

NITROGEN MASS BALANCE FOR SPRAY FIELDS FERTILIZED WITH LIQUID  
SWINE WASTE

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## **ABSTRACT**

**JEFFREY T. DEBERARDINIS: Nitrogen Mass Balance For Spray Fields Fertilized With  
Liquid Swine Waste  
(Under the direction of Dr. Stephen C. Whalen)**

The swine industry has expanded rapidly in North Carolina over the last two decades. Animals are raised in confined facilities where waste is flushed into open-air lagoons and the liquid phase is land-applied to receiving fields as an organic fertilizer. The post-application fate has not been fully documented. Therefore, on three occasions I experimentally applied liquid swine waste at typical doses (40 to 130 kg N ha<sup>-2</sup>) to defined plots in an active spray field on a representative North Carolina swine production facility and constructed an N mass balance for a 14 to 18 d period. Most of the N (52%) was assimilated into plants, while surprisingly little (9%) was volatilized. Microbial immobilization accounted for 10% of the applied N, while 12% migrated below the active soil zone (surface 20 cm) and was presumably lost to groundwater. The soil storage term averaged 16%, while the denitrification sink was inconsequential (<1%).

## **ACKNOWLEDGEMENTS**

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## **CHAPTER I**

### **INTRODUCTION**

Modern agriculture is characterized by increasing specialization, which began with crops and has extended to animal production. The swine industry has historically been located in the mid-west, but recent growth has shifted toward the southeast, particularly eastern North Carolina. Growth of North Carolina's swine industry accelerated enormously in the early 1990s while the number of hog farms actually decreased (Figure 1.1), as hogs were increasingly produced in Confined Animal Feeding Operations (CAFOs). Currently, North Carolina is the second largest swine producing state surpassed only by Iowa. December 2005 statistics show that 9.8 million hogs were resident in the state (NCDA 2006a). The most intense concentrations of swine occur in Duplin and Sampson counties, each housing over 2 million animals (NCDA 2006b). Trends toward farm consolidation, specialization and intensification may be sound from economic and management perspectives, but may fail to adequately address important environmental impacts resulting from amplified generation of organic waste in localized regions (Barker and Zublena 1995). With roughly 2 million more hogs than humans in the state, management of swine waste presents enormous problems for controlling nutrient loading to adjacent ecosystems.

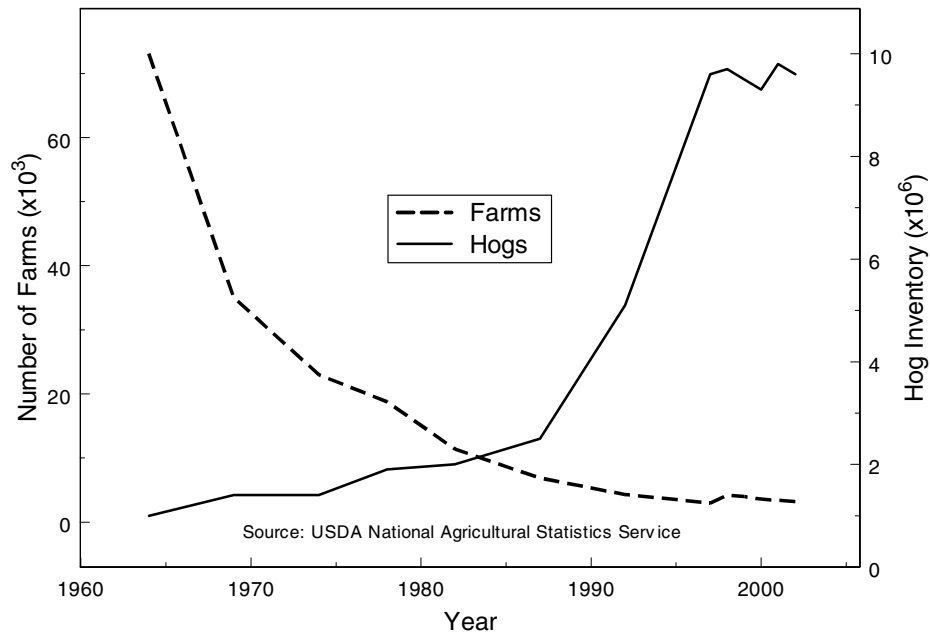


Figure 1.1. Consolidation of the swine industry in North Carolina.

The method of choice for swine waste disposal in North Carolina is the anaerobic lagoon/spray field system. Swine waste generated in confinement houses is typically flushed into open, anaerobic holding lagoons. While providing efficient and cost effective manure collection, these flush facilities produce a large volume of dilute liquid manure from a relatively small mass of concentrated feces and urine (Lowrance and Hubbard 2001). Although exposed to the atmosphere, lagoons are essentially anaerobic due to their high biochemical oxygen demand. The stored liquid is periodically land applied to crops as an organic fertilizer with solid set or traveling sprinkler systems. This management technique requires crops that assimilate high amounts of nutrients year round, so that applications can be made throughout the year if necessary (Crouse 1995). Typically, spray fields are planted with Coastal Bermudagrass (*Cynodon dactylon* L.) during the growing season and often over-

seeded in winter months with tall fescue (*Festuca* sp.). The accepted agronomic rate of application for these crops is based on the effluent N content and crop N requirements (Zublena *et al.* 1993).

The state vigorously regulates the industry with mandatory permitting and lagoon inspections. Nonetheless, ammonia volatilization and offsite runoff from hog farms has been implicated in the contemporary deterioration of coastal water quality in North Carolina (Paerl 1997). North Carolina is not unique in this regard. Agriculture in general is under increasing scrutiny, and the EPA is now working on regulations to control ammonia emissions from agricultural operations. Growth of the hog industry has plateaued in recent years both in North Carolina and the entire US because of a decline in price paid to the farmer for pork products. Further, in North Carolina, there is a moratorium on “general” permitting, which means no more lagoon/spray field systems can be implemented until the moratorium is lifted. This does not preclude CAFOs from employing other management systems. During the moratorium, the industry has undergone consolidation such that the majority of swine-producing CAFOs are owned or under contract with Smithfield Foods. During the moratorium, alternate technologies are being tested via a cooperative agreement between Smithfield Foods and NCSU. Once new technologies have been approved, Smithfield-owned farms (roughly 10% of NC CAFOs) must adopt the approved the waste management techniques and it is hoped that contract growers will follow suit.

Effluent stored in anaerobic lagoons contains nitrogen in the forms of organic-N, ammonium ( $\text{NH}_4^+$ -N), and ammonia ( $\text{NH}_3$ -N). During storage, 50-75% of the flushed waste-N is transformed into ammonium by mineralization or hydrolysis (Kirchman 1994). Once applied to spray fields, there are several fates for N (Figure 1.2). Organic-N can be

microbially mineralized to  $\text{NH}_4^+\text{-N}$ . Ammonium can be assimilated into plant material or microbial biomass, volatilized as  $\text{NH}_3$ , or can participate in cation exchange reactions in soil. Ammonia can be oxidized to nitrate ( $\text{NO}_3^-\text{-N}$ ) under aerobic conditions by nitrifying bacteria. Nitrate can be converted into plant biomass or reduced into inert dinitrogen gas ( $\text{N}_2$ ) within anaerobic microzones by various facultative anaerobes (denitrifying bacteria). All forms of gaseous N ( $\text{NH}_3$ ,  $\text{N}_2\text{O}$  and  $\text{N}_2$ ) can be lost to the atmosphere and dissolved forms ( $\text{NH}_4^+$  and  $\text{NO}_3^-$ ) can potentially be lost through surface water runoff and ground water leaching.

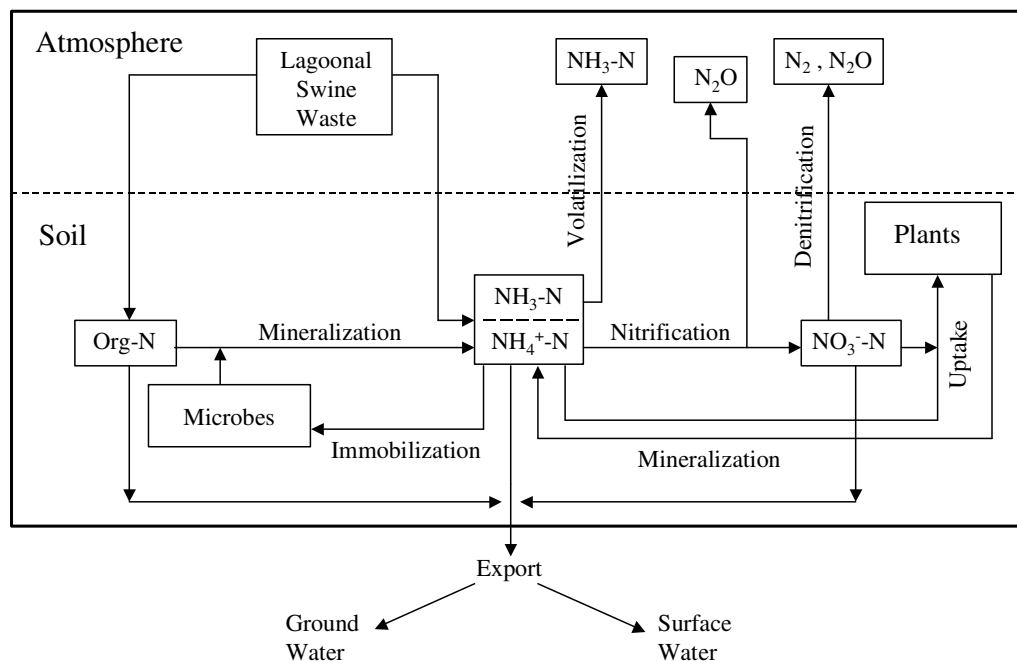


Figure 1.2. Nitrogen cycle for land-applied liquid lagoonal swine waste

To date, the emphasis on the fate of swine waste has been on ammonia volatilization (Hoff *et al.* 1981; Gordon *et al.* 1988; Safley *et al.* 1992; Halloran 1993; Sommer and Ersboll 1994; Sommer *et al.* 1997; Sharpe and Harper 1997, 2002; Dendooven *et al.* 1998; Rochette *et al.* 2001) and to a lesser extent on crop utilization (Cummings *et al.* 1975; Westerman *et*

*al.* 1977; Burns *et al.* 1985, 1990; Rees *et al.* 1993; Liu *et al.* 1997; Adeli and Varco 2001; Adeli *et al.* 2003). Fewer studies have focused on other individual processes such as nitrification and denitrification (Watanabe *et al.* 1997; Arcara *et al.* 1999; Harper *et al.* 2000; Whalen 2000; Whalen *et al.* 2000; Lowrance and Hubbard 2001 and Petersen *et al.* 2001). Others have also addressed microbial immobilization of N (Persson 1988; Kirchmann 1989; Fauci and Dick 1994) or ground water leaching and surface water runoff following fertilization (Cummings *et al.* 1975; Evans *et al.* 1984; King *et al.* 1985, 1990; Westerman *et al.* 1985, 1995, 1996; Karr *et al.* 2001).

A complete analysis of the fate of swine waste in the framework of an N mass balance has not been performed in North Carolina, but is necessary for perspective. Informed management and policy decisions rest on a more complete understanding of the fates, pools, transformations and the time course associated with each. Not only is an analysis of this nature lacking for land applied swine waste, it has been rarely performed for other forms of organic and inorganic fertilizers. Broadly, Best Management Practices (BMPs) for all facets of agriculture will benefit from improved knowledge of the fate of N-fertilizer in all forms, as N-fertilizer use has increased exponentially over the last 20 years and is projected to continue increasing (Howarth *et al.* 2002).

## **CHAPTER II**

### **OBJECTIVE**

The objective of this research was to investigate the rates of all fluxes and transformations among N pools associated with the microbially active soil zone (0-20 cm) in an agricultural spray field for approximately two weeks following liquid swine lagoon effluent irrigation. Specifically, on three occasions, rates of abiotic (ammonia volatilization and leaching) and biotic (denitrification, plant and microbial assimilation) processes were measured for two weeks in a crop of Coastal Bermudagrass after a simulated spray event of liquid effluent at two loading rates. The focus of each experiment was the construction of an event-specific N mass balance for a representative spray field in North Carolina in an effort to evaluate the relative importance of various post-application fates of land-applied liquid swine waste. Data will be useful to evaluate the effectiveness of lagoon/spray fields as a waste management tool and to aid managers and policy makers in determining the future role of this practice for swine production on North Carolina CAFOs.

## **CHAPTER III**

### **REVIEW OF THE LITERATURE**

#### **3.1. Importance of ammonia-N**

Atmospheric concentrations of biogenic gases such as  $\text{NH}_3$  are determined by the balance between local emissions and sinks (Schlesinger 1997). Despite a relatively short atmospheric residence time (10 d),  $\text{NH}_3$  is the third most abundant nitrogen gas following  $\text{N}_2$  and  $\text{N}_2\text{O}$ . Atmospheric  $\text{NH}_3$  readily reacts with acidic gasses and airborne particles that form hygroscopic salts containing ammonium sulfate and ammonium nitrate, which contribute to airborne particulate matter ( $\text{PM}_{2.5}$ ) (Krauter *et al.* 2003). These reactions occur in the troposphere within a few hours to days, rapidly returning  $\text{NH}_x$  salts and aerosols to the earth in wet and dry deposition, thereby fertilizing terrestrial and aquatic systems (Ferm 1998; Mosier 2001).

#### **3.2. Sources of ammonia-N**

Oxidized forms of N (eg.  $\text{NO}_x$ ) dominate in the atmosphere near sources of photochemical smog, while reduced forms ( $\text{NH}_4^+$  and  $\text{NH}_3$ ) are more prevalent in agricultural areas (Krauter and Potter 2002). The process by which gases are released from soils to the atmosphere is termed volatilization. In particular,  $\text{NH}_3$  is volatilized from animal waste and  $\text{NH}_4^+$ -based fertilizers, and to some extent by decomposition of organic matter by soil fungi and bacteria. Agricultural activity contributes significantly to the annual global emission of

NH<sub>3</sub> to the atmosphere. An estimated 50 to 75% of the ~55 Tg NH<sub>3</sub>-N yr<sup>-1</sup> emitted from terrestrial sources originates from animal- and crop- based agriculture, with domestic animals contributing over half of the agricultural component (Schlesinger and Hartley 1992; Mosier 2001). The three primary sources of NH<sub>3</sub> from livestock production are animal houses, waste storage systems and land-applied waste (Sharpe and Harper USDA/ARS).

### 3.3. Hydrolysis and NH<sub>3</sub> volatilization potential from waste storage

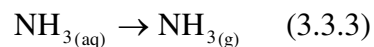
Prior to volatile loss as NH<sub>3</sub> from manures, N compounds must first be converted to ammoniacal (NH<sub>4</sub><sup>+</sup> and NH<sub>3</sub>) forms. Roughly 75% of the N found in mammalian urine is in the form of urea (Pinder *et al.* 2003). Excreted urea is rapidly hydrolyzed in the presence of the microbial enzyme urease that is present in fecal material (Whitehead and Raistrick 1990).



Following hydrolysis, NH<sub>4</sub><sup>+</sup> dissociates into aqueous phase NH<sub>3</sub> and H<sup>+</sup> according to a temperature and pH-dependent equilibrium constant (Monteny 2000). Ammonia then equilibrates between the aqueous and gas phase at the air-liquid boundary (Monteny 2000; Pinder 2003).



Gaseous NH<sub>3</sub> at this boundary may volatilize into ambient air by convective mass transfer depending on Henry's Law and the air temperature and wind velocity above the solution (Monteny 2000).



Additional nitrogenous compounds in urine include organic acids that are also subject to hydrolysis (Whitehead and Raistrick 1990). Efficient and rapid mineralization and



hydrolysis of urine ensure that stored liquid waste from CAFOs has a high potential for volatilization. Additional factors also contribute to a high potential for volatilization of stored waste. Deprotonation (eq. 3.3.2) ensures that the pH of stored waste remains high, favoring  $\text{NH}_3$  in the  $\text{NH}_4^+$  -  $\text{NH}_3$  acid-base pair ( $\text{pK} = 9.3$ ). High temperatures and the oxidation of fatty acids in manure also favor the formation of  $\text{NH}_3$  (Dendoovan *et al.* 1998; Pinder 2003). Physical attributes of the waste storage and animal holding facilities also contribute to the potential for  $\text{NH}_3$  volatilization. Confined or semi-enclosed facilities reduce air exchange, which increases the partial pressure of  $\text{NH}_3$  within the parlors and thereby discourages volatilization (Monteny 2000). Overall, 50 to 75% of the total-N in liquid manures is in the ammoniacal form (Kirchman 1994), and is therefore available for rapid volatilization.

