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Vegetative Treatment System Impacts on Groundwater Quality

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Abstract
Increased environmental awareness has prompted the need for improved feedlot runoff control. Vegetative treatment systems (VTSs) provide a cost-effective option that may enhance environmental security by protecting water quality. Vegetative treatment systems are typically designed on the basis of hydraulic performance, which may result in excess application of some nutrients, specifically nitrogen and phosphorus. Groundwater quality monitoring is required to determine the effect, if any, that VTSs have on groundwater. Shallow groundwater (2 to 10 m) quality beneath six VTSs in Iowa was monitored over a four-year period. Monitoring wells were located upgradient, within, and downgradient of the VTSs. Groundwater samples were collected on a monthly basis and analyzed for ammoniacal nitrogen, chloride, nitrate-nitrogen, and fecal coliforms. A trend analysis was conducted to evaluate groundwater response patterns to VTS construction and use. In general, monitoring wells located within and downgradient of the VTS showed increasing trends in chloride and decreasing trends in nitrate concentrations. No trends for fecal coliforms or ammoniacal nitrogen were seen. Statistical analysis was performed to test for concentration differences between upgradient, within, and downgradient monitoring wells. In general, no differences in ammoniacal nitrogen concentration were seen. Fecal coliform concentrations were generally highest at the monitoring well within the VTS, but no difference was found between upgradient and downgradient concentrations. Chloride concentrations were generally significantly higher within and downgradient of the VTS when compared to the upgradient well; nitrate concentrations were generally significantly lower within and downgradient of the VTA than upgradient.

Keywords
Feedlot runoff, Groundwater monitoring, Groundwater quality, Vegetative infiltration basins, Vegetative treatment areas, Vegetative treatment systems

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Comments
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VEGETATIVE TREATMENT SYSTEM IMPACTS ON GROUNDWATER QUALITY

D. S. Andersen, R. T. Burns, M. J. Helmers, L. B. Moody

ABSTRACT. *Increased environmental awareness has prompted the need for improved feedlot runoff control. Vegetative treatment systems (VTSs) provide a cost-effective option that may enhance environmental security by protecting water quality. Vegetative treatment systems are typically designed on the basis of hydraulic performance, which may result in excess application of some nutrients, specifically nitrogen and phosphorus. Groundwater quality monitoring is required to determine the effect, if any, that VTSs have on groundwater. Shallow groundwater (2 to 10 m) quality beneath six VTSs in Iowa was monitored over a four-year period. Monitoring wells were located upgradient, within, and downgradient of the VTSs. Groundwater samples were collected on a monthly basis and analyzed for ammoniacal nitrogen, chloride, nitrate-nitrogen, and fecal coliforms. A trend analysis was conducted to evaluate groundwater response patterns to VTS construction and use. In general, monitoring wells located within and downgradient of the VTS showed increasing trends in chloride and decreasing trends in nitrate concentrations. No trends for fecal coliforms or ammoniacal nitrogen were seen. Statistical analysis was performed to test for concentration differences between upgradient, within, and downgradient monitoring wells. In general, no differences in ammoniacal nitrogen concentration were seen. Fecal coliform concentrations were generally highest at the monitoring well within the VTS, but no difference was found between upgradient and downgradient concentrations. Chloride concentrations were generally significantly higher within and downgradient of the VTS when compared to the upgradient well; nitrate concentrations were generally significantly lower within and downgradient of the VTA than upgradient.*

Keywords. *Feedlot runoff, Groundwater monitoring, Groundwater quality, Vegetative infiltration basins, Vegetative treatment areas, Vegetative treatment systems.*

Open-lot animal feeding operation (AFO) runoff has been recognized as a potential pollutant to receiving waters because it contains nitrogen, phosphorus, organic matter, solids, and pathogens. The U.S. Environmental Protection Agency (EPA) developed a set of effluent limitation guidelines (ELGs) that described the design and operating criteria for feedlot runoff control systems on concentrated animal feeding operations (CAFOs) (Anschutz et al., 1979). These effluent limitation guidelines historically required collection, storage, and land application of feedlot runoff; however, recent modifications allowed the use of alternative treatment systems when the performance of the alternative systems, based on the mass of nutrients released, was equivalent to or exceeded that of an appropriately sized containment system (EPA, 2006).

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Vegetative treatment systems (VTSs) are one possible alternative runoff control technology that has been proposed and implemented. A VTS is a combination of treatment components, at least one of which utilizes vegetation, to manage runoff from open lots (Moody et al., 2006). Vegetative treatment areas (VTAs) and vegetative infiltration basins (VIBs) are two possible treatment component options for VTSs. A sloped VTA is defined as an area level in one dimension, to facilitate sheet flow, with a slight slope along the other dimension, planted and managed to maintain a dense stand of vegetation (Moody et al., 2006). Operation of the VTA consists of applying solids settling basin effluent uniformly across the top of the vegetated treatment area and allowing the effluent to sheet-flow down the slope (Moody et al., 2006). Ikenbery and Mankin (2000) identified several possible methods by which effluent was treated in VTAs, including settling solids, infiltrating runoff, and filtering as effluent flowed through the vegetation. A VIB is defined as a flat area, surrounded by berms, planted to permanent vegetation (Moody et al., 2006). Effluent is distributed over the VIB surface via a flood effect. VIBs have drainage tiles located 1 to 1.2 m (3.4 to 4 ft) below the soil surface to encourage infiltration of effluent. The tile lines collect effluent that percolates through the soil profile. The effluent then receives additional treatment, often in a VTA. Nutrient and pathogen removal in the VIB relies on filtration as the effluent percolates through the soil, plant uptake of nutrients, and micro-

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bial degradation or transformation of the nutrients and pathogens by soil fauna (Moody et al., 2006).

Two design approaches, one utilizing a hydraulic balance and the other a nitrogen balance, have been proposed for sizing VTAs (Woodbury et al., 2006). Previous work by Woodbury et al. (2005) showed that if VTSs are designed using the nitrogen balance approach, they can successfully utilize applied nitrogen. However, in many cases, VTSs have been designed based on hydraulic performance. This typically results in smaller VTSs, which may enhance opportunities for deep percolation of runoff water below the root zone and overapplication (i.e., in excess of agronomic demand) of nutrients, especially nitrogen (Woodbury et al., 2006). As VTSs rely heavily on the soil-plant-microbe system to filter, sorb, and transform nutrients and contaminants before water percolates through the soil profile, there is a need to understand the impacts that VTSs designed on a hydraulic performance basis have on groundwater quality.

The objective of this study was to evaluate the impact that VTS installation and use had on shallow groundwater quality. This article reports results from a four-year groundwater monitoring study at six VTS locations on open beef feedlots in Iowa. A trend analysis was used to evaluate temporal patterns in the groundwater concentrations, specifically those of chloride and nitrate. Chloride and nitrate concentrations were measured in each well at all six sites. The results of the trend analysis were used to compare groundwater concentrations before and after VTS use. An analysis of variance was then used to compare groundwater concentrations upgradient, within, and downgradient of the VTSs. Estimates of leached masses of nitrate and chloride along with soil concentrations of nitrate and ammonium were used to complement this analysis.

MATERIAL AND METHODS

SITE DESCRIPTIONS

The performance of six vegetative treatment systems was monitored. These treatment systems were located on CAFO-sized (>1000 head) open beef feedlots throughout the state of Iowa. At most of the locations, more than one VTS was installed. At these sites, one of the VTSs was monitored by Iowa State University (ISU) for nutrient outflow from each system component; only effluent outflow from the final treatment component was monitored for the other treatment systems (i.e., those not monitored by ISU). Table 1 shows the VTS components, the number of cattle, and the feedlot, VIB (where applicable), and VTA areas for both the on-farm and ISU-monitored portions of the feedlot and runoff control system. At sites with more than one VTS, performance of VTSs not monitored by ISU was assumed to be equal to that of the VTS monitored by ISU. This assumption is based on the VTSs being constructed at the same time, managed by the same individual, experiencing the same weather conditions, and having similar sizing characteristics to the ISU-monitored system. Groundwater wells were sited and installed at each farm by an Iowa Department of Natural Resources geologist. Maps showing locations of the wells in relation to the feedlot and VTS locations are shown in figure 1. Full descriptions of these sites are provided by Andersen et al. (2013); brief descriptions are provided here.

Central Iowa 1 (CN IA 1) was a 4.11 ha feedlot permitted for 1500 head of cattle. Runoff effluent drained into two solids settling basins (SSB) designed to hold 5640 m^3 of effluent. A gate valve on the SSB outlets was used to control outflow volumes and rates onto the VTA. The VTA consisted of three sections operated in parallel; each section was 24 m wide and averaged 311 m long. The VTA soil consisted of Clarion loam, Cylinder loam, and Wadena loam (USDA-NRCS, 2010). Long-term average precipitation at this location was 85 cm per year. Three groundwater wells were installed at CN IA 1. Depths of the upgradient, within, and downgradient wells were 7.8, 3.8, and 3.7 m, respectively; approximately the bottom meter of each well was screened. Average depths to groundwater were approximately 3.5, 0.65, and 1.1 m, respectively, at the three well locations.

