A FARM-SCALE TEST OF NITROGEN ASSIMILATION BY VEGETATED BUFFER SYSTEMS RECEIVING SWINE LAGOON EFFLUENT BY OVERLAND FLOW

R. K. Hubbard, G. L. Newton, J. M. Ruter

ABSTRACT. *A farm-scale study was conducted from 2000 to 2004 to determine the effectiveness of grass-forest vegetated buffers in assimilating nitrogen (N) from overland flow application of swine lagoon effluent. The rationale for the study was that replicated buffer plot studies had shown that vegetated buffers will effectively assimilate N, but it was not known whether or not they would work at a larger scale. The study was conducted on a commercial farm near Tifton, Georgia. Wastewater was pumped from a single-stage anaerobic lagoon to vegetated buffers composed of grass and mature or newly planted pines. The buffers approximated 60 m in length by 90 m in width. The upper 10 m of each buffer was in grass, while the downslope area was in mature or newly planted pines. Six buffers were instrumented for wastewater application and water quality monitoring. Two buffers received wastewater at a 1X rate (600 kg N ha^{-1} year^{-1}), two at a 3/4X rate (450 kg N ha^{-1} year^{-1}), and two served as controls. The wastewater was applied to the 10 m grassed portion of the buffers. Transects of shallow groundwater wells starting at the grass-forest interface and running downslope were used to monitor water quality N. The study showed mixed results concerning N assimilation by the buffers. Upslope land use changes by the producer during the study added significant N inputs to one set of buffers, and they were unable to assimilate sufficient N from both these inputs and the wastewater to protect shallow groundwater quality. In contrast, almost all samplings of shallow groundwater under the buffers receiving N only from the overland flow applied swine lagoon effluent showed nitrate (NO_3−N) concentrations 20 and 30 m downslope to be lower than 10 mg L^{-1} (drinking water standard). On these buffers, NO_3−N concentrations in shallow groundwater were near background levels five years after wastewater application commenced. The study indicated that the ratio of buffer area width to wastewater application area width on the landscape should be at least 1:1, and that buffers for protection of water quality should be continuous on the landscape. It was concluded from the study that buffers can be used at the farm scale to assimilate N from applied wastewater when they are sufficiently wide relative to waste application area, rate, and other N sources at the farm scale.*

Keywords. *Animal waste, Nitrogen, Vegetated buffers, Water quality.*

Pollution of surface and ground waters from animal wastes is of growing environmental concern. High nitrogen (N) loading rates to soils and waters can be associated with intensive animal operations. Concentrations of N in excess of 10 mg L^{-1} in the nitrate (NO_3−N) form render groundwater unsuitable as drinking water for humans (Alexander, 1972). In some states, approximately 10% of private rural wells tested exceeded the USEPA national drinking water standard of 10 mg L^{-1} for NO_3−N (Porter, 2001). A considerable portion of water quality degradation is directly attributed to agricultural land use, with N over-application with animal manures posing significant threats (Jackson-Smith et al., 2001). High N concentrations from animal wastes entering streams or lakes contribute to eutrophication (Hubbard et al., 2004). Eutrophication remains the most pervasive water quality problem in freshwater ecosystems (Carpenter et al., 1998).

A number of researchers have investigated use of vegetated filter strips (VFS) for treatment of animal wastes. Doyle et al. (1977) applied 850 kg N ha^{-1} year^{-1} of dairy manure to forest or grass buffers and found that 3.8 m of forest buffer or 4.0 m of grass buffer were useful in reducing N content of manure-polluted runoff. Thompson (1977) used a 24 m long waste area that received approximately 600 kg N ha^{-1} year^{-1} of dairy manure and found significant reductions in N concentrations with distance downslope for buffer strip lengths of 12.2 and 36.6 m. Edwards et al. (1981), Dickey and Vanderholm (1981), and Swanson et al. (1975) found that the combination of a 500 m heavily grassed waterway, vegetative filters, and a serpentine waterway permitted highly polluted initial runoff from barnlots and feedlots to be infiltrated into the soil and diluted by runoff from outside areas. Woodbury et al. (2002) investigated use of VFS to reduce or eliminate

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long-term liquid storage from feedlot runoff. A flat-bottom terrace was constructed to collect runoff, provide temporary liquid storage, and accumulate settleable solids, while distributing the nutrient-laden liquid fraction across a VFS. They concluded that their passive beef cattle feedlot runoff system was an improvement to traditional storage systems. Prantner et al. (2001) tested a soil infiltration and wetland system in series for removal of N and found that approximately 93% of the ammoniacal N (NH_3–N and NH_4–N) was removed by the soil infiltration area, with a corresponding 99% increase in NO_3–N concentrations. Their wetland system removed 94% of the remaining NH_3–N and NH_4–N and 95% of the NO_3–N.

Dillaha et al. (1987) found that 4.6 m filters removed 61% and 9.1 m filters removed 72% of the incoming N in runoff from areas to which dairy manure was applied. Carlson et al. (1974) used a 1.5 m wide × 6.0 m long constructed model with grass sod to evaluate an overland flow system for secondary effluent. Analyses showed that most of the NH_4–N (100%), NO_3–N (95%), and organic N (91%) were retained in the model. Fajardo et al. (2001) tested VFS of tall fescue with wastewater received from overland flow over manure stockpiles in the headland. They determined that the VFS reduced concentrations of NO_3–N in surface runoff by 97% to 99%. They concluded that dilution and residence time of water passing through the VFS were the most significant factors affecting reductions in NO_3–N.

Chaubey et al. (1994) used simulated rainfall to evaluate the effectiveness of VFS in reducing N losses from plots treated with liquid swine manure at 200 kg N ha^{-1}. Mass transport of NH_4–N and total Kjeldahl nitrogen (TKN) was reduced significantly ($p < 0.05$) by fescue (*Festuca arundinacea* Schreb.). Vegetated filter strips 3 m in length removed 65% of the incoming TKN and 71% of the incoming NH_4–N, while those 21 m in length removed 87% of the incoming TKN and 99% of the incoming NH_4–N. Mass transport of TKN and NH_4–N was minimized at 9 m VFS length. The VFS did not significantly reduce NO_3–N from the incoming runoff. Hawkins et al. (1998) used VFS on slopes of 5% and 11% as a means of treating wastewater effluent from the second cell of a swine waste anaerobic lagoon system. Their mass balance estimates suggested that VFS can be excellent treatment systems for liquid lagoon effluents, with mass reductions of N greater than 93% on an 11% slope and 60% on a 5% slope. However, they concluded that NO_3–N leaching could be a concern with extended long-term use of such systems.

