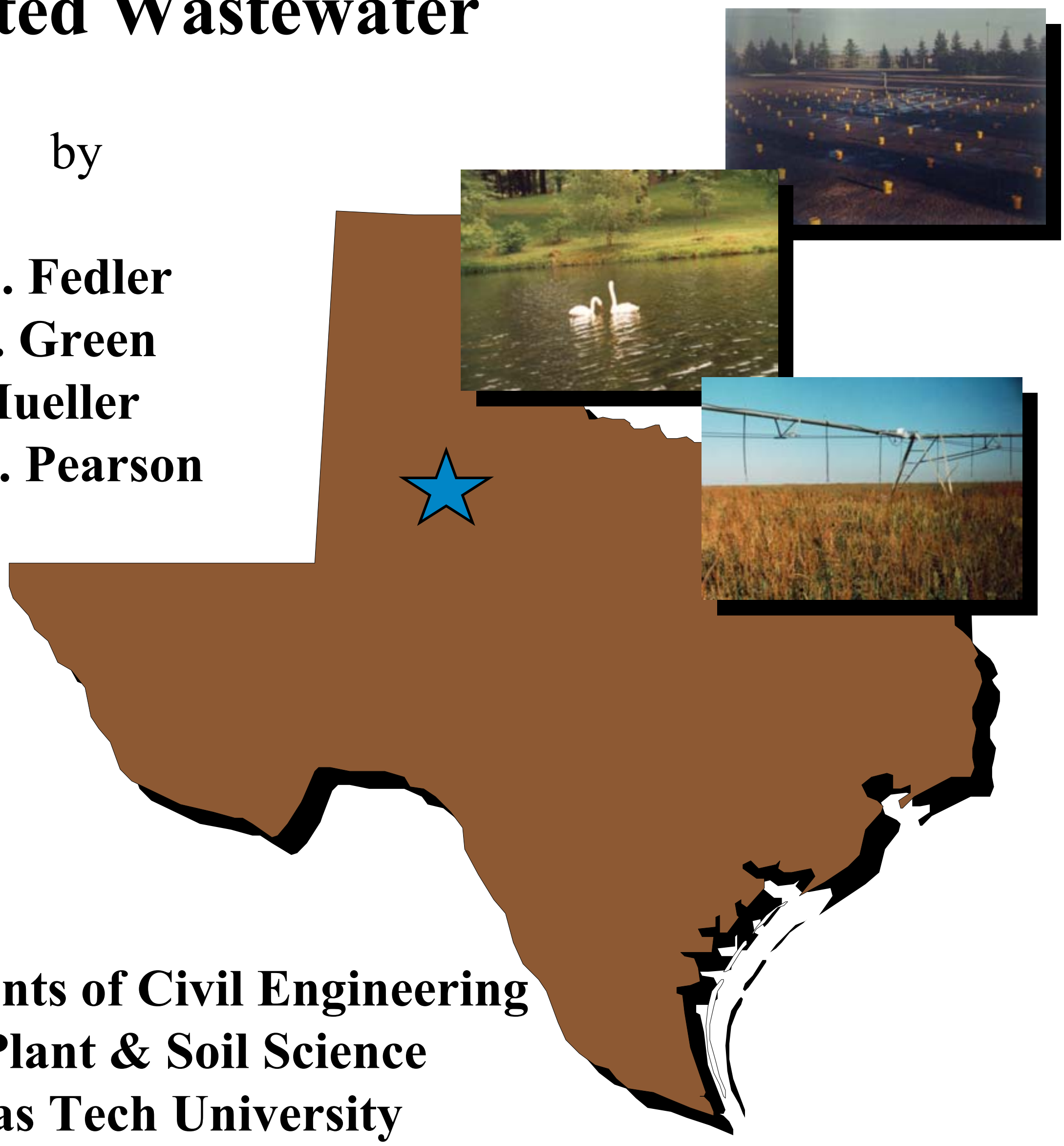
Denitrification for Land Applied Treated Wastewater



by

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SUMMARY

Even though the irrigation of wastewater on land has been officially recognized as a mechanism to complete the treatment of wastewater since the late 1800's, its real acceptance never occurred until about the 1970's. Rejection of land treatment of wastewater as a viable treatment alternative was due to incomplete understanding of the fate of nitrogen within the soil-wastewater system. Current research indicates a broad understanding of the nitrogen cycle, with nitrification as the best understood component. Denitrification, however, is less well defined, but is expanding. When it comes to the denitrification process, the understanding is still minimal because variations in the rate of denitrification of over 300 percent are not uncommon, as was shown in this research.

INTRODUCTION

Land treatment of municipal wastewater was introduced in the United States in 1872, yet by 1910 it was still not considered to be an acceptable treatment alternative. Discharge of partially treated municipal wastewater into receiving waters was believed to be a safe and cost effective alternative to land application. By 1930, land treatment was considered to be an uncontrolled natural process, and was classified in the same area as the purification that occurs in discharges or disposal to streams (Jewell and Seabrook, 1979).

Pollution of surface and groundwater in the United States became apparent during the 1950s causing the need for PL92-500, the "Clean Water Act," be created in an effort to improve the water quality in the U.S. Municipal wastewater treatment facilities consisting of unit operations and unit processes were essentially the technologies of choice. By the middle of the 1970s, it was apparent that mechanical methods were capital intensive and that the quality of treated effluent was inferior to that produced by land treatment systems. Around the mid-1970's, many states mandated that the use of land treatment be considered as an alternative when new treatment facilities were planned.

