Nitrate Dynamics in Riparian Forests: Groundwater Studies

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ABSTRACT

This study was conducted to assess the removal of groundwater nitrate (NO_3^-) in different soil drainage classes within three riparian forests located in Rhode Island. A solution of NO_3^- and a conservative tracer [either bromide (Br^-) or chloride (Cl^-)] was applied in the growing and the dormant seasons to trenches upgradient of wetland locations with hydric soils (poorly and very poorly drained soils) and transition zone locations with somewhat poorly and moderately well-drained soils located immediately upslope of the wetlands. To assess removal, the change in groundwater concentrations of NO_3^- relative to the concentration of the conservative tracer was observed in monitoring wells located in each soil drainage class from June 1989 through April 1990. Removal of groundwater NO_3^- was consistently high in the wetland locations, generally in excess of 80% in both growing and dormant seasons. In the transition zones, attenuation was less than 36% during the growing season, and ranged from 50 to 78% in the dormant season. Attenuation in the transition zones was positively correlated with water table elevations. Transition zone attenuation was high in the dormant season relative to the growing season likely because high water tables during the dormant season caused the contaminant plume to be exposed to soil with higher organic matter. The results suggest that both wetlands and transition zones between wetlands and uplands can be important sinks for groundwater NO_3^-.

NITRATE (NO_3^-) is a difficult pollutant to control due to its high mobility in soils and groundwater and the large inputs of this ion to groundwater from both agricultural practices (Keeney, 1986; Baker et al., 1975) and unsewered residential developments (Cain et al., 1989; Koppleman, 1978). Many studies have focused on quantifying the sources of NO_3^- loading to groundwater (Gold et al., 1990; Keeney, 1986). Recently several studies have focused on landscape features that remove NO_3^- after it has entered groundwater. In particular, strips of riparian forest have been determined to be important in maintaining stream water quality in areas of intensive agriculture (Karr and Schlosser, 1978; Lowrance et al., 1984; Jacobs and Gilliam, 1985; Peterjohn and Correll, 1985).

Groundwater NO_3^- moving through riparian zones is subject to denitrification, plant uptake and microbial immobilization. Marked differences in NO_3^- attenuation among riparian zones, however, have been reported and attributed to spatial variation in soil type, organic matter, hydrology, and vegetation (Whigham et al., 1988; Cooper, 1990). Soil organic matter levels and microbial activity are concentrated near the surface and decline sharply with depth (Parkin and Meisinger, 1989). For NO_3^- removal to occur through plant uptake, groundwater levels must be elevated within the root zone during the growing season. Plant uptake may be minimal during the late winter and early spring when groundwater flow is often at its annual maximum (Bormann and Likens, 1979).

Because the fate of groundwater NO_3^- may be influenced by the degree of interaction with the biologically active zone of the soil, natural soil drainage classes offer a potentially useful approach to evaluate broad-scale patterns of NO_3^- removal in riparian forests. Soils are classified into one of seven drainage classes based on morphological characteristics that reflect the frequency, duration, and seasonal timing of saturation or partial saturation during soil formation (Brady, 1974). Drainage classes differ in a number of characteristics that may affect groundwater NO_3^--N removal including depth to seasonal high water table, vertical distribution and percentage organic matter in the solum, and timing and location of anaerobic conditions within the soil profile.

The objectives of the research reported in this paper were to: (i) quantify the relative attenuation of groundwater NO_3^- in different soil drainage classes within riparian forests, (ii) compare attenuation in the dormant and growing season, and (iii) relate observed attenuation to soil and hydrologic characteristics. A companion paper describes a study conducted on the same sites that focused on NO_3^- removal processes in the riparian zone (Groffman et al., 1992).

EXPERIMENTAL APPROACH

Determining the fate of a groundwater contaminant moving through a riparian forest can be confounded by the influences of dilution as well as spatial and temporal variations of the contaminant. In this study we applied a solution of NO_3^--N along with a conservative tracer (Br^- or Cl^-) to trenches upgradient of both upland–wetland transition zones and wetland zones of a riparian forest. The change in concentration of NO_3^- relative to the concentration of the conservative tracer in the groundwater was then observed as the contaminants moved through the riparian zone. The dilution of the introduced contaminant plume was determined from the observed dilution of the conservative tracer. A decline in NO_3^- concentration in excess of that caused by dilution was attributed to attenuation (Trudell et. al., 1986).