### **3.4. Ammonia volatilization from waste**

Volatile losses of  $\text{NH}_3$  occur from environments with open surroundings such as fertilized agricultural fields, as the system attempts to reestablish equilibrium between aqueous and gas phases. Several studies show that these losses are influenced by the  $\text{NH}_3$  partial pressure gradient created between manure surfaces and the overlying air, evaporative conditions, and the method of application (Gordon *et al.* 2001). Evaporative conditions are the principal controllers of the rate and duration of  $\text{NH}_3$  loss from liquid effluents under field conditions and include air temperature and density, water vapor pressure, barometric pressure, and changes in net radiant energy (Lauer *et al.* 1976). Liquid manures are especially prone to N loss during application due to evaporation and drift. Westerman *et al.* (1996) observed effluent volume losses of 5 to 20% and Safley (1992) noted  $\text{NH}_3$  losses of

14 to 37% during field application of liquid swine waste with the big gun sprinkler systems commonly used in waste management at CAFOs in the southeastern US.

### **3.5. Ammonia volatilization from soil**

Ammonia also volatilizes from fertilized agricultural soils. In addition to the factors that control  $\text{NH}_3$  losses from the early stages of waste management (i.e. housing and storage), losses from agricultural fields are also determined by the characteristics specific to soil-cropping systems and by climate. Variables that influence volatilization from soils include moisture, temperature, buffering capacity, the form and rate of N applied, the depth of application and the resident urease activity (Demeyer *et al.* 1995; Dendoovan *et al.* 1998). Ammonia loss from soil depends partly on the rate at which effluent infiltrates the soil (Sommer and Ersboll 1994). Increasing resident soil water can promote  $\text{NH}_3$  loss by reducing soil porosity and, consequently, the rate of infiltration (Petersen and Andersen 1996). In extreme cases, ponding may result enhancing loss to volatilization. Additionally, crop residue can retard infiltration and provides additional surface area from which volatilization can occur (Rochette *et al.* 2001). Methods of waste application also regulate the depth of N integration into soils. Incorporation of waste (i.e. till or injection) into soil has been demonstrated to reduce  $\text{NH}_3$  emissions (Rochette *et al.* 2001). Sommer and Ersboll (1994) reported that  $\text{NH}_3$  loss from slurry applied to the surface of no-till soil plots was roughly two times the loss from tilled soil and three times the loss from injected slurry.

Waste characteristics such as the quantity and activity of urease also affect rates of  $\text{NH}_3$  volatilization from soil. The production of ammonium carbonate during the hydrolysis of urea generates localized increases in soil pH, which promote volatilization to an extent,

which is influenced by the initial soil pH, organic carbon content and cation exchange capacity CEC (Whitehead and Raistrick, 1990). Adsorption to soil and organic colloids influences the distribution of urease,  $\text{NH}_4^+$  and other N-compounds in a soil matrix. Urease activity is also dependant on soil temperature. The greatest loss of  $\text{NH}_3$  occurs in the temperature range of maximum urease activity (10 to 38°C), with the loss rate doubling for every 5.6°C temperature increase within this range ([www.misschem.com/scrReports/cotton.pdf](http://www.misschem.com/scrReports/cotton.pdf)).

### **3.6. Fate of emitted $\text{NH}_3$**

Depending on meteorological conditions, emitted  $\text{NH}_3$  may be rapidly redeposited or carried 10's of km from the source. (Mosier 2001). The latter has the potential to impact downwind ecosystems. Ammonia and its atmospheric products make up a substantial portion of total aerial N deposition worldwide, ranging from 47 to 87% (Marshall *et al.* 1998). Ammonia and  $\text{NO}_x$  have a relatively short atmospheric life span, so they are usually deposited in dryfall and precipitation, relatively close to their point of origin (Schlesinger 1997). Model estimates indicate that about 50% of the  $\text{NH}_3$  evolved from terrestrial systems is redeposited within 50 km of the source, and since most is redeposited within 20 km, total ammoniacal nitrogen (TAN) deposition near large animal operations can be substantial (Mosier 2001). Paerl (1997) noted that an increase in the ratio of  $\text{NH}_4^+:\text{NO}_3^-$  in wet precipitation recorded over the last 20 years at National Atmospheric Deposition Program (NADP) sites in Eastern North Carolina corresponds to the contemporary regional increase in the number of swine and poultry CAFOs.

### 3.7. Effects of NH<sub>3</sub> deposition

Natural ecosystems that receive NH<sub>3</sub> gas or its deposition products are thought to be net sinks for NH<sub>3</sub> (Sharpe and Harper 1997). Foliar absorption of NH<sub>3</sub> is recognized as a sink for airborne NH<sub>3</sub> and several studies have shown that agricultural crops can both absorb and emit NH<sub>3</sub> depending on their N status and atmospheric concentrations of NH<sub>3</sub> (Sharpe and Harper 1997). Above threshold levels, NH<sub>3</sub> can damage vegetation and reduce plant species biodiversity in natural ecosystems (McGinn and Janzen 1997). Ammonia emitted from agricultural sources has been implicated in forest decline (Nihlgard 1985; McLeod *et al.* 1990) and species changes within Europe (Van Hove *et al.* 1987).

Wet or dry deposition of the NH<sub>3</sub> may also contribute to undesired changes in aquatic ecosystems (Schulze *et al.* 1989). Excessive N deposition, has led to the acidification and eutrophication of aquatic systems (Mosier 2001). Eutrophication of aquatic systems is manifested as the development of harmful (toxic, hypoxia-inducing, food web altering) algal blooms (Paerl and Whitall 1999). In particular, the rapid proliferation of CAFOs in eastern North Carolina coincides with a contemporaneous increase in the frequency of fish kills, hypoxia events and development of Cyanobacterial blooms in the Neuse River estuary.

Comparison of global magnitudes of emission indicate that NH<sub>3</sub> is potentially more acidifying than emissions of sulfur dioxide (SO<sub>2</sub>) or nitrogen oxides (NO<sub>x</sub> = NO + NO<sub>2</sub>) (Bouwman *et al.* 1997). Deposition of NH<sub>3</sub> and NH<sub>4</sub><sup>+</sup> on poorly buffered soils can lead to an increase in soil acidification by increasing rates of nitrification and root uptake as well as increasing rates of acid formation from the chemical oxidation of NH<sub>4</sub><sup>+</sup> salts (Marshall *et al.* 1998). The effects of soil acidification include accelerated leaching of cations from plant tissues and soil, and increased mobilization of Al<sup>3+</sup>, which is toxic to plant roots (Marshall *et*

*al.* 1998). The input of one mole of ammonium sulfate ((NH<sub>4</sub>)<sub>2</sub>SO<sub>4</sub>) can result in the release of 4 moles of acidity (H<sup>+</sup>) by nitrification (Bouwman *et al.* 1997).

### **3.8. Nitrogen cycling**

Nitrogen is an essential component of amino acids, proteins and enzymes, all of which are critical to all life forms. The most bioavailable forms of nitrogen include ammonium (NH<sub>4</sub><sup>+</sup>-N) and nitrate (NO<sub>3</sub><sup>-</sup>-N), which dominate the inorganic pools of N. Other forms of nitrogen include nitrite (NO<sub>2</sub><sup>-</sup>-N), simple organic compounds such as free amino acids, urea, and poorly characterized, complex forms of organic N. Amino acids and urea are readily available to the biosphere, while NO<sub>2</sub><sup>-</sup>-N is toxic and little is known about the bioavailability of complex organics. The nitrogen cycle is the sum of physical and biochemical transformations and transfers of N among these pools, primarily through reactions that incorporate N into organic matter and subsequently release inorganic N into the environment (Stevenson and Cole 1999). Globally, the uptake of N from the inorganic N pool by organisms in the pedosphere is so rapid that little N remains in inorganic form, despite the large annual flux through this pool (Schlesinger 1997).

### **3.9. Immobilization and mineralization**

Immobilization and assimilation are processes where microbes and plants, respectively, transform inorganic N from the environment into biomass. Mineralization is the parallel process where the microbial decomposition of organic N releases NH<sub>4</sub><sup>+</sup>-N into the surrounding environment. Turnover of the immobilized or assimilated N occurs through the death and decomposition of biomass, once again transforming organic-N into

bioavailable inorganic forms (Fischer and Whalen 1999). Immobilization, assimilation and mineralization occur simultaneously in soils. Changes to the mineral N pool in the soil depend on the balance between plant and microbial demand for mineral N and the quality of carbon substrate for the microbes (Brady 2002).

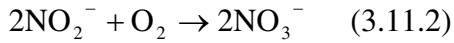
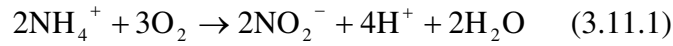
Microbial biomass dynamics play a critical role in mediating organic matter decomposition and turnover and therefore are central to nutrient cycling in soil (Fauci and Dick 1994). Studies have shown that fertilizer use impacts the relative size of microbial biomass present in soils, but there is little information about the effects of fertilizers upon microbial community structure (O'Donnell *et al.* 2001). Soils with a long-term history of organic amendments, such as animal manure, generally maintain high levels of microbial biomass and have a high potential to respond to new organic inputs (Fauci and Dick 1994).

### **3.10. Microbes**

A large number of biochemical transformations of nitrogen in soil are possible since N is found at valence states ranging from  $-3$  ( $\text{NH}_4^+$ ) to  $+5$  ( $\text{NO}_3^-$ ). A variety of specialized microorganisms found in soils participate in the transformations of N among these valence states and use the energy associated with redox changes for energy and growth (Rosswall 1982; Schlesinger 1997; Madigan *et al.* 2000). Collectively, these microbial activities are integral components of the soil N cycle (Schlesinger 1997). Two important microbially-mediated N cycling activities are nitrification and denitrification.

### 3.11. Nitrification

Nitrification is the microbial oxidation of  $\text{NH}_4^+\text{-N}$  to  $\text{NO}_3^-\text{-N}$  by chemolithotrophic bacteria. The complete oxidation of  $\text{NH}_4^+\text{-N}$  to  $\text{NO}_3^-\text{-N}$  is an eight-electron transfer and is accomplished by two bacterial genera acting in sequence (Madigan *et al.* 2000; Tate 2000):



The genus *Nitrosomonas* oxidizes  $\text{NH}_4^+$  to  $\text{NO}_2^-$  (eq. 3.11.1) and the genus *Nitrobacter* subsequently oxidizes  $\text{NO}_2^-$  to  $\text{NO}_3^-$  (eq. 3.11.2). Nitrification is an obligatory aerobic process that occurs readily in well-drained soils (<60% water filled pore space; WFPS) with neutral pH and low C/N ratios (Jones and Jacobsen 2001). The energy recovered by microbes during the conversion of  $\text{NH}_4^+\text{-N}$  to  $\text{NO}_3^-\text{-N}$  ranges from about 5 to 15%, indicating that large quantities of N must be nitrified for every  $\text{CO}_2$  molecule fixed by these microbes (Tate 2000).

### 3.12. Denitrification

Denitrification is the microbial respiratory process where nitrate is sequentially reduced and returned to the atmosphere as nitrogen gases ( $\text{NO}$ ,  $\text{N}_2\text{O}$ , and  $\text{N}_2$ ) (Schlesinger 1997).



This reductive sequence is coupled to electron transport phosphorylation and serves to support cellular metabolism (Fischer and Whalen 1999) Denitrifying bacteria are facultative anaerobes that utilize  $\text{NO}_3^-$  as an alternative electron acceptor for respiration when  $\text{O}_2$  is low

or absent (Crouse 1995). Oxygen supply is the most important environmental factor regulating denitrification in agricultural soils.

Soil oxygen supply is regulated by the partial pressure of O<sub>2</sub>, the physical characteristics of the soil (i.e. diffusion of O<sub>2</sub>), soil moisture content, and the rate of O<sub>2</sub> consumption through microbial respiration (Tiedje 1988; Fischer and Whalen 1999). Additional factors influencing rates of denitrification are soil temperature, NO<sub>3</sub><sup>-</sup> concentration, and availability of labile C, which acts as an electron donor (Fischer and Whalen 1999). Many denitrifying bacteria will also reduce other electron acceptors anaerobically such as ferric iron (Fe<sup>3+</sup>) and certain organic electron acceptors in the absence of NO<sub>3</sub><sup>-</sup>. Further, some denitrifying bacteria are also capable of fermentative metabolism (Madigan *et al.* 2000).

### **3.13. Coupled nitrification/denitrification**

Nitrification and denitrification are frequently ‘coupled’ in soils because the latter relies on the former to provide a substrate (NO<sub>3</sub><sup>-</sup>) that can be reduced. Consequently, denitrification rates in soil are often limited by rates of nitrification. Soil aggregates facilitate coupled nitrification-denitrification by the separation of aerobic soil and microaerophilic zones where oxidizers and reducers exist in close proximity (Seech and Beauchamp 1988). Rates of nitrification and denitrification are highly variable in soils. However, they affect the efficiency of fertilizer N use in agroecosystems, through the emissions of gaseous end-products (denitrification) and the production of NO<sub>3</sub> (nitrification), which is highly mobile and therefore subject to offsite transport. If denitrification leads to the evolution of large



proportions of N as  $\text{N}_2\text{O}$ , the net effect could be trading a potential water pollutant,  $\text{NO}_3$ , for an air pollutant,  $\text{N}_2\text{O}$  (Lowrance *et al.* 1998).

### **3.14. $\text{N}_2\text{O}$ and NO emissions from nitrification and denitrification**

Under unfavorable soil conditions (low  $\text{O}_2$ ),  $\text{N}_2\text{O}$  and NO increase in importance as an endproduct of denitrification, while  $\text{N}_2\text{O}$  becomes a byproduct of nitrification. Global monitoring over the last 20 years indicates that atmospheric concentrations of  $\text{N}_2\text{O}$  continue to rise about  $0.25\% \text{ yr}^{-1}$  (Houghten *et al.* 1996). Estimates indicate that nitrification and denitrification in agricultural soils contribute about 90% of the total anthropogenic  $\text{N}_2\text{O}$  emissions (Duxbury 1994). In fact, the acceleration of N fertilizer application in crop production is regarded as a major reason for enhanced  $\text{N}_2\text{O}$  emission from soils (Eichner 1990; Mosier *et al.* 1996). Cultivated soils are the largest source of atmospheric  $\text{N}_2\text{O}$ , emitting  $3.5 \text{ Tg } \text{N}_2\text{O-N yr}^{-1}$ , or  $\sim 24\%$  of the  $14.7 \text{ Tg}$  released annually from all sources (Houghten *et al.* 1996).

Gaseous products of nitrification and denitrification have several negative environmental consequences (Brady 2002). First, nitric oxide (NO) and  $\text{N}_2\text{O}$  from denitrification can contribute to the formation of nitric acid, a principal component of acid rain. Second, as  $\text{N}_2\text{O}$  rises into the upper atmosphere, it acts as a greenhouse gas (300 times as effective as  $\text{CO}_2$  on a per molecule basis) by absorbing infrared radiation that would otherwise exit the troposphere. Third and possibly most significantly,  $\text{N}_2\text{O}$  destroys stratosphere  $\text{O}_3$ , which shields the earth from harmful ultraviolet radiation.

### 3.15. Influence of fertilizers on N<sub>2</sub>O emissions

Interacting physiochemical and biological factors influence rates of N<sub>2</sub>O release from agricultural soils (Whalen *et al.* 2000). Enhanced short term or episodic N<sub>2</sub>O efflux is often associated with the application of synthetic fertilizers, which provide substrates readily utilizable (NH<sub>4</sub><sup>+</sup> and NO<sub>3</sub><sup>-</sup>) by nitrifiers or denitrifiers upon contact with these soil microbes (Ellis *et al.* 1998). In contrast, manures and slurries have medium term (i.e. several months) effects on soil nutrient status and microbial activity because these nutrients are made available by gradual decomposition of organics (Kirchmann 1994).

For three reasons, liquid swine waste contrasts sharply with slurries and solid organics with respect to microbial processing. First, the waste is largely (~80 to 90%) in the inorganic phase (NH<sub>4</sub><sup>+</sup>), and is therefore immediately available for microbial utilization. Second, liquid application ensures that the waste makes immediate contact with the microbial population as it drains downward in soil profile. Finally, management techniques frequently call for repeated application, ensuring the development of a highly responsive microbial community that can rapidly process the applied waste (Brown and Whalen 1999). In particular, liquid swine effluent provides not only a ready source of NH<sub>4</sub><sup>+</sup> for nitrifiers, but also a source of high quality C (volatile fatty acids) (Paul and Beauchamp 1989) for denitrifiers. Denitrifying activity is further enhanced by liquid waste through the reduction in soil O<sub>2</sub> concentration that occurs when air-filled pore spaces become liquid-filled and community respiration increases (Whalen *et al.* 2000; Sharpe and Harper 2002).

Liquid waste applied to an established community of microbes ensures a sharp post-fertilization decline in soil NH<sub>4</sub><sup>+</sup>-N and promotes a rapid and short lived (i.e. days) burst of nitrification, denitrification and N<sub>2</sub>O emission (Whalen *et al.* 2000). The limited information

on  $\text{N}_2\text{O}$  emission indicates that slurries and liquid manure lead to the highest denitrification rates (Lowrance and Hubbard 2001). However, comparisons of total N loss from various forms of organic waste (i.e. liquids and solids) show a wide range overall losses (Meisinger and Randall 1991).

