ABSTRACT

SEMUNEGUS, HILAWE. Comparative Assessment of Ammonia Emissions from Potential Environmentally Superior Technologies for Swine Facilities (Under the direction of Dr. S. Pal Arya and Dr. Viney P. Aneja)

Globally, domestic animal waste contributes nearly half of the total tropospheric ammonia (NH_3) emitted into the atmosphere. In North Carolina, swine waste from agricultural operations represents 47% of the total ammonia emissions and 21% of the state's nitrogen budget. Due to the short residence time of NH_3 (1-5 days) in the atmosphere, deposition occurs close to emission sources. Ammonia that does not deposit quickly combines with acidic species such as sulfuric acid, nitric acid, and hydrochloric acid to form ammonium aerosols. Ammonium aerosols travel farther from emission sources due to their lower deposition velocity leading to longer residence time (1-15 days) compared to gaseous ammonia. Detrimental effects of high concentrations and dry and wet depositions of reduced N species include harmful algal blooms, nitrogen enrichment, and eutrophication in aquatic ecosystems, soil acidification, and respiratory damage due to exposure of fine particulate matter (PM 2.5) formation, nitrogen saturation of forest soil, odor emanation, and visibility degradation.

In response to perceived adverse environmental impacts of the nearly 10 million hogs residing in North Carolina, alternative waste treatment methods were evaluated that could potentially reduce substantial emissions of ammonia. These are being evaluated by the OPEN Project Team. This study was a part of that evaluation, especially focusing on the ammonia emissions from two potential environmentally superior technologies (ESTs). Between April 2002 and February 2003, continuous NH_3 flux measurements were conducted over 2-week periods from two potential ESTs during two different (warm and cold) seasons and compared

with two conventional Lagoon and Spray Technology (LST) facilities, which were also sampled during different seasons. The two alternative technologies are (1) The Covered In-Ground Ambient Digester Technology and (2) The Constructed Wetlands/Solid Separator Technology. Using the dynamic flow-through chamber system, ammonia flux measurements were determined, along with measurements of meteorological parameters such as wind speed and direction, air temperature, relative humidity, and solar radiation from a 10 m tower at all the EST and LST sites.

For LST sites, the average ammonia fluxes for the storage ponds were 2,385 μg NH_3-N/m^2/min for Stokes Farm and 1,657 μg NH_3-N/m^2/min for Moore Farm during the fall season measurement campaigns. For winter season measurement campaigns, average ammonia flux measurements were 153 and 325 μg NH_3-N/m^2/min for the storage ponds at Stokes and Moore farms, respectively. For the April and November 2002 measurement campaigns at Barham Farm (EST), respectively, the average ammonia fluxes from the liquid waste storage ponds were 1,169 μg NH_3-N/m^2/min and 352 μg NH_3-N/m^2/min. At Howard Farm (EST), average measured ammonia fluxes were 790 μg NH_3-N/m^2/min and 243 μg NH_3-N/m^2/min during the June and December 2002 field campaigns.

Using the mass balance method, NH_3-N emissions from conventional LST or baseline sites were compared with those from ESTs. Ammonia emissions from different components such as liquid waste storage ponds, hog barns, and spray fields were also evaluated. To predict baseline NH_3-N emissions during the EST measurement periods, lagoon temperature (T_L) and air temperature – lagoon temperature difference (D) were inserted into an observational statistical model, developed from all baseline measurements. The observational model equation for average lagoon emissions at baseline sites is as follows,

Log_{10}(Area*flux/ton) = 3.8655 + 0.0449*T_L – 0.05946*D where T_L and D are in ^oC and Area*flux/ton represents the NH_3-N emissions from the lagoon per 1000 kg of live animal weight (LAW), expressed in units of μg NH_3-N/1000 kg-LAW/min. Emissions of NH_3-N normalized by the nitrogen excretion rate show that the Constructed Wetlands/Solid Separator Technology was not effective at all in reducing nitrogen emissions compared to the conventional Lagoon and Spray Technology. The Covered In-Ground Ambient Digester Technology was somewhat effective in reducing NH_3-N emissions compared to conventional Lagoon and Spray Technology, especially when the former attained steady-state.

COMPARATIVE ASSESSMENT OF AMMONIA EMISSIONS FROM POTENTIAL ENVIRONMENTALLY SUPERIOR TECHNOLOGIES FOR SWINE FACILITIES

by

HILAWE SEMUNEGUS

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APPROVED BY:

S. Pal Arya Chair of Advisory Committee	Viney P. Aneja Co-Chair of Advisory Committee
Sethu Raman	John J. Bates

BIOGRAPHY

Hilawe Semunegus was born on October 22, 1977, in Addis Ababa, Ethiopia to Dr. Semunegus Hailemariam and Tsega Berecket. Upon leaving Ethiopia with his family as a political asylee at the age of seven, Hilawe attended four different elementary schools and two different middle schools while his family moved around in Raleigh, NC. During the eighth grade, Hilawe lived with his parents for a year in Hanoi, Vietnam and during his sophomore year in high school, he moved to Bangladesh for three years with his parents and attended the American International School at Dhaka.

Upon graduating from high school in 1995, Hilawe entered North Carolina State University and graduated in May 2001, with a B.S. in Environmental Sciences with an Air Quality Concentration. While pursuing a B.S. at North Carolina State University, Hilawe was Student Patrol Director for the NCSU Campus Police for four years where he supervised up to 20 employees and was responsible for providing safety escorts for faculty and students of the university.

In the fall of 2001, Hilawe entered the Graduate School at North Carolina State University to study Atmospheric Sciences. While attending NCSU, Hilawe worked as a teaching assistant, meteorology lab instructor, and research assistant. In August of 2002, Hilawe was awarded a NOAA Education Partnership Program Scholarship as a Graduate Scientist, which provided training for permanent employment with the National Oceanic and Atmospheric Administration/National Climatic Data Center (NOAA/NCDC) in Asheville, NC, upon his graduation with a M.S. in Atmospheric Sciences. Between March 30 and April 2, 2003, Hilawe attended the 4th NOAA Expanding Opportunities Conference hosted by

Florida A&M University in Tallahassee, Florida, where he placed first overall, among 80 participants from several universities, in the graduate student poster presentation competition.

Presently, Hilawe is preparing to start his employment with the National Climatic Data Center in Asheville as a Physical Scientist. He will be working on satellite data, collected as part of the International Satellite Cloud Climatology Project (ISCCP), to study the effects of clouds on the radiation balance with emphasis on further understanding the hydrological cycle.

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Finally, I want to thank my parents, Dr. Semunegus Hailemariam and Tsega Berecket, and my sisters, Aida, Lulit, and Zema, which without their never-ending love, support, and prayers, I would not be where I am today. I am also thankful to my two little nieces, Leyu and Lela Wondafrash, for allowing me to realize what is really important in life.

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1.0 BACKGROUND AND INTRODUCTION

Atmospheric ammonia (NH_3) has a large influence on several environmental processes and therefore has several distinct fates within the atmosphere and in terrestrial and aquatic ecosystems. NH_3 reacts with acidic atmospheric species, such as sulfuric acid (H_2SO_4), nitric acid (HNO_3), and hydrochloric acid (HCl), to form ammonium aerosols, namely ammonium bisulfate (NH_4HSO_4), ammonium sulfate ($(NH_4)_2SO_4$), ammonium nitrate (NH_4NO_3), and ammonium chloride (NH_4Cl). Approximately 10% of atmospheric NH_3 also reacts with hydroxyl radicals ($^{\cdot}OH$) to form amide radicals ($^{\cdot}NH_2$) (Finlayson Pitts and Pitts 2000; McCulloch et al., 1998). These reactions can be summarized as follows:

$$NH_3(g) + H_2SO_4(l) \rightarrow NH_4HSO_4(l) \quad (1)$$

$$NH_3(g) + HNO_3(g) \leftrightarrow NH_4NO_3(s) \quad (2)$$

$$NH_3(g) + HCl(g) \leftrightarrow NH_4Cl(s) \quad (3)$$

$$NH_3(g) + H_2O(l) \rightarrow NH_4^+ + OH^- \quad (4)$$

$$NH_3 + {}^{\cdot}OH \rightarrow {}^{\cdot}NH_2 + H_2O(l) \quad (5)$$

(Finlayson Pitts and Pitts, 2000).

NH_3 undergoes wet and dry deposition to the surface, depositing close to its emission sources due to its relatively short residence time in the atmosphere of 1-5 days (Warneck 2000). This residence time increases when NH_3 transforms into the ammonium ion (NH_4^+) in aerosol form. Particulate NH_4^+ remains in the atmosphere for a longer period of time (1-15 days) due to a decrease in dry deposition velocity (Aneja et al., 1998). Therefore, it travels and deposits farther from its source and can affect ecosystems over larger spatial scales.

Some of the detrimental effects of high concentrations and depositions of reduced N species such as NH_3 include (Erisman et al., 1998):

1) Respiratory disease caused by exposure to high concentrations of photochemical and fine particulate matter
2) Nitrate contamination of drinking water
3) Eutrophication, harmful algal blooms and decreased surface water quality
4) Nitrogen saturation of forest soil
5) Soil acidification

Atmospheric deposition of nitrogen compounds contributes approximately 35-60% of nitrogen loading in coastal waters (Whitall et al., 2003; Pearl, 1995). An excess of nitrogen in coastal waters gives rise to environmental hazards such as toxic and non-toxic phytoplankton blooms, fish kills, and general declines in fisheries (Paerl 1995). In North Carolina, nutrient loading has influenced coastal river systems, such as the Neuse River Basin, for several years (Aneja et al., 1998). Elevated nitrate (NO_3^-) concentrations have been also observed in the North Carolina Coastal Plain Region (Showers, 1997).

Domestic animal waste produces the largest fraction of global atmospheric NH_3 emissions, contributing 20-35 Tg of nitrogen per year (Bouwman et al. 1997; Warneck 2000) (Figure 1.1). Swine waste produces the largest amount of NH_3 emissions in North Carolina, releasing an estimated 68,540 tons of nitrogen per year (Aneja et al., 1998). This amount represents 47% of the total NH_3 emissions for the state and accounts for approximately 21% of North Carolina's nitrogen budget (Aneja et al., 2001) (Figure 1.2). Soils, fertilizers, and biomass burning also release NH_3 into the atmosphere.

The statewide hog population has increased to almost 10 million in the last several decades (NCDA 1999) (Figure 1.3). In response to the extensive growth of the hog industry over the last decade in North Carolina, legislation was recently passed that extended the

previously imposed moratorium on the construction of new swine facilities and the expansion of existing swine facilities until September 2007. The growth of North Carolina's hog industry has largely occurred in the Coastal Plain Region, where more than 8.5 million hogs currently reside (Aneja et al., 2000) (Figure 1.4).

Under certain meteorological conditions, NH_3 emissions from the Coastal Plain Region have enhanced the wet deposition of NH_4^+ and NH_3 at National Atmospheric Deposition Program/National Trend Network (NADP/NTN) sites that lie up to 80 kilometers away from NH_3 sources (Walker et al., 2000). Many sensitive ecosystems lie within 80 kilometers of NH_3 area sources in North Carolina. Ecosystems in proximity to high NH_3 emission sources and NH_3/NH_4^+ deposition are subject to potential environmental consequences, including aquatic eutrophication and soil acidification. Quantification of NH_3 emission sources and development of emission factors are necessary in order to assess the potential extent of such environmental effects. Such a task requires extensive field measurements at confined animal feeding operations (CAFOs), i.e. swine waste agricultural operations.

There are several factors that influence NH_3 emissions from CAFOs (Griffing et al., 2003; Westerman et al., 2000):

- Nitrogen content of the feed
- Conversion factor between N in animal food and N in the meat and milk (amount of N waste available)
- Animal age and/or weight
- Animal housing system
- Manure storage system (pile, open/closed tanks, lagoon)

Recent studies, using a mass balance approach to estimate NH_3 emission rates, found that hog houses represent a more significant source than previously thought (Doorn et al., 2002). Based on a review of published swine data, Westerman et al. (2000) estimated the loss of ammonia from barnhouses to be around 15% of total nitrogen excreted. Griffing et al. (2003) used the mass balance method to estimate that about 80% of ammonia loss was due to volatilization from liquid waste storage systems. For this study, the mass balance method was employed to estimate NH_3-N loss rates relative to the nitrogen waste available for conventional waste lagoon storage systems for comparison with alternative waste treatment methods.

Program OPEN (Odor, Pathogens, and Emissions of Nitrogen) is an integrated study of the emissions of ammonia, odor and odorants, and pathogens from potential environmentally superior technologies (ESTs) for swine facilities. It is funded by the Animal and Poultry Waste Management Center at North Carolina State University. Its main purpose is to evaluate potential environmentally superior swine waste management technologies that have been developed and implemented under an agreement between the North Carolina Attorney General and several companies that own 10% of the swine farms in North Carolina, mostly employing the conventional Lagoon and Spray Technology (LST). Under Program OPEN, ESTs implemented at selected swine facilities are being evaluated to determine if they would be able to substantially reduce atmospheric emissions of NH_3, odor, and pathogens. This study focuses on the emissions of nitrogen in the form of ammonia (NH_3-N 14/17) from different components/processes involved in waste handling and treatment, including waste storage ponds, barnhouses, and spray fields at two selected EST farms and compares them with emissions from two conventional LST farms, called here as baseline

farms. This study is also a continuation and extension of a Master's Thesis research by Heather Arkinson (2003) who analyzed the data collected at the two EST farms (Barham and Howard) in April and June of 2002, both representing the warm season only.

The primary objective of this study was to quantitatively measure and estimate ammonia fluxes from various swine waste storage ponds at two potential Environmentally Superior Technologies (ESTs), by using a dynamic flow-through chamber system with an environmentally controlled mobile laboratory, and to compare them with fluxes from two conventional Lagoon and Spray Technology (LST) swine waste facilities. A special emphasis is placed on quantifying the emissions of ammonia, using a mass balance method, to develop emission factors based on nitrogen losses from major components of the swine waste treatment systems from LSTs and ESTs and to investigate their seasonal variability. A secondary objective was to analyze the measured NH_3 fluxes with respect to changes in physiochemical parameters such as lagoon temperature, air temperature, pH, and total ammoniacal nitrogen (TAN), and to obtain multiple regression relations for ammonia fluxes over the various storage ponds and lagoons used at different technology sites.

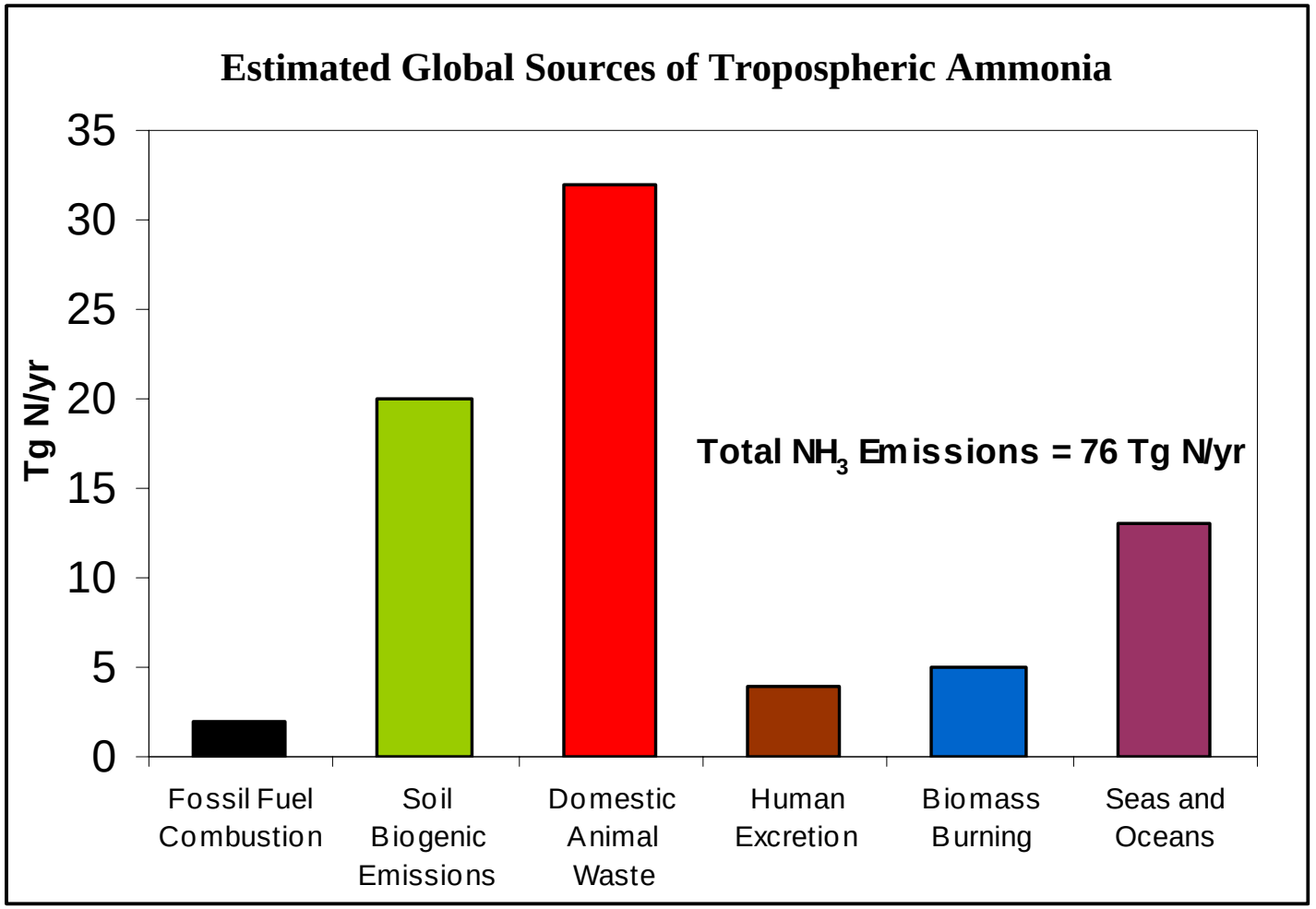


Figure 1.1 Global sources of tropospheric ammonia (Schlesinger & Hartley, 1992).