METHODS

Three sites were selected that were within a 35-km radius of Kingston, RI. Each site contained a soil drainage sequence ranging from moderately well drained to very poorly drained soils (Table 1 and 2). The length of each soil drainage sequence ranged from 25 to 60 m. Site A was located on soils derived from unstratified glacial drift, while sites B and C were located on soils derived from stratified glacial drift (outwash). A "growing season" study was conducted on all three sites between 15 June 1989 and 15 Oct. 1989 and a "dormant season" study was conducted between 15 Feb. 1990 and 19 Apr. 1990 at Sites A and B only. Due to loss of access, Site C was not available for the dormant season study. Site A was 100 m downslope from an unsewered low density residential development (lot sizes were approximately 0.8 ha) while Site B was 50 m downslope from an unsewered dense residential development (lot sizes were approximately 0.1 ha) and Site C was located in a

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Abbreviations: MWD, moderately well drained; SPD, somewhat poorly drained; PD, poorly drained; VPD, very poorly drained.

Table 1. Soil drainage class and classification at riparian study sites.

Drainage class	Classification
	Site A—Parent material: Unstratified glacial drift
MWD	Coarse loamy mixed mesic Aquic Dystrocrepts
SPD	Coarse loamy mixed mesic Aeric Haplaquepts
PD	Coarse loamy mixed mesic acid Typic Haplaquepts
VPD	Loamy mixed euic mesic Terric Medisaprists
	Site B—Parent material: Glacial outwash
MWD	Sandy mixed mesic Aquic Dystrocrepts
SPD	Sandy mixed mesic Aeric Haplaquepts
PD	Sandy mixed mesic Typic Haplaquepts
VPD	Sandy mixed mesic Humaqueptic Fluvaquents
	Site C—Parent material: Glacial outwash
MWD	Mixed mesic Typic Udipsamments
SPD	Mixed mesic Aquic Udipsamments
PD	Sandy mixed mesic Entic Haplumbrepts
VPD	Sandy mixed mesic Histic Humaquepts

forest area that was 300 m downslope from a golf course. All sites were forested with an upland hardwood forest dominated by oak (*Quercus rubra* L.) grading to a red maple (*Acer rubrum* L.) swamp.

Within each study area, "transition zone" and "wetland" sampling areas were established for placement of groundwater monitoring wells. Transition zone wells were located in moderately well-drained (MWD) and somewhat poorly drained (SPD) soils. These soils were immediately upslope of hydric wetland soils and below well-drained uplands. The wetland wells were located in poorly drained (PD) and very poorly drained (VPD) soils, all of which were classified as hydric soils (Tiner and Veneman, 1987). Well transects were placed perpendicular to the direction of groundwater flow within each drainage class and were approximately 6 m wide (Fig. 1 and 2). Groundwater flow direction was initially determined at each site by triangulation of surveyed water table elevations.

Wells were constructed of 4.04 cm diam. PVC pipe with slotted-length screens and were installed using a hand or power auger. To enhance the likelihood of intercepting the plume, we installed three shallow wells and three deep wells in each drainage class. Shallow wells were screened for 30 cm from the top of the seasonal water table and ranged from 0.8 to 1.0 m deep in the transition zone, to 0.4 to 0.6 m in the wetlands. The deep wells were screened for 70 cm beginning at the bottom of the shallow wells. A total of 30 wells were installed at each study site. The screened portion of the well was backfilled with a medium sand to prevent clogging. Bentonite pellets were used near the surface to prevent channeling of surface water.

Immediately upgradient of each transition zone and wetland sampling location, a 12.0 m long by 0.2 m wide chemical application trench was established perpendicular to the direction of groundwater flow. The bottom of the trench was 15 cm above the water table. At each location, NO_3^--N and a conservative tracer were applied to the trench at

Table 2. Selected characteristics of each site.

