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# Nitrate Movement in Shallow Ground Water from Swine-Lagoon-Effluent Spray Fields Managed under Current Application Regulations

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## ABSTRACT

**Rapid increases in the swine (*Sus scrofa domestica*) population in the 1990s and associated potential for nitrate N pollution of surface waters led the state of North Carolina to adopt stringent waste management regulations in 1993. Our objectives were to characterize (i) nitrate N movement from waste application fields (WAFs) in shallow ground water, and (ii) soil, hydrologic, and biological factors influencing the amount of nitrate N in the adjacent stream. A ground water monitoring study was conducted for 36 mo on a swine farm managed under new regulations. Water table contours and lack of vertical gradients indicated horizontal flow over most of the site. Nitrate N concentrations in water from shallow wells in WAFs averaged 30 ± 19 mg L$^{-1}$ and $\delta^{15}$N ratios for nitrate N were between +20 and +25 per mil. Nitrate N concentration decreased from field-edge to streamside wells by 22 to 99%. Measurement of $\delta^{18}$O and $\delta^{15}$N enrichment of nitrate in ground water throughout the WAF–riparian system indicated that denitrification has not caused significant $^{15}$N enrichment of nitrate. Over a 24-mo period, $\delta^{15}$N ratios for nitrate N in the stream approached $\delta^{15}$N ratios for nitrate N in ground water beneath WAFs indicating delivery of some waste-derived nitrate N to the stream in shallow ground water. Nitrate N concentrations in the stream were relatively low, averaging 1 mg L$^{-1}$. Dilution of high nitrate N water in shallow horizontal flow paths with low nitrate N water from deeper horizontal flow paths at or near the stream, some denitrification as ground water discharges through the stream bottom, and some denitrification in riparian zone contributed to this low nitrate N concentration.**

Manure handling and application practices on concentrated animal feeding operations (CAFOs) are currently undergoing critical revisions to reduce impacts on surface water quality (Harter et al., 2002). The swine population in North Carolina increased from a total inventory of 2.6 million head in 1989 to 9.6 million head in December 1997, and has remained in the range of 9.6 to 10 million head since 1997 (National Agricultural Statistics Service, 2005). Most of this inventory is concentrated in a few counties in the Coastal Plain of southeastern North Carolina. For example, the swine inventory in Duplin and Sampson Counties alone is 3.8 million animals. The predominant system for treatment of the enormous amount of waste generated by these animals is anaerobic lagoons from which effluent is pumped and spray irrigated onto agricultural fields (Sloan et al., 1999). The most common receiver crop used to remove nutrients from these WAFs is coastal bermuda grass [*Cynodon dactylon* (L.) Pers.] which is either grazed or cut for hay. Grain crops such as corn (*Zea mays* L.), wheat (*Triticum aestivum* L.), and soybean [*Glycine max* (L.) Merr.] are also used as receiver crops, and some effluent is pumped onto woodlands.

Strict regulations on waste management in the swine industry were imposed in February of 1993 with the adoption of North Carolina Administrative Code Section .0200: "Waste Not Discharged to Surface Waters" (North Carolina Department of Environment and Natural Resources, 2005), which specifically included animal waste management systems. These regulations mandated that producers develop and implement waste management plans for each field receiving animal waste, and that waste application rates not exceed the agronomic N needs of receiver crops. Agronomic needs for N are based on realistic yield expectations (RYEs) which take into account the soil type in the WAF.

The 1993 regulations governing application of swine-lagoon effluent in North Carolina increased the WAF acreage per animal unit produced. As the North Carolina swine inventory has nearly doubled since these regulations were imposed (National Agricultural Statistics Service, 2005), it is estimated that 50% or more of the current WAF acreage receiving swine-lagoon effluent in North Carolina probably came into use after 1993, and have been managed entirely under Section .0200 regulations.

Several researchers have found high nitrate N concentrations (as high as 100 mg L$^{-1}$) in ground water beneath fields receiving swine-lagoon effluent before 1993 (Mikkelsen, 1995; Sloan et al., 1999). There has been no systematic research on the extent of nitrate N movement from WAFs, managed under Section .0200 regulations throughout their existence, to surface waters in shallow ground water systems. Such research is essential for understanding the impact of current swine-waste management practices on nitrate N movement to surface waters in shallow ground water systems in the North Carolina Coastal Plain.

Several studies have identified contamination of shallow aquifers underlying active agricultural lands as the dominant source of eutrophication in many watersheds (Howarth et al., 2002; Böhlke, 2002). Extensive research has been done to understand nitrate N contamination and attenuation processes in ground water (Wassenaar, 1995; Böhlke and Denver, 1995; McMahon et al., 1999), but discharge rates of ground water nitrate N to streams are commonly not matched to field application rates.

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**Abbreviations:** DOC, dissolved organic carbon; WAF, waste application field.

In fact, discharge rates are usually significantly lower than field application rates. Measured riverine N fluxes typically only account for approximately 25% of the N input into watersheds (Kendall, 1998; Cane and Clark, 1999; Kendall and Aravena, 2000; van Breemen et al., 2002). Closing the N budget by correctly linking field application rates to contaminant loads in surface waters requires an understanding of the dynamics and time scales of contaminant transport through ground water systems and riparian zones that connect ground to surface waters.

The impact of nitrate N in ground water moving from agricultural fields on nitrate N concentrations in receiving streams can be influenced by vegetated riparian buffers between the fields and the streams, hyporheic zones, and by processes within the stream (Sloan et al., 1999; Spruill, 2000, 2004). Jacobs and Gilliam (1985) evaluated nitrate N movement in shallow ground water of the Beaverdam Creek watershed in the middle Coastal Plain with fields of traditional row crops receiving fertilizer N at recommended levels. The 3-yr average nitrate N concentrations in shallow ground water beneath three row crop fields ranged from 7 to 14 mg $L^{-1}$. Similar nitrate N concentrations were found in shallow ground water at the field edge, indicating nitrate N movement from the fields toward the stream draining the watershed. In contrast, nitrate N concentrations in shallow ground water sampled 16 and 47 m from the field edge in the vegetated riparian buffer between the field and the stream were less than 0.1 mg $L^{-1}$ (Jacobs and Gilliam, 1985). This decrease in nitrate N concentration in the shallow ground water in the riparian zone was related to reducing conditions in the sediments which promoted denitrification and uptake of N by the buffer vegetation (Jacobs and Gilliam, 1985). Denitrification in the upgradient aquifer due to presence of organic carbon and electron donors, long residence times (>50 yr) along flow paths allowing even slow reactions to completely remove nitrate, dilution of nitrate enriched waters with older water having little nitrate, by-passing riparian zones due to extensive use of ditches, and drains and movement of ground water along deep flow paths below reducing zones are factors that have been associated with wide variations in the efficiency of riparian zones in removing nitrate from ground water (Puckett, 2004; Puckett et al., 2002)

The impact of the intensive animal production in North Carolina on nitrate N movement to shallow ground water and streams has received limited research attention. High concentrations of nitrate N (>100 mg $L^{-1}$) have been observed in shallow ground water below some grazed pastures receiving swine-lagoon effluent in Sampson County, NC (Mikkelsen, 1995; Sloan et al., 1999). Sloan et al. (1999) also evaluated nitrate N movement from the spray fields toward an adjacent stream and denitrification in the riparian zone. They observed high levels of denitrification in certain areas of the riparian zone. Sloan et al. (1999) also demonstrated that stream samples taken downstream contained nitrate N concentrations that were 7 mg $L^{-1}$ higher than in samples taken upstream of the swine farm and that shallow ground water from some streamside wells contained as much as 40 mg $L^{-1}$ of nitrate N. They concluded that despite conditions being suitable for loss of nitrate N by denitrification in riparian areas, the riparian buffer did not fully protect the receiving stream from high nitrate N concentrations in the ground water. Karr et al. (2001), using the positive $\delta^{15}N$ natural abundance ratios of N in the lagoon effluent (Showers et al. 1999) to trace animal waste nitrate N through the same WAF–riparian–stream system, also concluded that animal waste–derived nitrate N is exported to the adjacent surface waters. Apparently the denitrification capacity of the riparian area was overwhelmed by the concentration of nitrate N in the ground waters moving from the WAF toward the stream, or the ground water flow paths were shallow and so fast that there was not enough time for denitrification in the field-edge buffers. Fields evaluated in this study had received lagoon-effluent applications for 20 yr before more stringent regulations on land application of animal waste were imposed in 1993.

The objective of this study was to assess nitrate N movement in shallow ground water from WAFs receiving swine-lagoon effluent to an adjacent stream buffered by a variable width riparian zone, and to assess the influence of biological and hydrologic factors on this nitrate N movement in the shallow ground water. The WAFs in this system have been managed according to Section .0200 regulations "Waste Not Discharged to Surface Waters" during the 6-yr period of swine-lagoon-effluent application.

## METHODS

### Study Site

The study site (Fig. 1) is located in a watershed along the upper reach of Six Runs Creek, which flows in a southerly direction in eastern Sampson County, NC. The study site is approximately 18 km north of Clinton, NC. The stream adjacent to WAF 1 flows in a channel, but the segment adjacent to WAF 2 is impounded by two beaver dams and forms an elongated pond (Fig. 1). Below the lower beaver dam the stream flows in a channel as it exits the producer's property. Aerial photographs taken in 1988 show two swine houses in this 275 ha watershed. Aerial photographs taken in 1998 showed four swine operations with 23 swine houses in this watershed. Fields receiving swine-lagoon effluent (approximately 40 ha) and cropped with coastal bermuda grass managed for hay or as grazed pastures are situated on both sides of the creek (Fig. 1). A forested riparian buffer of variable width is located between the WAFs and the creek.