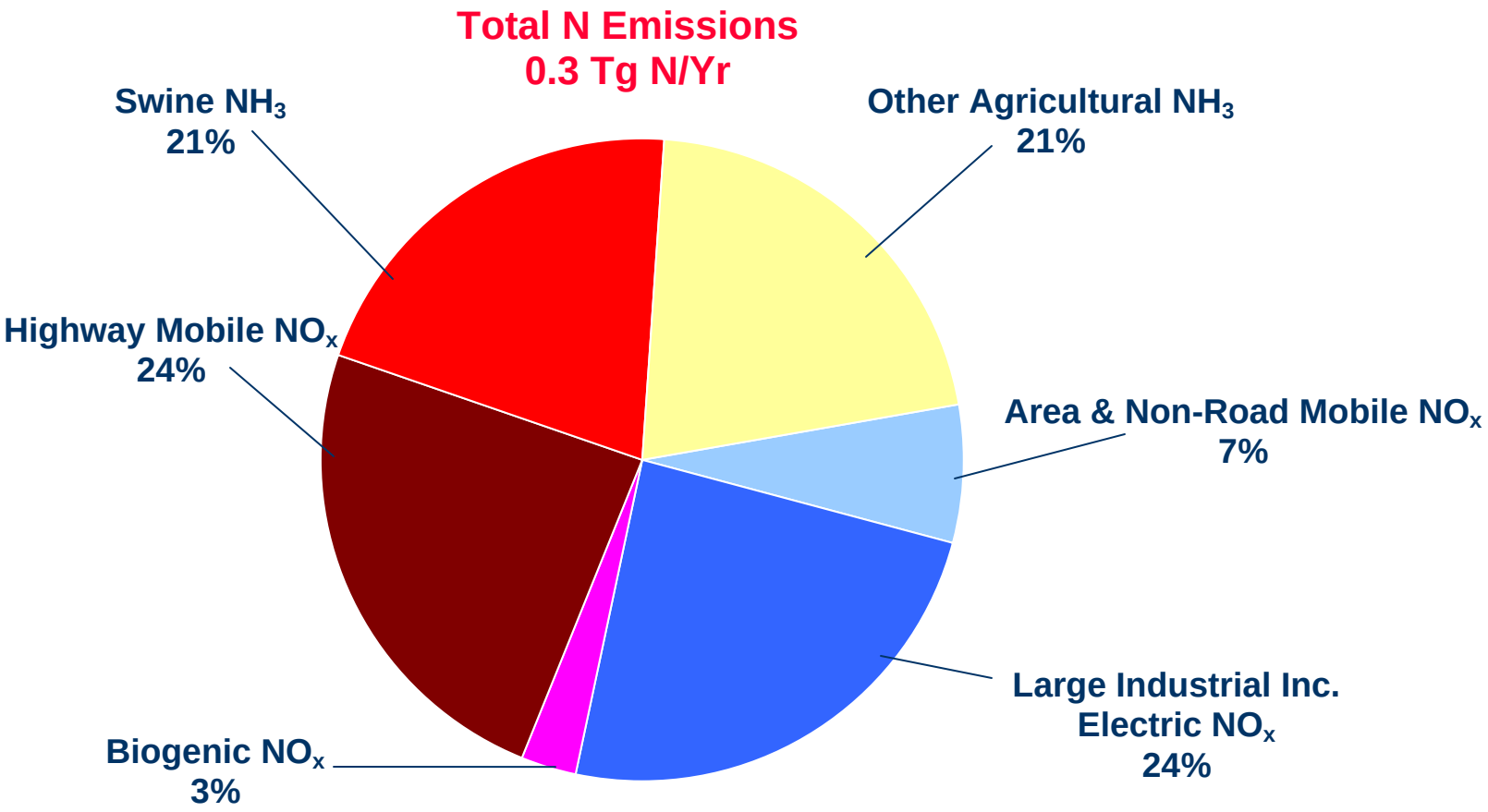


Figure 1.2 Percent nitrogen from NO_x and NH_3 sources in North Carolina (NCDA, 1996).

Figure 1.3 Increase of hog population in North Carolina over the last several decades (NCDA, 2000).

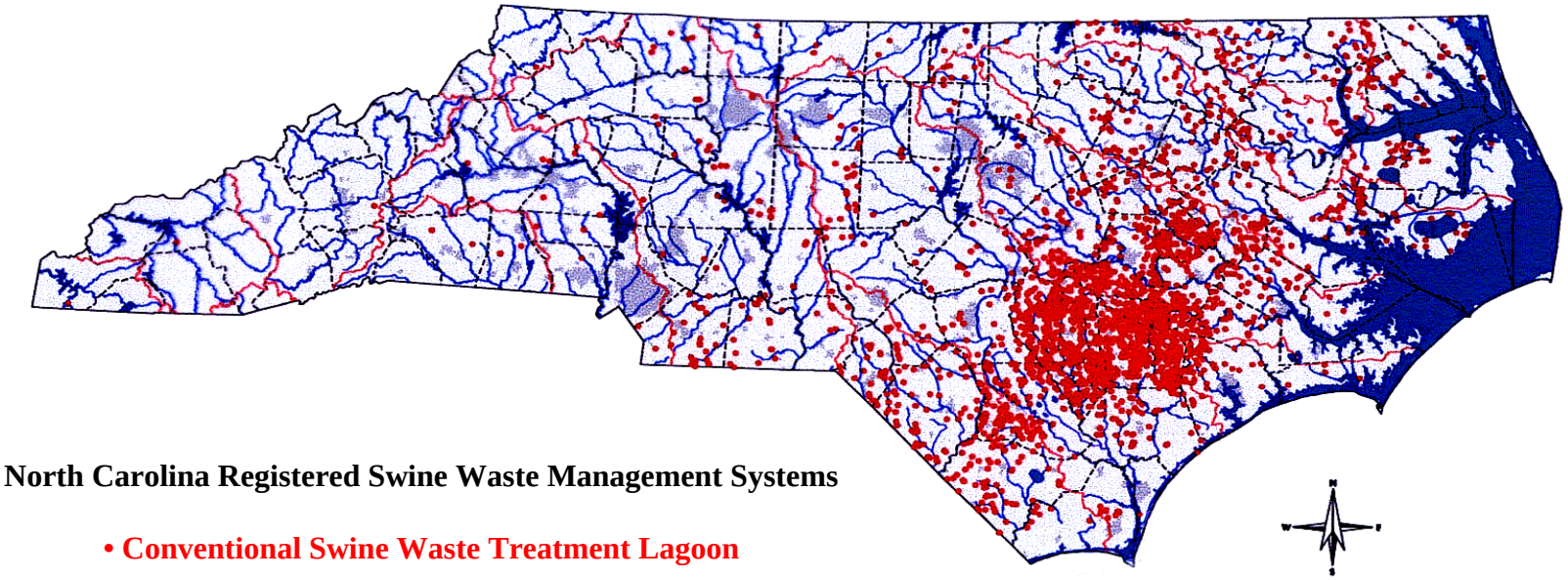


Figure 1.4 Swine farm distribution in North Carolina (NCDA, 2000).

2.0 METHODS AND MATERIAL

2.1 Sampling Sites

Ammonia flux measurements were conducted during two different seasons at four swine facilities in eastern North Carolina. The two conventional Lagoon and Spray Technology (LST) sites employed anaerobic lagoons, which did not require dissolved oxygen in the bacteria treatment of organic wastes in the lagoons. These LST sites are also referred to as "baseline" sites for comparison with EST sites. The two Environmentally Superior Technology (EST) sites were Barham and Howard farms, which employed the In-ground Ambient Digester Technology and Constructed Wetlands/Solid Separation Technology, respectively. Locations of these farms in eastern North Carolina are shown in Figure 2.1.

Lagoon and Spray Field Technology

Sampling was conducted at Stokes Farm (35.43^{o}N, 77.48^{o}W, 17 m MSL), located near Trenton, NC in Pitt County, during September 9-20, 2002 and January 6-17, 2003. Four naturally ventilated finishing barns housed 4,391 animals with an average weight of 104 kg in the fall season and 3,726 animals with an average weight of 88 kg in the winter season. Waste from the hog barns were flushed with recycled lagoon water and discharged into a storage pond from a single effluent pipe. Lagoon surface flux measurements were taken from various sampling locations from the lagoon illustrated in Figure 2.2.

Sampling at Moore Farm (35.14^{o}N, 77.47^{o}W, 13 m MSL) located near Kinston, NC in Jones County, was conducted during September 30 - October 11, 2002 and January 27 - February 7, 2003. Eight fan-ventilated finishing barns housed 7,617 animals with an average weight of 52 kg in the fall season and 5,784 animals with an average weight of 67 kg in the winter season. Waste from the hog barns were flushed with recycled lagoon water and

discharged into a storage pond from eight effluent pipes, one for each hog barn. Lagoon surface flux measurements plots are illustrated in Figure 2.3.

Environmentally Superior Technologies

1. In-ground Ambient Anaerobic Digester Technology

Sampling occurred at Barham Farm (35.70^{o}N, 78.32^{o}W, 130 m MSL), located near Zebulon, NC in Johnston County, during April 1-12, 2002 and November 11-22, 2002. The potential EST was an In-ground Ambient Digester Technology, which consisted of a covered anaerobic waste lagoon system (Figure 2.4). The primary anaerobic waste treatment lagoon was covered by an impermeable layer of 40 mm thick high-density polypropylene that prevented gaseous methane and other gases from escaping into the atmosphere during this digestion process. The trapped methane gas was extracted and burned in a biogas generator that provided electricity for the greenhouses and powered a water heater used for farm production activities. The effluent from this primary covered lagoon was discharged into a secondary stage lagoon, referred to as a storage pond. Biofiltration devices pumped effluent from the storage pond where bacteria oxidized the reduced forms of nitrogen from the effluent to nitrate (NO_3^-). This nitrified effluent was then used to flush out the hog production facilities and excess effluent was channeled into the larger overflow pond. A heavy polymer baffle separated the overflow and storage ponds. The overflow pond was used to store excess rainwater and overspills from the storage pond. The overflow pond was also pumped into a biofiltration system where the nitrates in the water were used to fertilize greenhouse tomatoes on the farm. Barham Farm was a farrow-to-wean operation, which included sows, gilts, boars, and piglet up to three weeks old. Four gestation barns housed 3,360 sows and two farrowing barns housed 640 piglets totaling 4,000 pigs with an average weight of 238 kg at

any given time. These figures were the same for both spring and winter measurement periods. This study conducted ammonia flux measurements from the surfaces of the storage and overflow ponds.

2. Constructed Wetlands/Solid Separation Technology

From June 3-14 and December 2-13, 2002, sampling was conducted at Howard Farm (34.84^{o}N, 78.40^{o}W, 5 m MSL) located near Richlands, NC, in Onslow County. The potential EST was a waste treatment system that utilized a wetland ecosystem to reduce ammonia emissions from the surface of the wetland cells (Figure 2.5). A solid separator was used to separate the solid from liquid waste. The solid portion was applied to crop fields as fertilizer and was also used to operate a vermiculture farm. The liquid effluent was broken down by microbes contained in the Cattail plant species through oxidation and reduction. At the end of the wetland cell process, the treated effluent was discharged into a finishing pond, which was used for spray field applications for adjacent croplands. Howard Farm was a finishing operation, which consisted of four hog houses with 3,618 pigs weighing an average of 64 kg in the summer season and 3,881 pigs weighing an average of 97 kg in the winter season. This study included ammonia flux measurements from the inlet, midpoint and outlet cells of the wetland system, from the finishing pond, and from the soil of adjacent cropland.

2.2 Ammonia Flux Measurement

Dynamic Flow-Through Chamber System

A flow-through dynamic chamber system with a variable-speed continuous impeller stirrer (Aneja et al., 2000; Chauhan 1999; Kim et al., 1994; Kaplan et al., 1988) was used to determine NH_3 flux from lagoon and soil surfaces at Stokes, Moore, Barham and Howard farms (Figure 2.6). The translucent plastic cylindrical chamber, 26 cm in diameter and 45.7

in height (a volume equal to 24.34 L), was fitted into a circular hole cut into the center of a 1.2 by 1.2 meter floating plywood platform, which penetrated the lagoon surface by ~7 cm. To create a closed system inside the chamber, a seal was formed between the bottom of the cylinder and the lagoon water. The cylindrical chamber was lined with a 5-mm thick fluorinated ethylene propylene (FEP) Teflon sheet throughout the inside surface of the chamber. Compressed zero grade air, which was used as a carrier gas, was passed through the chamber at a variable flow rate of 4-8 lpm using a mass flow controller. The air inside the chamber was ideally well mixed by a variable-speed, motor driven Teflon impeller ranging from speeds of 40-60 rpm for this study. Roelle (1996) found that varying the speed of Teflon impeller did not produce any significant changes in the calculated NO soil flux using the chamber method. Research conducted on pressure differences between the outside atmosphere and air within a chamber using a tilted water manometer indicated that pressure differences were below detection limits (0.2mm H_20) (Johansson and Granat, 1984). A vent line was fitted to the exiting sample line to prevent over pressurization and was bubble tested periodically to check for under pressurization or leaks in the enclosed system. Sample lines did not exceed 10 meters. Bunton (1999) conducted an experiment to explore possible differences between air temperature inside the chamber and the ambient temperature. The study found maximum temperature differences of 2.5^{o}C and 3.4^{o}C over two twenty-four hour experiments, which may have minimal effect on the temperature of the lagoon inside the chamber.

The sample exiting the chamber traveled through Teflon tubing (6.35 mm outside diameter, 3.97 mm inside diameter) to the ammonia detection instruments. The entire closed system was lined with Teflon or stainless steel fittings in order to minimize chemical

reactions with sample flow. For soil measurements, the chamber was placed on a stainless steel ring which was inserted into the soil, one day prior to sampling. Soil flux measurements were taken two hours after the insertion of the ring to ensure steady-state conditions in the chamber. NH_3 emission data were not collected during precipitation events.

Temperature Controlled Mobile Laboratory

A temperature-controlled mobile laboratory housed all detection instrumentation for this study. The mobile laboratory consisted of a modified Ford Aerostar van with a 13,500 BTU air conditioner unit. The temperature inside the van was regulated for effective performance of the ammonia analyzers. A 110-volt outlet was used to power the air conditioning and all detection instruments.

NH_3 Detection Instrumentation

Once the dynamic flow-through chamber system reached steady-state conditions, NH_3 concentrations were measured using a Thermo Environmental Instruments (TEI) Model 17c chemiluminescence Ammonia Analyzer from the sample flow exiting the chamber. The 17c ammonia analyzer separated the incoming sample flow into three separate parts. The first path mixed the sample flow with ozone (O_3) and all of the nitric oxide (NO) within the sample reacts to produce a reading of NO concentration via the standard chemiluminescence technique. The second path involved the sample flow passing through a molybdenum converter (325^oC), which converts all oxidized forms of nitrogen (NO_x) to NO. This sample then reacts with O_3 in order to quantify the concentration of NO_x. The third path flows through a stainless steel converter (775^oC) where both NO_2 and NH_3 are transformed to NO prior to reaching the reaction chamber to measure N_T ($NO + NO_2 + NH_3$). The concentration of NH_3 was calculated by taking the difference between the signal of N_T and the signal for

oxides of nitrogen (NO_x) (TEI, 2000). The following chemical reactions and other relations summarize the TEI 17c Ammonia Analyzer chemiluminescence technique for the TEI Model 17c Ammonia Analyzer:

$$NO + O_3 \rightarrow NO_2^* + O_2 \quad (6)$$

$$NO_2^* \rightarrow NO_2 + h\nu \quad (7)$$

$$N_T = NO_x + NH_3 \quad (8)$$

$$N_T - NO_x = NH_3 \quad (9)$$

Calibrations for the TEI Model 17c Ammonia Analyzer were conducted using a TEI Model 146 dilution-titration instrument in conjunction with cylinders of 20 ppmV and 900 ppmV of NH_3 in N_2 and zero grade air (Machine and Welding Purity Gases, NIST certified). The TEI 146 was serviced and calibrated to specification by the manufacturer and was re-certified by North Carolina Division of Air Quality (NCDAQ) technicians. A multipoint calibration was conducted before each two-week field measurement campaign. Zero and span checks were conducted every day of the experiment according to the TEI 17c Ammonia Analyzer Operator's Manual (TEI, 2000).

2.3 Automated Data Collection

A Gateway laptop computer and a Campbell Scientific CR21X Datalogger (PC208W software) were used as an automated data acquisition system. The CR21X datalogger recorded 15-minute averaged measurements for NH_3 concentrations inside the chamber, lagoon pH, and lagoon temperature. From a 10 m tower, the CR21X also collected 15-minute averaged measurements of ambient NH_3 concentrations at 10 m and meteorological parameters including wind speed and direction, air temperature, relative humidity, and solar radiation. The 15-minute averaged NH_3 concentrations in the chamber were used to calculate

ammonia flux during data analysis. Recorded values were checked against the front panel of detection instruments to ensure accuracy. There were no significant discrepancies found between datalogger stored values and instrument display readings for this study.

2.4 Meteorological Measurements

At each sampling site, a 10 m meteorological tower was erected to measure wind speed and direction, air temperature, relative humidity, and solar radiation. All meteorological instrumentation for this study were purchased from Campbell Scientific Incorporated (CSI), Logan, Utah. A Met One Instruments Model 034B-L Windset was used to measure wind speed and direction at 10 m above the surface. The Model 034B-L consists of an integrated cup anemometer and a wind vane. Accuracy of the measured wind speed component is ± 0.12 m/s for wind speeds below 10.1 m/s and ± 1.1 % of reading for wind speeds above 10.1 m/s. The wind direction component has an accuracy of ± 4^{o} and a threshold of 0.4 m/s. Air temperature and relative humidity (RH) measurements were made at 2 m height facing north with a Model HMP45C temperature and relative humidity probe housed in a radiation shield. RH accuracy is ± 2% (0-90% RH) and ± 3% (90-100% RH) while air temperature accuracy is 0.2-0.5^{o}C. Solar radiation measurements were also made at 2 m height but facing south using a Model LI200X Silicon Pyranometer probe. Solar radiation has an absolute error in natural daylight of 5% maximum and 3% typical. (CSI Operator Manual Reference)

2.5 Lagoon Parameters and Ambient NH_3

A CSI Model 11-L50 Innovative Sensors pH probe continuously monitored lagoon pH during lagoon NH_3 flux measurement periods. Two CS107 temperature probes measured lagoon temperatures simultaneously inside the chamber and 0.5 m outside of the chamber.

Differences in lagoon temperatures inside and outside the chamber were found to be insignificant (less than $1^{o}C$). These pH and temperature probes were submerged in the lagoon at a depth of 15~20 cm. Lagoon water samples were collected daily from measurement sites and submitted to the Department of Biological and Agricultural Engineering (BAE), North Carolina State University, for analysis of TAN (NH_3-N + NH_4^+-N) measurements. The BAE Environmental Analysis Laboratory used an ammonia-salicylate method for automated analysis of TAN.

Ambient NH_3 concentrations were measured from Teflon tubing extending from the top of a 10 m meteorological tower to a TEI Model 17c Ammonia Analyzer housed in the mobile laboratory unit. To prevent particles from entering into the sample inlet of the Model 17c Ammonia Analyzer, a one-stage Teflon filter (42 mm thick with 1 μm pore size) was placed at the sample inlet at 10 m height.

2.6 Sampling Scheme

The dynamic flow-through chamber system was flushed with compressed air for an hour, before a daily sampling period, to prevent accumulation of excess ammonia or moisture in the chamber and sample lines in order to account for variability in different areas or sections of each site's lagoon storage system. The whole floating chamber system, the mobile laboratory and meteorological tower, were moved one or two times to different sectors of the lagoons in order to examine possible variability. The floating chamber itself was periodically moved within a radius of 2 m from the previous plot on a daily basis.