Wastewater application rates in the West Texas area can range from 2.5 to 8 inches per month throughout the year with frequencies of application being nearly 20 days per month in order to achieve the desired application of water. During the summer months, the water in the soil profile of the root zone is seldom at field capacity, thus the nitrogen applied is either taken up by the plant or is denitrified. During the winter months, the soil within the root zone of the plant is always near or at saturation as a result of irrigation in order to leach accumulated salts from the root zone. During this winter period of leaching, plant uptake of nitrogen is reduced as compared to the summer. Due to potential for a high nitrate contamination to exist in the soil water, high nitrogen removal rates are desirable, and denitrification can be an important nitrogen removal mechanism (Monnett et al., 1995) by reducing the nitrate concentration in the deep percolation water.

Denitrification is a process by which nitrate undergoes dissimilatory reduction to NO, N_2O, or N_2 (Robertson, 2000). Under anaerobic conditions, many soil microorganisms respire nitrate, which is used as a terminal electron acceptor (Robertson, 2000). Denitrification is an important sink for soil nitrate. Additionally, denitrification can be an important source of atmospheric trace gases (Robertson, 2000), and the increasing concentration of atmospheric N_2O is an important component of global climate change (IPCC, 1996). Denitrification is influenced by soil moisture, temperature, nitrate concentration, and amounts of readily available carbon (Tiedje, 1988). Irrigation can influence denitrification by changing the extent and duration of anaerobic conditions as well as nitrate availability (Monnett et al., 1995).

NITROGEN IN IRRIGATED WASTEWATER SYSTEMS

Prior to 1890, process efficiencies were generally evaluated on their capability to remove nitrogen in the "albuminoid" or organic form. The operating hypothesis was that nitrification represented a fermentation process (Lloyd, 1993), which was responsible for purifying wastewaters, and that adequate treatment could be judged on the basis of the completeness of conversion of nitrogen to oxidized forms. The oxidation of nitrogen was believed to be the result of a "burning," which occurred due to the presence of oxygen (Jewell and Seabrook, 1979).

The evaluation approach described above was considered valid at a time prior to the definition of the nitrogen cycle. Scientific investigations concerning the role of microorganisms in nitrification and denitrification were in progress, but results of the research had not yet been applied to practical situations. Site managers knew of only one of the two major components of the nitrogen cycle, and did not realize the potential for contamination of groundwater by nitrate, which is one of the oxidized forms of organic nitrogen.

In the early years of irrigating with wastewater, the understanding was that decaying plant and animal material pour nitrogen into the atmosphere. This research was confirmed in 1886, and the modern era of study into nitrification and denitrification began (Lloyd, 1993; Payne, 1986). Schloesing and Muntz (Payne, 1986) established the bacterial etiology of nitrification in 1877. More recent investigation revealed that nitrification is the result of chemical and biological processes mainly attributable to two specific bacteria, *Nitrosomonas* and *Nitrococcus*.

The importance of the discovery of the nitrogen cycle to the agricultural and the sewage treatment communities would not become apparent for nearly fifty years. In 1940, methemoglobinemia (Sawyer et al., 1984), a blood disorder primarily affecting infants (NIANR, 1998), was discovered in the United States. Research into the illness discovered that methemoglobinemia is directly related to consumption of drinking water containing high concentrations of nitrate (Canter, 1997). Muchovej and Rechcigal (1994) report that nitrate contamination of drinking water may be linked to development of gastric cancer, non-Hodgkin's lymphoma, increased infant mortality, central nervous system birth defects, and hypertension.

While initially attributed to excessive fertilizer use by farmers, the discovery of nitrate contamination, and its harmful effects in humans, had a direct impact on land treatment systems. Prior to the discovery of methemoglobinemia and its cause, hydraulic loading rates, crop requirements, soil conditions, and the nitrogen cycle were not always considered simultaneously when irrigation rates were set at land treatment sites. Uncontrolled land application of partially treated municipal wastewater caused nitrate contamination of groundwater and created contaminated groundwater mounds (Fedler, 2000).

Modern land treatment sites are developed and operated under regulations adopted by the EPA and by state regulators. Designers are now aware that compromises between engineering efficiency and agricultural requirements are required, and that site conditions must be closely monitored to prevent groundwater contamination (Fedler and Borrelli, 1995).

Recently, many municipalities applying for new wastewater treatment permits are investigating some type of water reuse plan. This appears to be an attempt to reduce their dependence on groundwater supplies for irrigating crops and turf grasses. Since agriculture in many states use well over 50% of the state's fresh water resources, it seems appropriate that alternative sources of water, such as recycled wastewater, be investigated for their potential agricultural usage. Treated wastewater can provide a safe and viable alternative for the production of crops while reducing the demand on limited groundwater resources. One problem with the application of wastewater to the land surface where a crop is grown is our limited understanding of the fate of the nitrogen within the soil treatment system. There is little doubt that a large portion of the applied nitrogen is utilized by the crop and small amounts will be leached with the deep percolation water required for leaching the salts in the water. But the problem faced by engineers and scientist designing land application systems and surface applied on-site systems today is the denitrification process. This lack of understanding of the nitrification and denitrification processes has a significant boomerang effect on the operations of irrigated wastewater systems.