The VTS at Central Iowa 2 (CN IA 2) consisted of three SSBs, five VIBs, and two VTAs. Runoff from the 3.26 ha feedlot drained into concrete SSBs with a total volume of 136 m^3. Prior to reaching the SSB outlet, effluent flowed through a fence of round bales. A gate valve controlled when, how much, and at what rate effluent was applied to the VIBs. The SSB outlets applied effluent into a series of VIBs with a total area of 1.09 ha. Effluent from the VIBs was pumped onto one of two VTAs. Soils in the VIB consisted of Nicollet loam and Webster clay loam, and the VTA was Harps loam (USDA-NRCS, 2010). Long-term average annual precipitation in this region averaged 89 cm. Three groundwater wells were installed at CN IA 2. Well depths were approximately 4 m, with the bottom meter of each well screened. Average depths to groundwater were approximately 1.5, 1.5, and 1.2 m at the upgradient, within, and downgradient locations, respectively.

Table 1. Description of whole farm and Iowa State University monitored vegetative treatment systems at the six study feedlots. Information includes the number of cattle (head), the VTS components, and the size of the feedlot, solids settling basin (SSB), vegetative infiltration basin (VIB), and vegetative treatment area (VTA).

	Number of Cattle		VTS Components		On Farm				ISU Monitored			
Site[a]	Farm	ISU	On-Farm	ISU-Monitored	Feedlot (ha)	SSB (m^3)	VIB (ha)	VTA (ha)	Feedlot (ha)	SSB (m^3)	VIB (ha)	VTA (ha)
CN IA 1	1500	1000	2 SSB, 3 VTA	1 SSB, 2 VTA	4.11	5640	-	2.14	3.09	4290	-	1.49
CN IA 2	2400	650	3 SSB, 5 VIB, 2 VTA	1 SSB, 1 VIB, 1 VTA	3.26	136	1.09	0.72	1.07	51	0.32	0.22
NW IA 1	3400	1400	3 SSB, 5 VTA	1 SSB, 1 VTA	8.92	8906	-	4.06	2.91	3710	-	1.68
NW IA 2	4000	4000	1 SSB, 1 VIB, 1 VTA	1 SSB, 1 VIB, 1 VTA	2.95	110	1.01	0.60	2.96	110	1.01	0.60
SW IA 1	2300	2300	1 SSB, 10 VTA	1 SSB, 10 VTA	7.49	11,550	-	4.05	7.49	11550	-	4.05
SW IA 2	5500	1200	12 SSB, 7 VTA	1 SSB, 1 VTA	19.67	33,180	-	18.4	3.72	6275	-	3.44

[a] CN IA 1 = Central Iowa 1, CN IA 1 = Central Iowa 2, NW IA 1 = Northwest Iowa 1, NW IA 2 = Northwest Iowa 2, SW IA 1 = Southwest Iowa 1, and SW IA 2 = Southwest Iowa 2.

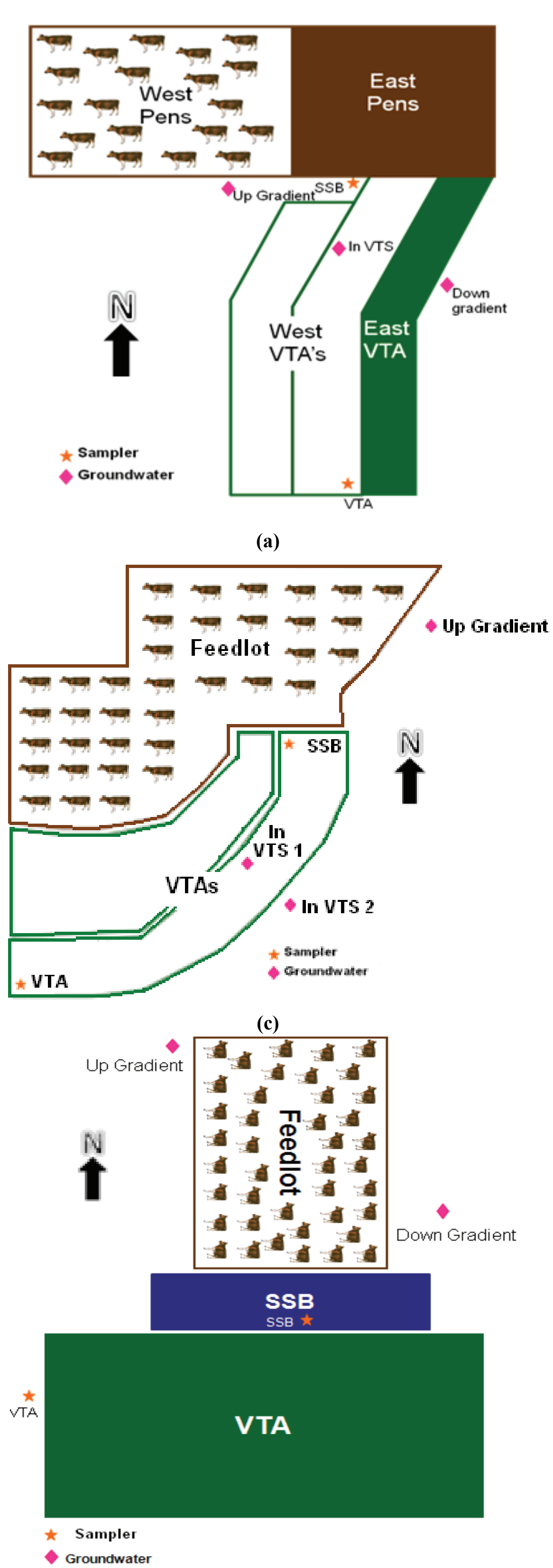

Figure 1. Groundwater well locations in relation to feedlot and VTS components for (a) Central Iowa 1, (b) Central Iowa 2, (c) Northwest Iowa 1, (d) Northwest Iowa 2, (e) Southwest Iowa 1, and (f) Southwest Iowa 2.

Northwest Iowa 1 (NW IA 1) consisted of an 8.92 ha feedlot permitted to hold 3400 head of cattle. Feedlot runoff was collected in 1.2 m deep SSBs having a total volume of 8906 m^3. The SSB outlet pipe discharged effluent uniformly along a concrete spreader located across the top width of the 4.06 ha VTA. A valve was used to actively control application of effluent from the SSB to the VTA. The VTA soil consisted of Galva silty clay and Radford silt loam (USDA-NRCS, 2010). Long-term average precipitation at this location was 66 cm per year. Three groundwater wells were installed at NW IA 1; the wells were installed upgradient, in VTA 1, and in VTA 2. Depths of the upgradient, VTA 1, and VTA 2 wells were 6, 9, and 6 m, respectively. Approximately the bottom meter of each well was screened. Average depths to groundwater were approximately 3.7, 3.9, and 1.9 m, respectively, at the three well locations. Based on groundwater level monitoring, the general flow direction appeared to be toward the well in VTA 2 from both the upgradient well and the well in VTA 1.

Northwest Iowa 2 (NW IA 2) had an SSB-VIB-VTA system designed to control runoff from a 2.95 ha concrete feedlot. An SSB with 110 m^3 capacity collected the feedlot runoff. Effluent from the SSB was applied to a 1.01 ha VIB. The VIB had 15 cm diameter perforated tiles installed 1.2 m deep and spaced 4.6 m apart. Flow from the tile lines was collected in a sump and pumped onto the VTA. A gated pipe was used to spread flow evenly across the top width of the VTA. The 0.6 ha VTA was divided into two 27 m wide sections. At a given time, effluent was pumped onto only one of the VTA sections. The section receiving effluent was switched manually by the producer. The soil at NW IA 2 consisted of Moody silty clay loam (USDA-NRCS, 2010). Long-term average annual precipitation at this location was 66 cm per year. Two groundwater wells were installed at NW IA 2. Depths of the upgradient and downgradient wells were 9 and 6 m, respectively. The bottom meter of the each well was screened. Average depths to groundwater at the upgradient and downgradient wells were 5.7 and 3.4 m, respectively.

Southwest Iowa 1 (SW IA 1) was a 7.49 ha feedlot. Runoff drained into an 11,550 m^3 solids settling basin. A gate valve on the settling basin outlet was used to control SSB outflow to the VTA. The 4.05 ha VTA was divided into ten sections. Effluent reaching the bottom of each VTA section was then directed to the westernmost VTA section. The VTA outlet was located 0.6 m above (in elevation) the bottom of the westernmost section. This provided storage of effluent in the VTA before outflow would occur. Tile lines, installed to control water table depth and enhance infiltration, surrounded each VTA section. A tile access point was installed in early 2008 to monitor the amount and quality of flow in the tile lines. This point was located such that all flow was from the VTA. Soils in the VTA consisted mostly of Judson silty clay loam with smaller areas of Colo-Ely complex (USDA-NRCS, 2010). Long-term average annual precipitation in this area was 91.5 cm. Two groundwater monitoring wells were installed. The locations of these two wells do not allow for analysis of the impact of the VTS but instead test the impact of the feedlot. Depths of both wells were approximately 6.1 m; average depths to groundwater were 1.9 and 2.9 m for the upgradient and downgradient wells, respectively.

Southwest Iowa 2 (SW IA 2) was a 19.67 ha feedlot. Runoff drained into solids settling basins that were designed to hold a 25-year, 24 h storm. Gate valves were installed on the SSB outlets to control effluent application onto the VTAs. The 18.4 ha VTA was constructed with earthen-berm level spreaders along the length. The spreaders slowed the flow of effluent through the system, increasing the time for infiltration to occur. The VTA soil consisted of Kennebec silt loam (USDA-NRCS, 2010). Long-term average precipitation at this location was 92 cm per year. Three groundwater monitoring wells were installed at SW IA 2 to collect samples upgradient, within, and downgradient of the VTS. Average water table depths at the three well locations were 5.5, 2.4, and 5.1 m, respectively. Groundwater depth monitoring indicated that the upgradient well was truly upgradient; however, the downgradient well is most likely a second monitoring well within the VTS.