Soil	Depth	Distance from application	pH	Organic Matter	Average water table depth: Growing	Average water table depth: Dormant	Rooting depth
		m		%	m	m	m
Site A—Unstratified drift, undeveloped upland							
Transition Zone							
MWD	Shallow	4.1	4.0	1.85	0.81	0.71	0.70
	Deep		5.1	0.86			
SPD	Shallow	8.6	4.4	1.68	0.49	0.42	0.60
	Deep		4.9	0.73			
Wetland							
PD	Shallow	2.7	4.3	3.88	0.18	0.10	>0.30
	Deep		5.1	0.70			
VPD	Shallow	6.6	4.5	27.95	0.12	0.05	>0.30
	Deep		4.8	1.22			
Site B—Outwash, developed upland							
Transition Zone							
MWD	Shallow	5.2	5.3	1.83	0.71	0.63	0.70
	Deep		5.6	2.14			
SPD	Shallow	14.9	5.5	1.78	0.42	0.38	0.65
	Deep		5.7	0.99			
Wetland							
PD	Shallow	2.4	5.1	7.17	0.21	0.17	>0.30
	Deep		5.9	2.56			
VPD	Shallow	5.8	5.4	30.68	0.06	0.02	>0.30
	Deep		5.4	5.88			
Site C—Outwash, undeveloped upland							
Transition Zone							
MWD	Shallow	2.4	5.1	1.27	1.11	NA†	0.80
	Deep		5.0	1.01			
SPD	Shallow	9.6	4.9	2.22	1.03	NA	0.65
	Deep		5.1	0.71			
Wetland							
PD	Shallow	2.5	4.7	2.13	0.27	NA	>0.30
	Deep		5.5	0.62			
VPD	Shallow	14.6	4.7	5.10	0.19	NA	>0.30
	Deep		5.1	0.33			

† NA = Data not available, site was not sampled in dormant season.

equal concentrations (180 mg/L) in solution form. The application rate (0.27 kg NO$_3^-$–N per application) was chosen to simulate the weekly total NO$_3^-$ –N mass output from a septic system leach field serving a three to four person household (USEPA, 1980). The application was split into three doses of 500 L each, spaced 48 h apart to minimize hydrologic disturbance. Dry, reagent grade chemicals were mixed into solution on site using water obtained from the associated nearby stream at each site. Stream concentrations of NO$_3^-$ at all sites were less than 2% of the application concentrations and were not considered during solution preparation.

At the outset of this study (growing season, wetland locations), Cl$^-$ was used as the conservative tracer; however, background concentrations of Cl$^-$ proved to be highly variable and Br$^-$ was substituted as the conservative tracer for the growing season–upland location study and all dormant season investigations.

Each well was sampled weekly beginning immediately prior to chemical application. Sampling was continued until after the end of the season of interest. Prior to each sampling, three well volumes of water were removed from each well to ensure that a representative sample of the ambient groundwater was obtained. Samples were filtered (2.5 μm)

