

DETERMINATION OF AMMONIA FLUX FROM
SWINE EFFLUENT APPLIED TO
CALCAREOUS SOILS

By

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FOREWORD

This document is presented as two chapters. Each chapter is formatted as a stand-alone article following the formatting specifications of the journal, *Atmospheric Environment*. This approach facilitates a more streamlined method of preparing manuscripts for publication without the necessity of rewriting the thesis.

CHAPTER I

EFFICIENCY DETERMINATIONS OF AN AMMONIA PASSIVE FLUX SAMPLING SYSTEM

ABSTRACT

Various methods have been used and studied to find an effective system for measuring the loss of ammonia (NH_3) from fertilizers and manure applications to croplands. Techniques vary, ranging from wind tunnels, ^{15}N -balance, closed systems, semi-closed, and micrometeorological methods. Of these approaches, most researchers agree that when evaluating NH_3 losses under field conditions, the micrometeorological techniques are preferred due to their ability to not disturb the environmental or soil conditions, which are extremely important in volatilization processes. Also, they minimize sampling variations from point to point due to their ability to measure average flux over a large area. Danish researchers have devised a mass balance micrometeorological passive flux sampling system, which has simplified many previous methods. Climatic variations do not need to be measured, electrical power and a large labor force are not needed; there are no specific restrictions to the surroundings of the experimental area, and they can be utilized cheaply without the requirement of expensive or complicated technologies.

The purpose of this present work was to determine the efficiency of these passive flux samplers to capture volatilized ammonia (i) in the laboratory under controlled

conditions using a standard ammonium solution, (ii) in a laboratory experiment using acidic and calcareous soils at five concentrations of NH_4^+ -N in effluent, and (iii) with varying wind speeds in a wind tunnel.

Ammonia capturing efficiency determined in the laboratory using a standard ammonium solution indicated that these samplers are very effective at recovering volatilized NH_3 . Recoveries of N ranged from 90 to 98%. Results from the soil-effluent laboratory experiments showed that volatilization was extremely rapid for the calcareous Richfield clay loam (pH 8.1) as compared to the acid Dennis silt loam (pH 5.8) over all effluent treatments. Mass balance results for soil-effluent experiments ranged from 71 to 148% recovery. Percent recovery decreased as effluent NH_4^+ -N concentration increased. However, when this sampling system was evaluated in a wind tunnel, the efficiency decreased to < 40%. There are some artifacts of working in this particular wind tunnel, which are believed to be the cause for such problems. Poor recovery is believed to have resulted from variable airflow and turbulence. Despite poor recovery of NH_3 in the wind tunnel, results from the laboratory experiments indicated that these passive samplers have a high efficiency in closed conditions and good repeatability in a soil-effluent system. Therefore, these samplers show promise to work well under field conditions.

INTRODUCTION

An accurate measurement of NH_3 volatilization from nitrogen sources such as nitrogen fertilizers or animal wastes is necessary in attaining optimum N-use efficiency and N recommendations. Published estimates of NH_3 volatilization losses from both of these sources vary tremendously. Variability in studies of NH_3 losses is due in large part to the countless types and numerous methods of experimentation. Sewage sludge applied to a bare field reportedly lost 60% of the applied ammoniacal N due to volatilization as measured by an open aerodynamic diffusion method (Beauchamp et al., 1978). Only 24 – 33% of the NH_4^+ -N applied in liquid dairy cattle manure applied to bare fields was lost when measured by this same method (Beauchamp et al., 1982). Losses of NH_3 from urea additions were found to be 28% by the enclosure method (EM) and integrated horizontal flux method (IHFM) while the nitrogen recovery method (NRM) estimated a 45% loss (Black et al., 1985). Ammonia volatilization measurements in the field have always presented a problem due to the inability to take readings without disturbing one or more of the many factors that influence volatilization. In contained trapping systems, wind speeds, rainfall, and soil/air temperature fluctuations are very difficult to simulate when trying to create a representative field model. Denmead (1983) classified methods for gaseous nitrogen loss measurements from the field into these general categories: 1) diffusion theory calculations of gas transport in the soil profile; 2) enclosures which measure flux densities of the gas in question at the surface; and 3) micrometeorological techniques which measure gaseous vertical flux density in the air above the surface. Therefore, studies to provide a systematic study of volatilization factors using forced-draft methods, such as those done by Ernst and Massey (1960), can not be used under

field conditions with any assurance of accuracy. Field research using closed or semi-closed methods such as a forced-draft or static enclosure (Harding et al., 1963; Volk, 1959) cannot simulate true environmental conditions due to extreme variation in the lower atmosphere and soil surfaces fluctuations. These experiments cannot be truly representative of field conditions because of the lack of a realistic near-surface turbulent flow (Gordon et al., 1988). Enclosures have typically been the method of choice by most researchers due to their relative simplicity, suitability for small experimental plots and the low sensitivity requirements for gas concentrations. An enclosure type method is more convenient for evaluating specific effects of variables and/or treatments. The use of wind tunnels as described by Lockyer (1984) has gained interest, yet not even with this semi-closed technique can all of the environmental field conditions be accurately represented. Studies using ^{15}N have been used with some success, yet this type of measurement requires specialized equipment and skills not readily available to all laboratories (Fox et al., 1996).

Micrometeorological methods are often better suited due to their ability to leave the surrounding area relatively undisturbed, enabling field conditions to be better represented. These methods are often necessary for determining losses from complex field conditions, because only with these methods can confidence be placed that measured losses have not been affected by the method (Denmead, 1983). However, these techniques are not without their own unique problems such as the need for a large labor force, large treatment areas, high expense, and technical equipment that make this system difficult to replicate and apply to variable treatments (Ferguson et al., 1988). Also, Leuning et al. (1985) noted that this type of measurement system is more effective when

sampling times are short, such as one to two hours, and in which NH_3 levels and wind speeds are not prone to large fluctuations.

Ferguson et al. (1988) compared two field methods (a microplot-forced-draft method and a micrometeorological method) for estimating NH_3 losses from urea solution applied to bare soil and wheat stubble residue. The microplot method places a cover over the treatment area while air is periodically drawn through the cover into an acid trap by a pump (Kissel et al., 1977). This was compared to the micrometeorological mass balance method (Beauchamp et al., 1978; Wilson et al., 1982). These studies revealed that the microplot method was more likely to affect the environmental parameters, which influence NH_3 volatilization measurements more than the open micrometeorological method. Ferguson et al. (1988) stated that the micrometeorological method more accurately reflects the actual levels of ammonia loss. However, due to the larger areas required for these types of experiments, it is more difficult to replicate and apply variable treatments.

Based upon the mass balance principle, Beauchamp et al. (1978) suggested that NH_3 volatilization losses from sludges applied in a field could be found by measuring the vertical profiles of the time averaged horizontal wind speed and NH_3 concentration. Ammonia molecules leaving a horizontal surface must be carried through a vertical plane by horizontal airflow. Wilson et al. (1982, 1983) simplified this model by eliminating the need for a large fetch, large labor force, and expensive instrumentation with the development of the ZINST mass balance approach. ZINST is the single height at which the rate of gaseous mass transfer to the atmosphere can be calculated as a function of roughness length and the radius source. Using this simple mass balance principle, a

passive flux sampler has been created by Danish researchers that employs a micrometeorological mass balance method for measuring NH_3 volatilization which should eliminate many of the problems associated with these open field measurements (Schjøerring et al., 1992). With this sampling system, climatic factors do not need to be measured and there are no restrictions to the experimental area or sampling periods, as with most other micrometeorological methods.

The purpose of the present work was to determine the efficiency of these passive flux samplers to capture volatilized ammonia (i) in the laboratory under controlled conditions using a standard ammonium solution, (ii) in a laboratory experiment using acid and calcareous soils at five concentrations of NH_4^+ -N in effluent, and (iii) with varying wind speeds in a wind tunnel.

MATERIALS AND METHODS

Samplers

The sampler, as described by Schjøerring et al. (1992) and Sommer et al. (1996), consists of 3 glass tubes joined together by silicon tubing with lengths of 100 mm, 100 mm, and 23 mm. All have an internal diameter of 7 mm. The inner surface of the two longer tubes is coated approximately 70 mm with oxalic acid. The shorter, 23 mm tube, has a 0.05 mm thick stainless steel disc with a centered hole of 1.0 mm in diameter glued to the outside end. The purpose of the stainless steel disc is to decrease the air speed inside the tubes in order to achieve a low friction resistance and a high NH_3 collection efficiency (Figure 1). Tubes and stainless steel discs were manufactured by Mikrolab Aarhus A/S, Axel Kiers Vej 34, DK-8270 Hoejbjerg, Denmark (Schjøerring et al., 1992).

Each sampler includes two parallel sampler units facing opposite directions so that while one is collecting from the NH_3 source, the other is collecting any background NH_3 that may be present in the experimental area (Figure 1). The oxalic acid coating is obtained by drawing acetone containing 3% oxalic acid into each tube. The acetone solution is allowed to drain and the tubes are dried with NH_3 free air and capped immediately to prevent any atmospheric NH_3 from contaminating the tubes. Oxalic acid was selected as the NH_3 trapping medium due to its ability to completely absorb NH_3 gas at all relative humidities of the air stream (Shendrikar and Lodge, 1975; Ferm, 1979). Acetone was used as the solvent for the oxalic acid due to its highly volatile nature, which provides a uniform coating of oxalic acid once it evaporates (Leuning et al., 1985). After being exposed to the NH_3 source, the tubes are disconnected and eluted with 3 mL

of deionized water. The dissolved coating, which now consists of ammonium oxalate and unreacted oxalic acid, was analyzed by using a Lachat 4 by Zellweger Analytics by the ammonia phenolate method (Bloxham, 1993).

Efficiency Determinations

These use of passive flux samplers is relatively new, with little published use of them. Therefore, laboratory experiments were conducted to determine the efficiency at which these samplers were able to collect volatilized NH_3 .

A system mass balance was performed to evaluate these samplers in the laboratory using standard NH_4^+ -N solutions. Fifty milliliters of 0.001 M NH_4Cl was placed in a 125 mL Erlenmeyer flask that was connected with a glass manifold consisting of several ports. Two of these ports contained a coated oxalic sampler, while the other ports were capped off. Air was passed through the flask so volatilized NH_3 would be carried to the samplers. The other side of the Erlenmeyer flask was attached to a helium air source, which would carry any volatilized NH_3 from the NH_4Cl source to and through the samplers (Figure 2). Helium (grade HP) was chosen as a carrier because it contains no detectable amounts of NH_3 . A needle and syringe were placed into the stopper of the flask for the addition of 2 mL of a base solution (0.1 M NaOH) to raise the pH to 12, thus promoting NH_3 volatilization from the solution. At this pH, any NH_4^+ -N in the solution would tend to volatilize and be carried into the air stream and through the coated samplers described above. After the sampling time was reached, 10 mL of 0.2 M H_2SO_4 was added to the flask via needle and syringe to reduce the pH of the solution to 2 in order to stop further volatilization. Titrations were performed to determine the amounts

of NaOH and H₂SO₄ needed to alter the solution to the desired pH. Samplers were removed and analyzed for NH₃ as described above. A Hewlett-Packard bubble flowmeter was used to ensure that airflow was maintained at 500 mL min⁻¹ through each efficiency determination experiment. The final NH₄⁺-N concentration for the NH₄Cl solution was determined for a mass balance of the system. The mass balance was corrected for the volumes of H₂SO₄ and NaOH added to obtain the desired pH to promote and halt NH₃ volatilization respectively.

Soil-Effluent Flask Volatilization Studies

To determine sampling efficiency under a soil-effluent system, a mass balance was performed similarly to the above efficiency studies. Five flasks containing soil were connected to a manifold, with samplers on the outlet flow, and compressed breathing air on the inlet (Figure 3). Breathing air was chosen instead of other gas carriers for fear that microbial populations would be altered due to a lack of O₂ in such high moisture environments after effluent additions. To determine the efficiency of a soil-effluent system similar to what might be found under true field conditions, the environment should not be modified in such a way to hamper microbial processes such as nitrification or immobilization. A Richfield clay loam (fine, smectitic, mesic Aridic Argiustolls) and a Dennis silt loam (fine, mixed, thermic Aquic Argiudolls) were used for comparison in these experiments. Soil characteristics are summarized in Table 1. All soil samples for this experiment and succeeding experiments were analyzed by the following procedures. Soils were air dried and extracted for inorganic N using a 2 M KCl and soil solution ratio

of 2:5 Bremner, 1965) and analyzed with an automated flow injection analysis system¹. Lachat Method 12-107-04-1-B was used for soil NO₃⁻-N while Method 12-107-06-1-B was used to analyze NH₄⁺-N (Lachat, 1993; Bloxham, 1993). Soil samples were air dried at ambient temperature and ground to pass a 100-mesh sieve (<0.15 mm) for the determination of total N and organic C analyses (Tabatabai and Bremner, 1970). Total N and organic carbon were analyzed using a Carlo-Erba (Milan, Italy) NA 1500 dry combustion analyzer (Schepers et al., 1989). The calcareous Richfield soil was first acidified with 10% HCl to remove free CaCO₃ in surface horizons before the organic carbon determinations. Soil pH was determined using a glass electrode and a soil:water ratio of 1:2. Soil phosphorus, extracted using Mehlich III procedures (Mehlich, 1984), was analyzed using a Milton Roy 401 spectrophotometer. Effluent characteristics for all experiments were determined as follows: inorganic N was analyzed with the automated flow injection analysis system as described above. Electrical conductivity (EC) was measured using a flow through cell Radiometer Copenhagen CDM 83 Conductivity Meter.

The Richfield clay loam, from the Oklahoma State University Panhandle Research Station, was chosen for this study because future field volatilization experiments using these samplers were to be used in this soil series. Soil from the Holdenville region of the state was used due to its lower pH value (5.8), which would make a good comparison to the much higher pH of the Richfield clay loam (8.1). Also, a significant amount of swine facilities are situated in the Holdenville area, warranting future attention to this area as well. Air dried soil was added to five 2000 mL Erlenmeyer

¹ Mention of model and company names is for reader information only and does not imply endorsement or preferential treatment by any party involved.

flasks to a depth of 2.54 cm for each experimental test. This required 500 g of the Richfield clay loam and 550 g of the Dennis silt loam. In order to determine sampler efficiency in capturing NH_3 volatilization over a range of NH_4^+ -N effluent concentrations, 5 different concentrations were evaluated (0, 100, 300, 500, 1000 mg NH_4^+ -N L^{-1}). Effluent was obtained from the Oklahoma State University swine facility, which typically is low in NH_4^+ -N (100 mg L^{-1}). Therefore, to attain the higher NH_4^+ -N concentrations, a 10,000 mg NH_4^+ -N L^{-1} stock solution was made in deionized water and added to the effluent to reach the desired concentrations. A high stock concentration was used to ensure that the characteristics of the effluent would not be altered to any significant degree when spiking the effluent. Before addition of the effluent, soils were brought to 16% field moisture by weight and allowed to equilibrate for approximately 30 minutes. Eighty-three mL of effluent was added to each flask, which is equivalent to a 2.54 cm rate, while 83 mL of deionized water was added to the control flask.

Preliminary studies were conducted to determine the rate of volatilization from this system to ensure that samplers could be removed prior to saturation of oxalic-acid adsorbent, thereby reducing NH_3 bypass. After determining the rate at which NH_3 was leaving the soil surface, samplers were taken off before they reached a point of possible NH_3 bypass due to saturated sites. According to Sommer et al. (1996), samplers were able to quantitatively absorb NH_3 from the air as long as their NH_4^+ -N concentration was below 50 μg , which was one-fifth of their total sorption capacity. The maximum capacity for NH_3 absorption of a sampler, after complete reaction of the oxalic acid, should be 252 μg NH_3 . Removal of the samplers at the proper time is imperative due to the absorption range. Tubes left on for too long could reach the maximum capacity and

lose NH_3 that bypasses filled sites, while tubes left on for too short of a time could be below instrument detection limits for NH_3 . Samplers for the Richfield clay loam necessitated removal every 10 minutes for the first hour due to extremely rapid volatilization rates immediately following the addition of the effluent. By the 20th hour, volatilization was still quite high, with samplers being removed every 60 minutes. The acid Dennis silt loam however, required removal of the samplers every 20 minutes for the first hour and after only 5 hours, the rate of volatilization was so low that samplers only needed to be removed every 60 minutes. After the 6th hour, samplers were only taken off three more times. Samplers were taken off less frequently for the 0 and 100 treatments for both soils due to a much lower volatilization rate.

After the 24-hour sampling period, soils were immediately extracted with 2 M KCl for NO_3^- -N and NH_4^+ -N determinations. Percent moisture was determined by oven drying. Results from two Richfield clay loam experiments were combined to determine a best-fit curve for all treatments using SigmaPlot (SPSS, 1997). A 2nd parameter power equation was empirically determined to model this data using an equation of $y = ax^b$. Three experiments for the Dennis silt loam were used to determine the same best-fit line as described for the Richfield soil. Only 2 of the Richfield flask runs were included in these comparisons, because the airflow rate for the first attempt was measured incorrectly after the stainless steel disc.

Wind Tunnel Studies

To evaluate sampling efficiency in environments similar to field conditions, effluent was applied to a soil in an environmental wind tunnel. This tunnel, developed to

study wind erosion by the Biosystems and Agricultural Engineering Department in the 1930's, is approximately 15 m long and 1.22m in height and width (Figure 4). Outside air is pulled through the tunnel by a 2.4 m diameter fan, while the exhaust was vented through open doors.

Three sets of plastic louvers were positioned upstream throughout the tunnel in an attempt to reduce turbulence in the airflow caused by the pulling motion of the fan. Roughness elements (64 mm x 64 mm) were installed over a 4 m span of the tunnel floor in front of the source to break up laminar airflow on the bottom.

Samplers were set up at the tunnel intake to measure background levels of NH_3 in the airflow. These values were subtracted from the samplers placed downwind of the source, which were capturing volatilized NH_3 . In initial tunnel runs, samplers captured volatilized NH_3 at a constant wind speed of 2.2 m s^{-1} from an NH_4Cl source. One thousand milliliters of a 0.1 M NH_4Cl source were raised to an alkaline pH with the addition of 0.2 M NaOH. After 12 hours, samplers were capped and the NH_4Cl solution was acidified to stop further volatilization. Final ammonium concentrations were analyzed from the source after the experiment to determine a mass balance for the system. Relative humidity and ambient air temperature inside the tunnel were monitored throughout each of the 12-hour experiments with a VelociCalc 8345 anemometer and a hand held sling psychrometer. After sample collection, $\text{NH}_4^+\text{-N}$ in the sampling tubes was determined as previously described.

After numerous runs with the NH_4Cl source, a soil-effluent system was initiated with the same procedure as described above. Thirteen kg of a Norge loam (fine mixed thermic Udertic Paleustoll) soil was added to a large pan across the entire width of the

tunnel at a 2.54 cm depth. The soil was brought to 16% field moisture and allowed to equilibrate for over 1 hour before 2.4 L of effluent was evenly distributed over the soil surface with a watering can. Effluent amounts were calculated to be applied at a ha - 2.54 cm rate. Due to the low NH_4^+ -N concentration of the effluent (100 mg L^{-1}), 50 mL of 1.067 M NH_4Cl was added to the effluent. This was performed in order to spike the effluent to levels that could be more accurately measured and which would be similar to NH_4^+ -N concentrations that is commonly found in the Panhandle region of the state.

RESULTS AND DISCUSSION

Efficiency Determinations

Under closed laboratory conditions, the passive samplers were relatively effective at capturing volatilized NH_3 as indicated by the efficiency of the tubes ranging from 90 - 98 % (Table 2). These preliminary tests indicated that the samplers should be effective for the collection of volatilized NH_3 .

Data from the original publications promoting these passive samplers found a strong linear relationship between the horizontal net flux densities of NH_3 measured by the passive flux samplers and by a reference method ($r^2 = 0.96$) under field conditions Schjøerring et al. (1992). An NH_3 source consisting of 140 beakers was evenly distributed throughout the experimental area in which a reference method was compared to the new passive flux samplers. Ammonia losses from an NH_4HCO_3 solution as estimated by the use of passive flux samplers was also compared to a reference technique by Sommer et al. (1996), in which estimations of losses were quite similar between the two methods.

Soil-Effluent Flask Volatilization Studies

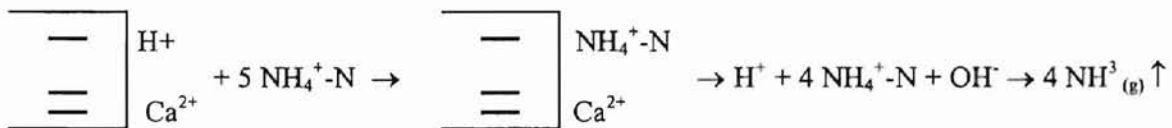
Early laboratory experiments found that small increases in soil pH resulted in higher rates of NH_3 volatilization (Jewitt, 1942) and research by Martin and Chapman (1951) demonstrated that NH_3 losses were dependent on pH and temperature. Many other studies have shown that generally, as pH increases the greater the loss of NH_3 due to volatilization (Chao and Kroontje, 1964; Du Plessis and Kroontje, 1964; Warren, 1962; Meyer et al., 1961; Freney et al., 1983). This was also demonstrated in these

laboratory flask runs using the calcareous soil versus the acid soil. Volatilization was extremely rapid for the Richfield clay loam as compared to the Dennis silt loam over all effluent treatments (Figures 5 and 6). Also, the difference in volatilization rates between the two soils is influenced by the amount of Ca^{2+} present in the calcareous soil, as can be seen by the soil cation exchange reaction shown below. Higher amounts of $\text{NH}_4^+\text{-N}$ are exchanged on the CEC with increasing amounts of $\text{NH}_4^+\text{-N}$ added to the Dennis silt loam. In comparison, as $\text{NH}_4^+\text{-N}$ concentrations increased in the Richfield clay loam, the presence of Ca^{2+} did not allow appreciable amounts onto the CEC due to the binding energy. With higher amounts of $\text{NH}_4^+\text{-N}$ present in soil solution, increased amounts of NH_3 will be volatilized due to the presence of OH^- in the high pH soil.

