Nitrogen Transformations in Ponds Receiving Polluted Water from Nonpoint Sources

G. B. REDDY AND K. R. REDDY

ABSTRACT

A laboratory study was conducted to determine the role of N transformations in ponds receiving inorganic N-rich effluents from septic fields, agricultural, and pasture watersheds. Undisturbed sediment columns were obtained from three ponds. Floodwater in the columns was enriched with either $^{15}$NH$_4$-N or $^{15}$NO$_3$-N. Ammonium removal rates ranged from 55 to 85 mg N m$^{-2}$ d$^{-1}$, while NO$_3$-N removal rates ranged from 48 to 71 mg N m$^{-2}$ d$^{-1}$. Twenty-three to 49% of floodwater $^{15}$NH$_4$-N diffused into the sediment during a 22-d incubation period and was recovered in $^{15}$NH$_4$-N (6-12% of the floodwater $^{15}$NH$_4$-N) and organic $^{15}$N fractions (14-37% of the floodwater $^{15}$NH$_4$-N). When $^{15}$NO$_3$-N was added to the floodwater, about 7% was tied up in the sediment. Ponds receiving effluents from septic tanks and pastured areas retained less floodwater N in the sediment compared to sediments of the pond receiving runoff from agricultural watershed.

Additional Index Words: N removal, nitrification, denitrification, retention ponds, aquatic systems, drainage effluents, watersheds.


Nitrogen usually enters into aquatic systems by surface-subsurface discharges from nonpoint sources such as faulty septic systems or sewage effluent discharges, agricultural drainage, and pasture watershed areas. Other sources of N input include (i) precipitation; (ii) N$_2$ fixation in the water and the sediment; and (iii) N release during the decomposition of dead aquatic plant and animal tissue.

About one-quarter of the inhabitants of the USA dispose sewage via sewage tanks and surface adsorption fields (Brown et al., 1979). Over one-third ($300 \times 10^8$ ha) of the land area of the continental USA is used for grazing livestock and receives 50% of all livestock wastes (Doran and Linn, 1979). Septic effluents contain large quantities of N and P (Sikora et al., 1976). Klausner et al. (1974) found that more soluble N escaped through surface runoff under heavy fertilization. Animal grazed lands contributed more N and P in runoff than nongrazed lands (Robbins, 1978). Therefore, nutrient discharge from these sources is a serious concern.

Nitrogen in the NO$_3$-N form discharged into aquatic systems is (i) assimilated by algae and other aquatic macrophytes, and (ii) diffused into underlying sediments where it undergoes denitrification (Chen et al., 1972a, b; Engler et al., 1976; Sain et al., 1977; Reddy et al., 1980). The loss of NO$_3$-N from overlying floodwater via denitrification has also been observed in flooded organic soil (Reddy et al., 1980), lake sediments (Chen et al., 1972b; Tiren et al., 1976), estuarine and marine sediments (Nishio et al., 1982; Kasper, 1982), and stream sediments (Engler and Patrick, 1974; Van Kessel, 1977; Sain et al., 1977). The NH$_4$ form of N discharged into an aquatic system is also assimilated by aquatic macrophytes, but a significant portion can be lost through nitrification in the water (Chen et al., 1972a; Tuffey et al., 1974) followed by denitrification in the sediment (Chen et al., 1972b) or by NH$_3$ volatilization (Bouldin et al., 1974). Inorganic N removal rate from overlying water is also to some ex-


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tent controlled by the underlying sediment characteristics. For example, sediments with a high concentration of available C remove floodwater NO₃⁻-N faster than sediments with low available C (Reddy et al., 1982).

Although the significance of N transformations in aquatic systems has been clearly established, very limited information is available on the potential of retention ponds functioning as a sink for N derived from nonpoint source waste discharges. The objectives of the present investigation were (i) to quantitatively determine under laboratory conditions the fate of ¹⁵N in sediment-water columns from ponds receiving drainage and runoff from septic fields, pasture and agricultural watersheds; and (ii) to evaluate the potential of pond sediment-water column as a sink for inorganic N.

### MATERIALS AND METHODS

The first pond (PSF = pond adjacent to septic fields) receives discharges from faulty septic systems located north of McConnell Road. The watershed with 6 to 8% slope is covered with fescue grass and has three soil types: Enon fine sandy loam (fine, mixed thermic Ultic Hapludalfs); Helena-Sedgefield sandy loam (clayey, mixed thermic Aquic Hapludults); and Iredell fine sandy loam (fine, montmorillonitic thermic Typic Hapludalfs). The pond surface area is 7376 m² with a mean depth (water and sediment) of 6.1 m. No management practices were administered at this location.