### **3.16. Loss of $\text{NO}_3^-$ -N to surface and ground water**

The addition of large quantities of  $\text{NH}_4^+$ -N in animal waste to soils may lead to  $\text{NO}_3^-$  losses to groundwater if rates of nitrification are stimulated and recovery by the receiving crop is inefficient (Cooper 1974). Nitrate has a low affinity for anion exchange sites in soils (Karr *et al.* 2001) and is therefore easily leached. Consequently, high  $\text{NO}_3^-$  levels are frequently associated with drinking water contamination (Evans *et al.* 1984). High levels of  $\text{NO}_3^-$  N in drinking water lead to methemoglobinemia in infants (Jennings and Sneed 1996), which reduces the oxygen carrying capacity of blood (Owens 1994). Leaching occurs mostly on sandy soils and at sites with shallow water tables, both of which are common on the Coastal Plain where swine producing CAFOs have proliferated.

Studies of  $\text{NO}_3^-$  movement through sandy soils in the southeast are limited, but show that subsurface loss to groundwater is more common than surface runoff of  $\text{NO}_3^-$  (Westerman *et al.* 1985a). Regionally, losses of  $\text{NO}_3^-$  to groundwater are most likely during winter, due to low temperatures (i.e. less active microbes) and increases in precipitation (Karr *et al.* 2001). In some sandy North Carolina soils, high concentrations of  $\text{NO}_3^-$  have been found as deep as 65 cm below the surface after only 7.5 cm of rain (Baird 1990). Although it is clear that water contamination is a problem for CAFOs in the southeast, most cases of  $\text{NO}_3^-$

contaminated groundwater have been documented when fertilizer application rates exceeded BMP recommendations (Barker 1995).

### **3.17. Cropping systems used for swine waste disposal**

Swine CAFOs in North Carolina are located primarily on sandy Coastal Plain soils (Aneja *et al.* 2001). The use of lagoon spray field waste handling systems in CAFOs has transformed a large segment of the Coastal Plain into highly productive grassland (Poore and Green 1996). Disposal of swine effluent on spray fields requires crops that assimilate large amounts of nutrients year round, so waste applications can be made whenever necessary (Crouse 1995; Liu *et al.* 1997). Sod crops such as coastal bermudagrass (*Cynodon dactylon*) and tall fescue (*Festuca sp.*) meet this requirement and are therefore preferred cover crops by the swine production industry.

The ability of a cropping system to assimilate nutrients from animal waste is largely dependent upon plant characteristics (Rouquette *et al.* 1973; Cummings *et al.* 1975). Bermudagrass has several desirable attributes that account for its widespread use in spray fields. First, bermudagrass is a long-lived perennial with a hot weather growth period from April through September, and is well suited for sandy soils (Westerman *et al.* 1977). Second, it has a deep root system, high yield potential and good forage quality (Ball *et al.* 1991; Liu *et al.* 1997). Finally, rhizomes and stolons give coastal bermudagrass a vigorous spreading habit that is critical for receiving yearly effluent applications (Burns *et al.* 1990). Often, spray fields are over-seeded with fescue, which has growth characteristics that compliment those of bermudagrass. Rye fescue is a cool season perennial with peak production in North

Carolina during March through May and a surge of re-growth in September through November (Westerman *et al.* 1977).

Plant recovery rates of N following effluent application vary depending on conditions that influence the form, availability and mobility of N in cropping soils (Adeli *et al.* 2003). These factors include crop density, rates and timing of application, waste composition (i.e. organic content, plant available N), soil physical characteristics and climate (Cummings *et al.* 1975; Westerman *et al.* 1977; Adeli *et al.* 2003). Most effluent applications are performed during the summer months when forages are actively growing to facilitate maximum nutrient uptake. (King *et al.* 1985). However, crop removal values reflect a minimum amount of N required by plants because they do not account for N losses nor the quality and availability of nutrient reserves already in the soil (Zublena 1991). Plant available N in swine effluent is only estimated at 50% of the total effluent N (i.e. availability coefficient of 0.5), in recognition of the potential for loss through volatilization, denitrification and leaching (Zublena 1991; Poore and Green 1996).

A number of studies in the southern United States have assessed the effectiveness of coastal bermudagrass and other forage grasses at assimilating the N in liquid swine effluent. (Cummings *et al.* 1975; Westerman *et al.* 1977; Burns *et al.* 1985, 1990; Liu *et al.* 1997; Adeli and Varco 2001; Adeli *et al.* 2003). Up to 90 percent or more of the plant available N applied to sod crops in North Carolina is commonly recovered when BMPs are followed (Baird 1990). Studies have also revealed the correlation between N removal and aboveground dry matter yield, both of which increase quadratically with increasing swine effluent irrigation rates (Burns *et al.* 1990; Adeli and Varco 2001; Adeli *et al.* 2003). However, the capacity to assimilate waste N is not infinite. Excess N loading to plants can

reduce yields by rendering plants more susceptible to diseases and insect lodging, and stimulating vegetative growth at the expense of grain production (Crouse 1995).

## CHAPTER IV

### MATERIALS AND METHODS

#### 4.1. Description of study site

The study site is a corporate, farrow to half finish swine production facility located in Sampson County (34°48'N, 78°18'W), one of several counties that collectively make up North Carolina's 'Hog Belt' (Figure 4.1.1). This facility maintains an on site population of approximately 2200 head and employs regionally representative waste management practices. Within animal confinement houses, swine waste is deposited through slatted floors into a belowground sump and then flushed into an outdoor, anaerobic lagoon of 1.52 ha surface area and  $44 \times 10^3 \text{ m}^3$  volume. The liquid phase is subsequently recirculated to flush the collection pits. Stored, liquid phase swine waste is eventually land applied via a set sprinkler system to a summer crop of coastal bermudagrass (*Cynodon dactylon L.*) that is often overseeded with tall fescue (*Festuca sp.*) in the winter. Fertilization is at the agronomic rate, which is roughly similar for both crops at 18 to 23 kg (dry tonne)<sup>-1</sup> (Zublena *et al.* 1995)

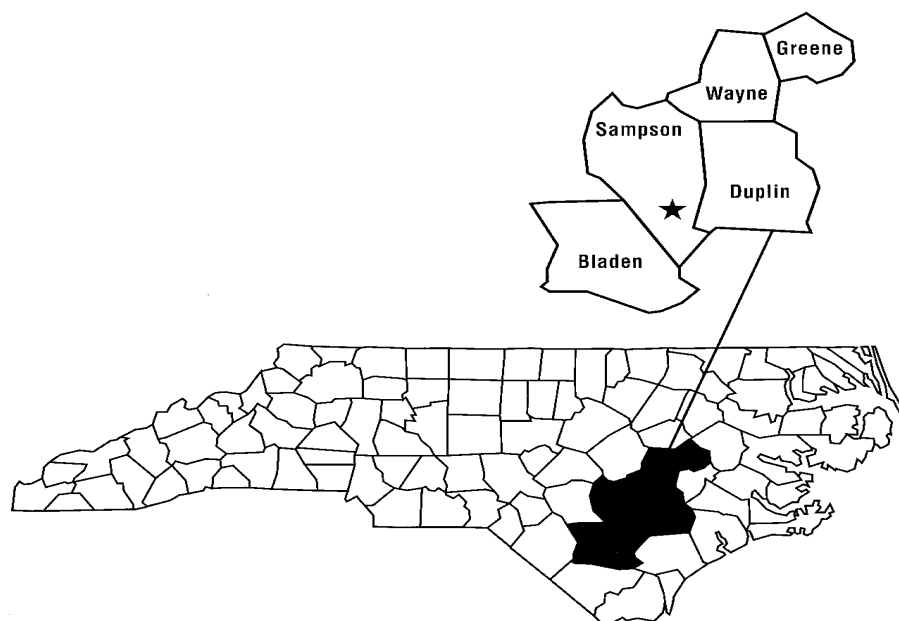


Figure 4.1.1. Location of the study site within North Carolina's hog belt

The spray field selected for this study has received liquid swine waste for approximately 12 years. The soil is a Blanton sand with zero to six percent slope and high permeability in the 0 to 20 cm layer (Brandon 1986). Soils are slightly acidic and are generally low in organic content. Concentrations of soil  $\text{NH}_4^+$  and  $\text{NO}_3^-$  are highly variable depending on proximity of the sampling date to the previous spray event. Table 4.1.1 gives ranges and means of selected soil physiochemical properties previously reported (Brown 1999; Nelson 1999) for the site.



Table 4.1.1. Ranges and means of selected soil physiochemical characteristics at the study site. Data are from homogenized samples collected monthly from the active 0 to 20 cm soil zone during the period December 1996 through October 1997.

Soil Characteristic	Range	Mean
pH	5.5 – 7.1	6.4
Moisture (%)	1.8 – 15.5	9.0
Particle Density (g cm <sup>-3</sup> )*	2.5 – 2.6	2.56
Bulk Density (g cm <sup>-3</sup> )*	1.19 – 1.40	1.31
Organic Matter (%)	1.2 – 5.0	3.4
Total-C (%)	0.79 – 2.66	1.16
Total-N (%)	0.05 – 0.14	0.07
NO <sub>3</sub> <sup>-</sup> -N (µg g dw <sup>-1</sup> )	1.1 – 55.8	8.5
NH <sub>4</sub> <sup>+</sup> -N (µg g dw <sup>-1</sup> )	0.5 – 60.0	2.1

\*Determined during this study.

## 4.2. Experimental design

**4.2.1. General.** Experiments were performed in the spring (March 2002), and late summer/early fall (August 2001 and 2002). Each experiment involved application of 0.5 and 1 inch (1.3 and 2.5 cm, respectively) of freshly collected liquid swine lagoon effluent to duplicate 1.5 m x 1.5 m (2.25 m<sup>2</sup>) plots. A fifth, unfertilized plot served as a control. Each plot was defined by an aluminum frame that extended 4 cm below the soil surface to prevent effluent loss. Prior to experimentation, vegetation within each plot and the surrounding area was cut to a uniform height of 18 cm for trial 1, not cut for trial 2 (vegetation < 12 cm at start of experiment) and 7-12 cm in height for trial 3 due to haying the previous week. Soil N

pools and transformations (Figure 1.2) were assessed for up to 19 d following waste addition. Sampling frequency (below) was guided by previous time course analyses conducted by this and other research groups for the response of various physicochemical and microbial processes to swine waste addition. The duration of each experiment was dictated by local weather and previous knowledge of the time required following waste application for soil N to decrease to pre-fertilization levels. Sample plots were positioned on level ground to minimize lateral off-plot transport of liquid swine waste during application or rainfall in the subsequent observational period.

All soils were collected with stainless steel sleeved (4.8 cm i.d.) or non-sleeved (5.4 cm i.d.) coring devices. The effect of destructive sampling was minimized by immediately replacing cores taken from experimental plots with cores from similarly fertilized companion plots. Locations of core removal and replacement in experimental plots were flagged to avoid later resampling. A previous study at this site (Nelson 1999) showed that >90% of the N-cycling microbial activity was localized in the surface 20 cm of soil. Except where noted, sampling was focused on this 20 cm “active zone”. All soil, liquid and plant materials intended for later analysis were kept at 4°C during storage and transport and were immediately frozen or processed on return to the laboratory.

Soil cores returned to the laboratory were sieved (4.75-mm mesh) and homogenized. Soil nutrients were extracted in 2 M KCl (10:1 volume/soil wet weight) and filtered (Whatman no. 42 paper). Nitrate plus  $\text{NO}_2^-$ -N (thereafter referred to as  $\text{NO}_3^-$ -N) and  $\text{NH}_4^+$ -N were determined by the copperized cadmium reduction and the indophenol blue methods (Keeney and Nelson 1982) using a Lachat QuikChem 4000 (Zellweger Analytics, Milwaukee, WI) automated flow injection analyzer. Soil moisture was measured

gravimetrically (oven dried at 105°C), particle density was determined pycnometrically, and bulk density was computed as the quotient of oven-dried mass divided by field volume. Percentage water-filled pore space (%WFPS) was calculated as the ratio of volumetric soil water content to total soil porosity.

Liquid swine waste was analyzed for total-N (TN) by persulfate oxidation (Solorzano and Sharpe 1980) and for  $\text{NH}_4^+$ -N as described above. A prior study at this site (Nelson 1999) showed no  $\text{NO}_3^-$ -N in liquid swine waste from this lagoon. TN in oven-dried, homogenized plant material was assayed by dry combustion with a Carlo Erba (Milan, Italy) NA 1500 elemental analyzer.

Fertilization was accomplished by adding either 29 or 58 L of liquid swine effluent to designated plots (no addition to the control) to achieve the target applications of 1.25 and 2.5 cm. Effluent was added slowly with a hand-held watering can over approximately 45 minutes to avoid ponding, and to allow ample time for infiltration. Collector cups (300 mL mason jars) were distributed throughout each plot to visually ensure uniform waste distribution. Following waste application, the contents of three randomly selected cups from each of two plots were composited for  $\text{NH}_4^+$ -N and TN analyses and the remaining cups were emptied onto the plots.

Soil moisture and temperature were measured continuously in the active soil zone during each experimental period. Soil moisture was determined with Campbell Scientific (Logan, UT) averaging water content reflectometers (Model CS 615) and temperature was determined with Campbell Scientific averaging thermocouple probes (Model TCAV) positioned at 5, 10, 15 and 20 cm below the soil surface. All data were recorded on a

Campbell Scientific Model CR10X data logger. Rainfall was determined with a Rainwise tipping bucket rain gauge.

**4.2.2. Ammonia volatilization.** Ammonia volatilization was measured using a dynamic chamber technique that has been demonstrated (Ruess and McNaughton 1988) to be 97% efficient at absorbing  $\text{NH}_3$  volatilized within the chamber. A single PVC soil collar (20 cm inside diameter x 14 cm height) equipped with an inlet and outlet was permanently emplaced to a soil depth of 5.5 cm in each plot. The system was sealed during measurement periods with a PVC lid fitted with an o-ring that had been lubricated with low-vapor vacuum grease. The system inlet consisted of a 2 cm inside diameter x 1.7 m long PVC pipe that extended beyond each plot. The distal end of the inlet was fitted with a removable cartridge of acid-coated (5% v/v  $\text{H}_3\text{BO}_4$ ) fiberglass batting to scrub inlet air of  $\text{NH}_3$ . Tests indicated that air entering the chamber was  $\text{NH}_3$ -free. A Gast (Benton Harbor, MI) rotary vane vacuum pump drew air through the chamber. A fraction of the outlet air was directed through Teflon tubing to a 500 ml gas-washing bottle where  $\text{NH}_3$  volatilized from the soil surface was trapped in 200 ml of 2%  $\text{H}_3\text{BO}_4$  solution. Flow rates through the chamber and  $\text{NH}_3$  trap were maintained at 15.7 and 0.7  $\text{L min}^{-1}$  with Dwyer (Michigan City, IN) gas flow meters.

Ammonia volatilization was presumed to represent a major loss term from waste-amended soils. To ensure accurate loading, each soil collar was covered during waste application to the remainder of the 2.25 $\text{m}^2$  plot and the exact volume of waste needed to achieve each target application (400 and 800 ml for 1.3 and 2.5 cm fertilizations) was added to the collar immediately thereafter. Ammonia flux measurements of 0.5 h duration were

made at increasing intervals from the time of application to 12 h (6 determinations), at 18, 24, 48 and 72 h and at 2 to 3 d intervals thereafter to the termination of the experiment. Inlet  $\text{NH}_3$  scrubbers and outlet  $\text{H}_3\text{BO}_4$  traps were replaced for each flux determination and soil collars were uncovered between flux determinations to allow free exchange of gases between the soil and atmosphere. Ammonia in boric acid traps was determined by automated colorimetry (above). Hourly, area-based rates of  $\text{NH}_3$  volatilization were determined from the fraction of chamber outlet air directed through the  $\text{NH}_3$  traps, the concentration of  $\text{NH}_3$  in the  $\text{H}_3\text{BO}_4$  solution and measurements of trapping efficiency.

**4.2.3. Nitrous oxide flux.** Nitrous oxide emission was determined using a static chamber technique (Whalen 2000). Briefly, a single soil collar of similar dimensions to those described above, but without inlet or outlet ports, was permanently installed in each plot. Covers were fitted with a capillary bleed to equalize pressure and a Swagelok o-seal fitting equipped with a septum for syringe sampling. A butyl o-ring on the cover effected the seal between it and the soil collar. Chamber headspaces were sampled using 10-cm<sup>3</sup> SESI nylon syringes equipped with pistons modified to accept a larger diameter sealing o-ring. Chamber headspaces were sampled on cover emplacement and at 20 min intervals thereafter to 60 min. Samples were stored prior to analysis by inserting the hypodermic needles of syringes into butyl stoppers. Covers were removed between sampling sessions, which occurred immediately following waste addition and at three-hour intervals to 11h (4 determinations), at 20, 24, 48, and 72 h, and at 2 to 3 day intervals thereafter to the end of each experiment.