MONITORING METHODS

Groundwater samples were collected monthly (between the 1st and 15th day of the month) from each monitoring well and tested for ammoniacal nitrogen (NH_3/NH_4-N; EPA Method 350.1 on a block digester with Foss automated titration), chloride (Cl^-; Standard Method 4500-Cl E on an ion chromatograph), nitrate-nitrogen (NO_3-N; EPA Method 353.1 by automated cadmium reduction), and fecal coliform concentrations (Standard Method 9222D). Occasionally, wells were dry and no sample could be collected. Prior to sample collection, stagnant water was purged from the well. The well was then allowed to recharge for five to seven days, after which a 250 mL sample was collected (100 mL of no treatment, 100 mL of acid treatment, and 50 mL in a sterile bottle for fecal coliform enumeration). After collection, the sample was placed on ice and shipped to a certified laboratory (Test America, Cedar Rapids, Iowa) for analysis following chain-of-custody protocol.

Portable samplers (6712, Teledyne ISCO, Lincoln, Neb.) equipped with either a pressure transducer (720 submerged probe module, Teledyne ISCO), an area velocity meter (750 area velocity module, Teledyne ISCO), or an analog-to-digital converter (4 to 20 mA sampler input interface, Teledyne ISCO) interfaced with a turbine flowmeter (turbine flowmeter with tricon/E 3 transmitter, Neptune Technology Group, Tallassee, Ala.) were installed at the outlets of the SSBs, VIBs, and VTAs at each site to measure effluent flow rates and total outflows from each VTS component. The ISCO samplers were programmed with site and VTS component specific programs that collected multiple samples (from two to eight samples depending on the size of the release event) from each runoff event based on cumulative flow volumes. Effluent samples from outflow events were collected and analyzed for ammoniacal nitrogen (NH_3-N; Standard Method 4500-NH3 B and E, macro digestion followed by titration), chloride (Cl^-; Standard Method 4500-Cl E on an ion chromatograph), total Kjeldahl nitrogen (TKN; EPA Method 351.2, block digester followed by Lachat autoanalyzer), nitrate-nitrogen (NO_3-N; EPA Method 353.3, manual cadmium reduction), and fecal coliform concentrations (Standard Method 9222 D). Concentrations of organic nitrogen (ON = TKN – NH_3-N) and total nitrogen (TN = TKN + NO_3-N) were calculated based on monitored concentrations. Summaries of the effluent concentrations in the outflow from each VTS component at each site are provided by Andersen et al. (2013). Nutrient loadings onto the VIBs and VTAs were calculated by multiplying the inflows and outflows from each event times the nutrient concentration measured for that event and dividing by the area of the VTA. Performances, in terms of nutrient concentrations and loadings, of the non-ISU monitored systems were assumed to be the same as the ISU-monitored system at that site.

Concentration Data Analysis

Regression analysis (independent variables of chloride, nitrate, ammoniacal nitrogen, and fecal coliform concentrations) was used to analyze temporal trends in the groundwater concentration data; trends were only found for chloride and nitrate. The regression equation fit was a model for an intervention at an unknown time, which is intended to evaluate if the intervention (construction and use of the VTS) impacted water quality, and if so how quickly. This equation fits the data to three distinct phases (eq. 1). The first phase of the equation was a "stationary" mean, i.e., the average concentration before VTS construction and use. At the intervention point, the equation began a linear concentration increase or decrease phase, which occurred until the concentrations reached a new mean. The linear increase or decrease portion indicated how quickly the VTS was affecting groundwater concentrations. Where applicable, the third stage of the equation represented the average groundwater concentration after implementation and use had created a new average, approximately steady-state groundwater concentration:

$$C_i = B_0 + \lambda(t_i - \tau_1)I_{[\tau_1,\infty)} - \lambda(t_i - \tau_2)I_{[\tau_2,\infty)} + \varepsilon_i \qquad (1)$$

where C_i is the sample concentration at the ith sampling time, B_0 is the average concentration before construction of the VTS, λ is the rate of change in groundwater concentration per day during the linear increase/decrease phase, τ_1 is the lag time (days) before the linear increase/decrease phase begins, τ_2 is the lag time (days) until the linear increase/decrease ends, $I_{[a,\infty)}$ is a step function defined as 0 for all times less than a and as 1 for all times greater than or equal to a, where a is a threshold value that the count variable (days) is compared to, ε_i is the model fit error of the ith sampling time, and t_i is the count variable that tracks the number of days since the background water sample was collected. Equation 1 was fit to the monitored data using the solver function in Microsoft Excel to minimize the sum of squares of error between the monitored and modeled concentrations.

After fitting equation 1, a before-and-after analysis was performed for each well to determine if the change in the average concentration was significant. This analysis was performed in Microsoft Excel as a comparison of means and assuming that the variances for both time periods (i.e., before and after intervention) had homogeneous variances (t-test with equal variance). Variances were estimated based on the residual error between the fitted model and the measured concentrations. An analysis of variance was also used to test for differences between upgradient, in VTS, and downgradient wells. This analysis was run as a repeated measures experiment. Only the concentration measurements falling into the third phase of equation 1 were used to evaluate differences between locations. The analysis was conducted as a repeated measures experiment using the PROC MIXED command in SAS (ver. 9.2, SAS Institute, Inc.). The analysis was conducted on a per site basis; location (i.e., upgradient, in VTS, and downgradient) was considered a fixed factor, and replication was with time.

Estimating Leaching Volumes and Masses

Along with evaluating the trends in chloride and nitrate concentration in groundwater, estimating the mass of these parameters leached also provides significant insight into system performance and environmental impacts. A water balance (eq. 2) was utilized to estimate the amount of leaching that occurred; this balance was conducted in the Soil-Plant-Air-Water (SPAW) model (Saxton, 2008):

$$L = P + I - R - ET - \Delta S \qquad (2)$$

where L is the volume of water leached (m^3 ha^{-1}), P is the volume of water added through precipitation (m^3 ha^{-1}), I is the volume of the water added through effluent application (m^3 ha^{-1}), R is the volume of water lost as runoff from the VTA (m^3 ha^{-1}), ET is the volume of water evaporated and transpired from the VTA (m^3 ha^{-1}), and ΔS is the change in soil moisture occurring during the monitoring period (m^3 ha^{-1}). Precipitation depths were measured using a tipping-bucket rain gauge (674, Teledyne ISCO, Lincoln, Neb.). A passive rain gauge installed on site was used as a backup in case of power failure. Iowa Environmental Mesonet data (http://mesonet.agron.iastate.edu) for the location closest to each site were used to determine precipitation depths for events occurring between 1 November and 1 April, which were mostly snowfall. Volumes of I and R were measured using ISCO 6712 samplers (Teledyne ISCO, Lincoln, Neb.) equipped with either ISCO 750 low-profile area-velocity sensors for pipe outlets or ISCO 720 submerged probes in conjunction with a 0.45 m (1.5 ft) H-flume for non-pipe outlet locations. The ET and ΔS volumes were estimated using the SPAW model (Saxton, 2008) to simulate the hydraulic budget of the site based on monitored site and weather conditions, as described by Andersen et al. (2010).

At sites with an in-VTS well (Central Iowa 1, Central Iowa 2, Northwest Iowa 1, and Southwest Iowa 2), the estimated leached volume was multiplied by the monitored groundwater concentration from the in-VTS well using the mean after-intervention concentrations as determined using equation 1. The groundwater sample was assumed to represent the concentration of the leachate, as empirical evidence suggested that the large volumes of effluent applied in the VTAs cause water table mounding (Machusick et al., 2011). Monitoring at these sites suggests that this was occurring, as water table levels within the VTAs were typically higher than those monitored before system operation commenced. Additionally, results indicated that in many cases chloride levels monitored at the in-VTA well approached those of the applied effluent, indicating little mixing with groundwater flow at the in-VTS well. No in-VTS well was installed at Northwest Iowa 2; thus, an alternative method was used to determine the mass leached at this site. In this case, equation 3 was used to estimate groundwater base flow. This equation represents a mass balance of a conservative tracer, in this case chloride:

$$Q = L\frac{C_{down} - C_{avg}}{C_{up} - C_{down}} \tag{3}$$

where Q is the volume of groundwater flow through the upper end of the VTA (m^3), L is the volume of leachate (m^3), C_{up} is the concentration of chloride in the upgradient well (mg L^{-1}), and C_{avg} is the average concentration in the applied effluent (mg L^{-1}) corrected for plant uptake, precipitation, VTA outflow, and evapotranspiration and scaled based on the relationship between the applied chloride concentration and the in-VTS groundwater concentration of the other sites. C_{down} is the concentration of chloride in the downgradient well (mg L^{-1}). Groundwater concentrations were taken as values obtained for the new steady-state conditions as determined in the trend analysis. This analysis relied on several assumptions, most notably that vertical leakage of groundwater through the aquatard below the monitored water table is negligible (conservation of mass of water), that leached water is completely mixed with groundwater by the time it is sampled at the downgradient well (complete mixing), and that effluent is uniformly applied over the VTA. The same concept was then applied to nitrate. The value of flow (Q) obtained from equation 3 was used in equation 4 to determine the concentration of nitrate-nitrogen in the leachate. This concentration was multiplied by the leaching volume to determine nitrate-nitrogen leaching:

$$C_{leach} = \frac{(Q + L)C_{down} - QC_{up}}{L} \tag{4}$$

Due to the siting of the groundwater wells at Southwest Iowa 1, neither of these methodologies could be used; however, tiles were installed around the VTAs at this site. Flow and concentration measurements from these tiles provided a measurement of the masses of chloride and nitrate leaching from the VTAs.

