

NITROGEN DYNAMICS IN A PIEDMONT ONSITE WASTEWATER TREATMENT SYSTEM

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Abstract. Total Maximum Daily Load (TMDL) analyses of lakes and reservoirs with nutrient impairments commonly identify onsite wastewater treatment systems (i.e. septic systems) as an important potential source of nitrogen (N). In most cases, however, the contribution from on-site wastewater systems (OWSs) is difficult to estimate because of uncertainty about how much of the N is lost due to denitrification. The objective of this study was to quantify wastewater N concentrations in the soil and the extent of denitrification in an OWS commonly used in the Piedmont region.

An OWS was installed in Griffin, GA and vadose zone N concentrations were monitored at different depths in the drainfield. Nitrate ($\text{NO}_3\text{-N}$) concentrations remained low for the first four months after wastewater dosing began and then increased monthly for the next 11 months for all depths beneath the drainfield. The average $\text{NO}_3\text{-N}$ concentration at 90 cm on the last sampling date was 20 mg L^{-1} . Denitrification was characterized in the drainfield by using Cl as a conservative tracer and calculating N/Cl ratios. We estimated that denitrification may account for up to 70% of N removal in OWSs. Currently, we are calibrating a model of the N dynamics using data from our experimental site. Once the model is calibrated, it will be used to estimate denitrification losses by OWSs in other regions by varying the soils and climate data.

INTRODUCTION

As of 2007, an estimated 20% of total U.S. houses use onsite wastewater systems (OWSs), an increase of 1.5% since 1985 (USEPA, 2007). Approximately 47% of OWSs serve suburban communities and 50% serve rural communities (USEPA, 2007). In the South, approximately 46% of homes are served by OWSs, followed by 22% in the Midwest, 19% in the Northeast, and 13% in the West (USEPA, 2007). A number of OWSs still rely on outdated technologies (outhouses, dry wells, and cess-pools) but with the increased awareness of the need to protect our nation's water supplies, a considerable amount of time and money is being devoted to the improvement and implementation of more efficient wastewater treatment systems. Still there are many aspects about OWS function that are not adequately understood and research in these areas is essential. Specific areas that need to be

better understood are biomat development, infiltration reductions, drainfield sizing, wastewater treatment efficiency, and the percentages of treated wastewater that are returned to aquifers.

Nutrient impairments are the third most common reason for state water bodies to be listed on the EPA 303(d) list (USEPA, 2011). Onsite wastewater treatment systems (OWS) are commonly listed as a potential source of N in TMDL reports on nutrient impairments (USEPA, 2007). An example is the recently developed TMDL for Lake Allatoona (GADNR, 2009). Load allocations (LA) are estimated for each source within a watershed when TMDLs are developed. Generalized load function models based on mass balances are often used to predict the LA for nonpoint sources, including OWSs. However, the contribution from OWSs is difficult to estimate because of uncertainty about how much of the N is lost due to denitrification. Common assumptions in OWS models for denitrification losses range from 0 to 50% of the input load (McCray et al., 2008).

N is a limiting nutrient for algal and microbial growth in lakes and reservoirs. High $\text{NO}_3\text{-N}$ concentrations in groundwater can accelerate eutrophication in surface waters and have also been linked to methemoglobinemia (i.e. blue baby syndrome) in infants who consume contaminated water. OWSs have been identified as contributing a significant amount of $\text{NO}_3\text{-N}$ to aquifers (Hinkle et al., 2007; Welhan and Poulson, 2009). However, there are conflicting results regarding the amount of $\text{NO}_3\text{-N}$ from OWSs reaching groundwater. Gold et al. (1990) reported $\text{NO}_3\text{-N}$ concentrations taken from lysimeters installed 1 m below several OWSs exceeded 10 mg L^{-1} . In contrast, groundwater $\text{NO}_3\text{-N}$ concentrations reported by Cogger and Carlile (1984) ranged <0.5 to 4.6 mg L^{-1} . The highest $\text{NO}_3\text{-N}$ concentrations were observed at the 1.5-m wells in seasonally saturated soil.