Nitrous oxide was determined with a Shimadzu (Columbia, MD) GC-14A  $^{63}\text{Ni}$  electron capture detector gas chromatograph fitted with a 0.5-mL sample loop. Gases were separated on a 1-m pre-column and a 3-m analytical column (both Porapak-Q) operated at 40°C with a 5%  $\text{CH}_4$  – 95% Ar carrier flowing at 25 mL min $^{-1}$ . The instrument was calibrated with commercial  $\text{N}_2\text{O}$  air blends (Scott Specialty Gases; Plumbstead, PA) following verification of stated concentrations with National Institute of Standards and Technology standards. Hourly, area-based rates of  $\text{N}_2\text{O}$  emission were calculated from the time-linear rate of concentration increase in the headspace during chamber deployment.

**4.2.4. Denitrification.** Rates of denitrification ( $\text{N}_2 + \text{N}_2\text{O}$  production) were determined with the  $\text{C}_2\text{H}_2$ -block technique (Yoshinari *et al.* 1977; Klemmedtsson *et al.* 1990), using duplicate 4.8 cm diameter soil cores collected from the 20 cm active zone of each plot. Stainless steel corer sleeves (total length 30.5 cm) containing the 20 cm intact soil cores were sealed at both ends with caps fitted with the Swagelok sampling ports described above. Cores were injected from the bottom with acid-washed  $\text{C}_2\text{H}_2$  (Paul and Zebarth 1997a) via a perforated stainless steel cannula (1 mm diameter x 20 cm length), which facilitated homogeneous distribution of  $\text{C}_2\text{H}_2$  through the soil profile. The volume of  $\text{C}_2\text{H}_2$  added was sufficient to achieve a partial pressure of 10 kPa, which has been demonstrated to inhibit the reduction of  $\text{N}_2\text{O}$  to  $\text{N}_2$  by denitrifiers (Balderston *et al.* 1976). Cores were equilibrated to atmospheric pressure by briefly cracking the headspace sampling port and  $\text{C}_2\text{H}_2$  was then allowed to diffuse through the soil for 1 h, a time period that has been previously demonstrated (Fischer 2000) to be sufficient to evenly distribute  $\text{C}_2\text{H}_2$  through similar soil cores. Headspaces were then sampled at 30 min intervals to 1 h using modified (above) 5 ml

SESI nylon syringes and  $\text{N}_2\text{O}$  was determined by electron capture gas chromatography. Hourly, area-based rates of denitrification were calculated from the time-linear rate of  $\text{N}_2\text{O}$  increase in the headspace of the core sleeve. Denitrification flux measurements were determined at 2, 6, 10, 20, and 24h, and at 1, 2, or 3d intervals thereafter, to the termination of each experiment.

**4.2.5. Microbial biomass-N.** Microbial biomass-N was determined by the chloroform fumigation method (Horwath and Paul 1994) using 5.4 cm diameter cores (n=4) from the active zone of each plot. A homogenized 50 g sample of field-moist soil from each core was fumigated with ethanol-free  $\text{CHCl}_3$  for 24 h at 25°C. Moisture content of unfumigated and fumigated samples was adjusted to 55% of water holding capacity and fumigated soil was inoculated with 2 g of unfumigated soil. All soils were incubated in sealed containers in the dark at 25°C for 10 d. Microbial biomass-N was calculated as the flush of  $\text{NH}_4^+$ -N due to fumigation, corrected for the  $\text{NH}_4^+$ -N mineralized from the unfumigated soil. Following Jenkinson (1988), an average extraction efficiency of 0.54 was assumed. Microbial biomass-N was determined prior to fertilization (day 0), on day 3 and at the termination of each experiment.

**4.2.6. Plant biomass-N.** Prior to experimentation, above- and belowground live plant biomass collected in 25 cm diameter cores (n=5) was separated from soil and detritus by elutriation. Cores were randomly collected from outside of the sample plots, but within an area where vegetation was cut to a height similar to that of the sample plots. At the termination of each experiment, cores (n=4) were collected from within each sample plot and

similarly processed. Oven-dried plant material was homogenized (Wiley Mill; 40 mesh screen) and analyzed for TN by dry combustion (Carlo Erba NA 1500 Elemental Analyzer).

**4.2.7. Soil-N concentrations.** After each denitrification rate determination, soil cores were homogenized and analyzed for  $\text{NH}_4^+$ -N and  $\text{NO}_3^-$ -N to assess the time course for change in soil inorganic-N during the experimental period. Additional 5.4 cm diameter cores were periodically collected to 50 cm, sectioned at 10 cm intervals and analyzed for inorganic-N. Five cores randomly collected near the study plots prior to experimentation and three collected from within each plot at the termination of the experiment were used to determine zero time ( $T_0$ ) and final time ( $T_F$ ) depth distributions of soil inorganic-N. Duplicate cores were collected within each plot and adjacent to each plot following each rainfall >2.5 cm, to assess vertical and lateral transport of inorganic-N from the 20 cm active zone.

### 4.3. Calculations.

Gaseous N losses through volatilization ( $N_V$ ),  $\text{N}_2\text{O}$  emission ( $N_{\text{N}_2\text{O}}$ ) and total denitrification ( $N_D = \text{N}_2\text{O} + \text{N}_2$ ) were calculated by time-integrating rate measurements over the observational period. The fractional loss of N to  $\text{N}_2\text{O}$  in denitrification was calculated as the quotient of the time-integrated rates,  $N_{\text{N}_2\text{O}}/N_D$ . Net N assimilation into plant material ( $N_P$ ) and microbial biomass ( $N_M$ ) was calculated as the difference between  $T_F$  and  $T_0$  assays. Inorganic-N storage in the 0 to 20 cm active zone at  $T_0$  ( $N_{\text{in-0}}$ ) and  $T_F$  ( $N_{\text{in-F}}$ ) was calculated from the total dry mass of soil and measured inorganic-N concentrations. Nitrogen loss below the active zone ( $N_G$ ) was calculated as the difference in inorganic-N stored in the 20 to



50 cm depth interval at  $T_F$  and  $T_0$ , based on dry soil mass and inorganic-N determinations. A nitrogen mass balance was constructed for each plot:

$$N_{in-0} + N_S = N_{in-F} + N_M + N_P - N_V - N_D - N_G \quad (4.3.1)$$

where  $N_S$  is the mass of N added to each plot as swine waste and all other terms are as previously defined. Due to the difficulties in monitoring fluxes into and out of soil dissolved organic-N reservoirs, I did not consider this reservoir. Instead, following Pakrou and Dillon (2000), I focused on the mineral-N pool in constructing a simplified N budget.

## CHAPTER V

### RESULTS

#### 5.1. Swine effluent-N

Ammonium-N and TN contents of the swine effluent collected from the lagoon prior to the three experiments ranged over factors of about 2 and 1.5, respectively (Table 5.1.1).

The waste was highly mineralized, as  $\text{NH}_4^+$ -N accounted for roughly 75% of the TN.

Table 5.1.1. Ammonium and TN concentrations in liquid swine lagoon effluent used to fertilize experimental plots.

Experiment	Date	$\text{NH}_4^+$ -N ( $\text{mg l}^{-1}$ )	TN ( $\text{mg l}^{-1}$ )	% TN as $\text{NH}_4^+$ -N
1	12 Aug 01	222	330	67
2	25 Mar 02	444	506	88
3	7 Aug 02	205	331	62
Average		290	389	72
SD		133	101	-

#### 5.2. Experiment 1

Immediately following the first experimental application,  $\text{NH}_3$  emission from the soil increased to  $79 \text{ mg m}^{-2} \text{ h}^{-1}$  and  $186 \text{ mg m}^{-2} \text{ h}^{-1}$  in the plots amended with 1.25 cm and 2.5 cm of swine waste, respectively (Figure 5.2.1). In contrast, emission from the control

plot was only  $7 \text{ mg m}^{-2} \text{ h}^{-1}$ , or 9% and 4% of the emission from the fertilized plots. The period of elevated emission was brief, as  $\text{NH}_3\text{-N}$  fluxes had declined to  $15 \text{ mg m}^{-2} \text{ h}^{-1}$  and  $40 \text{ mg m}^{-2} \text{ h}^{-1}$  in the respective fertilized plots only 1.5 h following waste addition. Within 6 h of waste addition,  $\text{NH}_3\text{-N}$  fluxes in the fertilized plots were indistinguishable from that of the control plot.

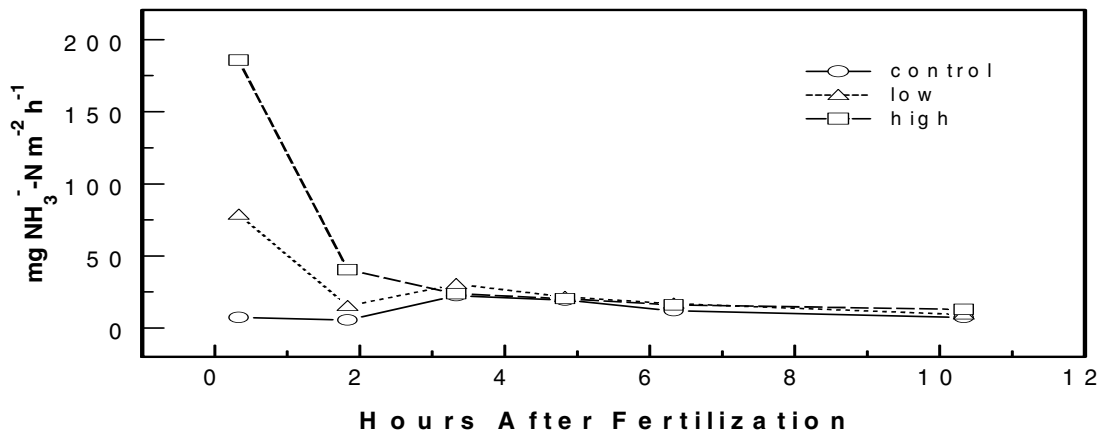


Figure 5.2.1. Time course for  $\text{NH}_3$  volatilization in experiment 1. Low and high doses represent 1.25 and 2.5 cm additions of waste, which correspond to area-based N additions of  $3.94$  and  $8.44 \text{ g m}^{-2}$ . The SD for the duplicate plots (error bars not shown) at each time point averaged  $8.1 \text{ mg NH}_3\text{-N m}^{-2} \text{ h}^{-1}$  and  $10.2 \text{ mg NH}_3\text{-N m}^{-2} \text{ h}^{-1}$  for the low and the high dose plots, respectively.

Soil percent water filled pore space responded immediately to the simulated spray event by increasing from a pre-fertilization value of around 30% to about 45% and 55% in the low and high dose plots, respectively (Figure 5.2.2a). Thereafter, these sandy soils rapidly drained with an accompanying decline in %WFPS. Soils in all plots showed increases in %WFPS, followed by rapid draining in response to storms that occurred roughly 1, 2, 6 and 8 d after waste addition. Soil %WFPS was essentially similar in all plots from about day 8 to the termination of the experiment. A total of about 18 cm of rain was recorded during the observational period.

Soil temperatures showed diurnal variations of about 3°C and declined gradually throughout the experiment (Figure 5.2.2b). The maximum soil temperature of 30°C was observed at the time of waste addition and corresponded to an air temperature of 32°C. The lowest soil temperature was 24°C, in the evenings of days 14 and 15. Thus, soils varied over only 6°C during the entire observational period.

Immediately after fertilization, soil  $\text{NH}_4^+$ -N increased from a pre-fertilization concentration of  $3 \mu\text{g g}_{\text{dw}}^{-1}$  to roughly  $9 \mu\text{g g}_{\text{dw}}^{-1}$  and  $17 \mu\text{g g}_{\text{dw}}^{-1}$  in the high and low dose plots, respectively (Figure 5.2.2c). Within 10 h of waste application, concentrations reached maxima of  $15 \mu\text{g g}_{\text{dw}}^{-1}$  and  $25 \mu\text{g g}_{\text{dw}}^{-1}$  in these plots. By day 6,  $\text{NH}_4^+$ -N concentrations had steadily declined to about  $9 \mu\text{g g}_{\text{dw}}^{-1}$  in both treatments, but still exceeded the  $6 \mu\text{g g}_{\text{dw}}^{-1}$  of the control plot. Thereafter, concentrations of  $\text{NH}_4^+$ -N fluctuated over a range of about  $10 \mu\text{g g}_{\text{dw}}^{-1}$  in all plots to the end of the experiment. Changes in  $\text{NH}_4^+$ -N concentrations in the fertilized plots tracked those of the control plot, suggesting that the fluctuations were unrelated to waste amendment. However, soil  $\text{NH}_4^+$ -N levels in the high dose treatment were consistently higher than those in the low dose treatment, and  $\text{NH}_4^+$ -N in the control plot was consistently lower than in the waste-amended plots.

Soil  $\text{NO}_3^-$ -N concentrations began a slow increase immediately after waste amendment from a pre-fertilization value of  $0.1 \mu\text{g g}_{\text{dw}}^{-1}$  to  $1.3 \mu\text{g g}_{\text{dw}}^{-1}$  and  $2.9 \mu\text{g g}_{\text{dw}}^{-1}$  by day 4 in the low and high dose treatments (Figure 5.2.2d). These increases in  $\text{NO}_3^-$ -N concentrations accompanied the decline in soil  $\text{NH}_4^+$ -N levels observed to day 4 (Figs. 5.2.2c & d). Nitrate values in the plots fertilized with the high dose of swine waste remained elevated until day 6, then rapidly decreased to about  $0.5 \mu\text{g g}_{\text{dw}}^{-1}$ . Thereafter, soil  $\text{NO}_3^-$ -N concentrations remained below  $1 \mu\text{g g}_{\text{dw}}^{-1}$  with small peaks on days 8, 14, and 18. Consistent

with the time course for change in soil  $\text{NH}_4^+$ -N, soil concentrations of  $\text{NO}_3^-$ -N reflected the level of fertilization. That is, concentrations in the high dose treatment exceeded those of the low dose treatment, which were elevated with respect to the control.

Highest denitrification rates were observed in all of the fertilized plots on days 2, 7 and 10 (Figure 5.2.2e). Fluxes of  $\text{N}_2$  at these sampling points varied from approximately 500 to 840  $\mu\text{g m}^{-2} \text{h}^{-1}$  in the low dose plots and from about 800 to 860  $\mu\text{g m}^{-2} \text{h}^{-1}$  in the high dose plots. All emissions of soil  $\text{N}_2$  corresponded with rain events of about 9, 7, and 2 cm immediately preceding these respective dates. Elevated rates of denitrification were observed in the control plot between day 7 and 12, corresponding to the increase in soil moisture following the day 7 shower.

Nitrous oxide, an additional endproduct of denitrification, showed much higher spatiotemporal variability in emission than  $\text{N}_2$  (Figure 5.2.2f). The  $\text{N}_2\text{O}$  flux abruptly increased from 26 to 615  $\mu\text{g m}^{-2} \text{h}^{-1}$  in the plots amended with the high dose of effluent on day 2. Reduced, but measurable fluxes were also observed on days 4 and 6. A small flux of 95  $\mu\text{g N}_2\text{O}^- \text{N m}^{-2} \text{h}^{-1}$  was also observed on day 2 from the plots amended with the low dose of swine effluent. Otherwise,  $\text{N}_2\text{O}$  fluxes were undetectable in the fertilized plots. Low levels of emission were observed on three dates in the control plot, with rates varying from 1 to 9  $\mu\text{g N}_2\text{O}^- \text{N m}^{-2} \text{h}^{-1}$ . Otherwise,  $\text{N}_2\text{O}$  emissions were not detected in control plots.