The denitrification process has the potential to remove as little as no nitrogen to as much as 80% of the nitrogen applied (USEPA, 1981). This variation in denitrification rate is due to the amount of moisture present in the soil and the temperature at which the soil mirocflora can complete the biological process. If the denitrification losses within the soil are at a soil's microfloras highest capacity, then the crop may suffer from a lack of available nitrogen, thus reducing the total treatment potential of the land application system. If the crop is not receiving the required amount of nitrogen, then the total biomass production is reduced. As the biomass production is reduced, so is the water uptake potential of the crop. If the crop water uptake is reduced, the potential for saturated soil conditions will increase thus further reducing the potential for producing a crop. All of this leads to a failed surface application system.

The objectives of this study are to evaluate the effect of wastewater application on the fate of nitrogen within the soil. More specifically the denitrification rates and the denitrification capacity of selected soils under field conditions where an actively growing crop is present were analyzed.

METHODOLOGY

The field site for this research is the City of Lubbock Land Application Site (LLAS), which has used land application as a part of its municipal wastewater treatment system for nearly 80 years. The LLAS began as a 200-acre farm in 1925, and by early 2001, the site contained nearly 6000 acres (Fedler 2000). Pivot irrigation systems are primarily used to apply secondary effluent to the land area of which totals about 2500 acres (Figure 1). Average effluent flow has increased from 1 MGD in 1925 to 12.75 MGD in 2001. During the early years, effluent was applied by furrow and border irrigation. The current operation uses self-propelled pivot irrigation systems to apply treated wastewater to over 30 plots ranging in area from 19 to 190 acres.

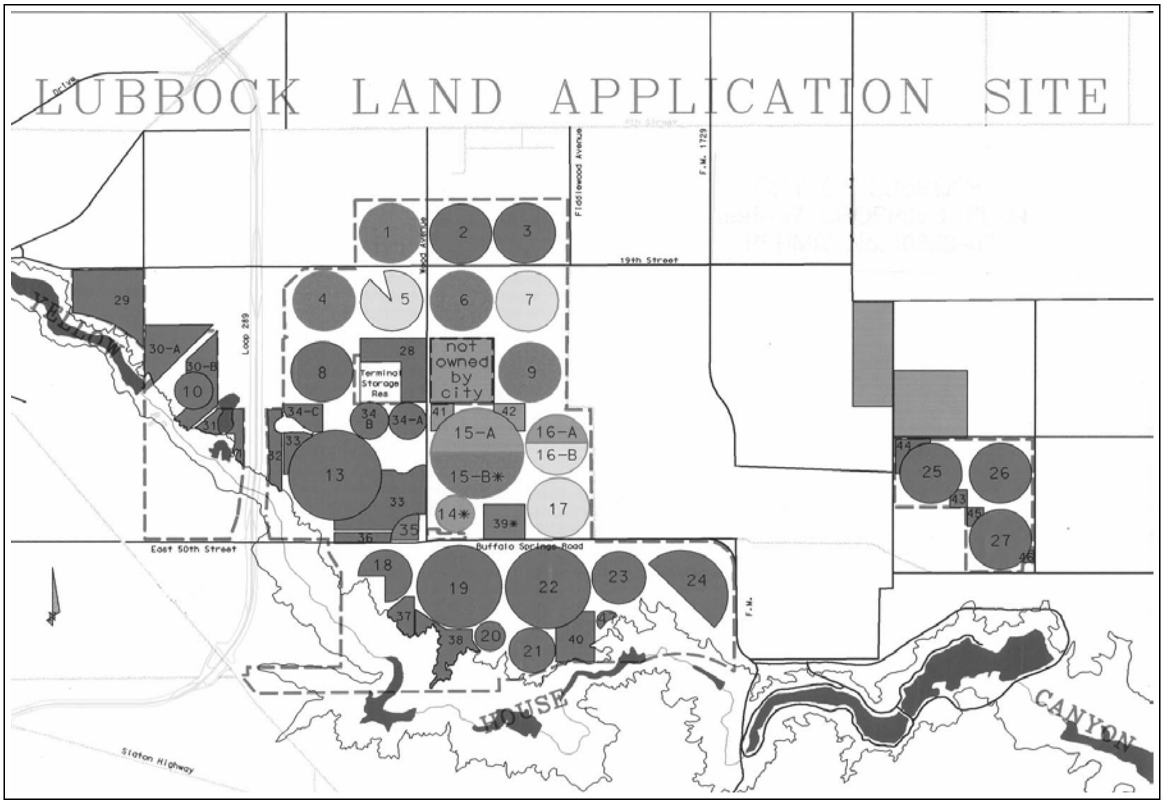

Figure 1. Layout of the site used to test the fate of nitrogen in the irrigated wastewater. Plots 3, 16, and 35 were used for the tests sites.

Nitrogen removal is the result of processes occurring in the storage reservoir and the soil matrix, from the harvest of crops, and through a cattle-grazing operation. Alfalfa (*Medicago sativa)*, winter wheat (*Triticum sp.*), Wheat Grass (A*gropyron sp.*), Bermuda grass (*Cynodon dactylon*), hay grazer (*Cynodon sp.)*, Italian Rye Grass (*Lolium multiflorum*), and native range grasses are typical crops grown on the LLAS.