Soil Sampling

Soil sampling was conducted before and then again after approximately two and three years of system operation. A soil sample was collected near the inlet and outlet of each VTA component. During the initial soil sampling, GPS coordinates were recorded for every sample location so that the same spot would be sampled in subsequent years. This allowed changes in soil nutrient contents with time to be tracked at various positions in the VTA. At each soil sampling location, a soil sampling probe (Giddings Machine Co., Windsor, Colo.) was used to collect a 2.54 cm (1 in.) diameter soil core that was 122 cm (48 in.) long. The sample was cut into segments to represent the 0-15.4 cm (0-6 in.), 15.4-30.5 cm (6-12 in.), 30.5-61 cm (12-24 in.), 61-94.4 cm (24-36 in.), and 94.4-122 cm (36-48 in.) depths. Each of these segments was put in a soil sampling bag and sent to the Soil and Plant Analysis Laboratory at Iowa State University for analysis of 2 M KCl extractable NO_3-N and NH_4-N using a QuikChem 8000 Series FIA (Lachat Instruments, Loveland, Colo.). Average concentrations of both parameters were calculated for each depth increment; the calculated average was assumed to represent the concentration of the midpoint of each sampling depth.

Results and Discussion

Chloride and Nitrogen Loadings

Chloride and nitrogen loadings were calculated from the measured inflows and outflows of the VTAs in this study (table 2). Chloride loadings were quite high, ranging from about 860 to over 3500 kg ha^{-1} $year^{-1}$. Similarly, total nitrogen loadings were also high, ranging from about 520 to over 1850 kg N ha^{-1}. Of this nitrogen, about 45% was ammoniacal (39% to 60%) and 55% was organic (40% to 62%). Less than 1% of the applied nitrogen was generally in nitrate form.

Groundwater Temporal Trends

The temporal trend analysis of groundwater concentrations was conducted by fitting equation 1 to the monitored concentration data for each parameter at each well and at each site. No trends for ammoniacal nitrogen or fecal coliform concentrations were found for any well at any site. Trends in chloride and nitrate were seen at most locations and are discussed below.

Chloride

Chloride was present in large quantities in the feedlot runoff; flow-weighted average concentrations of chloride in the SSB effluent were 234, 205, 596, 456, 232, and 500 mg L^{-1} for CN IA 1, CN IA 2, NW IA 1, NW IA 2, SW IA 1, and SW IA 2, respectively (concentrations for CN IA 2 and NW IA 2 are for the VIB effluent, as this was applied to the VTA). Chloride is relatively unreactive, i.e., it is not sorbed to soil, and only small amounts are incorporated into biomass (e.g., chloride can account for between 0.2% and 2% of dry mass of reed canary grass). In general, this means that 5% to 20% of the chloride could have been accounted for in the vegetation. As such, chloride was treated as a conservative tracer when analyzing and interpreting the groundwater data.

A plot of the chloride trends at the CN IA 1 monitoring wells is shown in figure 2a. In this figure, the x-axis represents the number of days the system was in operations; the

Table 2. Chloride (Cl^-) and nitrogen (TN = total nitrogen, TKN = total Kjeldahl nitrogen, NH_4-N = ammoniacal nitrogen, ON = organic nitrogen, and NO_3-N = nitrate-nitrogen) loadings onto the vegetative treatment areas at the six study sites. All values are in kg ha^{-1} $year^{-1}$.

Site[a]	Cl^-	TN	TKN	NH_4-N	ON	NO_3-N
CN IA 1	870	968	966	437	529	2
CN IA 2	1172	602	593	232	361	9
NW IA 1	3514	1865	1849	765	1084	15
NW IA 2	2711	1496	1486	705	780	10
SW IA 1	863	517	517	308	209	1
SW IA 2	2333	907	896	356	540	11

[a] CN IA 1 = Central Iowa 1, CN IA 1 = Central Iowa 2, NW IA 1 = Northwest Iowa 1, NW IA 2 = Northwest Iowa 2, SW IA 1 = Southwest Iowa 1, and SW IA 2 = Southwest Iowa 2.

0 value represents the date on which the background sample at each well was collected. VTA operation then commenced within one month. At CN IA 1, chloride concentrations in the upgradient well remained constant over the 3.5 years of monitoring; concentrations at the in-VTS and downgradient wells both increased after VTS operation began. Statistical results indicated that chloride concentrations were significantly different ($p < 0.0001$) after use of the VTS as compared to pre-VTS conditions at both the in-VTS and downgradient wells. Model fitting results indicated that in-VTS chloride concentrations increased by 124 mg L^{-1}, while downgradient concentrations increased by 15 mg L^{-1}. Concentrations at the VTS and downgradient wells quickly reached new steady-state levels, presumably due to the shallow depth to groundwater at this site. Groundwater chloride concentrations within the VTS well stabilized at an average of 200 ±30 mg L^{-1} (mean ±SD). The graphed data show a cyclical pattern: groundwater concentrations decreased in the winter and increased in the summer. This follows the effluent application pattern for the VTA, as no effluent was applied to the VTA during frozen ground conditions. Also of note are the high chloride concentrations at the upgradient well (273 ±36 mg L^{-1}). This well was located at the edge of the feedlot; it appears that leaching of chloride from the pen surface led to the elevated levels. Previous work (Olson et al., 2005; Maule

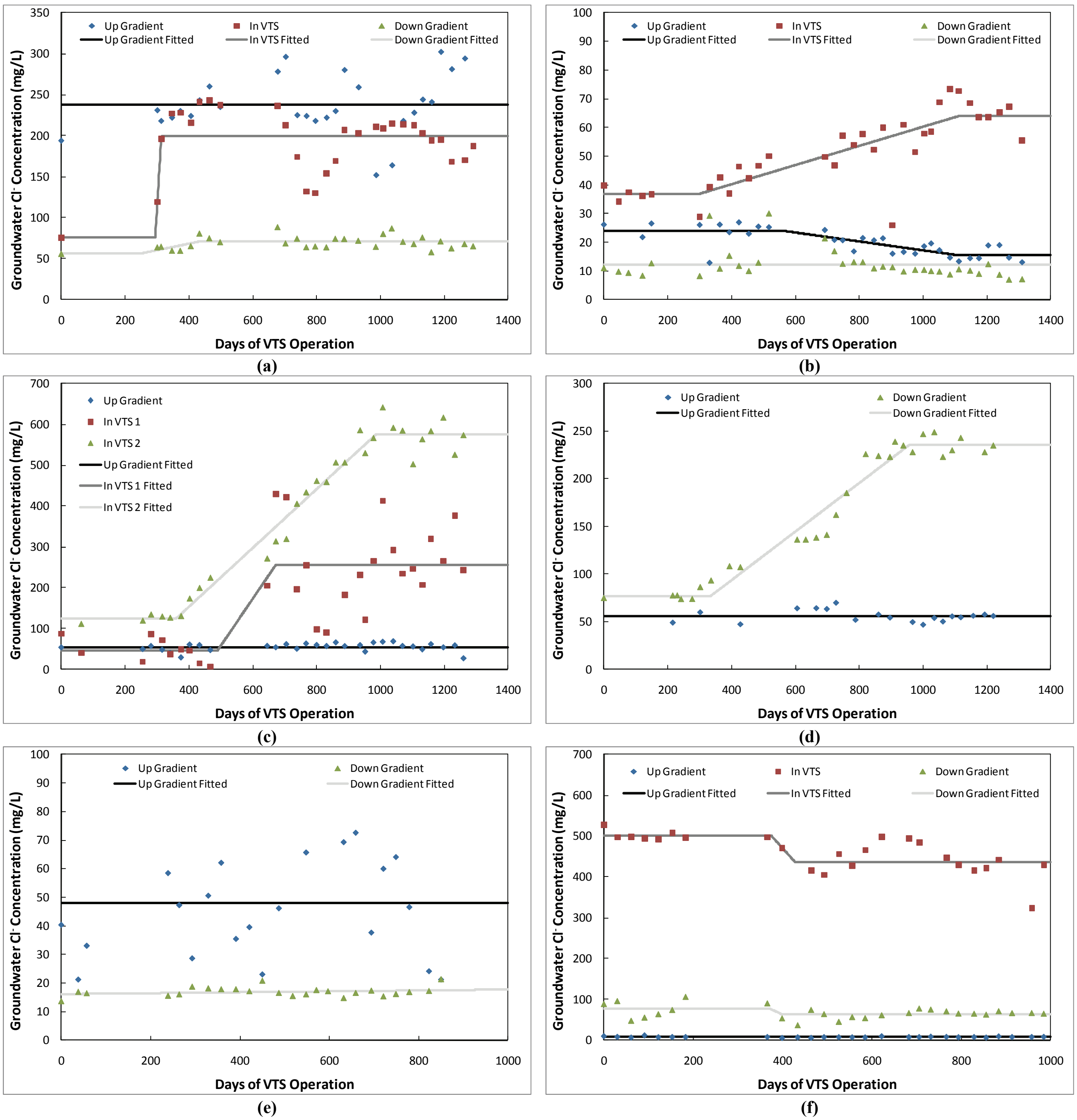

Figure 2. Groundwater chloride concentration trends at (a) Central Iowa 1, (b) Central Iowa 2, (c) Northwest Iowa 1, (d) Northwest Iowa 2, (e) Southwest Iowa 1, and (f) Southwest Iowa 2. Graphs are on different scales to make trends more evident.

and Fonstad, 2000) has shown that chloride concentrations in groundwater near feedlots are often elevated; concentrations ranging from 18 to 664 mg L^{-1}, depending on feedlot age and site conditions, have been reported.