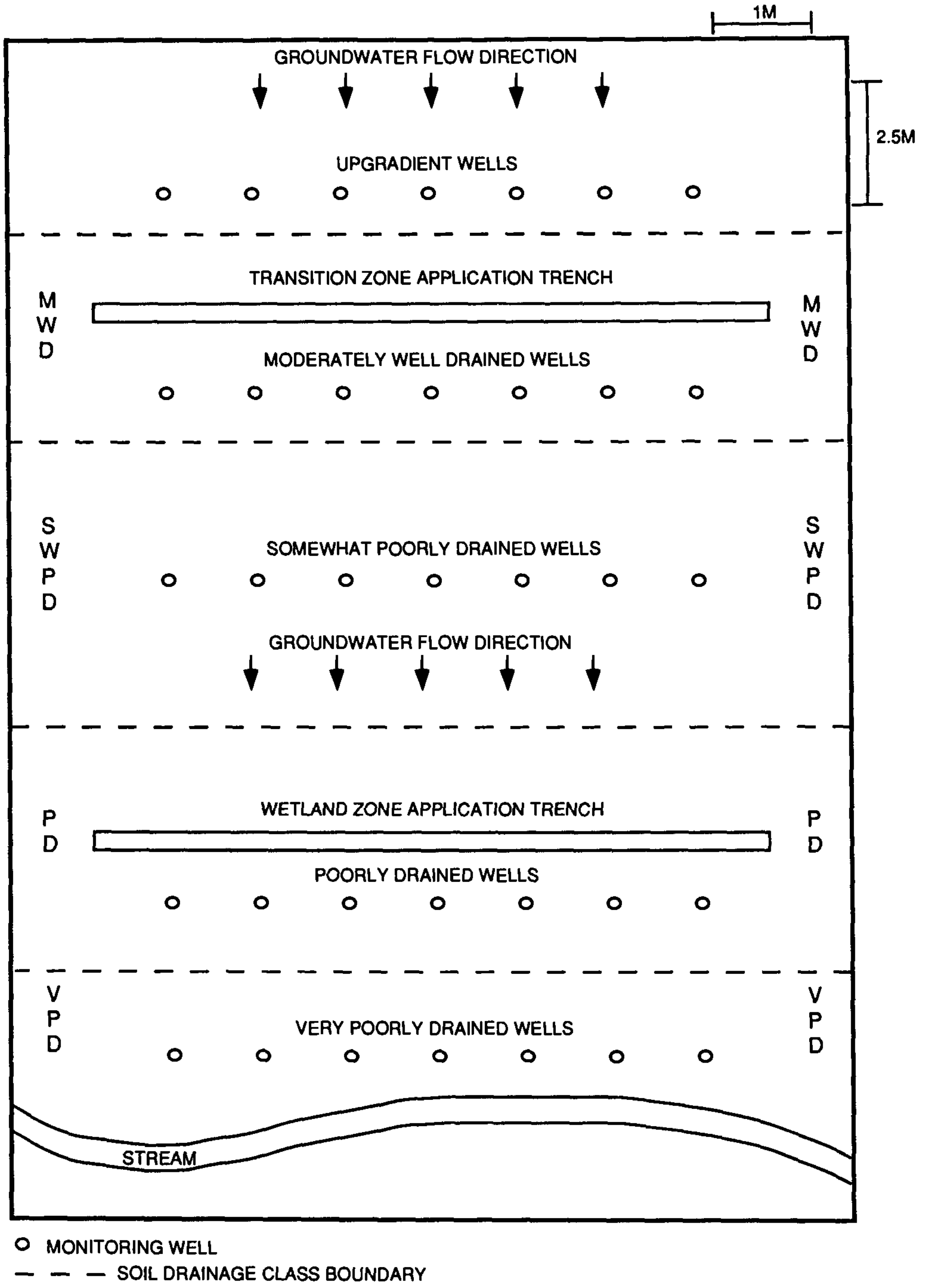

Fig. 1. Conceptual plan view of riparian forest experimental layout showing application trenches and well monitoring transects. MWD - moderately well-drained soil, SPD - somewhat poorly drained soil, PD - poorly drained soil, and VPD - very poorly drained soil.

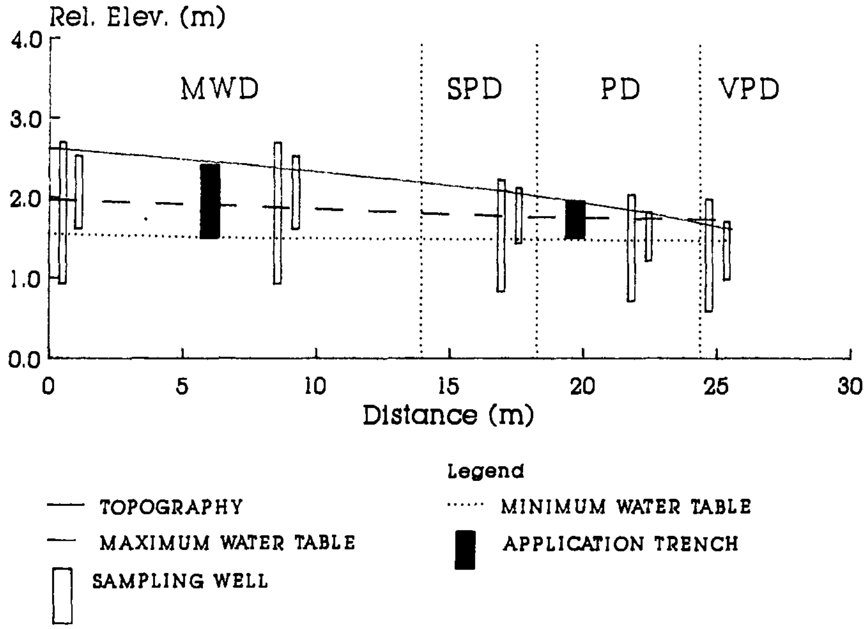

Fig. 2. Cross-section view of riparian forest experimental layout in the glacial outwash site with a developed upland showing application trenches and well monitoring transects. MWD - moderately well-drained soil, SPD - somewhat poorly drained soil, PD - poorly drained soil, and VPD - very poorly drained soil.