Dennis silt loam (pH 5.8)



Richfield clay loam (pH 8.1)



Initial runs using the calcareous soil found that the sorption capacity of the sampling tubes was saturated within 15 minutes after initial application, therefore the samplers were changed every 10 minutes for the first hour. The volatilization rate decreased with time and was negligible at 24 hours. When working with this system it is imperative that sampling begin as soon after application as possible for the most accurate measurements. The two Richfield clay loam flask experiments were very repeatable for

treatments 100, 300, and 1000 mg $\text{NH}_4^+\text{-N L}^{-1}$ ($r^2 = 0.99, 0.99, 0.99$ respectively), while treatments 0 and 500 mg $\text{NH}_4^+\text{-N L}^{-1}$ were much lower ($r^2 = 0.38$ and 0.81) (Figure 5). Regression statistics are shown in Table 3 for all treatments and both soils. The control treatment, in which only deionized water was added to the soil, displayed variations between the two replications resulting in a low r^2 (0.38). Variability should be higher at the lowest NH_3 volatilization rates (i.e. control). The 500 mg $\text{NH}_4^+ \text{L}^{-1}$ effluent treatment for the first experiment was considerably less than for the following experiment, resulting in a much lower coefficient of determination (0.81). The three Dennis silt loam experiments were quite variable (Figure 6). The initial flask run was lower than the following two experiments for treatments 1000 and 100, while treatment 500 for the second run was higher than all other treatments for all experiments. The reason for such variations in these three flask runs is unknown. Laboratory conditions, flowrate, source, and procedures were all held constant. Samplers possibly were just not able to pick up on these lower amounts of NH_3 .

Mass balances for these laboratory soil-effluent experiments demonstrated that generally, with increasing effluent concentrations, percent recovery of the system decreases (Table 4). Exchange competition becomes a factor at higher $\text{NH}_4^+\text{-N}$ solution concentrations. As exchange sites fill up with increasing concentrations of $\text{NH}_4^+\text{-N}$ in solution, more $\text{NH}_4^+\text{-N}$ will be subjected to volatilization. With increasing amounts of volatilized NH_3 , the higher the possibility of error, thus decreasing the percent recoveries of these systems at the higher concentrations. Recoveries greater than 100% for the control flasks could be attributed to increased microbial activity due to added moisture. These soils were air dried and stored for several months before use. Nitrogen could have

been mineralized, thus increasing the soil N balance at the conclusion of the experiments. Rice and Havlin (1994) discussed the factors that affect accessibility of organic N to microorganisms. Disruption of aggregates when working in a laboratory setting tends to increase net N mineralization (Craswell and Waring, 1972; Cabrera and Kissel, 1988). Also, when dry soils are rewet, microbial activity is generally stimulated which in turn increases mineralization (West et al., 1992). During the rewetting of soil, the flush of N mineralization could be due to the increase of the rapidly mineralizable pool (Cabrera, 1993). Therefore, it is possible that a flush of mineralization in the control flasks during this laboratory study could have contributed to the increased percent recoveries. Control flasks for the Dennis silt loam exhibited increases in NH_4^+ -N concentrations with a decrease in NO_3^- -N levels for two of the three experiments. However, for the Richfield clay loam the opposite was found. Soil concentrations of NH_4^+ -N in the control flasks for the last test had decreased by the end of the 24-hour testing period, whereas the NO_3^- -N levels increased. The first experimental run of the Richfield soil increased in both NO_3^- -N and NH_4^+ -N concentrations. However, these changes were small and are likely due to experimental error.

Wind Tunnel Studies

Application of swine effluent to the soil in the wind tunnel allowed for evaluation of the tubes in a simulated field condition. When this sampling system was moved to a wind tunnel setting, the efficiency of the samplers decreased significantly, ranging from 9.0 to 35.7% (Table 5). The concentrations collected on the tubes were much lower than those from the initial laboratory experiments. The NH_4^+ -N concentrations for many of

the samplers were near background levels. Known volatilization took place because ✓ concentrations of NH_4Cl solutions decreased with time. This is cause for concern that in a field experiment, NH_3 could be below detection limits of the analytical equipment when using these samplers. There were some anomalies associated with working in this wind tunnel, which are believed to be the cause for such problems. Due to the pulling motion of air by the fan, airflow inside the tunnel tended to form a vortex in the center, concentrating NH_3 in certain areas. Results from one of the initial NH_4Cl tunnel runs demonstrated that these samplers were able to capture only 2,285 mg of the total 25,433 mg that was volatilized. This is a recovery of only 8.9% of the NH_4^+ -N in the solution. Samplers were taken off at hourly intervals, and when analyzed were not beyond the point of maximum adsorptive capacity of the tubes. For this particular run, the average temperature was 18 °C with an average relative humidity of 72%. Perhaps the NH_3 was coming off so quickly in the first few hours of the NH_4Cl experiments that the samplers were not able to measure it adequately. However, if this were so, the first set of tubes should have been saturated. Samplers were placed < 1.6 cm from the bottom of the tunnel for several experimental runs in an effort to minimize any NH_3 losses that could potentially be escaping detection by passing beneath the coated tubes. Results for these runs were slightly higher, with a recovery of 35.7% (Table 5), but were still not at an acceptable level.

After plastic louvers were installed, the swirling effect in the center of the tunnel decreased somewhat, but still manifested itself in many of the following experiments. Soil-effluent experiments in the tunnel proved to be no more successful than the previous NH_4Cl experiments as shown in Figures 7 and 8. Percent recoveries of the soil-effluent

system for these two runs were 2.79 and 7.68% respectively. Figure 7 demonstrates that the airflow carried a higher concentration on the right side of the tunnel with a lower concentration on the left. This profile does not manifest the swirling effect in the center as Figure 8 does even though these tunnel runs were performed on consecutive days. The flow pattern was rarely repeatable, with laminar flow along the bottoms and sides of the wind tunnel often creating higher concentrations in the center, bottom or sides. In the tunnel run shown in Figure 9, samplers were set up higher than previous runs to determine if a measurable amount of NH_3 was flowing above the setup of the previous samplers. If this were true, it could possibly account for the extremely low percent recoveries. But, as can be seen, only negligible amounts were captured above the 15 cm level of the tunnel. Also, it should be noted that this particular run was tested under conditions not conducive to NH_3 volatilization, i.e. low temperatures. It is possible that changes in ambient air temperatures created changes in airflow patterns in a few of these tunnel experiments. The variability among runs limited the ability to evaluate these samplers under wind tunnel conditions. All attempts made to straighten airflow, such as airflow straightening screens and roughness elements met with limited success. However, there was a measurable quantity of NH_3 collected in the soil experiments, showing some promise for proper determination of flux under field conditions.

CONCLUSIONS

Initial work with these passive samplers indicated that this system is very efficient at capturing volatilized NH_3 under closed laboratory conditions as demonstrated by relatively high recoveries ranging from 90 to 98%. The laboratory soil-effluent flask experiments demonstrated that the calcareous Richfield clay loam (pH 8.1) has the potential to volatilize significant amounts of NH_3 within minutes after application. Therefore, any measurements with such soils in the field or laboratory must be done immediately following application of swine effluent. In comparison, the Dennis silt loam (pH 5.8) had a much lower volatilization rate across all treatments which is mostly likely due to the lower pH value for the Dennis silt loam. However, it could also be related to the presence of high amounts of Ca^{2+} on the calcareous Richfield clay loam's CEC, which would decrease the ability of NH_4^+ -N to exchange on the complex, thus leaving a greater quantity in solution which is more prone to loss via volatilization. The mass balances for these particular laboratory experiments ranged from 71 to 148% recovery, with recoveries decreasing with increasing effluent NH_4^+ -N concentrations. Exchange competition becomes a factor at higher NH_4^+ -N solution concentrations, so with increasing amounts of volatilized NH_3 , the higher the possibility of error, thus decreasing recoveries. Recoveries over 100% were most usually for the control flasks, which could be due to N mineralization from added moisture.

Applications of swine effluent to soil in a wind tunnel proved to be inconclusive due to difficulties in dealing with uneven airflow. Soil-effluent and NH_4Cl solution recoveries were < 40%. Perhaps a tunnel in which the air was pushed instead of being pulled through would have created an environment more conducive to this type of

atmospheric research. Even though wind tunnel NH_3 volatilization studies were inconclusive, they were not an integral focus of this project and did not affect the overall objective of quantifying NH_3 volatilization in the field. Overall, the passive flux sampling system should prove to be effective at correctly monitoring NH_3 volatilization from swine effluent applications to calcareous fields.

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Table 1. Average initial characteristics for both soils included in the laboratory flask volatilization experiments

| | Richfield | Dennis |
|--|------------------|---------------|
| Texture | clay loam | silt loam |
| pH | 8.1 | 5.8 |
| NH ₄ ⁺ -N (mg kg ⁻¹) | 3.0 | 12.0 |
| NO ₃ ⁻ -N (mg kg ⁻¹) | 22.0 | 2.5 |
| Total N (g kg ⁻¹) | 1.3 | 0.89 |
| M III P † (mg kg ⁻¹) | 29.0 | 12.5 |
| CEC (cmol _c kg ⁻¹) | 16.12 | 14.96 |
| Organic Carbon (g kg ⁻¹) | 12.1 | 10.4 |

† Mehlich III extractable

Table 2. Recoveries of NH₃ from a 15 mg N L⁻¹ in an NH₄Cl solution in closed system laboratory studies using oxalic acid coated passive samplers

| Minutes | Total NH₄⁺-N Recovery[†] | |
|----------------|--|--------------|
| | Run A | Run B |
| | ----- % ----- | |
| 60 | 92 | 97 |
| 90 | 92 | 97 |
| 120 | 92 | 98 |
| 180 | 93 | 98 |
| 210 | 93 | 96 |
| 270 | 91 | 95 |
| 300 | 98 | 90 |

† % Recovery of system = ((NH₄⁺-N final + NH₄⁺-N captured) / NH₄⁺-N initial) * 100

Table 3. Total NH₃ captured on oxalic acid coated passive samplers from soil-effluent flask experiments for five treatments on a Richfield clay loam (pH 8.1) and a Dennis silt loam (pH 5.8). Regression coefficient shown for volatilized NH₃ versus time

| Soil | NH ₄ ⁺ -N Treatment | Avg. Total NH ₃ Captured | Coefficient of Determination r ² | p value |
|-----------|--|---|---|---------|
| | ---- mg L ⁻¹ ---- | μg | | |
| Richfield | 0 | 5.15 | 0.38 | <0.05 |
| Richfield | 100 | 356.07 | 0.99 | <0.001 |
| Richfield | 300 | 507.83 | 0.99 | <0.001 |
| Richfield | 500 | 694.85 | 0.81 | <0.001 |
| Richfield | 1000 | 1161.47 | 0.99 | <0.001 |
| Dennis | 0 | 1.02 | 0.22 | >0.1 |
| Dennis | 100 | 76.45 | 0.45 | <0.001 |
| Dennis | 300 | 125.00 | 0.80 | <0.001 |
| Dennis | 500 | 172.89 | 0.63 | <0.001 |
| Dennis | 1000 | 171.43 | 0.58 | <0.001 |

Table 4. Mass balance average results for soil-effluent laboratory experiments

| Soil | NH ₄ ⁺ -N Treatment | Average Mass Applied | Recovery |
|-----------|--|-------------------------|-------------|
| | ---- mg L ⁻¹ ---- | ---- μg ---- | ---- % ---- |
| Richfield | 0 | 0.0 | 147.9 |
| Richfield | 100 | 8158.5 | 103.0 |
| Richfield | 300 | 21121.8 | 87.5 |
| Richfield | 500 | 35384.2 | 90.2 |
| Richfield | 1000 | 68439.6 | 73.6 |
| Dennis | 0 | 0.0 | 121.7 |
| Dennis | 100 | 9239.6 | 96.4 |
| Dennis | 300 | 21461.2 | 90.4 |
| Dennis | 500 | 35047.2 | 99.6 |
| Dennis | 1000 | 66669.9 | 71.1 |

Table 5. Percent recoveries of wind tunnel experiments at 2.2 m s^{-1} for 12 hours using NH_4Cl sources

| Run | Source | Total NH_4^+ -N Recovery |
|--------|------------------------|-----------------------------------|
| | | ---- % ---- |
| Run 8 | NH_4Cl | 9.0 |
| Run 9 | NH_4Cl | 11.6 |
| Run 14 | NH_4Cl | 35.7 |

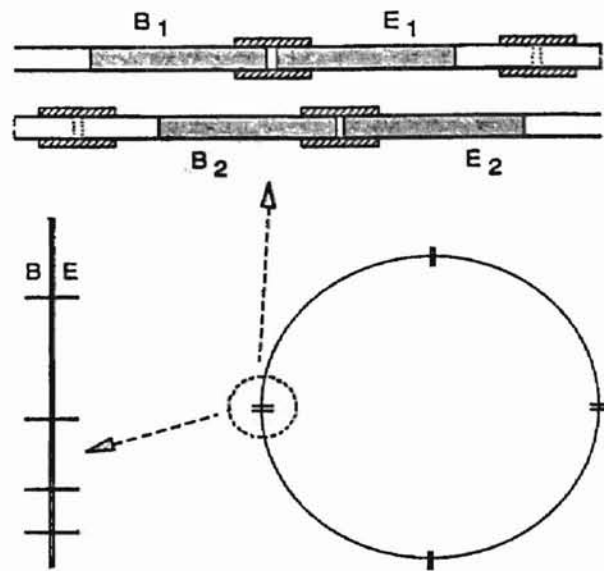


Figure 1: Passive flux sampler. Top: A passive flux sampler consisting of two parallel sampler units. Each unit is composed of 100 mm long glass tubes, coated with oxalic acid on approximately 70 mm of the inside (shaded), and a 23 mm tube with stainless steel disc at the end, all joined by silicone tubing. E₁, E₂ are exposed tubes and B₁, B₂ are background tubes. (Bottom left) Mast with four flux samplers seen from profile. (Bottom right) Four masts around a circular NH₃ source as seen from the top. From Sommer et al. (1996).

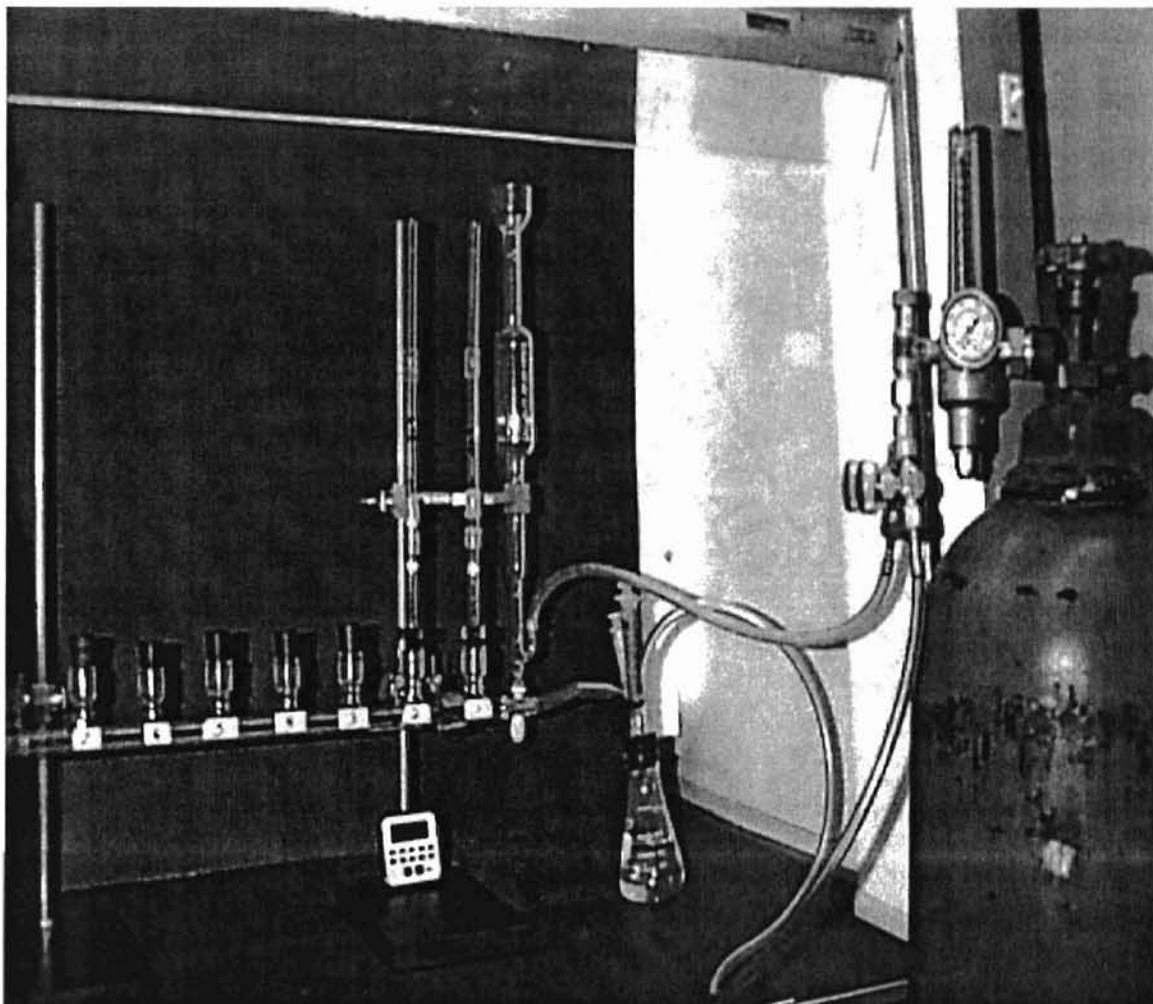


Figure 2. Setup for laboratory efficiency tests. Helium tank connected in series to flask containing known NH_4Cl solution, then to a manifold containing two samplers. Syringes containing 0.1 N H_2SO_4 and 0.1 N NaOH attached to flask stopper. Bubble flow meter seen at right.

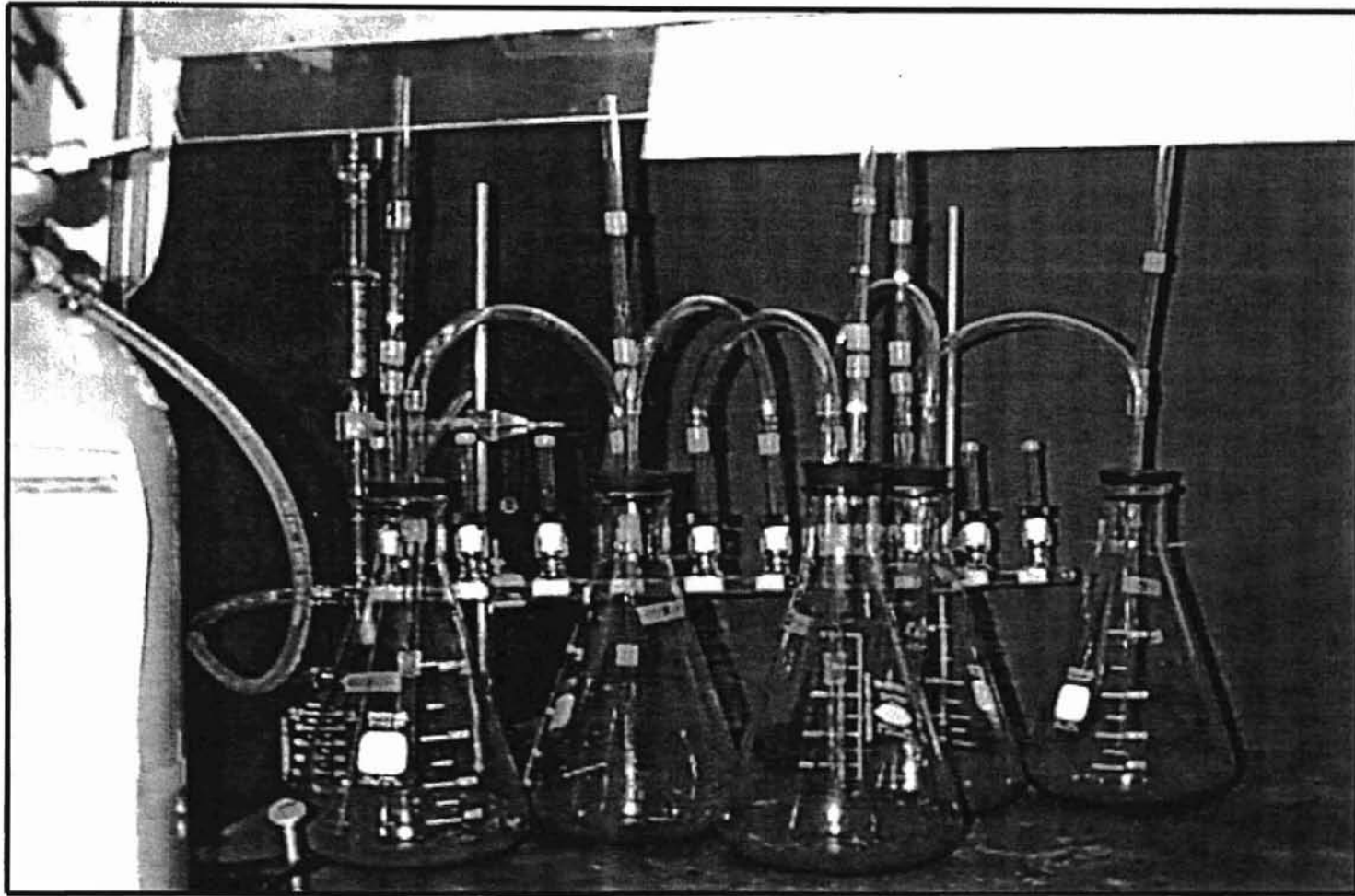


Figure 3. Setup for laboratory soil-effluent flask experiments. Oxygen tank connected in series to a manifold, joined to five flasks containing soil and effluent, with samplers attached to the outflow side of the air stream to capture any volatilized ammonia. Bubble flow meter seen at left.