Trends for chloride concentrations in CN IA 2 groundwater (fig. 2b) indicated decreasing concentrations at the upgradient well, no change in the downgradient well, and increasing concentrations at the in-VTS well. Concentration changes in the upgradient and in-VTS wells were significant ($p < 0.0001$). The increase (27 mg L^{-1}) at the in-VTS well indicates that wastewater was being infiltrated; however, at this site, the increase was slower than at Central Iowa 1. Although this site also had a shallow depth to groundwater, the well screen was installed in a clay layer with low permeability, which slowed chloride transport to the well and limited percolation of the applied effluent. Similar results were noted by Faulkner et al. (2011) for a New York VTS site with a shallow soil profile. The decrease in chloride concentrations at the upgradient well was unexpected; however, investigations of site conditions prior to VTS installation indicated that feedlot runoff pooled around this well location. Construction of the VTS decreased the effluent and chloride application in this area, thereby reducing the chloride loading to the groundwater near this well.

Northwest Iowa 1 (fig. 2c) had a constant chloride concentration in the upgradient well and significant ($p < 0.0001$) increases in both in-VTS wells; increases in chloride concentration were 210 and 451 mg L^{-1} at the VTS 1 and VTS 2 wells, respectively. The lag time before the VTS 1 chloride concentration started increasing was larger than the lag time for VTS 2. This was probably due to the greater depth to the water table at the VTS 1 well, resulting in increased travel time before chloride in the applied SSB effluent leached to groundwater. However, the concentration in VTS 2 stabilized after the VTS 1 concentration. Water table monitoring at this site indicated that groundwater was flowing from VTS 1 toward VTS 2; thus, the concentration at VTS 1 needed to stabilize before VTS 2, as it served as a chloride input to groundwater flowing to VTS 2. The results for Northwest Iowa 2 (fig. 2d) were similar to those for Northwest Iowa 1. Upgradient concentrations were stable over the 3.5 years of monitoring. The concentration increase in the downgradient well was again significant ($p < 0.0001$), increasing by 158 mg L^{-1}. The deeper water table at this site again delayed the time before the groundwater concentration began to respond.

Chloride concentrations in Southwest Iowa 1 (fig. 2e) groundwater remained constant. This was due to the siting of the monitoring wells. Both wells were installed upgradient of the VTS; thus, the monitoring wells did not allow the true impact of the VTS to be assessed. The chloride trends at Southwest Iowa 2 (fig. 2f) were different from those at the other locations. The concentrations in the VTS and downgradient wells both decreased significantly ($p < 0.0001$), by 64 and 15 mg L^{-1}, respectively, after initiating use of the VTS. The groundwater concentration decreases were presumably due to improved effluent distribution over the VTS. Previously, feedlot runoff at this site was allowed to pool in a grassed area below the feedlot. The VTS now spreads the applied SSB effluent over the VTA, rather than allowing unsettled feedlot runoff to pool in the location where the groundwater wells were installed. Groundwater concentrations in the upgradient well remained constant.

Table 3. Applied effluent chloride concentrations and groundwater chloride concentrations at the six study sites.

Site[a]	Cl^- Concentration (mg L^{-1}) Applied Effluent	Groundwater
CN IA 1	223	200
CN IA 2	228	64
NW IA 1	634	576
NW IA 2	430	235[b]
SW IA 1	175	180[c]
SW IA 2	525	437

[a] CN IA 1 = Central Iowa 1, CN IA 1 = Central Iowa 2, NW IA 1 = Northwest Iowa 1, NW IA 2 = Northwest Iowa 2, SW IA 1 = Southwest Iowa 1, and SW IA 2 = Southwest Iowa 2.
[b] Chloride concentration from the downgradient well.
[c] Chloride concentration in the tile lines around the VTA.

A correlation analysis was used to relate chloride concentrations monitored at the in-VTS wells (except for NW IA 2, where the downgradient well was used because no in-VTS well was available, and SW IA 1, where tile flow chloride concentrations were used) to the flow-weighted average chloride concentrations applied to the VTA. Chloride concentrations in the applied effluent (either from the SSB or VIB) were corrected for losses of chloride in VTA outflow and in harvested vegetation, and for volumes of water added from precipitation and lost to evapotranspiration prior to comparison (table 3). The correlation analysis indicated a strong relationship between the applied chloride concentration and the chloride concentration in the groundwater (Pearson's $r = 0.91$). With the exception of Central Iowa 2, where the well was located in a clay layer, groundwater chloride concentrations averaged 85% of the applied chloride concentration. At CN IA 2, chloride concentrations at the groundwater well were only 28% of the applied concentration. We hypothesize that the clay layer restricted percolation and limited the impact of effluent application on groundwater quality at this location.

Nitrate-Nitrogen

Nitrate-nitrogen (NO_3-N) trend analysis was conducted in a manner similar to that used for chloride. The Central Iowa 1 (fig. 3a) in-VTS and downgradient wells showed decreasing trends in NO_3-N concentration with time. Initial NO_3-N concentrations were 216 and 70 mg L^{-1} at these locations, respectively. After the linear decreasing trends had reached a new steady-state, the concentrations averaged 11 and 26 mg L^{-1}, respectively. This indicated that there was less NO_3-N leaching under the current land use as compared to previous conditions. During the summers of 2007 (around day 400) and 2008 (around day 800), the in-VTS groundwater NO_3-N concentration exhibited an annual peak; this trend was not noted in 2009. These peaks occurred in late summer, and we hypothesize that they indicate greater nitrate production than vegetative uptake and denitrification were capable of utilizing. The absence of a peak in 2009 may be due to greater vegetative uptake, as greater yields were obtained that year. Upgradient nitrate concentrations remained relatively constant; however, there was a period of abnormally high concentrations between day 700 and 800. This corresponded to construction of a

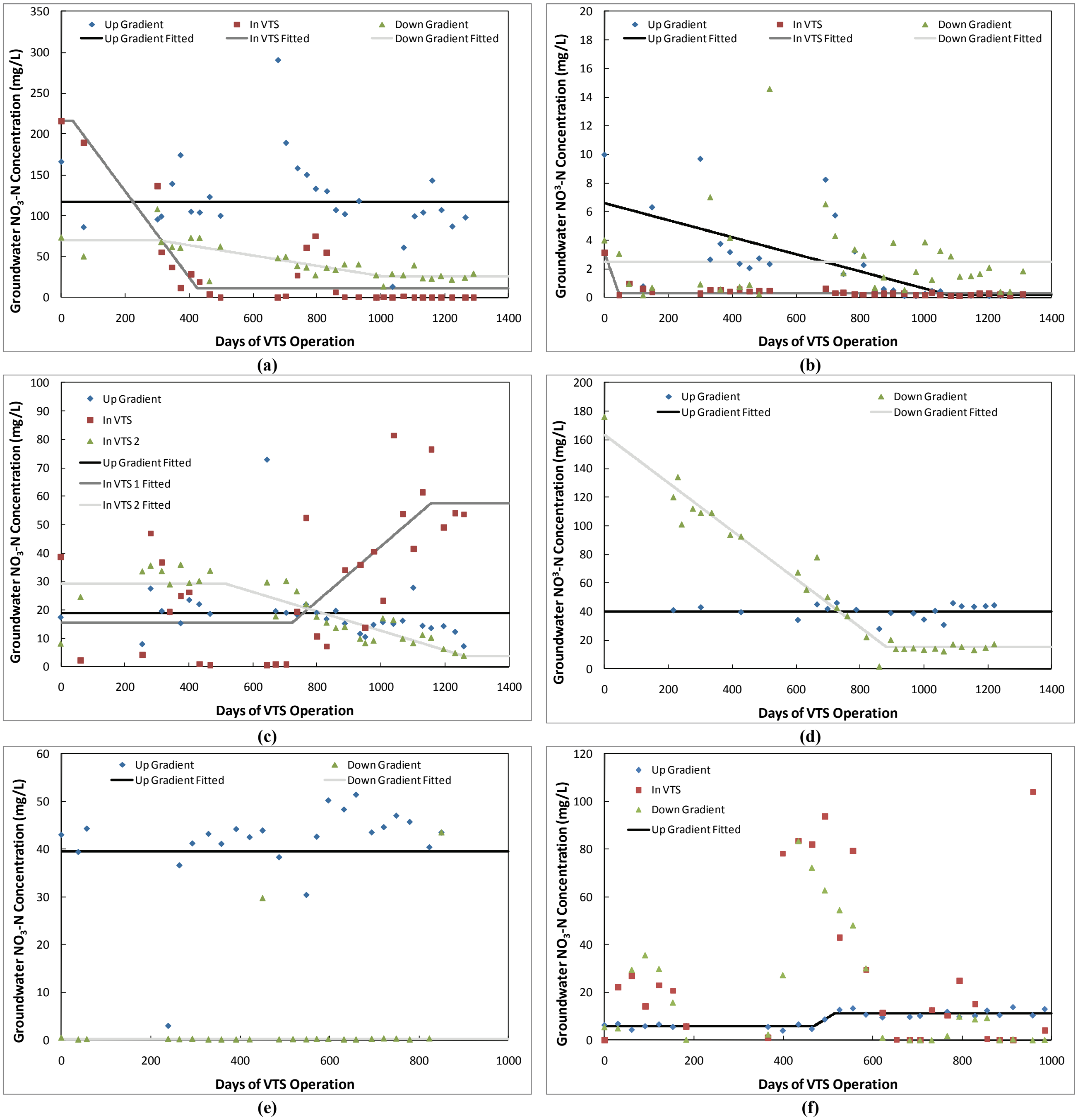

Figure 3. Groundwater nitrate-nitrogen concentration trends at (a) Central Iowa 1, (b) Central Iowa 2, (c) Northwest Iowa 1, (d) Northwest Iowa 2, (e) Southwest Iowa 1, and (f) Southwest Iowa 2. Graphs are on different scales to make trends more evident.

hoop building near the groundwater well. The construction disturbed the soil in this area and possibly mobilized nitrogen that had accumulated within the soil profile. The trends observed in the soil nitrate concentrations (fig. 4a) complement those observed in the groundwater. Prior to system operation, the nitrate concentrations in the surface soil (top 15 cm) averaged approximately 5 mg NO_3-N kg^{-1}, with increasing nitrate concentrations observed deeper in the profile (up to 15 mg NO_3-N kg^{-1} at the 94 to 122 cm depth). Two years after starting system operation, the nitrate concentrations in the surface soil (top 15 cm) were substantially higher, averaging 20 to 25 mg kg^{-1} in 2008 and 2009 (fig. 4a); however, these elevated nitrate concentrations were confined to the upper profile. At depths greater than 30 cm, the soil nitrate concentration was less than that under the previous land use (row crop agriculture).