2.7 Ammonia Flux Calculation

In order to calculate ammonia flux for this study, the following mass balance equation was used for the dynamic flow-through chamber system:

$$\frac{dC}{dt} = \left(\frac{q[C_{air}]}{V} + \frac{JA}{V} \right) - \left(\frac{LA_w}{V} + \frac{q}{V} \right)[C] \qquad (10)$$

where

C	NH_3 concentration in the chamber (ppbV)
C_{air}	NH_3 concentration in ambient air (ppbV)
q	flow rate of compressed air through the chamber (lpm)
V	volume of the chamber (24.34 L)
A	surface area covered by chamber (m^2)
A_w	inner surface area of the chamber of inner and upper wall surfaces (0.374 and 0.209 m^2 respectively)
L	total loss of NH_3 in the chamber per unit area (m min^{-1}) due to reaction with inner and upper walls of the chamber
h	internal height of the chamber (45.7 cm)
J	emission flux per unit area (μg NH_3-N m^{-2} s^{-1})

Zero air was used as a carrier gas, so C_{air} is assumed to be zero. Since the chamber is assumed to be well-mixed, concentration, C, is constant within the chamber. At steady-state conditions, the change of concentration with respect to time will be zero. Equation (1) can be simplified as:

$$\frac{J}{h} = \left(\frac{LAw}{V} + \frac{q}{V} \right) C_{eq} \qquad (11)$$

Loss term (L) was determined experimentally while equilibrium-state ammonia (C_{eq}), flow rate (q) and dimensions of the chamber (V and h) were all measured. Kaplan et al. (1988) devised a method for calculating loss term by calculating the slope of the plot of

$-\ln\left[\frac{C_{eq} - C(t)}{C_{eq} - C_o}\right]$ versus time (t). For this experiment, C_o is the initial equilibrium state NH_3 concentration measured by the chamber system at a constant flow rate (12-14 lpm). C_{eq} is the measured NH_3 concentration at a second equilibrium state at a reduced flow rate (4-6 lpm) into the chamber system. C(t) depicted NH_3 concentration at any time, t, during the transition between the first and second equilibrium states. The following equation shows how L is determined:

$$L = \left(slope \quad of \quad line \quad - \frac{q}{V}\right)\left(\frac{V}{A_w}\right) \tag{12}$$

where A equals the area of the inner walls of the chamber.

2.8 Climatological Data

Climatological data provided by the State Climate Office of North Carolina, located at North Carolina State University, were used to compare monthly averages of air temperature and precipitation for the experimental period month against the monthly averaged 10-year average data for the same month. Historical climate data revealed normal or abnormal variations of air temperature and precipitation parameters during the field experimental periods. Significant climatological variations from 10-year averages for the months of the experimental periods can be used to identify abnormal meteorological conditions that might suggest atypical data obtained from an EST or LST site which may not be representative of that particular month or season.

2.9 Nitrogen Excretion based on Animal Feed Analysis

Hog feed analysis was used to calculate the mass of nitrogen excretion produced from each experimental site. Based on hog feed, weight and population, the following equation was used to determine the amount of nitrogen excretion for each farm:

$$E = \frac{F \times \%N \times (1 - ER)}{W} \times 1000 \tag{13}$$

where

E	Nitrogen Excretion (kg N/1000 kg LAW/ year) where LAW = Live Animal Weight
F	Feed consumed (kg/pig/year)
%N	percentage of nitrogen in feed, expressed as a fraction
ER	Feed efficiency rate (PigCHAMP, 1999) depending on farm operation type (e.g. wean to farrow, feeder to finish)
W	Average weight of hog for each farm (kg)

Nitrogen excretion data were used to determine the total nitrogen introduced into a swine CAFO and for normalizing the NH_3-N emissions from its individual components (barns, lagoon, etc.).

2.10 Baseline Observational Statistical Model

In order to evaluate the potential reduction of total nitrogen based on animal feed for ESTs (Barham and Howard farms), Stokes and Moore farms were chosen as baseline farms (LSTs) for obtaining background measurements of ammonia. These LST sites were used to compare the effectiveness of EST sites in reducing nitrogen based on the ratio of total nitrogen emissions from the liquid waste storage system and the total nitrogen excretion based on animal feed analysis. This ratio is denoted as %E and is estimated from the following parameters:

$$\%E = \left[\frac{Total\ Nitrogen\ Emissions\ \ from\ Lagoons\ (N\ emitted)}{Nitrogen\ Excretion\ based\ on\ Animal\ Feed\ (total\ N\ excreted)} \right] \times 100 \qquad (14)$$

Experimental periods for ESTs were chosen at different times of the year, therefore environmental conditions were different at each site. In order to account for these differences, a multiple regression analysis of the dependence of ammonia emission on measured environmental parameters at the two baseline sites was conducted. Figure 2.7 depicts the algorithm employed to evaluate ESTs with lagoon and/or storage ponds based on the strong multiple linear regression relationship between NH_3 flux and lagoon temperature and the difference between air and lagoon temperature. First, data analysis of NH_3-N measurements from baseline farms (Stokes and Moore farms) and the corresponding air and lagoon temperatures were performed to obtain the observational statistical model relationship based on multiple linear regression analysis as shown below:

$$Log_{10}(Area * Ammonia\ \ \ Flux/ton) = a + (b * T_{lagoon}) + (c * \Delta T) \qquad (15)$$

where a, b and c are experimental constants, T_{lagoon} represents lagoon temperature, T_{air} represents air temperature, $\Delta T = T_{air} - T_{lagoon}$. *Area* denotes the surface area of the lagoon at the baseline farm where flux was measured and ton represents the total live animal weight at the farm in metric tons (1000 kg). When ΔT was found to be positive or $\Delta T>0$, then $(c*\Delta T)$ was used in the equation but when ΔT was negative or $\Delta T<0$, then $(c*\Delta T)$ was zero. The reasoning for employing ΔT in the statistical model will be discussed in later sections. By inserting averaged measurements of lagoon temperature and ΔT (only when $T_{air} > T_{lagoon}$) for the experimental period and appropriate constants/coefficients of the observational model, an averaged NH_3-N baseline flux value, $F_{projected}$, is estimated for each EST experiment. $F_{projected}$ is then compared with the measured averaged NH_3-N flux value, $F_{measured}$, during the actual

experimental period. This comparison was performed by calculating the ratios $F_{measured}$/total N excreted from the EST sites and $F_{projected}$//total N excreted from baseline farms. The potential reduction of normalized NH_3-N emission from EST sites was determined from the following equation:

$$\Delta(\%E) = \%E\ of\ \ baselines - \%E\ of\ \ ESTs \tag{16}$$

where positive Δ(%E) would show that an EST is effective in reducing NH_3-N relative to the total N excreted while a negative Δ(%E) would show no NH_3-N reduction from an EST site. An important assumption for the potential reduction of nitrogen emission from EST sites is that the total N excretion for baseline farms should be fairly constant throughout the year as is shown in the results of this study. Hog weight, hog population, %N of hog feed, and hog feed efficiency rates (ER) based on age are all considered in the evaluation of EST sites using the observational statistical model.

2.11 Measurement Error Calculations

In order to determine confidence intervals for the calculation of overall %E, from the baseline statistical model and during the EST experimental periods, conservative measurement errors were assigned to nitrogen excretion, lagoon emission, and barn emission values. Standard error estimates for calculated nitrogen excretion values for EST and LST sites were 10%. The average NH_3-N baseline flux value from the statistical model, $F_{projected}$, and the measured averaged NH_3-N flux value, $F_{measured}$, during the actual EST experimental period were estimated to have standard errors of 20%. The standard error that was assigned to barn emissions (OP-FTIR method) conducted by Dr. Lori Todd's ammonia group, from the University of North Carolina at Chapel Hill, at EST and LST sites was also 20%.

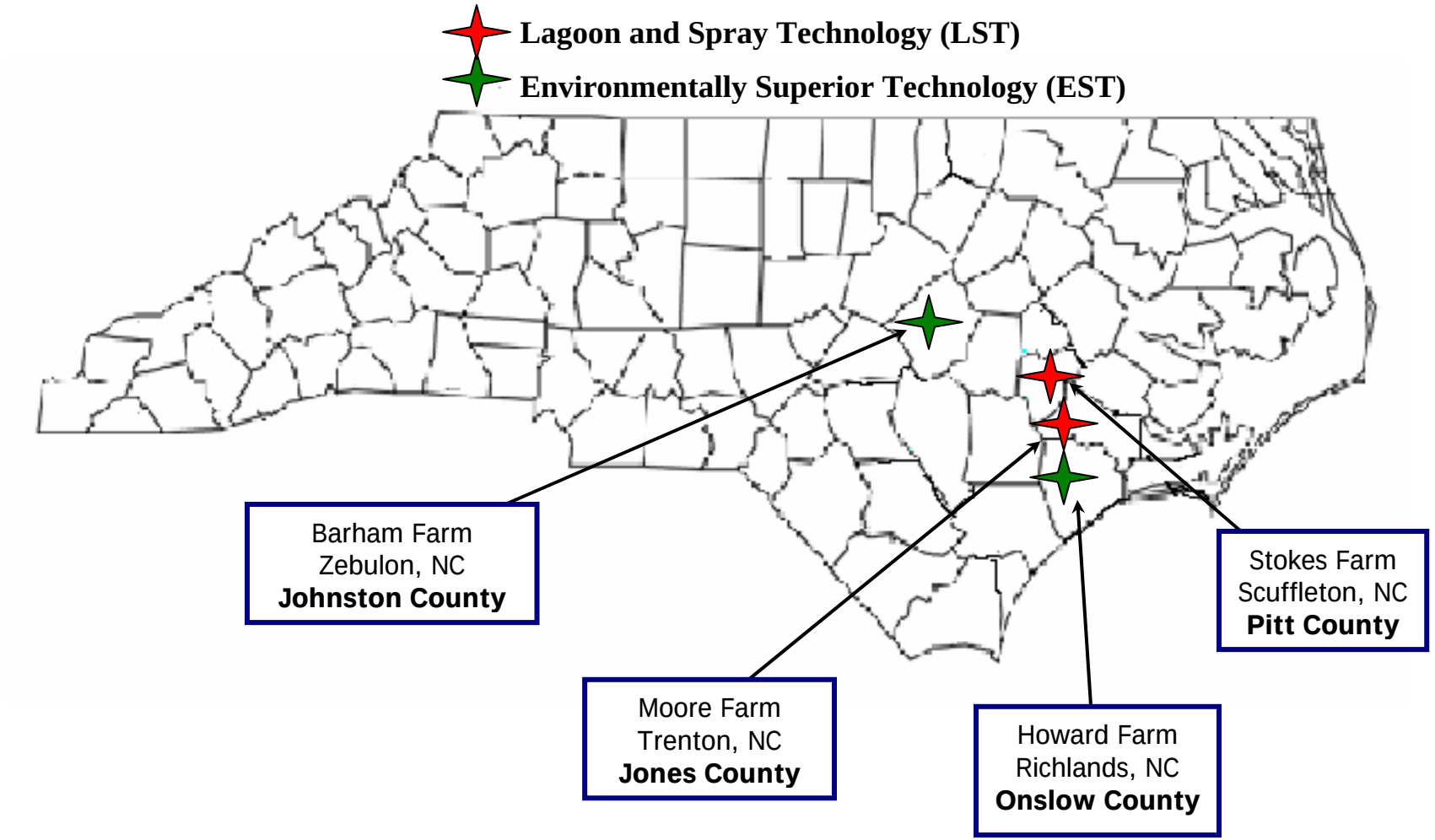


Figure 2.1 Experimental research site locations in eastern North Carolina

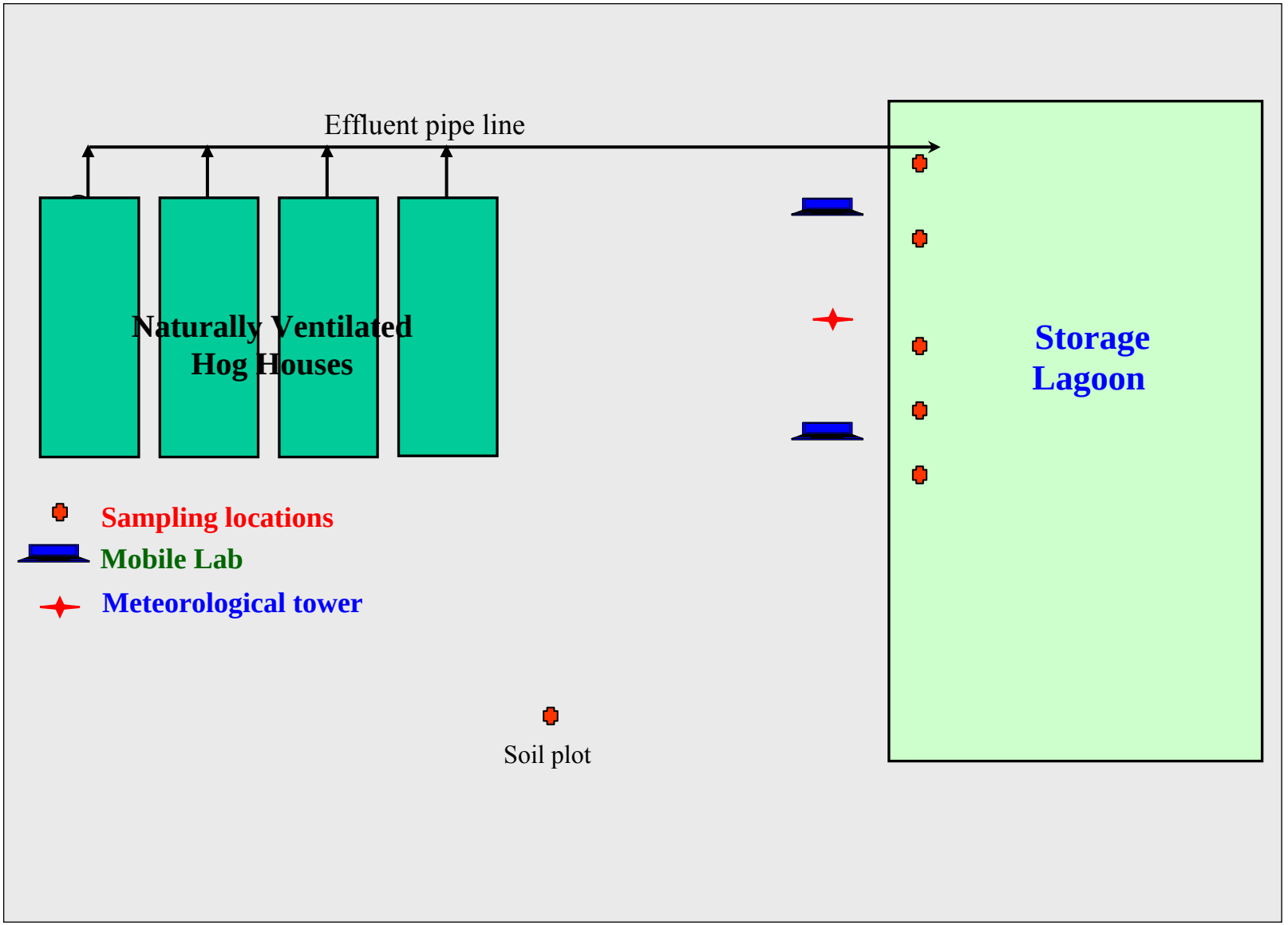


Figure 2.2 Schematic layout of Stokes farm (LST site). Note that hog houses are naturally ventilated.

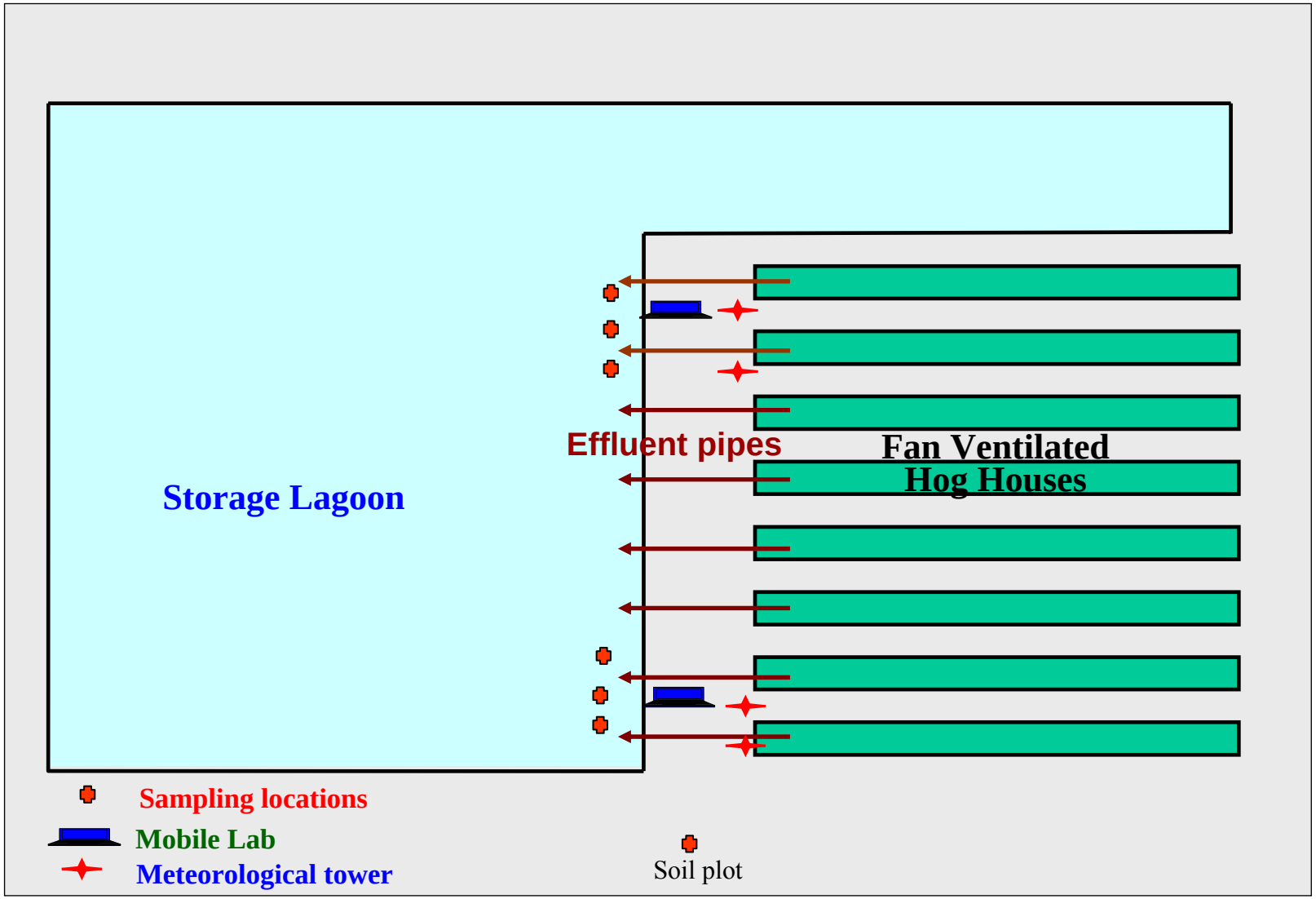


Figure 2.3 Schematic layout of Moore Farm (LST site). Note that hog houses are fan ventilated.