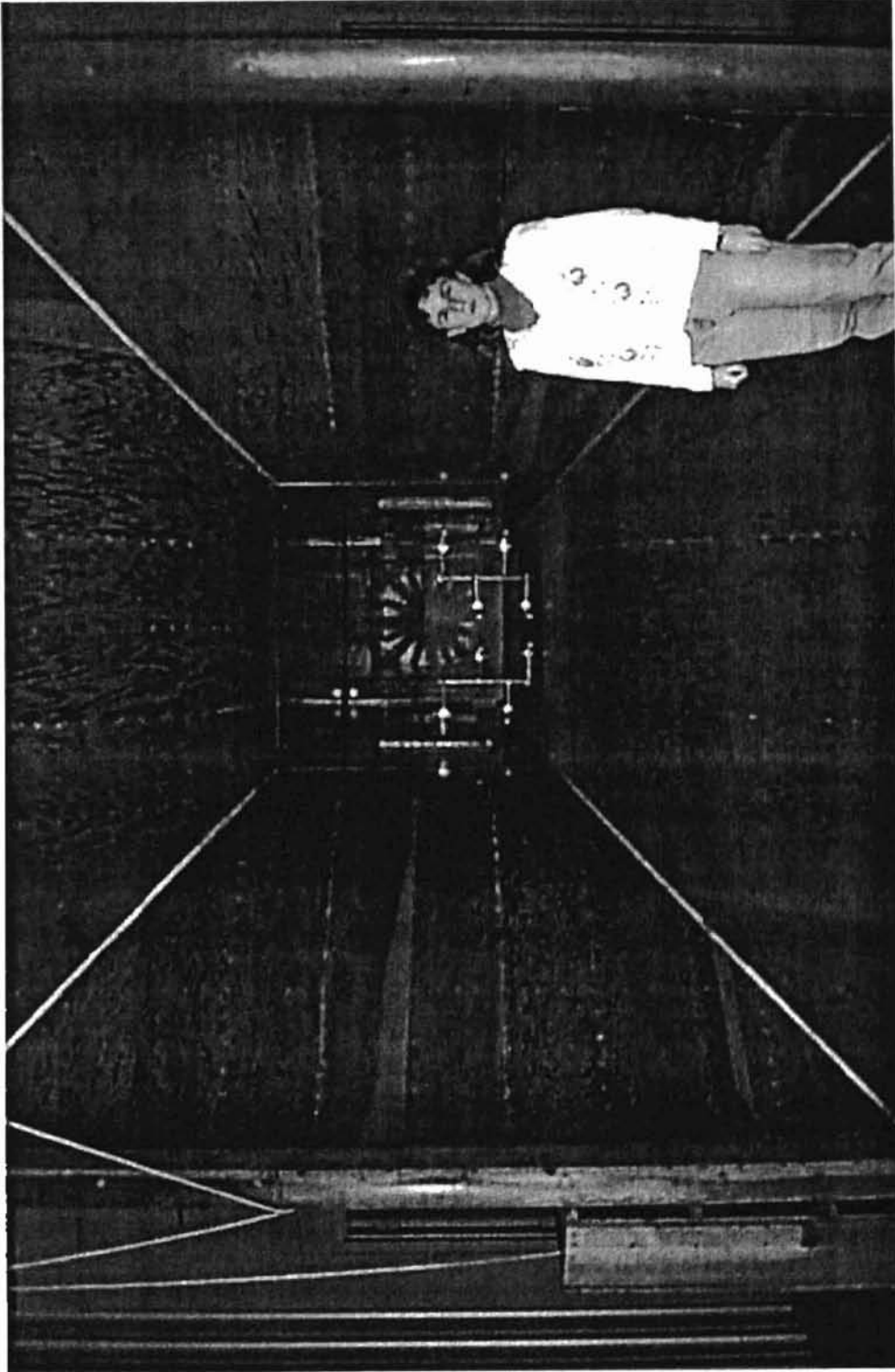


Figure 4. Wind tunnel as seen from air intake, with background samplers seen inside tunnel.

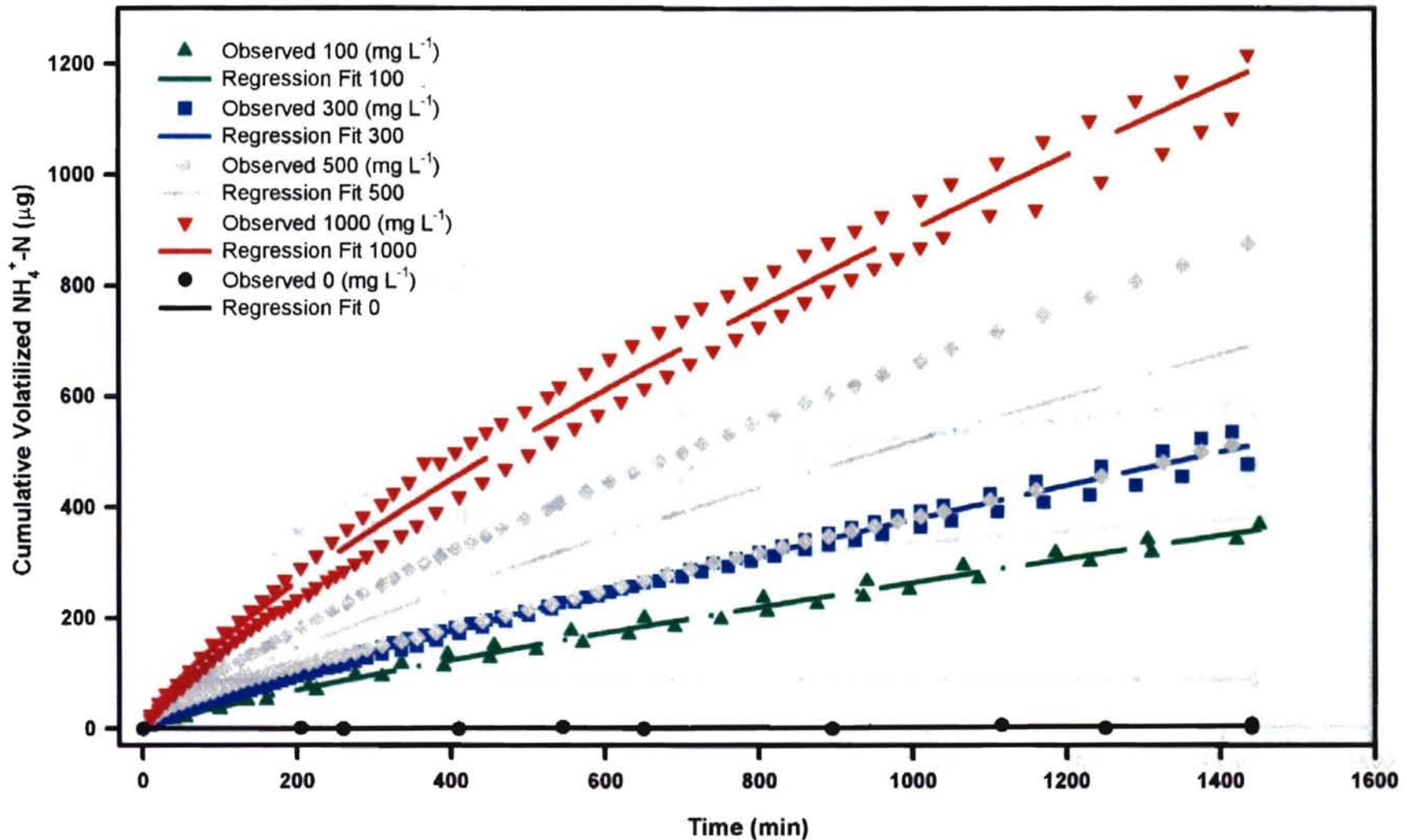


Figure 5. Regression lines of plotted cumulative NH_4^+-N versus time for the Richfield clay loam in laboratory flask experiments with five concentrations (mg N L^{-1}) of swine effluent.

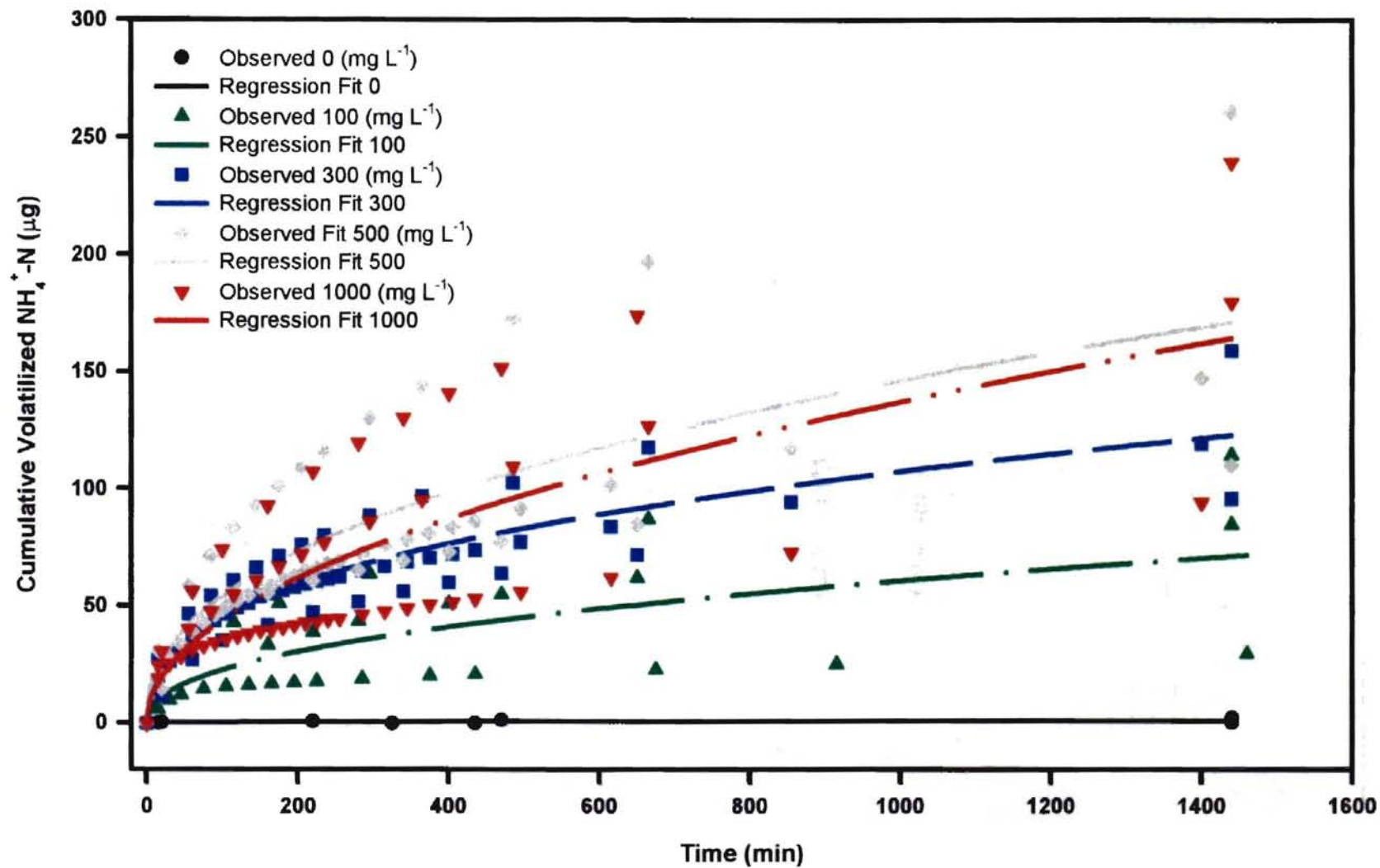


Figure 6. Regression lines of plotted cumulative $\text{NH}_4^+\text{-N}$ versus time for the Dennis silt loam in laboratory flask experiments with five concentrations ($\text{mg NH}_4^+\text{-N L}^{-1}$) of swine effluent.

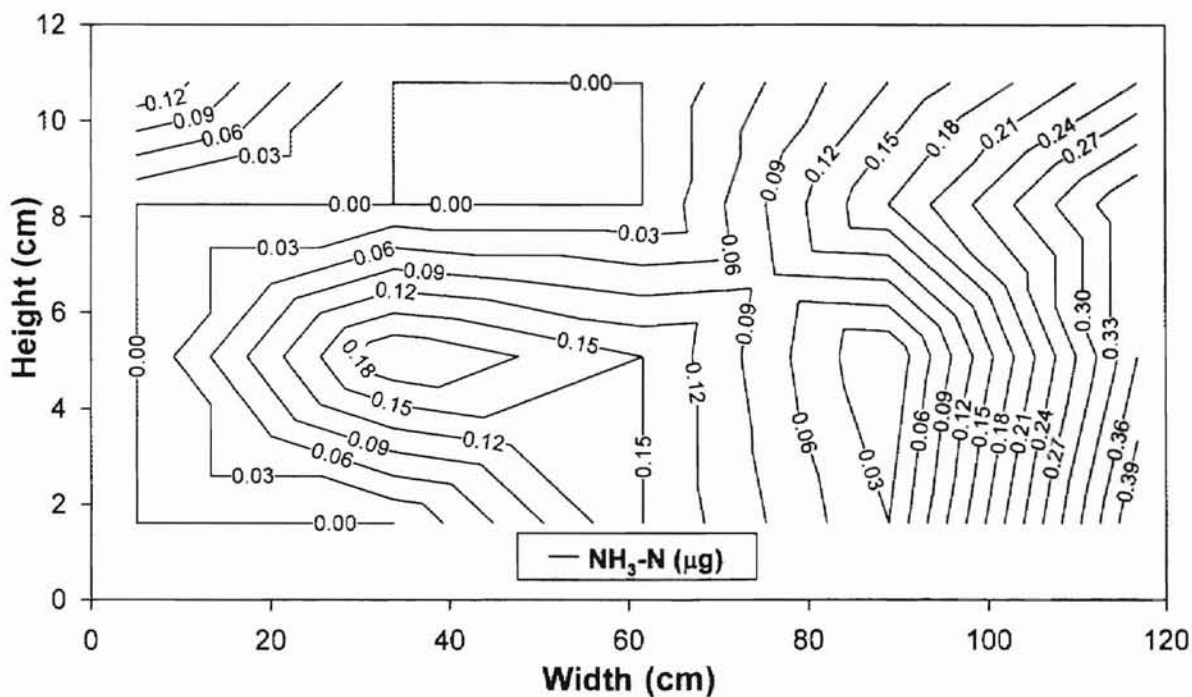


Figure 7. Ammonia profile 1 in wind tunnel. After the addition of swine effluent to soil at a wind speed of 2.23 m s^{-1} for 12 hours. Air temperature ranged from 19 to 26°C and relative humidity from 33 to 59%.

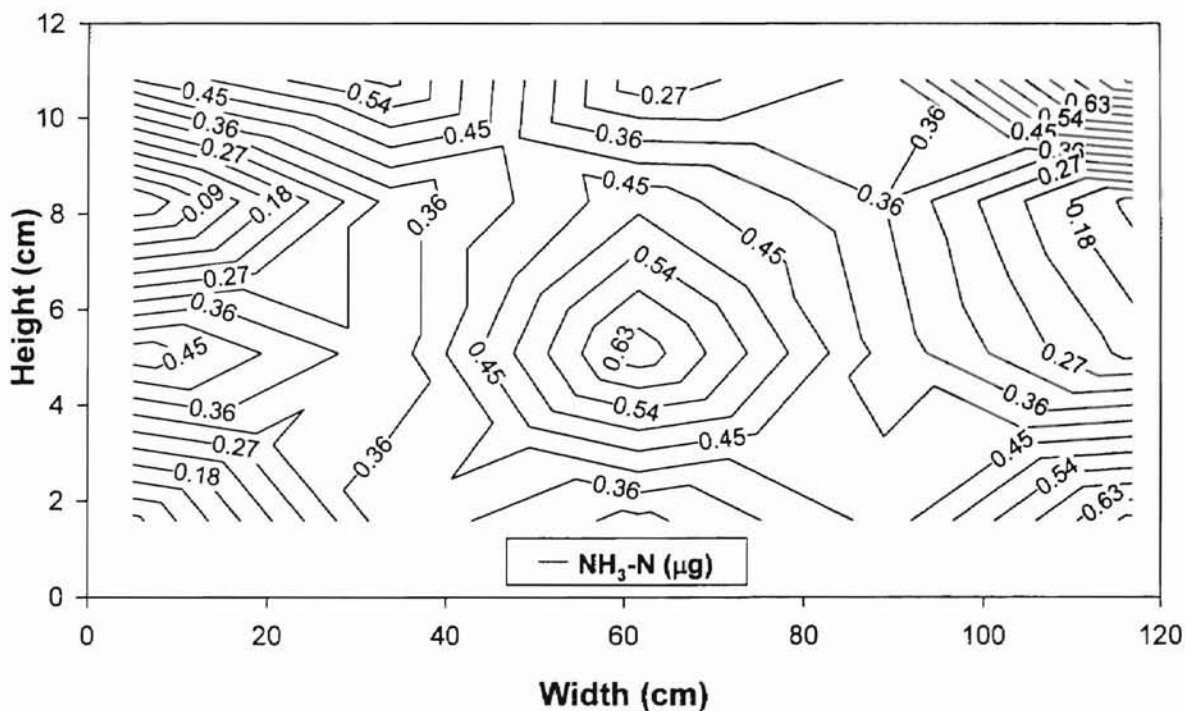


Figure 8. Ammonia profile 2 in wind tunnel. After the addition of swine effluent to soil at a wind speed of 2.23 m s^{-1} for 12 hours. Air temperature ranged from 8 to 24°C and relative humidity from 59 to 80%.

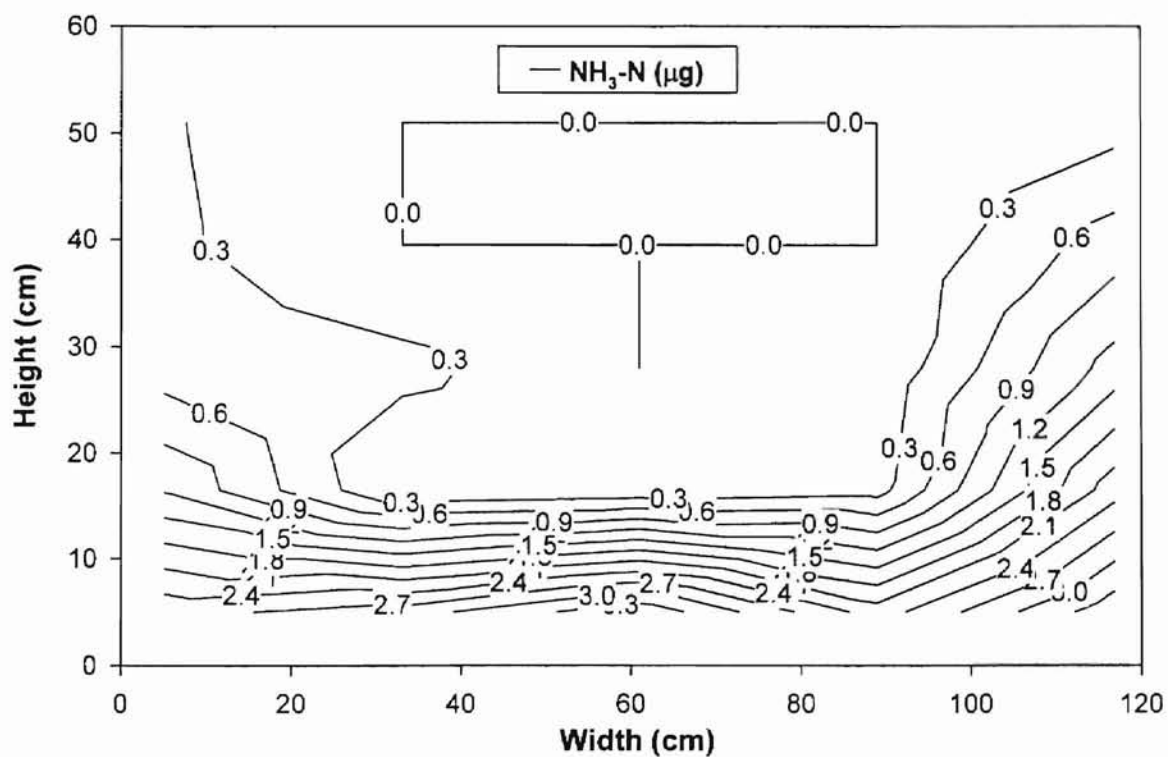


Figure 9. Ammonia profile 3 in wind tunnel. After addition of $1200 \text{ mg NH}_4^+\text{-N L}^{-1}$ from an NH_4Cl solution at a wind speed of 2.23 m s^{-1} for 11 hours. Air temperature ranged from 1 to 4°C and relative humidity from 35% to 62%.

CHAPTER II

DETERMINATION OF AMMONIA FLUX FROM SWINE EFFLUENT APPLIED TO CALCAREOUS SOILS

ABSTRACT

In the Panhandle of Oklahoma, swine effluent is pumped from lagoons through center pivot irrigation systems or big guns where many of the environmental conditions promote the potential loss of a significant amount of nitrogen when applied to cropped fields. This region of the United States is located in the high plains of the interior Great Plains, where the soil pH is 7 to 9, summer daytime temperatures are consistently 23 – 37°C, with brisk wind speeds of 4.4 - 17.8 m s⁻¹, and low relative humidities (3-30%). The Panhandle of Oklahoma is of heightened interest to the state due to the large increase in the number of confined swine feeding facilities, which have located there in the last decade. Between 1996 and 1998, the total number of swine in the Panhandle district has increased from approximately 370,000 to 905,000 head (245% increase), making it one of the leading areas of swine growth in the United States. Given the increasing number of swine in a region very conducive to loss of nitrogen via volatilization losses, the need for further study of ammonia volatilization has become extremely important. More information unique to this region will not only enable proper nutrient utilization and management for the producer, but will also ensure that the release of unnecessary atmospheric NH₃ is minimized. This will benefit the air quality of the area and help decrease a potential odor problem. Therefore, the objective of these field research tests

was to determine the NH_3 flux during land application of swine effluent in the Southern Great Plains using a simple passive sampling system. Effluent from a confined swine production facility, with an average $\text{NH}_4^+\text{-N}$ concentration of 963 mg L^{-1} , was applied to circular test plots at a $\text{ha}^{-1} - 2.54 \text{ cm}$ rate while samplers continuously monitored the horizontal flux of NH_3 at four different heights. Depending on the climatic conditions, field volatilization losses of NH_3 ranged from 37 - 57% of the total $\text{NH}_4^+\text{-N}$ that was applied. The 57% loss of $\text{NH}_4^+\text{-N}$ was encountered during hot, dry, and windy conditions, with the lowest loss (37%) during a more humid, rainy period. Mass balances performed for these field experiments ranged between 79 – 134% recovery. Overall, these samplers seem to be adequate at capturing and providing an estimate of NH_3 volatilization losses under variable field conditions.