The major mechanisms for reducing $\text{NO}_3\text{-N}$ concentrations in OWSs are dilution and denitrification. The effect of dilution can be estimated using mass balance models, but denitrification rates in OWSs are largely unknown. McCray et al. (2008) analyzed 83 studies of OWS systems, 51 of which were field studies (the others were done in the lab). None of the field studies reported a denitrification rate. The objective of this study was to quantify N concentrations and estimate the extent of denitrification

in the vadose zone beneath an OWS installed in a well-drained Piedmont soil.

MATERIALS & METHODS

A conventional OWS was installed at the Dempsey Research Farm located in Griffin, GA in August, 2008. The system consisted of a 4170 L dosing tank where wastewater was temporarily stored prior to entering the septic tank. There was a 3875-L, 2-compartment, concrete septic tank installed in the ground with approximately 15 cm protruding from the surface to allow easy access for sampling. The septic tank was connected to a concrete distribution box (D-box) that was placed on a level concrete slab. The drainfield consisted of three 10 m long trenches that were 2.5 m apart from center-to-center, each connected to the D-box. The gravel bed of each trench was 30 cm thick, with a 10 cm diameter perforated pipe 10 cm above the infiltrative surface, and covered with 20 cm of gravel. Contractor paper was placed on top of the gravel bed prior to backfilling with soil. A Georgia Department of Community Health (GADCH) certified septic installer installed the system, adhering to GADCH guidelines for installation. A GADCH inspector approved the system before any of the trenches were backfilled.

The OWS was installed in a Cecil series soil (fine kaolinitic thermic typic kanhapudult). The landscape was convex with 10-15% slope. Two pits were excavated up-gradient from where the system was to be installed and samples were collected for soil characterization. Characterization of soil samples included particle size distribution (PSD), cation exchange capacity (CEC), extractable base analysis, and bulk density (ρ_b). Methods used for soil characterization can be found in USDA (1996). Hydraulic loading rate was estimated by measuring the field saturated hydraulic conductivity (K_s) of boreholes using Compact Constant Head Permeameters in the subsoil horizons.

Suction lysimeters (Soil Moisture Equipment Corp., Santa Barbara, CA) were installed beneath and adjacent to each trench at 3.3 m and 6.6 m from where effluent entered. Figure 1 shows the location of lysimeters relative to the trench bottoms. Samples were collected from the lysimeters bi-weekly for the first 2 months of the study and monthly thereafter. Wastewater samples were also collected from the dosing tank and septic tank (inlet and outlet). The wastewater and soil water samples were analyzed for pH, Cl (Labconco Chloridometer, Kansas City, MO), $\text{NO}_3\text{-N}$, $\text{NH}_4\text{-N}$, and Total Kjeldahl Nitrogen (TKN). The raw wastewater samples were also analyzed for 5-day biochemical oxygen demand (BOD_5) and total suspended solids (TSS). Methods for BOD_5 , TSS, and N analysis can be found in Eaton and Franson (2005). Means for lysimeter depths in the drainfield and ANOVA for the dose and septic tank samples were calculated using

the PROC MEANS and PROC GLM procedures, respectively, in SAS (SAS Inc, Cary, NC).

Tensiometers (Decagon Devices, Pullman, WA) were installed beneath trenches at 10 and 15 cm, and down-slope of trenches at 1 cm above and 15 cm below trench bottoms. CR1000 data loggers (Campbell Scientific, Logan, UT) were used to continuously record pressure heads. Tensiometers were also clustered 3.3 and 6.6 m from where effluent entered the trenches.

Wastewater was supplied by the Cabin Creek Wastewater Treatment Plant in Griffin, GA. Wastewater was collected from the inlet of the wastewater treatment plant prior to entering the plant and transported by truck to the research site. The storage tank was filled such that there was a maximum of 3 days storage before the wastewater was dosed to the septic tank.