Central Iowa 2 (fig. 3b) also shows a trend of decreasing NO_3-N in the in-VTS well. A decreasing trend in NO_3-N concentrations was also seen in the upgradient well; this corresponded with the decreasing trend in chloride seen in this well. This may indicate that installation and use of the VTS improved effluent handling over previous conditions at this location. No trend in NO_3-N concentration was seen at the downgradient well. In general, nitrate-nitrogen concentrations were consistently low (<10 mg NO_3-N L^{-1}) at this site, which contradicts observations at the other sites.

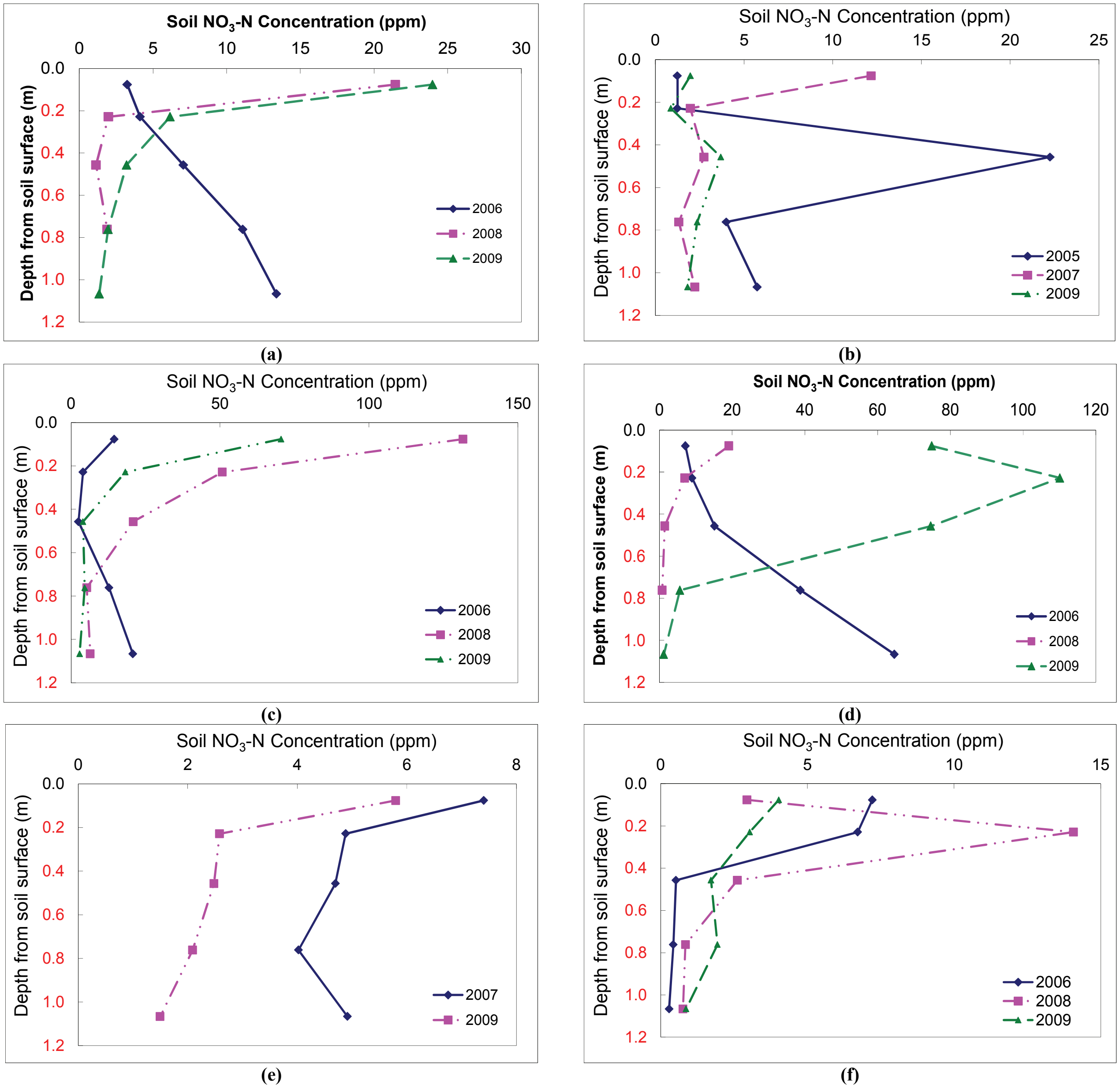

Figure 4. Soil nitrate-nitrogen concentrations as a function of depth at (a) Central Iowa 1, (b) Central Iowa 2, (c) Northwest Iowa 1, (d) Northwest Iowa 2, (e) Southwest Iowa 1, and (f) Southwest Iowa 2. Graphs are on different scales to make trends more evident.

Soil samples at this site (fig. 4b) tended to corroborate the patterns observed in groundwater, as nitrate concentrations deeper in the soil profile were again lower than under the previous land use.

Northwest Iowa 1 (fig. 3c) showed no trend in upgradient NO_3-N concentration, increasing NO_3-N concentrations in the VTS 1 well, and a decreasing trend in the VTS 2 well. At this site, the VTAs contributing flow to VTS 1 are located at a higher elevation than the VTA contributing to VTS 2. We observed that this led to fewer saturated soil conditions. This soil water profile could lead to conditions that encourage nitrification and possibly limit denitrification opportunities. The VTS 2 well was positioned below a VTA that was lower in elevation, had a shallower groundwater table, and stayed consistently wetter. The wetter conditions are more favorable for denitrification and could potentially limit nitrification. Monitoring of soil nitrate concentrations at this site showed a similar pattern to those observed at CN IA 1, i.e., high nitrate concentrations in the surface soil but low concentrations deeper in the profile (fig. 4c).

Northwest Iowa 2 (fig. 3d) groundwater trends were similar to those observed at NW IA 1; nitrate concentrations remained constant in the upgradient well, while the downgradient well showed a consistent trend of decreasing NO_3-N concentrations. At the downgradient well, NO_3-N levels were initially monitored to be 164 mg L^{-1}. At the end of the 3.5 years of monitoring, the nitrate-nitrogen concentrations had stabilized at 15 mg L^{-1}. This indicates that the VTA reduced nitrate leaching concentrations as compared to the previous land use (row crop production). The trends in soil nitrate concentrations (fig. 4d) shared some similarities to those observed at other sites, but in this case elevated nitrate concentrations (>80 mg NO_3-N kg^{-1}) down to a depth of 60 cm were observed in 2009. This may indicate the potential of nitrogen movement through the soil profile; however, below this depth, concentrations dropped rapidly to average values lower than observed for the previous land use.

No trends in NO_3-N groundwater concentrations were seen at Southwest Iowa 1 (fig. 3e). This was attributed to the monitoring well siting around the feedlot, rather than around the treatment system. Soil nitrate concentrations (fig. 4e) exhibited a small decrease in nitrate-nitrogen concentrations throughout the soil profile, but in both cases (before and after system use) the actual nitrate concentrations observed were relatively low, averaging between 2 and 8 mg NO_3-N kg^{-1}. Southwest Iowa 2 showed a small but significant ($p < 0.001$) increase in NO_3-N concentration at the upgradient well (fig. 3f). Model fits were extremely poor for the in-VTS and downgradient wells; as such, the models are not shown. These two wells exhibited a sinusoidal pattern, with maximum NO_3-N concentrations occurring during the summer and minimum concentrations occurring in the winter, similar to the seasonal pattern of effluent application. This seems to indicate that during the warmer, drier, summer months larger amounts of the applied ammonium and organic nitrogen were being nitrified, increasing the leaching potential. During the winter and spring, nitrate-nitrogen concentrations dropped to levels near the detection limit. Groundwater level monitoring at this site indicated the presence of a seasonal high water table. This led to saturation of the soil profile in the spring. During the summer, the water table dropped rapidly. Soil sampling (fig. 4f) showed that the deeper soil profile exhibited small increases in nitrate concentration (~1 to 2 mg NO_3-N kg^{-1}) as opposed to the decreases in nitrate observed at the other locations.