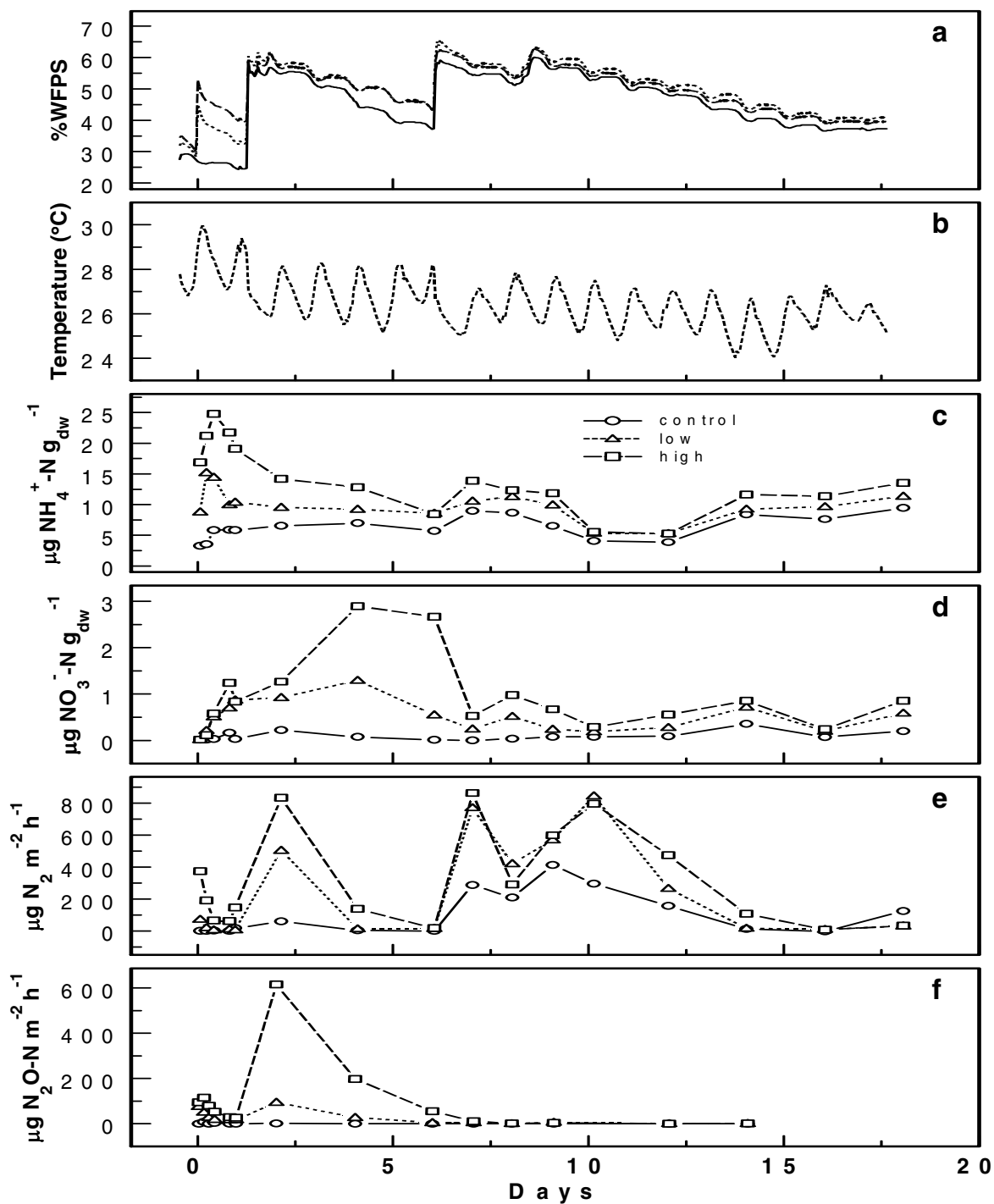


Figure 5.2.2. Swine waste fertilization experiment 1. Time course for change in: (a) percent water filled pore space (%WFPS); (b) temperature; (c)  $\text{NH}_4^+\text{-N}$ ; (d)  $\text{NO}_3^-\text{-N}$ ; and soil emissions of (e)  $\text{N}_2$ ; and (f)  $\text{N}_2\text{O-N}$  from plots amended with low and high doses (as per Figure 5.2.1) of liquid swine effluent. Data points for all variables are the average of duplicate plots at each dose except temperature ( $n = 3$ ). The SD for duplicate plots at the high and low doses (error bars not shown) averaged: 1.4 and 1.8  $\mu\text{g NH}_4^+\text{-N g}_{\text{dw}}^{-1}$ , 0.2 and 0.3  $\mu\text{g NO}_3^-\text{-N g}_{\text{dw}}^{-1}$ , 74.3 and 66.8  $\mu\text{g N}_2\text{-N m}^{-2}\text{ h}^{-1}$  and 5.1 and 17.4  $\mu\text{g N}_2\text{O-N m}^{-2}\text{ h}^{-1}$ .

### 5.3. Experiment 2

Consistent with the first simulated spray event,  $\text{NH}_3\text{-N}$  emissions from soils immediately increased to  $50 \text{ mg m}^{-2} \text{ h}^{-1}$  and  $68 \text{ mg m}^{-2} \text{ h}^{-1}$  in the low and high dose plots, respectively (Figure 5.3.1). In contrast, the immediate emission from the control plot was only  $1.5 \text{ mg m}^{-2} \text{ h}^{-1}$ , or 3% and 2% of the flux from the fertilized soils. Within 1.5 h of waste addition,  $\text{NH}_3\text{-N}$  fluxes had dropped to  $8 \text{ mg m}^{-2} \text{ h}^{-1}$  in the low dose plots and  $20 \text{ mg m}^{-2} \text{ h}^{-1}$  in the high dose plots. However, by 10 h,  $\text{NH}_3\text{-N}$  fluxes were indistinguishable among all treatments.

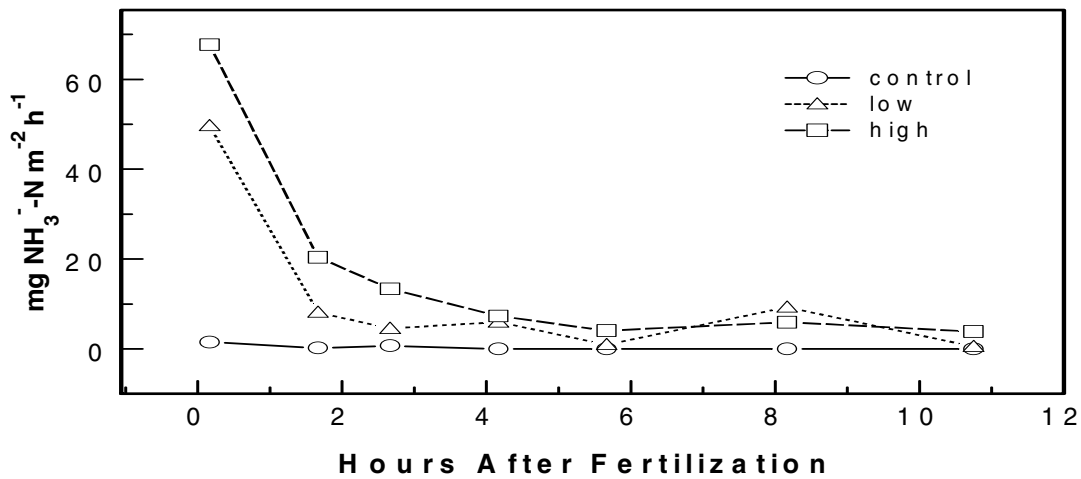


Figure 5.3.1. Time course for  $\text{NH}_3$  volatilization in experiment 2. Low and high doses represent 1.25 and 2.5 cm additions of waste, which correspond to area-based N additions of 6.1 and 12.9  $\text{g m}^{-2}$ . The SD for the duplicate plots (error bars not shown) at each time point averaged  $1.7 \text{ mg NH}_3\text{-N m}^{-2} \text{ h}^{-1}$  and  $2.0 \text{ mg NH}_3\text{-N m}^{-2} \text{ h}^{-1}$  for the low and the high dose plots, respectively.

Soil percent water filled pore space responded immediately to the simulated spray event by increasing from a pre-fertilization value of around 35% to about 46% and 56% in the low and high dose plots, respectively (Figure 5.3.2a). As in the first experiment, %WFPS immediately increased in waste-amended plots following fertilization, but rapidly declined as

soils drained. Soils in all plots also showed increases and subsequent decreases in %WFPS in response to 1 and 4 cm storms that occurred on days 1 and 6. As soils drained and dried following the day 6 rainfall, %WFPS steadily declined in all plots until reaching pre-fertilization values at the end of the experiment. A total of 6.4 cm of rain was recorded during the observational period.

Soil temperatures showed diurnal variations ranging from 1 to 5°C as daily temperatures generally increased to the middle of the experiment (Figure 5.3.2b). The maximum soil temperature of about 19°C was observed on day 9 and corresponded to an air temperature of 20°C. The lowest soil temperature of 12°C was observed in the mornings of days 0 and 13. Thus, soils varied over only 7°C during the entire observational period.

Immediately following waste addition, soil  $\text{NH}_4^+\text{-N}$  increased from a pre-fertilization concentration of  $3 \mu\text{g g}_{\text{dw}}^{-1}$  to roughly  $13 \mu\text{g g}_{\text{dw}}^{-1}$  and  $45 \mu\text{g g}_{\text{dw}}^{-1}$  in the low and high dose plots, respectively (Figure 5.3.2c). Ammonium concentrations continued to increase to a maximum of  $25 \mu\text{g g}_{\text{dw}}^{-1}$  at 10 h post-fertilization in the low dose plots, but declined from the  $45 \mu\text{g g}_{\text{dw}}^{-1}$  observed immediately after fertilization in the high dose plots to about  $30 \mu\text{g g}_{\text{dw}}^{-1}$  in the same time period. By day 6,  $\text{NH}_4^+\text{-N}$  concentrations had steadily declined to about  $9 \mu\text{g g}_{\text{dw}}^{-1}$  in both treatments, but still exceeded the  $2.5 \mu\text{g g}_{\text{dw}}^{-1}$  of the control plot. Thereafter, concentrations of  $\text{NH}_4^+\text{-N}$  fluctuated over a range of roughly  $12 \mu\text{g g}_{\text{dw}}^{-1}$  in fertilized plots until the end of the experiment. In agreement with experiment 1, changes in  $\text{NH}_4^+\text{-N}$  concentrations in fertilized plots generally tracked those of the control plot, suggesting that the fluctuations were unrelated to waste amendment. Furthermore, soil  $\text{NH}_4^+\text{-N}$  levels in the high dose plots were consistently higher than those in the low dose plots and  $\text{NH}_4^+\text{-N}$  in the control plot was consistently lower than all waste-amended plots.



Soil  $\text{NO}_3^-$ -N concentrations slowly increased after waste amendment from a pre-fertilization value of  $0.5 \mu\text{g g}_{\text{dw}}^{-1}$  to  $4 \mu\text{g g}_{\text{dw}}^{-1}$  and  $7 \mu\text{g g}_{\text{dw}}^{-1}$  by day 6 in the low and high dose treatments, respectively (Figure 5.3.2d). These increases in  $\text{NO}_3^-$ -N concentrations accompanied the decline in soil  $\text{NH}_4^+$ -N observed to day 6 (Figure 5.3.2c). Nitrate concentrations at both doses remained elevated between days 6 and 10, fluctuating between about  $2$  and  $4 \mu\text{g g}_{\text{dw}}^{-1}$  in the low dose plots and between  $5$  and  $8 \mu\text{g g}_{\text{dw}}^{-1}$  in the high dose plots. Thereafter,  $\text{NO}_3^-$ -N values generally declined in both fertilized plots to the end of the experiment, but remained elevated above concentrations in the control. Overall, the time course for change in soil  $\text{NO}_3^-$ -N and  $\text{NH}_4^+$ -N and concentration differences among treatments in this experiment corroborated well the data from experiment 1. That is, concentrations in the high dose treatment always exceeded those of the low dose treatment, which always exceeded those of the control, and soil  $\text{NO}_3^-$ -N increased as  $\text{NH}_4^+$ -N declined in waste-amended plots.

Noteworthy increases in rates of denitrification in fertilized plots were observed only on day 7 (Figure 5.3.2e). Peak fluxes of  $\text{N}_2$  reached approximately  $2600 \mu\text{g m}^{-2} \text{h}^{-1}$  in the low dose plots and  $5400 \mu\text{g m}^{-2} \text{h}^{-1}$  in the high dose plots subsequent to a 4 cm rainfall on day 6. Dinitrogen gas emission from the control plot also increased on day 7 to  $365 \mu\text{g m}^{-2} \text{h}^{-1}$  from a baseline value of about  $5 \mu\text{g m}^{-2} \text{h}^{-1}$ . However, this elevated value for the control was only 14% and 7% of the  $\text{N}_2$  emitted from the high and low dose fertilized plots, respectively.

Nitrous oxide emissions tracked that of  $\text{N}_2$  in the fertilized plots (Figure 5.3.2f). Emission of  $\text{N}_2\text{O}$ -N was limited to a single event in response to the day 6 rain, and resulted in fluxes of  $175 \mu\text{g m}^{-2} \text{h}^{-1}$  from the low dose plot and  $1000 \mu\text{g m}^{-2} \text{h}^{-1}$  from the high dose plot.

Otherwise, N<sub>2</sub>O emissions were undetectable from the fertilized plots. Nitrous oxide emission was never observed from the control plot.

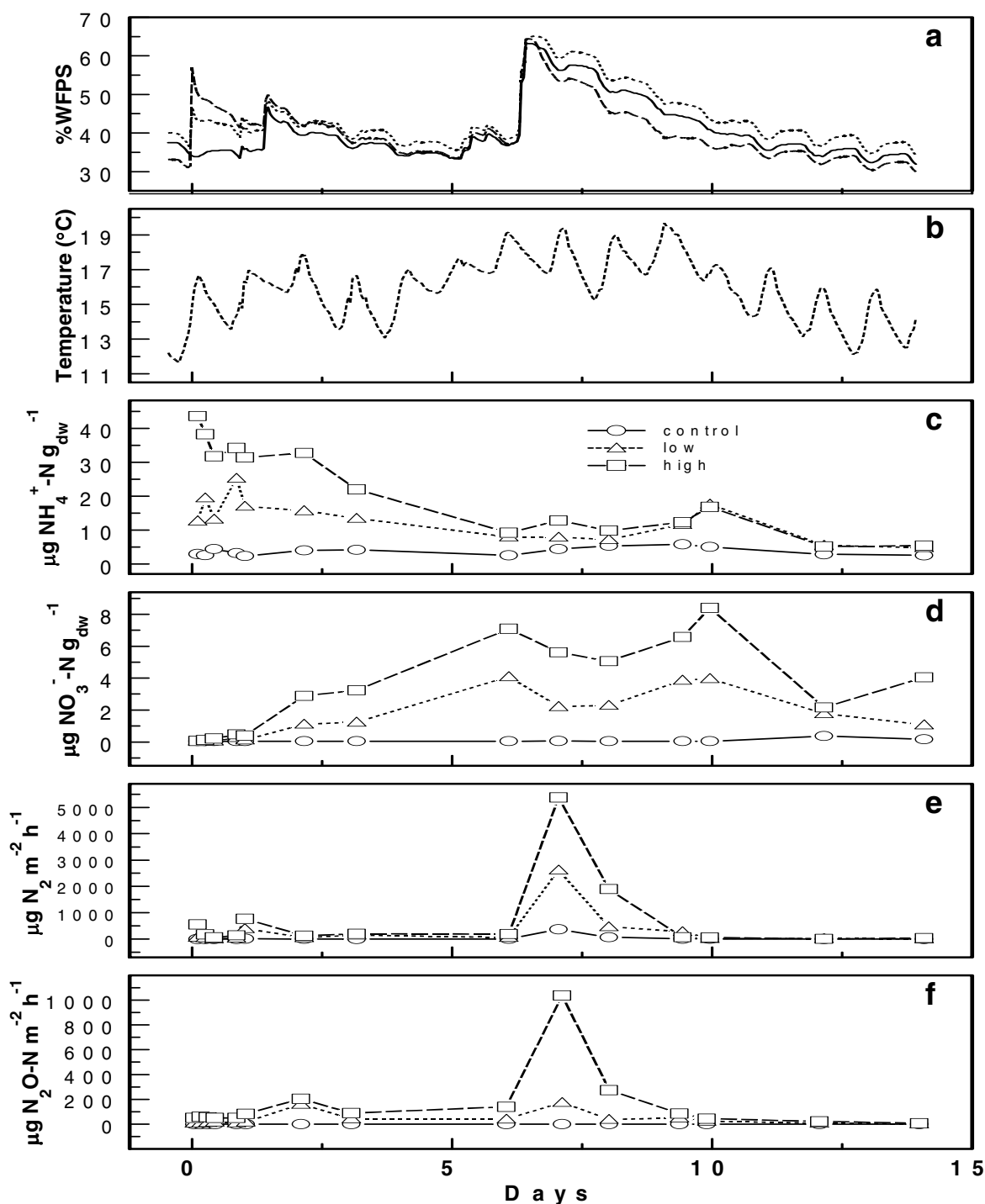


Figure 5.3.2. Swine waste fertilization experiment 2. Time course for change in: (a) percent water filled pore space (%WFPS); (b) temperature; (c)  $\text{NH}_4^+-\text{N}$ ; (d)  $\text{NO}_3^--\text{N}$ ; and soil emissions of (e)  $\text{N}_2$ ; and (f)  $\text{N}_2\text{O}-\text{N}$  from plots amended with low and high doses (as per Figure 5.3.1) of liquid swine effluent. Data points for all variables are the average of duplicate plots at each dose except temperature ( $n = 3$ ). The SD for duplicate plots at the high and low doses (error bars not shown) averaged: 2.9 and 4.3  $\mu\text{g NH}_4^+-\text{N g}_{\text{dw}}^{-1}$ , 0.9 and 1.3  $\mu\text{g NO}_3^--\text{N g}_{\text{dw}}^{-1}$ , 148.3 and 493.9  $\mu\text{g N}_2-\text{N m}^{-2} \text{ h}^{-1}$  and 34.3 and 104.0  $\mu\text{g N}_2\text{O}-\text{N m}^{-2} \text{ h}^{-1}$ .