The pasture pond (PP = pond adjacent to a pasture) has 9547-m² area with a mean depth (water and sediment) of 5.5 m, is located south of McConnell Road and receives runoff/seepage from 30 beef cattle (Bos taurus) and 20 dairy cattle from March through November. The watershed is covered with fescue (Festuca sp.) grass and clover (Trifolium sp.), and has 6 to 10% slope. The grazing area has two soil types: Enon fine sandy loam and Mecklenburg clay loam (fine, mixed thermic Ultic Hapludalfs).

The third nonpoint source contamination pond that received runoff and drainage from an active agricultural land (PAF = pond adjacent to agricultural farm land) has an area of 7038 m² with an average depth (water and sediment) of 4.6 m. Soybean [Glycine max (L.) Merr.] and corn (Zea mays L.) were grown on this land every year on a rotational basis and was kept fallow without cover crop in the nonagricultural season. This watershed area has a slope

<table>
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<th>Parameter</th>
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ranging from 5 to 7% with Enon sandy loam and Mecklenburg clay loam soil.

Intact sediment columns were obtained by driving a polyvinyl chloride (PVC) pipe (5-cm o.d. by 60-cm length) 40-cm deep into the pond sediment. Sediment columns were brought to the laboratory by closing the column bottoms with a rubber stopper. The water above the sediment columns was replaced with 0.365 L (18-cm depth) of floodwater obtained from respective ponds and incubated for 24 h. After incubation, 0.005-L labeled $^{15}$NH$_4$Cl (99 atom % $^{15}$N excess) containing 0.73 mg N L$^{-1}$ was added to the floodwater to one set of sediment columns (nine columns, three from each pond), and 0.005 L of K$^+NO_3$ (99 atom % $^{15}$N) containing 0.73 mg N L$^{-1}$ to the second set of sediment columns to obtain a final concentration of 10 mg N L$^{-1}$ in all floodwater columns. In a separate treatment, floodwater (without underlying sediment) was treated in the same manner; this treatment was included in order to differentiate the reactions occurring in the water from those in the sediment. Each treatment was replicated three times. All columns were incubated in the dark at 30°C for a period of 22 d. A 0.005-L sample of floodwater was removed from each column on 1, 2, 4, 9, 15, and 22 d and analyzed for NH$_4$-N and NO$_3$-N. After 22 d, floodwater was drawn off the sediment columns and the sediment columns were sectioned horizontally for organic $^{15}$N, $^{15}$NH$_4$-N, and $^{15}$NO$_3$-N determinations. Floodwater was also analyzed for $^{15}$NH$_4$-N and $^{15}$NO$_3$-N. Dissolved O$_2$ in the water was measured once every 10 d using an oxygen sensor and oxygen meter (Yellow Springs Instruments, Yellow Springs, OH).

Ammonium-N and NO$_3$-N in the water samples were determined by an autoanalyzer (APHA, 1980, p. 1134). Floodwater at the end of 22 d was treated with 0.002 L of concentrated H$_2$SO$_4$, and the volume of water was reduced to about 0.1 L by evaporation at 60°C. Ammonium-N from the concentrated sample was again distilled using steam distillation, and subsequently analyzed for $^{15}$N content. A known amount of fresh sediment sample was extracted with 2 M KCl solution, and filtered solutions were analyzed for NH$_4$-N, NO$_3$-N, and $^{15}$N content. The residual sediment sample was dried at 60°C, ground, and analyzed for total Kjeldahl-N (TKN) (Bremner and Mulvaney, 1982) and $^{15}$N content (Hauck, 1982). Organic C in sediment was determined by the ashing method of Jackson (1965).

**RESULTS**

Organic C content of sediments collected from PSF and PP was higher than PAF pond sediments (Table 1). Ammonium-N and organic N contents were higher in these ponds sediments. Effluents discharged from septic tanks and pasture operations were usually higher in NH$_4$-N and TKN as compared to agricultural discharges. Concentrations of organic C and N were greater in surface sediments and decreased with depth.

Added $^{15}$NH$_4$-N was rapidly lost from the water in each system (Fig. 1). In all cases, nonlinear decrease of NH$_4$-N from the floodwater occurred and NH$_4$-N concentrations reached negligible levels in 15 d. Decrease in NH$_4$-N was directly correlated with the increase in NO$_3$-N concentration of the floodwater. This relationship was clearly demonstrated in the floodwater without underlying sediment. The nitrification rate was lower in the floodwater of PAF compared to PSF and PP sediment-water columns. Nitrate-N concentration in the floodwater with underlying sediment was generally low (<2.5 mg L$^{-1}$), while essentially all of the added NH$_4$-N was recovered as NO$_3$-N in the floodwater without sediment. Rapid decrease in floodwater NO$_3$-N was observed in PSF and

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**Fig. 3.** Sediment profiles of PAF, PSF, and PP showing the percent of added $^{15}$NH$_4$-N after 22 d of incubation.