INTRODUCTION

Land application of livestock wastes has been practiced and studied around the world for centuries as a principal source of plant nutrients (Sims, 1995). Since the early 1900's, research has shown that significant losses of nitrogen can occur from land applied animal fertilizers through ammoniacal nitrogen volatilization (Russell, 1915; Heck, 1931). In more recent times, the loss of nitrogen via this pathway in Europe has led to an increased concern over its impacts on the surrounding environment. These emissions contribute to local odor associated with swine production and to regional air quality problems. Schulze et al. (1989) stated that the deposition of NH_3 and NH_4^+ -N can contribute to eutrophication and acidification of some nitrogen limited ecosystems. Ammonia influences the pH of aerosols in cloud water and may be important in determining regional air quality characteristics. In the atmosphere, ammonia is the dominant alkaline gas and can have a significant effect on oxidation rates which affects deposition rates of acidic airborne constituents such as sulfuric, nitric and hydrochloric acid (ApSimon et al., 1987). Ammonia has a short half life in the atmosphere of 5-5 ½ days (Warneck, 1988); however, studies imply that NH_3 inputs over Europe have increased over 50% between 1950 and 1980 with doubling of emissions in some countries (ApSimon et al., 1987).

The emission of ammonia from animal wastes represents one of the most important sources of atmospheric ammonia in Europe (Buijsman et al., 1987). According to Groenestein and Van Faassen (1996), the Dutch legislation intends on reducing NH_3 emissions by the year 2005 by 70% in relation to their 1980 emissions. It is estimated that in Denmark alone, 46% of the environmental acidification is due in large part to

ammonia emissions from agricultural sources (Heij and Schneider, 1995). Ammonia originates from many natural sources, but the largest contributions are from domestic animal waste, fertilizers, and other agricultural operations (Buijsman et al., 1987; ApSimon et al., 1987). Ammonia emissions from confined animal production facilities may arise from several sources: inside the buildings, manure storage facilities (e.g. anaerobic lagoon or stock-piled manure), during the application process (irrigation or land application), and from the soil or plant surface following application. Ammonia deposition will vary greatly depending on local sources of volatile ammonia and local transport conditions.

Depending on the nutrient content and method of application, the net value of manure spread as a source of nutrients in the United States ranges from approximately -\$1.00 to + \$6.00 per metric ton on a dry matter basis (Hoff et al., 1981) or \$2.50 to \$3.50 per hog (Sutton, 1992). However, more than half of the N content in cattle and swine slurries is in the ammoniacal form which can be lost by volatilization of NH_3 , often within the first few hours after land application (Sommer and Ersbøll, 1994). Therefore, losses of NH_3 via this pathway may significantly lower the N value of the effluent for crop production, which must then be replaced by commercial fertilizers. Losses from swine lagoon effluent applied to grassland were reported as high as 62% of the applied NH_4^+ -N (Pain et al., 1989). Beauchamp et al. (1982) reported that between 24 and 33% of the ammoniacal N applied in liquid dairy manure was lost by volatilization within a 7 day period.

Ammonia volatilization is influenced by a number of soil and environmental factors with significant interactions occurring between the variables. Svensson (1994)

categorized these factors in three main groups: meteorological, soil/manure, and application technique. When examining meteorological factors, air temperature, relative humidity, rainfall events, evapotranspiration, NH_3 concentration in the atmosphere, wind speeds and air movements must be taken into account. Soil characteristics that need to be considered when discussing ammonia losses are the soil pH, texture, temperature, porosity, moisture, cation exchange capacity (CEC), hydrogen ion buffering capacity, calcium carbonate content, urease activity, immobilization and nitrification rates and the presence of vegetation. Manure properties that can affect ammonia volatilization are the pH, buffering capacity, total ammoniacal nitrogen content (TAN), and dry matter content. Fertilization management, such as timing, application method and rate, type, fertilizer placement and irrigation droplet size are factors that should be taken into consideration from an application viewpoint (McInnes et al., 1986; Nelson, 1982; Svensson, 1994; Freney et al., 1983).

Although interpretation of previous experiments concerning volatilization is difficult due to the variety of techniques utilized, most research has shown a positive correlation between wind speed and the amount of NH_3 volatilized. The influence of wind speed upon volatilization increased with the approximate square of the wind speed (Denmead et al., 1982). It was also determined that the exchange coefficient for NH_3 is affected by the aerodynamic roughness of the surface and the surface area. Ryden and McNeill (1984) were able to show that the wind speed decreased with decreasing height and neared zero at a point close to the base of a grazed sward. However, Bouwmeester et al. (1985) reported that volatilization from urea fertilized soils was affected negatively by increased wind speed. Sommer et al. (1991) could find no apparent correlation between

ammonia loss when wind speeds exceeded 2.5 m s^{-1} . Work by Thompson et al. (1990) using wind tunnels, indicated that the most noticeable amount of volatilization released was within the first twenty-four hours of application. This study also revealed that increasing the wind speed from 0.5 to 3.0 m s^{-1} elevated the total five day loss by a factor of 0.29 and that wind speed effects may have been altered by a temperature interaction. Lower temperatures appear to increase the formation of $\text{NH}_4^+ \text{-N}_{(\text{aq})}$ which reduces the amount of 'volatilizable' NH_3 present in the soil solution (Sherlock and Goh, 1984). Perhaps this temperature interaction could explain the conflicting findings as to the effects increased wind speed has on volatilization. Beauchamp et al. (1978) noted that of the many meteorological parameters measured for their sewage sludge study, temperature appeared to be the most closely related to the measured flux rate, especially 2 to 3 days after application. However, researchers in Australia found that temperature had very little influence on their research of cattle slurry application to grasslands (Thompson et al., 1990). Nelson (1982) stated that increased temperature has the potential to elevate the rate of NH_3 diffusion from the soil surface, which permits a more rapid conversion of $\text{NH}_3_{(\text{aq})}$ to $\text{NH}_3_{(\text{g})}$. A laboratory study using pig slurry on a dry, well-cultivated clay soil by Svensson (1993) showed that the NH_3 concentration increased 3 fold when the temperature increased from 14 to $24 \text{ }^\circ\text{C}$. Fenn and Kissel (1974) found that the effect of temperature on total NH_3 volatilization with ammonium nitrogen salts was dependent upon the presence or absence of CaCO_3 in the soil and upon the type of $\text{NH}_4^+ \text{-N}$ compound applied. They concluded that the influence of temperature was most pronounced in the loss rate; at the lowest temperature the smallest amount of first day NH_3 was released and the highest temperature produced the highest first day NH_3 losses.

An explanation for the effect on temperature is given according to Beauchamp et al.

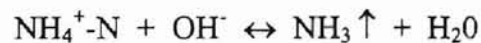
(1982):

Lauer et al. (1976) suggested that the partial pressure gradient between the manure and the atmosphere was a major factor determining flux. It follows that the ammoniacal-N concentration in manure is a major determinant of the flux as suggested by Vlek and Stumpe (1978). The diurnal pattern involving maxima near midday and minima in the early morning hours must then be related to a varying ammoniacal-N concentration in the manure. It may be surmised, therefore, that during the later morning hours as dew-water evaporation occurs, the concentration of ammoniacal-N in the aqueous phase increases, the partial pressure gradient also increases and its release would be enhanced. After midday, the flux peak is usually reached and is followed by a decrease coinciding with a depletion in ammoniacal-N available for volatilization. Also, as the temperature decreases, the water vapor pressure deficit in the atmosphere would decrease, thereby lessening the evaporation rate.

Relative humidity of the air has an inverse effect on the rate of water loss from the soil surface. Hargrove et al. (1977) demonstrated that the rate of ammonia loss after the application of ammonium salts in a field experiment followed a diurnal pattern, which followed the fluctuations of the atmospheric relative humidity. During intervals of high humidity, Black et al. (1987) suggested that water was absorbed on to the urea granules which stimulated hydrolysis leading to volatilization. Ryden and McNeill (1984) found that rainfall and low rates of evapotranspiration lowered the NH_3 flux. Rainfall tended to decrease the ammonia flux according to Beauchamp et al. (1982). However, since cooler air temperatures sometimes occurred with precipitation, it was difficult to determine whether the lower rate was due to the rainfall event, the lower temperatures, or a combination. They theorized that the water could have leached a portion of the soluble ammoniacal N into the ground, thus decreasing volatilization rates. Brunke et al. (1988) stated that the primary parameter determining volatilization is the drying rate of the manure according to the partial pressure of NH_3 . They also suggest that a derived

meteorological variable, such as the hay drying index already utilized by some network agrometeorological forecasts, may be the most suitable single indicator of high-volatilization weather conditions. Results were inconclusive in an attempt to correlate observed volatilization rates of NH_3 to specific meteorological parameters such as temperature, wind speed, net radiation, etc. (Brunke et al., 1988). These Canadian researchers found that these variables were not independent under field conditions.

Ammonia volatilization is controlled by a complex series of interactions and reactions governed by many soil characteristics. The potential for volatilization is highly dependent on the texture and type of soil to which the manure is applied. Researchers often disagree as to which soil characteristics are the most important. The pH effect is expected to play a fundamental role in this process due to the equilibrium equation as discussed below; as pH increases, the concentration of NH_3 present in the soil solution and soil air will increase, therefore increasing the potential for NH_3 loss from the soil system (Freney et al., 1983). DuPlessis and Kroontje (1964) compared soils with a range of pH values (4.5 to 7.1) and demonstrated that NH_3 volatilization increased with higher soil pH and with increasing concentrations of NH_4^+ -N applied to the soil as $(\text{NH}_4)_2\text{SO}_4$. The equilibrium equation can be described as:



with a shift in the reaction to the right when higher concentrations of OH^- are present which enhances the possibility of volatilization.

Upward movement of water through the soil profile aids the transport of ammonia to the soil surface, therefore a relationship would be expected between water evaporation and NH_3 loss (Freney et al., 1983). Numerous researchers have shown that water

evaporation and NH_3 loss are related (Jewitt, 1942; Martin and Chapman, 1951; Wahhab et al., 1957; Denmead et al., 1976; Fenn and Escarzaga, 1977). Under drying conditions, the amount of volatilization increased when applied to soils that had a greater initial moisture content. However, when the soil moisture content remained consistent, there was less volatilization from the soil with the greater moisture content (Burch and Fox, 1989). Australian and New Zealand scientists, working with urea, found that application of water shortly after spreading urea significantly decreased the NH_3 loss. The urea was carried by water into the soil before any hydrolysis processes could occur, thus decreasing the rate of NH_3 flux (Black et al., 1987). However, Ferguson and Kissel (1986) stated that rapid drying of the soil was found to significantly decrease the rate of NH_3 loss from soils when working with urea. They found that when the soil-surface water content declined to the point in which soil-water content/potential was not adequate to support urea hydrolysis that NH_3 losses would diminish.

Since ammonium is a positively charged ion, it reacts with the exchange complexes of the soil (Freney et al., 1983). In general, the higher the CEC value the less potential for ammonia volatilization. According to Whitehead and Raistrick (1993) the controlling soil factor related to volatilization of NH_3 with cattle urine experiments, is the cation exchange capacity. Nonetheless, a critical CEC level for NH_3 volatilization has not been proven to our knowledge, but it appears that a CEC value greater than $25 \text{ cmol}_c \text{ kg}^{-1}$ is required to reduce NH_3 losses substantially (Freney et al., 1983). Volatilization losses decreased as clay content increased, yet increased with additional amounts of municipal biosolids applied (Beauchamp et al., 1978). An inverse relationship should be found between NH_3 volatilization and clay content, since CEC is a function of the

quantity and types of organic matter or clay mineral types present in the soil (Freney et al., 1983). A linear inverse relationship was determined between NH_3 evolution and clay content (Ryan and Keeney, 1975). However, results on the relationship between soil factors and NH_3 loss can easily be misleading due to various interactions between soil parameters. With increased CEC, water holding capacity and buffering capacity of soils typically increase; both of these factors have been documented to influence NH_4^+ -N reactions in the soil (Fenn and Escarzaga, 1977; Ferguson et al., 1984; Clay et al., 1990).

The combination effects of a soil/manure system such as the buffering capacity and pH value are difficult to isolate and analyze separately. However, a few of the characteristics can be investigated further with some beneficial conclusions. Manure fluidity is a characteristic affecting volatilization rates, which can be described as the capability of a fluid to infiltrate a soil (Svensson, 1993). Svensson states that this characteristic should be considered the most effective manure parameter influencing NH_3 volatilization rates because fluidity has a major influence on the capacity of the manure to enter the soil. This parameter seems to be a more representative measure of the infiltration capacity of slurry than the often used total solid contents. Rapid infiltration of slurries into soils immediately after application may reduce the rate of NH_3 loss compared with thicker slurries, which remain on the surface for longer periods of time (Pain and Thompson, 1989). Sommer and Olesen (1991) found that the loss of NH_3 was linearly related to the dry matter content; small changes in dry matter content at low and high amounts had very limited influence on NH_3 volatilization. Volatilization from surface-applied cattle slurry can be decreased by more than 90% if the pH is lowered to 5.5 with strong acids (Stevens and Logan, 1987). An unreplicated wind tunnel study by

Pain et al. (1990) indicated that NH_3 volatilization from acidified slurry (pH 5) prior to application reduced losses by 30 to 54% when applied to perennial ryegrass in the UK. Using 2 M H_2SO_4 , Pain's research project required 30 to 85 mL L^{-1} to reduce the pH of the slurry to approximately 5.5. However, the economic feasibility of this treatment is questionable (Miyamoto et al., 1975). Freney et al. (1983) suggested that additions of acids to animal wastes being applied as fertilizer could be a useful practice where cheap by-product acids are available. However, this increased the losses of N from denitrification by approximately 40%. Results from Sharpe and Harper (1997) indicated that NH_3 emissions ($29.7 \text{ kg N ha}^{-1}$) contribute to atmospheric N-loading much more than do N_2O emissions (4.7 kg N ha^{-1}) from irrigated swine effluent applications, therefore, volatilization is still a greater concern than denitrification. Again, accessibility and economic feasibility of this type of treatment was not addressed.

Animal manures have the potential to lose significant amounts of NH_3 in the atmosphere when applied to the soil surface and not incorporated. However, in the experiments carried out by Svensson (1993), the mode of application was found to be the factor of highest influence in volatilization. He concluded that there was no significant difference between broadcast and banded application rates as a percentage of total volatilized NH_3 . Nonetheless, he did find that application technique made a substantial difference; injected swine manure decreased volatilization rates considerably when compared to surface spreading. However, contrary to other reports, Sommer and Christensen (1990) pointed out that NH_3 volatilization could be excessive if slurry is injected into a very wet and compressed soil. Application uniformity also seemed to have a notable effect during surface application. Band spreading seemed to produce

lower amounts of NH_3 in the air during the first hour as compared to broadcast spreading (Thompson et al., 1990). Interestingly, they also found that broadcast and band application follow different time courses for the rate of NH_3 volatilized and became similar as the volatilization neared an end. The process of ammonia volatilization is a highly complex system and is affected by a combination of biological, chemical and physical factors (Svensson, 1993). Many interact with each other, making it difficult to differentiate among them. By taking a more in depth examination of these many factors, combinations, and systems, a better understanding of how the pathway of ammonia volatilization functions will be gained.

Combining the information concerning the characteristics that could potentially increase NH_3 volatilization from livestock waste application and the recent micrometeorological techniques, the Panhandle of Oklahoma warranted research of this type. The majority of swine operations in the state use facultative anaerobic lagoon systems in which the top layer aerobically oxidizes and breakdowns much of the volatile organics to form CO_2 and H_2O , which reduces odors. The middle portion anaerobically digests a portion of the materials and the solids sink to the bottom of the lagoon to form a slurry. Most effluent, taken from the top layer of lagoons, is applied in this region of the state through center pivots. Ammonia solutions applied through sprinkler systems usually allow for greater exposure to the air, increasing the chance for water evaporation and NH_3 losses (Warnock, 1966). Henderson et al. (1955) reported that NH_3 losses could be greater than 60 % when anhydrous ammonia was applied through a jet type sprinkler system. Since the partial pressure of NH_3 under normal atmospheric conditions is low,

dissolved NH_3 readily volatilizes upon exposure to the atmosphere (Miyamoto et al., 1975).

Wind speeds are often very high ($4.4 - 17.8 \text{ m s}^{-1}$) during the spring in the Panhandle of Oklahoma; as much as 15.64 m s^{-1} wind speeds have been recorded for a 12-hour period (Climatological, 1996). Wind velocities of more than 12 m s^{-1} over a period of 24 hours have been reported at the Goodwell Station (USDA, 1984). Relative humidities are often very low (3 – 30%), especially during the summer months (Climatological, 1996), and rainfall in Texas County averages approximately 43 cm year^{-1} (USDA, 1984). The swine population has increased exponentially over the last decade, growing from 370,000 to 905,000 total swine between 1996 and 1998 (National, 1997). Fifteen years ago, the vast majority of swine owners raised their livestock as free range hogs. Now, only small portions are raised in this fashion, with confined swine feeding operations dominating the area. Therefore, in light of these area environmental factors, which typically tend to increase NH_3 volatilization from N additions, the objective of these field experiments was to determine the NH_3 flux during land application of swine effluent in the Southern Great Plains using a simple passive sampling system.

MATERIALS AND METHODS

Samplers

Simple passive samplers, as described previously in Chapter 1, were utilized under field conditions.

Field Measurements

Field tests were established in the spring, early summer, and late summer of 1998 to monitor the NH_3 volatilization of swine effluent applied to the land surface at the Oklahoma State University Research Station in Goodwell, Oklahoma. A Richfield clay loam (fine, smectitic, mesic Aridic Argiustolls) had an average pH of 8.1, 3 mg kg^{-1} NH_4^+ -N, 22 mg kg^{-1} NO_3^- -N (KCl extractable), and 0.13 % Total N (Table 1). Samplers were placed on four masts placed at right angles to each other on the boundary of a circular plot (Figure 1). On each mast, four samplers were positioned at 17, 47, 109, and 184 cm above ground, in which each sampler had one unit having the stainless steel disc facing toward the NH_3 source and the other facing away from the experimental area. As noted by other researchers and as demonstrated from the wind tunnel experiments described previously, volatilized NH_3 tends to move laminarly along the surface with only small amounts being carried into the higher sampling areas (Ferm and Svensson, 1993). Therefore, these heights were chosen based on research from previous experiments using micrometeorological methods (Beauchamp et al., 1978; Beauchamp et al., 1982; Schjoerring, 1995) and are intended to capture NH_3 that volatilizes and stays relatively close to the surface. The highest sampling position should capture very little NH_3 in order to ensure that the flux is not underestimated due to large amounts of NH_3 moving above the sampling area. Horizontal flux can be determined by utilizing the

mass balance approach, which measures the mean atmospheric gas concentration minus the background gas concentration and the mean horizontal wind speed at several heights downwind from the leading edge of a plane source (McInnes et al., 1985). The average horizontal flux of NH_3 (F_{hm} , $\mu\text{g NH}_3\text{-N m}^{-2} \text{ s}^{-1}$) through two glass tubes facing in the same direction at each height (h) measured on each mast (m) either toward (the exposed tubes) or from (background tubes) the NH_3 source was calculated by using Equation 1 proposed by Schjøerring et al. (1992):

$$F_{hm} = \frac{(C_1 + C_2) V}{2\pi r^2 \cdot K \cdot \Delta t} \quad [1]$$

where C_1 and C_2 are the concentrations of $\text{NH}_4^+\text{-N}$ ($\mu\text{g NH}_4^+\text{-N L}^{-1}$) in either of the two exposed tubes E_1 and E_2 or the background tubes B_1 and B_2 (Figure 1); V is the volume of water used to dissolve the NH_4^+ sorbed in the tubes (3 mL), r is the radius of the hole in the stainless steel disc (0.05 mm), t is the time (s) between the start and conclusion of the measurement and K is the correction factor ($K = 0.77$).

The horizontal net flux ($F_{net}(h)$, $\mu\text{g NH}_3\text{-N m}^{-2} \text{ s}^{-1}$) at each height was calculated with Equation 2 by Schjøerring et al. (1992):

$$F_{net}(h) = \sum_{m=1}^{m=4} (F_{hm,s} - F_{hm,b}) \quad [2]$$

where $F_{hm,s}$ is the flux of NH_3 from the source including the exposed tubes and $F_{hm,b}$ the flux measured from the background tubes at each height (h) and at each mast ($m=4$).

Sommer et al. (1996) noted that in high wind environments, a slight bypass was observed due to rather high wind velocity in the center of the tube. In these instances, the horizontal net fluxes of NH_3 were calculated by adding the $\text{NH}_4^+\text{-N}$ in the background and exposed tubes instead of subtracting them (contrary to the procedure shown in the

previous equation). Test plots in these field experiments also demonstrated this phenomenon; so, to determine NH_3 flux during high wind events, the background and the exposed tubes were added together. Captured NH_3 from the zero application plot was subtracted from the treatment plot values to ensure that any ambient NH_3 was not included into flux calculations.

Vertical flux (F_v , $\mu\text{g NH}_3\text{-N m}^{-2} \text{ s}^{-1}$) was determined from Equation 3 (Sommer et al., 1996):

$$F_v = \frac{1}{x} \sum_{h=1}^{h=n} F_{\text{net},h} \Delta h \quad [3]$$

where x is the diameter (m) of the plot, Δh is the height interval (m) of the samplers, and n is the number of measuring heights at which the samplers were mounted to the masts ($n=4$). Once the vertical flux was calculated for each treatment plot, the values were averaged and integrated using a second order inverse polynomial, which was empirically determined using SigmaPlot to find the best-fit equation to model the data (SPSS, 1997). Sommer and Ersbøll (1994) described NH_3 losses from surface applied swine and cattle slurry using a Michaelis-Menten like equation, which is a first-order reaction. Regression equations for each test period are shown in Table 2.

Swine lagoon effluent was obtained from a production facility and effluent was collected using a large tank. Using a flowmeter to monitor application amounts, 4625 L of effluent were applied from a 5.1 cm hose onto each treatment plot. The effluent from the same lagoon was used in each of the 3 field tests and remained fairly consistent in nutrient content and other characteristics (Table 1). The ammonium content averaged $963 \text{ mg NH}_4^+\text{-N L}^{-1}$, ranging from $868 - 1081 \text{ mg NH}_4^+\text{-N L}^{-1}$ with a standard deviation

of 86. Nitrate in the effluent was negligible ($< 1 \text{ mg L}^{-1}$), the average pH was 7.3, Total N averaged 1002 mg L^{-1} , and the effluent had $< 1\%$ solid.

Field areas chosen for all three sampling times on the research station were situated very far from any obstructions of airflow such as structures, trees etc. Special care was taken to avoid any potential sources of NH_3 , i.e., center pivots, hog facilities etc. However, to demonstrate this, samplers were allowed to monitor ambient air approximately 16 hours before effluent additions during the September tests to measure background levels of NH_3 in the atmosphere (Figure 2). Level portions of the field areas were chosen for the 15.24 m diameter plots to maintain equal coverage and small levees ($> 5 \text{ cm}$ in height) were built around the plots to ensure that the liquid would not run off once applied. Bare, recently disked fields were used for each test plot, with a 1-3% wheat stubble residue. Three treatment plots and one zero application plot were initiated on May 28, July 28, and September 12, 1998, with new experimental areas chosen for each test period. Treatment and zero application plots were placed over 91 m apart from each other to ensure that no cross contamination between plots was possible. Samplers collected NH_3 for a 7 day period, (except for September where only 5 days were used) in which samplers were taken off and replaced with new samplers approximately every 12 hours for the first 4 days and every 24 hours for the last 3 days. September's testing date was shortened to 5 days because the previous two testing periods had yielded no significant NH_3 losses after day 5.