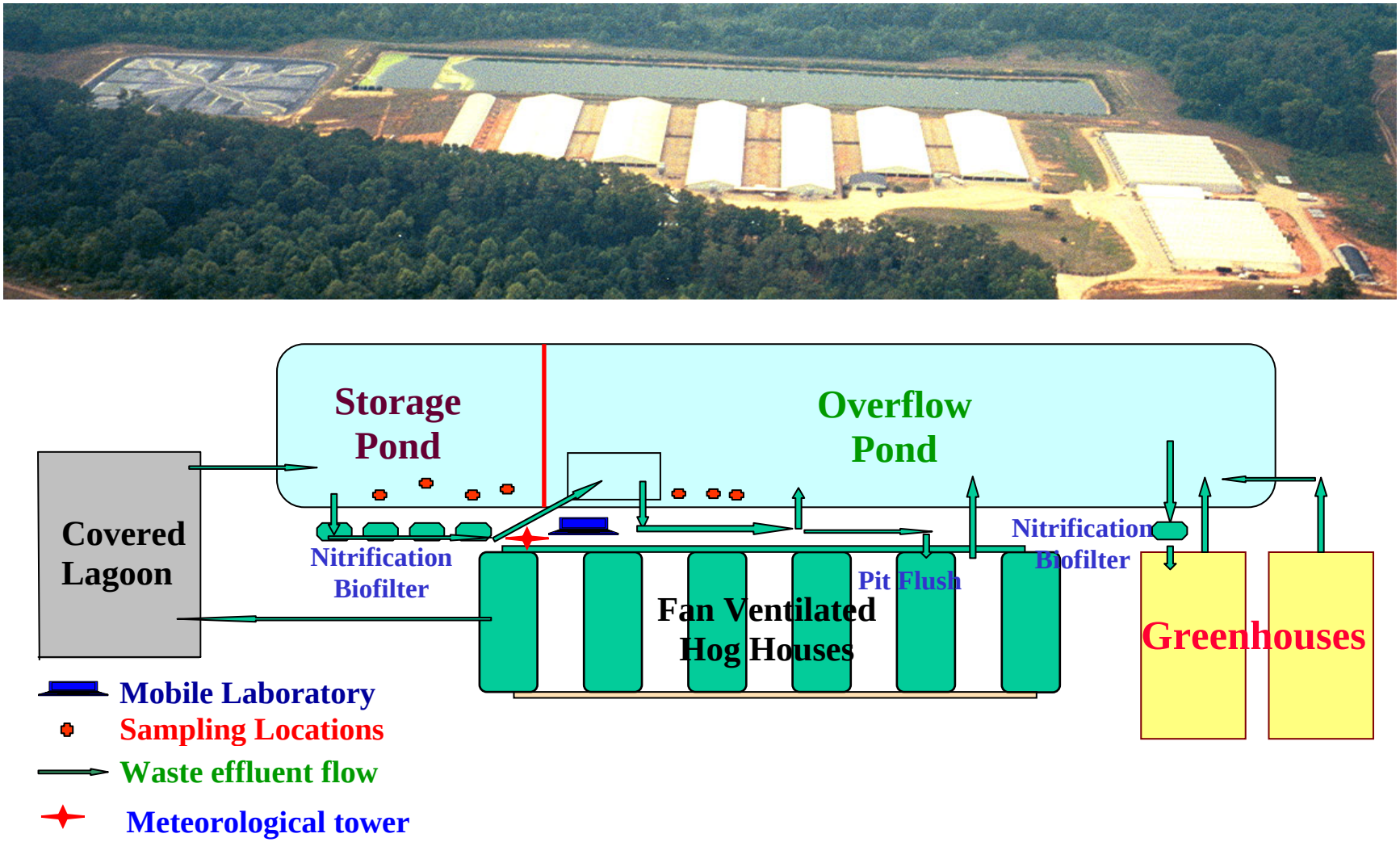


Figure 2.4 Schematic layout of Barham Farm which employed the Covered In-Ground Ambient Anaerobic Digester Technology (potential EST).

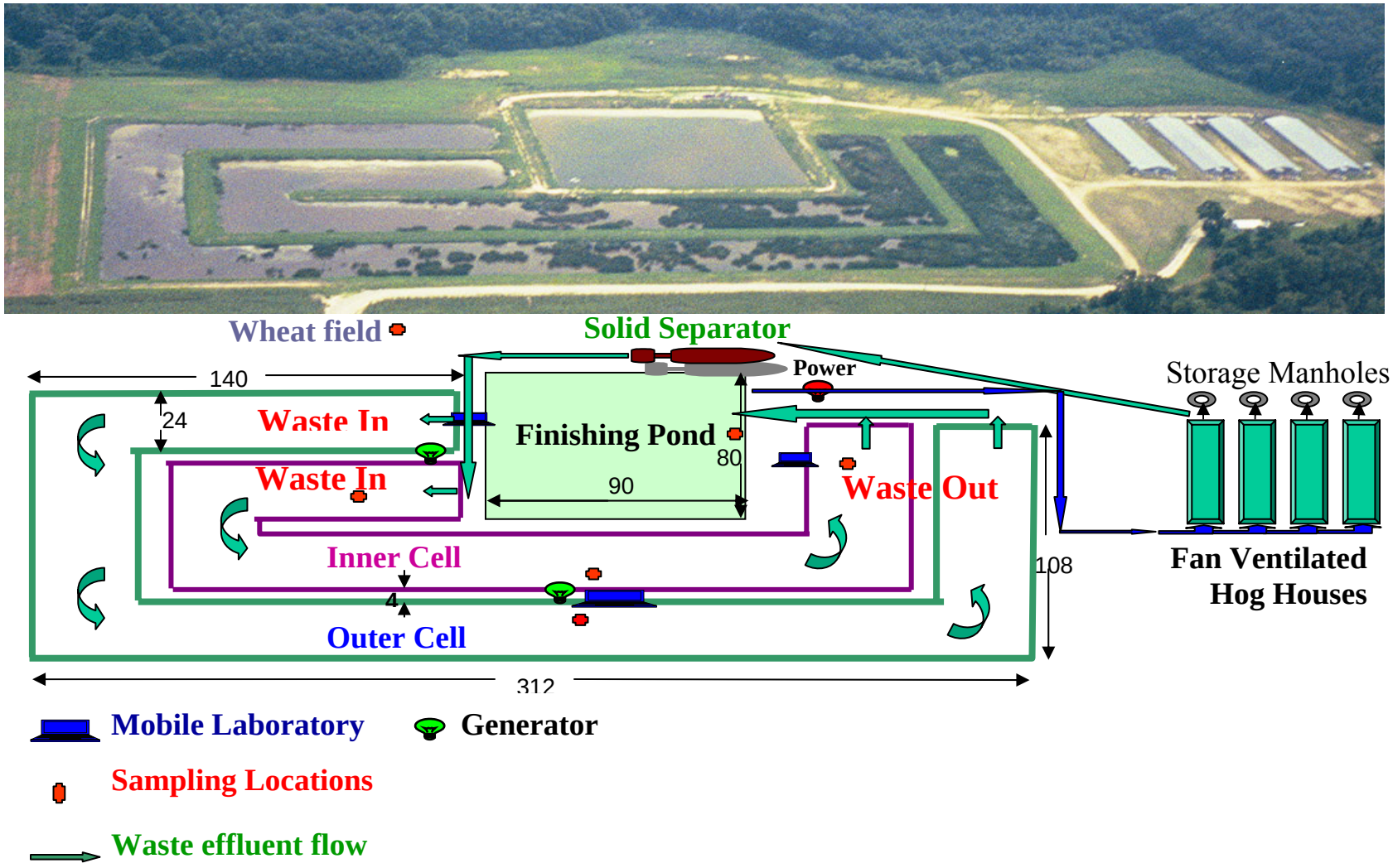


Figure 2.5 Schematic layout of Howard Farm which employed the Constructed Wetlands/Solid Separator Technology (potential EST).

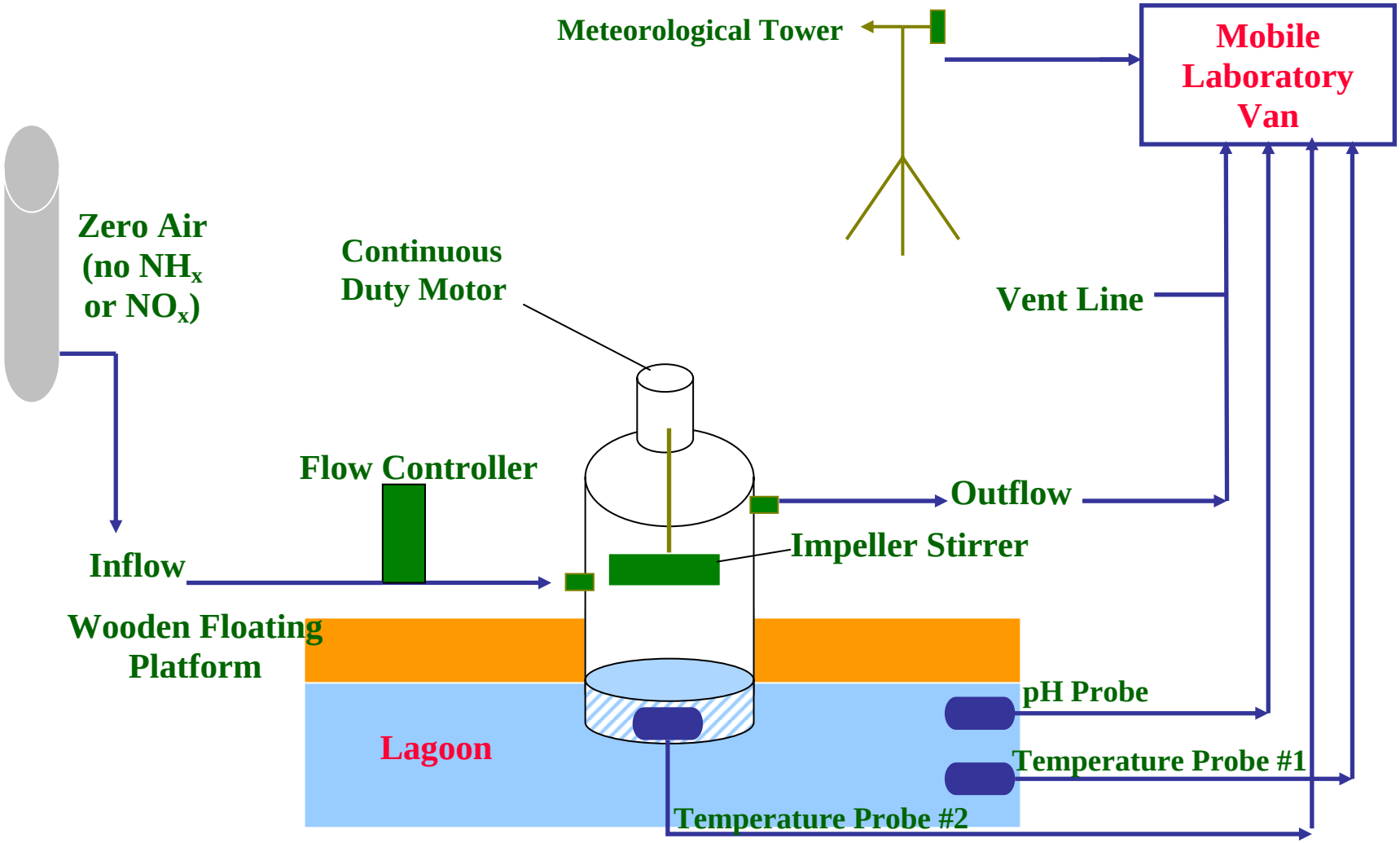


Figure 2.6 Schematic of the dynamic flow-through chamber system. The chamber fits inside a wooden floatation device, which is placed on the lagoon surface and allowed to reach steady-state before measurements are taken.

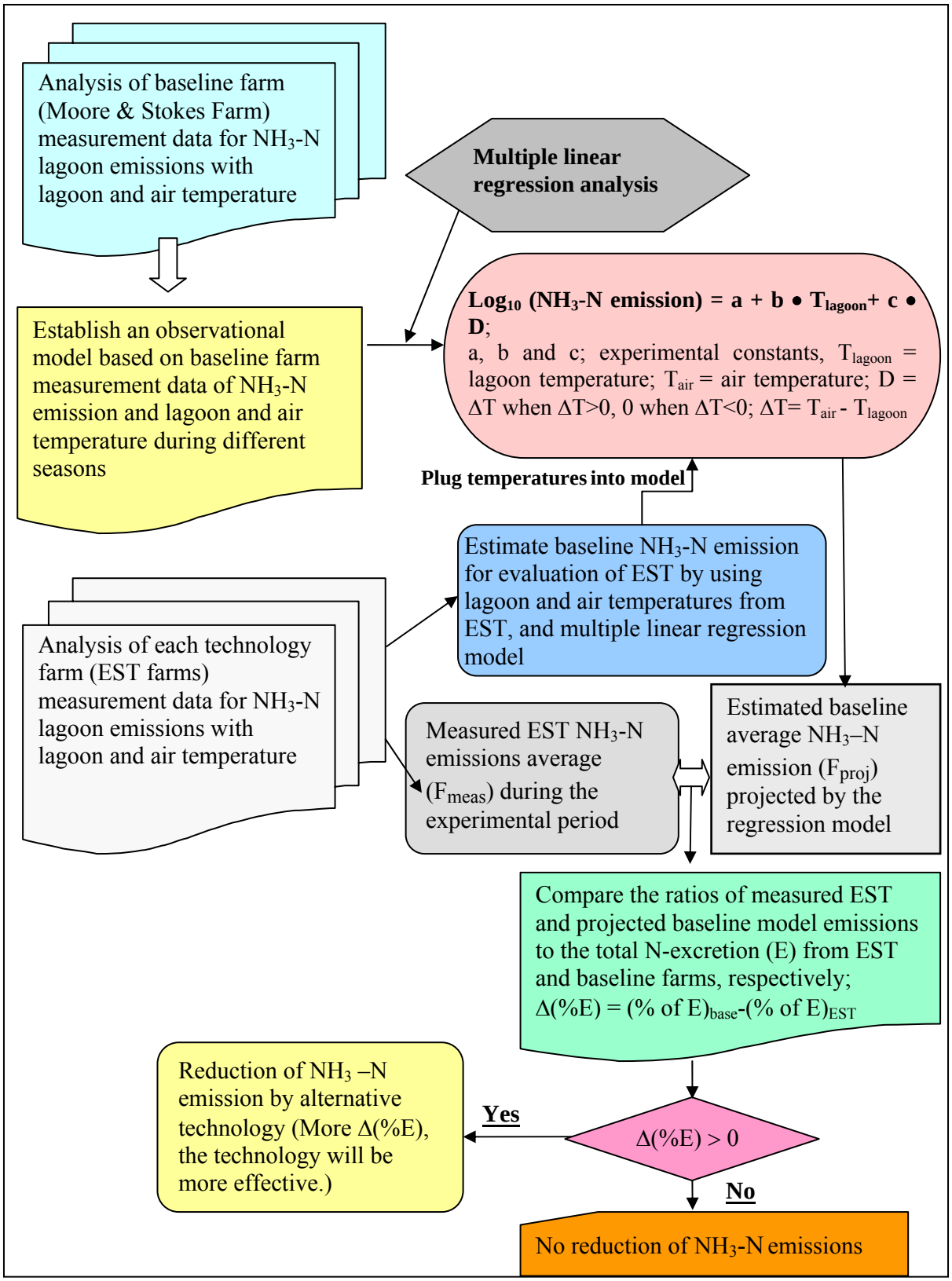


Figure 2.7 Algorithm for evaluation of ammonia emissions from EST lagoons and/or storage ponds.

3.0 RESULTS AND DISCUSSION

3.1 Climatological Data Analysis

Based on analysis of precipitation and temperature over a 50 year period (1949-1998) from 75 recording stations, precipitation in North Carolina has increased, especially during the cooler seasons (Boyles and Raman, 2003). Although that study did not include any data after 1998, individual analyses generally show that the last decade was the warmest and wettest in the later half of the last century. Other statewide general patterns, over the 50 year period analysis, include decreased precipitation during the summer and a decreasing trend between the difference of maximum and minimum temperatures possibly due to increased cloud cover and precipitation (Boyles and Raman, 2003; Dai et al., 1999; Easterling et al., 1997). Note that statewide climate patterns do not always reflect long term climate trends for individual recording stations.

EST sites

Climatological analysis of precipitation and air temperature trends were performed to compare monthly 10-year averages (1992-2001) with monthly averages during which measurements were conducted. By performing this analysis, abnormal meteorological conditions based on precipitation and air temperature difference may help to explain atypical data obtained during the experimental periods. Table 3.1 summarizes the climatological findings from this analysis for ESTs and LSTs. Climatological data were obtained from the State Climate Office (SCO) of North Carolina ECONET and National Weather Service Cooperative Observer Network (NWS/CON) recording stations.

Barham Farm climatological data were obtained from the Clayton station for the April and November 2002 measurement campaigns. The 10-year monthly averaged

precipitation from 1992-2001 for the month of April was 7.3 cm and 4.1 cm for the month of April 2002. For the month of November, the 10-year monthly averaged precipitation (1992-2001) was 8.3 cm and 9.0 cm for November 2002. For the 10-year monthly averaged air temperature for the month of April from 1992-2001, air temperature was 15.4^{o}C and 17.3^{o}C for the month of April 2002. For the November 2002 campaign, the 10-year monthly averaged air temperature was 11.0^{o}C and 9.3^{o}C for November 2002.

For Howard farm, the Hoffman Forest station was used to obtain climatological records for the June and December 2002 campaigns. The 10-year monthly averaged precipitation from 1992-2001 for the month of June was 15.2 cm and 9.4 cm for June 2002. For the month of December, the 10-year precipitation data was 7.9 cm and 7.6 cm for December 2002. The 10-year air temperature data was 24.4^{o}C for June and 24.6^{o}C for June 2002. For December, the 10-year air temperature data was 8.8^{o}C and 9.3^{o}C for December 2002.

The spring campaign for Barham Farm showed a decrease of precipitation of 4.1 cm compared to 10-year precipitation records for the months of April 1992-2001. With a higher monthly air temperature average of 17.3^{o}C for April 2002 compared to 15.4^{o}C for April 1992-2001. It may be inferred that ammonia flux measurements were taken during atypically warmer and drier conditions. It is possible that the warmer and drier atmospheric conditions during the April 2002 campaign would positively influence ammonia fluxes for Barham due to a potential increase of lagoon temperatures. For Howard farm, air temperature data for long term and current records were almost identical at 24.5^{o}C for the June 2002 measurement period, but precipitation was lower during the same experimental period at 9.4 cm compared to 15.2 cm for the 10-year climatological data for June, a difference of 5.8 cm. For the

December 2002 campaign, only minor differences were noted in precipitation data but the average air temperature for December 2002 was slightly higher compared to the 10-year average for the same month from 1992-2001.

LST sites

Stokes Farm climatological data were obtained from the Greenville station for the September 2002 and January 2003 measurement campaigns. The 10-year monthly averaged precipitation from 1992-2001 for the month of September was 18.5 cm and 11.9 cm for the month of April 2002. For the month of January, the 10-year monthly averaged precipitation (1992-2001) was 12.2 cm and 18.3 cm for January 2003. For the 10-year monthly averaged air temperature for the month of September from 1992-2001, air temperature was 22.4^{o}C and 24.2^{o}C for the month of September 2002. For the month of January, the 10-year monthly averaged air temperature was 6.3^{o}C and 6.9^{o}C for January 2003.

For Moore farm, the Kinston station was used to obtain climatological records for the October 2002 and January-February 2003 campaigns. The 10-year monthly averaged precipitation from 1992-2001 for the month of October was 9.0 cm and 6.6 cm for October 2002. For the months of January-February, the 10-year precipitation data were 11.4-7.4 cm and 8.4-9.4 cm for January-February 2003, respectively. The 10-year air temperature data was 16.1^{o}C for October (1992-2001) and 18.6^{o}C for October 2002. For January-February, the 10-year air temperature data were 6.1-7.4^{o}C and 5.5-7.2^{o}C for January-February 2003.

Climatological analysis conducted for LST experimental periods showed some deviation from 10 year averages for precipitation and little deviation in air temperature parameters. Stokes Farm experienced generally drier conditions during the fall season measurement campaign and wetter periods during the winter season measurement campaign.

Air temperature data did not show abnormal patterns for both fall and winter season research periods at Stokes farm. A similar climatological trend is observed for Moore Farm where fall season measurement periods were generally drier and winter measurement periods were wetter but no significant differences can be seen in air temperature data for both seasons.