The WAF–riparian system evaluated in this study (Fig. 1) is located on the west side of Six Runs Creek. The width of the forested riparian buffer ranges from 41 to 87 m. This swine operation has a standing herd of 4400 finishing animals. Lagoon effluent is applied to 10 ha of WAFs cropped with coastal bermuda grass. Waste Application Field 1 (5.2 ha), which lies on the north end of the farm (Fig. 1), had been cropped with coastal bermuda grass cut for hay and had received lagoon effluent for 6 yr by the end of the sampling period. Waste Application Field 2 (1.8 ha) was cropped with coastal bermuda grass cut for hay for 2 yr and the last 4 yr it had been grazed intensively (100 feeder calves) between 1 July and 15 September with an occasional cutting of hay removed.

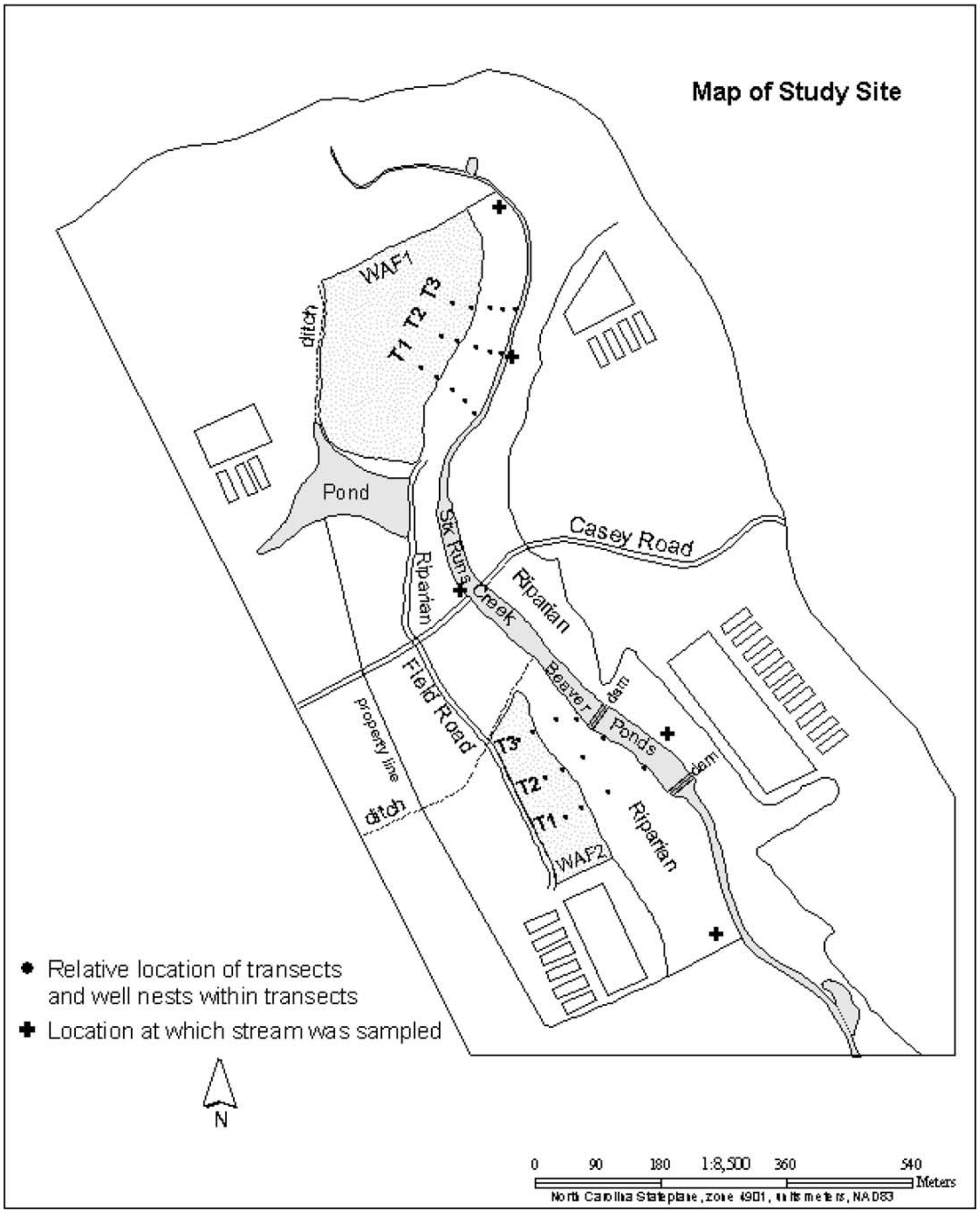


**Fig. 1. Map of the study site.**

Both WAFs (Fields 1 and 2; Fig. 1) received an average of 250 kg $ha^{-1}$ $yr^{-1}$ of plant available N for six growing seasons as swine-lagoon effluent (Table 1). The amount of N applied was derived from the producer's pumping records and effluent analysis reports from the North Carolina Department of Agriculture and Consumer Services, Agronomic Division. The laboratory uses a Kjeldahl procedure as described by Bremner (1960). Within 30 d of each waste application during the 36-mo monitoring period, effluent samples were taken from the lagoon for analysis. The TKN concentration of 20 such samples averaged $600 \pm 155$ mg $L^{-1}$. Ammonium N typically comprises 80 to 85% of the total Kjeldahl N in swine-lagoon effluent (Barker and Zublena, 1995). An N availability coefficient of 0.5 was used in calculating the amount of plant available N applied to the crops. This N availability coefficient corrects for ammonia volatilization during spraying. The producer has followed certified waste management plans approved by North Carolina Department of Environment and Natural Resources since the operation was established. His records are subject to periodic inspection by North Carolina Department of Environment and Natural Resources personnel.

Most of the soils in the WAF–riparian system are well drained. Soil types were derived from soil survey maps of Sampson County, North Carolina (USDA, 1985). Soil in WAF 1 is classified as a Norfolk series (fine-loamy, kaolinitic, thermic Typic Kandiudults). Soil in WAF 2 is classified as a Wagram series (loamy, kaolinitic, thermic Arenic Kandiudults). Soils in the riparian zone are classified as a Marvyn series (fine-loamy, kaolinitic, thermic Typic Kanhapludults) and Blanton (loamy, siliceous, semiactive, thermic Grossarenic Paleudults).

**Table 1. Plant available N applied to waste application fields (WAFs) as swine-lagoon effluent.**

| Year | Plant available N† | |
|---|---|---|
| | WAF 1 | WAF 2 |
| | kg ha$^{-1}$ | |
| 1997 | 118 | 120 |
| 1998 | 292 | 380 |
| 1999 | 360 | 348 |
| 2000 | 286 | 262 |
| 2001 | 226 | 208 |
| 2002 | 228 | 156 |
| Total | 1510 | 1474 |
| Mean ± SD | 252 ± 82 | 246 ± 103 |

† The amount of N applied was derived from the producer's pumping records and effluent analysis reports from the North Carolina Department of Agriculture and Consumer Services, Agronomic Division.

One area of the riparian zone adjacent to WAF 1 is a wetland and always has water on the surface even during extended dry periods (Fig. 1). Apparently shallow ground water from upland areas discharges in this area. The distance across this wetland area to the stream channel is 60 m. The soil in this area is black indicating the accumulation of organic matter and has a loamy texture. Soil samples along Transect 1 across the wetland area averaged 3.3 ± 0.8% C while samples taken from the field and field edge averaged 0.9 ± 0.2%. Soil in this wetland area is distinctly different from other soils at the study site, but was not mapped as a different series in the soil survey because of its small areal extent.

When this study was initiated in 2000, the riparian zone had a mixture of pine (*Pinus taeda* L.) and hardwoods such as tulip poplar (*Liriodendron tulipifera* L.), beech (*Fagus grandifolia* Ehrh.), american holly (*Ipex opaca* Ait.), and sweet gum (*Liquidambar styraciflua* L). Pine tended to grow near the outer edge of the riparian zone and hardwoods closer to the stream. In March of 2002, the landowner had marketable timber harvested from the riparian zone leaving the prescribed 15-m tree buffer along the stream. During the following spring and summer dense growth of shrubs and vines covered the logged area. The riparian zone has not been disturbed since the logging was completed. Nitrate N concentrations in ground water from riparian wells adjacent to WAF 1 were 8.6 ± 6.7 mg L$^{-1}$ before and 8.0 ± 6.5 mg L$^{-1}$ after logging. For ground water from riparian wells adjacent to WAF 2 nitrate N concentrations were 9.3 ± 4.7 mg L$^{-1}$ before and 14.4 ± 9.9 mg L$^{-1}$ after logging. It does not appear that logging had a large effect on the concentration of nitrate N in the shallow ground water moving through the riparian zone.

In the upper and middle Coastal Plain, surficial sediments usually overlie fine textured Tertiary and Cretaceous marine sediments. Daniels et al. (1975) concluded that little or no nitrate N moves through or below these clay aquitards or aquicludes into underlying deep ground water. As a result of these shallow aquitards, shallow ground water moves laterally down topography toward streams or seepage outlets along side slopes (Heath, 1980). Therefore, nitrate N movement from agricultural fields can be detected by sampling unconfined shallow ground water beneath the fields, the riparian zone, and in the first and second order streams.

## Piezometer Installation

Three transects of piezometers (hereafter referred to as wells for simplicity) were installed in each of two WAFs and the adjacent forested riparian system on the west side of Six Runs Creek for sampling of shallow ground water. Each transect consisted of four or five piezometer nests positioned on the side slope of the field, at the field edge, in the riparian zone, and at the stream edge (Fig. 1). In the WAFs, wells within a nest were placed 1 m apart and screened at three different depths: near top of water table, and at two greater depths below the water table. The depth of the screens for the two deeper wells did not overlap. Most other locations within the transects outside the WAFs had two wells per nest with the screen of the shallow well positioned near the top of the water table and the screen of the deeper well at least 0.6 m deeper than the bottom of the screen for shallow well. Streamside well nests in transects off WAF 1 had only one well. The depth of these wells ranged from 1.5 to 2.1 m. Well depths ranged from 1.2 to 6.1 m.