and preserved with 0.8 mL/L concentrated sulfuric acid (pH < 2) (APHA, 1985). Samples were stored at 4 °C until analysis.

Water analysis for NO$_3^-$ and Cl$^-$ was performed using an automated colorimetric Cd reduction method and an automated colorimetric ferricyanide method, respectively, on a Alpkem RFA 300 Rapid Flow Analyzer (Alpkem Corp., 1986). Bromide analysis was performed using an electrometric method with an Orion Research 940EA Ionanalyzer (Orion Research, 1982).

Water table elevations and temperatures were measured monthly during the growing season study and biweekly during the dormant season study. Soil samples were obtained from each transect at all sites, at depths corresponding to the monitoring well screens. The pH of each of the soil samples was measured in triplicate using a electrometric method (USEPA, 1979) with a glass electrode (Corning Glass Works, Corning, NY). The approximate depth of the root zone was determined in the well transects at each site by visual inspection of an open trench. Percentage organic matter of each soil sample was determined in the well transects at each site by visual inspection of an open trench. Percentage organic matter of each soil sample was determined in duplicate by a C loss on ignition method (Nelson and Sommers, 1982). Particle size analysis was performed on soils obtained from the screened sampling depth using a dry sieve method (Gee and Bauder, 1986).

To quantify the amount of NO$_3^-$-N loss that was observed in each sampling well, we compared the application ratio of NO$_3^-$-N to the conservative tracer with the ratios observed in samples taken from the monitoring wells (Trudell et al., 1986). The percentage loss due to attenuation was determined as follows

Percentage Attenuation = [1 − (observed ratio/application ratio)] × 100% where

Application ratio = NO$_3^-$-N applied (mg/L)/tracer applied (mg/L)

Observed ratio = NO$_3^-$-N observed (mg/L)/tracer observed (mg/L).

The ambient groundwater concentrations (background concentrations) of Br$^-$, Cl$^-$, and NO$_3^-$-N, were established by sampling each well a least three times prior to attenuation analysis. Background concentrations were subtracted from the observed value prior to calculating the percentage attenuation. Background concentrations for the applied contaminants were determined for each well from samples obtained prior to application in the dormant season study, and during the initial sampling period (before the tracer was observed to significantly increase) in the growing season study.

Less than 5% of the background samples had Br$^-$ concentrations above the detection limit of 0.4 mg/L. The maximum background Br$^-$ concentration observed at any site was 0.8 mg/L. Following application of the contaminant, samples with Br$^-$ concentrations greater than the maximum background value were considered to be part of the experimental plume. Background NO$_3^-$-N concentrations were undetectable (<0.10 mg/L) at Site A and ranged from <0.10 to 0.56 mg/L and from <0.10 to 4.4 mg/L at Sites C and B, respectively. When concentrations were observed to be below detection, the detection limit of the method of analysis was used in the attenuation calculations.

As described earlier, only Cl$^-$ was used as the conservative tracer in the wetland sites during the growing season. We chose to represent the background Cl$^-$ concentration at each contaminated well by the mean background concentration plus three standard deviations because of the high spatial and temporal variations observed in ambient groundwater Cl$^-$ concentrations. By selecting a high background concentration for the conservative tracer, the attenuation estimates were less likely to overstate the actual attenuation. Because the NO$_3^-$-N levels were so low in the wells of the wetland locations during 1989, the attenuation estimates were not particularly sensitive to the background concentration of chloride selected for computation.