Soil samples were taken before and after application in which soil moisture was measured along with $\text{NH}_4^+\text{-N}$ and $\text{NO}_3^-\text{-N}$. A soil mass balance was performed for each testing period using collected soil samples, effluent concentrations, and captured $\text{NH}_4^+\text{-N}$

on the samplers. Using a Campbell Scientific CR500 datalogger, wind speed, direction, soil temperature, relative humidity, and air temperature were monitored. Weather conditions were variable between the 3 test times and are summarized in Table 4.

RESULTS AND DISCUSSION

The vertical flux and cumulative loss of NH_3 for the May 28 – June 3 sampling period as determined by the passive flux samplers is shown in Figure 3. The initial NH_3 flux for these test plots was $151 \mu\text{g NH}_3 \text{ m}^{-2} \text{ s}^{-1}$, as shown by the best fit line, a second order inverse polynomial with a p value < 0.001 . Ammonia volatilization decreased rapidly and within 3 days very little NH_3 was detected. Any NH_3 volatilizing from these plots after day 3 were insignificant. Measured levels of NH_3 before swine effluent applications in September indicate that background NH_3 fluxes are typically below $40 \mu\text{g m}^{-2} \text{ s}^{-1}$. Background NH_3 levels, as detected by the zero application plot after application during this May field test, were initially quite high, with horizontal fluxes ranging from 344 to $200 \mu\text{g NH}_3 \text{ m}^{-2} \text{ s}^{-1}$ and eventually declining to $20 \mu\text{g NH}_3 \text{ m}^{-2} \text{ s}^{-1}$ within 24 hours. This would seem to suggest that the zero application plot was not far enough from the treatment plots. Ammonia volatilization from the 3 treatment plots showed little deviation among reps (Figure 4), but followed expected diurnal fluctuations for the first 3 days (McGarity and Rajaratnam, 1973; Denmead et al., 1974; Beauchamp et al., 1978; Beauchamp et al., 1982; Ryden and McNeil, 1984). Cumulative NH_3 losses for this test period totaled 154 kg ha^{-1} , which was a loss of 57% of the total amount of NH_4^+ -N that was applied initially (Table 5). The weather conditions during this test period (Table 4), which were hot, dry, with very low relative humidity, and brisk wind speeds, likely influenced the loss of 57% NH_3 , which would be a significant loss for producers in this region. Research in Georgia using swine effluent applications to an irrigated oat field during the heading stage resulted in 82% of the NH_4^+ -N in the effluent lost to the atmosphere (Sharpe and Harper, 1997). This value includes a 13% NH_3 loss during

application with a solid-set sprinkler irrigation system. These results account for volatilization before the effluent even reached the crop or soil surface. These higher results could be due to applications to a sandy soil, and to increased losses due to interception by the crop canopy which reduces the opportunity for the NH_4^+ -N to enter the soil. On the contrary, effluent for these Panhandle field experiments was surface applied to bare ground to reduce and remove any potential application variable that might be encountered when effluent adheres to leaf canopies or to volatilization of NH_3 during the actual application itself. For these field research trials, measurements of NH_3 were restricted to losses encountered after application to determine volatilization rates without these additional variables.

Due to rainfall events 5 nights through the 7 day test period, ammonia losses were much less during the July 28 – August 3 experiments with an initial vertical flux of $101 \mu\text{g NH}_3 \text{ m}^{-2} \text{ s}^{-1}$ (Figure 3). Air temperature was not significantly different than during the May fields tests, but due to rainfall, minimum and average humidities were much higher (Table 4). Danish research, such as that done by Van der Molen et al. (1990) demonstrated similar volatilization declines during rainfall events when cattle slurry was applied to arable land. Weather conditions create highly variable results due to interactions between the various climatic conditions, manure characteristics, and manure handling as described by Christensen (1986). The sampling period for this study was characterized by higher humidity, thunderstorms, and high soil moisture, all of which could have contributed to lower relative amounts of volatilization. Texas County, OK, receives only 43 cm yr^{-1} on average, so that a 5.05 cm rainfall total for a 7 day period is significant enough on this clay loam soil to ensure that NH_4^+ -N in the effluent will

probably not volatilize to any large degree. Volatilization rates dropped off very quickly and after approximately 2 days, NH_3 losses were extremely low. Temperatures during the day warmed up considerably, creating a rapid drying effect from the previous night's rainfall. This created diurnal fluctuations, which were somewhat more noticeable than for the other 2 test periods (Figure 4). Cumulative NH_3 losses were calculated to be 82 kg ha^{-1} of the total 221 $\text{kg NH}_4^+ \text{-N ha}^{-1}$ that was applied which was a 37% loss (Table 5). It is believed that larger standard deviations for this field test resulted from thunderstorms and high winds encountered, which may have contaminated some samplers with water or blown soil. Ammonia measurements from the zero application plot were initially high (between 120 and 30 $\mu\text{g NH}_3 \text{ m}^{-2} \text{ s}^{-1}$) but decreased by day 2 and remained below 30 $\mu\text{g NH}_3 \text{ m}^{-2} \text{ s}^{-1}$ for the remainder of the sampling period.

In a cooperative effort between Oklahoma State University and North Carolina State University, an enclosure method currently being used by NCSU was brought to the Oklahoma Panhandle Research Station during the July field test to be compared to passive samplers already in use. This method resulted in a 56% loss of applied ammoniacal N 49 hours after effluent application (D. Byers, 1999, personal communication). This is a tremendous difference from the 37% loss of volatilized ammonia after 7 days, as measured by the micrometeorological method during this same field test. Without further replications or data, any conclusions concerning these differing results, other than as just a simple comparison are somewhat fruitless, except to show the significant differences between methods even under the same conditions.

Ammonia flux from field plots during the September 12 – 17 test period was initially 115.2 $\mu\text{g NH}_3 \text{ m}^{-2} \text{ s}^{-1}$, which then dropped quickly and was insignificant after

approximately 3 days (Figure 3). Sharpe and Harper (1997) also reported high NH_3 emissions immediately following irrigation with swine effluent, with rates dropping to within background levels after 12 to 24 hours. Cumulative amounts of volatilized NH_3 during this September run were calculated to be 117 kg ha^{-1} over a 7 day period, which is a 50% loss of the $236 \text{ kg NH}_4^+ \text{-N ha}^{-1}$ that was applied in the swine effluent (Table 5). Background levels of NH_3 were low before application ($< 20 \mu\text{g NH}_3 \text{ m}^{-2} \text{ s}^{-1}$) and followed a diurnal pattern after application but unlike the first field run in May, rarely exceeded $40 \mu\text{g NH}_3 \text{ m}^{-2} \text{ s}^{-1}$. This would seem to indicate that the zero application plot was placed far enough away so as to not receive any NH_3 from the treatment plots as was noted in the first two field experiments. Weather data for this sampling period was milder than the previous two experiments, which would account for the moderate NH_3 loss (Table 4). Lower air temperatures and wind speeds were also noted to be the probable cause for lower NH_3 flux measurements during effluent applications in Georgia (Sharpe and Harper, 1997).

During the September field tests, samplers were allowed to monitor ambient background air for approximately 16 hours before effluent applications to demonstrate that no measurable amounts of NH_3 were in the atmosphere in the testing area. From these preapplication measurements in September, it can be seen that NH_3 levels in the atmosphere were negligible before effluent was applied (Figure 2).

For all sampling periods, the samplers at the lowest height consistently detected more NH_3 than the 3 higher heights (Figure 2). This would be expected due to lateral movement of air currents across the plot being more significant than the vertical movement over such short fetches. Ferm and Svensson (1993) reported similar results in

NH₃ emission flux measurements using a simple passive sampler. They found that horizontal NH₃ flux is higher near the ground and decreases approximately exponentially with increasing height. This trend was also detected when using passive samplers in the wind tunnel as described previously (Figure 9, Chapter 1). Only at the initial sampling periods could any detectable amounts of NH₃ be measured at the highest position (184 cm).

Reports from other NH₃ volatilization research projects using animal waste resulted in very similar values to those found in the experiments performed for this study. A micrometeorological study in the Netherlands by Van der Molen et al. (1990) found that 32 and 67% of the ammoniacal N applied from cattle slurry to bare soil was lost to the atmosphere from two, nine day experiments. The higher value for the first test was theorized to be due to a higher surface pH, higher wind speed and lower humidity throughout the testing period. They also noted diurnal NH₃ fluctuations from cattle slurry applications, with maximum fluctuations occurring at midday and minimum fluctuations at midnight. Ammonia volatilization experiments in the UK and the Netherlands found 24 and 62% losses from applied ammoniacal N in pig slurries to ryegrass using a micrometeorological mass balance method (Pain et al., 1989). A micrometeorological mass balance determination of NH₃ volatilization from surface applied cattle slurry in Northern Ireland resulted in a 51% loss of applied NH₄⁺-N over a 7 day period (Stevens and Logan, 1987). Beauchamp et al. (1978) reported 60 and 56% of the NH₄⁺-N in surface applied sewage sludge volatilized within a week's sampling period. To our knowledge, the only research with manure application measurements using the integrated horizontal flux method and the passive flux samplers used for this experiment was done

in the Southeast United States between 1995-1997. Using the samplers, as described by Schjøerring et al. (1992), 28 to 46% of the NH_4^+ -N applied from broiler litter to tall fescue pastures was volatilized in 3 southeastern states (GA, AL, TN) over a 14 day testing period (Marshall et al., 1998). Other work resulted in an average 23% loss of total applied N as NH_3 when swine slurry was applied to bermudagrass pastures (Sullivan et al., 1998).

Mass balance calculations for the 3 test periods were variable and prone to overestimations and underestimations in several parameters (Table 3). Average percent recoveries for the three field trials were 90, 101, and 117 for the May, July, and September periods respectively with standard deviations of 14.2, 15.9, and 15.2 respectively. Mass balances of this type are often inaccurate due to the many transformations of nitrogen that are difficult to account for such as immobilization from microbial growth. Surface amounts of soil total nitrogen are only 0.05 to 0.10 %, while inorganic N represents only a small fraction ($< 2\%$ of the total N in soils) (Bremner, 1965). Total nitrogen values, with a precision of 0.01%, are known to be $\pm 224 \text{ kg ha}^{-1}$ (Raun et al., 1998) and are therefore prone to large miscalculations. Overestimations of NH_3 losses were found using a nitrogen recovery mass balance method ($\text{N loss} = \text{N recovered immediately following application} - \text{N recovered after time}$) from the broadcasting of urea granules onto a pasture (Black et al., 1985). They summarize that this method resulted in higher estimated underestimations due to other processes, such as immobilization, and that this calculation became more prone to error over longer time intervals. Also, for our experiments, assumptions were made that could introduce error, such as the soil bulk density (D_B) being equal to 1.33 g cm^{-3} , sampling depths being

consistently at 15.24 cm, etc. However, a 10% change in bulk density would result in only a 2% variation, which would therefore not result in a large source of error. Although a flowmeter was used to apply effluent to each plot, this could also be a potential source of error. However, a 10% change in the volume of effluent that was added would contribute only an 8 % variation. The concentration of effluent that was measured from tank to tank would more likely be a source of error. A 10% change in the concentration of effluent would produce only a 6% change. Another large source of variation could be the actual concentration of captured NH_4^+ -N on the samplers. Although these samplers had an excellent recovery rate under laboratory tests, contaminated tubes are always a concern. Under some high wind environments and thunderstorms, some samplers collected small amounts of blown soil or rain. A 10% variation in the tubes' ability to capture ammonia under such circumstances would result in a 5% change in the percent recovery of the system. If a 10% change in all four of the above parameters were to occur, only a 6% variation is anticipated. So, the percent recoveries for the system seemed to turn out quite well considering the above potential sources of error. During the May test, 85, 79, and 106% recovery of the system was calculated for each plot. An 83, 103, and 115% recovery was calculated for July's tests, while September's recoveries were all over 100, at 134, 108, 107%. Recoveries over 100% could be due to potential errors in the mass balance itself as described above.

Despite inherent problems when dealing with soil-N mass balance estimates, percent recoveries were all above 78%, indicating that these samplers have the ability to monitor NH_3 emissions under varying field environments. The objective was to determine to what degree NH_3 losses were occurring, due to the environmental

characteristics and to the large increase in confined swine feeding facilities in this area of the state. Based on our results, these samplers should enable researchers to provide more accurate estimates of the potential NH_3 losses that can be anticipated in this region under the environmental conditions studied. Producers can then utilize this information for better N budgets when applying swine effluent to their cropland. Nitrogen buildups do not appear to be a major concern with this production system in the Southern Great Plains. The majority of N being introduced into these corn/wheat cropping systems from swine effluent additions will most likely be utilized by the plant or volatilized after application from center pivots. However, P and salinity levels should be monitored if effluent applications are added at levels to counter NH_3 volatilization losses.

CONCLUSIONS

Ammonia volatilization from swine effluent can be affected by several different factors such as soil pH, CEC, texture, swine effluent pH, application method, soil moisture, irrigation droplet size, air temperature, humidity, and wind speed. Climatic conditions seemed to have the most dramatic affect on NH_3 volatilization from field tests, with results ranging from 37 - 57% loss of the total NH_4^+ -N that was applied from swine effluent. These are significant losses to the producers of the region and should be taken into account when managing N.

Obviously there has been a tremendous amount of research on NH_3 volatilization from both commercial fertilizers and also from varying animal wastes. However, research specific to this region of the United States, which has so many factors prone to increasing NH_3 volatilization from swine effluent applications is lacking. Therefore, being able to provide a reliable range of numbers over differing environmental conditions specific to this region of the state will enable producers in this area to more properly manage their N. Also, these numbers could potentially yield useful information on the amount of NH_3 that is entering the atmosphere, and could be used to minimize the amount of unnecessary NH_3 being lost into the air. This will benefit the air quality of the area and help decrease potential odor problems, which will become more important as the population of swine continues to rise in this area of Oklahoma.

Even though soil mass balance calculations are prone to error, the percent recoveries for the three test periods were all within an acceptable range (79 – 134 %). Overall, from the soil mass balance calculations and the volatilization results, these

samplers seem to be quite adequate at capturing and providing an estimate of NH_3 volatilization losses under variable field conditions.

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Table 1. Average initial soil characteristics over all sampling periods for Richfield clay loam and swine effluent characteristics

| | Soil Characteristics | Effluent Characteristics |
|---------------------------------|--|--------------------------|
| Soil Series | Richfield | ----- |
| Texture | clay loam | ----- |
| pH | 8.1 | 7.4 |
| NH ₄ ⁺ -N | 3 mg kg ⁻¹ | 963 mg L ⁻¹ |
| NO ₃ ⁻ -N | 22 mg kg ⁻¹ | 1 mg L ⁻¹ |
| Total N | 1.3 mg kg ⁻¹ | 1002 mg L ⁻¹ |
| P | 29 mg kg ⁻¹ | 95 mg L ⁻¹ |
| CEC | 16.12 cmol _c kg ⁻¹ | ---- |
| Organic Carbon | 12.1 g kg ⁻¹ | ---- |
| EC | 0.47 dS m ⁻¹ | ---- |

Table 2. Second order inverse polynomial regression equations for vertical NH₃-N flux from swine effluent applications on a Richfield clay loam

| | Regression Equation † | Coefficient of Determination r ² | p value |
|------|--|--|----------|
| May | $y = 0.1079 + 578.8759/x + 14710.3642/x^2$ | 0.99 | < 0.0001 |
| July | $y = 8.8903 + (-162.2165)/x + 6325.0944/x^2$ | 0.90 | < 0.01 |
| Sept | $y = 2.1418 + 454.6661/x + 4986.8851/x^2$ | 0.95 | < 0.001 |

† x = time, y = vertical NH₃-N flux

Table 3. Soil mass balance recoveries from each plot calculated on 7-day field volatilization studies

| Date | % Recovery | | | Mean | Standard Deviation |
|-------|------------|--------|--------|------|--------------------|
| | Plot 1 | Plot 2 | Plot 3 | | |
| May | 85 | 79 | 106 | 90 | 14.2 |
| July | 84 | 103 | 115 | 101 | 15.9 |
| Sept. | 134 | 108 | 108 | 117 | 15.2 |

Table 4. Weather data for the field volatilization studies

| | Temp | | | Humidity | | | Rain | Wind Speed | |
|------|---------------|-----|-----|-----------------|-----|-----|-------------|-------------------|---------------------------|
| | Min | Avg | Max | Min | Avg | Max | | Max | Avg |
| | ----- C ----- | | | ----- % ----- | | | | cm | --- m s ⁻¹ --- |
| May | 12 | 24 | 38 | 6 | 42 | 94 | 0.00 | 17 | 5 |
| July | 17 | 24 | 36 | 32 | 72 | 97 | 5.05 | 18 | 4 |
| Sept | 12 | 21 | 31 | 27 | 61 | 94 | 0.28 | 12 | 4 |

Table 5. Summary of field volatilization studies

| Date | NH₄⁺-N Added | NH₃-N lost | NH₄⁺-N lost |
|-------------|---|------------------------------|--|
| | -- kg ha ⁻¹ -- | -- kg ha ⁻¹ -- | --- % --- |
| May | 271 | 154 | 57 |
| July | 221 | 83 | 37 |
| Sept. | 236 | 117 | 50 [†] |

[†] based on 7 day regression outputs instead of 5 day

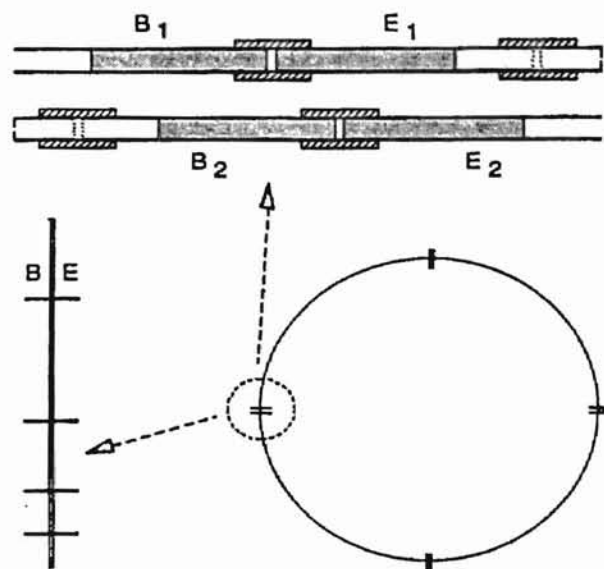


Figure 1: Passive flux sampler. Top: A passive flux sampler consisting of two parallel sampler units. Each unit is composed of 100 mm long glass tubes, coated with oxalic acid on approximately 70 mm of the inside (shaded), and a 23 mm tube with stainless steel disc at the end, all joined by silicone tubing. E₁, E₂ are exposed tubes and B₁, B₂ are background tubes. (Bottom left) Mast with four flux samplers seen from profile. (Bottom right) Four masts around a circular NH₃ source as seen from the top. From Sommer et al. (1996).

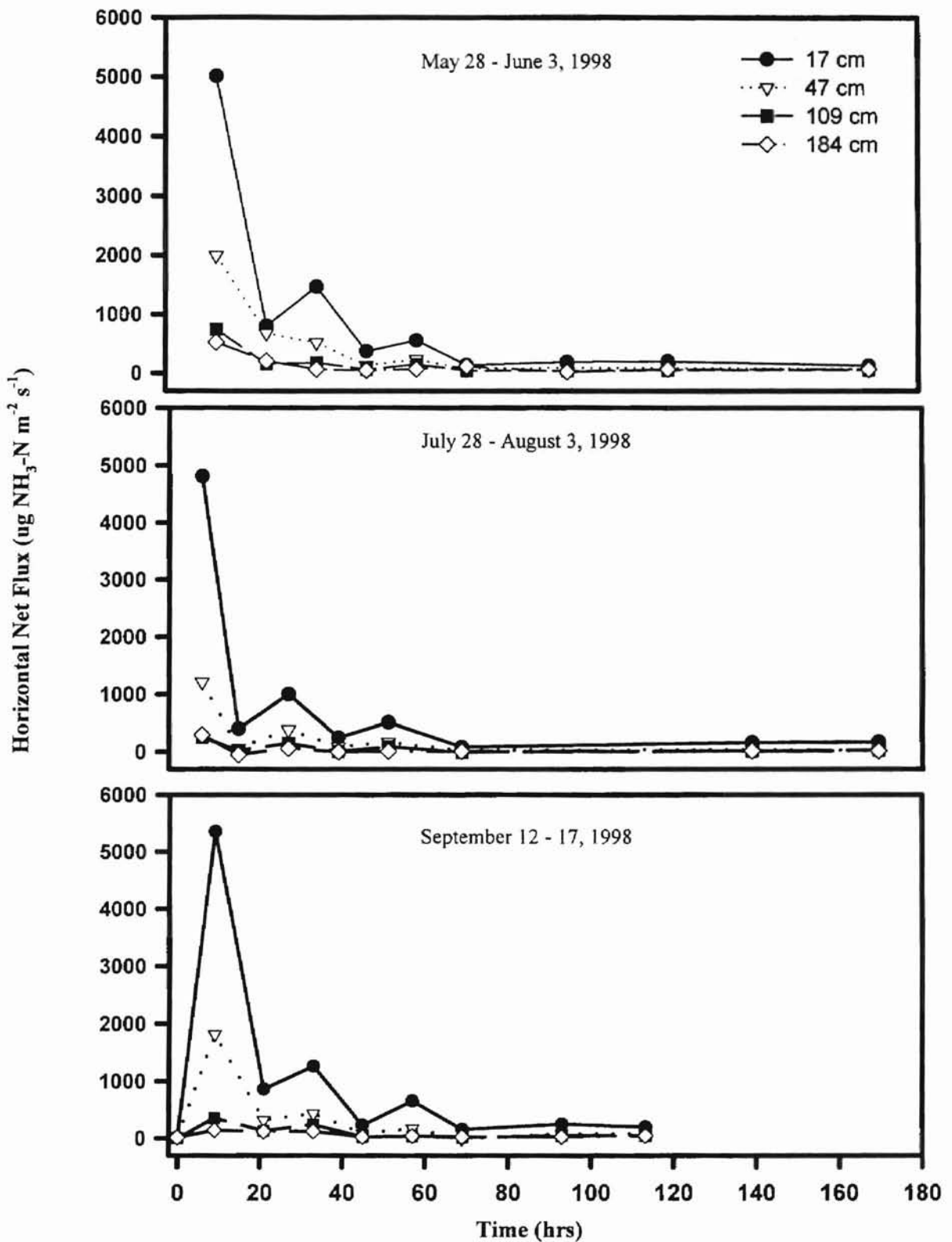


Figure 2. Average horizontal net NH₃ flux (n = 3) at each height for three field tests in Goodwell, OK. September tests were began approximately 16 hours before initial application of effluent. This explains the difference in the initial peak from the first two sampling periods.