In accessible areas wells were drilled with a screw type auger attached to a Giddings (Windsor, CO) probe mounted on a truck and in riparian areas wells were drilled with hand augers. Well casings were constructed from 5.1-cm-diameter PVC pipe. The bottom 0.6 m of each casing was screened by drilling 3-mm holes and covering with screen sock. Sand was used to backfill space between the casings and the wall of the bore holes to 0.3 m above the screen. Then a 0.3-m layer of bentonite was placed on top of the sand layer. Spoil from the bore holes was used to backfill to within 0.45 m of the surface. Then another 0.3-m layer of bentonite capped with a 0.15-m layer of sand brought the backfill to the soil surface. This well construction prevented entry of surface water into the well casings.

Vertical and horizontal hydraulic gradients were derived from water table elevations measured with a water level tape. Relative elevations of well casings were determined by surveying.

## Well Sampling and Chemical Analyses

Wells were sampled at 30-d intervals for 24 mo and at 3-mo intervals thereafter. Sampling involved purging two well volumes and taking 100-mL samples of the water that recharged the wells after the second purging. This procedure ensured that a sample represented fresh water from the shallow aquifer at the intended depth. Water was also sampled at several points along the adjacent stream (Fig. 1) so that ground water constituent concentrations could be related to stream concentrations. Samples were transported to the laboratory in ice chests at ambient temperature, and stored at −20°C until chemical analyses could be performed. Time between initiation of sample collection at each sampling date and transfer of samples to freezer did not exceed 6 h. All water samples were passed through 0.45-μm Millipore (Billerica, MA) filters before chemical analyses were performed. Nitrate N and ammonium N were measured with a Lachat QuikChem 8000 Automated Ion Analyzer using QuikChem methods 10-107-04-1-A and 10-107-06-2-A, respectively (Lachat Instruments, 1992). The total N concentration in water samples from two sampling dates was determined with a TOC-Vcs/cm carbon analyzer equipped with a TNM unit which quantifies NO produced during sample combustion with a chemiluminescence detector (Shimadzu, Kyoto, Japan). This analysis showed that on average 90% of the total N was nitrate N and 3% was ammonium N (data not shown). Chloride concentrations were measured with a Haake Buchler digital chloridometer (Labconco Corp, Kansas City, MO). Dissolved organic carbon concentrations (DOC) were measured with a Shimadzu TOC-5050 carbon analyzer.

## Nitrogen-15 and Oxygen-18 Measurements

Many fields currently receiving animal waste previously produced row crops that received fertilizer N. Therefore, it is important to determine how much of the nitrate N in ground water beneath the riparian zone is derived from the various N sources applied to the field. Nitrogen in poultry litter and swine-lagoon effluent has $\delta^{15}N$ natural abundances ratios three to six times higher than fertilizer N sources (+15 to +25 per mil $\delta^{15}N$; Karr et al., 2001; Showers et al., 1999) as a result of ammonia volatilization (Shearer et al., 1974). Denitrification also causes an increase in the $\delta^{15}N$ natural abundance ratios in residual nitrate N in soil–water systems. This causes concern about using $\delta^{15}N$ natural abundance ratios to identify sources of nitrate N and to trace its movement through ground water systems. However, denitrification also causes enrichment of $\delta^{18}O$ ratios in residual nitrate N (Böttcher et al., 1990; Kendall, 1998; Kendall and Aravena, 2000). Measurement of both $\delta^{15}N$ and $\delta^{18}O$ natural abundance ratios of nitrate N in ground water from different parts of field-riparian systems can be used to assess the impact of denitrification on $\delta^{15}N$ natural abundance ratios of residual nitrate N in the ground water system.

At four sampling dates (March and August of 2001, February 2002, and March 2003), $\delta^{15}N$ and $\delta^{18}O$ natural abundance ratios of nitrate N in ground water beneath WAFs receiving swine-lagoon effluent and the riparian zone and adjacent stream samples were determined. The $\delta^{15}N$ and $\delta^{18}O$ natural abundance ratios of nitrate N in water samples were measured with a EA/TCEA system connected to a Finnigan MAT DELTA+XL-IRMS (Thermo-Finnigan, Bremen, Germany) after extraction using a modified version of the ion exchange method of Chang et al. (1999). The water samples were filtered with a 0.45-μm filter (GWV #12178) and nitrate N was extracted on 2 mL of Bio-Rad (Hercules, CA) AG2 8X (100–200 mesh) anion exchange resin after being pretreated with 2 mL of Bio-Rad AG50 8X (100–200 mesh) cation exchange resin. The anion exchange resin was first rinsed with 3 *M* HCl, and then repeatedly washed with deionized water until the pH of the wash water was neutral. The nitrate N was eluted from the anion exchange resin with 3 *M* HCl, neutralized, and converted to $AgNO_3$ with AgO, and then filtered to remove solid $AgCl_2$. The remaining solution was freeze dried to produce a white $AgNO_3$ powder. If the powder produced was not white, the sample was rehydrated, filtered through a Supelco (Bellefonte, PA) SPE DSC-18 (3 mL/500 mg) filter to remove organic carbon compounds, and then freeze dried again to produce white powder. Comparison of $\delta^{15}N$ of nitrate N in ground water samples with $\delta^{15}N$ of ammonium N in the applied animal waste provides an indication of the source of nitrate N in ground water from different parts of the WAF–riparian systems.

## RESULTS

### Hydrology of the Study Site

Annual rainfall at the study site for years 2000 through 2002 averaged 1168 ± 76 mm. Measurements made over a 36-mo period indicate that the mean water table depth in the riparian zone of five transects ranged from 0.5 to 2.7 m and that water table depths were greater in the riparian zone adjacent to WAF 2 (Table 2). Mean depths to the water table in streamside wells ranged from 0.14 to 1.0 m. The water table was at or near the surface throughout the riparian zone of Transect 1 adjacent to WAF 1 (Table 2).

Water table contour maps were developed from water elevation measurements and GPS coordinates of well locations using the Surfer program (Golden Software, 2003) This allowed generation of flow paths using water table data (Fig. 2 and 3). Ground water flow from both WAFs is nearly perpendicular to the stream, with little seasonal or annual variation (Fig. 2 and 3). The flow direction essentially follows the topography as each transect slopes down from field to stream. The hydraulic gradient increases as the slope increases near the stream.

**Table 2. Mean water table depths measured in the waste application field (WAF) riparian system over a 36-mo period. Values in parentheses are standard deviations.**

| Location | Water table depth | | |
|---|---|---|---|
| | Transect 1 | Transect 2 | Transect 3 |
| | m | | |
| | WAF 1 | | |
| Field | 2.7 (0.3) | 2.9 (0.3) | 2.0 (0.4) |
| Field edge | 1.3 (0.2) | 1.6 (0.3) | 1.6 (0.3) |
| Riparian 1 | 0.03 (0.12) | 0.6 (0.4) | 1.1 (0.2) |
| Riparian 2 | 0.14 (0.18) | 1.0 (0.3) | 0.9 (0.2) |
| Streamside | 0.18 (0.16) | 0.3 (0.1) | 0.9 (0.2) |
| | WAF 2 | | |
| Field | 3.6 (0.3) | 3.2 (0.1) | 1.6 (0.2) |
| Field edge | 3.4 (0.2) | 3.8 (0.2) | 1.2 (0.2) |
| Riparian | 2.8 (0.2) | 2.6 (0.2) | 0.7 (0.3) |
| Streamside | 0.7 (0.1) | 0.5 (0.1) | 0.9 (0.1) |

Vertical gradients, measured at each well cluster, were too small to measure in all well clusters, except the wells in riparian zone of Transects 1 and 2 from WAF 1. The lack of vertical hydraulic gradient, indicating near-vertical equipotential lines, indicates shallow horizontal flow over most of the transects. Recharge further from the stream will follow deeper flow paths, rising near the stream. This is consistent with the Cl data, which shows separation of shallow and deep flow paths (Fig. 4). When averaged over the 36-mo monitoring period, Cl concentrations in ground water from deep wells along six transects were consistently lower than in ground water from shallow wells (Fig. 4). In general, standard deviations for Cl concentrations in ground water from deep wells were smaller than those for ground water from shallow wells (Fig. 4). This indicates that ground water Cl concentration in the deeper flow paths is not as sensitive to activities at the soil surface as ground water from shallower flow paths.