Depth to groundwater for each sampling date was compared to percentage attenuation using the Pearson's product moment correlation coefficient (Bhattacharyya and Johnson, 1977). A student's *T* test was used to compare attenuation between the transition zone and wetland locations within each site during each season (Bhattacharyya and Johnson, 1977). Within each site, the results from the SPD well transects represented the cumulative attenuation that occurred as the introduced plume traveled from the application trench through the MWD soil. Likewise, the results from the wells in the VPD soil were not independent from the results of the PD wells. Given this lack of independence, intrasite statistical tests of attenuation differences were limited to comparisons between the wetland and transition zone locations rather than between all four drainage classes located at a site. The Statistical Analysis System (SAS Inst., Inc., 1985) was used for all statistical comparisons. All results are reported at the 95% confidence level.

RESULTS

A contamination plume was observed at nearly all transects during both the dormant and growing season within 8 to 37 d following application. The only exceptions occurred at Site A in the 1989 growing season where a plume was not detected in the SPD, PD, and VPD well transects. The experimental design was not adequate to determine whether the plume moved away from the wells or was diluted by recharge and dispersion. The high sensitivity of the analytical methods, and the fact that the plumes were easily detected elsewhere, suggest that the plume moved away from the wells, however.

Although the contaminant solutions were evenly applied upgradient of the well transects, the plumes showed evidence of preferential flowpaths. As shown in Table 3, at some transects only a portion of the wells intercepted the plume. The time period that the plume intercepted a given well was also quite variable. Within several transects, selected wells intercepted the plume two to three times longer than other wells at the same depth. Preferential flowpaths were more evident in the growing season when the water table was deeper and often below the screened portion of the shallow wells.

As expected, a strong association was found between the movement of Br^- and NO_3^-. Figures 3 and 4 depict temporal changes in Br^- and NO_3^--N concentrations at wells located in separate transects. Figure 3 illustrates a situation with minimal attenuation, where Br^- and NO_3^- N concentrations were nearly identical throughout the sampling period. Figure 4 shows a situation with considerable attenuation, with Br^- concentrations at least tenfold greater than NO_3^--N concentrations during the period (Day 10–35) when elevated (>0.5 mg/L) Br^- concentrations were detected. Figure 5 shows the relationship between Cl^- and NO_3^--N concentrations in a well location where Cl^- was used as the conservative tracer. The temporal patterns of Cl^- and NO_3^--N are similar but attenuation analysis requires subtraction of a background Cl^- concentration. For the results depicted in Fig. 5, a background Cl^- concentration of 8.7 mg/L was used, yielding a mean attenuation of 51%. We restricted plume analysis to Day 35 to 73 thereby including only those observations with Cl^- concentrations elevated above the computed background level.

For both the growing season and dormant season studies, high NO_3^- attenuation was consistently observed in the wetland locations (Table 3). Most of the transition zone well transects had low NO_3^- attenuation during the growing season with mean attenuation generally less than 36%. The four transition zone transects sampled during the dormant season demonstrated more uniform results with attenuation ranging from 50 to 78%. There was no consistent pattern of attenuation with time for either season.

Table 3. Mean groundwater NO_3^- removal and plume distribution in three riparian forest sites. Summer 1989 (growing season) and winter 1990 (dormant season). Values are mean ± standard deviation.

Site	NO_3^- removal	Wells intercepting	Plume interception period
	%	No.	wk/well
		Growing season	
A. Unstratified drift			
Transition zone			
MWD	20 ± 28 (n = 5)	1	5
SPD	ND†		
Wetland location			
PD	ND		
VPD	ND		
B. Outwash (developed upland)			
Transition zone			
MWD	36 ± 28* (n = 11)	3	3.7 ± 1.15
SPD	91 (n = 1)	1	1
Wetland location			
PD	93 ± 17 (n = 51)	5	10.2 ± 5.4
VPD	87 ± 28 (n = 13)	2	6.5 ± 7.8
C. Outwash (undeveloped upland)			
Transition zone			
MWD	1 ± 3 (n = 6)	3	2.0 ± 1.7
SPD	17 ± 15 (n = 8)	4	2.7 ± 1.5
Wetland location			
PD	80 ± 21 (n = 8)	4	2.0 ± 1.4
VPD	84 ± 24 (n = 14)	2	7 ± 1.4
		Dormant season	
A. Unstratified drift			
Transition zone			
MWD	62 ± 18 (n = 9)	3	3.0 ± 3.5
SPD	78 ± 38 (n = 3)	3	1.0 ± 0.0
Wetland location			
PD	89 ± 17 (n = 5)	2	2.5 ± 0.7
VPD	97 ± 8 (n = 15)	3	5 ± 0.0
B. Outwash (developed upland)			
Transition zone			
MWD	50 ± 25 (n = 33)	6	5.5 ± 1.4
SPD	59 ± 31 (n = 16)	6	2.7 ± 1.2
Wetland location			
PD	61 ± 27 (n = 37)	6	6.2 ± 0.4
VPD	90 ± 16 (n = 6)	4	1.5 ± 0.6
C. Outwash (undeveloped upland)			
Not sampled in dormant season			