### 5.4. Experiment 3

Immediately following the third experimental fertilization,  $\text{NH}_3$  emissions from soils increased to  $38 \text{ mg m}^{-2} \text{ h}^{-1}$  and  $74 \text{ mg m}^{-2} \text{ h}^{-1}$  in the plots amended with 1.25 cm and 2.5 cm of swine waste, respectively (Figure 5.4.1). In contrast, emission from the control plot was  $16 \text{ mg m}^{-2} \text{ h}^{-1}$ , or 42% and 22% of the emission from the fertilized plots. Consistent with the previous experiments, the period of elevated emission was brief. Ammonia fluxes declined to about  $10 \text{ mg m}^{-2} \text{ h}^{-1}$  in all of the fertilized plots 5 h after fertilization. Thereafter,  $\text{NH}_3\text{-N}$  fluxes from the fertilized plots were indistinguishable from that of the control plot.

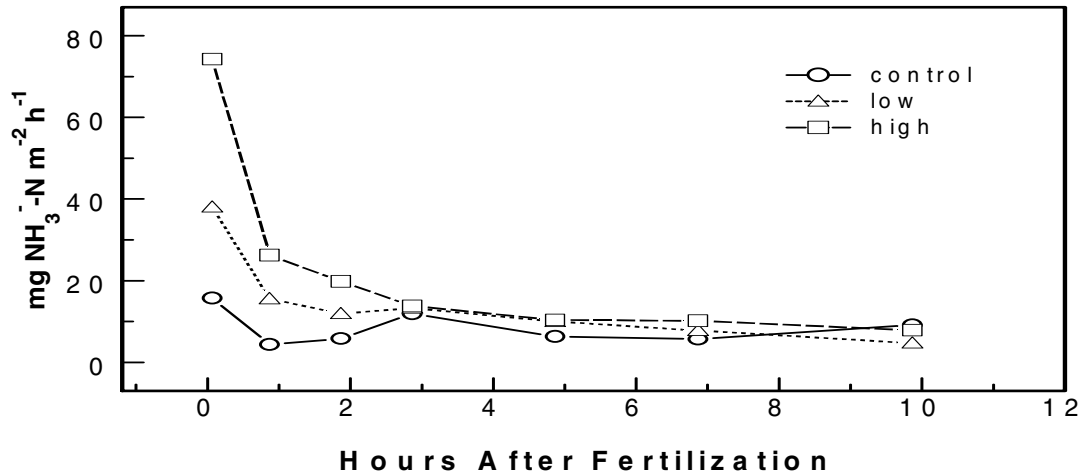


Figure 5.4.1. Time course for  $\text{NH}_3$  volatilization in experiment 3. Low and high doses represent 1.25 and 2.5 cm additions of waste, which correspond to area-based N additions of  $4.0$  and  $8.5 \text{ g m}^{-2}$ . The SD for the duplicate plots (error bars not shown) at each time point averaged  $2.6 \text{ mg NH}_3\text{-N m}^{-2} \text{ h}^{-1}$  and  $4.3 \text{ mg NH}_3\text{-N m}^{-2} \text{ h}^{-1}$  for the low and the high rate plots, respectively.

Soil percent water filled pore space responded immediately to the third simulated spray event by increasing from a pre-fertilization value of around 25% to about 34% and 42% in the low and high dose plots, respectively (Figure 5.4.2a). As soils drained over the next 6 d, moisture values in all fertilized plots declined to about 18% WFPS. Soils showed

increases in %WFPS on day 12 following a 4 cm rain on the preceding day. In total, soil moisture values fluctuated over about 25%WFPS throughout the 16 d experiment. A total of 4.8 cm of rain was received during the experimental period. This value excludes a malfunction of the rain gauge on day 9 where an apparent 9 cm of rainfall was not evident in continuously recorded soil moisture data (Figure 5.4.2a).

Fertilized soils showed diurnal temperature variations of 3 to 6°C and generally increased throughout the 16 d observational period (Figure 5.4.2b). The lowest soil temperature of 24°C was observed the morning of day 1 in response to an air temperature of 14°C the previous evening. The maximum soil temperature of 31°C was observed on days 6, 11 and 16, corresponding to air temperatures of around 40°C. Thus, soil temperatures varied over about 7°C during the entire 16 d observational period.

Soil  $\text{NH}_4^+$ -N rapidly increased from a pre-fertilization concentration of  $4 \mu\text{g g}_{\text{dw}}^{-1}$  to roughly  $23 \mu\text{g g}_{\text{dw}}^{-1}$  in both the high and low dose plots immediately following waste application (Figure 5.4.2c). Ammonium concentrations fluctuated over the next 6 h at both doses, but by day 12, soil  $\text{NH}_4^+$ -N concentrations had declined to about  $5 \mu\text{g g}_{\text{dw}}^{-1}$  in both treatments. However,  $\text{NH}_4^+$ -N concentrations still exceeded the  $3 \mu\text{g g}_{\text{dw}}^{-1}$  of the control plot. As in the first two fertilizations, changes in the control plot tracked those of the fertilized plots, suggesting that the fluctuations were unrelated to waste amendment. Once again, soil  $\text{NH}_4^+$ -N levels were consistently highest in the high dose treatment, and  $\text{NH}_4^+$ -N in the control plot was consistently lower than the waste-amended plots.

Soil  $\text{NO}_3^-$ -N concentrations started increasing in the high dose plot about 1 d after waste amendment from a pre-fertilization value of roughly  $1 \mu\text{g g}_{\text{dw}}^{-1}$  to a maxima of 11 and  $8 \mu\text{g g}_{\text{dw}}^{-1}$  on day 6 and day 12 in the high and low dose treatments, respectively (Figure

5.4.2d). Nitrate concentrations in the low dose treatment were only slightly elevated relative to controls until the day 12 increase, when the concentration exceeded that of the high dose treatments, which was declining. In accord with the entire data for fertilized plots in previous experiments, the temporal decrease in soil  $\text{NH}_4^+\text{-N}$  at the high dose coincided with the increase in  $\text{NO}_3^-\text{-N}$ . However, this pattern was not readily apparent for the low dose treatment.

Maximum denitrification rates of 149 and 386  $\mu\text{g m}^{-2}\text{h}^{-1}$  were observed immediately following waste addition in the low and high dose treatments (Figure 5.4.2e). Small peaks in denitrification rates were also observed in the low dose treatment on days 1, 6 and 12, while peaks in the high dose treatment occurred on days 3 and 12. Fluxes of  $\text{N}_2$  on these dates varied from approximately 40 to 150  $\mu\text{g m}^{-2}\text{h}^{-1}$  in the low dose plots and from about 100 to 400  $\mu\text{g m}^{-2}\text{h}^{-1}$  in the high dose plots. The day 12 pulses were likely related to the increase in % WFPS from precipitation the preceding day (Figure 5.4.2a), while the cause for other small increases was unclear. Rates of  $\text{N}_2$  emission from the control plot generally varied from 0 to 5  $\mu\text{g m}^{-2}\text{h}^{-1}$ . However, the flux increased to 50  $\mu\text{g m}^{-2}\text{h}^{-1}$  on day 13, which may have also been related to the day 11 rain.

Nitrous oxide emissions showed temporal variations similar to the  $\text{N}_2$  flux in fertilized plots (Figures 5.4.2e & f). Nitrous oxide fluxes increased immediately following waste addition in both treatments to maximum values of 35  $\mu\text{g m}^{-2}\text{h}^{-1}$  from the low dose plots and 76  $\mu\text{g m}^{-2}\text{h}^{-1}$  from the high dose plots. Smaller peaks were observed on days 2 and 6. All fluxes of  $\text{N}_2\text{O}$  were considerably less than those of the first and second experiment, and by day 12, were indistinguishable from that of the control plot, which showed little or no  $\text{N}_2\text{O}$  flux.

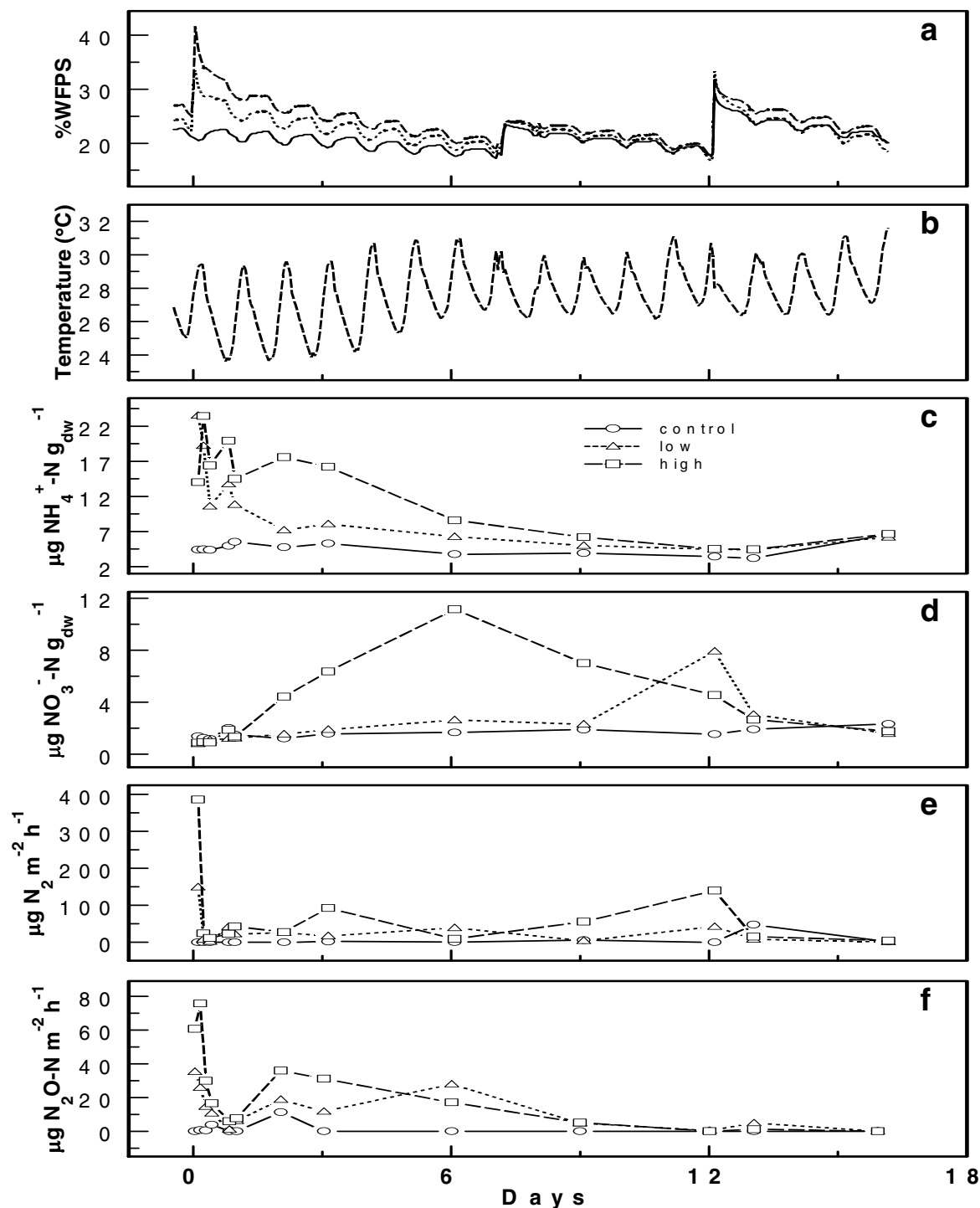


Figure 5.4.2. Swine waste fertilization experiment 3. Time course for change in: (a) percent water filled pore space (%WFPS); (b) temperature; (c)  $\text{NH}_4^+-\text{N}$ ; (d)  $\text{NO}_3^--\text{N}$ ; and soil emissions of (e)  $\text{N}_2$ ; and (f)  $\text{N}_2\text{O}-\text{N}$  from plots amended with low and high doses (as per Figure 5.4.1) of liquid swine effluent. Data points for all variables are the average of duplicate plots at each dose except temperature ( $n = 3$ ). The SD for duplicate plots at the high and low doses (error bars not shown) averaged: 2.1 and 3.4  $\mu\text{g NH}_4^+-\text{N g}_{\text{dw}}^{-1}$ , 0.9 and 1.5  $\mu\text{g NO}_3^--\text{N g}_{\text{dw}}^{-1}$ , 23.0 and 81.1  $\mu\text{g N}_2-\text{N m}^{-2} \text{ h}^{-1}$  and 8.2 and 9.4  $\mu\text{g N}_2\text{O}-\text{N m}^{-2} \text{ h}^{-1}$ .

## 5.5. Microbial biomass-N

Microbial biomass showed a consistent pattern of change in all experiments (Table 5.5.1). Fertilization at the high dose resulted in a ~100% increase in microbial biomass at day 3, while a proportionately lesser increase of about 50% was observed in low dose plots. Microbial biomass declined in all plots by the termination of all three experiments, but, relative to the control, remained elevated by about and 22 to 30% and 35 to 50% in the low and high dose treatments.

The mass of N immobilized by the end of the ~two week experiments varied from 0.6 to 0.9 g m<sup>-2</sup> in the low dose treatments and from 0.7 to 1.2 g m<sup>-2</sup> in the high dose treatments. This represented 13 to 22% and 8 to 14% of the total waste-N applied in the respective treatments.

Table 5.5.1. Percent increase in microbial biomass relative to control plots at selected time points during three liquid swine waste fertilization experiments. Waste was applied at two doses (high and low) corresponding to 1.25 and 2.5 cm additions. See text for mass of N applied during each experiment. The last sampling date for each experiment corresponded to the termination of that experiment.

Experiment		1		2		3	
Day		3	19	3	14	3	16
Microbial Biomass	Low	63	27	77	22	47	30
	High	109	35	141	35	94	50

## 5.6. Plant biomass-N

Relative to the control, plant biomass also increased in waste-amended plots (Table 5.6.1), but the response was more variable than for microbial biomass. The low dose plots in experiment 1 showed only a 4% increase in plant biomass relative to the control, compared



with the 43% increase observed in the high dose plots. The mass of swine effluent-N assimilated into plant biomass varied from 2.0 g m<sup>-2</sup> in the low dose treatment of experiment 1 to 9.7 g m<sup>-2</sup> in the high dose plot of experiment 2. A dose response was evident in experiments 1 and 2, as the mass of effluent-N assimilated into plant biomass was about two-fold or more higher for the high versus low dose plots. In contrast, plant assimilation accounted for about 4.5 g N m<sup>-2</sup> at both application levels in experiment 3. Overall, incorporation into plant biomass represented 46 to 117% of the waste-N.

Table 5.6.1. Percent increase in plant biomass relative to control plots at selected time points during three liquid swine waste fertilization experiments. Waste was applied at two doses (high and low) corresponding to 1.25 and 2.5 cm additions. See text for mass of N applied during each experiment. The last sampling date for each experiment corresponded to the termination of that experiment.

Experiment		1		2		3	
Day		3	19	3	14	3	16
Plant Biomass	Low	-	4	-	33	-	69
	High	-	43	-	87	-	67

### 5.7. N transport below the active 20 cm soil zone

Fertilizer-N was transported below the microbially active surface 20 cm of soil at both loading rates for each experiment (Table 5.7.1). The mass of N loss was roughly similar at both loading rates for Experiment 1, but was seemingly dose-related during the two successive trials, as the higher dose showed higher N loss. Leaching losses varied over a factor of about 20, with values ranging from 0.2 g N m<sup>-2</sup> for the low dose in Experiment 3 to 4.5 g N m<sup>-2</sup> for the high dose in Experiment 2. The mass of N transported below the active zone appeared to be unrelated to rainfall, as largest values were associated with relatively

little rainfall (6.4 cm). However, the largest mass losses did correspond with the highest loading rate in all experiments.

Table 5.7.1. Rainfall received, fertilizer-N loading rate and N lost to leaching below the 20 cm microbially active soil zone for each experiment. All N loss rates are the means of duplicate plots at each loading rate.

Experiment	Date	Rainfall (cm)	Loading rate (g N m <sup>-2</sup> )	Leaching loss (g N m <sup>-2</sup> )
1	12 Aug 01	18.1	4.0	0.5
			8.4	0.6
2	25 Mar 02	6.4	6.1	1.2
			12.9	4.5
3	7 Aug 02	4.8	4.0	0.2
			8.5	0.8

## 5.8. Mass balance

The relative importance of sink terms for applied N was assessed by a mass balance approach at the termination of each experiment (Table 5.8.1). Assimilation by plants was consistently the most important sink, accounting for 46 to 117% of the liquid waste when the entire data were considered. Losses to the atmosphere through denitrification and volatilization were relatively minor. Denitrification was negligible in all experiments, accounting for  $\leq 2\%$  of the N applied, while NH<sub>3</sub> volatilization accounted for only 5 to 17% of the N loss. The storage term was important in experiment 1 only, representing 30 to 50% of the applied N in this trial, but only about 10% on other dates. Microbial immobilization and leaching loss were of about equal importance, accounting for roughly 10 to 15% of the

applied-N, with the exception of experiment 2 when leaching loss assumed greater significance.

