Fig. 2). Posts numbered 31–40 are distributed along the path of overland storm-water flow.

Post no.	Accretion	Burial of washer	Litter
		(mm)	
4	0	0	14
5	0	2	6
6	0	0	54
7	−3	0	14
8	−5	0	33
9	−3	0	28
10	18	7	0
11	−1	0	30
12	−1	0	2
31	−3	27	190
32	2	5	60
33	0	10	5
34	4	9	20
35	−10	0	8
36	−21	4	0
37	−30	7	12
38	0	0	8
39	−3	0	5
40	−1	0	8

most between 0 and 10 kg N/ha (Davidson et al., 1990), but higher annual rates have been reported for riparian forests (32 kg/ha, Lowarance et al., 1984b; 6–194 kg/ha, Pinay and Decamps, 1988; 475 kg/ha, Fustec et al., 1991). Such rates could account for much of the NO_3^- removal we estimated (≈ 60 kg NO_3^- -N/ha per yr). Assimilation could also account for much of the annual removal. Net long-term incorporation of N into aboveground biomass in other riparian forests has been estimated as 15 kg/ha per yr (Peterjohn and Correll, 1984) and 52 kg/ha per yr (Lowrance et al., 1984b).

Denitrification of a mole of NO_3^- requires 1 to 2.5 moles of organic C (Peterjohn and Correll, 1986). Although there was little change in DOC concentrations throughout the forest (Fig. 3), DOC may be consumed and continuously replenished by transformation of particulate organic C. Temperate forests produce more than enough organic matter to support denitrification at rates high enough to account for our estimated removal of NO_3^- (Peterjohn and Correll, 1986), but it is not clear how much of the organic matter produced is available to denitrifiers.

The removal of NO_3^- by either assimilation or denitrification probably helped raise the pH of groundwater traversing the forest (Fig. 3). The increase in pH would protect stream waters from acidification, which has been detrimental to aquatic ecosystems in this region (Correll et al., 1987). The pH trend may have been dampened by the concomitant increase in SO_4^{2-} in groundwater (Fig. 3), but the net change in concentrations of the anions we measured was more than enough to account for the pH change. Of course, cations and anions that we did not measure would also be involved in determining groundwater pH.

The increase in SO_4^{2-} indicates that reduction of SO_4^{2-} to sulfides was not a dominant process. This is consistent with our Eh measurements, which were usually above − 220 mV, too high to favor SO_4^{2-} reduction (Correll and Weller, 1989). Oxidation of sulfides such as particulate iron sulfide may have occurred instead. Oxidation of sulfides can drive denitrification and result in concomitant increases in groundwater SO_4^{2-} with decreases in NO_3^- (Postma et al., 1991).

The increase in dissolved phosphate concentration near

the stream bank (Fig. 3) may be due to reduction of iron oxyhydroxides that bind phosphate to soil particles (Richardson, 1985). Iron reduction can occur at Eh below − 200 mV after the more energetically favored reduction of NO_3^- is complete (Correll and Weller, 1989). Low NO_3^- concentrations (Fig. 3) and Eh of − 510 mV (Table 1) were found near the stream.

Other studies have stressed the importance of riparian forests in trapping sediment and particulate nutrients borne in overland flow. Peterjohn and Correll (1984) sampled surface runoff and found a large decrease in suspended load with distance into the forest. Cooper et al. (1986) measured sediment accumulation above the soil horizon labelled with Cs-137 from bomb testing of the 1950s to 1960s. Lowrance et al. (1984b) observed accumulation of sediment above a layer of buried woody vegetation deposited in the last century. In contrast, we found net erosion along a surface runoff pathway through our study site. However, deposition or erosion of sediment is extremely variable spatially, and erosion was confined to the relatively narrow flow path of surface runoff at our sites. At other sites in the forest, there may be little surface flow from the field into the forest, or surface flow may be spread out rather than concentrated along one path. The channeling of suface flow promotes erosion by producing high flow velocities.

Differences among riparian forests clearly affect their nutrient retention properties. The forest of our present study, near Centreville, differs in important ways from one we previously studied (e.g., Peterjohn and Correll, 1984), near the Rhode River. First, the area of corn field draining into the Centreville forest is about 800 m wide, whereas the area draining into the Rhode River forest is about 150 m wide. This suggests greater hydrologic and nutrient loading of the Centreville forest than of the Rhode River forest. Greater hydrologic loading at the Centreville forest may account for the channelization of overland storm flow there. Groundwater flow may be further enhanced at the Centreville forest by the presence of a

thick layer of sandy sediment (Fig. 2) not present at the Rhode River forest. Also, the Centreville forest has two distinct zones, the hill slope and the floodplain (Fig. 2), whereas the Rhode River site lacks a hill slope. These differences may account for the observation that NO_3 uptake occurred at the field–forest boundary of the Rhode River forest, but did not occur until the edge of the floodplain in the Centreville forest. Despite the differences, both forests seem to remove large amounts of NO_3^- from groundwater, perhaps as a result of lateral flow of groundwater confined near the surface by underlying aquicludes. Functional differences among riparian forests and within zones of particular forests suggest the need for further research to determine the environmental conditions that promote nutrient removal and how commonly such conditions are met.

ACKNOWLEDGMENTS

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