Differences in Cl concentration with depth could reflect sampling from unconfined and confined aquifers or sampling from horizontal flow paths at different depths in an unconfined surficial aquifer. Water table and Cl data indicate that both situations exist in different areas of the study site. Crude profile descriptions were made at each well nest as bore holes were drilled for installation of wells. Clay layers with red and gray mottles were observed at various depths but continuity of these apparent low-permeability layers could not be determined. In the riparian zone of Transects 1 and 2 from WAF 1, water-table levels measured in deep wells were 0.15 to 0.3 m higher than in shallow wells (data not shown) indicating a strong upward hydraulic gradient. The lack of a vertical gradient between shallow and deep wells in the riparian zone of Transect 3 from WAF 1 and all transects from WAF 2 indicates that flow is nearly horizontal over most of the site. The clear separation of Cl concentrations in ground water from shallow and deep wells indicating little mixing of water in flow paths

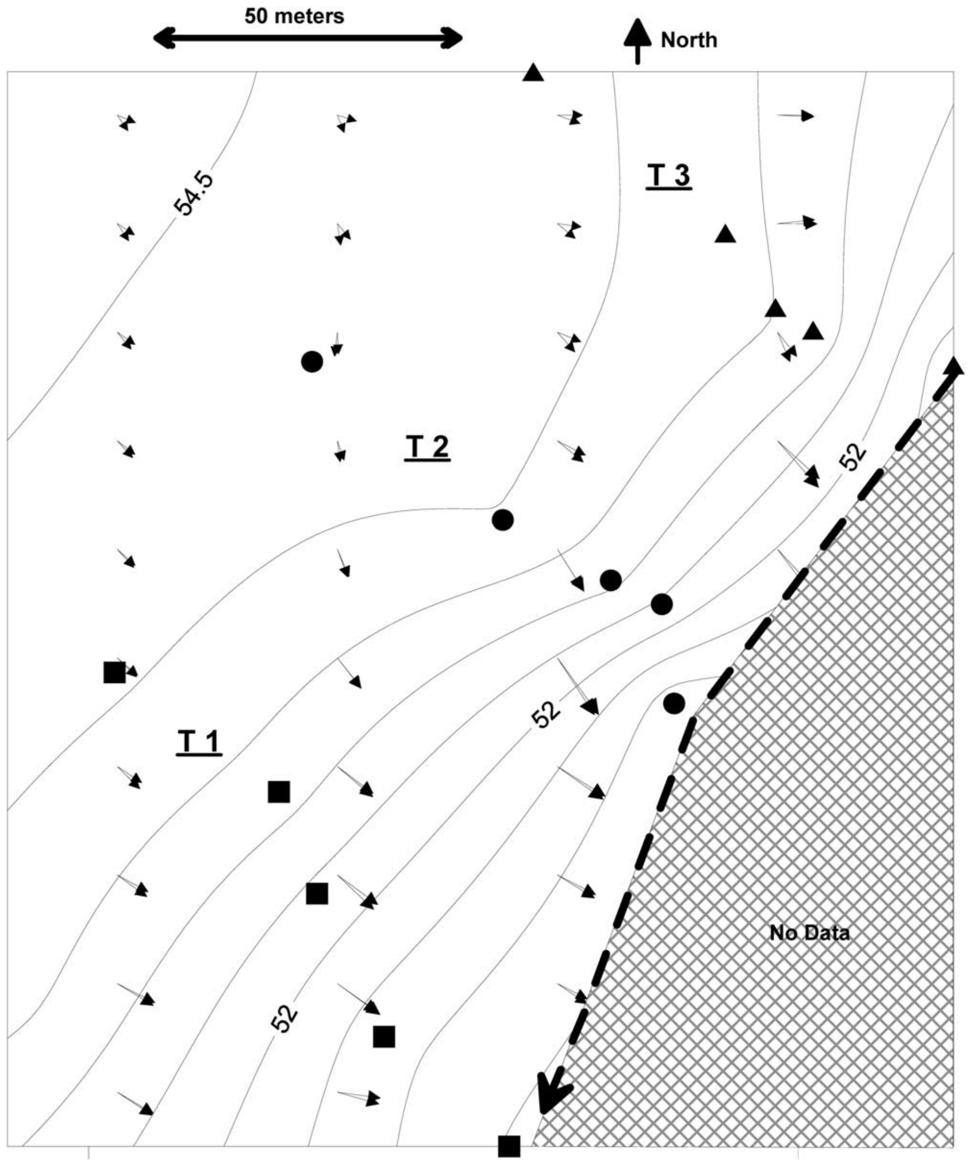


**Fig. 2.** Water table contours (meters above sea level) for Waste Application Field (WAF) 1 with horizontal gradient vectors at two different dates to show seasonal consistency. Black symbols indicate location of well nests and dashed line represents the stream. The contour interval is 0.5 m. The length of vectors represents size of the hydraulic gradient. Transect 1, ■; Transect 2, ●; Transect 3, ▲.

at different depths in the surficial aquifer is consistent with the lack of vertical hydraulic gradients.

## Composition of Shallow Ground Water under the Waste Application Fields

Since ammonium N comprised only 5% of the inorganic N in shallow ground water and stream samples, data for this N constituent are not presented in this report. Waste effluent sprayed onto the receiving fields contains primarily ammonium N and organic N, but the ammonium N must be rapidly nitrified as little is present in any shallow ground water samples collected in the WAFs. Average nitrate N concentrations in ground water beneath the WAFs ranged from 24 to 35 mg $L^{-1}$ in wells screened at 1.2- to 3.0-m depths and 6 to 9 mg $L^{-1}$ in wells screened at 3.7- to 6.1-m depths (Table 3). When averaged over all wells, nitrate N concentration in ground water beneath WAF 2 which has been used as a grazed pasture for the past 4 yr was similar to that in ground water beneath the WAF 1 which has been used for hay production (12 ± 10 mg $L^{-1}$ vs. 18 ± 14 mg $L^{-1}$; Table 3). Concentrations in ground water from shallow wells were also similar for the two spray fields that had been managed differently for the last 4 yr (24 ± 19 mg $L^{-1}$ to 35 ± 12 mg $L^{-1}$; Table 3). Large standard deviations result from spatial and temporal variation in nitrate N concentrations. This large variation precludes meaningful inferences about the impact of crop management on nitrate N movement to shallow ground water beneath the WAFs.

Dissolved organic carbon (DOC) concentrations in shallow ground water beneath the WAFs averaged less than 3 mg $L^{-1}$ (Table 4). Dissolved organic carbon levels in this range have been shown to support low rates of

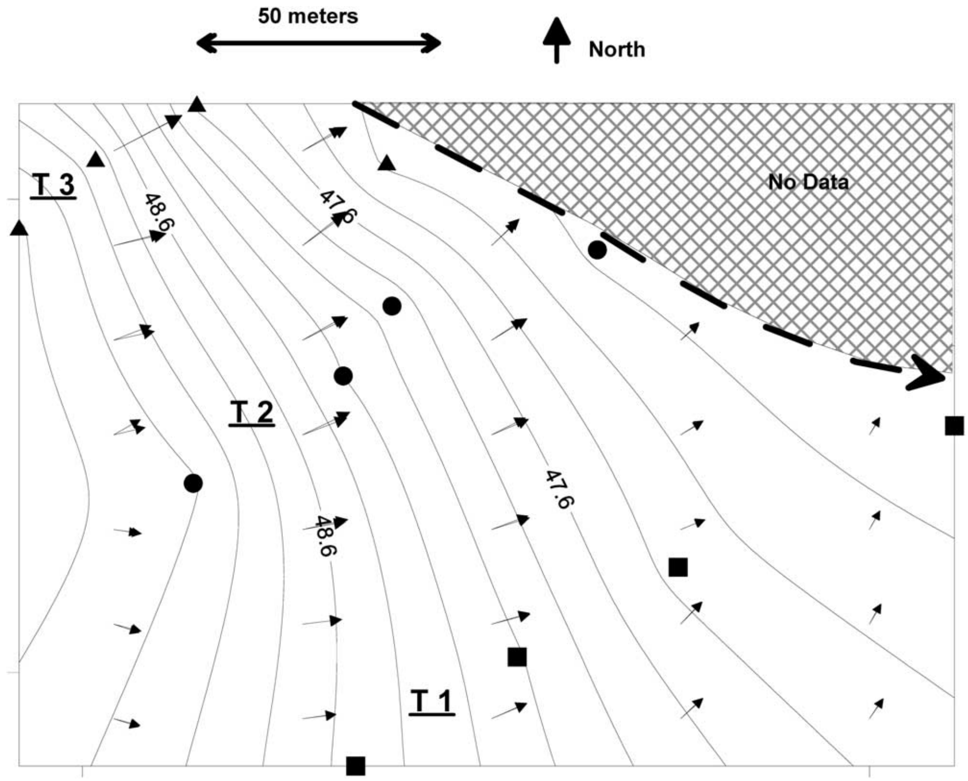


**Fig. 3. Water table contours (meters above sea level) for Waste Application Field (WAF) 2 with horizontal gradient vectors at two different dates to show seasonal consistency. Black symbols indicate location of well nests and dashed line represents the stream. The contour interval is 0.2 m. The length of vectors represents size of the hydraulic gradient. Transect 1, ■; Transect 2, ●; Transect 3, ▲.**

denitrification in subsoil (Buffington, 1994; Sloan, 1999). Thus DOC concentrations measured in this study suggest a low potential for denitrification in the soil–ground water system in these fields located on landscapes with well drained soils (Typic and Arenic Kandiudults).

## Composition of Shallow Ground Water beneath the Riparian Zone

Mean nitrate N and Cl concentrations in ground water from shallow wells in the WAF–riparian system for the 36-mo monitoring period are presented in Fig. 5. In Transect 1, ground water from the nest of riparian wells closest to the stream and from the streamside well had very low mean nitrate N concentrations ($<$0.1 mg L$^{-1}$; Fig. 5). In Transect 1, Cl concentrations in ground water from shallow field-edge to streamside wells were relatively constant at 40 mg L$^{-1}$ (Fig. 5). Thus, the [Cl] to [nitrate N] ratio is elevated dramatically in the riparian zone of this transect (Fig. 5). The decrease in nitrate N concentration in riparian zone of Transect 1 could have resulted from ground water bypassing the area due to low permeability of soil and sediments. However, the stable Cl concentration in ground water from field-edge to streamside wells indicates that ground water was moving through this area. Under suitable environmental conditions nitrate N can be converted to $N_2O$ and $N_2$ by denitrifying bacteria, whereas Cl is a conservative tracer that is not transformed by biological processes. The large increase in the [Cl] to [nitrate N] ratio in ground water from riparian and streamside wells of Transect 1 compared to that in ground water from field-edge wells indicates that denitrification contributed significantly to the decrease in nitrate N concentration in ground water moving through this portion of the riparian buffer. We cannot rule out the possibility that some of the decrease in nitrate N concentration in the shallow ground water resulted from N uptake by the vegetation. Peterjohn and Correll (1984) reported that forest vegetation removed 15 kg N ha$^{-1}$ yr$^{-1}$ from shallow ground water moving beneath a riparian zone. This accounted for only 33% of the nitrate N lost from the ground water moving through the riparian zone (Peterjohn and Correll, 1984).