Chaubey et al. (1995) also used simulated rainfall to evaluate the effectiveness of VFS in reducing N losses from plots amended with poultry litter (5 Mg ha^{-1}). Vegetated filter strips of 3.1 m reduced mass transport of TKN by 39% and NH_4–N by 47%, while 21.4 m VFS reduced TKN by 81% and NH_4–N by 98%. As with their 1994 study with liquid swine manure, they found that the VFS were ineffective in removing NO_3–N from the incoming runoff from poultry litter. Srivastava et al. (1996) assessed the effects of pollutant source area (fescue pasture treated with poultry litter) length and VFS (fescue pasture) length on VFS removal of NO_3–N, NH_3–N, and TKN using a rainfall simulator. They found that runoff mass transport of NH_3–N and TKN increased with increasing litter-treated length (due to increased runoff) and decreased (approximately first-order exponential decline) with increasing VFS length. Bingham et al. (1980) evaluated pollutant concentrations in runoff at various distances downslope from an area where caged-layer poultry manure was applied regularly, and found that a buffer-area-length to waste-application-area-length ratio of 1.0 was usually required to reduce concentrations to those measured in runoff from a similar plot receiving no manure.

Limited information exists on the effectiveness of grass-forest vegetated buffer systems on lower landscape positions for assimilation of N from animal wastewater entering via direct application, surface runoff, or shallow subsurface flow. The N assimilation concepts for such systems are that N can be removed from the wastewater by both vegetative uptake and denitrification. Nitrogen taken up by vegetation is then removed from the buffer system by cutting the grassed zone for hay and by removal of the trees as harvestable timber. Denitrification is a microbial process requiring NO_3–N, a carbon (C) source, and anaerobic conditions. In these vegetated buffer systems, N and C are supplied by the wastewater, while anaerobic conditions occur due to both soil wetness (lower landscape position) and consumption of oxygen (O_2) by decomposing organic matter.

Hubbard et al. (1998) conducted a three-year study (1993-1996) to determine the feasibility of using grass-forest vegetated buffer systems to assimilate N from swine lagoon effluent. Wastewater from the third lagoon of the University of Georgia Coastal Plain Experiment Station swine research unit at Tifton, Georgia, was applied to replicated buffer plots by overland flow from tanks at the top ends of each plot. The wastewater, which contained an average N concentration of 160 mg L^{-1}, primarily as NH_4–N, was applied to the plots at two different rates: 1X (800 kg N ha^{-1} year^{-1}), or 2X (1600 kg N ha^{-1} year^{-1}). The study tested three different vegetated buffer treatments: 10 m grass buffer draining into 20 m existing riparian zone vegetation, 20 m grass buffer draining into 10 m existing riparian zone vegetation, and 10 m grass buffer draining into 20 m maidencane (*Panicum hematomon* Schult. 'Halifax'). The effects of the wastewater on surface runoff and groundwater quality were evaluated by transects of surface runoff collectors, suction lysimeters, and shallow groundwater wells, which extended from the top to the bottom of each plot. Results from the study showed water quality differences due to wastewater application rate and distance downslope from the wastewater application pipe. Nitrate concentrations in shallow groundwater at the top ends of the plots increased over time, but little or no increase was observed at the bottom ends of the plots.

Long-term effectiveness of these buffer systems in assimilating N from swine lagoon wastewater was determined by continued wastewater application and shallow groundwater sampling for a total of eight years (Hubbard et al., 2004). Results from 1997-2000 showed increasing NO_3–N concentrations in shallow groundwater at the upper ends of the buffers (i.e., where the wastewater was applied most heavily), but little or no increase in NO_3–N concentrations at the lower ends of the buffers. In fact, NO_3–N concentrations in shallow groundwater at the bottom end of the buffers in 2000 were about the same as the concentrations observed at the beginning of the study in 1993. As part of this same study, Hubbard et al. (2003) determined N removal by the grassed portion of the grass-forest buffers. Results showed that the N content of the grass at the 1X rate was less than that at the 2X rate. At the 1X rate, grass buffers 20 m in width removed 44% of the N as biomass. Nitrogen removal via uptake (percentage of applied) decreased by 62.5% when wastewater was applied at the 2X rate. Overall, the grass N uptake part of the study showed that while uptake into the grass biomass accounted for a portion of the N removal in the grass-forest buffer systems, the N concentrations in surface runoff and subsurface water exiting these systems indicated that other factors (denitrification, forest uptake, and adsorption) played a greater role in the N assimilation and filtering by the buffers. The study indicated that whether for animal wastes or other sources of N, grass buffers alone will not meet the needs of protecting environmental quality in this region unless the buffers are quite wide.

In another subpart of this buffer study, Lowrance and Hubbard (2001) quantified the denitrification rate and the changes in N pools over time. Denitrification and soil N pools were determined bimonthly for three years (1993-1996). It was found that denitrification rates and soil NO_3–N were greater under the 2X wastewater application rate than under the 1X rate. The soil surface depth (0-6 cm) had greater denitrification, NO_3–N, and NH_4–N than the 6-12 cm soil depth. For all buffer treatments, denitrification did not vary significantly with position in the plot (7, 14, 21, or 28 m downslope), but NO_3–N decreased in the downslope direction, while NH_4–N increased downslope. Results from the study indicated that about 200 kg N ha^{-1} year^{-1} may be the maximum denitrification rate possible for liquid manure application systems on Coastal Plain soils where most of the N in the effluent is NH_4–N.

The replicated buffer plot studies of the early 1990s through 2000 indicated that overland flow wastewater application to lower-landscape position grass-forest buffers may serve as a useful technology for wastewater management by animal producers. However, the study was conducted with wastewater from the third lagoon of the University of Georgia swine research facility, and N concentrations were lower than what is commonly found in the single anaerobic lagoons of commercial hog producers. In addition, the buffer plot study was conducted under relatively ideal conditions with carefully controlled wastewater application and plot management. This does not match what generally occurs on commercial farms, where wastewater applications are less intensively managed or monitored. The objective of this study was to determine the effectiveness of farm-scale grass-forest buffers in assimilating N from swine lagoon wastewater applied by overland flow. Effectiveness was measured by monitoring water quality under the buffers over time. Grass-forest buffers were selected (as opposed to other wetland vegetation) since grass upslope of pine forest commonly occurs in the Coastal Plain region of the southeastern U.S. and is easy for land owners to manage.