3.2 Site Meteorological Data

Means and standard deviations of meteorological measurements are summarized for all farms in Table 3.2. Wind speed averages and standard deviations for spring and fall seasons at Barham Farm were similar at 1.9 ± 0.9 m/s and 1.5 ± 1.1 m/s, respectively. Compared to Barham Farm, Howard Farm had somewhat higher wind speed averages and standard deviations of 2.2 ± 1.4 m/s for the summer and 2.4 ± 1.5 m/s for the winter sampling period. During the early fall season, Stokes Farm exhibited similar wind speeds and standard deviations, as that of Howard Farm during the summer and Barham Farm during the spring, at 2.0 ± 1.5 m/s, while winter measurements were highest for all experimental sites at 3.3 ± 2.3 m/s. Moore Farm had wind speed measurements of 1.5 ± 1.1 for the October 2002 experimental period, which was identical to Barham Farm measurements during the November 2002 experimental period, and higher measurements, during the winter measurement period, of 2.3 ± 1.7 m/s.

Wind directions for the spring season sampling period at Barham Farm were 180 ± 120 degrees or generally from the south, but with large variability in wind directions originating from the southeast and southwest. In the fall season, measurements of wind direction at Barham farm were 266 ± 75 degrees, or predominantly from the west. A similar trend is observed at Howard Farm, during summer season measurements, where wind directions were from 156 ± 105 degrees or from the southwest with wind directions varying

from the east and south directions, and during the winter measurement period measured wind directions were from 252 ± 100 degrees or from the southwest to west. At Stokes Farm, wind directions were 143 ± 95 degrees or mainly from the southeast direction with significant variations during the fall measurement period. During the winter sampling period at Stokes Farm, wind directions were 232 ± 41 degrees or mainly from the southwest with small variability. Wind direction measurements were from 142 ± 87 degrees or mainly from the southeast for the fall measurement period at Moore Farm, which was almost identical to fall measurements at Stokes Farm, and measurements during the winter season were from 277 ± 93 degrees or generally from the west. Warm season measurements of wind directions seemed to generally come from the south to southeast directions, while cooler season measurements were mostly from the southwest to west directions.

Air temperature measurements were 14.8 ± 4.4^{o}C for the spring field measurement campaign and 10.3 ± 5.3^{o}C for the fall measurement period at Barham Farm. At Howard Farm, air temperature averages and standard deviations were 23.3 ± 5.2^{o}C for the summer experimental period, and during the winter experimental period, air temperatures were 4.2 ± 4.3^{o}C. Air temperature measurements at Stokes Farm were 24.1 ± 4.1^{o}C during the early fall measurement period and 4.6 ± 5.8^{o}C during the winter season, which were comparable with Howard Farm measurements for both warm and cold seasons. Averages and standard deviations for Moore Farm were 23.5 ± 4.4^{o}C and 6.8 ± 5.5^{o}C for the fall and winter experimental periods, respectively. Moore Farm also showed similar air temperature measurements with Stokes and Howard farms for warm and cold seasons. For this study, summer and early fall measurement campaigns had air temperature averages between 23~24^{o}C and winter season measurements had air temperature averages between 4~7^{o}C.

Hourly averaged wind speed, wind direction and air temperature were measured from a 10 m tower, and their overall (composite) averages and standard deviations as functions of time of day for each sampling period are illustrated in Figures 3.1-3.4 for all LST and EST sites. Figures 3.1a and 3.1b represent such composite meteorological measurements from Stokes Farm for fall and winter measurement periods, respectively. At Stokes farm, composite hourly averaged wind speeds ranged from 0.8 to 3.7 m/s with highest wind speeds and standard deviations during the early afternoon hours and lower wind speeds and standard deviations during the early morning hours during the fall measurement campaign. Standard deviations of hourly averaged wind direction measurements were fairly constant diurnally and composite wind directions ranged from 111^{o} to 201^{o} during the same period. Composite hourly averaged air temperature data ranged from 19.3^{o}C to 29.4^{o}C and standard deviations were relatively small (~2-3^{o}C) and did not vary significantly throughout the measurement period in the fall season. Winter season composite hourly averaged wind speeds, at Stokes farm, ranged from 2.2 to 5.5 m/s with higher wind speeds and minimum standard deviations in the mid-afternoon hours and lower wind speeds and maximum standard deviations during the night. Average wind directions ranged from 213^{o} to 251^{o} and did not vary significantly throughout the winter experimental period, especially during the day. Composite hourly averaged air temperatures ranged from 0.9^{o}C to 9.8^{o}C for the winter season measurement period at Stokes farm.

Figures 3.2a and 3.2b represent composite hourly averaged data from Moore Farm for fall and winter experimental periods, respectively. During the fall season measurement campaign, average wind speeds ranged from 0.7 to 2.5 m/s with minimum wind speeds before and after midnight and maximum wind speeds during the late morning to mid-day

hours. Wind direction measurements showed high standard deviations during the fall measurement period and their composite mean values ranged from 112^{o} to 175^{o}. Mean air temperatures ranged from 19.0^{o}C to 29.6^{o}C with slightly higher standard deviations from hourly air temperature means from the late morning to late afternoon hours. Composite mean wind speeds for winter season measurements at Moore ranged from 1.5 to 3.7 m/s with maximum wind speeds during the mid-afternoon hours and minimum wind speeds during the night. Mean wind directions ranged from 257^{o} to 304^{o} with large standard deviations throughout the winter measurement period. Composite mean air temperature ranged from 3.2^{o}C to 13.4^{o}C and exhibited fairly constant standard deviations diurnally.

Figures 3.3a and 3.3b illustrate composite hourly averaged meteorological parameters for Barham Farm during the spring and fall measurement periods, respectively. Composite mean wind speeds ranged from 1.4 to 2.8 m/s, while wind direction measurements ranged from 101^{o} to 284^{o} during the spring experimental period. For the same period, composite mean air temperatures ranged from 9.5^{o}C to 19.1^{o}C. Standard deviations for wind speed averages were higher during the night, while those for wind directions were large throughout the diurnal period. Standard deviations for air temperature were higher during the mid-day to late afternoon hours. Fall season values of composite mean wind speed ranged from 0.9 to 2.7 m/s and showed large standard deviations for all hours. Mean wind directions ranged from 230^{o} to 286^{o} with lower standard deviations during the day. Air temperature data showed high standard deviations during the night and composite averages ranged from 6.7^{o}C to 15.7^{o}C.

Composite hourly averaged wind speeds, wind directions, and air temperatures for Howard Farm are illustrated in Figures 3.4a and 3.4b for summer and winter season

measurements, respectively. During the summer experimental period, mean wind speeds ranged from 0.8 to 3.9 m/s and showed more deviations in the middle to late afternoon hours. Wind direction measurements ranged from 105^{o} to 230^{o} and had higher standard deviations from midnight to the late morning hours. Air temperatures exhibited higher standard deviations during the night and their composite mean values ranged from 17.8^{o}C to 29.1^{o}C. During the winter measurement campaign at Howard farm, mean wind speeds ranged from 1.5 to 3.8 m/s while wind directions ranged from 223^{o} to 307^{o} with a minimum standard deviation at noon. Composite mean air temperatures ranged from 1.7^{o}C to 9.3^{o}C and standard deviations were diurnally consistent.

3.3 Ammonia Emissions from Spray Fields

The In-ground Ambient Anaerobic Digester Technology employed at Barham farm, during the April and November 2002 measurement periods, was measured for ammonia emissions from a soil surface that was not used for agricultural spray fields. Howard farm, which employed the Constructed Wetlands/Solid Separation Technology, had also not received land application of treated lagoon spray during the June and December 2002 experimental periods. Ammonia concentrations for non-spray and spray fields at EST sites were below the detectable limit of 2 ppb for 200 ppb range (TEI, 2000). The same results were found for the baseline sites at Stokes and Moore farms where soil flux measurements did not coincide with land application of lagoon spray due to scheduling conflicts.

For mass balance considerations, soil measurements of ammonia from soils on adjacent cropland should be considered for determining emission factors for subsystems of swine operations for various seasonal conditions (National Research Council Report, 2003). Roelle (2001) found that soils fertilized with wastewater slurry emitted an average of thirty-

two times more ammonia than soils that were unfertilized. Although both lagoon effluent and cropland soils contribute significant amounts of ammonia to the atmosphere, this study did not find reportable data on soil emissions of ammonia for the summer, spring, fall and winter measurement campaigns. Future measurements of ammonia emissions, from spray fields used at swine farms, should coincide with times of lagoon spray from waste treatment systems to establish a relationship between them.

3.4 Lagoon and Environmental Parameters

Means and standard deviations of NH_3-N flux measurements and lagoon parameters are summarized for all farms in Table 3.3. For the April 2002 measurement campaign, Barham Farm had average ammonia fluxes of 1,102 ± 640 μg NH_3-N/m^2/min from the storage pond and 1,249 ± 175 μg NH_3-N/m^2/min from the overflow pond with an overall average flux of 1,169 ± 519 μg NH_3-N/m^2/min. For the November 2002 measurement campaign, average ammonia flux measurements were 436 ± 39 and 234 ± 34 μg NH_3-N/m^2/min for the storage and overflow ponds, respectively, with overall average ammonia flux of 352 ± 27 μg NH_3-N/m^2/min. Figure 3.5 illustrates hourly averaged ammonia flux measurements from storage and overflowing ponds, and compares overall measurements of hourly averaged ammonia fluxes for warm and cold seasons. In the warm season, ammonia flux measurements from the storage pond were higher than overflow pond measurements during the day and lower during the night. Non steady-state operation of the EST may explain higher average ammonia fluxes in the overflow pond during the warm season sampling period at Barham Farm. During the cold season, the treated effluent in the overflow pond exhibited consistently lower values of ammonia flux compared to storage pond measurements. Diurnally, ammonia flux measurements were found to be the highest during

the midday to late afternoon hours in the warm season, but this trend was less noticeable in the cold season. Standard deviations from overflow pond measurements of NH_3-N were smaller compared to storage flow pond measurements for both seasons, and also much lower for the overall NH_3 flux measurements for the cold season.

At Howard Farm, average measured ammonia fluxes were 914 ± 437 μg NH_3-N/m^2/min from the wetland cells and 434 ± 136 μg NH_3-N/m^2/min from the finishing pond for the June 2002 field campaign and 289 ± 47 μg NH_3-N/m^2/min and 152 ± 8 μg NH_3-N/m^2/min for the wetland cells and finishing ponds, respectively, for the December 2002 campaign. The overall average ammonia fluxes for Howard farm were 790 ± 336 μg NH_3-N/m^2/min for summer measurements and 243 ± 29 μg NH_3-N/m^2/min for winter measurements. Figure 3.6 illustrates hourly averaged ammonia flux measurements from wetland cells and finishing pond, and compares overall measurements of hourly averaged ammonia fluxes for warm and cold seasons at the EST. For warm and cold seasons, ammonia flux measurements from the wetland cells were higher, for each hourly averaged NH_3-N flux measurement, compared to measurements from the finishing pond. The summer sampling period also had consistently higher hourly averages of ammonia flux compared to winter sampling period. Again, ammonia flux measurements varied diurnally with highest NH_3 fluxes during the midday to late afternoon hours in the warm season, with a much weaker diurnal trend in the cold season. Standard deviations of ammonia fluxes were also larger for warm season measurements than cold season, and also larger from wetland cells compared to finishing pond measurements.

For the fall season measurement campaigns, the average ammonia fluxes for the storage ponds were 2,385 ± 565 μg NH_3-N/m^2/min for Stokes Farm during the September

2002 campaign and 1,657 ± 207 μg NH_3-N/m^2/min for Moore Farm during the October 2002 measurement period. For the winter season measurement campaigns, average ammonia flux measurements were 153 ± 19 and 325 ± 45 μg NH_3-N/m^2/min for the storage ponds at Stokes and Moore farms, respectively. Both baseline farms exhibited maximum hourly averaged measurements of ammonia flux in the mid to late afternoon hours with higher standard deviations in the warm season as illustrated in Figure 3.7. The diurnal trend of ammonia flux is weaker during the winter season measurement periods.

Lagoon temperatures for Barham Farm ranged from 6°C to 24°C and 11°C to 19°C during the April and November 2002 experimental periods, respectively. At Howard farm, lagoon temperatures ranged from 20°C to 31°C for the June 2002 measurement campaign and 4°C to 10°C for the December 2002 measurement campaign. Lagoon temperatures for Stokes Farm ranged from 24.7°C to 33.6°C and 4.8°C to 9.3°C during the September 2002 and January 2003 experimental periods, respectively. At Moore farm, lagoon temperatures ranged from 20°C to 30°C for the October 2002 measurement campaign and 1.6°C to 12.0°C for the January-February 2003 measurement campaign.

For EST sites, pH values ranged from 7.14 to 8.47 at Barham Farm for the April 2002 experimental period and for the June 2002 experimental period at Howard Farm, pH values ranged from 6.84 to 8.29 (Arkinson, 2003). For the November 2002 measurement campaign at Barham Farm, lagoon pH values ranged from 8.32 to 8.85 and at Howard Farm for December 2002, pH values ranged from 7.78 to 9.10. For LST sites, lagoon pH values ranged from 7.43 to 8.35 and from 8.06 to 8.46 at Stokes Farm for the fall and winter season experimental periods, respectively. At Moore farm, lagoon pH ranged 7.86 to 9.12 and from 7.85 to 8.25 for the fall and winter season measurement campaigns.

Lagoon pH has an important role in the potential for ammonia volatilization. In a lagoon, pH is the primary controller of the direction of equilibrium for NH_3 in solution with NH_4^+ as shown in the following equation (Warneck, 2000):

$$NH_3(aq) + H_2O\ (aq) \leftrightarrow NH_4^+(aq) + OH^-(aq) \quad (1)$$

An increase in the pH of a lagoon will increase hydroxyl ion concentrations [OH^-] which shifts the equilibrium to the left and leads to more ammonia being released into the atmosphere. On the other hand, an increase in the water content of the lagoon, such as precipitation events, shift the equilibrium to the right, causing ammonia to be more tightly bound or stifling ammonia volatilization.

Waste lagoon slurry, however, has a high buffer capacity which limits lagoon pH to a relatively narrow range (Aneja et al., 2001a; Aneja et al., 2000, Bunton 1999). As ammonium ions (NH_4^+) dissociates, it releases surplus hydrogen ions (H^+) which is shown in the following equation:

$$NH_4^+(aq) \leftrightarrow NH_3(aq) + H^+(aq) \quad (2)$$

The pH in a lagoon is neutralized by bicarbonate ions which are proton acceptors for hydrogen ions released into solution by ammonium dissociation as shown in Equation 3. Bicarbonate ions are formed as a product of the hydrolysis of urea and microbial conversion of organic matter.

$$CO_2(aq) + H_2O(aq) \leftrightarrow HCO_3^-(aq) + H^+(aq) \quad (3)$$

The pH of a waste lagoon slurry can undergo several changes. When fresh slurry is added to a lagoon, higher concentrations of inorganic carbon are present compared to [NH_4^+] and [NH_3] (Oleson and Sommer, 1993). Since the water solubility of ammonia is much higher than that of carbon dioxide, pH will increase as carbon dioxide volatilizes from the lagoon

surface at a faster rate than ammonia. When carbon dioxide concentrations in the lagoon decline, lagoon pH will increase and gaseous ammonia will escape into the atmosphere. As more gaseous ammonia continues to escape into the atmosphere, pH will decrease (Bunton, 1999).

Measurements of TAN (NH_3-N + NH_4^+-N) concentration from Barham Farm, during the April 2002 measurement campaign, ranged from 371 to 551 mg N/l, while TAN concentrations ranged from 292 to 401 mg N/l during the November 2002 field measurement campaign. Measurements of TAN concentration from Howard Farm ranged from 33 to 306 mg N/l and 51 to 399 mg N/l for summer and winter measurement periods, respectively. Measurements of TAN concentration from Stokes Farm ranged from 410 to 477 mg N/l for the September 2002 experimental period and from 545 to 593 mg N/l for the January 2003 experimental period. For Moore Farm, TAN concentrations ranged from 316 to 409 mgN/l and 545 to 665 mgN/L for the October 2002 and Jan.-Feb. 2003 field experimental periods, respectively. Lagoon TAN samples were collected daily and analyzed by the BAE laboratory. The number of samples collected for lagoon TAN were limited for each field campaign, therefore statistical analysis was not performed on this parameter. At steady-state farm conditions, a conventional LST site would have a balance of nitrogen inflow and outflow. For an EST site, this balance may differ from an LST site due to various biological, chemical and physical processes that could potentially alter the nitrogen cycling of the technology systems.

Two conventional lagoon and spray technology (LSTs) or baseline sites for handing animal waste were chosen for a comparative evaluation of ESTs. The baseline farm measurements were also conducted in two different seasons during the fall and winter

(September – October, 2002; January – February, 2003) measurement campaigns (recently, two more late spring/summer campaigns have also been conducted at Moore farm).

It has been suggested that lagoon temperature is the most important parameter regulating the loss of NH_3 from the hog waste lagoons to the atmosphere (Aneja et al., 2000). The pH (range from 7.0 to 8.5) and TAN (range from 300 to 750 ml/l) are relatively constant in most North Carolina hog waste lagoons. For this study, no statistical significant correlation was found between ammonia flux and pH or TAN for the two baseline farms. Therefore, lagoon temperature, usually varying diurnally and seasonally, is considered the most important environmental parameter to estimate NH_3 flux, using an empirical regression model. A more fundamental mass transfer coupled with chemical reaction model indicates that wind speed, lagoon pH and lagoon TAN may also have strong influence on NH_3 flux from a lagoon (Arkinson, 2003). It has been applied to both the baseline and EST farms to compare and contrast the results. In order to evaluate an alternative technology with lagoon and storage ponds, using the algorithm depicted in the flow chart of Figure 2.7, a multiple regression relationship between NH_3 flux and lagoon temperature and the difference between air and lagoon temperatures was developed over a relatively wide range of lagoon temperatures (~2^{o}C to 35^{o}C) observed during the field campaigns of fall and winter months at both baseline farms.

First, a simple linear regression between $\log_{10}$(Ammonia Flux) and lagoon temperature; $\log_{10}$ Flux = 0.0516 T_{lagoon} + 1.9433 (R^2 = 0.82 and $p < 0.0001$) was found as illustrated in Figure 3.8. Each data point in Figure 3.8 represents a 15-min averaged NH_3 flux plotted against the corresponding 15-min lagoon temperature at both baseline farms during experimental periods Figures 3.9 and 3.10 also show strong, but different linear relationships

between $\log_{10}$(Ammonia Flux) and lagoon temperature for ESTs at Barham Farm ($R^2 = 0.78$, $p < 0.0001$) and Howard Farm ($R^2 = 0.88$, $p < 0.0001$). Each data point in Figures 3.9 and 3.10 represents log of hourly averaged NH_3 flux plotted against the corresponding hourly averaged lagoon temperature at ESTs. Reasons for the exponential relationship, between NH_3 flux and lagoon temperature, are that the liquid phase mass transfer coefficients of NH_3 in water are exponential functions of temperature in the range 5°C to 30°C (Ibusuki and Aneja, 1984), and the dependence of Henry's law on temperature (Dasgupta and Dong, 1986). The transfer of NH_3 across the liquid-gas interface, thus, follows an exponential model; and the flux increases exponentially with surface lagoon temperature. An increase in lagoon temperature causes the rate of decomposition of waste sludge from the bottom of the lagoon to increase. As ammonia volatilization to the atmosphere occurs from the surface of the lagoons, ammonia is replenished by upward diffusion from the decomposition of sludge from the bottom of the lagoon (Muck and Steenhuis, 1982).