**Fig. 4.** Sediment profiles of PAF, PSF, and PP showing the percent of added $^{15}$NO$_3$-N after 22 d of incubation.
PP systems with about 70% reduction in the first 5 d (Fig. 2). In the PAF system the rate of decrease in NO\textsubscript{3}-N levels was lower. Nitrate-N was stable in the floodwater without sediment.

Distribution of floodwater \textsuperscript{15}N in the sediment profile is shown in Fig. 3 and 4. The largest fraction of added \textsuperscript{15}NH\textsubscript{4}-N was recovered in the surface 2-cm depth and \textsuperscript{15}N content decreased with sediment depth. When \textsuperscript{15}NH\textsubscript{4}-N was added to the floodwater, about 8 and 28% \textsuperscript{15}N was recovered in inorganic- and organic-N fractions in the 2-cm depth of the PAF sediment system, respectively. Whereas in PSF and PP systems, recovered \textsuperscript{15}N was 7 and 6% in organic-N, and 4 and 2% in inorganic-N fractions, respectively (Fig. 3). When \textsuperscript{15}NO\textsubscript{3}-N was added to the floodwater, recovery of \textsuperscript{15}N in organic-N fraction of the sediment in the 2-cm depth was <4% in all systems (Fig. 4). In all systems evaluated, \textsuperscript{15}N was detected even at the 30-cm depth.

Mass balance of added \textsuperscript{15}N is shown in Table 2. In the PAF system about 12 and 37% of the floodwater \textsuperscript{15}NH\textsubscript{4}-N was present in NH\textsubscript{4}-N and organic-N fractions of the sediment, respectively. In PSF and PP systems, recovery of floodwater \textsuperscript{15}NH\textsubscript{4}-N in the sediments was 14 and 19% in organic-N fraction and 9 and 6% in the NH\textsubscript{4}-N fraction, respectively. These results indicate a significant movement of NH\textsubscript{4}-N from the floodwater to the underlying sediment. When \textsuperscript{15}NO\textsubscript{3}-N was added to the floodwater, recovery of \textsuperscript{15}N in the sediment was <7 and 2% in organic-N and \textsuperscript{15}NO\textsubscript{3}-N fractions, respectively. Total recovery of added \textsuperscript{15}N in the systems containing underlying sediment and floodwater ranged from 39 to 79% when \textsuperscript{15}NH\textsubscript{4}-N was added to the floodwater, while a much lower recovery (10-32%) was observed when \textsuperscript{15}NO\textsubscript{3}-N was added to the floodwater. Losses of added \textsuperscript{15}N were minimal in the system without underlying sediment with 95 to 100% recovery of both added NH\textsubscript{4}-N and NO\textsubscript{3}-N. The highest amount of added \textsuperscript{15}N was recovered by the PAF system.

**DISCUSSION**

Results demonstrate the effectiveness of sediment functioning as a sink for inorganic N discharges into ponds. Added NH\textsubscript{4}-N was lost from the floodwater due to oxidation of NH\textsubscript{4} to NO\textsubscript{3} and downward diffusion of NO\textsubscript{3} from the floodwater into the underlying sediment. Decrease in NH\textsubscript{4}-N concentration in the floodwater correlated with the increase in NO\textsubscript{3}-N concentration, demonstrating the occurrence of nitrification. This observation is clearly noted in the floodwater without sediment. Low NO\textsubscript{3}-N in the floodwater with sediments suggests diffusion of NO\textsubscript{3} followed by denitrification. Dissolved O\textsubscript{2} concentration of the floodwater in all systems was in the range of 3 to 5 mg l\textsuperscript{-1}, which is adequate to support nitrification. Removal of NH\textsubscript{4}-N from floodwater of various aquatic systems was also observed by several researchers (Curtis et al., 1975; Reddy and Graetz, 1981). Recovery of added \textsuperscript{15}N in the underlying sediment demonstrated diffusion of floodwater inorganic N into the sediment.