Materials and Methods

Site Description

The study was conducted from 2000 to 2004 on a commercial hog farm located near Tifton, Georgia. The vegetated buffer site on the farm was selected during 1998. The selection criteria included landscape position, soils, slope, vegetation, and land availability, i.e., the willingness of the land owner/farm manager to allow wastewater to be applied to the specified portion of his land. Our criteria were: lower landscape position with accompanying soils, gentle slope (1% to 2%), and some forest vegetation at least 10 years old. The soils at the selected site were Tifton loamy sand (fine-loamy, siliceous, thermic, Plinthic Kandiudults) grading into Grady sandy loam (fine, kaolinitic, thermic Typic

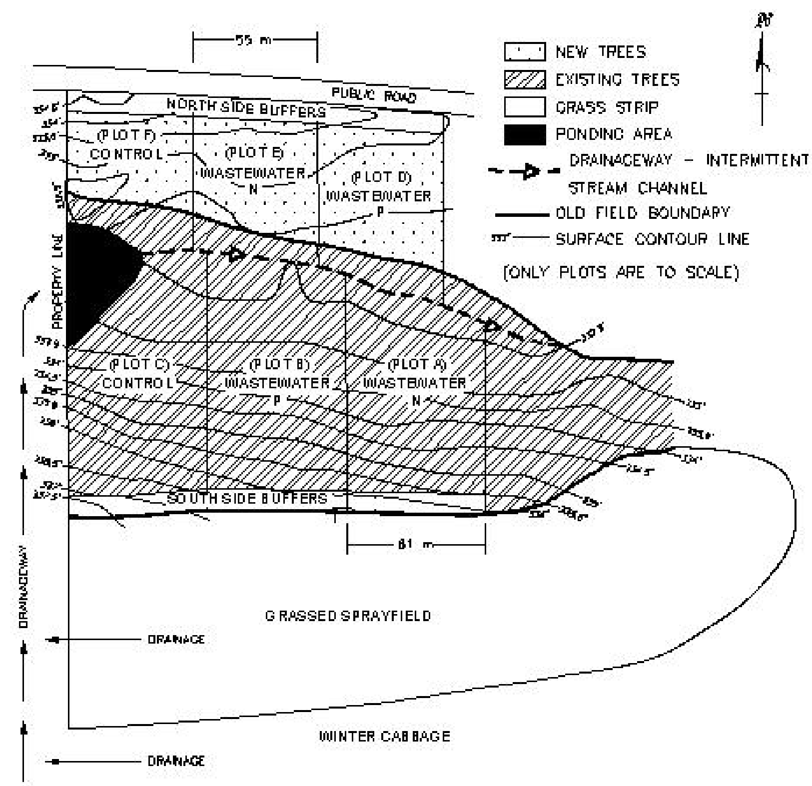

Figure 1. Diagram showing overland flow buffers.

Paleaquults). The site selected and agreed upon included a natural drainageway-intermittent stream channel through the center with mature slash pine trees (*Pinus elliotti* L. var. *elliottis* Engelm) on the south side and land cropped to cotton (*Gossypium hirsutum* L.) on the north side (fig. 1). Although a site comprised entirely of mature pine (except for the grass buffer) would have simplified the experimental design, such a site did not exist on the producer's land. We requested that the land cropped to cotton be planted to pines, and the farm manager agreed, since the area of land involved was not large. In addition, previous research (Hubbard and Lowrance, 1997) showed that mature pines, selectively thinned pines, and newly planted pines are all effective in assimilating N.

Experimental Design

The study was designed to compare the environmental impacts on shallow groundwater quality of swine lagoon wastewater applied to grass-forest buffers at different rates. Six large buffer areas were selected: two serving as controls, two receiving swine lagoon wastewater based on N content (1X rate), and two receiving wastewater based on P content (3/4X rate) (fig. 1). The target N application rate for our 1X buffers was 800 kg N ha^{-1} $year^{-1}$, our lower N rate from the 1993-1996 study (Hubbard et al., 1998; Hubbard et al., 2003). This rate was comparable to N commonly applied to agronomic triple cropping systems in the Georgia Coastal Plain. The target P application rate for our 3/4X buffers was 100 kg P ha^{-1} $year^{-1}$ (P results from the study will be reported in a separate article). The producer decreased swine numbers early in the study, which resulted in the actual 1X rate being about 600 kg N ha^{-1} $year^{-1}$ and the 3/4X rate being about 450 kg N ha^{-1} $year^{-1}$.

There were three buffer areas on each side of the natural drainageway-intermittent stream channel (fig. 1). A 10 m buffer of coastal bermudagrass (*Cynodon dactylon* L. Pers 'Tifton 85') was planted upslope from the mature forest on the south side, and upslope from the "cotton land" on the north side during spring 1998. Young slash pines were planted on the "cotton land" during summer 1998. Each south side buffer (grass plus mature forest) was 61 m in length (parallel to the drainageway-intermittent stream channel) and 98 to 133 m in width (perpendicular to the drainageway-intermittent stream channel) (fig. 1). Each north side buffer (grass plus planted pines and some existing mature pines) was 55 m in length and an average of 83 m in width. The buffer sizes hence were somewhat uneven but represented real-world conditions, since buffers as used at the farm scale will be on areas of available land most often of uneven sizes, as opposed to the same-sized areas commonly used for replicated scientific studies.

Wastewater Application

Swine lagoon wastewater was first applied to the buffers on 6 March 2000 (fig. 2). Wastewater was then applied to the plots on a weekly schedule until 31 December 2004. The wastewater was pumped approximately 1.6 km from the single anaerobic lagoon to the vegetated buffer plots using a Paco Type L pump (18 cm impeller, 10 hp motor, Paco Pumps, Brookshire, Texas; www.paco-pumps.com) located in the lagoon. The wastewater then exited from application pipes (10.2 cm dia. aluminum irrigation pipe with 0.64 cm dia. holes with rubber grommets drilled 48.3 cm apart), which were located immediately upslope of the grassed portions of the buffers. Individual in-line water meters (Neptune T-10, Neptune Equipment Co., Cincinnati, Ohio; http://neptuneequipment.com) were used to measure the volume of wastewater applied to each buffer. The shape of the landscape was such that during each application almost all of the wastewater was applied only to the grassed portions of the buffers. Some exceptions to the weekly wastewater application schedule occurred when equipment malfunctioned (broken pump, pump would not prime, clogged inlet pipe, broken water meters, etc.) or when the producer needed to use the pipeline for applying wastewater to his grassed sprayfield, which was installed on the farm during spring-summer 2002 (fig. 2). The producer maintained the lagoon at a level where it would not overflow and contaminate the drainage–