Denitrification Rates

Evaluating denitrification under field conditions was modeled after the soil core method for measuring denitrification described by Mosier and Klemedtsson (1994) and Welzmiller (2001). Cores (15 cm long by 3.18 cm diameter) were collected at 10 locations within 3 different soil series at the Lubbock Land Application Site (LLAS) during the months of August, January, and May. The soils were Amarillo fine sandy loam from Plot number 3 (Fine-loamy, mixed, superactive, thermic Aridic Paleustalfs), Acuff loam from Plot number 16 (Fine-loamy, mixed, superactive, thermic

Aridic Paleustolls), and Estacado clay loam from Plot number 35 (Fine-loamy, mixed, superactive, thermic Aridic Paleustolls). Plot number 3 was planted to Bermuda grass only whereas Plots 16 and 35 were planted to a mix of Bermuda Grass and Italian Rye Grass. The core sampler contained an inner plastic sleeve, which had been perforated with numerous 2-mm diameter holes (Mosier and Klemedtsson, 1994). Following sampling, the plastic sleeve was removed from the core sampler. The intact cores and sleeves were then placed into 1 L incubation jars fitted with a rubber septa. The lid of the jar had been treated with "Grip Guard" to form a tight seal (Welzmiller, 2001). After sealing the jars, 40 mL of air was drawn out of each of the incubation jars and replaced with 40 mL of acetylene to reach a final concentration of 7 to 10% (v/v) depending on the moisture content of the core (Welzmiller, 2001). Once 40 mL of acetylene was introduced, the gas was mixed with the gas in the macro pores by alternatively reducing and increasing the pressure by pumping with a large gas tight syringe. The cores were then placed into holes in the ground near where they were collected and were incubated under ambient conditions for 24 hours. Following incubation, the N_2O collected was quantified by using gas chromatography.

To simulate the effects of irrigation, 5 cores were collected from each soil type as described previously. Each core was saturated with effluent water obtained from the three plots. The soils were treated with acetylene and incubated under laboratory conditions for 24 hour periods corresponding to 1, 2, 6, and 7 days following saturation with the wastewater effluent. Nitrous oxide-N was determined by using gas chromatography.

Denitrification Capacity

Denitrification capacity is being determined by using procedures similar to those described by Myrold and Tiedje (1985). Briefly, a 10 g sample of soil was mixed with 5 ml of solution containing factorial combinations of carbon and nitrogen treatments. The carbon treatments were glucose additions to supply 0, 1000, and 2000 mg carbon per kg of soil. The nitrogen treatments were KNO_3 additions to supply 0 and 100 mg nitrogen per kg of soil. The samples were placed in 125 ml flasks, flushed with He, treated with C_2H_2, and incubated for 48 hours at 25 ^{o}C. Each treatment was replicated 5 times and for soil collected at the three depths of 0 to 15 cm, 15 to 30 cm, and 30 to 45 cm.

RESULTS AND DISCUSSION

A summary of the denitrification rates measured in the field core studies is shown in Tables 1to 3. In general, the denitrification rates were low during the August 2002 sampling period (Table 1) and averaged 1.5, 0.4, and 1.3 g N_2O-N ha^{-1} d^{-1} for the Amarillo, Acuff, and Estacado soils, respectively. These denitrification rates are in the range of those reported by Welzmiller (2001) for Arizona soils. The low denitrification rates found likely reflect the low levels of moisture present in the soil when samples were collected. The moisture content averaged from 3.9 to 7.4% for the August sampling, which provide a preliminary indication of "background" emissions under the relatively dry conditions. The negative values suggest that some N_2O was utilized by the soil microflora. Overall, this translates to less than 890 g-N/ha (1 lb-N/acre) being removed by denitrification.

During the January 2003 sampling period (Table 2), higher denitrification rates were observed relative to the August sampling period. The rates averaged 124, 174, and 210 g N_2O-N ha^{-1} d^{-1} for the Amarillo, Acuff, and Estacado soils, which is equivalent to 37, 51, and 61 kg-N/ha (41, 57, and 68 lb-N/acre), respectively. For these same soils, the average soil moisture content was 17.1%, 16.3%, and 20.8% for the three soil types. The higher denitrification rates during January likely reflect the higher soil moisture content that existed. Furthermore, the ambient temperature at the time of sample collection was favorable, yet not optimal, for biological activity.

Table 1. Denitrification (nitrous oxide emission) rates from 3 soil series at the Lubbock Land Application Site in August, 2002.

		Amarillo			Acuff			Estacado	
Month	Rep	Moisture %	g N/ha/d	Rep	Moisture %	g N/ha/d	Rep	Moisture %	g N/ha/d
August	1	3.8	0.5	1	3.5	-0.6	1	6.9	0.1
	2	5.3	-0.5	2	4.3	-0.2	2	6.3	0.9
	3	9.2	5.0	3	4.8	-0.3	3	7.3	0.3
	4	4.7	0.0	4	3.3	3.1	4	6.5	0.0
	5	4.6	2.6	5	3.5	-0.3	5	5.6	3.6
	6	4.4	3.4	6	4.6	1.3	6	8.6	1.7
	7	4.7	2.8	7	3.2	2.9	7	6.0	0.4
	8	5.5	1.0	8	3.2	-0.4	8	8.6	4.2
	9	5.8	0.3	9	4.8	-0.6	9	10.0	1.0
	10	4.8	0.3	10	4.2	-0.3	10	7.8	0.9
Average		5.3	1.5		3.9	0.4		7.35	1.32
CV		28%	116%		16%	323%		19%	111%
Min		3.77	-0.51		3.20	-0.65		5.55	0.05
Max		9.23	4.95		4.80	3.06		9.95	4.20