Table 5.8.1. Fate of applied N in three experimental fertilizations of agricultural fields with liquid swine waste at two loading rates. All percentages are the means of duplicate plots at each loading rate.

Experiment	12-Aug-01		25-Mar-02		7-Aug-02	
Loading rate (g N m <sup>-2</sup> )	4.0	8.4	6.1	12.9	4.0	8.5
Percent of applied N						
Denitrification (N <sub>D</sub> )	2	1	1	1	0	0
Plant assimilation (N <sub>P</sub> )	49	46	70	75	117	52
Microbial immobilization (N <sub>M</sub> )	22	14	13	9	14	8
Leaching loss (N <sub>G</sub> )	14	15	19	35	6	9
NH <sub>3</sub> volatilization (N <sub>V</sub> )	5	6	17	17	13	14
Storage (N <sub>in F-0</sub> )	50	30	10	10	14	9
Total	142	112	130	147	164	92

## CHAPTER VI

### DISCUSSION

#### 6.1. Swine effluent TN and $\text{NH}_4^+\text{-N}$

Concentrations of total-N and  $\text{NH}_4\text{-N}$  in the liquid swine waste used in my experimental fertilizations (Table 5.5.1) agree with previous chemical analyses of lagoonal swine waste. Safely *et al.* (1992) reported average TN and  $\text{NH}_4^+\text{-N}$  concentrations of 381 and 308  $\text{mg l}^{-1}$  for three mid-summer experiments at an eastern North Carolina farm. Likewise, Adeli *et al.* (2003) found TN and  $\text{NH}_4^+\text{-N}$  concentrations of 341 and 280  $\text{mg l}^{-1}$  in liquid swine waste during two late summer studies on a commercial swine facility in Mississippi. Finally, Sharpe and Harper (2002) provide somewhat higher concentrations for TN and  $\text{NH}_4^+\text{-N}$ , 561 and 482  $\text{mg l}^{-1}$ , for lagoonal swine effluent at another eastern North Carolina facility.

The percentage of TN accounted for by  $\text{NH}_4^+\text{-N}$  in the effluent applied in my three experiments was about 75%. These results are consistent with previous reports for swine lagoons in this region, which show  $\text{NH}_4^+\text{-N}$  percentages ranging from 64 to 92% of the TN (Westerman *et al.* 1985; Safely *et al.* 1992; Zublena *et al.* 1996; Sharpe and Harper 1997, 2002; Adeli and Varco 2001; Adeli *et al.* 2003). This predominance of  $\text{NH}_4^+\text{-N}$  over organic-N reflects the efficiency of anaerobic decomposition in swine lagoons (Adeli and Varco 2001), whereby mineralization converts the majority of organic N to  $\text{NH}_4^+\text{-N}$  and lack of  $\text{O}_2$  prevents oxidation of the  $\text{NH}_4^+\text{-N}$  to  $\text{NO}_3^-\text{-N}$ .

## 6.2. Ammonia volatilization

The time trajectory, hourly rates and fractional loss of fertilizer-N to  $\text{NH}_3$  volatilization following animal waste applications to cropping systems are highly variable depending on waste characteristics, method of application, field conditions, and the technique employed to measure  $\text{NH}_3$  emission. However, all investigations consistently show maximum rates of  $\text{NH}_3$  efflux shortly after waste application and my study is consistent with these previous reports. Maximum  $\text{NH}_3$ -N efflux rates from my experimental plots were observed within 1 h after swine waste amendment in all three experimental fertilizations (Figures 5.2.1, 5.3.1 & 5.4.1). These transient  $\text{NH}_3$  fluxes agree with other studies from this region (Safely *et al.* 1992; Sharpe and Harper 1997, 2002), and elsewhere for this waste type (Hoff *et al.* 1981; Beauchamp *et al.* 1982; Lockyer *et al.* 1989; Pain *et al.* 1989; Bless 1991) that report large pulses of  $\text{NH}_3$  immediately upon application, followed by gradual returns to background fluxes within ~1 d.

Despite the common observation of peak  $\text{NH}_3$ -N emissions shortly after waste amendment, maximum rates differ among studies. My low dose plots subjected to total N loading rates of 40 to 60  $\text{kg ha}^{-1}$  produced maximum volatilization rates that varied from 0.4 to 0.8  $\text{kg NH}_3\text{-N ha}^{-1} \text{ h}^{-1}$  while the high dose plots amended with 84 to 129  $\text{kg ha}^{-1}$  showed rates about twice as high, with maxima varying from 0.7 to 1.9  $\text{kg NH}_3\text{-N ha}^{-1} \text{ h}^{-1}$ . My results compare favorably with a regional (Georgia) whole field spray study (Sharpe and Harper 1997), where somewhat lower total-N loading rates (13 to 18  $\text{kg ha}^{-1}$ ) gave maximum effluxes of 1.1 to 1.6  $\text{kg NH}_3\text{-N ha}^{-1} \text{ h}^{-1}$ . My maximum rates are also consistent with other reports involving swine waste addition in other locales, employing measurement technologies both similar and different from those used here. Hoff *et al.* (1981) showed a

maximum  $\text{NH}_3\text{-N}$  efflux of  $1.3 \text{ kg ha}^{-1} \text{ h}^{-1}$  from a similar microplot/closed chamber study in the Midwestern U.S. (Indiana), but the swine effluent loading rates of 273 to  $551 \text{ kg N ha}^{-1}$  exceeded those used here by a factor of 2 to 14. Pain *et al.* (1990) observed a maximum  $\text{NH}_3\text{-N}$  emission of  $1.5 \text{ kg ha}^{-1} \text{ h}^{-1}$  in the UK with a series of wind tunnels on small plots following swine slurry treatments at loading rates of about  $250 \text{ kg NH}_4^+\text{-N ha}^{-1}$ , which are 2 to 10 times those used here. Other investigations involving liquid swine waste fertilization (Lockyer *et al.* 1989; Pain *et al.* 1989; Thompson *et al.* 1990; Bless *et al.* 1991; Sharpe and Harper 2002) give higher  $\text{NH}_3$  emission maxima (ranging from 2.5 to  $12.1 \text{ kg ha}^{-1} \text{ h}^{-1}$ ) in European and North American studies involving wind tunnels and whole field fertilization.

Maximum  $\text{NH}_3\text{-N}$  effluxes similar to mine have also been shown following application of other liquid animal wastes. Beauchamp *et al.* (1982) showed maximum  $\text{NH}_3\text{-N}$  effluxes varying from approximately 1 to  $3 \text{ kg ha}^{-1} \text{ h}^{-1}$  immediately following whole field fertilizations with liquid cattle slurry at TN loading rates of 325 to  $387 \text{ kg ha}^{-1}$  in Ontario Canada, while Lockyer *et al.* (1989) observed maximum  $\text{NH}_3$  emissions of  $1 \text{ kg ha}^{-1} \text{ h}^{-1}$  following cattle slurry additions at TN loading rates similar to mine (75 and  $102 \text{ kg ha}^{-1}$ ).

Over the courses of my ~two week experiments,  $\text{NH}_3$  volatilization losses represented 7 to 22% of the swine effluent  $\text{NH}_4^+\text{-N}$  applied to the low dose plots and 10 to 22% of the  $\text{NH}_4^+\text{-N}$  applied to the high dose plots. These fractional losses corroborate well the values of about 10 to 15% given by Hoff *et al.* (1981) and Pain *et al.* (1990) involving liquid or slurried swine waste in closed chamber/microplot and wind tunnel studies, respectively, and are somewhat lower than the 15 to 30% loss reported by Sharpe and Harper (2002) in a whole field fertilization within several km of my study site. Overall, my fractional losses fall toward the low end of values given for many whole field and wind tunnel studies involving

slurried or liquid animal wastes, which center around 30 to 40% (Beauchamp *et al.* 1982; Lockyer *et al.* 1989; Thompson *et al.* 1990; Bless *et al.* 1991; Safley *et al.* 1992), but are as high as 82% (Sharpe and Harper 1997).

My fractional losses of  $\text{NH}_4^+$ -N to volatilization are likely underestimated relative to expected losses in whole field fertilizations. The regional standard for liquid swine waste application to agricultural soils involves the use of traveling big gun sprinkler systems or less common solid set sprinkler systems. Significant  $\text{NH}_3$  losses are incurred during these high-pressure spray events due to evaporation and offsite drift of waste droplets. For example, in a full-scale swine effluent spray event in Georgia, Sharpe and Harper (1997) demonstrated that out of 82% of the  $\text{NH}_4^+$ -N lost as  $\text{NH}_3$  within 24 h, 13% occurred due to evaporation and drift during application. Similarly, another study at a Georgia facility (Safley *et al.* 1992) noted that 62 to 100% of the  $\text{NH}_4$ -N lost from liquid swine waste during volatilization occurred as volumetric loss throughout the spray event. My method of pouring liquid swine effluent into small field plots does not represent the regional industry standard of large-scale sprinkler application to cropping systems. Instead, my treatment method was more representative of more conservative techniques such as trail hose application, injection or tillage incorporation (Sommer and Ersboll 1994; Sommer *et al.* 1997; and Dendooven *et al.* 1998b), whereby losses via small droplet production and drift are eliminated.

An additional factor that may have reduced volatilization in my study is soil texture. Anaerobic liquid lagoon effluents have lower organic dry matter contents than animal slurries (Kirchman 1994), and will rapidly infiltrate a sandy soil upon application if directly applied to the surface. This was clearly evident in my study as liquid effluent permeated the soil surface within a few minutes of fertilization. In comparison to soils with high organic

matter, sandy soils improve the ionic mobility of an effluent by way of a reduced CEC (USDA 1992). Accordingly, this rapid infiltration may have reduced  $\text{NH}_3$  fluxes in all three of my experiments. Rather than being volatilized, the majority of the applied N in my three experiments was made available for plant and microbial uptake or leaching.

### 6.3. Denitrification/ $\text{N}_2\text{O}$

Much like  $\text{NH}_3$ -N volatilization, the time courses, hourly rates, and fractional losses of fertilizer-N to denitrification ( $\text{N}_2 + \text{N}_2\text{O}$ ) following animal waste applications are highly variable depending on waste characteristics, loading rates, field conditions, and the duration and frequency of flux measurements. Because denitrification is confined to heterogeneously distributed anaerobic microsites in arable soils, it has the largest spatial and temporal variability of any of the microbial processes involved in N cycling (Tiedje *et al.* 1989), with coefficients of variation exceeding 100% for multiple measurements at the same site (Parkin and Robinson 1989).

The rapid post-fertilization increase in denitrification-N fluxes from my plots was clearly dose-related, as soils receiving 40 to 60 kg total-N  $\text{ha}^{-1}$  showed emissions ranging from 70 to 149  $\mu\text{g N}_2 \text{ m}^{-2} \text{ h}^{-1}$ , while the soils amended with 85 to 130 kg total-N  $\text{ha}^{-1}$  generated initial  $\text{N}_2$  peaks of 373 to 550  $\mu\text{g m}^{-2} \text{ h}^{-1}$ . These rates are 4 to 8-fold lower than the 600 and 1720  $\mu\text{g N}_2 \text{ m}^{-2} \text{ h}^{-1}$  given by Fischer and Whalen (2005) in a study involving liquid swine waste additions at comparable loading rates to intact soil cores at another eastern North Carolina swine production facility.

Denitrification rates returned to baseline levels in  $\leq 1$  d in my study as well as in these previous reports. In contrast, surface applications of slurried animal wastes to intact



soils (Thompson *et al.* 1987; Loro *et al.* 1997; Paul and Zebarth 1997a, b; Ellis *et al.* 1998) have generally shown enhanced rates of denitrification lasting weeks or months. Overall, high variability in the duration of the immediate post-application denitrification pulse for organic fertilizers indicates that the factors driving enhanced emissions are complex and interact nonlinearly, discouraging generalizations.

The immediate response of soil denitrifiers to liquid swine effluent application suggests that fertilizer addition rapidly promoted soil conditions favoring denitrification. Specifically, the immediate infiltration of liquid swine effluent rapidly increased soil WFPS (Figures 5.2.2a, 5.3.2a, & 5.4.2a) and likely produced local anaerobic microzones. Increased rates of denitrification in normally aerated soils are generally associated with increases in soil moisture that impede O<sub>2</sub> diffusion by reducing air-filled pore spaces and enhance O<sub>2</sub> consumption by stimulating microbial respiration (reviewed by Firestone 1982; Granli and Bockman 1994; Beauchamp 1997; and Mosier 1998). Further, rapid waste infiltration quickly established contact between the extant microbial community and a substrate immediately available for nitrification (NH<sub>4</sub><sup>+</sup>-N) in aerobic zones. Finally, the waste provided labile C substrates (Whalen 2000) such as volatile fatty acids (Paul and Beauchamp 1989a; Sommer *et al.* 1996) that support high rates of coupled nitrification-denitrification (Peterson *et al.* 1991; Nielsen and Revsbeck 1994) at the oxic-anoxic boundary within soil aggregates. Following these initial increases in denitrification, emissions quickly settled to pre-fertilization levels because the coarse soil texture encouraged rapid drainage, ensuring that anaerobic conditions necessary for denitrification were temporary.

Rates of denitrification following rainfall were frequently higher than the initial post-fertilization spikes. Fluxes of roughly 850 µg N<sub>2</sub> m<sup>-2</sup> h<sup>-1</sup> were observed for both doses

following rainfall on days 2, 7, and 10 of experiment 1 (Figure 5.2.2e), while fluxes of about  $2600 \mu\text{g N}_2 \text{ m}^{-2} \text{ h}^{-1}$  and  $5,300 \mu\text{g N}_2 \text{ m}^{-2} \text{ h}^{-1}$  were associated with the low and high dose treatments on day 7 in response to 1.6 inches of rain in experiment 2 (Figure 5.3.2e). Rainfall-induced peaks in denitrification, which lasted 1 to 4 d, persisted longer than the fertilization-induced peaks (Figures 5.2.2e, 5.3.2e, & 5.4.2e). Collectively, higher peaks of longer duration following rainfall suggest  $\text{N}_2$  emission is a consequence of denitrification uncoupled to nitrification. Accumulated soil  $\text{NO}_3^-$ -N (Figures 5.2.2d, 5.3.2d & 5.4.2d) from ongoing nitrification provided nitrogenous substrate, while increased soil moisture promoted the development of anaerobic microzones. In contrast, denitrification immediately following waste application was dependent in part on  $\text{NO}_3^-$ -N supply from nitrification, as pre-fertilization  $\text{NO}_3^-$ -N concentrations were low, at  $<0.5 \mu\text{g g}_{\text{dw}}^{-1}$  soil (Figures 5.2.2d & 5.3.2d). Denitrification uncoupled from nitrification has been reported in laboratory (Nielsen and Revsbech 1998) and field (Whalen 2000) studies involving soil-applied liquid cattle manure and swine waste, respectively. Episodic pulses of denitrification are likely associated with each post-fertilization rainfall as long as  $\text{NO}_3^-$ -N persists in the soil and, depending on weather, are probably a more important loss term for N than the initial, post-application burst.

Within each experiment, rates of denitrification closely tracked % WFPS (Figures 5.2.2a&e, 5.3.2a&e, & 5.4.2a&e), with maximum rates observed in experiments 1 and 3 at ~60 to 65% WFPS. This is in close accord to previous observations involving liquid organic wastes that show maximum rates above 60% WFPS (Linn and Doran 1984), although the optimum moisture content is likely dependent on soil texture. Fischer and Whalen (2005) observed similar correlations between moisture and denitrification, as their homogenized

sandy soils from a NC swine waste-receiving field approached or exceeded water-holding capacity. Denitrification rates were 39-fold higher for their saturated soils compared to the soils below saturation. Lowrance *et al.* (1998) also showed significant correlations between denitrification and soil moisture following liquid dairy amendments to loamy sand plots of bermudagrass. Maximum denitrification rates ( $2,080$  to  $8,250 \mu\text{g N}_2 \text{ m}^{-2} \text{ h}^{-1}$ ) exceeded my maxima by about a factor of two, which may be related to differences in soil physicochemical characteristics.