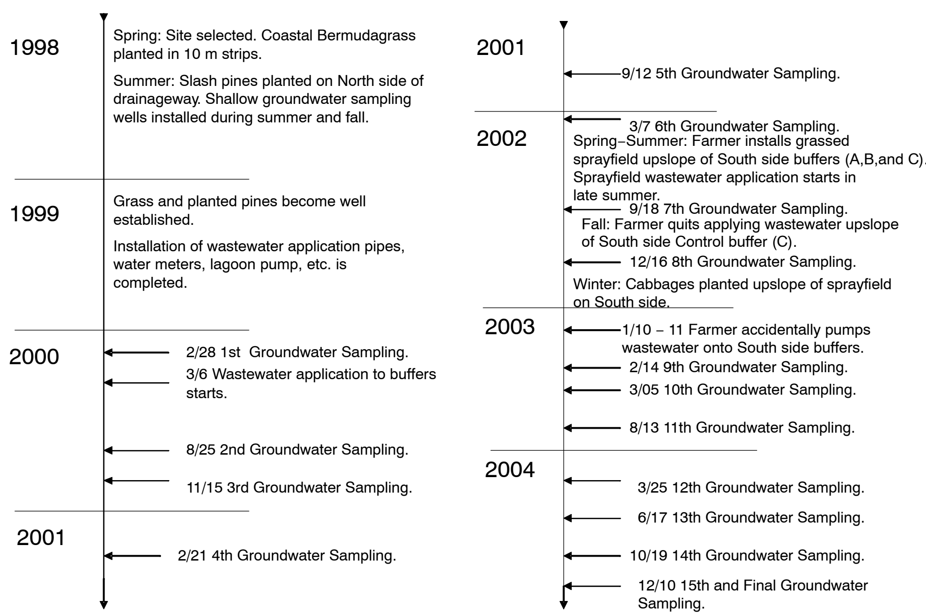

Figure 2. Timeline.

way adjacent to his lagoon or other lands, and at times, particularly when there was high rainfall, this required him to pump a considerable volume of wastewater onto his grassed sprayfield. There was some inaccuracy in wastewater application during March 2001, when the flow meters malfunctioned for several weeks. In addition, an extended period with no wastewater application occurred from 6 August through 13 October 2004 due to electrical problems in the lagoon pump breaker box.

The application rates were 6804 and 9072 L buffer^{-1} (3/4X or 1X rates) of wastewater averaging 225 mg L^{-1} total N. Ninety to 95% of the N was in the NH_4–N form. Actual annual N rates for the study were 450 and 600 kg N ha^{-1} year^{-1} for the 3/4X and 1X rates, respectively. The mean Cl concentration of the wastewater was 125 mg L^{-1}. Buffers B and D received the 3/4X rate, while buffers A and E received the 1X rate (fig. 1). Two other buffers (C and F) served as controls and did not receive direct application of wastewater by overland flow.

Several farm management events/decisions occurred during the study that were not part of the original experimental design, but which impacted results (fig. 2). During spring/summer 2002, the farm manager installed a grassed sprayfield (fig. 1) for application of his lagoon wastewater directly upslope of our south side buffers (A, B, and C). This land management decision was made without our input, and we were not made aware of the decision until it was implemented. We would have very strongly encouraged the farm manager to choose an alternate site so that there would be no potential for additional N input to our study site. However, the farm manager had the sprayfield designed by NRCS based on their guidelines for the minimum area needed to manage the N generated on the farm. A large-gun irrigation system was used to spray the wastewater onto the sprayfield grass on an as-needed basis to lower the lagoon level. After discussion with the farmer concerning the potential negative effects of this sprayfield on our study, he ceased applying the wastewater above our south side control buffer (C) beginning in fall 2002, but buffers A and B continued to have direct upslope application of the wastewater for the duration of the study.

An event impacting our study occurred on 10 to 11 January 2003, when the farm manager accidentally pumped approximately 1,058,000 L of wastewater onto all of the south side buffers, including the control buffer. In addition, during winter 2002 to spring 2003, cropland directly upslope of both the grassed sprayfield and our south side buffers was used to grow winter cabbage (fig. 1). This land was not managed by our farm cooperator, so information on N application rates was not available. However, N application rates to winter vegetables grown on sandy Coastal Plain soils are generally high. A center-pivot irrigation system was located on the land used for winter cabbages, and some of the N applied by the grower was via injection through the irrigation system. The shape of the land was such that, during rainfall/runoff events, it would be possible for surface runoff from the cabbage to move directly into an upslope drainageway connecting with the drainageway-intermittent stream channel located midway between our south and north side buffer systems (fig. 1). All of these farm management decisions/events placed N and Cl sources directly or indirectly upslope of the south side buffers, with potential effects on the amount of N that the buffers could assimilate and N and Cl concentrations in the buffer drainageway-intermittent stream channel.

Although the original design of our experiment appeared to be compromised in 2002 with the installation of the lagoon wastewater sprayfield directly upslope of our south side buffers, a decision was made to continue the study on all buffers. Assuming that N from wastewater applied to the sprayfield might impact the buffers on the south side, it was decided that monitoring of shallow groundwater quality under these buffers would determine if they could assimilate sufficient N from all sources to still protect water quality. Nitrogen from the sprayfield potentially could enter the buffer system via shallow lateral subsurface groundwater flow or in extreme events by runoff. The original experimental design was also compromised by the accidental wastewater spill in 2003, and potentially compromised by N applied to cabbages grown in winter spring 2002-2003.

Shallow Groundwater Wells

A network of shallow groundwater sampling wells was established in each of the buffers during summer 1998 (fig. 3). There were three transects of wells within each buffer, with the wells located at 10, 20, 30, and 40 m downslope from the wastewater application pipe. The wells were constructed of 5.1 cm dia. PVC pipe with 15.2 cm of slotted well screen glued to the bottom of the PVC pipe. A PVC well cap was glued to the bottom of the well screen. During well installation, coarse sand was poured around the slotted well screen, and bentonite clay was poured from the sand up to about 5.1 cm from the soil surface. The sand was placed around the slotted screen to enhance entry of water. The bentonite clay was used to prevent leakage of water and solutes from the soil surface down the well casing to the screen. The area around the interface of the well and soil surface was dug out to about 30.5 cm dia. and 5.1 cm depth and filled with poured concrete to further minimize possible contamination of the shallow groundwater samples by leakage around the well casing. There were two wells at each of the sampling locations, one at 1 m and one at 2 m depth. Two different well depths were used to determine if there were significant differences in water quality relative to sampling depth within treatments. The wells were sampled using a peristaltic pump. There were 15 samplings of the shallow groundwater during the study: 28 February 2000 (Prior to wastewater application) through 10 December 2004 (fig. 2). Samples were collected when the groundwater table was high enough for the wells to contain water. Although we originally anticipated relatively even numbers of water samples from the 1 and 2 m well depths, periods of low rainfall (fig. 4) coupled with high temperatures and evapotranspiration resulted in long periods with a low water table where only the 2 m wells could be sampled. Consequently, there were more samples collected from the 2 m than from the 1 m wells.