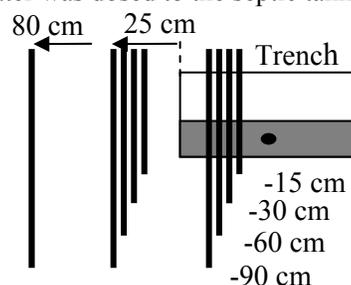


Figure 1. Drawing of lysimeter locations relative to trench bottoms.

The septic tank was dosed every 8 h. Based on the soil classification and K_s measurements, we determined a hydraulic loading rate of 2.5 cm d^{-1} (GADCH, 2007). The drainfield was dosed at 2 times the hydraulic loading rate for 2 months to accelerate biomat development. Wastewater application began 9 April 2009. For the first several months of wastewater application a time dosed method was used to dose the system (i.e. we opened the valve for approximately 50-s to apply the dose). We determined that the waste application rate was too variable with this method due to differences in the hydraulic head in the dosing tank over time and modified the dosing method by installing an ultrasonic sensor (Senix Corp., Bristol, VT) to measure the depth of wastewater in the dosing tank and to accurately and consistently dose the system.

RESULTS & DISCUSSION

The drainfield was installed in a typical piedmont Ultisol. Ultisols are characterized by low cation exchange capacity (CEC) and low base saturation. Table 1 shows the CEC and base saturation for an excavated pit adjacent to where the drainfield was installed. Base saturation ranged from 8.6 % to 21.4% in the subsoil and decreased

with depth. The major diagnostic features for classifying

Ultisols are CEC <16 cmol kg⁻¹ and base saturation

Table 1. Site soil characteristics.

Horizon	Depth (cm)	Texture [‡]	CEC (cmol kg ⁻¹)	Base Sat. (%)	K_s (cm hr ⁻¹)	ρ_b (g cm ⁻³)	K_{d2} (cm ³ g ⁻¹)	R
Ap	0-10	scl	10.1	30.6	†	†	†	†
Bt1	10-58	c	9.70	21.4	1.4	1.59	0.73	4.1
Bt2	58-83	sc	9.40	10.5	1.1	1.63	1.1	5.4
BC	83-145+	scl	10.9	8.60	0.1	1.85	0.89	5.3

†not measured; ‡ sandy clay loam (scl), clay (c), sandy clay (sc)

<35% in the upper boundary of an argillic or kandic horizon (Buol et al., 1997).

Residential strength wastewater, not requiring pre-treatment, was used in this study to mimic typical household waste. The wastewater is characterized by having BOD₅ < 200 mg L⁻¹ and TSS <200 mg L⁻¹ (GADCH, 2007). Table 2 shows mean wastewater characteristics in the dosing and septic tanks. In the dosing tank, mean TSS and BOD₅ were 89 and 154 mg L⁻¹, respectively. These data show the wastewater used in the study met the requirements for residential strength wastewater, as defined by the State of Georgia.

Table 2. Wastewater characteristics. †Values with different letters are significantly different (p<0.05).

Sample	TSS [†] (mg L ⁻¹)	BOD ₅ (mg L ⁻¹)	NH ₄ -N (mg L ⁻¹)	NO ₃ -N (mg L ⁻¹)	Total N (mg L ⁻¹)
Dose Tank	89 a	154 a	38 a	0.02 a	60 a
Inlet	44 b	115 ab	33 a	0.03 a	39 a
Outlet	52 b	105 b	33 a	0.02 a	38 a

BOD₅ was significantly lower in the septic tank outlet than in the dose tank (Table 2). TSS concentrations were also significantly different between the dose tank and the septic tank. These data indicated that primary treatment occurred in the dose tank, as well as in the septic tank. Mean TSS and BOD₅ concentrations for the septic tank outlet were 52 and 105 mg L⁻¹, respectively. These values represent the average TSS and BOD₅ concentration dosed to the drainfield trenches. In properly functioning OWSs, TSS is removed from effluent by the first few centimeters of soil through filtration at the infiltrative surface. Colloidal and dissolved BOD are removed through aerobic biological processes in the vadose zone where oxidation occurs (USEPA, 2002). Since TSS and BOD are almost completely removed, they are not considered contaminants of concern in properly functioning OWSs.