Based on the waste storage and treatment lagoon measurements at Farm 10 (also lagoon and spray system) in eastern North Carolina over one year, Aneja et al. (2000) reported an ammonia flux observational model in terms of lagoon temperature, $\log_{10}$ Flux = 0.048 T_{lagoon} + 2.1, which is also plotted in Figure 3.8, and shows very good agreement with the regression relation based on baseline measurements from this study, with almost identical fluxes at higher lagoon temperatures. Both models seem to underestimate large fluxes at high lagoon temperatures ($T_{lagoon} > 30$°C). Figure 3.8 also shows larger variability of NH_3 flux during the cold season (lagoon temperature from 2 to 10°C).

Large scatter in the data around the best-fitted regression relationship suggests that there could be other environmental parameters, such as air temperature, or difference

between air and lagoon temperature (ΔT), lagoon pH, etc., to affect loss of NH_3 from waste lagoon, especially during cold season. In order to investigate the relative contributions of these parameters to a multiple regression relationship for the $\log_{10}$(NH_3-N flux) at baseline farms, 15-minute averages of ($\log_{10}$ NH_3-N flux) and corresponding lagoon temperature (T_L), air temperature (T_a), or the difference $\Delta T = T_a - T_L$, and lagoon pH were included and statistically analyzed in a multiple linear regression. The 15-min. averaged measured flux over a lagoon was first multiplied by the lagoon surface area to get the emission rate and then normalized by the appropriate live animal weight (LAW) for the baseline farm; this is called *A flux/ton*, where A is the lagoon surface area and *ton* represents a metric ton or 1000 kg. Figure 3.11 shows a plot of log (*A flux/ton*) versus the difference ($\Delta T = T_{air} - T_{lagoon}$) between the air temperature and the lagoon temperature. For reference, a vertical line is plotted at ΔT=0, the point where air and lagoon temperatures are equal. To the left of the ΔT = 0 line in Figure 3.11, the four color categories, each representing a range of lagoon temperatures, appear in more or less horizontal bands with higher emissions corresponding to higher lagoon temperatures (red, cyan) and lower emissions to lower lagoon temperatures (blue, black). This indicates that when ΔT < 0 (air temperature < lagoon temperature), $\log_{10}$ (*A*flux/ton*) is determined largely by lagoon temperature, i.e., air temperature has no effect on log (*A*flux/ton*). However, the characteristics of the plot to the right of the line ΔT = 0 (air temperature > lagoon temperature) are strikingly different. When ΔT > 0 the colors have the same ordering as on the left, indicating that lagoon temperature still has the expected effect (log (*A*flux/ton*) higher for higher lagoon temperature). But the apparent downward slope to the right of ΔT = 0 indicates that log (*A*flux/ton*) decreases approximately linearly with ΔT in all four lagoon temperature groups. In other words, if air temperature is less than lagoon

temperature, then log (*A*flux/ton*) is solely dependent on lagoon temperature (independent of air temperature); however, when air temperature exceeds lagoon temperature ($\Delta T > 0$), log (*A*flux/ton*) decreases by an amount approximately proportional to the amount by which air temperature exceeds lagoon temperature (in addition to its dependence on lagoon temperature) (D. Dickey and L. Stefanski, 2003, Personal communication).

The air-lagoon temperature difference, ΔT, may be considered as a simple measure of atmospheric stability, which is found to strongly influence surface–air turbulent exchanges/fluxes including that of ammonia from a lagoon surface. When $\Delta T < 0$, unstable conditions enhance turbulence and lead to large fluxes. On the other hand, stable or inversion conditions with $\Delta T > 0$ represent weaker turbulence and smaller fluxes. The air-surface temperature difference is the simplest measure of the stability of the atmospheric surface layer, for which a variety of parameters are used in micrometeorology and air pollution meteorology (Arya, 1999, 2001). Better measures include wind shear or wind speed, such as the bulk Richardson number $R_{iB} = (g\, z_r / T_0)(\Delta T + 0.01\, z_r) / U^2$, where $z_r = 10$m which is the reference height for wind speed U and air temperature measurements, and T_0 is the surface temperature in absolute units. Thus, R_{iB} depends on both ΔT and U, but in a different manner. Very unstable and convective conditions are characterized by large $-\Delta T$ and weak winds, near-neutral conditions by small $|\Delta T|$ and strong winds, and very stable conditions by large ΔT and weak winds (S. P. Arya, 2003, Personal communication).

According to the well-known bulk-transfer relations used for estimating surface fluxes (Arya, 2001), the ammonia flux from a lagoon surface can be expressed as a product of the wind speed at a reference height (say, 10m), the difference in ammonia concentrations at the surface and the reference height, and a dimensionless mass transfer coefficient C_M. Judging

from the behavior of the transfer coefficients, for heat and water vapor, C_M is expected to depend on atmospheric stability, monotonically decreasing with increasing stability (R_{iB}). In particular, C_M decreases rapidly with increasing R_{iB} for $R_{iB} > 0$. This explains the stronger dependence of ammonia flux on ΔT for $\Delta T > 0$, as shown in Figure 3.11. The weaker or lack of dependence of flux on ΔT for neutral and unstable conditions ($\Delta T < 0$) can be explained by the opposing effects of wind speed and C_M on flux. Note that, under unstable conditions, as wind speed decreases, C_M is expected to increase (S.P. Arya, 2003, Personal communication).

Table 3.4 summarizes the statistical findings from multiple regression analysis for all data point representing 15-min averaged $\log_{10}$(A*flux/ton) against the corresponding 15-min averaged lagoon temperature and ΔT at both baseline farms during the four experimental periods (D. Dickey and L. Stefanski, 2003, Personal communication). The dependent variable, $\log_{10}$(A*flux/ton), was found to be mainly dependent on lagoon temperature (T_L) and the 'hot air' variable D, defined such that D = 0, when the lagoon is warmer than the air ($\Delta T \leq 0$) and D = ΔT, when the air is hotter than the lagoon ($\Delta T > 0$). The model also incorporates a farm indicator variable (F_i) in order to account for differences in responses to lagoon temperature and ΔT between farms. For the purposes of the multiple regression analysis, $F_i = 0$ for Stokes Farm and $F_i = 1$ for Moore Farm. The statistical analysis also accounts for autocorrelation since most of the data points are in consecutive 15 minute intervals and not taken as random samples.

The multiple regression analysis showed a high correlation ($R^2 = 0.95$) between ammonia flux and the regressors, which were lagoon temperature (T_L), 'hot air' variable (D) and the farm indicator variable (F_i). The overall response of ammonia flux between farms

differed and was statistically significant (t=646, p<0.0001) as indicated by the farm variable. The multiple regression also shows statistically significant relationships between ammonia flux and lagoon temperature (t=166, p<0.0001) and ammonia flux and D (t=-55, p<0.0001). The multiple regression equations for lagoon emissions at Stokes (top) and Moore (bottom) farms are given below:

$$\text{Log}_{10}(\text{A*flux/ton}) = 3.6264 + .04491(T_L) - .05946(D) \quad (4)$$

$$\text{Log}_{10}(\text{A*flux/ton}) = 3.6264 + 0.4782(F_i) + .04491(T_L) - .05946(D) \quad (5)$$

The multiple regression equation for average lagoon emissions at the two baseline farms is given below:

$$\text{Log}_{10}(\text{A*flux/ton}) = 3.8655 + 0.0449*T_L - 0.05946*D \quad (6)$$

In order to statistically analyze the relative effect of lagoon pH, multiple regression analysis for all data points representing 15-min averaged $\log_{10}$(A*flux/ton) plotted against the corresponding 15-min lagoon temperature, ΔT, and lagoon pH at both baseline farms during the four experimental periods (D. Dickey and L. Stefanski, 2003, Personal communication). Table 3.5 summarizes the findings of this statistical analysis. The dependent variable, $\log_{10}$(A*flux/ton), was still found to be mainly dependent on lagoon temperature (T_L) ($p < 0.0001$) and the 'hot air' variable D ($p < 0.0001$) but was found to be statistically insignificant for lagoon pH ($p < 0.1098$). The statistical analysis also accounted for autocorrelation since most of the data points are in consecutive 15 minute intervals and not collected as random samples.

3.5 Comparison of Ammonia Emissions from EST and LST Farms

The comparison of total NH_3-N emissions (kg-N/yr/1000 kg-LAW) and relative emissions of NH_3-N (% E) from the two ESTs with those from the appropriate baseline farms

are summarized in Table 3.6. This analysis included hog population, weight, %N in feed, and N-excretion at baseline and EST farms. Potential losses of NH_3-N from lagoons, barns, and spray fields were also evaluated individually for Barham and Howard farms. Soil NH_3-N emissions or from spray fields were found to be below the detectable limit of the chamber or Model 17C Ammonia Analyzer used for measuring NH_3 concentrations in the chamber, which was < 2 ppb at 200 ppb measurement range for all the measurement periods for LSTs and ESTs. This was because lagoon effluent was not sprayed on the field shortly before or during any of the measurement periods.

An evaluation of Barham farm, which is summarized in Table 3.7, shows that the measured lagoon NH_3-N emissions relative to N-excretion (%E) was 18.8%, as compared to the relative emissions projected from the Baseline Model which was 11.3% for the April 2002 measurement campaign. For the November 2002 measurement period, however, the projected relative emission from the Baseline Model was higher at 9.6%, compared to 3.9% from Barham farm. For the April 2002 experimental period, the relative barn emissions from Barham Farm were slightly lower at 20.6%, compared to 23.9% relative barn emissions from Moore Farm (fan-ventilated barnhouses). For the November 2002 measurement period, the relative barn emissions from Barham Farm were higher (27.9%) than those from Moore Farm (22.8%). The total lagoon and barn emissions, as % of E, were about the same at Barham Farm (39.4 ± 6.8%) and from the Baseline Model (40.8 ± 7.0%) during the April 2002 measurement period. For the cold season measurement period in November 2002, % E for Barham Farm was lower at 31.8 ± 3.9% compared to 37.1 ± 6.4% for baseline farms. Thus, the EST at Barham Farm was effective in reducing NH_3-N from lagoon (storage and overflowing ponds) surfaces, only during the second (cold season) experimental period. It is

likely that the EST did not attain steady-state during the first (April 2002) experimental period.

The evaluation of Howard farm, which is summarized in Table 3.8, shows that lagoon NH_3-N emissions relative to N-excretion (%E) were 32.1% and 22.6% for Howard Farm and Baseline Model projection, respectively, for the June 2002 measurement campaign. For the December 2002 measurement period, these relative lagoon emissions were 9.8% for Howard Farm and 3.9% for baseline farms. For the June 2002 experimental period, barn NH_3-N emissions from Howard Farm also had a higher relative emission of 38.0% compared to 23.9% from Moore Farm (fan-ventilated barnhouses). For the December 2002 measurement period, the relative barn emissions were 33.4% from Howard Farm and 22.8% from Moore farm. The total lagoon and barn relative emissions from June 2002 measurements at Howard Farm were significantly higher (70.1 ± 12.2%) compared to 52.2 ± 8.3% for baseline emissions. For December 2002, the total relative emission were reduced at both EST and baseline sites, but emissions at Howard Farm were still higher at 43.3 ± 7.2% compared to 31.4 ± 6.0% for baseline farms. Thus, the EST at Howard Farm was not effective in reducing ammonia emissions; but it appeared to enhance them.

Except for the November 2002 measurement period at Barham farm, lagoon NH_3-N emissions relative to N-excretion were higher for ESTs compared to the baseline farms. Barnhouse emissions of NH_3-N relative to N-excretion were about the same or higher than those from baseline farms. The comparison of total relative emissions at both EST and baseline sites shows that only Barham Farm during the November 2002 measurement campaign, showed significantly lower NH_3-N emissions when compared to the baseline modeled and measured emissions during the same season.

3.6 Emission Factors

LST sites

Emission factors for Stokes and Moore farms were estimated for the liquid waste portions and the barn housing system of the waste technology systems. Table 3.9 provides a summary of emission factors for liquid waste storage systems at LST and EST sites for fall and winter seasons. The ratio of nitrogen losses from lagoons or swine liquid waste storage systems to the total nitrogen excretion available was used to calculate a % NH_3-N loss. To make estimations of emission factors, each of the ammonia flux averages from storage ponds, overflow ponds, finishing ponds, and wetland cells were assumed to be constant across the liquid surfaces. Figure 3.12 illustrates the seasonal variation of % NH_3-N loss from swine liquid waste storage systems for LST and EST sites.

The storage ponds at Stokes and Moore farms had surface areas of 15,170 m^2 and 30,630 m^2. Stokes Farm had 4,391 finishing hogs for the September 2002 measurement period and 3,726 finishing hogs for the January 2003 measurement period. Moore Farm had 7,617 finishing hogs for the October 2002 measurement period and 5,784 finishing hogs for the January-February 2003 measurement period. For the fall season measurement campaigns, the average ammonia fluxes for the storage ponds were 2,362 μg NH_3-N/m^2/min for Stokes Farm during the September 2002 campaign and 1,667 μg NH_3-N/m^2/min for Moore Farm during the October 2002 measurement period. For winter season measurement campaign, average ammonia flux measurements were 155 and 332 μg NH_3-N/m^2/min for the storage ponds at Stokes and Moore farms, respectively. Stokes Farm was measured in January 2003, while Moore Farm was measured in Jan-Feb 2003. During the fall season campaigns, Stokes Farm emitted an estimated 0.0358 kg NH_3-N/min from its storage pond, while Moore Farm

emitted an estimated 0.0511 kg NH_3-N/min. During the winter season campaigns, Stokes Farm emitted an estimated 0.0024 kg NH_3-N/min from its storage pond, while Moore Farm emitted an estimated 0.0102 kg NH_3-N/min. Stokes Farm had a total pig weight of 457,372 kg for the September 2002 measurement period and 329,661 kg for the January 2003 measurement period. Moore Farm had 397,711 kg for the October 2002 measurement period and 387,771 kg for the February 2003 measurement period. The estimated emission factors for the early and middle fall season were 41.1 and 67.5 kg NH_3-N/1000 kg-LAW/year for Stokes and Moore farms, respectively. The estimated emission factors for the winter season were 3.8 and 13.8 kg NH_3-N/1000 kg-LAW/year for Stokes and Moore farms, respectively. Normalizing by the rate of nitrogen excretion for each farm in different seasons, NH_3-N losses from storage ponds at Stokes Farm were 29.2% and 2.9% for fall and winter seasons measurements, respectively. For Moore farm, % NH_3-N losses from storage ponds were 29.6% and 6.8% of the total nitrogen excretion (N waste available) for the fall and winter seasons experimental periods, respectively. For the two baseline farms, winter season relative losses of NH_3-N (%E) from lagoons were 26% and 23% less than to the warmer season measurement periods for Stokes and Moore farms, respectively. Thus, NH_3-N emissions from conventional lagoons strongly depend on the season, with highest values expected to occur in the summer and lowest in winter.

The estimated emission factors for LST sites were compared with published scientific studies summarized in Table 3.10 (Aneja et al., 2000; Aneja et al., 2001b; Todd et al., 2001; Harper et al., 2000;; Van der Hoek, 1998). These are based on a recent review by Griffing et al. (2003). Griffing et al. (2003) calculated relative %N loss based on TKN entering the lagoon, while this study calculated %N loss based on total nitrogen excretion available.

Aneja et al. (2000) and Aneja et al. (2001b) used the dynamic flow-through chamber to measure ammonia emissions from lagoons in North Carolina. Average fluxes were interpolated from a best fit equation that used the average difference between lagoon and air temperatures to integrate the yearly flux. Aneja et al. (2000) had an integrated yearly loss rate of 20% with actual measurements taken during various times of the year. December and February relative losses of NH_3-N from lagoons for Aneja et al. (2000) were 10% and 4%, respectively, which are similar to relative LST emissions of NH_3-N from lagoons for the cold season for this study. An average percent N loss of 2% based on TKN input was calculated for Aneja et al. (2001b) from two research lagoons and two lagoons on commercial farms. Ammonia flux measurements by Aneja et al. (2001b) were conducted from smaller lagoons (~3600 to 7800 m^2), while measurements of Aneja et al. (2000) were conducted from a larger lagoon (25,000 m^2), which was more representative of the lagoon surface areas that were researched for this study. Todd et al. (2001) made measurements of ammonia emission using the OP-FTIR technique at the same farm during the same year as Aneja et al. (2000). Ammonia emission measurements for the same month (May, 1998) from Todd et al. (2001) and Aneja et al. (2000), using different methods differed considerably. The average ratio (Todd et al. 2001/Aneja et al. 2000) of emissions measured by their respective methods was found to be 2.55. Using this ratio, Griffing et al. (2003) estimated that measurements by Todd et al. (2001) accounted for an average of 51% N loss of incoming lagoon TKN, which was higher than measurements taken from this study for the fall and winter measurements periods. Harper et al. (2000) employed the micrometeorological technique to measure ammonia flux by collecting ammonia concentrations and wind speed at several heights above the lagoon surface to calculate fluxes. They also measured dinitrogen (N_2), nitrous oxide

(N_2O), and nitrate fluxes from an anaerobic lagoon. Emissions of NH_3-N relative to lagoon TKN estimated for Harper et al. (2000) were 40% with a high estimated error of 56%. A review article, based on European estimates by Van der Hoek (1998), estimated an average percent loss of 18% from lagoons. Recent ammonia flux measurements from Moore Farm (LST) conducted for the late spring and summer seasons of 2003 should provide more emission factors for future comparisons with published literature of NH_3-N loss from lagoons.

Variations in emission factors between swine agricultural farm operations could be due to differing waste treatment technologies, animal management, and feeding practices, environmental parameters and experimental error. Potential sources of error in calculating ammonia emissions based on mass balance approach include (Griffing et al., 2003):

• Determining how much waste enters the lagoon (N excretion) ~7% error for N/LAW

• Steady-state assumption of lagoons

• Additional emissions of N_2 or other nitrogen containing compounds

EST sites

The storage and overflow ponds at Barham Farm had surface areas of 4,459 m^2 and 19,398 m^2, respectively. For the April 2002 measurement campaign, the average ammonia fluxes were 1,102 μg NH_3-N/m^2/min from the storage pond and 1,249 μg NH_3-N/m^2/min from the overflow pond. For the November 2002 measurement campaign, average ammonia flux measurements were 436 and 234 μg NH_3-N/m^2/min for the storage and overflow ponds, respectively. During the spring season campaign, both the storage and overflow ponds emitted an estimated .0291 kg NH_3-N/min while in the fall season the same ponds emitted an estimated 0.0065 kg NH_3-N/min. Barham Farm had 4,000 finishing hogs for both

measurement periods. The estimated emission factors of the liquid storage systems for the spring and fall seasons were 16.1 and 3.6 kg NH_3-N/1000 kg-LAW/yr, respectively. For Barham Farm, %NH_3-N losses from storage and overflow ponds were 18.8% and 3.9% of the total nitrogen excretion (N waste available) for the spring and winter experimental periods, respectively. For Barham farm, November 2002 relative emissions of NH_3-N (%E) from the storage and overflow ponds were around 15% less compared to those during the warmer April 2002 experimental period.