Samples from the groundwater wells were transported to the laboratory following collection and immediately frozen. The water samples were analyzed for NO_3–N, NH_4–N, and Cl concentrations on a Lachat flow injector analyzer (QuikChem AE, Lachat Chemicals, Inc., Milwaukee, Wisc.) according to standard methods (APHA, 1989). The Lachat methods for NO_3–N, NH_4–N, and Cl were 10-107-04-1-A, 10-107-06-2-A, and 10-117-07-1-B, respectively. All samples were run in duplicate, and where substantial disagree–

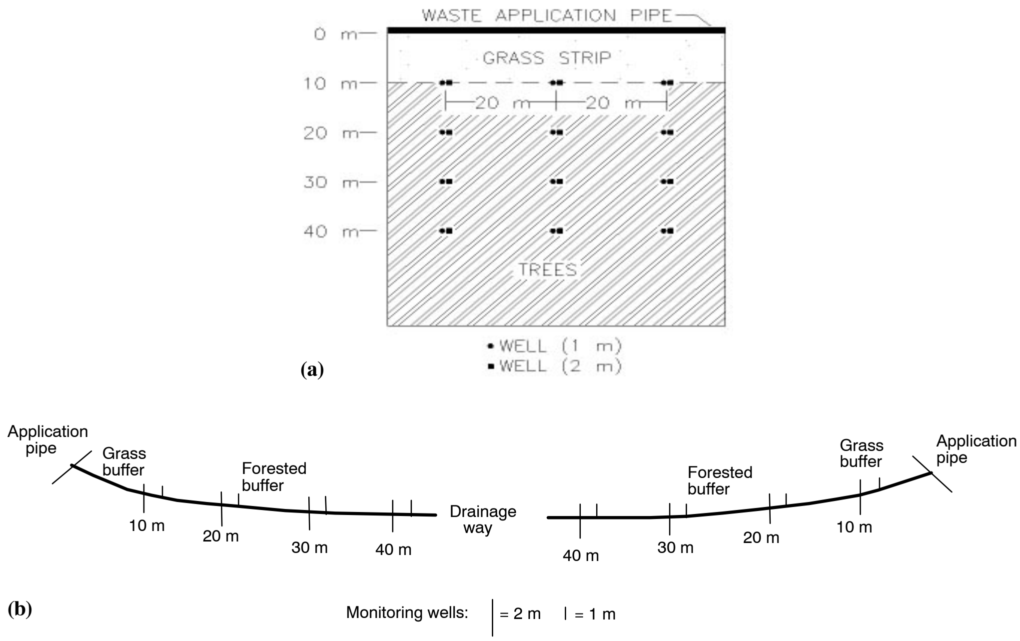

Figure 3. Diagrams showing (a) well sampling positions and (b) a cross-section for each buffer.

ment occurred, samples were reanalyzed. Extreme outliers were eliminated from the data set. The shallow groundwater samples were analyzed for Cl (chloride anion) concentration for use in interpreting water movement and N uptake patterns. Chlorine is a micronutrient and is used by plants only in very small quantities. When Cl is added to soils at agronomic rates (generally as KCl) or as a discrete ion in animal waste (swine lagoon wastewater), uptake by plants is so small, compared to the amount added to the soil that can be transported by leaching, that it serves as a conservative tracer for water movement (Hubbard et al., 1982).

Soil Samples

Soil samples were collected from the grassed portion of the buffers in November 2005 for C and N analyses. The decision to sample was prompted primarily by observed differences (discussed in the Results and Discussion section) in solute concentrations in shallow groundwater between the south and north side buffers. The objective was to determine if the extra NO_3-N appearing under both the south side buffers and the lowermost wells of the north side buffers was primarily from the sprayfield or from upslope cabbage production. No soil samples were collected in these grassed buffer areas at the start of the study in 2000, since we had not anticipated that our south side buffers would be impacted during the study by wastewater from an upslope sprayfield or by upslope cabbage production. The 2005 samples were collected from three locations in the grassed portion of each buffer using a tractor-mounted auger (15.2 cm dia.). The locations were centered in the grass buffer in line with the well transects (fig. 3). The soil was sampled at 0-15, 15-30, 30-60, and 60-90 cm depth. Samples were ball milled and analyzed for C and N using a Carolo Erba NA 1500 Series 2 nitrogen/carbon analyzer (Nelson and Sommers, 1996).

Statistical Analyses

The NH_4-N, NO_3-N, and Cl concentrations observed in shallow groundwater from 2000-2004 were statistically analyzed according to the variables year, wastewater application rate, sampling position, buffer location, and well depth. Statistical analyses by buffer location were to determine if additional N sources on the south side had impacted experimental results, while analyses by well sampling depth determined whether or not there were differences in groundwater quality between 1 and 2 m wells. The C and N data from the soil samples were also analyzed statistically. Statistical comparisons were made using the t-test least significant difference (LSD) procedure of the General Linear Models (GLM) at the 0.05 level of significance (SAS, 1985). All differences discussed in this article are significant unless otherwise noted.

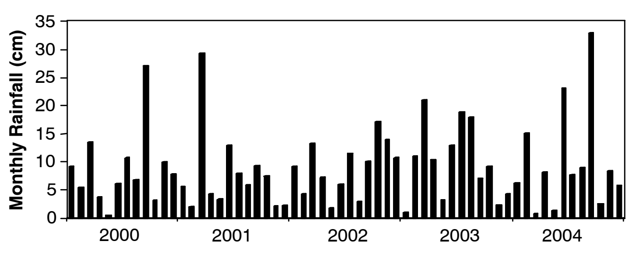

Figure 4. Monthly precipitation during the study.

Results and Discussion

Patterns in Shallow Groundwater Quality

Analyses of shallow groundwater quality (NH_4-N, NO_3-N, and Cl concentrations) were initially done with the data sorted by the single variables year, wastewater application rate, shallow groundwater sampling position, buffer location, and well depth. Uneven sample sizes, associated with non-separation of the data according to variables that significantly affected water quality, resulted in high standard deviations. However, the results (not shown) showed statistically significant trends for increasing NH_4-N, NO_3-N, and

Cl concentrations in shallow groundwater from 2000-2004, greater concentrations under the treatments receiving wastewater than under the control, greater concentrations at the locations nearest the application pipe with decreasing downslope concentrations (sampling position), much greater NO_3−N and Cl concentrations under the south side buffers than under the north side buffers, and little or no difference in water quality due to well depth (1 or 2 m). Since fewer water samples could be collected from the 1 m wells than from the 2 m wells, due to the water table being lower than 1 m much of the time, and since there were no significant differences in shallow groundwater solute concentrations associated with well depth, most of the discussion in the remainder of this article focuses on the data from 2 m, where sample collection was more consistent. Mean NH_4−N concentrations under all treatments and sampling positions were quite low, generally less than 1 mg L^{-1}, and were not of environmental quality concern (data not shown).