Effect of VTS on Groundwater Quality

Ammoniacal Nitrogen

Most of the groundwater samples were at or below the ammoniacal nitrogen detection limit of 0.20 mg NH_3-N L^{-1}. When means and standard deviations were calculated, all samples that were reported as below the detection limit were assumed to be at the detection limit (alternative assumptions of zero or half of the detection limit did not impact statistical results). Means, standard deviations, and significant differences between well locations are summarized in table 4. The majority (>90%) of samples at CN IA 1, NW IA 2, and SW IA 1 were at or below the detection limit (0.20 mg L^{-1}). At NW IA 1, more than 80% of samples were below the detection limit, and no significant differences in ammoniacal nitrogen concentrations were detected. At SW IA 2, ammoniacal nitrogen was not detected at the upgradient well; the in-VTS and downgradient wells were significantly different from the upgradient well. The higher levels were present at the start of the study and may indicate previous contamination of the shallow groundwater. The VTS area had received runoff from the feedlot for more than 30 years; thus, the higher levels can probably be attributed to historic ammonium and organic nitrogen accumulation in the soil. At CN IA 2, ammoniacal nitrogen was rarely (~20% of the time) detected at the in-VTS well. The wells upgradient and downgradient of the feedlot were above the detection limit more than 95% of the time. At this site, all wells were significantly different from

Table 4. Means (and standard deviations) of ammoniacal nitrogen concentrations in the upgradient, vegetative treatment system (VTS), and downgradient monitoring wells at the six study sites.[a]

Site[b]	Well Location		
	Upgradient	In VTS	Downgradient
CN IA 1	0.22 (0.07) a	0.21 (0.07) a	0.20 (0.00) a
CN IA 2	3.02 (1.02) a	0.23 (0.07) b	1.37 (0.69) c
NW IA 1	0.36 (0.82) a	0.69 (2.30) a	0.51 (0.93) a
NW IA 2	0.20 (0.00) a	NA	0.21 (0.06) a
SW IA 1	0.21 (0.02) a	NA	0.21 (0.03) a
SW IA 2	0.20 (0.00) a	1.08 (2.05) b	0.65 (0.56) b

[a] Within a row, means followed by different letters are significantly different at the $\alpha = 0.05$ level.

[b] CN IA 1 = Central Iowa 1, CN IA 1 = Central Iowa 2, NW IA 1 = Northwest Iowa 1, NW IA 2 = Northwest Iowa 2, SW IA 1 = Southwest Iowa 1, and SW IA 2 = Southwest Iowa 2.

each other. The upgradient well had the highest concentrations, and the in-VTS well had the lowest concentrations.

Overall, the groundwater monitoring results seem to indicate that the use of VTSs did not increase ammoniacal nitrogen concentrations in the groundwater; however, the deep soil sampling within the VTAs provided significantly more insight into what may be occurring. At many of the sites, soil ammonium-nitrogen concentrations were elevated compared to the previous land use (fig. 5). This result is consistent with lands that are frequently dosed with nitrogen-rich wastewaters. At two locations (CN IA 1 and NW IA 2) increases (~15 and 28 mg NH_4-N kg^{-1}, respectively) in soil ammonium content were observed deeper (below 0.6 m) in the soil profile. At both sites, this trend of increased ammonium was only observed in cores collected near the VTA inlet. These soil cores tended to have high nitrate contents in the surface and then low concentrations lower in the profile. The opposite trend observed for ammonia.

SW IA 2 exhibited a trend in ammonium-nitrogen concentrations that was opposite the trend seen at CN IA 1 and NW IA 2. As discussed previously, the in-VTS well at SW IA 2 had numerous groundwater samples that were above the ammoniacal nitrogen detection limit, but we speculate that this was due to previous contamination. Soil samples support this hypothesis, as high levels (200 to 250 mg NH_4-N) were observed at the lower depths in the soil profile during collection of the background soil sample. After several years of VTS operation, these levels had decreased below 50 mg NH_4-N kg^{-1}.

Chloride

Chloride concentrations at the in-VTS wells were higher than at the upgradient wells (table 5). As the applied wastewater had high concentrations of chloride, this serves as an indicator of manure infiltration into the VTA. At CN IA 1, chloride concentrations at the upgradient well were significantly higher than at the in-VTS and downgradient wells. This well was located near the feedlot, and the high concentrations are probably influenced by this location (Olson et al., 2005; Maule and Fonstad, 2000). At SW IA 1, chloride concentrations at the downgradient well were significantly lower than at the upgradient well, but the proximity of the well with respect to the feedlot limits the use of the data. All other locations exhibited statistically significant increases in chloride concentrations at the in-VTS and downgradient wells.

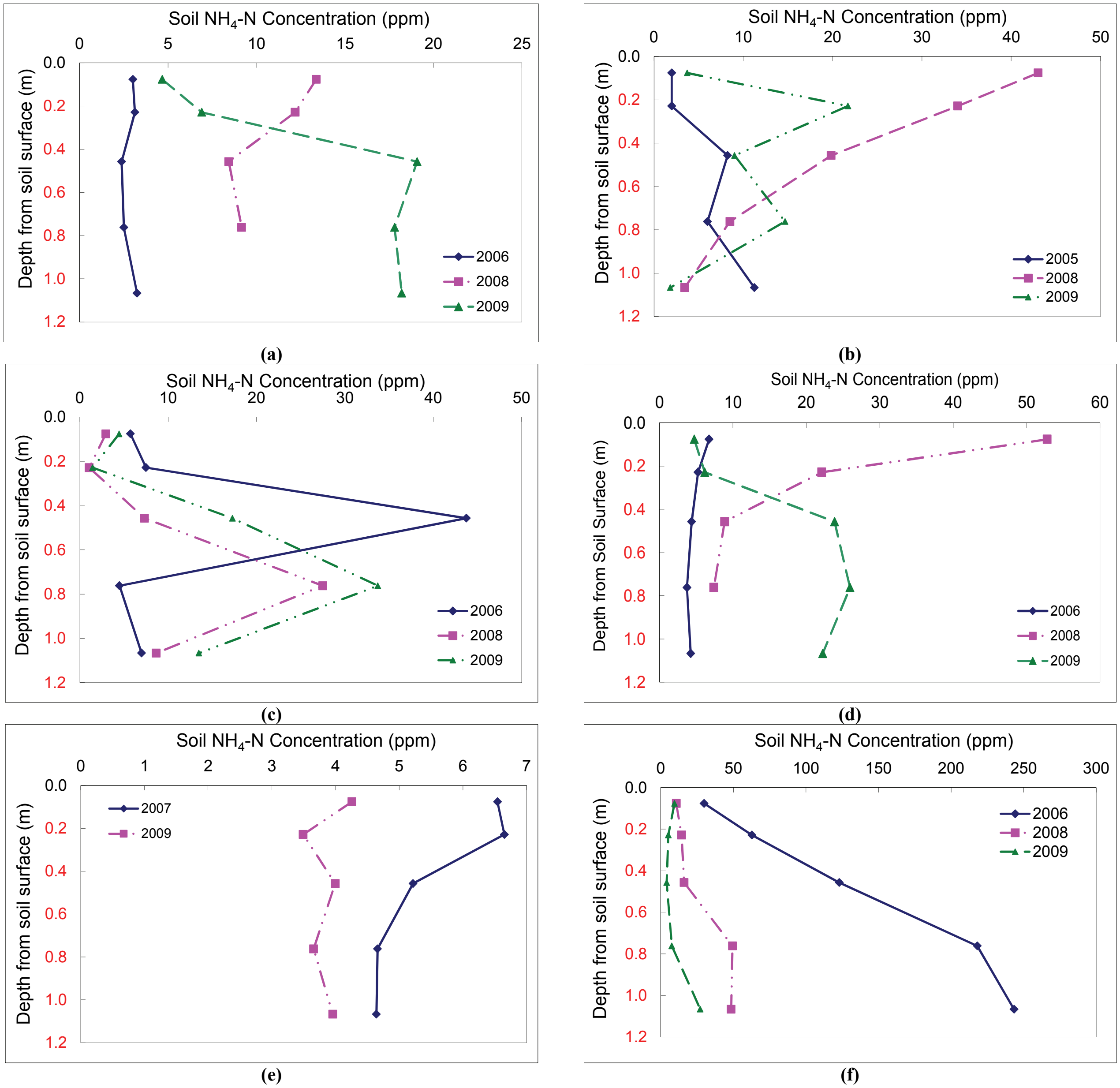

Figure 5. Soil ammonium-nitrogen concentrations as a function of depth at (a) Central Iowa 1, (b) Central Iowa 2, (c) Northwest Iowa 1, (d) Northwest Iowa 2, (e) Southwest Iowa 1, and (f) Southwest Iowa 2. Graphs are on different scales to make trends more evident.

Table 5. Means (and standard deviations) of chloride concentrations in the upgradient, vegetative treatment system, and downgradient monitoring wells at the six study sites.[a]

	Well Location		
Site[b]	Upgradient	In VTS	Downgradient
CN IA 1	273 (36) a	200 (30) b	71.0 (7.0) c
CN IA 2	15.4 (2.9) a	64.0 (7.1) b	12.1 (5.2) c
NW IA 1	54.4 (9.8) a	256 (82) b	576 (31) c
NW IA 2	55.8 (6.4) a	NA	235 (13) b
SW IA 1	48.1 (24.7) a	NA	17.7 (1.7) b
SW IA 2	8.14 (1.26) a	437 (34) b	63.2 (13.9) c

[a] Within a row, means followed by different letters are significantly different at the $\alpha = 0.05$ level.

[b] CN IA 1 = Central Iowa 1, CN IA 1 = Central Iowa 2, NW IA 1 = Northwest Iowa 1, NW IA 2 = Northwest Iowa 2, SW IA 1 = Southwest Iowa 1, and SW IA 2 = Southwest Iowa 2.