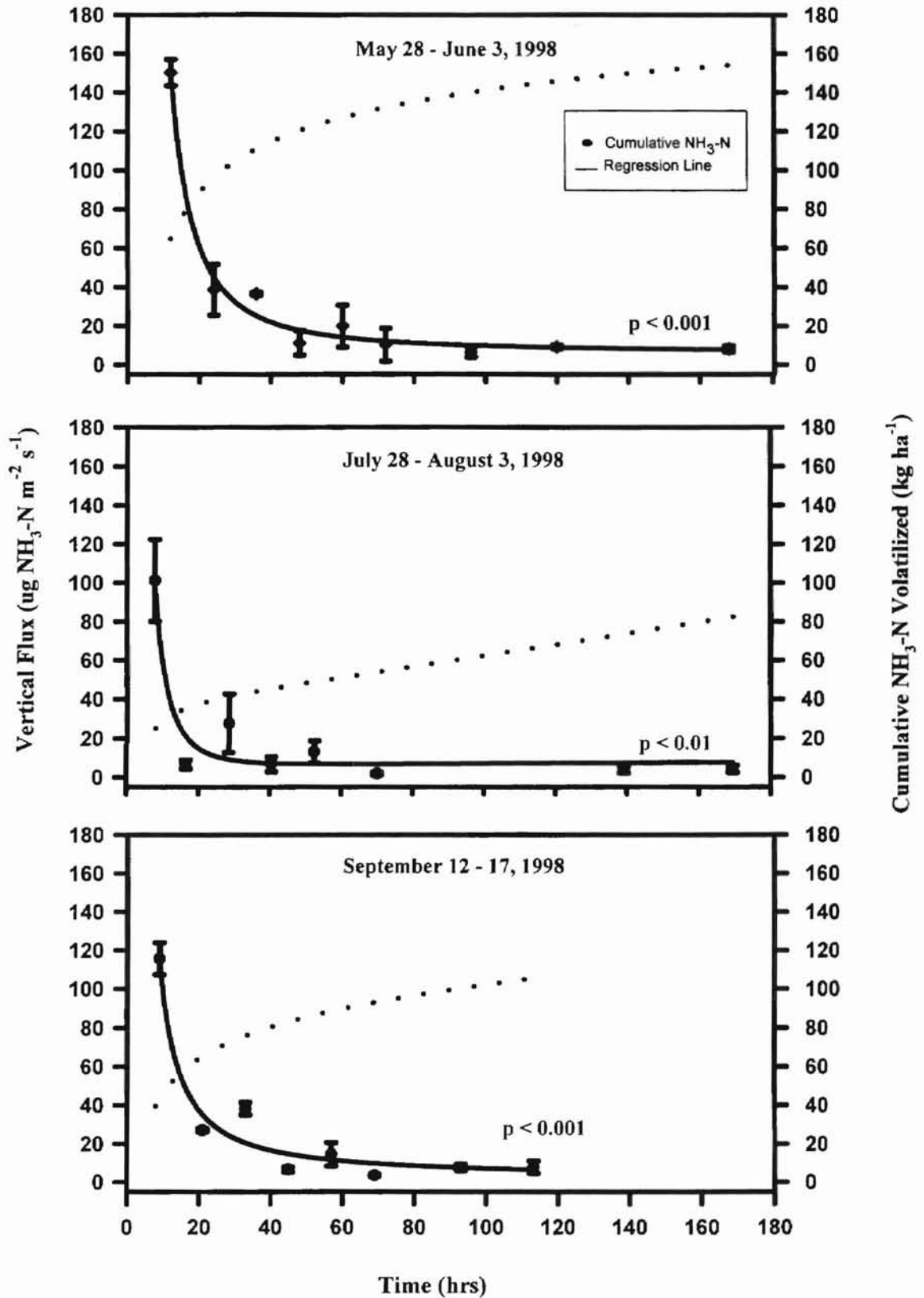


Figure 3. Vertical $\text{NH}_3\text{-N}$ flux and cumulative $\text{NH}_3\text{-N}$ lost from field plots during 3 field tests in Goodwell, OK. Bars represent standard errors.

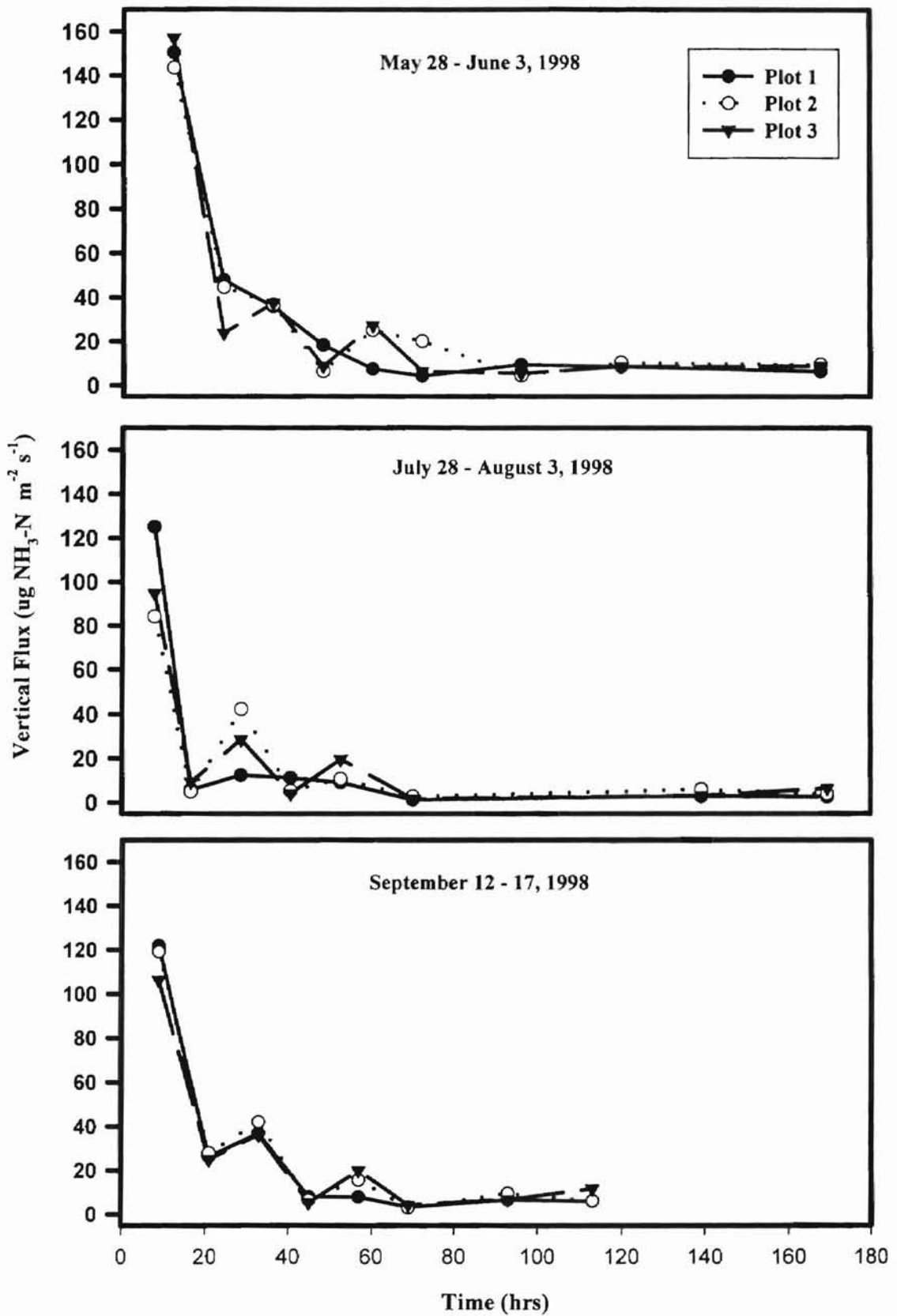


Figure 4. Vertical NH₃ Flux for each plot during three field tests in Goodwell, OK.

APPENDICES

APPENDIX I
Efficiency Determinations

Table I - 1
Percent Efficiency of Oxalic Coated Tubes: Run A

| Sample (3 mL) | Corresponding Solution | Minutes | Tube ($\mu\text{g mL}^{-1}$) | Tube (μg) | $\text{NH}_4^+\text{-N}$ captured in tubes (mole) | $\text{NH}_4^+\text{-N}$ calculated in tubes (mole) | % Eff. of tubes (mole) | % Recovery of System |
|---------------|------------------------|---------|--------------------------------|------------------------|---|---|------------------------|----------------------|
| 32 | 21 | 60 | 2.32 | 6.96 | 4.97E-07 | 5.27E-06 | 18.023 | 92.15 |
| 33 | 21 | 60 | 2.11 | 6.33 | 4.52E-07 | | | |
| 34 | 22 | 60 | 1.68 | 5.04 | 3.60E-07 | 5.29E-06 | 13.410 | 91.67 |
| 35 | 22 | 60 | 1.63 | 4.89 | 3.49E-07 | | | |
| 36 | 23 | 60 | 2.32 | 6.96 | 4.97E-07 | 5.16E-06 | 18.701 | 92.38 |
| 37 | 23 | 60 | 2.18 | 6.54 | 4.67E-07 | | | |
| 38 | 24 | 120 | 4.55 | 13.65 | 9.75E-07 | 6.40E-06 | 30.251 | 91.89 |
| 39 | 24 | 120 | 4.48 | 13.44 | 9.60E-07 | | | |
| 40 | 25 | 120 | 11.82 | 35.46 | 2.53E-06 | 1.01E-05 | 50.599 | 90.89 |
| 41 | 25 | 120 | 12.12 | 36.36 | 2.60E-06 | | | |
| 42 | 26 | 120 | 12.77 | 38.31 | 2.74E-06 | 9.56E-06 | 54.967 | 92.17 |
| 43 | 26 | 120 | 11.76 | 35.28 | 2.52E-06 | | | |
| 44 | 28 | 180 | 0.03 | 0.09 | 6.43E-09 | 4.05E-06 | 0.159 | 92.65 |
| 45 | 28 | 180 | 0 | 0 | 0.00E+00 | | | |
| 46 | 29 | 180 | 0.05 | 0.15 | 1.07E-08 | 3.89E-06 | 0.275 | 92.94 |
| 47 | 29 | 180 | 0 | 0 | 0.00E+00 | | | |
| 48 | 30 | 90 | 0.06 | 0.18 | 1.29E-08 | 4.18E-06 | 0.307 | 92.42 |
| 49 | 30 | 90 | 0 | 0 | 0.00E+00 | | | |
| 50 | 31 | 180 | 17.23 | 51.69 | 3.69E-06 | 1.16E-05 | 64.612 | 92.55 |
| 51 | 31 | 180 | 17.68 | 53.04 | 3.79E-06 | | | |
| 52 | 32 | 210 | 19.59 | 58.77 | 4.20E-06 | 1.24E-05 | 67.549 | 92.71 |
| 53 | 32 | 210 | 19.35 | 58.05 | 4.15E-06 | | | |
| 54 | 33 | 210 | 14.74 | 44.22 | 3.16E-06 | 9.67E-06 | 63.287 | 93.54 |
| 55 | 33 | 210 | 13.83 | 41.49 | 2.96E-06 | | | |
| 56 | 34 | 210 | 14.69 | 44.07 | 3.15E-06 | 1.02E-05 | 60.637 | 92.68 |
| 57 | 34 | 210 | 14.25 | 42.75 | 3.05E-06 | | | |
| 58 | 35 | 270 | 21.63 | 64.89 | 4.64E-06 | 1.35E-05 | 66.471 | 91.77 |
| 59 | 35 | 270 | 20.26 | 60.78 | 4.34E-06 | | | |
| 60 | 36 | 270 | 25.3 | 75.9 | 5.42E-06 | 1.51E-05 | 73.966 | 92.86 |
| 61 | 36 | 270 | 26.74 | 80.22 | 5.73E-06 | | | |
| 62 | 37 | 270 | 17.26 | 51.78 | 3.70E-06 | 1.52E-05 | 54.769 | 87.53 |
| 63 | 37 | 270 | 21.5 | 64.5 | 4.61E-06 | | | |
| 64 | 38 | 300 | 31.4 | 94.2 | 6.73E-06 | 1.32E-05 | 102.072 | 100.50 |
| 65 | 38 | 300 | 31.66 | 94.98 | 6.78E-06 | | | |
| 66 | 39 | 300 | 29.3 | 87.9 | 6.28E-06 | 1.49E-05 | 80.541 | 94.73 |
| 67 | 39 | 300 | 26.7 | 80.1 | 5.72E-06 | | | |
| 68 | 40 | 0 | 0.01 | 0.03 | 2.14E-09 | | | |
| 69 | 40 | 0 | 0.05 | 0.15 | 1.07E-08 | | | |

Table I - 1 cont.
Percent Efficiency of Oxalic Coated Tubes: Run A

| 62 mL final solution amount | Final Solution ($\mu\text{g mL}^{-1}$) | Final Solution (μg) | Final Solution (moles $\text{NH}_4^+\text{-N}$) |
|-----------------------------|--|----------------------------------|--|
| Solution21 | 11.23 | 696.26 | 4.97332E-05 |
| Solution22 | 11.225 | 695.95 | 4.9711E-05 |
| Solution23 | 11.255 | 697.81 | 4.98439E-05 |
| Solution24 | 10.975 | 680.45 | 4.86039E-05 |
| Solution25 | 10.13 | 628.06 | 4.48617E-05 |
| Solution26 | 10.26 | 636.12 | 4.54374E-05 |
| Solution28 | 11.505 | 713.31 | 5.0951E-05 |
| Solution29 | 11.54 | 715.48 | 5.1106E-05 |
| Solution30 | 11.475 | 711.45 | 5.08182E-05 |
| Solution31 | 9.805 | 607.91 | 4.34224E-05 |
| Solution32 | 9.63 | 597.06 | 4.26474E-05 |
| Solution33 | 10.235 | 634.57 | 4.53267E-05 |
| Solution34 | 10.11 | 626.82 | 4.47731E-05 |
| Solution35 | 9.37 | 580.94 | 4.1496E-05 |
| Solution36 | 9.015 | 558.93 | 3.99238E-05 |
| Solution37 | 8.995 | 557.69 | 3.98352E-05 |
| Solution38 | 9.43 | 584.66 | 4.17617E-05 |
| Solution39 | 9.055 | 561.41 | 4.0101E-05 |

| Original Solution | |
|-------------------|--|
| 50 | Amount of Solution (mL) |
| 15.4 | Original concentration of $\text{NH}_4^+\text{-N}$ ($\mu\text{g mL}^{-1}$) |
| 770 | Original concentration of $\text{NH}_4^+\text{-N}$ (μg) |
| 5.50003E-05 | Moles of $\text{NH}_4^+\text{-N}$ |

| EQUATIONS |
|---|
| $x \mu\text{g} (1\text{g} / 1,000,000 \mu\text{g}) (1 \text{ mole} / 14 \text{ g } \text{NH}_4^+\text{-N}) = 7.1429 \text{ E}^{-8} \text{ moles } \text{NH}_4^+\text{-N} / \mu\text{g} (x \mu\text{g})$ 7.1429E-08 |
| $\text{NH}_4^+\text{-N (mole) calculated in tubes} = \text{NH}_4^+\text{-N (mole) original } \text{NH}_4^+\text{-N (mole) final solution}$ |
| $\% \text{ Efficiency of Tubes} = (\text{NH}_4^+\text{-N on tubes collected} / \text{NH}_4^+\text{-N on tubes calculated}) * 100$ |
| $\% \text{ Recovery of System} = ((\text{NH}_4^+\text{-N final} + \text{NH}_4^+\text{-N in tubes}) / \text{NH}_4^+\text{-N original}) * 100$ |

Table I - 2
Percent Efficiency of Oxalic Coated Tubes: Run B

| Sample (3 mL) | Corresponding Solution | Minutes | Tube ($\mu\text{g mL}^{-1}$) | Tube (μg) | $\text{NH}_4^+\text{-N}$ captured in tubes (mole) | $\text{NH}_4^+\text{-N}$ calculated in tubes (mole) | % Eff. of tubes (mole) | % Recovery of System |
|---------------|------------------------|---------|--------------------------------|------------------------|---|---|------------------------|----------------------|
| 71 | 41 | 60 | 6.35 | 19.05 | 1.361E-06 | 3.71E-06 | 61.283 | 97.05 |
| 72 | 41 | 60 | 4.25 | 12.75 | 9.107E-07 | | | |
| 73 | 42 | 60 | 6.5 | 19.5 | 1.393E-06 | 4.02E-06 | 69.678 | 97.50 |
| 74 | 42 | 60 | 6.56 | 19.68 | 1.406E-06 | | | |
| 75 | 44 | 90 | 9 | 27 | 1.929E-06 | 5.61E-06 | 70.847 | 96.64 |
| 76 | 44 | 90 | 9.55 | 28.65 | 2.046E-06 | | | |
| 77 | 45 | 90 | 9.12 | 27.36 | 1.954E-06 | 5.10E-06 | 74.685 | 97.35 |
| 78 | 45 | 90 | 8.66 | 25.98 | 1.856E-06 | | | |
| 79 | 46 | 120 | 10 | 30 | 2.143E-06 | 5.41E-06 | 77.416 | 97.49 |
| 81 | 46 | 120 | 9.55 | 28.65 | 2.046E-06 | | | |
| 82 | 47 | 120 | 11.64 | 34.92 | 2.494E-06 | 6.10E-06 | 80.438 | 97.55 |
| 83 | 47 | 150 | 11.25 | 33.75 | 2.411E-06 | | | |
| 84 | 48 | 150 | 13.67 | 41.01 | 2.929E-06 | 7.32E-06 | 78.793 | 96.81 |
| 85 | 48 | 150 | 13.23 | 39.69 | 2.835E-06 | | | |
| 86 | 49 | 150 | 14.84 | 44.52 | 3.18E-06 | 7.32E-06 | 84.212 | 97.63 |
| 87 | 49 | 150 | 13.91 | 41.73 | 2.981E-06 | | | |
| 88 | 50 | 180 | 17.42 | 52.26 | 3.733E-06 | 8.69E-06 | 85.235 | 97.36 |
| 89 | 50 | 180 | 17.14 | 51.42 | 3.673E-06 | | | |
| 90 | 51 | 210 | 16.71 | 50.13 | 3.581E-06 | 9.93E-06 | 79.014 | 95.72 |
| 91 | 51 | 210 | 19.9 | 59.7 | 4.264E-06 | | | |
| 92 | 52 | 180 | 16.32 | 48.96 | 3.497E-06 | 7.91E-06 | 85.676 | 97.67 |
| 93 | 52 | 180 | 15.32 | 45.96 | 3.283E-06 | | | |
| 94 | 53 | 210 | 18.62 | 55.86 | 3.99E-06 | 9.42E-06 | 84.356 | 96.97 |
| 95 | 53 | 210 | 18.46 | 55.38 | 3.956E-06 | | | |
| 96 | 54 | 270 | 20.52 | 61.56 | 4.397E-06 | 1.12E-05 | 63.456 | 91.57 |
| 97 | 54 | 270 | 12.75 | 38.25 | 2.732E-06 | | | |
| 98 | 56 | 330 | 30.02 | 90.06 | 6.433E-06 | 1.34E-05 | 88.858 | 96.92 |
| 99 | 56 | 330 | 25.75 | 77.25 | 5.518E-06 | | | |
| 100 | 57 | 270 | 24.84 | 74.52 | 5.323E-06 | 1.11E-05 | 89.582 | 97.63 |
| 101 | 57 | 270 | 21.48 | 64.44 | 4.603E-06 | | | |
| 102 | 58 | 330 | 0 | 0 | 0 | 1.36E-05 | 37.681 | 82.53 |
| 103 | 58 | 330 | 24 | 72 | 5.143E-06 | | | |

Table I - 2 cont.
Percent Efficiency of Oxalic Coated Tubes: Run B

| 62 mL final solution amount | Final Solution ($\mu\text{g mL}^{-1}$) | Final Solution (μg) | Final Solution (moles $\text{NH}_4^+\text{-N}$) |
|-----------------------------|--|----------------------------------|--|
| Solution41 | 10.155 | 629.61 | 4.49724E-05 |
| Solution42 | 10.085 | 625.27 | 4.46624E-05 |
| Solution44 | 9.725 | 602.95 | 4.30681E-05 |
| Solution45 | 9.840 | 610.08 | 4.35774E-05 |
| Solution46 | 9.770 | 605.74 | 4.32674E-05 |
| Solution47 | 9.615 | 596.13 | 4.2581E-05 |
| Solution48 | 9.340 | 579.08 | 4.13631E-05 |
| Solution49 | 9.340 | 579.08 | 4.13631E-05 |
| Solution50 | 9.030 | 559.86 | 3.99902E-05 |
| Solution51 | 8.750 | 542.50 | 3.87502E-05 |
| Solution52 | 9.205 | 570.71 | 4.07652E-05 |
| Solution53 | 8.865 | 549.63 | 3.92595E-05 |
| Solution54 | 8.455 | 524.21 | 3.74438E-05 |
| Solution56 | 7.955 | 493.21 | 3.52295E-05 |
| Solution57 | 8.490 | 526.38 | 3.75988E-05 |
| Solution58 | 7.910 | 490.42 | 3.50302E-05 |

| Original Solution | |
|-------------------|--|
| 50 | Amount of Solution (mL) |
| 13.63 | Original concentration of $\text{NH}_4^+\text{-N}$ ($\mu\text{g mL}^{-1}$) |
| 681.5 | Original concentration of $\text{NH}_4^+\text{-N}$ (μg) |
| 4.87E-05 | Moles of $\text{NH}_4^+\text{-N}$ |

| EQUATIONS |
|---|
| $x \mu\text{g} (1 \text{ g} / 1,000,000 \mu\text{g}) (1 \text{ mole} / 14 \text{ g } \text{NH}_4^+\text{-N}) = 7.1429 \text{ E}^{-8} \text{ moles } \text{NH}_4^+\text{-N} / \mu\text{g} (x \mu\text{g})$ 7.1429E-08 |
| $\text{NH}_4^+\text{-N (mole) calculated in tubes} = \text{NH}_4^+\text{-N (mole) original } \text{NH}_4^+\text{-N (mole) final solution}$ |
| % Efficiency of Tubes = $(\text{NH}_4^+\text{-N on tubes collected} / \text{NH}_4^+\text{-N on tubes calculated}) * 100$ |
| % Recovery of System = $((\text{NH}_4^+\text{-N final} + \text{NH}_4^+\text{-N in tubes}) / \text{NH}_4^+\text{-N original}) * 100$ |