In our original experimental design, we did not anticipate differences in shallow groundwater quality due to buffer location. However, as discussed earlier, we inferred during the experiment that our design and results potentially had been affected by the addition of a sprayfield upslope of the south side buffers. During data analyses it also became apparent that groundwater quality under the south side buffers, and to a much lesser degree under the north side buffers, may also have been affected by winter cabbage production upslope of the sprayfield. Because of the observed differences in the data, we made statistical comparisons of groundwater quality for location by year (both 1 and 2 m well data combined) to determine if differences existed throughout the entire study period or if they only occurred after sprayfield wastewater application commenced (fig. 5). No differences in mean NO_3−N or Cl concentrations were found between the south and north side buffers in 2000. In 2001, mean NO_3−N concentrations were greater under the south side buffers than under the north side buffers, but there were no differences in mean Cl concentrations. Beginning in 2002, the year when the wastewater sprayfield was installed, and continuing through 2004, mean NO_3−N and Cl concentrations in shallow groundwater were significantly greater under the south side buffers than under the north side buffers (fig. 5). Overall, the data indicated that at the start of the experiment (2000), NO_3−N and Cl concentrations in shallow groundwater were the same under

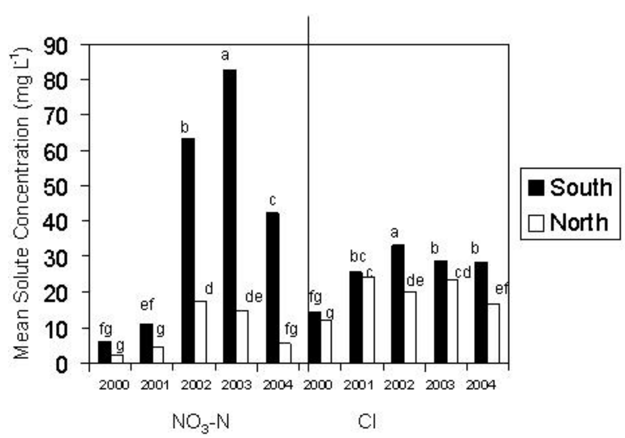

Figure 5. Mean NO_3−N and Cl concentrations in shallow groundwater by buffer location.

both the south and north side buffers. From 2002 through 2004, significantly greater mean NO_3−N and Cl concentrations under the south side buffers indicated that they were heavily impacted by additional wastewater applied to the sprayfield, by N and Cl from cabbage production on the upland field, or both.

Comparison was made of the ratio of NO_3−N to Cl in shallow groundwater between the south and north side buffers. The comparison showed that the south side buffers had a much higher ratio (1.66) than the north side buffers (0.47). Sources for the excess NO_3−N under the south side buffers could have been N from the cabbage production, N from the excess swine lagoon effluent applied to the sprayfield, or both. Logically, the N to Cl ratio of the applied wastewater was the same whether it was applied by overland flow to our buffers or to the sprayfield by large-gun irrigation, since all wastewater was from the same single lagoon pumped through a common underground pipe. Some of the excess NO_3−N contributing to the much higher NO_3−N:Cl ratio under the south side buffers could be attributable to overwhelming of soil NO_3−N assimilation processes by the wastewater, which then caused increased NO_3−N leaching to groundwater and an increased ratio. However, since the ratio was 3.5 times greater under the south side buffers than under the north side buffers, and since high NO_3−N values were also found in the lowermost wells under the north side control buffer, where the source could only have been the N entering the drainageway-intermittent stream channel from the cabbage production, we concluded that N fertilizer from the cabbages was the most likely source of the much higher shallow groundwater NO_3−N concentrations that were found under the south side buffers than under the north side buffers.

Test of Vegetated Buffers

Results from the south side buffers were clearly confounded by N and Cl sources in excess of what was applied with the overland flow of wastewater. Therefore, only the north side data were statistically analyzed as the test of the buffer system's ability to assimilate N. The statistical analyses of shallow groundwater quality under the north side buffers compared mean solute concentrations (NO_3−N and Cl only, as NH_4−N concentrations were quite low) by wastewater rates among all sampling positions within each year at 2 m depth (figs. 6 and 7). The analyses within years was to determine if there were water quality differences under buffers receiving wastewater as compared to the control, and between the 3/4X and 1X rates. The data were also separated within years and rates by sampling position to determine if there was a pattern of greater solute concentrations at the upslope ends of the buffers, where the wastewater was applied, and lower concentrations at the downslope sampling positions, as had been observed in a previous overland flow wastewater application study (Hubbard et al., 1998).

Mean NO_3−N concentrations in shallow groundwater at 2 m varied with year, wastewater rate, and sampling position (fig. 6). The greatest concentrations at all positions and treatments were observed in 2002 and 2003. Comparison of mean NO_3−N concentrations between the 10 and 20 m positions showed no significant differences for any of the years under the control buffer, but concentrations were significantly greater at 10 m than at 20 m for most years under both 3/4X and 1X. Little or no difference was found in water

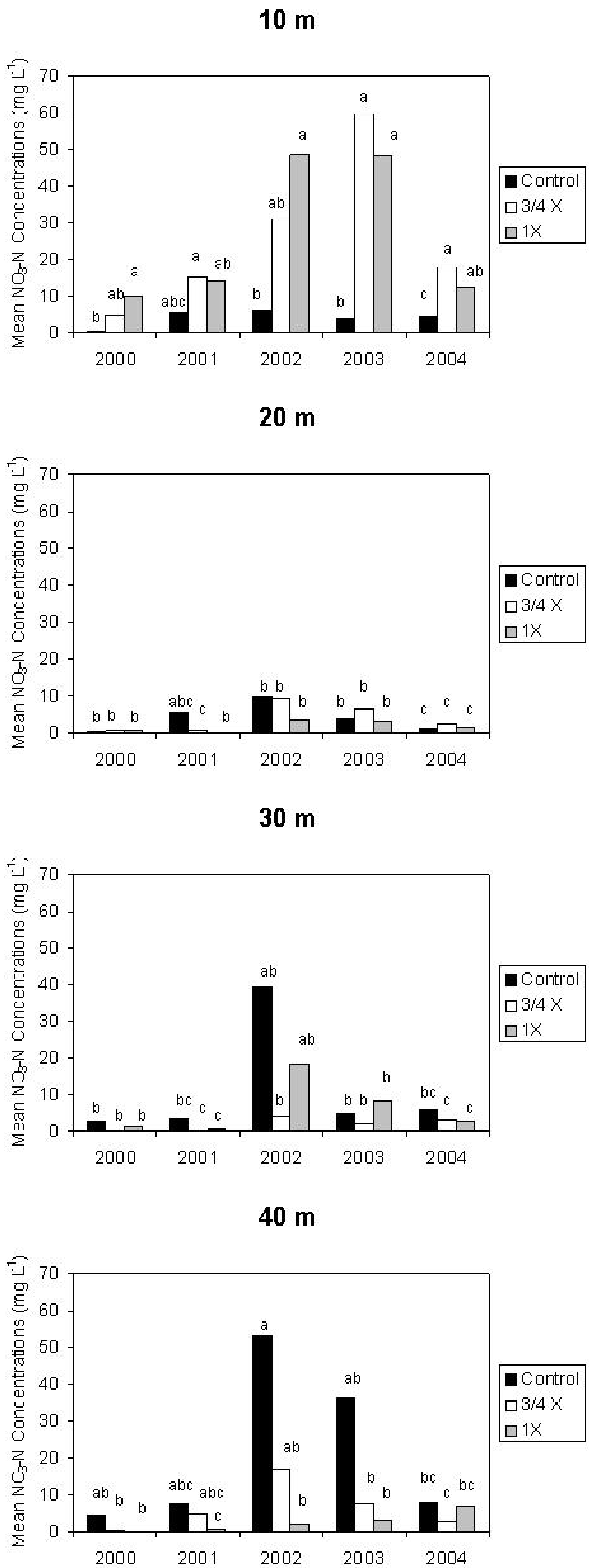

Figure 6. Mean NO_3–N concentrations (mg L^{-1}) in shallow groundwater at 2 m depth on the north side within years by wastewater rate and sampling position. Letters denote t-test least significant difference (LSD). The tests were performed within years across wastewater rate and sampling position. Means followed by the same letters are not significantly different at the 0.05 level.