* Within a site means in a column followed by the same letter are not significantly different at the 0.05 level based on a Student's *T* Test. Statistical analysis limited to cumulative removals observed in transition zone (SPD) vs. wetland (VPD). Test performed on initial removal in transition zone (MWD) vs. wetland (PD) in Site B, growing season due to insufficient replicates on SPD transect.

† ND = plume not detected.

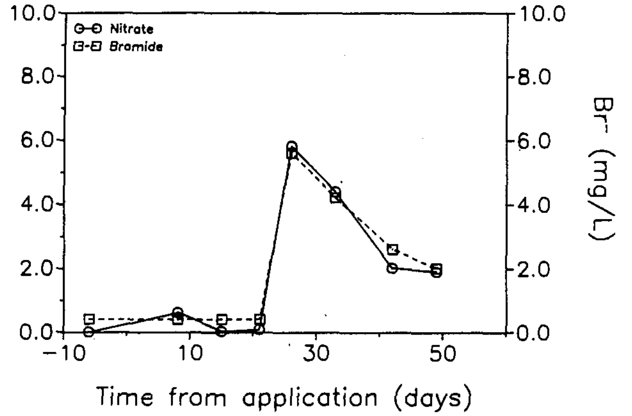

Fig. 3. Nitrate–nitrogen and Br^- concentrations in a well from the moderately well drained transect of the glacial outwash site with an undeveloped upland during the growing season, 1989.

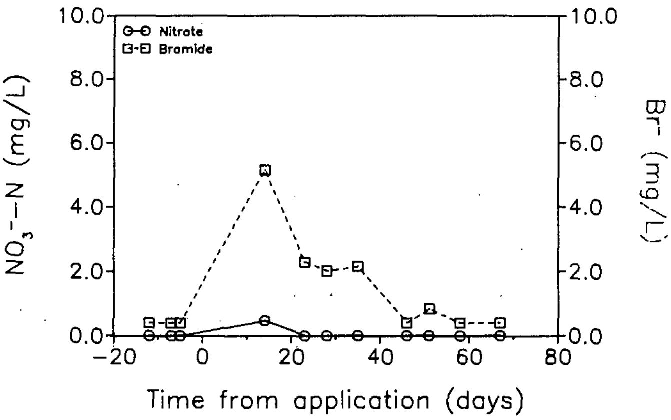

Fig. 4. Nitrate–nitrogen and Br^- concentrations in a well from the very poorly drained transect of the unstratified drift site with an undeveloped upland during the dormant season, 1990.

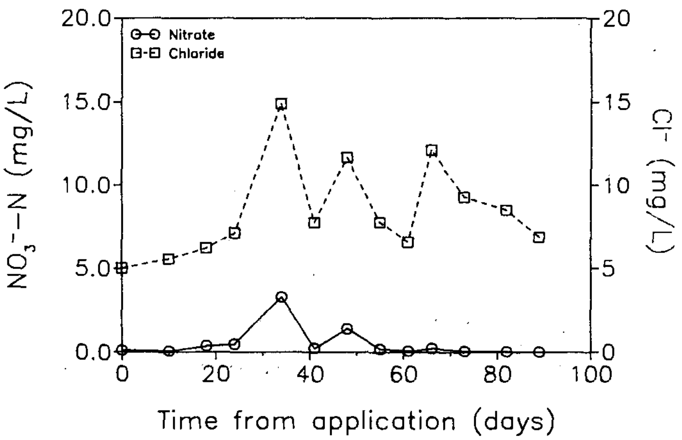

Fig. 5. Nitrate–nitrogen and Cl^- concentrations in a well from the poorly drained transect of the glacial outwash site with an undeveloped upland during the growing season, 1989.