The wetland cells and finishing pond at Howard Farm had surface areas of 29,591 and 7,428 m^2, respectively. Average measured ammonia fluxes were 914 μg NH_3-N/m^2/min from the wetland cells and 434 μg NH_3-N/m^2/min from the finishing pond for the June 2002 field campaign and 289 and 152 μg NH_3-N/m^2/min for the wetland cells and finishing ponds, respectively, for the December 2002 campaign. During the summer season campaign, both the wetland cells and finishing ponds emitted an estimated 0.0276 kg NH_3-N/min, while in the winter season the same ponds emitted an estimated 0.0094 kg NH_3-N/min. Howard Farm had 3,618 finishing hogs for the spring season and 3,881 finishing hogs for the winter season. The estimated emission factors of the liquid storage systems for the summer and winter seasons were 62.2 and 13.0 kg NH_3-N/1000 kg-LAW/yr, respectively. This accounted for 32.0% and 9.0% of NH_3-N emissions from the wetland cells and finishing pond relative to the nitrogen excretion (N waste available) for the spring and winter measurement periods, respectively. For Howard farm, December 2002 relative losses of NH_3-N (%E) from the wetland cells and finishing pond were about 22% less compared to the warm June 2002 experimental period.

Table 3.1 Climatological comparison between 10-year monthly normals and experimental period months for EST and LST sites.

	Research Experimental Sites	**Precipitation** (cm)		**Air Temperature** (^{o}C)	
	(State Climate Office of North Carolina ECONET or NWS/CON site closest to field experiment)	10-year Monthly Averaged Normals	Average for Experimental Month	10-year Monthly Averaged Normals	Average for Experimental Month
ESTs	**Barham Farm—April 2002** (Clayton station)	7.3	4.1	15.4	17.3
	Barham Farm—November 2002 (Clayton station)	8.3	9.0	11.0	9.3
	Howard Farm—June 2002 (Hoffman Forest station)	15.2	9.4	24.4	24.6
	Howard Farm—December 2002 (Hoffman Forest station)	7.9	7.6	8.8	9.3
LSTs	**Stokes Farm—September 2002** (Greenville station)	18.5	11.9	22.4	24.2
	Stokes Farm—January 2003 (Greenville station)	12.2	18.3	6.3	6.9
	Moore Farm—October 2002 (Kinston station)	9.0	6.6	16.1	18.6
	Moore Farm—Jan-Feb. 2003 (Hoffman Forest station)	11.4 and 7.4 respectively	8.4 and 9.4 respectively	6.1 and 7.4 respectively	5.5 and 7.2 respectively

Table 3.2 Mean wind speed, wind direction and air temperature data for all sampling sites.

Research Locations	Wind Speed	Wind Direction	Air Temperature
	(m/sec)	(degrees)	(^{o}C)
Barham Farm (April 2002)	1.9 ± 0.9	188 ± 120	14.8 ± 4.4
Barham Farm (November 2002)	1.5 ± 1.1	266 ± 75	10.3 ± 5.3
Howard Farm (June 2002)	2.2 ± 1.4	156 ± 105	23.3 ± 5.2
Howard Farm (December 2002)	2.4 ± 1.5	252 ± 100	4.2 ± 4.3
Stokes Farm (September 2002)	2.0 ± 1.5	143 ± 95	24.1 ± 4.1
Stokes Farm (January 2003)	3.3 ± 2.3	232 ± 41	4.6 ± 5.8
Moore Farm (October 2002)	1.5 ± 1.1	142 ± 87	23.5 ± 4.4
Moore Farm (Jan.-Feb. 2003)	2.3 ± 1.7	277 ± 93	6.8 ± 5.5

Table 3.3 Mean NH_3-N flux measurements and lagoon parameters for all sampling sites.

Research Locations	Subsection	Ammonia Flux	Lagoon Temperature	Lagoon pH	Lagoon TAN
		μg NH_3-N/m^2/min	(^oC)		(mg N/l)
Barham Farm (April 2002)	**Overall**	**1169 ± 519**	**17.2 ± 3.7**	**8.09 ± 0.18**	**428 ± 56**
	Storage Pond	1102 ± 640	18.5 ± 2.5	7.97 ± 0.06	--
	Overflow Pond	1249 ± 175	16.6 ± 4.6	8.26 ± 0.13	--
Barham Farm (Nov. 2002)	**Overall**	**352 ± 27**	**14.2 ± 1.9**	**8.70 ± 0.02**	**347 ± 45**
	Storage Pond	436 ± 39	15.2 ± 0.6	8.64 ± 0.10	--
	Overflow Pond	234 ± 34	12.5 ± 0.6	8.80 ± 0.02	--
Howard Farm (June 2002)	**Overall**	**790± 336**	**24.6 ± 1.0**	**7.70 ± 0.25**	**115 ± 90**
	Wetland Cells	914 ± 437	24.1 ± 1.7	7.71 ± 0.28	--
	Finishing Pond	434 ± 136	27.5 ± 1.8	7.67 ± 0.00	--
Howard Farm (Dec. 2002)	**Overall**	**243 ± 29**	**6.2 ± 2.1**	**8.69 ± 0.05**	**195 ± 110**
	Wetland Cells	289± 47	6.9 ± 1.8	8.43 ± 0.41	--
	Finishing Pond	152 ± 8	6.5 ± 0.4	9.08 ± 0.01	--
Stokes Farm (Sep. 2002)	**Overall**	**2385 ± 565**	**26.6 ± 1.0**	**8.07 ± 0.02**	**442 ± 18**
Stokes Farm (Jan. 2003)	**Overall**	**153 ± 19**	**7.2 ± 0.4**	**8.41 ± 0.01**	**560 ± 19**
Moore Farm (Oct. 2002)	**Overall**	**1657 ± 207**	**25.0 ± 1.1**	**8.20 ± 0.01**	**364 ± 36**
Moore Farm (Jan.-Feb. 2003)	**Overall**	**325 ± 45**	**6.8 ± 0.6**	**8.07 ± 0.01**	**636 ± 37**

Table 3.4 Statistical summary of multiple regression analysis of ammonia flux measurements from Stokes and Moore Farms (baselines) (D. Dickey and L. Stefanski, 2003, Personal communication).

Baseline farms with 'Hot air' variable
Area Adjusted

The GLM Procedure

Dependent Variable: Lflux log10(A*flux/(tons wt))

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	3	821.2709351	273.7569784	14905.6	<.0001
Error	2911	53.4636834	0.0183661		
Corrected Total	2914	874.7346185			

R-Square	Coeff Var	Root MSE	Lflux Mean
0.938880	3.006981	0.135522	4.506897

Source	DF	Type I SS	Mean Square	F Value	Pr > F
farm	1	208.7160632	208.7160632	11364.2	<.0001
Tlagoon	1	556.7377574	556.7377574	30313.4	<.0001
'Hot air'	1	55.8171144	55.8171144	3039.14	<.0001

Parameter		Estimate	Standard Error	t Value	Pr > |t|
Intercept		3.626352543 B	0.00561299	646.06	<.0001
farm	Moore	0.478227645 B	0.00504468	94.80	<.0001
farm	Stokes	0.000000000 B	.	.	.
Tlagoon		0.044912075	0.00027042	166.08	<.0001
'Hot air'		-0.059456319	0.00107851	-55.13	<.0001

NOTE: The X'X matrix has been found to be singular, and a generalized inverse was used to solve the normal equations. Terms whose estimates are followed by the letter 'B' are not uniquely estimable.

Table 3.5 Statistical summary of multiple regression analysis of ammonia flux measurements from Stokes and Moore farms (Baselines) with lagoon pH (D. Dickey and L. Stefanski, 2003, Personal communication).

AUTOREGRESSION

Variable	DF	Standard Estimate	Approx Error	t Value	Pr > |t|
Intercept	1	3.4434	0.3446	9.99	<.0001
Stokes	1	-0.6559	0.0350	-18.74	<.0001
Fall	1	-0.6657	0.0814	-8.17	<.0001
Stokes Fall	1	0.3442	0.0448	7.68	<.0001
Tlagoon	1	0.0702	0.003759	18.68	<.0001
D 'hot air'	1	-0.0347	0.002413	-14.37	<.0001
Lagoon pH	1	0.0682	0.0427	1.60	0.1098

Table 3.6 Summary of animal weight, feed consumed N-excretion and NH_3-N emissions at baseline (Stokes and Moore) and EST (Barham and Howard) Farms.

Farm	Number of hogs	Average hog weight (kg/hog)	Total hog weight (kg)	Feed consumed (kg/pig/yr)	N-excretion, E (kg-N/yr/ton-LAW)
Stokes (September 2002)	4,392	104.3	457,372	667.8	**140.7**
Stokes (January 2003)	3,727	88.5	329,661	667.8	**133.2**
Moore (October 2002)	7,611	52.3	397,711	571.5	**228.3**
Moore (Jan.-Feb. 2003)	5,784	67.0	387,771	643.1	**202.8**
Barham (April 2002)	4,000	238.1	952,560	827.8	**85.6**
Barham (November 2002)	4,000	238.1	952,560	827.8	**92.0**
Howard (June 2002)	3,618	64.3	232,777	688.5	**194.0**
Howard (December 2002)	3,881	96.8	375,762	810.1	**131.8**

Total NH_3-N Emissions from Lagoons, Barns and Land Application at ESTs		N-excretion by Mass Balance at EST, E	Total Baseline Model Emission (F_{proj})[a]	% of E_{base}	Total Measured Emission (F_{meas})[b]	% of E_{EST}	Δ(%E) = (%E_{base})-(%E_{EST}) (if Δ(%E)>0, then reduction in NH_3-N)
BARHAM FARM	**April 2002**	**85.6**	**75.2**	**40.8 ± 7.0**	**33.7**	**39.4 ± 6.8**	**1.4 ± 9.8**
	November 2002	**92.0**	**62.3**	**37.1 ± 6.4**	**29.3**	**31.8 ± 3.9**	**5.3 ± 7.5**
HOWARD FARM	**June 2002**	**194.0**	**96.2**	**52.2 ± 8.3**	**136.0**	**70.1 ± 12.2**	**-17.9 ± 14.8**
	December 2002	**131.8**	**52.7**	**31.4 ± 6.0**	**57.0**	**43.3 ± 7.2**	**-11.9 ± 9.4**

[a] used average of Warm Season or Cold Season total hog weight to calculate F_{proj} which has units of [kg-N/yr/1000 kg-LAW].

[b] Used Barham total hog weight to calculate F_{meas} which has units of [kg-N/yr/1000 kg-LAW].

Table 3.7 Evaluation of the total emissions of NH_3-N from Barham Farm from different components of the EST.

Emission Sources at Barham Farm		EST Average Lagoon Temperature (^oC)	EST Average ΔT Temperature (^oC)	Baseline Model Emission (F_{proj})	% of E	Barham Measured Emission (F_{meas})	% of E
LAGOON EMISSIONS	Apr. 2002	17.2	0.7	20.8	11.3	16.1	18.8
	Nov. 2002	14.2	0.3	16.1	9.6	3.6	3.9
BARN EMISSIONS	Apr. 2002	---	---	54.4[a]	23.9	17.6	20.6
	Nov. 2002	---	---	46.2[b]	22.8	25.7	27.9
LAND APPLICATION	Apr. 2002	---	---	0[c]	0[c]	0[c]	0[c]
	Nov. 2002	---	---	0[c]	0[c]	0[c]	0[c]

Total NH_3-N Emissions from Lagoons, Barns and Land Application	N-excretion by Mass Balance at Barham Farm, E	Total Baseline Model Emission (F_{proj})[d]	% of E_{base}	Total Barham Measured Emission (F_{meas})[e]	% of E_{EST}
April 2002 (Warm)	**85.6**	**75.2**	**40.8 ± 7.0**	**33.7**	**39.4 ± 6.8**
November 2002 (Cold)	**92.0**	**62.3**	**37.1 ± 6.4**	**29.3**	**31.8 ± 3.9**

[a] barnhouse emissions measured at Moore Farm during October 2002.

[b] barnhouse emissions measured at Moore Farm during Jan-Feb 2003.

[c] soil emissions were below the Model 17C Ammonia Analyzer detection limit of 2 ppb @ 200 ppb range.

[d] used average of Warm Season or Cold Season total hog weight to calculate F_{proj} which has units of [kg-N/1000 kg-LAW/yr].

[e] Used Barham total hog weight to calculate F_{meas} which has units of [kg-N/yr/1000 kg-LAW].

Table 3.8 Evaluation of the total emissions of NH_3-N from Howard Farm from different components of the EST.

Emission Sources at Howard Farm		EST Average Lagoon Temperature (^oC)	EST Average ΔT Temperature (^oC)	Baseline Model Emission (F_{proj})	% of E	Howard Measured Emission (F_{meas})	% of E
LAGOON EMISSIONS	Jun. 2002	24.6	1.2	41.7	22.6	62.2	32.1
	Dec. 2002	6.2	0.9	6.47	3.85	13.0	9.8
BARN EMISSIONS	Jun. 2002	---	---	54.4[a]	23.9	73.8	38.0
	Dec. 2002	---	---	46.2[b]	22.8	44.1	33.4
LAND APPLICATION	Jun. 2002	---	---	0[c]	0[c]	0[c]	0[c]
	Dec. 2002	---	---	0[c]	0[c]	0[c]	0[c]

Total NH_3-N Emissions from Lagoons, Barns and Land Application	N-excretion by Mass Balance at Howard Farm, E	Total Baseline Model Emission (F_{proj})[d]	% of E_{base}	Total Howard Measured Emission (F_{meas})[e]	% of E_{EST}
June 2002 (Warm)	**194.0**	**96.2**	**52.2 ± 8.3**	**136.0**	**70.1 ± 12.2**
December 2002 (Cold)	**131.8**	**52.7**	**31.4 ± 6.0**	**57.0**	**43.3 ± 7.2**

[a] barn house emissions measured at Moore Farm during October 2002.

[b] barn house emissions measured at Moore Farm during Jan-Feb 2003.

[c] soil emissions were below the Model 17C Ammonia Analyzer detection limit of 2 ppb @ 200 ppb range.

[d] used average of Warm Season or Cold Season total hog weight to calculate F_{proj} which has units of [kg-N/yr/1000 kg-LAW].

[e] Used Howard total hog weight to calculate F_{meas} which has units of [kg-N/yr/1000 kg-LAW].

Table 3.9 Estimated emission factors for liquid waste storage lagoons/ponds (lagoon NH_3-N loss/N excreted) from LSTs and ESTs.

Research Sites	Liquid Waste Storage Subsection	Averaged Ammonia Flux	Surface Area of Emission	Total Estimated Emission	Total Animal Weight	Estimated NH_3-N Loss from Liquid Waste Storage System Surfaces	Nitrogen Excretion (E)	%N loss from liquid waste storage systems (%E)
		µg NH_3-N/m^2/min	(m^2)	kg NH_3-N/min	kg	kg N/1000 kg-LAW/yr	kg N/1000 kg-LAW/yr	--
Stokes Farm (September 2002)	Storage Pond	2,385	15,170	0.0358	457,372	**41.1**	**140.7**	**29.2**
Stokes Farm (January 2003)	Storage Pond	153	15,170	0.0024	329,661	**3.8**	**133.2**	**2.9**
Moore Farm (October 2002)	Storage Pond	1,657	30,630	0.0511	397,711	**67.5**	**228.3**	**29.6**
Moore Farm (Jan-Feb 2003)	Storage Pond	325	30,630	0.0102	387,771	**13.8**	**202.8**	**6.8**
Barham Farm (April 2002)	Storage Pond	1,102	4,459	0.0291	952,560	**16.1**	**85.6**	**18.8**
	Overflow Pond	1,249	19,398					
Barham Farm (November 2002)	Storage Pond	436	4,459	0.0065	952,560	**3.6**	**92.0**	**3.9**
	Overflow Pond	234	19,398					
Howard Farm (June 2002)	Wetland Cells	914	29,591	0.0276	232,777	**62.2**	**194.0**	**32.0**
	Finishing Pond	434	7,428					
Howard Farm (December 2002)	Wetland Cells	289	29,591	0.0094	232,777	**13.0**	**131.8**	**9.9**
	Finishing Pond	152	7,428					

Table 3.10 Comparison of emission factors (%N Loss of LSTs) for this study with previous studies.

This Study	**Estimated %N Lost based on Nitrogen Excretion (total N input) from Direct NH_3-N Measurements for Swine Waste Storage Ponds using the Dynamic Flow-Through Chamber**				
Stokes Farm (Sept. 2002)	**29**				
Stokes Farm (Jan. 2003)	**3**				
Moore Farm (Oct. 2002)	**30**				
Moore Farm (Jan.-Feb. 2003)	**7**				
Previous Studies	**Estimated %N Lost based on Lagoon TKN from Direct NH_3-N Measurements for Swine Waste Storage Ponds using Different Technologies (Griffing et al., 2003)**				
Aneja et al., 2000[a]	August	December	February	May	**Average**
	47	**10**	**4**	**20**	**20**
Aneja et al., 2001b[a]	September	November	March	March	**Average**
	2.5	**1.6**	**1.9**	**1.2**	**2.0**
Todd et al., 1999[b]	**51 (OP-FTIR/Dynamic Flow-Through Chamber)**				
Harper, Sharpe, and Parkin (2000)c	**40 (Micrometeorologica1Technique)**				
Van der Hoek, 1998	**18 (Estimated for Europe-Review Article)**				

[a] Average fluxes were interpolated from a best fit equation using an integrated yearly loss rate based on seasonal flux measurements and average temperatures from AccuWeather (dynamic flow-through chamber method).

[b] %N lost was based on the average ratio (2.55) of the differences in measurements of OP-FTIR and Dynamic Flow-Through Chamber which were made from the same farm as Aneja et al. (2000).