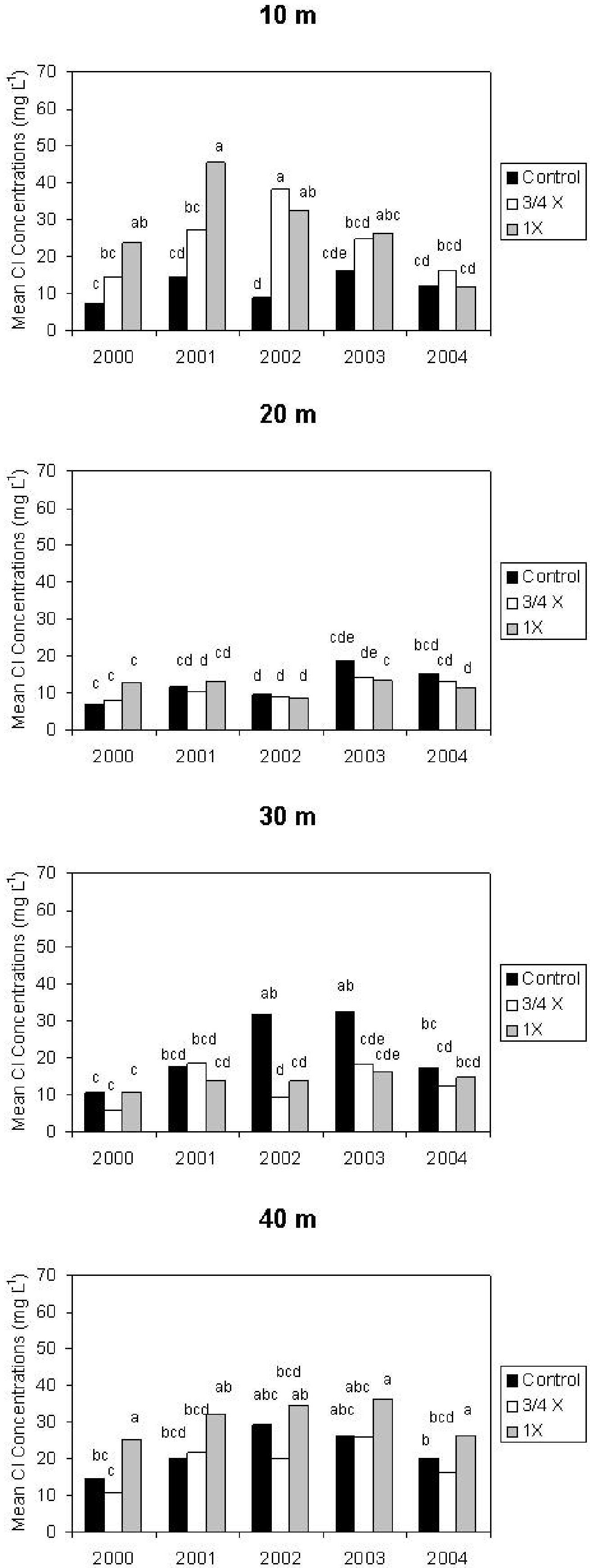

Figure 7. Mean Cl concentrations (mg L^{-1}) in shallow groundwater at 2 m depth on the north side within years by wastewater rate and sampling position. Letters denote t-test least significant difference (LSD). The tests were performed within years across wastewater rate and sampling position. Means followed by the same letters are not significantly different at the 0.05 level.

quality between the 3/4X and 1X treatments. The data indicated that wastewater application to the grass portion of the buffers (0 to 10 m) caused elevated NO_3-N concentrations in the 2 m wells at the 10 m sampling position, but not at the 20 m sampling position. The implication was that N assimilation processes (uptake and denitrification) on the buffers receiving wastewater at both the 3/4X and 1X rates removed sufficient N for concentrations at 20 m to remain about the same as those found under the control buffer. Concentrations of NO_3-N in shallow groundwater at 20 m downslope from the application pipe were 2 to 6 fold less than those found at 10 m downslope and, with the exception of some samples in 2002, were generally below the 10 mg L^{-1} drinking water standard. The NO_3-N concentrations in shallow groundwater at 20 m versus those at 10 m indicated that for our wastewater application rates, a 1:1 or greater ratio of buffer area width to wastewater application area width was sufficient for N assimilation and filtering. At much greater wastewater application rates, this conclusion probably would not hold, since the land area required to filter nutrients from wastewater is a function of the amount applied. This 1:1 ratio finding was the same as that found by Bingham et al. (1980) for caged-layer poultry manure, as was discussed in the introduction.

Mean NO_3-N concentrations in shallow groundwater under all treatments at the 30 m sampling position were similar to what was observed at the 20 m position, except for a relatively high mean concentration (not significantly different) observed under the control in 2002 (fig. 6). Mean NO_3-N concentrations at the 40 m position differed from those at the 20 and 30 m positions primarily in that the concentrations found under the control buffer in 2002 and 2003 were unexpectedly high. These concentrations were inconsistent with the low concentrations found at the 10 and 20 m positions under the control buffer, and also were inconsistent with the fact that no wastewater or other source of N was applied to the control buffer on the north side. The only possible source of N for the 40 m position wells under the control buffer, which were located in the buffer site drainageway/ponded area (fig. 1), was from upslope south side sources. Any N sources entering the portion of the drainageway upslope of our buffer study site would have first encountered the ponded area; hence, NO_3-N would have first appeared in the 40 m wells of the north side control buffer.