Applications of liquid swine effluent to soils also produced transient  $\text{N}_2\text{O}$  emissions (an additional endproduct of denitrification) that generally showed higher spatiotemporal variability than  $\text{N}_2$  emissions (Figures 5.2.2f, 5.3.2f, & 5.4.2f). The immediate  $\text{N}_2\text{O}$  fluxes were low ( $8$  to  $115 \mu\text{g m}^{-2} \text{ h}^{-1}$ ), and returned to prefertilization levels within 1 d of application. Maximum  $\text{N}_2\text{O}$  fluxes closely tracked the initial  $\text{N}_2$  peaks in all trials, again pointing to the close coupling between soil moisture and denitrifying activity. Maximum post-fertilization  $\text{N}_2\text{O}$  fluxes in my study were generally lower than the maxima observed elsewhere for liquid swine waste. Peak fluxes from my three fertilization experiments averaged  $559 \mu\text{g N}_2\text{O-N m}^{-2} \text{ h}^{-1}$ , which agrees with the average maxima of  $650 \mu\text{g N}_2\text{O-N m}^{-2} \text{ h}^{-1}$  reported by Fischer and Whalen (2005). However, this mean is around five times lower than the maxima of  $2,500$  and  $3,250 \mu\text{g N}_2\text{O-N m}^{-2} \text{ h}^{-1}$  inferred from the data of Sharpe and Harper (1997), and Sharpe and Harper (2002) respectively, during two field studies involving triplicate sprinkler applications of swine effluent on Coastal Plain soils in GA and NC. Moreover, this  $\text{N}_2\text{O}$  maximum is roughly 16.5 times lower than the maximum  $\text{N}_2\text{O}$  flux of  $9,240 \mu\text{g N m}^{-2} \text{ h}^{-1}$  reported by Whalen *et al.* (2000) during an investigation in North Carolina involving multiple swine effluent fertilizations interspersed with rainfall. Nitrous

oxide is an endproduct of denitrification, but is also a byproduct of nitrification (Bandibas *et al.* 1994). Close coupling between denitrification and N<sub>2</sub>O spikes suggest that denitrifiers dominate N<sub>2</sub>O production during pulsed emission while nitrifiers provide a constant, baseline N<sub>2</sub>O flux.

My three soil fertilization experiments indicate losses to denitrification of 0 to 2.06% of the applied N from liquid swine waste (Table 5.8.1). This percentage of N lost to denitrification compares favorably with the 0.57 to 1.63% reported by Fischer and Whalen (2005) for an NC receiving field. Comfort *et al.* (1990) also reported comparable denitrification-N loss (1.0 to 1.3%) for relatively dilute cattle slurry (5% total solids) injected into repacked cores. Nonetheless, my data for N loss to denitrification are generally much lower than previous reports for other types of animal wastes. Losses vary from 0.6 to 21.3% for slurried cattle and pig wastes (Thompson *et al.* 1987; Thompson 1989; Loro *et al.* 1997; Paul and Zebarth 1997b; Ellis *et al.* 1998; Acara *et al.* 1999), 4.2 to 18.8% for solid manures (Loro *et al.* 1997; Mahmood *et al.* 1998; Mogge *et al.* 1999) and are as high as 11 to 37% for liquid dairy cattle waste (Lowrance *et al.* 1998). Generally lower fractional losses of applied N to denitrification for liquid swine waste than slurried and solid wastes likely reflects the lower mineralization potential of lagoonal effluent (Table 5.5.1) as well as the coarser soil texture characteristic of this region (Brandon 1986). Further, many of these estimates involve longer-term (weeks to seasonal) measurements. In a review of N emissions from agroecosystems, Bouwman (1996) noted that fractional loss estimates increased with increasing duration of the observational period. Fractional loss values given here and by Fischer and Whalen (2005) for liquid swine waste are undoubtedly underestimates.

The fractional loss of applied N to  $\text{N}_2\text{O}$ -N varied from 0 to 0.62%. These estimates compare well with the values of 0.05 to 2.3% given by Whalen (2000) and Whalen *et al.* (2000) from short term (1 to 3 weeks) static chamber-derived  $\text{N}_2\text{O}$  field measurements (Whalen 2000; Whalen *et al.* 2000), as well as the values of 0.47 to 0.84% reported for short term (8 days) intact core-derived measurements (Fischer and Whalen 2005) from North Carolina soils. Consistent with estimates for fractional total gaseous N loss to denitrification, these percentages fall toward the low end of reports for annual  $\text{N}_2\text{O}$  emissions for a variety of mineral and organic fertilizers (range to 6.8%; summarized by Bouwman 1994; Dobbie and Smith 2003). Again, the short observational period ensures that fractional loss values are underestimates.

#### **6.4. Plant uptake**

My values of around 45 to 70% for recovery of applied N as plant material (Table 5.8.1) in short term experiments fall midway among previous regional reports involving liquid swine waste application to coastal bermudagrass in experiments of variable duration. Several multiyear investigations for repeated waste application give values that center around 70% (Cummings *et al.* 1975; Westerman *et al.* 1977; Burns *et al.* 1985; Burns *et al.* 1990). Otherwise, Adeli *et al.* (2003) showed an overall recovery of 40% of waste-N following two applications of swine effluent over a growing season, but the maximum recovery of 51% was observed in the fall fertilization. Liu *et al.* (1997) reported even lower dry matter N recoveries of 13 to 32% over the course of a 2y study in Alabama.

Decreased efficiency of recovery appears to be loosely correlated with increased loading rates. Studies cited above that report ~70% recovery have annual loads of 300 to 400

kg N ha<sup>-1</sup>, somewhat lower than the 400 to 800 kg N ha<sup>-1</sup> my sites would experience if it is reasonably assumed that fertilization occurred monthly for an 8 mo growing season. Lowest dry matter N recoveries (20 and 13%) of Liu *et al.* (1997) were associated with loading rates of 1120 and 2240 kg N ha<sup>-1</sup>, with a 32% recovery observed at 560 kg N ha<sup>-1</sup>. However, the 40% recovery of Adeli *et al.* (2003) was observed at a loading rate of 300 kg N ha<sup>-1</sup> that was elsewhere affiliated with ~70% recovery. Studies aimed at determining the maximum efficiency of N recovery at harvest recommend annual application rates of 335 to 550 kg N ha<sup>-1</sup> (Westermann 1977; King *et al.* 1990; Adeli and Varco 2001). However, high variability in recovery indicates that many additional factors such as physical soil characteristics, dosage per application and variations in weather influence the efficiency of bermudagrass recovery of waste N in this geographical region.

## **6.5. Microbial biomass**

Immobilization represented an important sink for swine waste-N, as 8 to 22% of the added effluent was incorporated into microbial biomass (Table 5.8.1). Liquid phase waste ensures immediate post-application transport to microbially active soil zones, while rapid (within 3 d) immobilization of waste-N points to a highly active microbial population. To my knowledge, this is the first study explicitly relating microbial community dynamics to liquid swine waste application. Nonetheless, my data are consistent with direct (<sup>15</sup>N) observations of accelerated microbial immobilization of N following fertilization with slurried manures (Persson 1988; Kirchmann 1989) as well as reported increases of 50 and 140% (Fauci and Dick 1994) in microbial biomass relative to the control for greenhouse soils amended with poultry and beef wastes, respectively.

Soils managed with long-term solid or slurried organic fertilizers generally have larger and more active microbial populations than those managed with mineral fertilizers (Bolton *et al.* 1985; McGill *et al.* 1986; Dick *et al.* 1988; Alef *et al.* 1988; Anwarzay *et al.* 1990) because of the larger C:N ratios associated with organic wastes. My results suggest that this generalization can be extended to liquid swine waste, although the C:N ratio is lower than the value of ~10:1 which is frequently associated with solid and slurried organics (see review by Kirchmann 1994). Although liquid swine waste has a relatively low mineralization potential, the standard management practice of repeatedly fertilizing receiving fields ensures a continuous C and N supply for microbes in a manner similar to the slower decomposition of solids and slurries.

#### **6.6. N transport below the active soil zone**

Inorganic-N concentrations showed an increase below the active soil zone in all fertilized plots at the termination of all experiments relative to day 0 values (Tables 5.7.1, and 5.8.1). The mass of N draining below my defined 0 to 20 cm active zone provides an estimate of the maximum waste-N lost to subsurface transport. Although, ~95% of the microbial activity has been shown to be localized in the active zone of NC spray fields (Whalen *et al.* 2000), roots of coastal bermudagrass extend to a greater depth and can potentially withdraw additional waste-N. Remaining, unassimilated fertilizer-N below the active zone is likely subject to downward transport to shallow groundwater at this site and additional, lateral subsurface flow in areas of greater relief. My observations are consistent with previous reports of high  $\text{NO}_3^-$ -N concentrations ( $>5 \text{ mg N l}^{-1}$ ) in subsurface drainage waters and wells beneath receiving fields in these regionally sandy soils (Evans *et al.* 1984;

Mikkelsen 1995; Gilliam *et al.* 1996; Stone 1998; Sloan *et al.* 1999; Karr *et al.* 2001; Israel *et al.* 2005). However, the susceptibility of a swine production facility to groundwater contamination and severity of the problem must be evaluated on a case-by-case basis (e.g. Stone *et al.* 1998).

My limited data suggest that the mass of N lost to leaching below the active zone is unrelated to the rainfall. It is more likely that timing of rainfall rather than absolute volume in the post-application observational period is a stronger determinant of leaching loss of fertilizer-N. Nitrate is more mobile than  $\text{NH}_4^+$ -N, which is frequently substituted for common interlayer cations within the lattices of soil particles (Stevenson and Cole 1999). The potential for leaching then increases with time after fertilization, as applied waste is increasingly transformed into  $\text{NO}_3^-$ -N by nitrification. Nitrate loss to leaching is apparent in experiments 1 and 2, where accumulated  $\text{NO}_3^-$ -N declines and % WFPS increases (Figs. 5.2.2 & 5.3.2, panels a & d), in response to rainfall events of 7 and 4 cm, respectively on day 6 of both experiments. Residual N at the termination of an experiment (e.g Experiment 1; Figure 5.2.2c & d) is also subject to continued nitrification and down-profile transport during subsequent rains.

## **6.7. N mass balance**

My study gives a comprehensive N mass balance for agricultural soils amended with liquid swine effluent, as I have attempted to assess most important N reservoirs, transfers and transformations in receiving soils for a post application period sufficient to allow most rates of transformation and the storage term to approach pre-fertilization values. The single previous effort (King *et al.* 1985) was not aimed at a complete mass balance, and focused on



plant recovery and the soil storage term. The authors reported recovery of 34 to 73% of applied N in plant material and total recovery (defined as plant + soil-N) of 44 to 58% of applied-N in a 6 yr study involving annual loading rates of 335 to 1340 kg N ha<sup>-1</sup>. Partial mass balances have also been attempted for cattle slurry applied to grasslands. In a 3 month study, Thompson *et al.* (1987) reported recoveries of 49 to 109% of cattle slurry N as plant material when waste was surface-applied or injected into grassland soils as well as 1 to 31% and 2 to 21% losses to volatilization and denitrification. Estavillo *et al.* (1997) showed 16 to 70% incorporation of applied N into the cover crop, 17 to 78% immobilization and losses of 2 to 5% and 3 to 11% to leaching and denitrification. The relative importance of immobilization and plant assimilation showed high interannual variability in this 2 yr study.

My attempt at assessing all major terms in the budget helped ensure rigorous evaluation of the error of closure and I consistently accounted for a higher mass of N than was applied (Table 5.8.1). As discussed above, my estimates for most terms fall within the range of values reported for this waste management practice except perhaps the percent loss to volatilization. My estimates of 5 to 17% loss to volatilization are low relative to the 23 to 69% given by Sharpe and Harper (1997, 2002) in whole field fertilization experiments. If my data are underestimates, the error term will increase. This is not likely, however.

Gaseous emissions are affected by waste characteristics, climate and physicochemical soil properties (Misselbrook *et al.* 1996; Lowrance *et al.* 1998) that differ among sites. The estimate of 23% loss to volatilization by Sharpe and Harper (2002) is reasonably close to my estimates, but was reported for a receiving field characterized by soils of higher clay content where localized ponding was observed (Whalen 2000) for several hours following waste application. Rapid infiltration of liquid waste in my soil will decrease volatile loss.

Ammonia volatilization loss for cattle slurry injected into grassland soil was only 0.9% compared to 20 to 31% for surface applied material (Thompson *et al.* 1987).

Other sources of error may have contributed to the error of closure in my experiments. I did not account for atmospheric N inputs as rainfall or dry deposition. No data exist for dry deposition in the study area. However, this is likely a negligible source of N input to my plots during the ~2 wk observational period of my experiments, assuming the maximum value of 136 kg N ha<sup>-1</sup> given in a review of observations for Europe (Pearson and Stewart 1998). Likewise, direct precipitation is a minor input. Using an average concentration of 1.25 mg N L<sup>-1</sup> (NO<sub>3</sub><sup>-</sup> + NH<sub>4</sub><sup>+</sup>) in rainfall recorded at the NADP observatory in Clinton, NC ([nadp.sws.uiuc.edu/trends](http://nadp.sws.uiuc.edu/trends)) and my precipitation volumes (Table 5.7.1), I calculate inputs of 0.08 to 0.22 g N m<sup>-2</sup> via precipitation for the three experiments. Incorporation of these values into the mass balances improves the mass balances an average of only 2%. The largest source of error may involve transformations of endogenous organic-N, a reservoir that was not assessed in this study. Organic-N may comprise 96 to 98% of the total-N pool in soils (Bardgett 2005). The organic-N pool may amount to 3400 kg N ha<sup>-1</sup> 15 cm<sup>-1</sup> in agroecosystems (Meisinger and Randall 1991), dwarfing the total-N loads of 40 to 129 kg ha<sup>-1</sup> experimentally added as swine effluent. Mineralization of a small fraction of this reservoir and subsequent immobilization or plant assimilation can lead to budgetary errors.

## **6.8. Summary and conclusions**

It is impossible to assess the error involved in individual terms of the mass balances (Table 5.8.1). I therefore assumed that the error was proportional to the magnitude of each term in scaling down my overall N recovery to force a mass balance (e.g. total = 100%) in an

effort to evaluate the fate of land-applied liquid swine waste-N at the study site (Figure 6.8.1). Roughly half (52%) of the waste-N is sequestered in plant material that can be removed from the site and recycled into livestock, while a smaller percentage (10%) remains in the soil as microbial biomass. A total of 22% is involved in unintended offsite transport. This is about equally divided between losses to the atmosphere via volatilization plus denitrification (10%) and leaching loss (12%). The 16% of applied waste remaining in soil storage is subject to any of these fates.

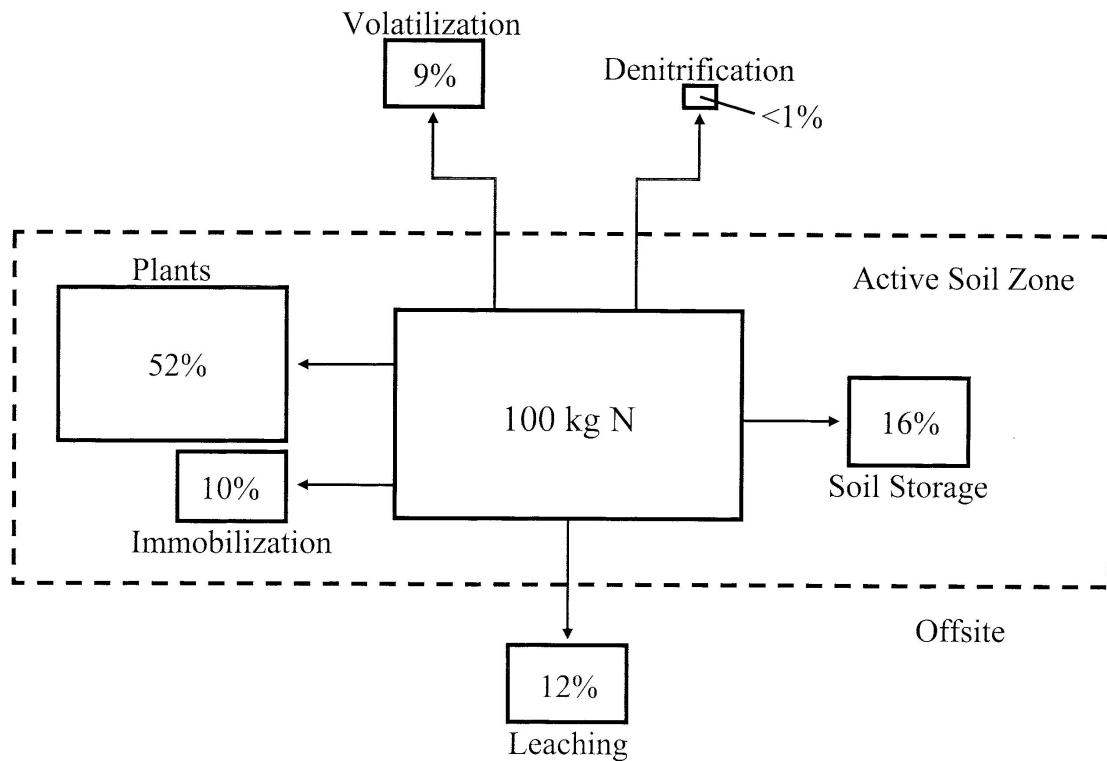


Figure 6.8.1. Generalized mass balance for 100 kg of liquid swine effluent-N applied to a receiving field associated with a North Carolina CAFO.

My study best addresses the post-application fate of N in liquid lagoonal swine waste and does not include losses incurred during surface application by traveling or center pivot sprinkling systems, which can be as high as 40% (Safley *et al.* 1992). The post-application

fate of N in liquid swine waste does not differ fundamentally from other organic and inorganic fertilizers, as the relative importance of all loss and storage terms fall within the ranges of values given for other fertilizers (comprehensively summarized in Meisinger and Randall 1991). However, liquid swine waste does differ from other N fertilizers in the rate of processing, due to its immediate contact with and availability for plants and microbes.

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