Mean nitrate N and Cl concentrations in ground water from shallow wells at each nest decreased from WAF 1 to streamside wells in Transects 2 and 3 (Fig. 5). The [Cl] to [nitrate N] ratio was relatively constant across much of the riparian zone and was approximately 10-fold higher in ground water from streamside wells. A comparison of the chemical composition of ground water from shallow and deep wells in the riparian zone of Transects 2 and 3 (Table 5) reveals marked differences in [Cl] to [nitrate N] ratio with depth. The [Cl] to [nitrate N] ratios were 1.4 to 17-fold greater in ground water from deep than from shallow wells at these riparian nests. Therefore, mixing of ground water with different [Cl] to [nitrate N] ratios at or near the stream could account for some of the increase in [Cl] to [nitrate N] ratios in ground water from streamside wells in these two transects (Fig. 5).

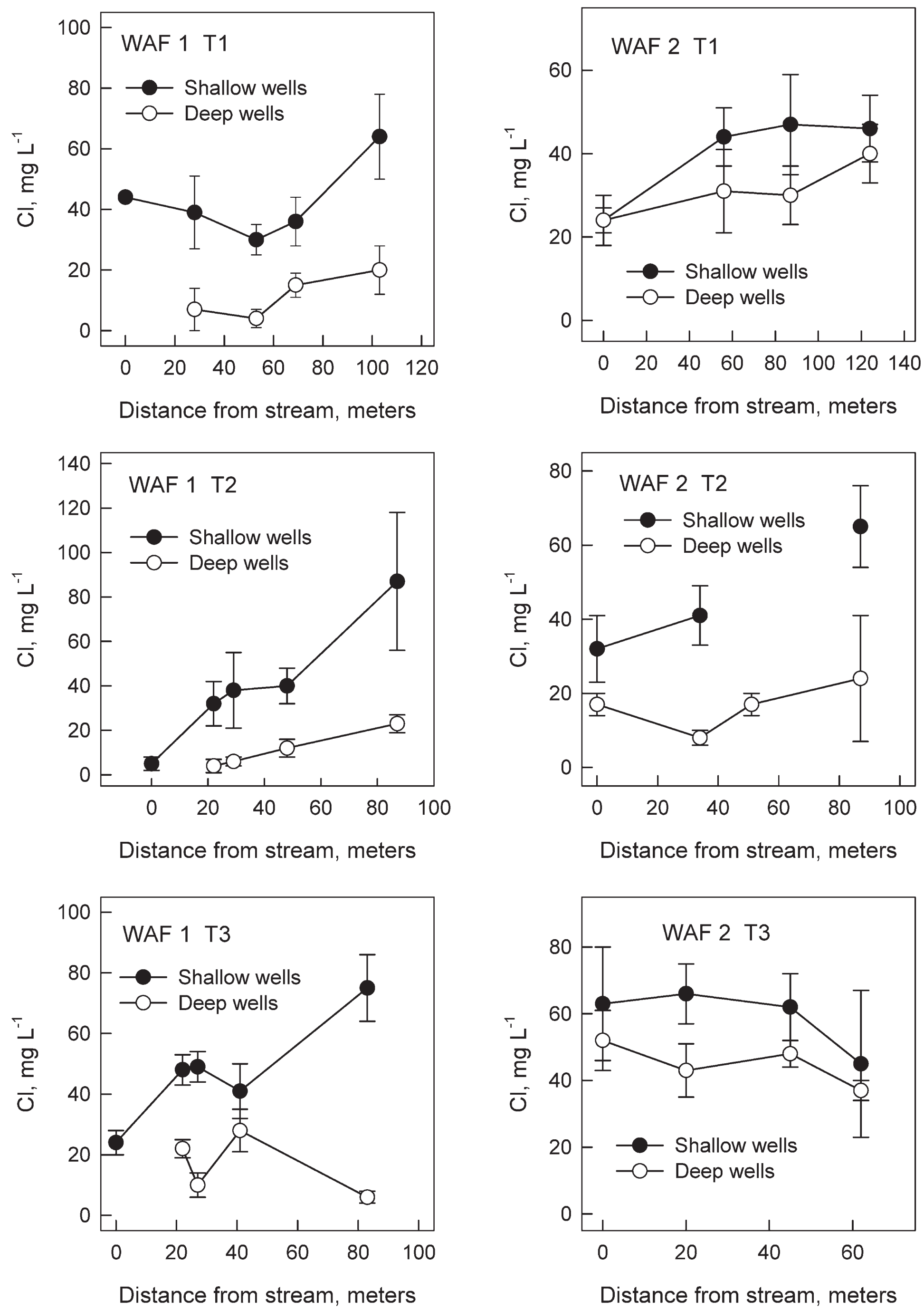


**Fig. 4. Chloride concentrations in ground water from shallow and deep wells averaged over the 36-mo sampling period.**

In WAF 2, Transects 2 and 3, nitrate N concentrations decreased in water from riparian wells compared to water from field-edge wells, but the [Cl] to [nitrate N] ratios were relatively constant, indicating dilution rather than denitrification was the cause of decreasing nitrate N concentration in ground water moving through this portion of the riparian zone. Mean nitrate N concentration in ground water from streamside wells adjacent to WAF 2 varied from <0.5 mg L$^{-1}$ in Transect 1 to 15 mg L$^{-1}$ in Transect 3. Nitrate N concentrations and [Cl] to [nitrate N] ratios (Fig. 5) for ground water from streamside wells of Transect 1 from WAF 2 appear to indicate denitrification in the riparian area. However, water table depths (Table 2) across this transect and DOC concentrations (Fig. 6) in ground water from the streamside and riparian wells do not favor denitrification. While the mean nitrate N concentration in ground water from the streamside wells of this transect was <0.5 mg L$^{-1}$, concentrations began to increase 20 mo into the monitoring study reaching 3 mg L$^{-1}$ at the 36-mo sampling (data not shown). The distance across the riparian zone is 87 m. Thus, it appears that nitrate N from the WAF did not reach streamside wells until 20 mo after sampling began.

Table 3. Summary of nitrate N concentrations measured over a 36-mo period in shallow ground water under waste application fields (WAFs) receiving lagoon effluent and cropped in coastal bermuda grass hay for 6 yr (WAF 1) or managed for coastal bermuda grass hay production for the initial 2 yr and as a grazed pasture for the following 4 yr (WAF 2).

| | Well screen depth below soil surface (m) | | | |
|---|---|---|---|---|
| | 1.8–2.4 | 2.4–3.0 | 3.7–4.3 | All depths |
| | | WAF 1 | | |
| Observations | 27 | 84 | 78 | 189 |
| Mean nitrate N concentration, mg $L^{-1}$ | 35 ± 12 | 21 ± 14 | 6 ± 4 | 18 ± 14 |
| Maximum nitrate N concentration, mg $L^{-1}$ | 65 | 74 | 13 | 74 |
| ≥1 mg $L^{-1}$, % | 100 | 100 | 100 | 100 |
| ≥10 mg $L^{-1}$, % | 100 | 82 | 36 | 65 |
| | Well screen depth below soil surface (m) | | | |
| | 1.2–3.0 | 3.0–4.3 | 4.9–6.1 | All depths |
| | | WAF 2 | | |
| Observations | 56 | 127 | 129 | 312 |
| Mean nitrate N concentration, mg $L^{-1}$ | 24 ± 19 | 9 ± 3 | 9 ± 2 | 12 ± 10 |
| Maximum nitrate N concentration, mg $L^{-1}$ | 84 | 19 | 12 | 84 |
| >1 mg $L^{-1}$, % | 100 | 100 | 100 | 100 |
| >10 mg $L^{-1}$, % | 80 | 50 | 32 | 49 |

Table 4. Summary of dissolved organic carbon (DOC) concentrations measured over a 36-mo period in shallow ground water under waste application fields (WAFs) receiving lagoon effluent and managed for hay production or as a grazed pasture.

| | Well screen depth below the soil surface (m) | | | |
|---|---|---|---|---|
| | 1.2–3.0 | 2.4–4.3 | 3.7–6.1 | All depths |
| Observations | 60 | 208 | 250 | 518 |
| Mean DOC concentration, mg $L^{-1}$ | 2.0 ± 1.0 | 2.3 ± 1.4 | 2.1 ± 1.4 | 2.2 ± 1.4 |
| Maximum DOC concentration, mg $L^{-1}$ | 7 | 8 | 15 | 15 |

The large increase in mean nitrate N concentration in ground water from streamside wells compared to riparian wells in Transect 3 from WAF 2 is probably associated with the streamside well nest being offset from the line of the other three well nests and, as a result, ground water from a different flow path was sampled. The low nitrate N in ground water from the riparian nest of Transect 3 may be associated with some level of denitrification. The mean water table depth of 0.7 m is favorable for denitrification (Table 2) but the mean DOC concentration averaged only 2.5 mg $L^{-1}$. Groffman et al. (1996) demonstrated that dentrification rates in microcosms that simulated ground water conditions in a riparian forest were carbon limited when DOC concentrations in the ground water were in the 2 to 5 mg $L^{-1}$ range.