APPENDIX II

Soil-Effluent Flask Experiments

Table II - 1
Richfield Clay Loam Mass Balances

| Run 6 | Soil beginning ($\mu\text{g N}$) | Soil end ($\mu\text{g N}$) | Effluent added ($\mu\text{g N}$) | Captured ($\mu\text{g NH}_4^+\text{-N}$) | Missing ($\mu\text{g N}$) | % Missing | % Recovery |
|-------|---------------------------------------|---------------------------------|---------------------------------------|---|--------------------------------|-----------|------------|
| 0 | 3250 | 5587.50 | 0.00 | 1.86 | -2339.36 | -71.98 | 171.98 |
| 100 | 3250 | 11032.50 | 8551.49 | 342.21 | 426.78 | 3.62 | 96.38 |
| 300 | 3250 | 22745.00 | 21517.75 | 537.39 | 1485.36 | 6.00 | 94.00 |
| 500 | 3250 | 29430.00 | 34542.11 | 512.82 | 7849.29 | 20.77 | 79.23 |
| 1000 | 3250 | 46059.50 | 66613.97 | 1104.51 | 22699.96 | 32.49 | 67.51 |

| Run 7 | | | | | | | |
|-------|------|----------|----------|---------|----------|--------|--------|
| 0 | 4335 | 5355.00 | 0.00 | 8.43 | -1028.43 | -23.72 | 123.72 |
| 100 | 4335 | 12890.00 | 7765.48 | 369.93 | -1159.45 | -9.58 | 109.58 |
| 300 | 4335 | 19830.00 | 20725.93 | 478.26 | 4752.67 | 18.96 | 81.04 |
| 500 | 4335 | 40120.00 | 36226.18 | 876.87 | -435.69 | -1.07 | 101.17 |
| 1000 | 4335 | 58260.00 | 70265.31 | 1218.42 | 15121.89 | 20.27 | 79.73 |

Table II - 2
Dennis Silt Loam Mass Balances

| Run 8 | Soil beginning ($\mu\text{g N}$) | Soil end ($\mu\text{g N}$) | Effluent added ($\mu\text{g N}$) | Captured ($\mu\text{g NH}_4^+\text{-N}$) | Missing ($\mu\text{g N}$) | % Missing | % Recovery |
|-------|---------------------------------------|---------------------------------|---------------------------------------|---|--------------------------------|-----------|------------|
| 0 | 6663 | 8868.75 | 0.00 | 0.21 | -2205.96 | -33.11 | 133.11 |
| 100 | 6663 | 14971.00 | 9284.38 | 29.73 | 946.65 | 5.94 | 94.07 |
| 300 | 6663 | 28776.00 | 22332.40 | 119.58 | 99.82 | 0.34 | 99.66 |
| 500 | 6663 | 40315.00 | 35113.57 | 147.45 | 1314.12 | 3.15 | 96.85 |
| 1000 | 6663 | 52981.50 | 67028.73 | 94.24 | 20615.99 | 27.98 | 72.02 |

| Run 9 | | | | | | | |
|-------|------|----------|----------|--------|----------|---------|--------|
| 0 | 7830 | 9746.00 | 0.00 | 0.36 | -1916.36 | -24.478 | 124.48 |
| 100 | 7830 | 16593.50 | 9076.88 | 114.69 | 198.69 | 1.18 | 98.83 |
| 300 | 7830 | 24343.00 | 20902.72 | 159.33 | 4230.39 | 14.72 | 85.23 |
| 500 | 7830 | 48364.25 | 33960.28 | 261.12 | -6835.09 | -16.36 | 116.36 |
| 1000 | 7830 | 58047.00 | 66200.8 | 180.12 | 15803.68 | 21.35 | 78.65 |

| Run 10 | | | | | | | |
|--------|------|----------|----------|--------|----------|-------|--------|
| 0 | 7420 | 7975.00 | 0.00 | 2.49 | -557.49 | -7.51 | 107.51 |
| 100 | 7420 | 16071.00 | 9357.42 | 84.93 | 621.49 | 3.70 | 96.30 |
| 300 | 7420 | 24557.50 | 21148.40 | 96.09 | 3914.81 | 13.70 | 86.30 |
| 500 | 7420 | 37130.50 | 36067.65 | 110.10 | 6247.05 | 14.37 | 85.64 |
| 1000 | 7420 | 46161.50 | 66780.14 | 239.76 | 27798.88 | 37.47 | 62.54 |

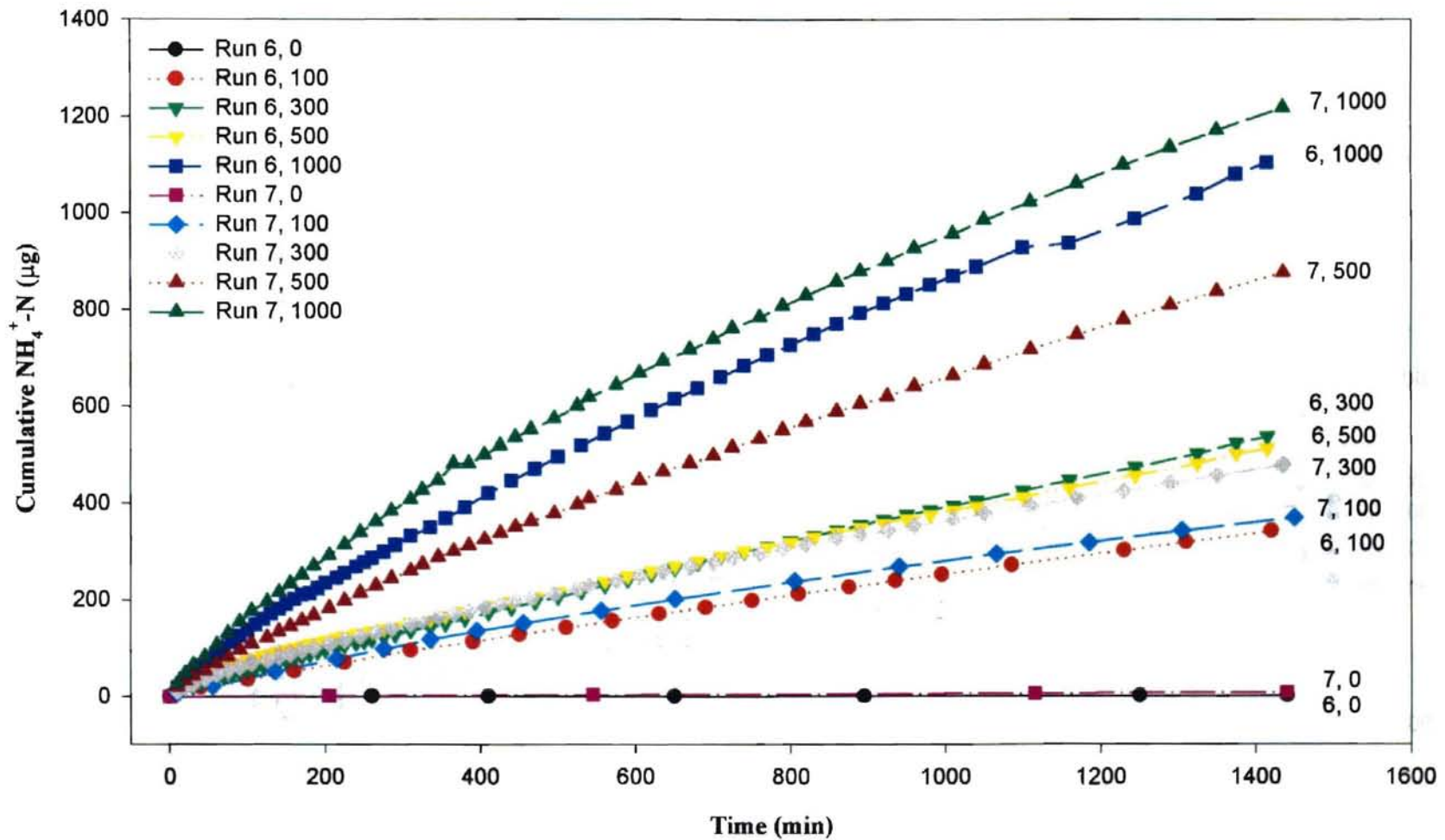


Figure II - 1. $\text{NH}_3\text{-N}$ volatilization results from application of five concentrations of swine effluent on two Richfield clay loam laboratory flask experiments

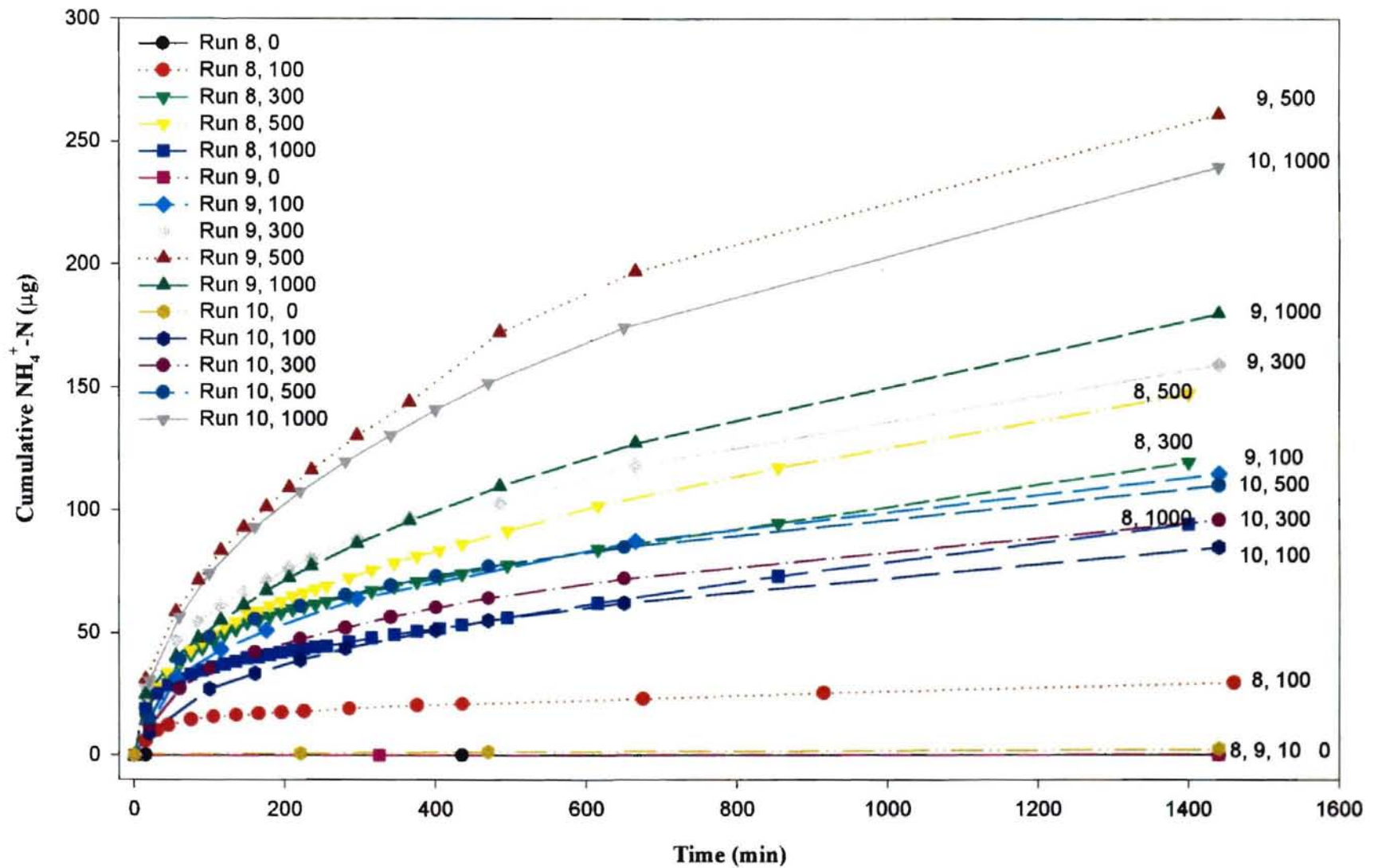


Figure II - 2. $\text{NH}_3\text{-N}$ volatilization results from application of five concentrations of swine effluent on three Dennis silt loam laboratory experiments. Note difference in scale from previous graph.

APPENDIX III
Wind Tunnel Studies

Table III - 1
Wind Tunnel Run 8

| A | B | C | D | E | F | G | H | I | J | K | L |
|-----------|---------------------------------|--------------------------------|--------|---|--|--|---------------------|--------------------|---|---|--|
| Quad-rant | Area of Tube (in ²) | Area of Tube (m ²) | Tube # | NH ₃ -N (μg mL ⁻¹) | NH ₃ -N Average in Background Samplers (μg mL ⁻¹) | NH ₃ -N after background subtraction (μg mL ⁻¹) | Water ellutant (mL) | Sample period (hr) | NH ₃ -N flux (μg m ⁻² s ⁻¹) | NH ₃ -N (mg m ⁻² hr ⁻¹) | NH ₃ -N through tube in 12 hours (mg) |
| 1 | 20.869 | 0.013 | 657 | 3.05 | 0.01625 | 3.034 | 3 | 12 | 174.27 | 627.38 | 101.36 |
| 2 | 8.859 | 0.006 | 650 | 1.81 | 0.01625 | 1.794 | 3 | 12 | 103.04 | 370.95 | 25.44 |
| 3 | 11.813 | 0.008 | 649 | 1.46 | 0.01625 | 1.444 | 3 | 12 | 82.94 | 298.57 | 27.30 |
| 4 | 11.813 | 0.008 | 525 | 1.15 | 0.01625 | 1.134 | 3 | 12 | 65.13 | 234.46 | 21.44 |
| 5 | 29.481 | 0.019 | 527 | 4.63 | 0.01625 | 4.614 | 3 | 12 | 265.03 | 954.12 | 217.77 |
| 6 | 12.516 | 0.008 | 653 | 2.06 | 0.01625 | 2.044 | 3 | 12 | 117.40 | 422.65 | 40.95 |
| 7 | 16.688 | 0.011 | 662 | 1.10 | 0.01625 | 1.084 | 3 | 12 | 62.26 | 224.12 | 28.95 |
| 8 | 16.688 | 0.011 | 659 | 0.12 | 0.01625 | 0.104 | 3 | 12 | 5.96 | 21.46 | 2.77 |
| 9 | 28.156 | 0.018 | 651 | 6.77 | 0.01625 | 6.754 | 3 | 12 | 387.96 | 1396.67 | 304.45 |
| 10 | 11.953 | 0.008 | 660 | 4.73 | 0.01625 | 4.714 | 3 | 12 | 270.78 | 974.80 | 90.21 |
| 11 | 15.938 | 0.010 | 661 | 3.39 | 0.01625 | 3.374 | 3 | 12 | 193.80 | 697.69 | 86.09 |
| 12 | 15.938 | 0.010 | 654 | 1.51 | 0.01625 | 1.494 | 3 | 12 | 85.81 | 308.91 | 38.12 |
| 13 | 28.819 | 0.019 | 658 | 8.53 | 0.01625 | 8.514 | 3 | 12 | 489.07 | 1760.64 | 392.82 |
| 14 | 12.234 | 0.008 | 655 | 6.80 | 0.01625 | 6.784 | 3 | 12 | 389.69 | 1402.88 | 132.88 |
| 15 | 16.313 | 0.011 | 516 | 5.58 | 0.01625 | 5.564 | 3 | 12 | 319.61 | 1150.58 | 145.31 |
| 16 | 16.313 | 0.011 | 514 | 2.90 | 0.01625 | 2.884 | 3 | 12 | 165.66 | 596.36 | 75.31 |
| 17 | 19.875 | 0.013 | 652 | 7.65 | 0.01625 | 7.634 | 3 | 12 | 438.52 | 1578.66 | 242.91 |
| 18 | 8.438 | 0.005 | 528 | 6.92 | 0.01625 | 6.904 | 3 | 12 | 396.58 | 1427.69 | 93.26 |
| 19 | 11.250 | 0.007 | 656 | 6.48 | 0.01625 | 6.464 | 3 | 12 | 371.31 | 1336.70 | 116.42 |
| 20 | 11.250 | 0.007 | 526 | 5.64 | 0.01625 | 5.624 | 3 | 12 | 323.05 | 1162.99 | 101.29 |

Total NH₃-N captured (mg) = 2285.06
 Beginning NH₄⁺-N concentration (mg) = 25433.44
 Ending NH₄⁺-N concentration (mg) = 8.07
 Percent recovery = 9.02

Table III - 2
Wind Tunnel Run 9

| A | B | C | D | E | F | G | H | I | J | K | L |
|-----------|---------------------------------|--------------------------------|--------|---|--|--|---------------------|--------------------|---|---|--|
| Quad-rant | Area of Tube (in ²) | Area of Tube (m ²) | Tube # | NH ₃ -N (µg mL ⁻¹) | NH ₃ -N Average in Background Samplers (µg mL ⁻¹) | NH ₃ -N after background subtraction (µg mL ⁻¹) | Water ellutant (mL) | Sample period (hr) | NH ₃ -N flux (µg m ⁻² s ⁻¹) | NH ₃ -N (mg m ⁻² hr ⁻¹) | NH ₃ -N through tube in 12 hours (mg) |
| 1 | 20.869 | 0.013 | 700 | 3.05 | 0.1125 | 2.938 | 3 | 12 | 168.74 | 607.47 | 98.15 |
| 2 | 8.859 | 0.006 | 701 | 1.81 | 0.1125 | 1.698 | 3 | 12 | 97.51 | 351.04 | 24.08 |
| 3 | 11.813 | 0.008 | 702 | 1.46 | 0.1125 | 1.348 | 3 | 12 | 77.41 | 278.66 | 25.48 |
| 4 | 11.813 | 0.008 | 703 | 1.15 | 0.1125 | 1.038 | 3 | 12 | 59.60 | 214.55 | 19.62 |
| 5 | 29.481 | 0.019 | 704 | 4.63 | 0.1125 | 4.518 | 3 | 12 | 259.50 | 934.22 | 213.23 |
| 6 | 12.516 | 0.008 | 705 | 2.06 | 0.1125 | 1.948 | 3 | 12 | 111.87 | 402.74 | 39.02 |
| 7 | 16.688 | 0.011 | 706 | 1.10 | 0.1125 | 0.988 | 3 | 12 | 56.73 | 204.21 | 26.38 |
| 8 | 16.688 | 0.011 | 707 | 0.12 | 0.1125 | 0.007 | 3 | 12 | 0.43 | 1.55 | 0.20 |
| 9 | 28.156 | 0.018 | 708 | 6.77 | 0.1125 | 6.658 | 3 | 12 | 382.44 | 1376.77 | 300.11 |
| 10 | 11.953 | 0.008 | 709 | 4.73 | 0.1125 | 4.618 | 3 | 12 | 265.25 | 954.90 | 88.37 |
| 11 | 15.938 | 0.010 | 710 | 3.39 | 0.1125 | 3.278 | 3 | 12 | 188.27 | 677.79 | 83.63 |
| 12 | 15.938 | 0.010 | 711 | 1.51 | 0.1125 | 1.398 | 3 | 12 | 80.28 | 289.00 | 35.66 |
| 13 | 28.819 | 0.019 | 712 | 8.53 | 0.1125 | 8.418 | 3 | 12 | 483.54 | 1740.74 | 388.38 |
| 14 | 12.234 | 0.008 | 713 | 6.80 | 0.1125 | 6.688 | 3 | 12 | 384.16 | 1382.97 | 130.99 |
| 15 | 16.313 | 0.011 | 714 | 5.58 | 0.1125 | 5.468 | 3 | 12 | 314.08 | 1130.68 | 142.79 |
| 16 | 16.313 | 0.011 | 715 | 2.90 | 0.1125 | 2.788 | 3 | 12 | 160.13 | 576.45 | 72.80 |
| 17 | 19.875 | 0.013 | 716 | 7.65 | 0.1125 | 7.538 | 3 | 12 | 432.99 | 1558.75 | 239.85 |
| 18 | 8.438 | 0.005 | 717 | 6.92 | 0.1125 | 6.808 | 3 | 12 | 391.05 | 1407.79 | 91.96 |
| 19 | 11.250 | 0.007 | 718 | 6.48 | 0.1125 | 6.368 | 3 | 12 | 365.78 | 1316.80 | 114.69 |
| 20 | 11.250 | 0.007 | 719 | 5.64 | 0.1125 | 5.528 | 3 | 12 | 317.52 | 1143.08 | 99.56 |

Total NH₃-N captured (mg) = 2234.95
Beginning NH₄⁺-N concentration (mg) = 19380.71
Ending NH₄⁺-N concentration (mg) = 15.33
Percent recovery = 11.61