In almost all cases, attenuation in the wetland soils was significantly higher ($P < 0.05$) than in the transition zone soils. The only case where a significant difference was not observed was between the SPD and VPD soils at Site A during the dormant season where the SPD soils had high (78%) attenuation.

Differences in NO_3^- attenuation among the transition zone sites in the growing season may be related to variation in water table depths (Table 2). The transition zone transects at Site C had the lowest growing season attenuation and the lowest water table elevations. For all observations in the transition zone sites, independent of season, NO_3^- attenuation was significantly correlated ($P < 0.05$) to water table elevation at the time of observation. This relationship was also significant when the analysis was applied only to the growing season data in the transition zone locations, but was not significant in transition zone locations during the dormant season. Water table elevations were significantly ($P < 0.05$) lower in the growing season than the dormant season on all sites in the transition zone drainage classes.

DISCUSSION

Spatial and seasonal variations in water table elevations appear to strongly influence the fate of groundwater NO_3^- in the riparian zone. During both the dormant and growing season studies, the water table and the contaminant plume were located in the root zone of all the wetland locations (Table 2). The shallow water table within the wetlands placed the contaminant plume into contact with soil characterized by elevated (>2.5%) organic matter (Table 1) and high denitrification potential (Groffman et al., 1992).

Although attenuation in the wetland locations was consistently high in both the dormant and growing season studies, the relative contribution of different attenuation mechanisms may have varied seasonally. During the growing season plant uptake, denitrification, and microbial immobilization may have contributed to NO_3^- attenuation, while dormant season attenuation was likely the result of only microbial processes (Groffman et al., 1992). Groundwater temperatures throughout the dormant season at all wetland and upland locations were consistently between 6.5 and 8.0 °C suggesting that microbial activity was not limited by temperature (Broadbent and Clark, 1965).

The high attenuation observed in the transition zone transects during the dormant season may have been generated by seasonal changes in water table elevations. During the dormant season, water table elevations within each of the soil drainage classes were higher than during the growing season and were relatively uniform over time and between the two sites. The mean water table was less than 72 cm from the surface and within the root zone at all four upland transects in the dormant season, whereas three of five upland transects sampled in the growing season had mean water tables below the root zone. For the two MWD transects included in the seasonal comparison studies, the average depth to the water table was 8 to 10 cm deeper during the growing season than during the dormant season.

The rise in the water table noted during the dormant season exposed the upper portion of the groundwater to different soil conditions than during the growing season. In nearly all of the upland transects the soil surrounding the shallow wells had almost twice the organic matter content of the media in which the deep wells were screened (Table 2). The shallow wells were in the unsaturated zone during most of the growing season study. During the dormant season study, however, the shallow wells were consistently below the water table. The higher organic matter content encountered by the upper portion of the groundwater in the dormant season could have supported enough microbial immobilization and denitrification to account for the observed NO_3^--N removal during the dormant season. Microbial biomass and activity have been shown to decline exponentially with depth in the soil profile (Parkin and Meisinger, 1989).

The spring, summer and fall of 1989 were extremely wet, and water tables during the 1989 growing season study were higher than those that occur in most years (USGS, 1989). During the 1990 dormant season studies (February–May), groundwater levels in southern Rhode Island were described by USGS (1990) as "normal." Normal groundwater levels are defined as monthly values that are not within the highest or lowest 25% of recorded levels for the period of record. These data suggest that the growing season attenuation results may have been higher than normal due to elevated water tables, but that the dormant season results were obtained during "average" groundwater conditions.

Much of the annual groundwater flow through riparian zones is expected in early spring when water tables are at or near their annual high. During this period riparian forests located on MWD or SPD soils may be expected to have groundwater within their root zone. If local hydrology results in flow patterns that carry NO_3^--contaminated water into the shallow groundwater of riparian zones, these areas should be a major sink for NO_3^-. Our companion paper (Groffman et al., 1992) explores the mechanisms involved in the observed NO_3 removal and focuses on long-term effectiveness of these sites.

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