In five of the six transects the mean DOC concentrations in ground water from shallow wells in the riparian zone were below 5 mg $L^{-1}$ and as low as 2 to 3 mg $L^{-1}$ in many wells (Fig. 6). This indicates denitrification in much of the riparian system would be limited by the availability of carbon. The mean DOC concentrations in ground water from riparian and streamside wells of Transect 1 from WAF 1 were 12 to 14 mg $L^{-1}$ (Fig. 6). Sloan (1999) reported high rates of dentrification (up to 2 g N $m^{-2}$ $d^{-1}$) in a forested riparian zone when ground water passing through the soil and sediments contained 25 mg $L^{-1}$ DOC. Therefore, the level of DOC is consistent with denitrification being a major contributor to the low nitrate N concentrations in ground water moving through the riparian zone of Transect 1 from WAF 1 (Fig. 5).

## Tracing Nitrate Nitrogen Movement with $\delta^{15}N$

The $\delta^{15}N$ ratio of nitrate N was measured in shallow ground water from WAF wells to streamside wells over a 24-mo period (Fig. 7). At the beginning of this period the WAFs had received swine-lagoon effluent for 4 yr. At all sampling dates, $\delta^{15}N$ ratios of nitrate N in ground water from field and field-edge wells were in the range of 18 to 30‰ which is well above the upper limit of $\delta^{15}N$ ratios reported for N fertilizers, 10‰ (Shearer and Kohl, 1993), and for N in soil organic matter, 6‰ (Heaton, 1984), and similar to values reported for $\delta^{15}N$ ratio of ammonium N in swine-lagoon effluent (Showers et al., 1999; Karr et al. 2001). By the last sampling date (3 March; Fig. 7), $\delta^{15}N$ ratios of nitrate N in ground water from streamside wells of four transects were ≥20‰.

When propagation of a $\delta^{15}N$ signal is used to trace nitrate N movement through ground water, the impact of denitrification on the signal must be assessed. Karr et al. (2001) concluded that as long as denitrification in natural waters or ammonia loss during land application of lagoon effluent do not remove most of the N being applied, isotopic shifts induced by these processes do not greatly alter the propagated mean or weighted mean $\delta^{15}N$ signal of nitrate N in affected ground water and adjacent surface waters. Denitrification in ground water systems causes enrichment of $^{18}O$ as well as $^{15}N$ in nitrate (Aravena and Robertson, 1998; Böttcher et al., 1990; Kendall and Aravena, 2000). If significant denitrification occurs as nitrate-containing ground water moves through a riparian zone, a significant linear relationship between $\delta^{18}O$ ratio of nitrate (*y* axis) and $\delta^{15}N$ ratio of nitrate (*x* axis) with a slope of 0.5 should be observed

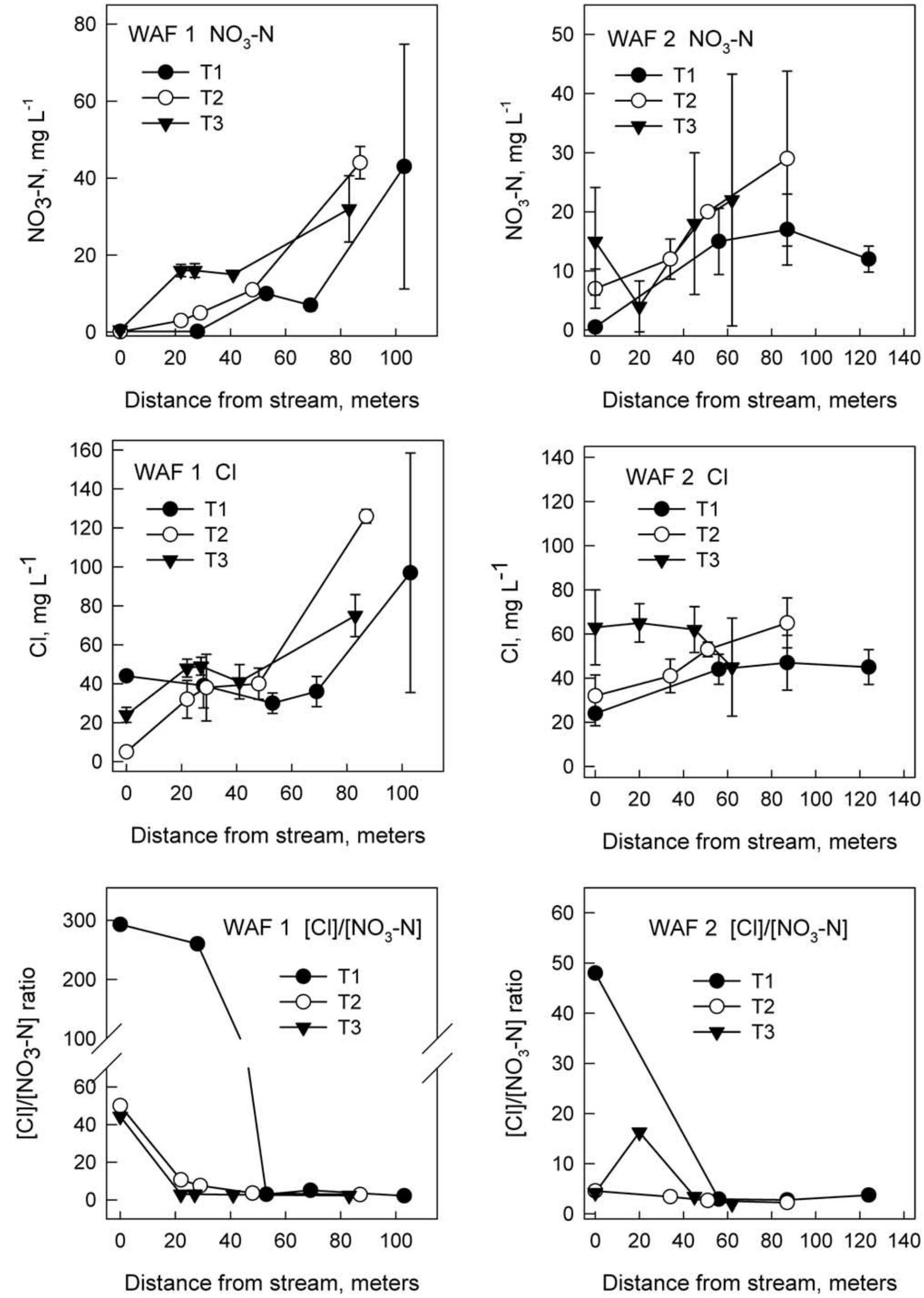


**Fig. 5.** Nitrate N and Cl concentrations and [Cl] to [nitrate N] ratios in shallow ground water moving from waste application fields (WAFs) to stream. Values are means for all samples taken from the shallow wells at each nest over a 36-mo period.

(Kendall, 1998; Kendall and Aravena, 2000). Data plotted in Fig. 8 show that in general there is little relationship between $\delta^{18}O$ ratio of nitrate and $\delta^{15}N$ ratio of nitrate in most of the WAF–riparian system. The only values that fit a line with a slope of 0.5 were derived from riparian wells in Transect 3 from WAF 2. Other measurements (Fig. 5) also indicate high levels of denitrification in the riparian zone of Transect 1 from WAF 1; however, nitrate N concentrations in the ground water were so low that stable isotope measurements were not feasible. Evidence for significant denitrification in only two of six transects indicates that denitrification is not a major factor contributing to $^{15}N$ enrichment of nitrate N in shallow ground water in much of the WAF–riparian system. Thus, it is reasonable to use the $\delta^{15}N$ signal of nitrate N to trace movement of effluent-derived N in the shallow ground water between the WAFs and Six Runs Creek. The $\delta^{15}N$ ratios of nitrate N of ≥20‰ in streamside wells of four transects indicate some discharge of N derived from WAF effluent to the stream. Nitrate N concentrations ($<0.2$ mg $L^{-1}$) in ground water from riparian and streamside wells of

**Table 5. Chloride and nitrate N concentrations in ground water from shallow and deep wells in the riparian zone between Waste Application Field (WAF) 1 and the stream. Values are means for 36-mo sampling period. Standard deviations are in parentheses.**

| Transect | Well depth | Nitrate N | Cl | [Cl] to [nitrate N] ratio |
|---|---|---|---|---|
| | | mg L$^{-1}$ | | |
| 1 | shallow | 0.15 (0.13) | 39 (11) | 259 |
| | deep | 0.25 (0.18) | 7 (7) | 29 |
| 2 | shallow | 3.4 (4.1) | 32 (10) | 9 |
| | deep | 0.3 (0.5) | 4 (3) | 13 |
| 3 | shallow | 15.5 (1.6) | 48 (5) | 3 |
| | deep | 0.43 (0.93) | 22 (3) | 52 |

Transect 1 from WAF 1 indicate that denitrification removes most of the nitrate N moving through this part of the system and that water discharging to the stream in this area contains very low levels of nitrate N.

## Nitrate Nitrogen in Six Runs Creek and Its Origin

The headwaters of Six Runs Creek originate in a large field that has produced cotton (*Gossypium hirsutum* L.) each year since the year 2000. Nitrate N concentrations in water from the stream entering the farm averaged 3.0 mg L$^{-1}$ and remained the same in the free flowing stream segment adjacent to WAF 1 (Table 6). Nitrate N concentrations in water from the Casey Road bridge to the McCullen Road bridge 2.0 km downstream (not shown on map) from the farm averaged <1.0 mg L$^{-1}$ (Table 6). The standard deviations given in Table 6 reflect seasonal variation in nitrate N concentrations at different sampling locations. Highest values were measured in winter and lowest values in spring and summer. The Cl concentrations in the stream were relatively constant (18–25 mg L$^{-1}$) along the entire the 3.3 km stream segment while the nitrate N concentrations decreased from 3 to 4 mg L$^{-1}$ in the stream segment adjacent to WAF 1 to 1 mg L$^{-1}$ or less from the Casey Road bridge to the McCullen Road bridge 2.0 km downstream from the farm. The [Cl] to [nitrate N] ratios were 6 for water from the upper stream and ranged from 21 to 35 for sampling points from Casey Road bridge to McCullen Road bridge. Nitrate N concentrations in the stream are 8- to 20-fold lower than in the shallow ground water under the WAFs (Tables 2 and 3).