During the May sampling period (Table 3), higher denitrification rates were observed relative to the other sampling periods. The rates averaged 468, 807, and 1990 kg-N/ha (524, 905, and 2229 g N_2O-N ha^{-1} d^{-1}) for the Amarillo, Acuff, and Estacado soils, which is equivalent to 153, 263, and 648 kg-N/ha (171, 295, and 726 lb-N/acre), respectively. For these three soils, the average soil moisture content was 17.3%, 18.5%, and 19.4%, which is similar to the samples collected in during the January sampling period. These higher rates of denitrification are most likely due to the higher, more optimal for microbial activity, average temperatures that occurred in May compared to those found in January.

Table 2. Denitrification (nitrous oxide emission) rates from 3 soil series at the Lubbock Land Application Site in January, 2003.

		Amarillo			Acuff			Estacado	
Month	Rep	Moisture %	g N/ha/d	Rep	Moisture %	g N/ha/d	Rep	Moisture %	g N/ha/d
January	1	15.9	-0.1	1	14.7	4.1	1	24.6	214.7
	2	17.0	132.6	2	10.2	513.6	2	21.9	205.4
	3	14.5	100.4	3	19.9	189.0	3	22.7	209.7
	4	15.2	284.8	4	12.9	227.2	4	18.8	215.1
	5	19.0	278.0	5	15.9	53.3	5	23.7	202.9
	6	16.4	286.2	6	18.7	113.5	6	21.4	228.2
	7	22.0	24.8	7	17.8	51.8	7	21.7	208.5
	8	14.9	123.2	8	18.6	2.1	8	20.6	200.4
	9	15.6	9.4	9	16.9	433.8	9	18.2	207.0
	10	20.2	5.7	10	16.9	154.3	10	14.1	208.0
Average		17.1	124.5		16.3	174.3		20.8	210.0
CV		15%	96%		18%	101%		15%	48%
Min		14.5	-0.1		10.2	2.1		14.1	200.4
Max		22.0	286.2		19.9	513.6		24.6	228.2

Table 3. Denitrification (nitrous oxide emission) rates from 3 soil series at the Lubbock Land Application Site in May, 2003.

		Amarillo			Acuff			Estacado	
Month	Rep	Moisture %	g N/ha/d	Rep	Moisture %	g N/ha/d	Rep	Moisture %	g N/ha/d
May	1	17.3	90.5	1	19.6	105.2	1	19.2	1861.4
	2	17.5	112.2	2	18.9	762.8	2	20.6	2663.5
	3	17.8	853.2	3	18.7	842.3	3	18.6	2426.4
	4	15.8	264.0	4	18.7	807.8	4	20.1	2056.7
	5	18.1	526.4	5	18.2	921.8	5	19.6	1062.6
	6	17.1	447.0	6	20.1	2725.0	6	19.3	2676.8
	7	15.9	256.0	7	18.6	371.7	7	19.7	4306.7
	8	20.0	761.8	8	17.3	337.6	8	18.2	2082.0
	9	18.0	1275.8	9	16.5	523.6	9	20.3	1757.7
	10	15.6	650.2	10	18.3	1654.0	10	18.3	1398.4
Average		17.3	523.7		18.5	905.2		19.4	2229.2
CV		8%	71%		6%	85%		4%	40%
Min		15.6	90.5		16.5	105.2		18.2	1062.6
Max		18.0	733.9		18.9	901.9		20.0	2604.2

Relatively high rates of denitrification were observed in the samples saturated with effluent water (Figure 2, Table 4). Of the sampling periods evaluated in this study, the highest rates were observed 2 days after saturation (Figure 2). At Day 2, the denitirfication rates averaged 290, 414, and 598 g N_2O-N ha^{-1} d^{-1} for the Amarillo, Acuff, and Estacado soils, respectively. Higher levels of denitrification are expected with higher soil moisture levels such as that following an irrigation event or rainfall event. Indeed, denitrification rates have been reported to increase sharply for short periods after soils are wetted (Westerman and Tucker, 1978; Groffman and Tiedje, 1988). Furthermore, denitrification can occur in anaerobic microsites in an otherwise aerobic soil (Knowles, 1981).