Figure 6 shows that where wastewater was applied by overland flow, elevated NO_3-N concentrations were found in shallow groundwater at the 10 m sampling position but not at the 20, 30, or 40 m positions in 2000, 2001, 2003, or 2004. Somewhat elevated NO_3-N concentrations were found at the 30 and 40 m positions in 2002, but given that elevated concentrations were not found at the 20 m position and that the control buffer also had high NO_3-N concentrations at the 30 and 40 m positions in 2002, it would appear that the N source for the concentrations found in these wells was not from the overland flow applied wastewater. As a test of farm-scale application of wastewater by overland flow, the results from the north side buffers indicated that this method can effectively assimilate N. Nitrate concentrations in shallow groundwater were high immediately downslope of the application area, but less than the drinking water standard at 20, 30, and 40 m downslope. However, what this study shows from the NO_3-N concentrations found in shallow groundwater under both the north and south side buffers, and perhaps what is more important, is that buffer systems are easily bypassed with concentrated flows, with resulting contamination of surface and ground waters. It would appear from the south side data that the combination of winter cabbage and a grassed sprayfield easily overwhelmed the capacity of the buffer systems to assimilate N. Installation of extended buffers alongside the drainageway adjacent to cabbage production would have reduced the high NO_3-N concentrations found under the south side buffers and at the 40 m position under the north side control buffer, although total N from both the cabbages and sprayfield may still have overloaded the buffers.

Mean Cl concentrations in shallow groundwater at 2 m on the north side also varied with year, wastewater rate, and sampling position (fig. 7). Concentrations increased in 2001-2004 with application of the wastewater, and concentrations at the 10 m sampling position were elevated under the buffers receiving wastewater as compared to the control buffer for all years except 2004, when the control and 1X rates had about the same concentrations. Comparison of water quality at the 10 and 20 m positions showed no significant differences in mean Cl concentrations under the control for any of the years. Under 3/4X, mean concentrations at 10 m were significantly greater than at 20 m in 2001 and 2002, while under 1X, mean Cl concentrations were significantly greater at the 10 m than at the 20 m position in 2000, 2001, and 2002. The significant differences in mean Cl concentrations between the 10 and 20 m positions showed that groundwater quality had been significantly affected at 10 m by the wastewater applied to the grass, but the concentration of Cl was attenuated by 20 m.

Significantly different mean Cl concentrations at the 30 m position between rates within years only occurred in 2002 and 2003 under the control buffer, where concentrations were greater than under the buffers where wastewater was applied. Since the control buffer had no overland flow applied source for Cl, it is hypothesized that these significantly greater concentrations were associated with the fertilizers applied to the cabbages. This result for Cl was similar to what was observed for NO_3-N at the 30 m position under the control buffer. Mean Cl concentrations at the 40 m position tended to be greater, although not significantly, than those at 30 m.

Carbon/Nitrogen Ratios of Soil Under the Grassed Buffers

Table 1 shows mean C/N ratios by soil depth for both the south and north side buffers. The general trends were that C/N ratios decreased with increasing soil depth and were lower under the 1X rate than under the 3/4X or control due to the greater amount of N applied at 1X. No differences in C/N ratios were found between the south and north side buffers. The C/N ratio information confirmed that the control, 3/4X, and 1X buffers on both the south and north sides received different wastewater N rate applications. The information also indicated that if N from the sprayfield entered the buffers by surface runoff or shallow subsurface flow on the south side, it did not affect the basic experimental design of two N rates and a control. Little or no difference in the C/N ratios of the grassed buffers between the south and north sides indicated that most likely the high NO_3-N concentrations in both the south side wells and the lowermost drainageway-intermittent stream associated wells on the north side were from N associated with the cabbage production rather than

Table 1. Mean soil C/N ratios by buffer area, wastewater rate, and soil depth.[a]

Buffer Area	Treatment Rate	Statistic	Soil Depth (cm) 0-15	15-30	30-60	60-90
North side	Control	Mean	18.68 ab	17.99 ab	15.10 ab	14.85 ab
		SD	1.13	2.73	2.74	3.46
South side	Control	Mean	22.63 ab	21.96 a	20.62 a	19.20 a
		SD	1.87	2.92	1.48	3.69
North side	3/4X	Mean	19.74 ab	16.89 ab	12.68 b	12.75 ab
		SD	3.13	3.42	3.71	4.30
South side	3/4X	Mean	27.04 a	18.92 ab	13.48 b	18.80 a
		SD	11.72	5.55	--	0.96
North side	1X	Mean	15.10 b	14.46 b	12.81 b	14.92 ab
		SD	2.44	2.51	3.71	5.35
South side	1X	Mean	16.78 b	15.34 ab	11.90 b	9.69 b
		SD	2.55	4.82	1.73	0.44
		LSD	10.07	6.79	5.96	7.94

[a] Letters denote t-test least significant difference (LSD). The tests were performed across buffer area and wastewater rate by soil depth. Means in the same column followed by the same letters are not significantly different at the 0.05 level.

from the wastewater sprayfield. If significant quantities of N from the sprayfield had been moving downslope via surface runoff or lateral subsurface flow beneath the south side grassed buffers, then this additional N should have resulted in lower C/N ratios in the soils on the south side grassed buffers than in those under the north side buffers. This would be particularly true for the 2005 soil sampling, since wastewater application to the sprayfield was ongoing throughout 2005, while wastewater application to all buffers ceased at the end of 2004. However, since there were no differences in the C/N ratios between the north and south side soils under the grassed buffers, the conclusion must be that the greater NO_3−N concentrations observed in wells on the south side and the lowermost sections of the north side were primarily from the cabbage production. The fact that the high NO_3−N concentrations were found in 2002 and 2003, years with cabbage production, and ceased in 2004, also adds to the conclusion that the cabbages were the source of excess N.

Conclusions

The study showed that overland flow vegetated buffer systems receiving swine lagoon effluent at the farm scale can effectively assimilate N such that NO_3−N and NH_4−N concentrations in shallow groundwater at the lower end of buffers remain below the drinking water quality standard of 10 mg L^{-1}. This was demonstrated on buffers with planted pine receiving wastewater only from applied overland flow.

The study also clearly showed that vegetated buffers cannot assimilate large quantities of N when wastewater or other N sources are applied at rates exceeding the assimilative capacity of the buffers. This was demonstrated on our buffers with mature pines that received N from multiple sources including overland flow applied wastewater, an upslope sprayfield, and winter cabbage production. The study also illustrated that complex water and solute movement in the natural landscape can strongly affect N assimilation by buffers, and that water quality cannot be protected by buffers when drainage patterns allow bypass of the buffer systems by nutrients applied to upland areas.

The overall conclusion from this study is that lower landscape vegetated buffers can assimilate N from applied wastewater at the farm scale, and hence can be a useful tool for animal wastewater management. However, it must be cautioned that wastewater application rates must be carefully controlled so that N input is in balance with the assimilative capacity of the buffers. Key variables determining the assimilative capacity of buffers include width, soils, and vegetation, with width in relation to wastewater application rate being the most critical. The study indicated that the ratio of buffer area width to wastewater application area width should be at least 1:1 when land application of wastewater is at or near agronomic rates. In addition, vegetated buffers must be continuous on the landscape to protect water quality from all contaminant sources.

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