Nitrate-Nitrogen

Nitrate-nitrogen concentration differences were site-specific. At most sites (except CN IA 2), nitrate-nitrogen concentrations exceeded the 10 mg NO_3-N L^{-1} drinking water standard (table 6). At CN IA 1, the average concentration in the upgradient well was 117 mg L^{-1}. As mentioned earlier, this well was located near the feedlot, which may have impacted the monitored groundwater concentrations. Similarly, Maule and Fonstad (2000) reported concentrations ranging from 2.5 to 233 mg NO_3-N L^{-1} in wells around feedlots. At CN IA 2, the downgradient well had a significantly ($p < 0.0001$) higher average concentration than either the upgradient or in-VTS wells; however, the actual concentrations were relatively low, with an average of 2.52 mg L^{-1}. At NW IA 1, the NO_3-N concentrations were lowest at the VTS 2 well (shown as downgradient in table 5) and highest at the VTS 1 well. As discussed previ-

Table 6. Means (and standard deviations) of nitrate-nitrogen concentrations in the upgradient, vegetative treatment system (VTS), and downgradient monitoring wells at the six study sites.[a]

Site[b]	Well Location		
	Upgradient	In VTS	Downgradient
CN IA 1	117 (53) a	11.0 (23.1) b	26.0 (12.4) b
CN IA 2	0.18 (2.03) a	0.33 (0.19) a	2.52 (2.74) b
NW IA 1	19.0 (11.3) a	57.7 (16.6) b	3.80 (5.56) c
NW IA 2	40.3 (5.1) a	NA	15.3 (8.5) b
SW IA 1	39.6 (12.5) a	NA	0.18 (10.51) b
SW IA 2	11.4 (1.2) a	33.6 (32.1) b	2.72 (19.4) a

[a] Within a row, means followed by different letters are significantly different at the $\alpha = 0.05$ level.

[b] CN IA 1 = Central Iowa 1, CN IA 1 = Central Iowa 2, NW IA 1 = Northwest Iowa 1, NW IA 2 = Northwest Iowa 2, SW IA 1 = Southwest Iowa 1, and SW IA 2 = Southwest Iowa 2.

Table 7. Log-transformed means (and standard deviations) of fecal coliform concentrations in the upgradient, vegetative treatment system (VTS), and downgradient wells at the six study sites.[a]

Site[b]	Well Location		
	Upgradient	In VTS	Downgradient
CN IA 1	2.24 (1.37) a	2.63 (1.06) a	1.55 (0.69) b
CN IA 2	1.66 (0.84) a	1.21 (0.49) b	1.87 (0.89) a
NW IA 1	1.28 (0.44) a	1.71 (0.65) b	1.60 (0.74) ab
NW IA 2	1.30 (0.67) a	NA	1.47 (0.69) a
SW IA 1	1.47 (0.79) a	NA	1.93 (1.09) a
SW IA 2	1.49 (0.56) a	3.70 (1.59) b	2.17 (0.94) c

[a] Within a row, means followed by different letters are significantly different at the $\alpha = 0.05$ level.

[b] CN IA 1 = Central Iowa 1, CN IA 1 = Central Iowa 2, NW IA 1 = Northwest Iowa 1, NW IA 2 = Northwest Iowa 2, SW IA 1 = Southwest Iowa 1, and SW IA 2 = Southwest Iowa 2.

ously, VTS 1 was sited at a location with greater depth to groundwater than VTS 2. As a result, conditions in the VTS 1 well were drier and presumably favored nitrification and limited denitrification. The VTS 2 well was located below another VTA that was lower in elevation and stayed much wetter; potentially favoring denitrification. Similarly, downgradient NO_3-N concentrations at NW IA 2 were significantly lower than upgradient concentrations. At SW IA 2, concentrations in the VTS were significantly higher than either upgradient or downgradient. Both the in-VTS and downgradient wells had high amounts of variability in NO_3-N concentrations; this was caused by the seasonal trend of higher NO_3-N concentrations in summer and lower concentrations in winter. Overall, it appears that the VTSs did not cause significant increases in groundwater NO_3-N concentrations. In some cases, they even reduced NO_3-N levels as compared to the pre-VTS conditions, although seasonal trends of high NO_3-N concentrations in the summer were seen at several locations. More research is required to determine the mechanisms that cause these trends in NO_3-N concentrations.

Fecal Coliforms

All fecal coliform concentrations were log-transformed prior to statistical analysis (table 7). With the exception of Central Iowa 2, fecal coliform concentrations were highest at the in-VTS wells. At Central Iowa 2, the monitoring well was installed in a clay layer that slowed percolation and reduced transport of contaminants to groundwater, similar to the function of the fragipan described by Faulkner et al. (2011) for their New York VTS site. At most sites (CN IA 1, CN IA 2, NW IA 1, NW IA 2, and SW IA 1), concentrations at the upgradient and downgradient wells were not significantly different.

Estimated Leaching of Chloride and NO_3-N

The methods described previously were used to calculate the average volumes of water and masses of chloride and nitrate leached. The results are summarized in table 8. In general, the calculated masses of leached chloride were 30% to 85% of the applied chloride masses, with another 5% to 20% being removed with harvested vegetation. Based on our calculations, this approach accounted for 80% to 100% of the applied chloride, although at CN IA 2 and NW IA 2 it only accounted for 30% and 55% of the applied chloride, respectively. Although not perfect, this level of tracing provides strong evidence that the leaching estimates are reasonable. Following the same methodology, NO_3-N leaching was estimated to range from 2 to 140 kg NO_3-N ha^{-1} year^{-1} (table 8). At SW IA 1, where tile lines surrounded the VTA, approximately 14 kg NO_3-N ha^{-1} year^{-1} was monitored in the tile flow. This estimate is reasonable in comparison to the estimated leached masses of nitrogen occurring at four of the sites. In general, these results are similar to those for tile-drained fields under a corn-soybean rotation in the upper Midwest. For instance, results from the Midwest have ranged from 0 to 50 kg NO_3-N ha^{-1} (Randall et al., 1997; Randall et al., 2003; Randall and Vetsch, 2005), while Bahksh et al. (2005, 2006) found losses in Iowa of 11 to 14 kg N ha^{-1}. These nitrogen leaching losses only account for a small portion, i.e., 0.1% to 16%, of the applied nitrogen at these sites. In this regard, SW IA 2 is an outlier, as the estimated nitrate leaching at

Table 8. Volumes and masses of chloride and nitrate-nitrogen estimated to be leached by the vegetative treatment areas at the six study sites based on long-term hydraulic balances and monitored groundwater concentrations.

Site[a]	Water				Chloride		NO_3-N	
	Precipitation (cm year^{-1})	Inflow (cm year^{-1})	Outflow (cm year^{-1})	Leached[b] (cm year^{-1})	Leached[c] (kg ha^{-1} year^{-1})	% of Applied	Leached[c] (kg ha^{-1} year^{-1})	% of Applied
CN IA 1	95	56	33	36 ±4	700 ±100	90	39 ±20	1
CN IA 2	103	103	62	62 ±6	400 ±50	30	2.0 ±0.4	>0.1
NW IA 1	67	71	9	58 ±6	3300 ±400	91	22 ±8	0.2
NW IA 2	73	111	47	67 ±7	1600 ±200	55	15 ±3	0.2
SW IA 1[d]	97	50	22	13	250	83	14	0.5
SW IA 2	100	64	16	42 ±4	1800 ±200	99	140 ±40	16

[a] CN IA 1 = Central Iowa 1, CN IA 1 = Central Iowa 2, NW IA 1 = Northwest Iowa 1, NW IA 2 = Northwest Iowa 2, SW IA 1 = Southwest Iowa 1, and SW IA 2 = Southwest Iowa 2.

[b] Uncertainty assumes 10% error in the SPAW-modeled leaching volume; Andersen et al. (2010) reported 8% bias in estimated VTA outflow.

[c] Uncertainty assumes 10% error in leaching volume and SEM of chloride and NO_3-N concentrations.

[d] Leached volume and masses estimate based on monitored tile flow measurements.

this site was substantially larger than at the other sites. This corresponds to the high concentration peaks in groundwater seen during the summer at this site and with the soil samples showing an increase in nitrate concentration at lower soil depths as compared to the samples collected prior to system operation.

CONCLUSIONS

A trend analysis was conducted to evaluate groundwater chloride and nitrate response patterns to VTS construction and use for treatment of feedlot runoff. In general, groundwater below the VTS exhibited trends of increasing chloride concentrations and decreasing nitrate concentrations. No trends for fecal coliforms or ammoniacal nitrogen were seen. Statistical analysis was performed to test for differences between upgradient, in-VTS, and downgradient monitoring wells. In general, no differences in ammoniacal nitrogen concentration were seen, with most samples being below the ammonia-nitrogen detection limit. Fecal coliform concentrations were generally highest within the VTS monitoring well but showed no difference between upgradient and downgradient concentrations. Chloride concentrations were generally significantly higher within and downgradient of the VTS when compared to the upgradient well; nitrate concentrations were generally significantly lower at these locations. A water balance model was then used to estimate the volumes of water that were leached, which was used to estimate chloride and nitrate leaching. In general, the results suggested that 30% to 99% of the chloride was in the leachate, but only 0.1% to 16% of the applied nitrogen was leached. Nitrate-nitrogen leaching masses were estimated to range from 2 to 140 kg ha^{-1}; these values are similar to those reported for corn-soybean rotation tile drainage in Iowa and suggest that more study is needed to better understand the fate of the applied nitrogen.

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