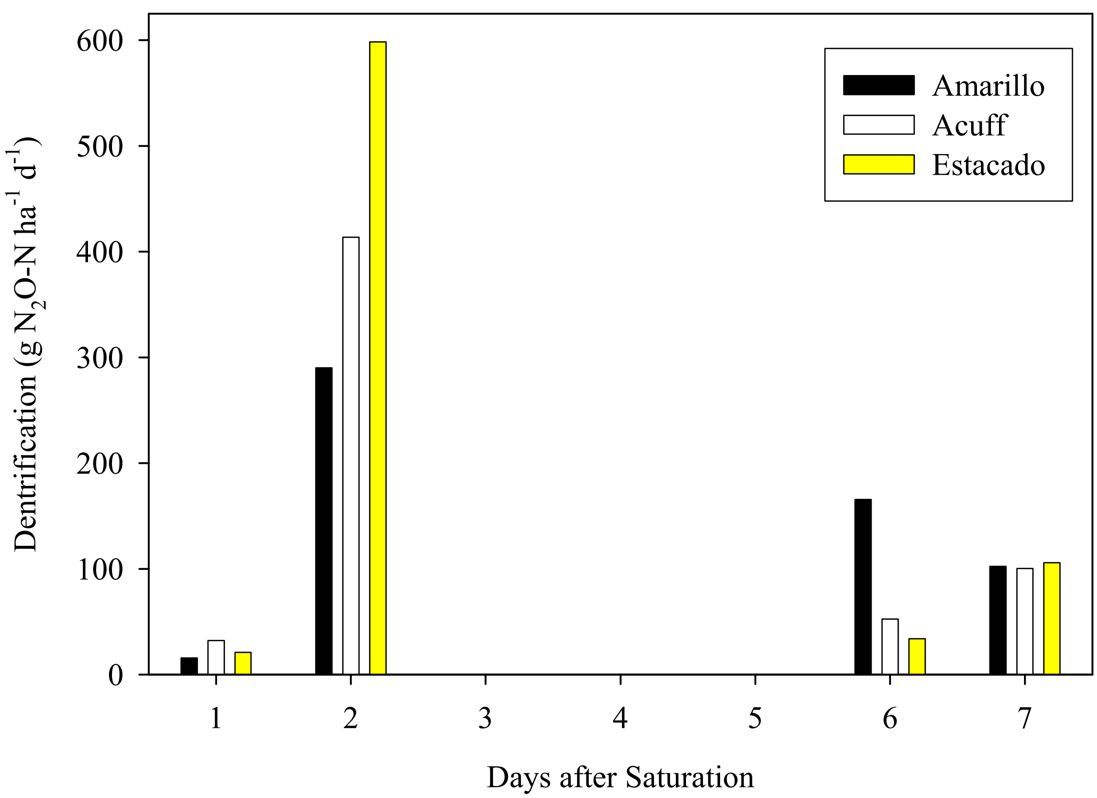

Figure 2. Denitrification rates from 3 soils at selected days following saturation with wastewater effluent.

It is generally observed that soil properties vary across a field or landscape. This variability is called "spatial variability." The presence of spatial variability is an important factor in determining appropriate sampling protocols to ensure that a given sample represents the area to be sampled. Furthermore, some soil properties especially nitrate and moisture contents can vary in time. This variability is called "temporal variability." Since denitrification is strongly influenced by nitrate concentration and moisture content in the field, denitrification rates can exhibit strong spatial and temporal variability. To fully characterize denitrification under field conditions, it is important to take into account this variability.

Table 4. Denitrification (nitrous oxide emission) rates from 3 soils at selected days following saturation with effluent water under laboratory conditions.

	Amarillo				Acuff				Estacado			
Rep	Day 1	Day 2	Day 6	Day 7	Day 1	Day 2	Day 6	Day 7	Day 1	Day 2	Day 6	Day 7
						g N_2O-N ha^{-1} d^{-1}						
1	8.2	344.3	548.7	95.4	19.7	234.1	33.5	102.4	26.5	580.4	15.9	130.8
2	28.8	290.7	23.9	32.5	1.7	729.3	42.4	78.8	5.2	707.4	78.8	190.6
3	22.3	457.1	117.3	143.5	44.1	648.3	146.2	190.3	11.1	601.7	58.9	80.5
4	4.8	73.2	11.3	18.6	40.3	174.1	13.1	44.1	15.8	195.1	15.1	59.4
5	13.7	285.3	127.0	221.2	55.6	281.7	26.9	85.9	46.3	907.2	0.6	67.8
Avg	15.6	290.1	165.6	102.2	32.3	413.5	52.4	100.3	21.0	598.3	33.9	105.8
CV	64%	48%	133%	82%	67%	62%	102%	54%	77%	43%	98%	52%
Min	4.8	73.2	11.3	18.6	1.7	174.1	13.1	44.1	5.2	195.1	0.6	59.4
Max	28.8	457.1	548.7	221.2	55.6	729.3	146.2	190.3	46.3	907.2	78.8	190.6

Denitrification was found to have high spatial variability (Schlesinger, 1991). Coefficients of variation for N_2O emissions can range from 6-1800% (Aulakh, 1992). Data collected in these tests clearly demonstrate the high spatial variability of denitrification rates in the field. For these sets of 10 data points in each of 3 soil types, coefficient of variations ranged from 111 to 323% for the August sampling and from 96 to 101% for the January sampling event and 40 to 85% for the May sampling period. Only 4 cores were analyzed for the Estacado soil in January, and these samples had a coefficient of variation of 64%.

The effects of soil depth on denitrification for the Amarillo, Acuff, and Estacado soils are shown in Table 5. Compared to data obtained from the intact cores incubated at ambient oxygen levels discussed previously, the denitrification data for these anaerobically treated, sieved samples is quite high. Important to note is that the denitrification rates for the soils are similar to each other in magnitude. Furthermore, the rates are fairly uniform with depth. This suggests that there is potential for denitrification to occur at in the top 45 cm, and perhaps, at even greater depths. This can be very important in the overall N balance of systems receiving relatively high rates of effluent water application. Compared to more traditional agronomic deficit-irrigation rates, effluent field rates may induce substantial denitrification rates at depth.

Table 5. Denitrification (nitrous oxide emission) rates from two soil series incubated anaerobically.