N in the dose and septic tanks was mostly in the form of NH₄-N and organic-N due to anaerobic conditions (Table 2). Mean NH₄-N concentrations ranged

from 33-38 mg L⁻¹ in the septic and dose tanks, respectively, and were not significantly different. NO₃-N concentrations were near the detection limit (0.02 mg L⁻¹) and also not significantly different. Mean wastewater N concentrations in this study were consistent with values found in the literature (Androli et al., 1979; Cogger and Carlile, 1984; Reneau, 1977; Siegrist, 1987).

Positive pressure heads from tensiometers installed 10 and 15 cm below trench bottoms showed that the soil remained saturated most of the time in that region (Figure 2a-d). Nitrification would be inhibited and denitrification favored in this region of the drainfield because of saturated conditions. Pressure heads down-slope of the trenches varied over a larger range and were negative more often (Figure 2e-h). Nitrification would be favored in the down-slope region where large soil pores are unsaturated. Denitrification may also occur in soil aggregates containing small pores that do not drain as rapidly as larger pores.

In an OWS, NH₄-N has several potential cycling pathways. In the first pathway, the positively charged NH₄⁺ ion may be bound to negatively charged cation exchange sites on clay particles in the soil. Piedmont soils exhibit a net negative charge due to isomorphous substitution in tetrahedral and octahedral clay layers, as well as pH dependent charge associated with iron and aluminum oxides. Cation exchange capacities are low in Piedmont soils due to low pH, however, they still act as a sink for NH₄⁺. Biomats in drainfield trenches may also adsorb NH₄⁺ because organic matter typically has a large CEC. In the second cycling pathway, NH₄-N in solution will undergo nitrification where aerobic conditions and nitrifying microbes are present in the soil matrix. Nitrifying microbes require an inorganic carbon source to build cell mass, which is provided in the wastewater effluent (McCray et al. 2008). Microbial immobilization of NH₄-N may also occur. Lastly, NH₄-N may be taken up by plants.

Mean NH₄-N concentrations at the different sampling depths below the OWS drainfield verses time are shown in Figure 3. The concentrations of NH₄-N were near zero at all depths for the first 4 months following

initiation of wastewater dosing. However, $\text{NH}_4\text{-N}$ concentrations began to increase at 15 and 30 cm below the drainfield in September, 2009 and increased monthly thereafter.

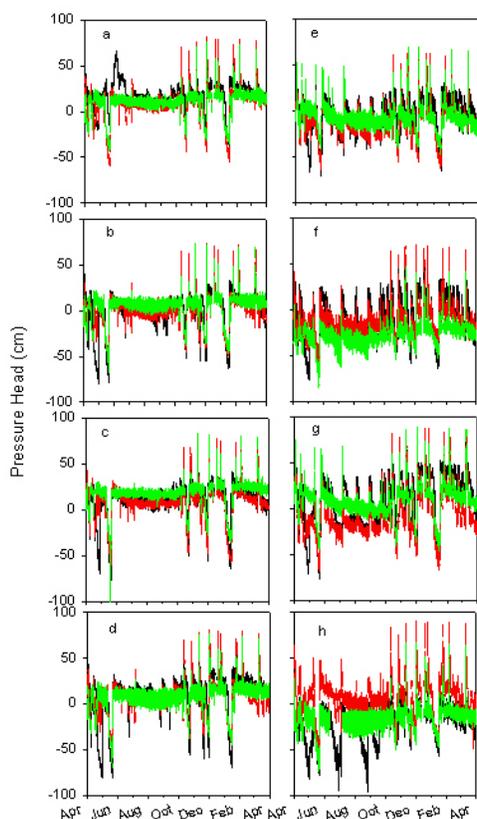


Figure 2. Drainfield tensiometer pressure heads from April 2009 to April 2010. Beneath trenches: (a) 10 cm—3.3 m from inlet, (b) 10 cm—6.6 m from inlet, (c) 15 cm—3.3 m from inlet, (d) 15 cm—6.6 m from inlet; 10 down-slope of trench sidewall: (e) 1 cm above trench bottom—3.3 m from inlet, (f) 1 cm above trench bottom—6.6 m from inlet, (g) 15 cm below trench—3.3 m from inlet, and (h) 15 cm below trench—6.6 m from inlet. Trench 1 (black line), Trench 2 (red line), & Trench 3 (green line).