The Cl concentration in the stream (Table 6) was about 50% of the concentration in ground water from streamside wells adjacent to WAF 2 (Fig. 5). This indicates dilution with low Cl ground water from deeper flow paths near the stream. Some of the decrease in nitrate N concentration in the stream compared to the ground water from streamside wells can be attributed to this dilution effect. However, the constant Cl concentration along the 3.3-km section of the stream and the lower nitrate N concentration downstream from Casey Road compared to upstream (Table 6) indicate biological processes in the stream remove nitrate N. Absorption by aquatic plants or denitrification as nitrate N passes through the hyporheic zone in the stream bottom (Spruill 2000, 2004) may account for this removal.

Changes in the $\delta^{15}N$ ratios of nitrate N in the stream over a 24-mo period compared to the $\delta^{15}N$ ratios of

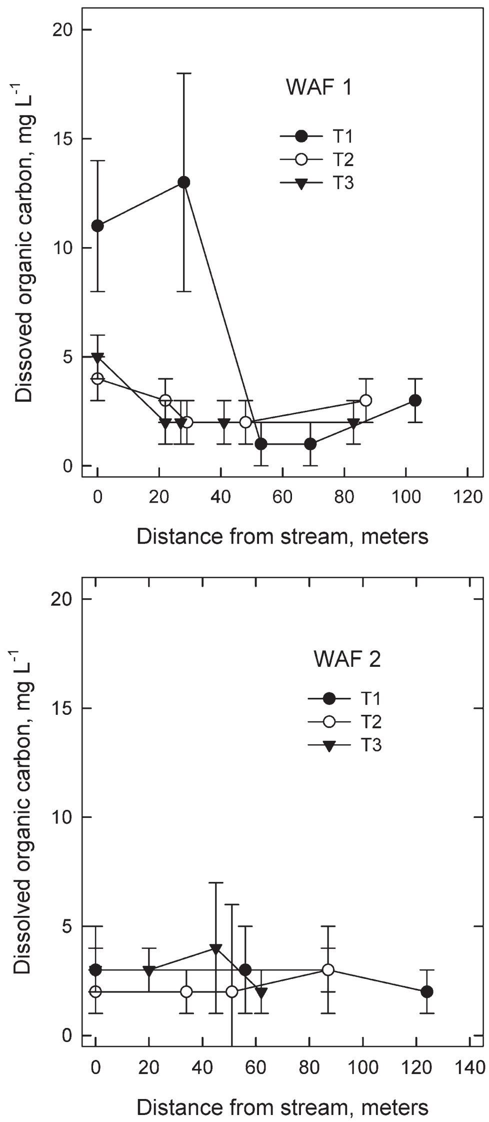


**Fig. 6. Dissolved organic carbon (DOC) concentrations in shallow ground water moving from waste application fields (WAFs) to the stream. Values are means for all samples taken from the shallow wells of each nest over a 36-mo period.**

nitrate N in ground water under the spray fields are illustrated in Fig. 9. At the beginning of the 24-mo period (4 yr after waste application to the fields was initiated), $\delta^{15}N$ ratios of nitrate N in the stream averaged 5‰ (Fig. 9). This value is within the range of $\delta^{15}N$ ratios reported for fertilizer N (Shearer et al., 1974). At the end of the 24-mo period the $\delta^{15}N$ ratios of nitrate N in the stream averaged 16‰ (Fig. 9). This is at the low end of the range of $\delta^{15}N$ values reported for N in swine-lagoon effluent (Karr et al., 2001). This mean value for the stream after 24 mo is approaching the mean $\delta^{15}N$ of nitrate N in shallow ground water under the WAFs (Fig. 9). These observations clearly demonstrate that some nitrate N derived from swine-lagoon effluent applied to the WAFs is discharging to the stream.

## DISCUSSION

In the Coastal Plain region of the United States, forested riparian buffers have been shown to be extremely

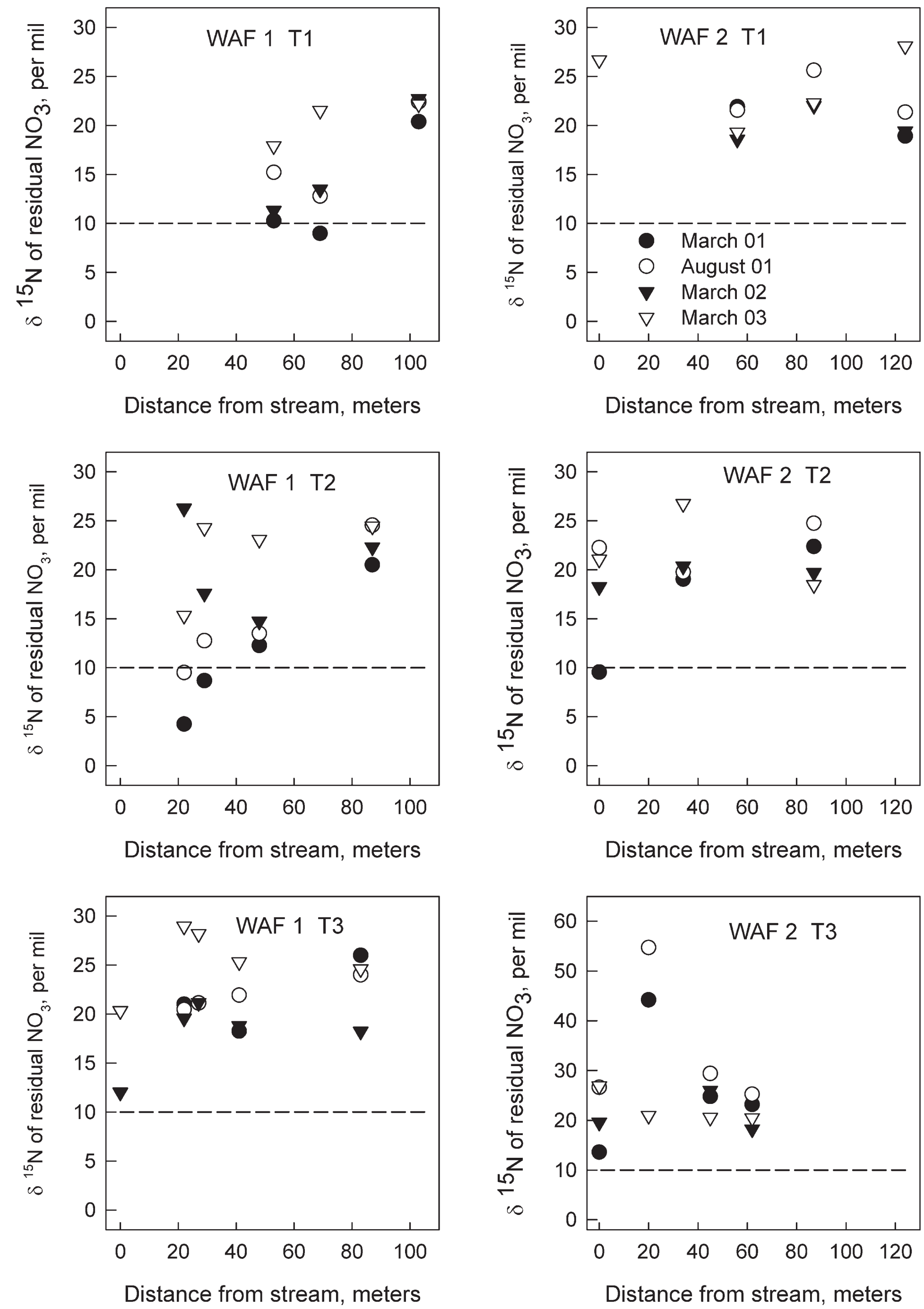


**Fig. 7.** $\delta^{15}N$ **natural abundance ratios for nitrate N in shallow ground water moving from waste application fields (WAFs) to the stream. The horizontal dashed lines represent the upper end of the range of** $\delta^{15}N$ **natural abundance ratios reported for fertilizer N sources.**

effective in removing 80 to 90% of the nitrate N in ground water moving from fields receiving N toward streams (Jacobs and Gilliam, 1985; Lowrance et al., 1984; Peterjohn and Correll, 1984; Simmons et al., 1992). Many studies indicate that denitrification is the major mechanism in riparian buffers for removal of nitrate N in ground water (Haycock and Burt, 1993; Jacobs and Gilliam, 1985; Pinay et al., 1993; Schipper et al., 1993, 1994). Shallow (0.5–2 m) flow paths which ensure intermittent saturation of upper soil horizons and saturated soil zones containing soluble organic carbon create an anaerobic environment suitable for denitrification (Dukes et al., 2003)

The riparian buffer widths at our study site ranged from 41 to 87 m which exceeds the 15-m buffer widths required for streams in the Neuse River basin of North Carolina (North Carolina Department of Environment and Natural Resources, 2004). Yet some effluent-derived nitrate N is moving through the riparian zone in shallow ground water into the stream (Fig. 7 and 9). This is especially true for the riparian system between WAF 2 and the stream where DOC levels in the shallow ground