Soil Type	Depth (cm)	g N/ha/d
Acuff	0-15	1.66E+05
	15-30	1.97E+05
	30-45	1.78E+05
Estacado	0-15	1.61E+06
	15-30	1.35E+05
	30-45	2.75E+05

The data presented in Figures 3 and 4 shows a rapid decline in soil TKN concentration at two depths from Day Two through Day Four. A 287 mg/kg (46%) reduction in soil TKN at the 0 to 6-inch depth is indicated, while the TKN concentration at the 18 to 24-inch depth shows a 395 mg/kg (44%) decline. It is estimated that there are three phenomena that may have contributed to the rapid decline in soil TKN concentration. The possible causes include anaerobic denitrification, aerobic denitrification, and ammonia volatilization.

At the end of an irrigation event the upper horizons of the soil matrix were saturated and the water occupying the soil pores contained substantial amounts of carbon. These factors combined to develop an environment that was hospitable to anaerobic denitrifiers. Monnett et al. (1995) found that rapid rates of denitrification can occur in the topsoil as a result of carbon availability and aeration status caused by irrigation with wastewater. As the soil matrix became less saturated an environment was created in which aerobic heterotrophs (*Pseudomonas, Alcaligenes, Agrobacterium, Azospirillum, Bacillus, and Flavobacterium*) could function as denitrifiers (Lloyd, 1993).

During both the anaerobic and the aerobic periods, ammonification of amino acids occurred. Some fraction of the ammonia was lost through volatilization as a result. The EPA (1981) states that ammonia volatilization can be significant, especially in sandy, low organic soils. The test plot consists of Amarillo Fine Sandy Loam.

The TKN Model

An examination of the data presented in Figures 3 and 4 indicated that the soil TKN concentration at both the 0 to 6-inch and the 18 to 24-inch test depth decayed at a rapid rate from Day Zero until Day Four, reached an equilibrium or constant level at Day Five, and remained at approximately that level through Day Thirteen. A mathematical model described by Equation 1 was developed as a result. The TKN Model was developed during mild winter conditions, in Amarillo Fine Sandy Loam under an alfalfa crop.

$$y_t = ae^{-kt} + C, \qquad \text{Equation (1)}$$

where

y_t = TKN concentration in the soil at day t.
k = nitrogen decay rate
0 to 6-inch depth = 0.765
18 to 24-inch depth = 0.537.
a = TKN concentration in the soil at Day Zero, or 2060 mg/kg
C = TKN constant for the particular soil
0 to 6-inch depth = 278 mg/kg
18 to 24-inch depth = 373 mg/kg.
t = time in days from Day Zero.

The TKN Model is both a predictive and an analytical tool, and can be used to develop simulated TKN decay curves against which actual TKN analyses can be compared. If, for example, the TKN concentration at Day Zero is known, then the approximate TKN concentration at Day Four can be determined. The Model can also be used to find the approximate TKN concentration at Day Zero, if the TKN concentration on Day X is known. The analytical capability of the Model can be used to evaluate the accuracy of laboratory analyses performed on soil samples. Laboratory test results that are materially higher or lower than results predicted by the simulated decay curve may indicate procedural errors or unusual field conditions.

The TKN Model was used to develop simulated TKN data for the 0 to 6-inch and the 18 to 24-inch test depths and the coefficients as shown. The model fit the data very well with an R^2 of 0.95, that was highly significant ($\alpha = 0.02$).

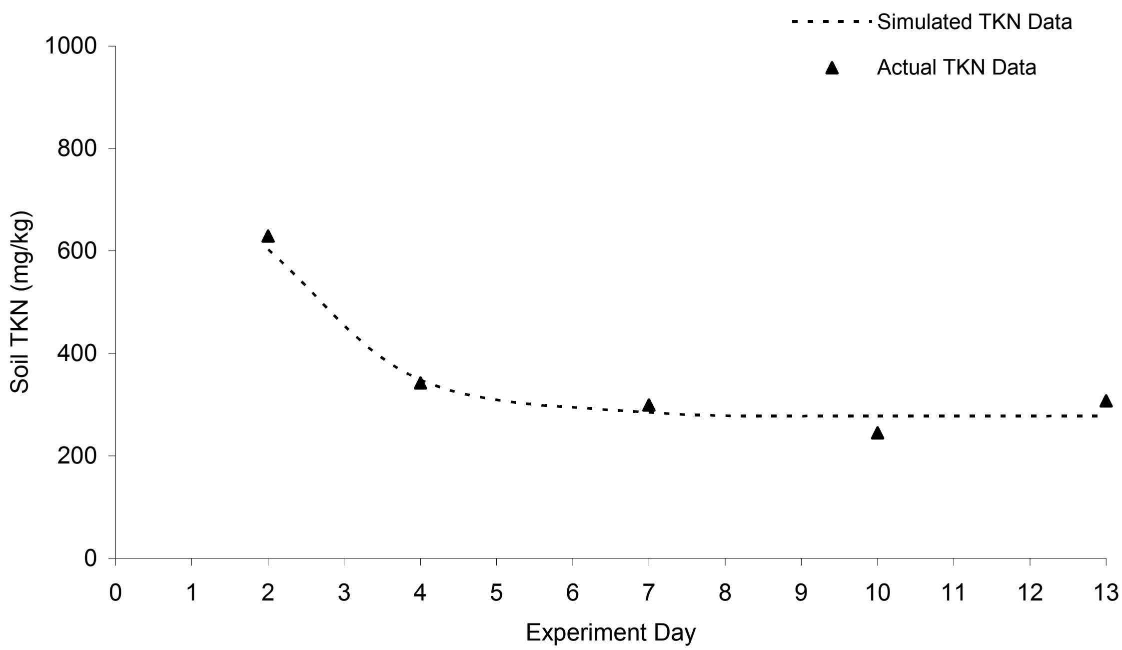

Figure 3. Measured versus predicted data of TKN at a soil depth between 0 and 6 inches.