After the initial 4 months, mean $\text{NH}_4\text{-N}$ concentrations varied between 4 and 8 mg L^{-1} 15 cm below the trenches. Concentrations stayed below 1.0 mg L^{-1} at the 60 and 90 cm depths. The relatively constant and high concentrations at 15 cm below the drainfield indicated that equilibrium had been reached between adsorbed and solution $\text{NH}_4\text{-N}$ and conversion to $\text{NO}_3\text{-N}$. At 30 cm, the increasing concentrations indicated that the maximum adsorption capacity had not been reached. Most N that reached 60 and 90 cm below the trenches was in the form of $\text{NO}_3\text{-N}$.

As with $\text{NH}_4\text{-N}$, there are several potential cycling pathways for $\text{NO}_3\text{-N}$. One cycling mechanism for the

NO_3^- ion is electrostatic binding by anion adsorption, characteristic of acidic soils. A study by McVay et al (2004) showed the average anion exchange capacity of 7 subsoil samples from the Georgia Piedmont was 1.98 cmol kg^{-1} . The authors calculated the potential NO_3^- adsorption from a 1 m thick subsoil be 9.2 Mg ha^{-1} with an estimated travel time through subsoil and saprolite 4x that of water. We calculated retardation coefficients (R) using distribution coefficients (K_{d2}) from Cl adsorption isotherms on three subsoil horizons at the Griffin site. R values indicated that travel times for monovalent anions (i.e. Cl^- and NO_3^-) may be 4.1 to 5.4 times slower than that of water (Table 1).

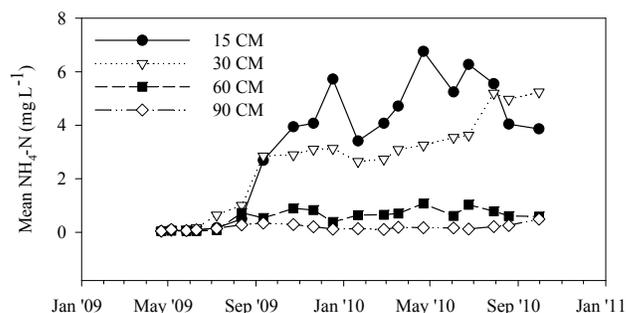


Figure 3. Mean $\text{NH}_4\text{-N}$ concentrations at 15, 30, 60, and 90 cm below drainfield trenches.

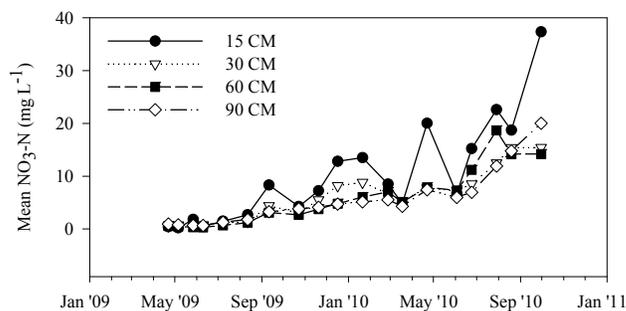


Figure 4. Mean $\text{NO}_3\text{-N}$ concentrations at 15, 30, 60, and 90 cm below drainfield trenches.

Leaching of $\text{NO}_3\text{-N}$ to groundwater may also occur. Another possible cycling mechanism for $\text{NO}_3\text{-N}$ is denitrification. Favorable conditions for denitrification include the presence of $\text{NO}_3\text{-N}$, anoxic (i.e. saturated) soil, and carbon (C) to act as an energy source for the reduction reaction. OWSs undergo constant wetting and drying cycles due to dosing and drainage of wastewater, thus conditions favorable for denitrification can occur throughout the drainfield. Root uptake of $\text{NO}_3\text{-N}$ may also occur if drainfield trenches are installed near the root zone.