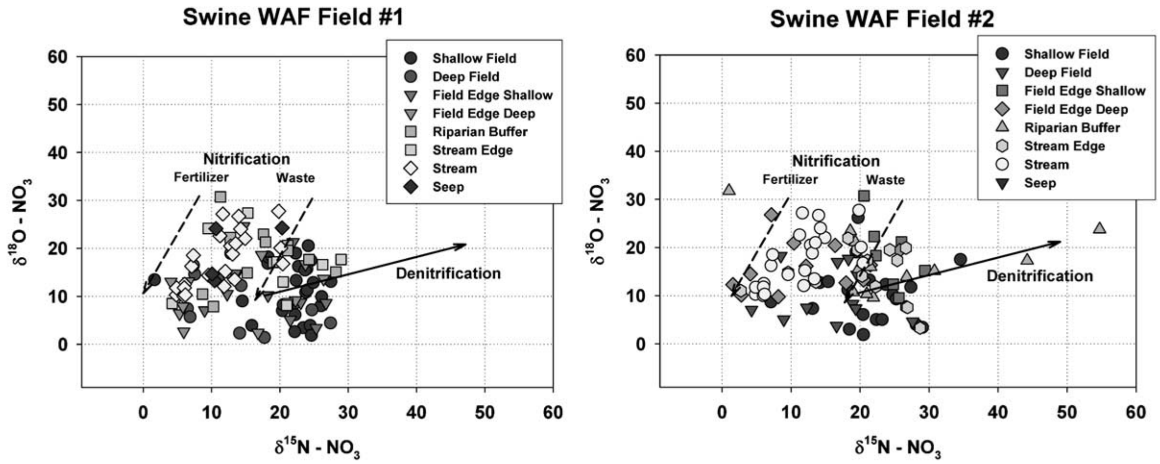


**Fig. 8. Relationships between $\delta^{18}$O of nitrate and $\delta^{15}$N of nitrate in shallow ground water moving from waste application fields (WAFs) to the stream. Solid line represents the predicted $\delta^{18}$O-nitrate vs. $\delta^{15}$N-nitrate relationship when nitrate N is denitrified in the system. Dashed lines represent predicted $\delta^{18}$O-nitrate vs. $\delta^{15}$N-nitrate relationships when fertilizer N and animal waste N are nitrified.**

water are low and water table depths are greater than 2 m in two transects (Table 2 and Fig. 7). Evidence of significant levels of denitrification was noted in only two of six transects (Fig. 5, 6, and 8). These observations indicate large spatial variations in denitrification activity in this WAF–riparian buffer system across a relatively small area. The hydrology of this riparian system is quite different from that of systems studied by Jacobs and Gilliam (1985) and Dukes et al. (2003). These systems had water table depths of 1 m or less in the riparian zone and significant denitrification was observed in the riparian buffers.

Approximate transport times from the WAFs to the stream can be estimated using Darcy's law. Hydraulic gradients were averaged over transects (from the WAFs to the stream) for different seasons and years of data. Approximate values were 0.036 for WAF 1 and 0.024 for WAF 2 with standard deviations of 0.005 and 0.010. An approximate saturated hydraulic conductivity of $5 \times 10^{-4}$ cm s$^{-1}$ and a porosity of 0.40 were estimated for the heterogeneous sandy soil with discontinuous clay lenses from the tables in Freeze and Cherry (1979, p. 29 and 37). The calculated velocities, $v_x = KI/n$, equal $4.5 \times 10^{-5}$ cm s$^{-1}$ and $3 \times 10^{-5}$ cm s$^{-1}$ for WAFs 1 and 2, respectively. This translates into annual velocities of 10 to 14 m yr$^{-1}$. With distance of travel from field edge to the stream averaging 50 m for both WAFs, travel times for ground water movement from edge of field to the stream are calculated as 3.5 and 5.3 yr for WAFs 1 and 2, respectively. While these values are general estimates of travel time, they are consistent with the time from beginning of effluent application to the time of measurable increase in $\delta^{15}$N of nitrate in the stream (4.5 yr; Fig. 9).

The short travel times for shallow ground water in this system indicate that changes in management of WAFs could influence the amount of nitrate N delivered to the stream in a relatively short period of time. This contrasts with situations where agricultural activities in small watersheds in Iowa have resulted in elevated nitrate N concentrations in deep ground water with travel times of decades (Tomer and Burkart, 2003). In such a system decades would be required for altered management practices to impact nitrate N delivery to the stream.

The intensity of animal production in this small watershed, location of WAFs adjacent to a stream with riparian buffer zones, and management of WAFs within the watershed according to recommended best management practices make this an excellent system for assessing the impact of the natural system on movement of nitrate N from WAFs to the stream in shallow ground water and on the concentration of nitrate N in the stream. Four swine operations in this 275-ha watershed have a standing herd of 17 000 finishing animals and all have been under Section .0200 regulations throughout their existence. While $\delta^{15}$N natural abundance ratio measurements indicate that some waste-derived nitrate N is delivered to the stream in shallow ground water, the nitrate N concentration in the stream averaged 1.0 mg L$^{-1}$ from the beaver pond to McCullen Road (Table 6). The relatively low nitrate N concentrations in this stream contrast with nitrate N concentrations of 10 mg

**Table 6. Summary of nitrate N and Cl concentrations in water samples taken along a 3.3-km segment of Six Runs creek including a 1.2-km segment flowing by the waste application fields (WAFs) and associated riparian buffers. Values in parentheses are standard deviations.**

| Location sampled | Monthly observations | Nitrate N | Cl | [Cl] to [nitrate N] ratio |
|---|---|---|---|---|
| | | mg L$^{-1}$ | | |
| Stream as enters farm | 6 | 3 (1.6) | 19 (5.0) | 6.3 |
| Stream off Transect 3, WAF 1 | 9 | 3.6 (1.0) | 24 (6.1) | 6.7 |
| Stream Casey Rd. bridge | 11 | 1.9 (1.2) | 24 (5.1) | 12.6 |
| Stream off Transect 1, WAF 2 | 11 | 0.6 (1.0) | 21 (3.9) | 35.0 |
| Stream as exits farm | 7 | 1.0 (0.9) | 21 (3.5) | 21.0 |
| Stream McCullen Rd. bridge† | 9 | 0.8 (0.7) | 18 (3.7) | 22.5 |

† Not shown in Fig. 1.

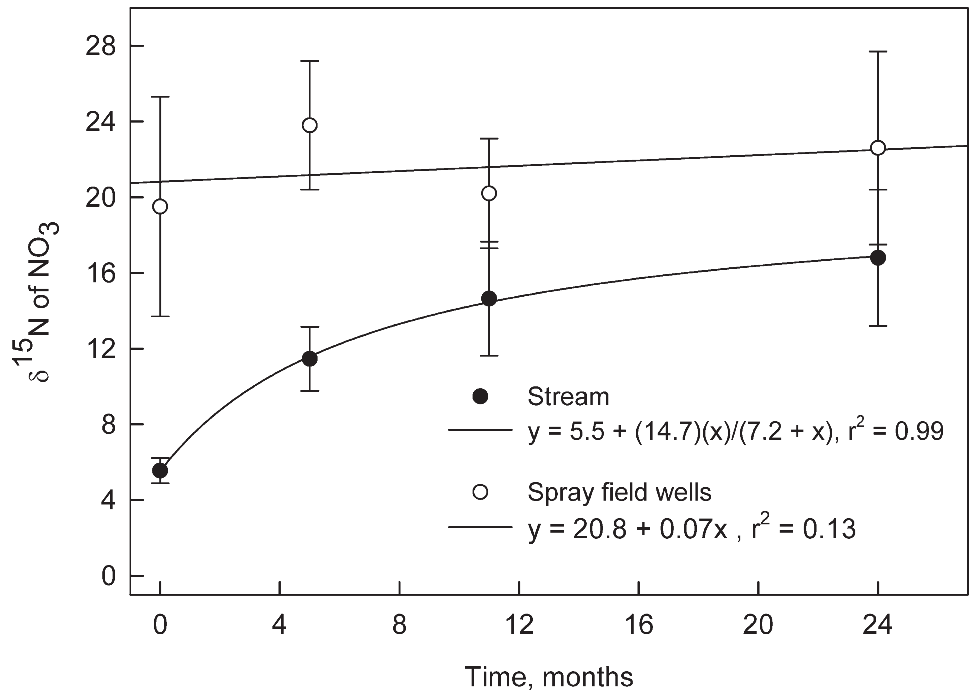


**Fig. 9. Comparison $\delta^{15}N$ natural abundance ratios for nitrate N in water from the stream and from wells in the waste application fields (WAFs) over a 24-mo period.**

$L^{-1}$ measured in streams draining small watersheds in Iowa with a high intensity (80–90% of the land area of the watersheds) of row crop production (Schilling and Libra, 2000). Likewise, Sloan et al. (1999) reported nitrate N concentrations in a stream draining a watershed with 20-yr-old WAFs and intervening riparian buffers as high as 25 mg $L^{-1}$. Results from this study indicate that several factors may contribute to the relatively low nitrate N concentration in the upper reach of Six Runs Creek compared to other systems: (i) dilution of high nitrate N water in shallow horizontal flow paths with low nitrate N ground water from deeper horizontal flow paths at or near the stream, (ii) nitrate N absorption by aquatic plants or denitrification as ground water passes through the hyporheic zone during discharge through the bottom of the stream, and (iii) denitrification in some areas of the riparian zone.

In summary, managing the WAFs at this site according to Section .0200 regulations has not prevented movement of some waste-derived N through the riparian buffers (41–87 m in width) into the adjacent stream. While some forested riparian buffers may not be highly effective in removing nitrate N from shallow ground water, producers should be encouraged to maintain existing riparian buffers between effluent spray fields and adjacent streams. Buffers may function in concert with other biological and hydrologic factors to maintain relatively low nitrate N concentrations in streams draining watersheds with a high intensity of swine production.

## ACKNOWLEDGMENTS

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