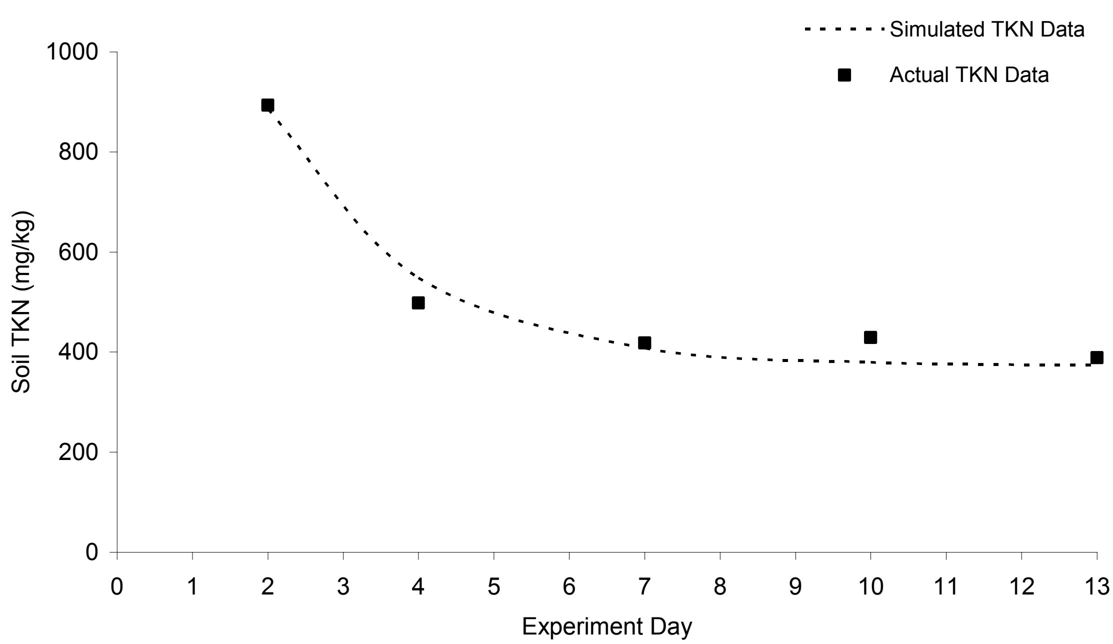

Figure 4. Measured versus predicted data of TKN at a soil depth between 18 and 24 inches.

CONCLUSIONS

The primary objective of this study was to understand the mechanisms affecting the fate of wastewater nitrogen at a wastewater irrigation site, and to use the knowledge gained from the study to improve the effectiveness and efficiency of wastewater treatment. Samples were collected from three test plots over a period of time in order to take into account the changes caused by the climatic conditions throughout the year.

To accurately assess denitrification losses under field conditions, it is necessary to evaluate temporal and spatial variability. Certainly, on an annual basis, denitrification losses reflect a time-weighted average of flux rates from throughout the year. Lower fluxes are expected for periods of lower water content and higher fluxes are expected for periods during and following significant rainfall or irrigation events. These fluxes are also influenced by temperature and several other soil properties. These baseline data can be used to improve the overall nitrogen balance estimates for a typical wastewater irrigation site for treated effluent.

It has been found that there is a large variation in the denitrification rates that occur within an irrigated wastewater site, especially under field conditions. In addition, when looking at the total nitrogen at the site, a fairly rapid decline in nitrogen concentration occurred within 4-5 days following an irrigation event and then remained almost constant after that time indicating that natural background conditions were achieved.

FUTURE ISSUES AND PRIORITIES

It is generally observed that soil properties vary across a field or landscape. This variability is called “spatial variability”. The presence of spatial variability is an important factor in determining appropriate sampling protocols to ensure that a given sample represents the area to be sampled. Furthermore, some soil properties especially nitrate and moisture content can vary in time. This variability is called “temporal variability”. Since denitrification is strongly influenced by nitrate and moisture contents in the field as well as temperature, denitrification rates can exhibit strong spatial and temporal variability. To fully characterize denitrification under field conditions, it is important to take into account this variability.

To accurately assess denitrification losses under field conditions, it is necessary to evaluate temporal and spatial variability. Certainly, on an annual basis, denitrification losses reflect a time-weighted average of flux rates throughout the year. Lower fluxes are expected for periods of lower water content and higher fluxes are expected for periods during and following significant rainfall or irrigation events. These fluxes are also influenced by temperature and several other soil properties. Furthermore, denitrification may occur at depths greater then the top 15 cm (6 inches) and losses of N that may occur at the deeper depths need to be evaluated to assess the overall N balance of a system receiving effluent water for irrigation.

The baseline data provided in this preliminary study can be used to improve the overall nitrogen balance estimates for a surface applied on-site sewage facility and at the Lubbock Land Application Site.

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