Mean $\text{NO}_3\text{-N}$ concentrations at the different sampling depths below the OWS drainfield verses time are shown in Figure 4. Mean $\text{NO}_3\text{-N}$ concentrations were near 1 mg L^{-1} at all depths and remained relatively con-

stant for the first 4 months following initiation of wastewater dosing. $\text{NO}_3\text{-N}$ concentrations began to increase at all depths in the drainfield in September, 2009 and increased linearly with time thereafter. The $\text{NO}_3\text{-N}$ concentration was 10 mg L^{-1} at 90 cm in September, 2010. This concentration of $\text{NO}_3\text{-N}$ would probably be leached to groundwater as denitrification may not occur beyond 1 m below the drainfield. Increased nitrogen loading to groundwater can be detrimental to water quality because groundwater flow to streams can carry $\text{NO}_3\text{-N}$ that will stimulate eutrophication in lakes and reservoirs.

It appeared that most of the N in the OWS was cycled to $\text{NO}_3\text{-N}$ as the system matured because the mean wastewater $\text{NH}_4\text{-N}$ concentration decreased from 38 mg L^{-1} $\text{NH}_4\text{-N}$ to 20 mg L^{-1} $\text{NO}_3\text{-N}$, 90 cm below the drainfield. $\text{NO}_3\text{-N}$ concentrations increased in the 15 cm zone below the drainfield indicating that the soil was unsaturated zone part of the time or that nitrification may have occurred in the trenches as effluent trickled through the gravel bed. Denitrification and dilution are probably the mechanisms responsible for N that was removed between 15 and 90 cm.

In the first 4 months following the initiation of wastewater dosing there was not much change in $\text{NH}_4\text{-N}$ or $\text{NO}_3\text{-N}$. There are several possible explanations for the initial lag in N cycling. Cation and anion adsorption may have played an important role in removing NH_4^+ and NO_3^- until exchange sites became saturated. Microbial populations necessary for N cycling are generally present in most soils however, there may have been an initial microbial growth period following the start of wastewater applications where most $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ would have been assimilated. Finally denitrification may have occurred under saturated conditions, removing most of the N from the drainfield. There is a strong likelihood that these processes occurred simultaneously.

Cl is a conservative tracer in soils and can be used to detect the occurrence of denitrification because it does not undergo transformations due to microbial activity. Nitrogen-to-chloride ratios (N/Cl), in $\text{mg N mg}^{-1} \text{ Cl}$, were calculated using Equation 1

$$\frac{N}{Cl} = \frac{\text{Total}_N + \left(\text{NH}_4\text{-N} * \frac{\rho_b}{\theta_s} * K_{D1} \right) + \left(\text{NO}_3\text{-N} * \frac{\rho_b}{\theta_s} * K_{D2} \right)}{\text{Cl} + \left(\text{Cl} * \frac{\rho_b}{\theta_s} * K_{D2} \right)}$$

Equation 1. N/Cl ratio.

where: Total_N , $\text{NH}_4\text{-N}$, and $\text{NO}_3\text{-N}$ (mg L^{-1}) are the total-, ammonium-, and nitrate-N concentrations in the soil solution, respectively, ρ_b (g cm^{-3}) and θ_s are the bulk density and the saturated water content of the soil horizon where the lysimeter was installed, respectively, K_{D1} and K_{D2} ($\text{cm}^3 \text{ g}^{-1}$) are the adsorption coefficients for NH_4^+

and Cl^- , respectively. The value for K_{D1} was $1.32 \text{ cm}^3 \text{ g}^{-1}$ in a similar soil (Johson et al., 1999). K_{D2} was also used for NO_3^- assuming monovalent anion adsorption was similar.