A significant amount of floodwater NH\textsubscript{4}-N diffused into the underlying sediment of the PAF system, thus reducing the loss of N through nitrification-denitrification. In this system, sediment contained low levels of exchangeable NH\textsubscript{4}, thus resulting in a downward concentration gradient. Because of low exchangeable NH\textsubscript{4} in the sediment, about 37% of added \textsuperscript{15}NH\textsubscript{4}-N probably was assimilated by microorganisms during the decomposition of organic matter. In 22 d about 49% of the floodwater NH\textsubscript{4}-N had diffused into the underlying sediment. The amount of NH\textsubscript{4}-N diffused into sediments of PSF and PP systems was 22.9 and 24.4% of added N, respectively. Low diffusion rates in these systems were due to the high extractable NH\textsubscript{4}-N present in the sediment. In all systems, diffusion of NH\textsubscript{4}-N into sediment indicates pore water NH\textsubscript{4}-N concentration in the sediment was probably lower compared to the floodwater NH\textsubscript{4} concentration. Some diffusion of NH\textsubscript{4}-N was also probably due to the concentration gradient of isotopic \textsuperscript{15}N added in the floodwater, which contained 99 atom % \textsuperscript{15}N. Differences in \textsuperscript{15}N and \textsuperscript{14}N diffusion in soils is not established, and warrants further investigation before any definite conclusions are drawn.

Organic matter content of the sediment also influenced the NO\textsubscript{3}-N removal rate from the floodwater. Since NO\textsubscript{3}-N reduction is dependent on substrate C availability (Bremner and Shaw, 1958; Burford and Bremner, 1975; Reddy et al., 1982) it is not surprising that NO\textsubscript{3}-N removal rates from floodwater were affected by organic matter content of the sediments. Organic C content of the sediments used ranged from 59 to 77 g C kg\textsuperscript{-1}} in sur-
Table 3. Nitrogen removal rate from floodwater enriched with $^{15}$NH$_4$-N and $^{15}$NO$_3$-N.

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<th>System</th>
<th>$^{15}$NH$_4$-N Inorganic Nt</th>
<th>$^{15}$NO$_3$-N Inorganic Nt</th>
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<tr>
<td>PAF</td>
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</tr>
<tr>
<td>PSF</td>
<td>55.2</td>
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<td>PP</td>
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Removal rates

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<th>System</th>
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<td>PSF</td>
<td>71.3</td>
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<tr>
<td>PP</td>
<td>70.2</td>
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</tbody>
</table>

$^{1}$NH$_4$N + NO$_3$N.

face 0- to 2-cm sediment and decreased with depth. Although the difference in total C of PSF and PP sediments is minimal and the fact that NO$_3$-N removal rates for each pond were different suggests that the available C in these sediments was probably variable. Significant relationships were observed between available C and total C, and available C and NO$_3$-N loss through denitrification (Burford and Bremer, 1975; Reddy et al., 1982).

The water column was enriched with $^{15}$NO$_3$-N and $^{15}$NH$_4$-N in order to follow the transport of N into the sediment. This enrichment elevated NO$_3$-N and NH$_4$-N concentrations of the floodwater compared to those observed under field conditions (Reddy et al., 1986). Ammonium-N and NO$_3$-N contents of the ponds were in the range of 1.8 to 4.6 mg L$^{-1}$.

Although elevated levels of NH$_4$-N and NO$_3$-N of the pond water indicate heavy N loading to these ponds, it is very difficult to quantify the amount of N entering each pond from the watersheds. A number of factors play a vital role in the movement of N from source to pond; these include soil characteristics, N transformations in the watersheds, surface and subsurface runoff, vegetation, rainfall, and temperature. The results presented in this paper, however, show the ability of ponds to remove inorganic N transported from external sources.

In conclusion, NH$_4$-N removal rates from floodwater ranged from 55 to 85 mg N m$^{-2}$ d$^{-1}$ ($0.55$ to $0.85$ kg N ha$^{-1}$ d$^{-1}$, Table 3). Overall inorganic N removal in NH$_4$-N enriched systems ranged from 40 to 63 mg N m$^{-2}$ d$^{-1}$. In NO$_3$-N enriched systems, floodwater NO$_3$-N removal rates were 48 to 71 mg N m$^{-2}$ d$^{-1}$, while overall inorganic N removal rates were 46 to 70 mg N m$^{-2}$ d$^{-1}$. Lower inorganic N removal rates in NH$_4$-N enriched system were due to some of the NH$_4$-N being nitrified to NO$_3$-N in floodwater. In a NO$_3$-N enriched system, the lower inorganic N removal rates reflect upon the release of NH$_4$-N from sediment to the floodwater. Under field conditions, inorganic N removal rates would probably be greater than those observed because of other sinks. For example, uptake by aquatic macrophytes (McCarthy et al., 1977) and NH$_3$ volatilization (Bouldin et al., 1974) can contribute to significant losses of N from aquatic systems.

REFERENCES


J. Environ. Qual., Vol. 16, no. 1, 1987