Table III - 3
Wind Tunnel Run 14

| A | B | C | D | E | F | G | H | I | J | K | L |
|-----------|---------------------------------|--------------------------------|--------|---|--|--|---------------------|--------------------|---|---|--|
| Quad-rant | Area of Tube (in ²) | Area of Tube (m ²) | Tube # | NH ₃ -N (μg mL ⁻¹) | NH ₃ -N Average in Background Samplers (μg mL ⁻¹) | NH ₃ -N after background subtraction (μg mL ⁻¹) | Water ellutant (mL) | Sample period (hr) | NH ₃ -N flux (μg m ⁻² s ⁻¹) | NH ₃ -N (mg m ⁻² hr ⁻¹) | NH ₃ -N through tube in 12 hours (mg) |
| 1 | 10.336 | 0.007 | 1 | 0.76 | 0 | 0.760 | 3 | 12 | 43.66 | 157.17 | 12.58 |
| 2 | 10.336 | 0.007 | 2 | 0.74 | 0 | 0.740 | 3 | 12 | 42.51 | 153.03 | 12.25 |
| 3 | 8.859 | 0.006 | 3 | 0.47 | 0 | 0.470 | 3 | 12 | 27.00 | 97.20 | 6.67 |
| 4 | 8.859 | 0.006 | 4 | 0.47 | 0 | 0.470 | 3 | 12 | 27.00 | 97.20 | 6.67 |
| 5 | 14.602 | 0.009 | 5 | 1.69 | 0 | 1.690 | 3 | 12 | 97.08 | 349.49 | 39.51 |
| 6 | 14.602 | 0.009 | 6 | 1.34 | 0 | 1.340 | 3 | 12 | 76.98 | 277.11 | 31.33 |
| 7 | 12.516 | 0.008 | 7 | 0.93 | 0 | 0.930 | 3 | 12 | 53.42 | 192.32 | 18.64 |
| 8 | 12.516 | 0.008 | 8 | 0.39 | 0 | 0.390 | 3 | 12 | 22.40 | 80.65 | 7.81 |
| 9 | 13.945 | 0.009 | 9 | 1.35 | 0 | 1.350 | 3 | 12 | 77.55 | 279.18 | 30.14 |
| 10 | 13.945 | 0.009 | 10 | 1.35 | 0 | 1.350 | 3 | 12 | 77.55 | 279.18 | 30.14 |
| 11 | 11.953 | 0.008 | 11 | 0.68 | 0 | 0.680 | 3 | 12 | 39.06 | 140.62 | 13.01 |
| 12 | 11.953 | 0.008 | 12 | 0.86 | 0 | 0.860 | 3 | 12 | 49.40 | 177.85 | 16.46 |
| 13 | 14.273 | 0.009 | 13 | 0.96 | 0 | 0.960 | 3 | 12 | 55.15 | 198.53 | 21.94 |
| 14 | 14.273 | 0.009 | 14 | 0.89 | 0 | 0.890 | 3 | 12 | 51.13 | 184.05 | 20.34 |
| 15 | 12.234 | 0.008 | 15 | 1.30 | 0 | 1.300 | 3 | 12 | 74.68 | 268.84 | 25.46 |
| 16 | 12.234 | 0.008 | 16 | 0.72 | 0 | 0.720 | 3 | 12 | 41.36 | 148.90 | 14.10 |
| 17 | 9.844 | 0.006 | 17 | 0.85 | 0 | 0.850 | 3 | 12 | 48.83 | 175.78 | 13.40 |
| 18 | 9.844 | 0.006 | 18 | 1.35 | 0 | 1.350 | 3 | 12 | 77.55 | 279.18 | 21.28 |
| 19 | 8.438 | 0.005 | 19 | 0.79 | 0 | 0.790 | 3 | 12 | 45.38 | 163.37 | 10.67 |
| 20 | 8.438 | 0.005 | 20 | 0.33 | 0 | 0.330 | 3 | 12 | 18.96 | 68.24 | 4.46 |

Total NH₃-N captured (mg) = 356.84
 Beginning NH₄⁺-N concentration (mg) = 1007.00
 Ending NH₄⁺-N concentration (mg) = 7.765
 Percent recovery = 36.21

Table III - 4

| Equations for wind tunnel tables | |
|---|--|
| Col B | $A = l \times w$ |
| Col C | $A \text{ (m}^2\text{)} = (\text{in}^2 \times 2.54\text{cm}^2) / 100\text{cm}^2$ |
| Col G | $\mu\text{g mL}^{-1}$ after background subtraction = $\text{IF}(\text{E3-}\$F\$3 < 0, 0, \text{IF}(\text{E3-}\$F\$3 > 0, (\text{E3-}\$F\$3)))$ |
| Col J | $\text{NH}_3\text{-N flux } ((\mu\text{g/mL}) / \text{m}^2 \text{ s}) = (\mu\text{g/mL}) (3\text{mL}) / 2 \times 3.14 \times (0.0005\text{m})^2 \times 0.77 \times \text{hrs} \times 3600\text{s}$ |
| Col K | $\text{NH}_3\text{-N flux (mg / m}^2 \text{ hr)} = ((\mu\text{g/mL}) / \text{m}^2 \text{ s}) / 1000) \times 3600\text{s}$ |
| Col L | $\text{NH}_3\text{-N (mg)} = (\text{mg} / \text{m}^2 \text{ hr}) \times \text{m}^2 \times \text{hrs}$ |
| Percent Recovery of system = $((\text{NH}_4^+\text{-N final} + \text{NH}_3 \text{ captured}) / \text{NH}_4^+\text{-N initial}) * 100$ | |

APPENDIX IV
Field Measurements

Table IV - 1
Panhandle Field Run 2
Soil Mass Balance

| TRT | Plot 1 | Plot 2 | Plot 3 |
|---|---------------|---------------|---------------|
| Soil beginning | | | |
| $\mu\text{g NH}_4^+\text{-N g}^{-1}$ | 3.13 | 4.18 | 3.14 |
| $\mu\text{g NO}_3^-\text{-N g}^{-1}$ | 18.94 | 15.08 | 18.36 |
| $\mu\text{g g}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 22.07 | 19.26 | 21.5 |
| g soil | 36955254.61 | 36955254.61 | 36955254.61 |
| Total $\mu\text{g N}$ | 815602469.2 | 711758203.8 | 794537974.1 |
| Soil end | | | |
| $\mu\text{g NH}_4^+\text{-N g}^{-1}$ | 22.63 | 13.71 | 53.23 |
| $\mu\text{g NO}_3^-\text{-N g}^{-1}$ | 35.58 | 29.4 | 36.75 |
| $\mu\text{g g}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 58.21 | 43.11 | 89.98 |
| g soil | 36955254.61 | 36955254.61 | 36955254.61 |
| Total $\mu\text{g N}$ | 2151165371 | 1593141026 | 3325233810 |
| Effluent | | | |
| $\mu\text{g NO}_3^-\text{-N mL}^{-1}$ | 1 | 1 | 1 |
| $\mu\text{g NH}_4^+\text{-N mL}^{-1}$ | 1081 | 1052 | 1078 |
| $\mu\text{g mL}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 1082 | 1053 | 1079 |
| mL effluent | 4625283.88 | 4625283.88 | 4625283.88 |
| Total $\mu\text{g N added}$ | 5004557153.83 | 4870423921.43 | 4990681302.20 |
| Captured on tubes ($\mu\text{g NH}_4^+\text{-N}$) | 2812684826 | 2812684826 | 2812684826 |
| MISSING (μg) | 856309426.2 | 1176356273 | -352699359.5 |
| % Missing | 15 | 21 | -6 |
| % Recovery | 85 | 79 | 106 |

Table IV - 2
Panhandle Field Run 2
Soil Mass Balance with + 10% change of soil weight

| TRT | Plot 1 | Plot 2 | Plot 3 |
|---|-------------|-------------|-------------|
| Soil beginning | | | |
| $\mu\text{g NH}_4^+\text{-N g}^{-1}$ | 3.13 | 4.18 | 3.14 |
| $\mu\text{g NO}_3^-\text{-N g}^{-1}$ | 18.94 | 15.08 | 18.36 |
| $\mu\text{g g}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 22.07 | 19.26 | 21.5 |
| g soil | 40650780.07 | 40650780.07 | 40650780.07 |
| Total $\mu\text{g N}$ | 897162716.1 | 782934024.1 | 873991771.5 |
| Soil end | | | |
| $\mu\text{g NH}_4^+\text{-N g}^{-1}$ | 22.63 | 13.71 | 53.23 |
| $\mu\text{g NO}_3^-\text{-N g}^{-1}$ | 35.58 | 29.4 | 36.75 |
| $\mu\text{g g}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 58.21 | 43.11 | 89.98 |
| g soil | 40650780.07 | 40650780.07 | 40650780.07 |
| Total $\mu\text{g N}$ | 2366281908 | 1752455129 | 3657757191 |
| Effluent | | | |
| $\mu\text{g NO}_3^-\text{-N mL}^{-1}$ | 1 | 1 | 1 |
| $\mu\text{g NH}_4^+\text{-N mL}^{-1}$ | 1081 | 1052 | 1078 |
| $\mu\text{g mL}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 1082 | 1053 | 1079 |
| mL effluent | 4625283.876 | 4625283.876 | 4625283.876 |
| Total $\mu\text{g N added}$ | 5004557154 | 4870423921 | 4990681302 |
| Captured on tubes ($\mu\text{g NH}_4^+\text{-N}$) | | | |
| | 2812684826 | 2812684826 | 2812684826 |
| MISSING (μg) | | | |
| | 722753136.1 | 1088217991 | -605768943 |
| % Missing | 12 | 19 | -10 |
| % Recovery | 88 | 81 | 110 |

* changed by + 10%

** + or - 2% change when g soil changes by 10%, so bulk density assumption should not matter

Table IV - 3
Panhandle Field Run 2
Soil Mass Balance with + 10% change in effluent volume

| TRT | Plot 1 | Plot 2 | Plot 3 |
|---|-------------|-------------|-------------|
| Soil beginning | | | |
| $\mu\text{g NH}_4^+\text{-N g}^{-1}$ | 3.13 | 4.18 | 3.14 |
| $\mu\text{g NO}_3^-\text{-N g}^{-1}$ | 18.94 | 15.08 | 18.36 |
| $\mu\text{g g}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 22.07 | 19.26 | 21.5 |
| g soil | 36955254.61 | 36955254.61 | 36955254.61 |
| Total $\mu\text{g N}$ | 815602469.2 | 711758203.8 | 794537974.1 |
| Soil end | | | |
| $\mu\text{g NH}_4^+\text{-N g}^{-1}$ | 22.63 | 13.71 | 53.23 |
| $\mu\text{g NO}_3^-\text{-N g}^{-1}$ | 35.58 | 29.4 | 36.75 |
| $\mu\text{g g}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 58.21 | 43.11 | 89.98 |
| g soil | 36955254.61 | 36955254.61 | 36955254.61 |
| Total $\mu\text{g N}$ | 2151165371 | 1593141026 | 3325233810 |
| Effluent | | | |
| $\mu\text{g NO}_3^-\text{-N mL}^{-1}$ | 1 | 1 | 1 |
| $\mu\text{g NH}_4^+\text{-N mL}^{-1}$ | 1081 | 1052 | 1078 |
| $\mu\text{g mL}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 1082 | 1053 | 1079 |
| mL effluent | 5087811 | 5087811 | 5087811 |
| Total $\mu\text{g N added}$ | 5505011502 | 5357464983 | 5489748069 |
| Captured on tubes ($\mu\text{g NH}_4^+\text{-N}$) | | | |
| | 2812684826 | 2812684826 | 2812684826 |
| MISSING (μg) | | | |
| | 1356763774 | 1663397335 | 146367407.3 |
| % Missing | 21 | 27 | 2 |
| % Recovery | 79 | 73 | 98 |

* changed by + 10%

** + or - 8% change when volume of effluent changes by 10%, so proper effluent amounts are important

Table IV - 4
Panhandle Field Run 2
Soil Mass Balance with + 10% change in effluent concentration

| TRT | Plot 1 | Plot 2 | Plot 3 |
|---|-------------|-------------|-------------|
| Soil beginning | | | |
| $\mu\text{g NH}_4^+\text{-N g}^{-1}$ | 3.13 | 4.18 | 3.14 |
| $\mu\text{g NO}_3^-\text{-N g}^{-1}$ | 18.94 | 15.08 | 18.36 |
| $\mu\text{g g}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 22.07 | 19.26 | 21.5 |
| g soil | 36955254.61 | 36955254.61 | 36955254.61 |
| Total $\mu\text{g N}$ | 815602469.2 | 711758203.8 | 794537974.1 |
| Soil end | | | |
| $\mu\text{g NH}_4^+\text{-N g}^{-1}$ | 22.63 | 13.71 | 53.23 |
| $\mu\text{g NO}_3^-\text{-N g}^{-1}$ | 35.58 | 29.4 | 36.75 |
| $\mu\text{g g}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 58.21 | 43.11 | 89.98 |
| g soil | 36955254.61 | 36955254.61 | 36955254.61 |
| Total $\mu\text{g N}$ | 2151165371 | 1593141026 | 3325233810 |
| Effluent | | | |
| $\mu\text{g NO}_3^-\text{-N mL}^{-1}$ | 1 | 1 | 1 |
| $\mu\text{g NH}_4^+\text{-N mL}^{-1}$ | 1181 | 1152 | 1178 |
| $\mu\text{g mL}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 1182 | 1153 | 1179 |
| mL effluent | 4625283.876 | 4625283.876 | 4625283.876 |
| Total $\mu\text{g N added}$ | 5467085541 | 5332952309 | 5453209690 |
| Captured on tubes ($\mu\text{g NH}_4^+\text{-N}$) | 2812684826 | 2812684826 | 2812684826 |
| MISSING (μg) | 1318837814 | 1638884661 | 109829028.1 |
| % Missing | 21 | 27 | 2 |
| % Recovery | 79 | 73 | 98 |

* changed by +10%

** + or - 6% change when conc. of effluent changes by 10%, so proper effluent testing is important

Table IV - 5
Panhandle Field Run 2
Soil Mass Balance with + 10% change in tube collection values

| TRT | Plot 1 | Plot 2 | Plot 3 |
|---|-------------|-------------|--------------|
| Soil beginning | | | |
| $\mu\text{g NH}_4^+\text{-N g}^{-1}$ | 3.13 | 4.18 | 3.14 |
| $\mu\text{g NO}_3^-\text{-N g}^{-1}$ | 18.94 | 15.08 | 18.36 |
| $\mu\text{g g}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 22.07 | 19.26 | 21.5 |
| g soil | 36955254.61 | 36955254.61 | 36955254.61 |
| Total $\mu\text{g N}$ | 815602469.2 | 711758203.8 | 794537974.1 |
| Soil end | | | |
| $\mu\text{g NH}_4^+\text{-N g}^{-1}$ | 22.63 | 13.71 | 53.23 |
| $\mu\text{g NO}_3^-\text{-N g}^{-1}$ | 35.58 | 29.4 | 36.75 |
| $\mu\text{g g}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 58.21 | 43.11 | 89.98 |
| g soil | 36955254.61 | 36955254.61 | 36955254.61 |
| Total $\mu\text{g N}$ | 2151165371 | 1593141026 | 3325233810 |
| Effluent | | | |
| $\mu\text{g NO}_3^-\text{-N mL}^{-1}$ | 1 | 1 | 1 |
| $\mu\text{g NH}_4^+\text{-N mL}^{-1}$ | 1082 | 1052 | 1078 |
| $\mu\text{g mL}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 1083 | 1053 | 1079 |
| mL effluent | 4625283.876 | 4625283.876 | 4625283.876 |
| Total $\mu\text{g N added}$ | 5009182438 | 4870423921 | 4990681302 |
| Captured on tubes ($\mu\text{g NH}_4^+\text{-N}$) | 3093953309 | 3093953309 | 3093953309 |
| MISSING (μg) | 579666227.1 | 895087790 | -633967842.5 |
| % Missing | 10 | 16 | -11 |
| % Recovery | 90 | 84 | 111 |

* changed by + 10%

** + or - 5% change when conc. of effluent changes by 10%, tubes could easily be off that much

Table IV - 6
Panhandle Field Run 2
Soil Mass Balance with + 10% change in all 4 parameters

| TRT | Plot 1 | Plot 2 | Plot 3 |
|---|-------------|-------------|-------------|
| Soil beginning | | | |
| $\mu\text{g NH}_4^+\text{-N g}^{-1}$ | 3.13 | 4.18 | 3.14 |
| $\mu\text{g NO}_3^-\text{-N g}^{-1}$ | 18.94 | 15.08 | 18.36 |
| $\mu\text{g g}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 22.07 | 19.26 | 21.5 |
| g soil | 40650780.07 | 40650780.07 | 40650780.07 |
| Total $\mu\text{g N}$ | 897162716.1 | 782934024.1 | 873991771.5 |
| Soil end | | | |
| $\mu\text{g NH}_4^+\text{-N g}^{-1}$ | 22.63 | 13.71 | 53.23 |
| $\mu\text{g NO}_3^-\text{-N g}^{-1}$ | 35.58 | 29.4 | 36.75 |
| $\mu\text{g g}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 58.21 | 43.11 | 89.98 |
| g soil | 40650780.07 | 40650780.07 | 40650780.07 |
| Total $\mu\text{g N}$ | 2366281908 | 1752455129 | 3657757191 |
| Effluent | | | |
| $\mu\text{g NO}_3^-\text{-N mL}^{-1}$ | 1 | 1 | 1 |
| $\mu\text{g NH}_4^+\text{-N mL}^{-1}$ | 1181 | 1152 | 1178 |
| $\mu\text{g mL}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 1182 | 1153 | 1179 |
| mL effluent | 5087811 | 5087811 | 5087811 |
| Total $\mu\text{g N added}$ | 6013792602 | 5866246083 | 5998529169 |
| Captured on tubes ($\mu\text{g NH}_4^+\text{-N}$) | 3093953309 | 3093953309 | 3093953309 |
| MISSING (μg) | 1450720101 | 1802771669 | 120810440.8 |
| % Missing | 21 | 27 | 2 |
| % Recovery | 79 | 73 | 98 |

* changed by + 10%

** + or - 6% change when all 4 parameters are changed by 10%

Table IV - 7
Panhandle Field Run 2
Soil Mass Balance Equations

| |
|---|
| <p>diameter of plot = 15.24m Sampling depth = 1/2 ft = 15.24 cm Assume bulk density of 1.33 g/cm³ Surface area of plot = 3.14 X r² Volume of plot = 3.14 X r² X sampling depth g of soil = vol of plot X bulk density Soil $\mu\text{g N} = \text{total } \mu\text{g N g}^{-1} \times 36955254.61 \text{ g soil}$ mL effluent = 1222 gal / 0.2642gal X 1000 mL (1L = 0.2642 gal) Effluent $\mu\text{g N} = \text{total } \mu\text{g N mL}^{-1} \times 4625283.876 \text{ mL effluent}$ $\mu\text{g captured on tubes from "transformed" equation spreadsheet total of}$ column $\text{g/m}^2 \times \text{SA of plot} \times 1 \times 10^6 \mu\text{g}$ Missing = ($\mu\text{g N}$ in beginning soil + $\mu\text{g N}$ in effluent) - ($\mu\text{g N}$ in ending soil + $\mu\text{g NH}_4^+ \text{-N}$ captured) % Missing = missing value/($\mu\text{g N}$ beginning soil + $\mu\text{g N}$ in effluent) % Recovery = 100 - % missing</p> |
|---|

Table IV - 8
Panhandle Field Run 3
Soil Mass Balance

| TRT | Plot 2 | Plot 3 | Plot 4 |
|---|-------------|--------------|--------------|
| Soil beginning | | | |
| $\mu\text{g NH}_4^+\text{-N g}^{-1}$ | 3.33 | 3.33 | 3.33 |
| $\mu\text{g NO}_3^-\text{-N g}^{-1}$ | 28.95 | 28.95 | 28.95 |
| $\mu\text{g g}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 32.28 | 32.28 | 32.28 |
| g soil | 36955254.61 | 36955254.61 | 36955254.61 |
| Total $\mu\text{g N}$ | 1192915619 | 1192915619 | 1192915619 |
| Soil end | | | |
| $\mu\text{g NH}_4^+\text{-N g}^{-1}$ | 5.67 | 25.13 | 46.25 |
| $\mu\text{g NO}_3^-\text{-N g}^{-1}$ | 71.51 | 80.47 | 78.15 |
| $\mu\text{g g}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 77.18 | 105.6 | 124.4 |
| g soil | 36955254.61 | 36955254.61 | 36955254.61 |
| Total $\mu\text{g N}$ | 2852206551 | 3902474887 | 4597233673 |
| Effluent | | | |
| $\mu\text{g NO}_3^-\text{-N mL}^{-1}$ | 1 | 1 | 1 |
| $\mu\text{g NH}_4^+\text{-N mL}^{-1}$ | 868 | 873 | 888 |
| $\mu\text{g mL}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 869 | 874 | 889 |
| mL effluent | 4625283.876 | 4625283.876 | 4625283.876 |
| Total $\mu\text{g N}$ added | 4019371688 | 4042498108 | 4111877366 |
| Captured on tubes ($\mu\text{g NH}_4^+\text{-N}$) | 1506145520 | 1506145520 | 1506145520 |
| MISSING (μg) | 853935236.3 | -173206680.4 | -798586208.9 |
| % Missing | 16 | -3 | -15 |
| % Recovery | 84 | 103 | 115 |

Table IV - 9
Panhandle Field Run 4
Soil Mass Balance

| TRT | Plot 2 | Plot 3 | Plot 4 |
|---|-------------|--------------|-------------|
| Soil beginning | | | |
| $\mu\text{g NH}_4^+\text{-N g}^{-1}$ | 1.98 | 1.65 | 2.36 |
| $\mu\text{g NO}_3^-\text{-N g}^{-1}$ | 24.47 | 25.87 | 17.06 |
| $\mu\text{g g}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 26.45 | 27.52 | 19.42 |
| g soil | 36955254.61 | 36955254.61 | 36955254.61 |
| Total $\mu\text{g N}$ | 977466484.4 | 1017008607 | 717671044.5 |
| Soil end | | | |
| $\mu\text{g NH}_4^+\text{-N g}^{-1}$ | 82.23 | 38.8 | 47.05 |
| $\mu\text{g NO}_3^-\text{-N g}^{-1}$ | 53.11 | 54.36 | 45.09 |
| $\mu\text{g g}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 135.34 | 93.16 | 92.14 |
| g soil | 36955254.61 | 36955254.61 | 36955254.61 |
| Total $\mu\text{g N}$ | 5001524159 | 3442751519 | 3405057160 |
| Effluent | | | |
| $\mu\text{g NO}_3^-\text{-N mL}^{-1}$ | 1 | 1 | 1 |
| $\mu\text{g NH}_4^+\text{-N mL}^{-1}$ | 938 | 894 | 957 |
| $\mu\text{g mL}^{-1}$ total $\text{NO}_3^-\text{-N}$ & $\text{NH}_4^+\text{-N}$ | 939 | 895 | 958 |
| mL effluent | 4625283.876 | 4625283.876 | 4625283.876 |
| Total $\mu\text{g N added}$ | 4343141560 | 4139629069 | 4431021953 |
| Captured on tubes ($\mu\text{g NH}_4^+\text{-N}$) | 2131503120 | 2131503120 | 2131503120 |
| MISSING (μg) | -1812419235 | -417616963.6 | -387867282 |
| % Missing | -34 | -8 | -8 |
| % Recovery | 134 | 108 | 108 |

VITA 2

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