Figure 5 shows the mean N/Cl ratio versus time for wastewater effluent and at 15, 30, 60, and 90 cm below drainfield trenches. Mean N/Cl ratios ranged from 0.8 to $1.6 \text{ mg N mg}^{-1} \text{ Cl}$ in the effluent and decreased to $0.3 \text{ mg N mg}^{-1} \text{ Cl}$ at 60 and 90 cm. Mean N/Cl ratios at 15 cm varied from 0 to $1.5 \text{ mg N mg}^{-1} \text{ Cl}$ in the first year of OWS operation. Low N/Cl ratios at deeper depths (i.e. 60 and 90 cm) indicated most of the N was lost via denitrification and not dilution. If dilution was the mechanism for the decreased mass of N, then N/Cl ratios would have remained near $1 \text{ mg N mg}^{-1} \text{ Cl}$ because the mass of Cl would have decreased equivalently. Assuming an average N/Cl ratio of $1 \text{ mg N mg}^{-1} \text{ Cl}$ for the wastewater effluent and an average ratio of $0.3 \text{ mg N mg}^{-1} \text{ Cl}$ at 90 cm below the drainfield, we estimated that denitrification may account for up to 70% of the total N lost in the drainfield above 90 cm.

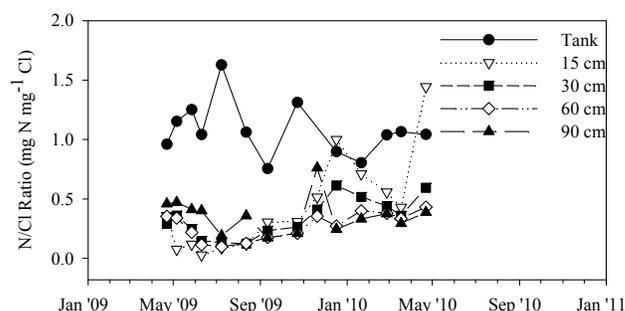


Figure 5. Mean N/Cl ratios in wastewater effluent and at 15, 30, 60, and 90 cm below drainfield trenches.

Currently, a more complex HYDRUS model of the N dynamics is being calibrated using data from our experimental site. Once the model is calibrated, it will be used to estimate denitrification losses by OWSs in other regions by varying the soils and climate data. Estimates of the N load from OWS in other regions can then be determined.

CONCLUSIONS

Saturated conditions dominated the soil matrix directly beneath the trenches while unsaturated conditions dominated the soil matrix as depth and distance from trench bottoms increased. Pressure heads indicated favorable soil water conditions for both nitrification and denitrification to occur in the first 15 cm below the infiltrative surface.

The N contribution from OWSs in nutrient impaired watersheds is difficult to determine because there are no field studies that have measured the loss due to denitrifi-

cation. After an initial lag period of 4 months, $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ concentrations increased for the next 11 months in our OWS field experiment. Almost all $\text{NH}_4\text{-N}$ was nitrified in the drainfield and concentrations of $\text{NO}_3\text{-N}$ reached 20 mg L^{-1} at the deepest depth after 15 months. Using N/Cl ratios we measured N reductions from approximately $1.0 \text{ mg N mg}^{-1} \text{ Cl}$ in wastewater effluent to approximately $0.3 \text{ mg N mg}^{-1} \text{ Cl}$ at 90 cm below the drainfield. This data suggests that up to 70% of N may be lost through denitrification.

Anion adsorption may have been an important renovation mechanism in the subsoil and underlying saprolite. Retardation of anions by saprolite was reported to be 4.4 by McVay et al. (2004) and we calculated an average R of 4.9 for the drainfield subsoil, meaning $\text{NO}_3\text{-N}$ may travel through soil and saprolite greater than 4 to 5 times slower than water. Thus, there is a strong likelihood that $\text{NO}_3\text{-N}$ leaching beyond piedmont drainfields will be retained by thick layers of saprolite underlying OWSs. However, $\text{NO}_3\text{-N}$ may contaminate groundwater in areas with shallow water tables.

Combining N and Cl data with accurate water fluxes from models, we can obtain better estimates of N losses and therefore more accurate estimates of N loading from OWSs. Using a complex model of the N dynamics calibrated to our experimental site, estimates of denitrification losses by OWSs in other regions can be determined and used to assist in TMDL assessments of nutrient impaired watersheds.

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