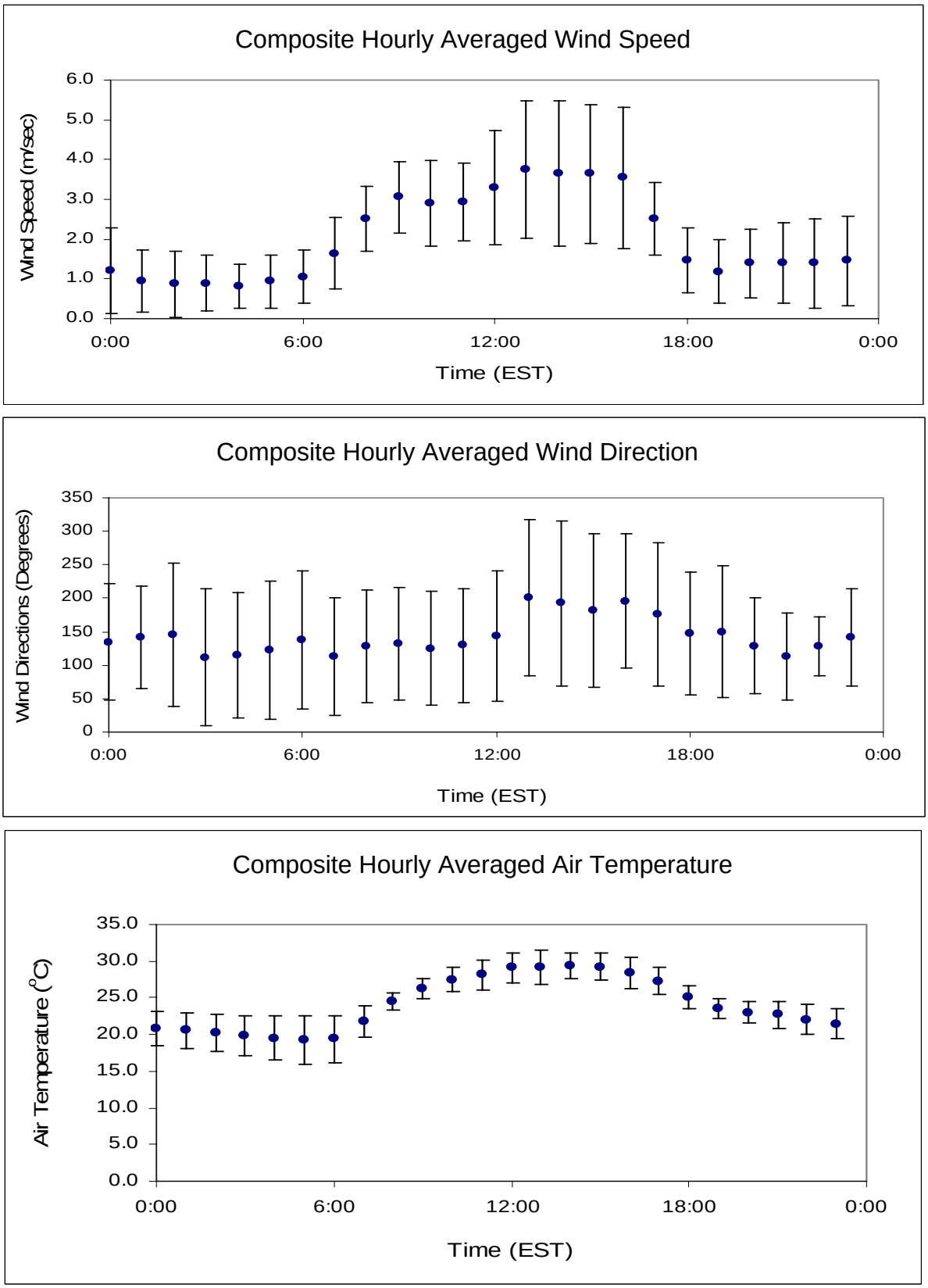
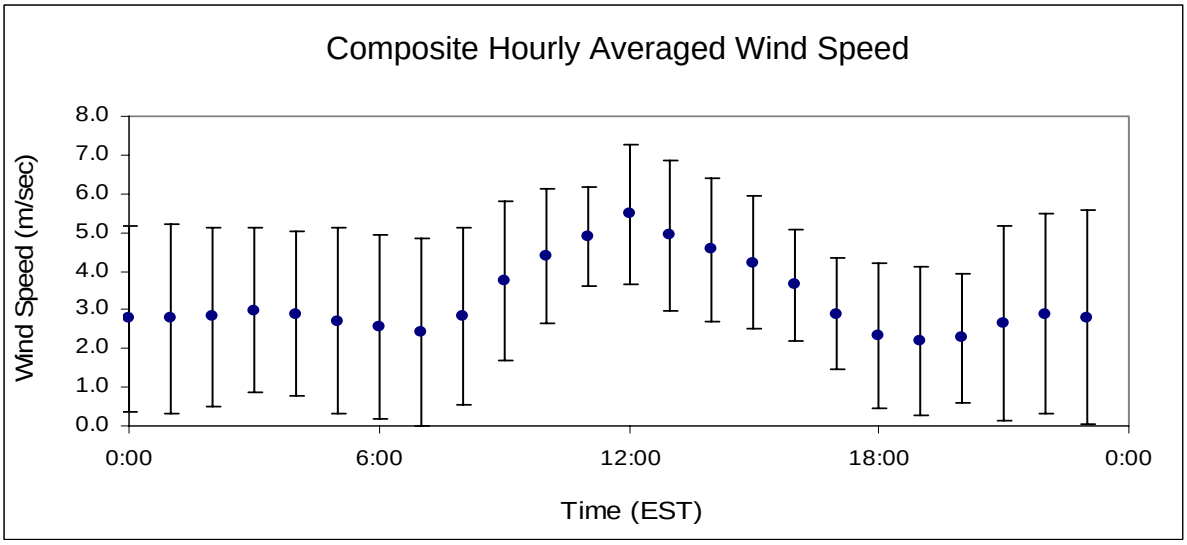
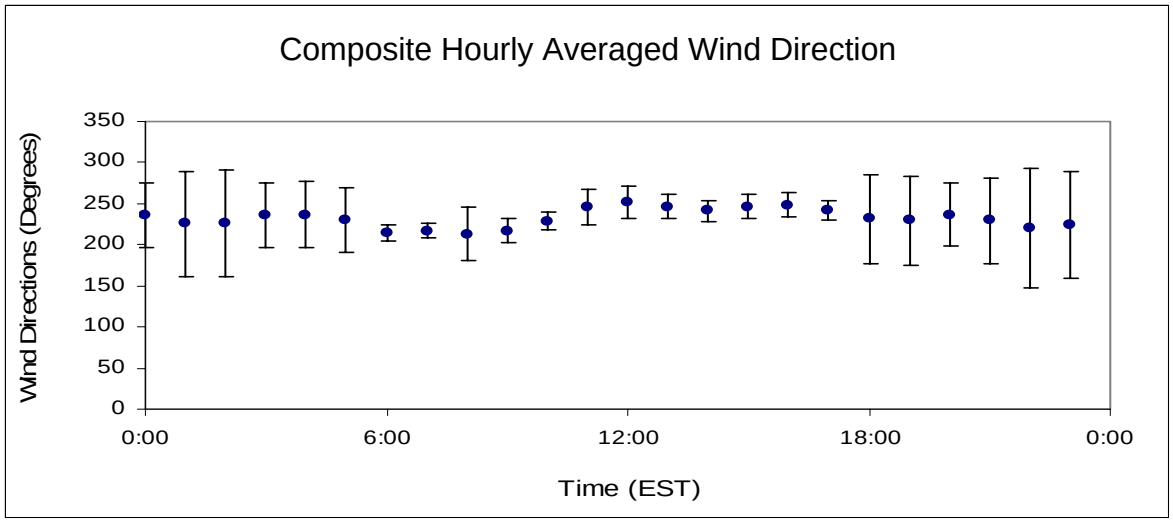


Figure 3.1a Site meteorological data during the 1st Stokes Farm measurement period (September 9 – 20, 2002).





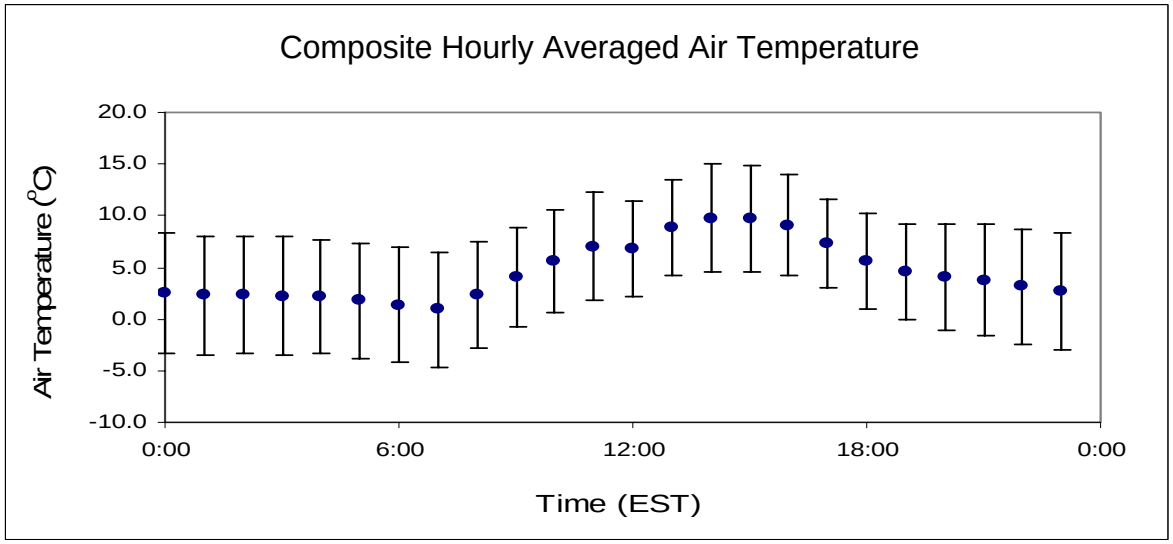


Figure 3.1b Site meteorological data during the 2nd Stokes Farm measurement period (January 6-17, 2003).

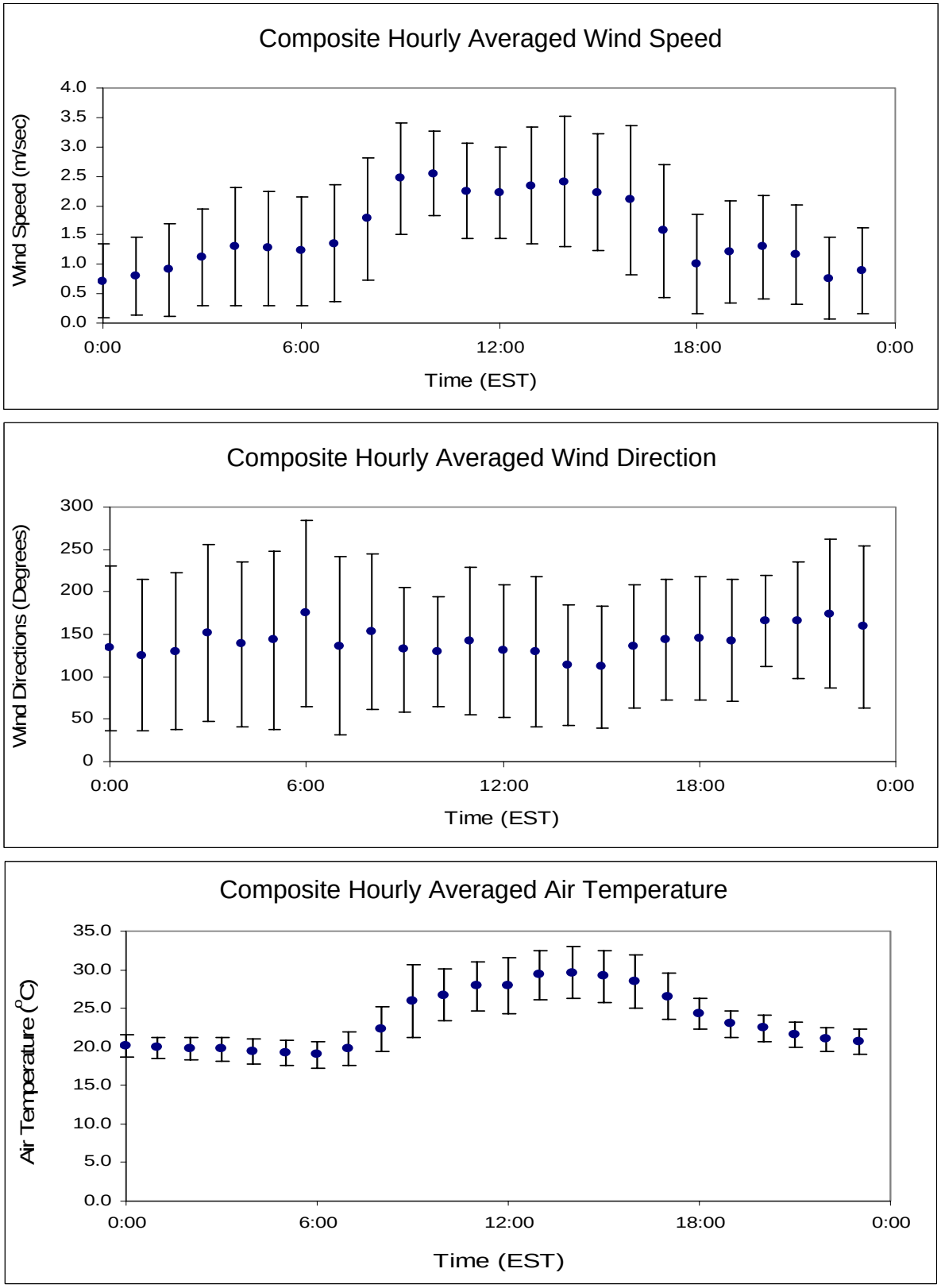


Figure 3.2a Site meteorological data during the 1st Moore Farm measurement period (September 30 – October 11, 2002).

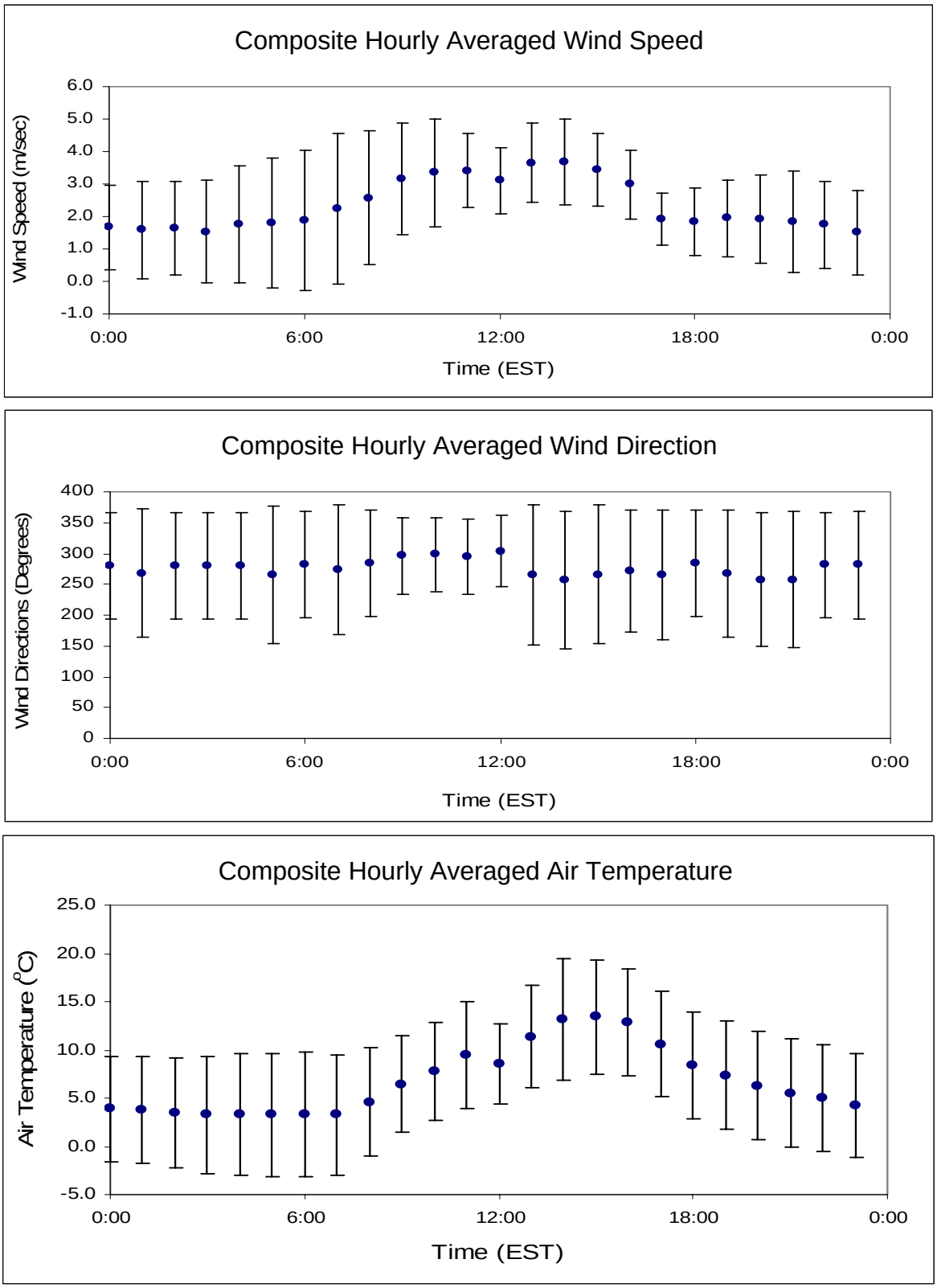


Figure 3.2b Site meteorological data during the 2nd Moore Farm measurement period (January 27 – February 7, 2003).

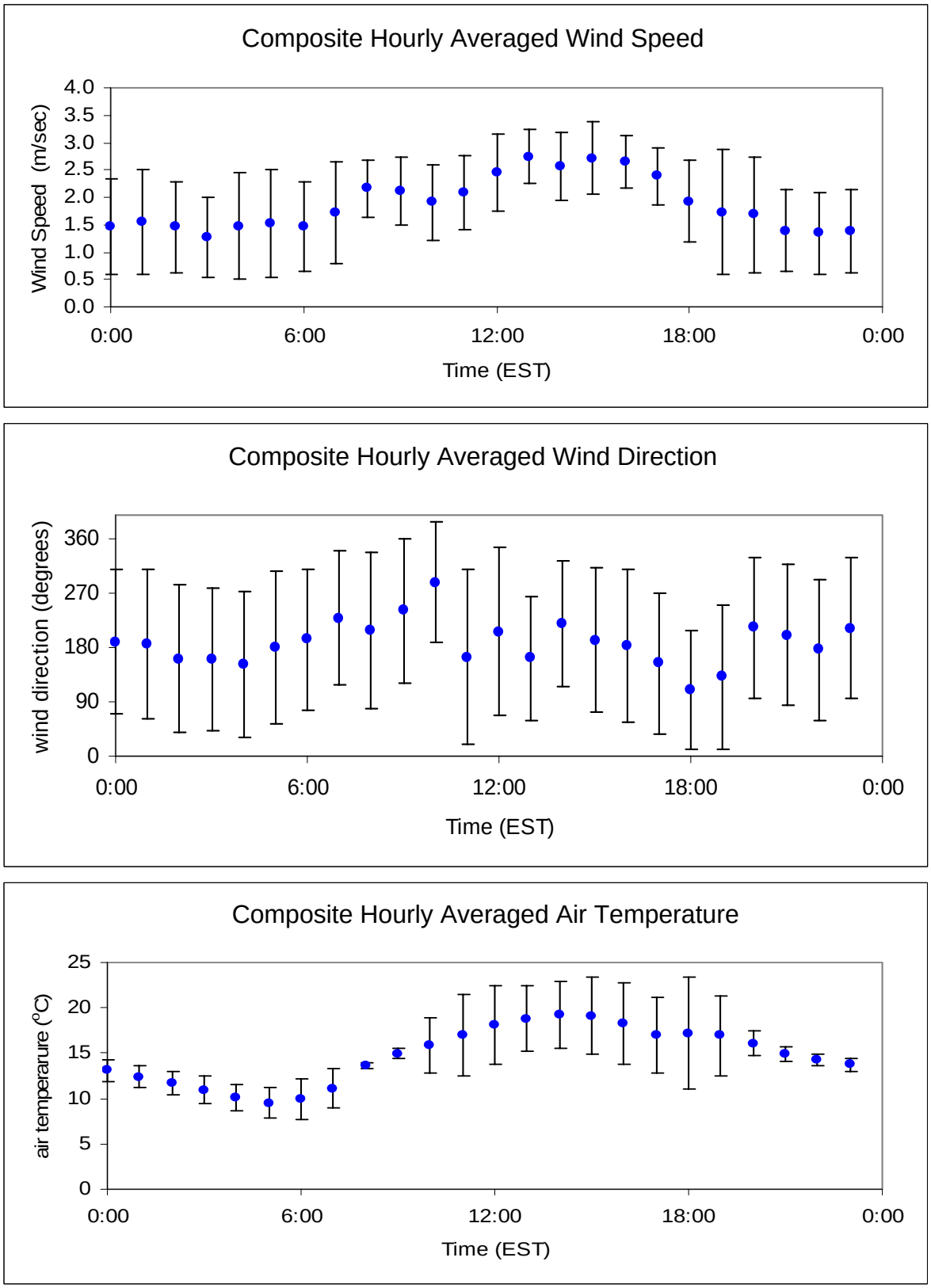


Figure 3.3a Site meteorological data during the 1st Barham Farm measurement period (April 1 – 12, 2002)

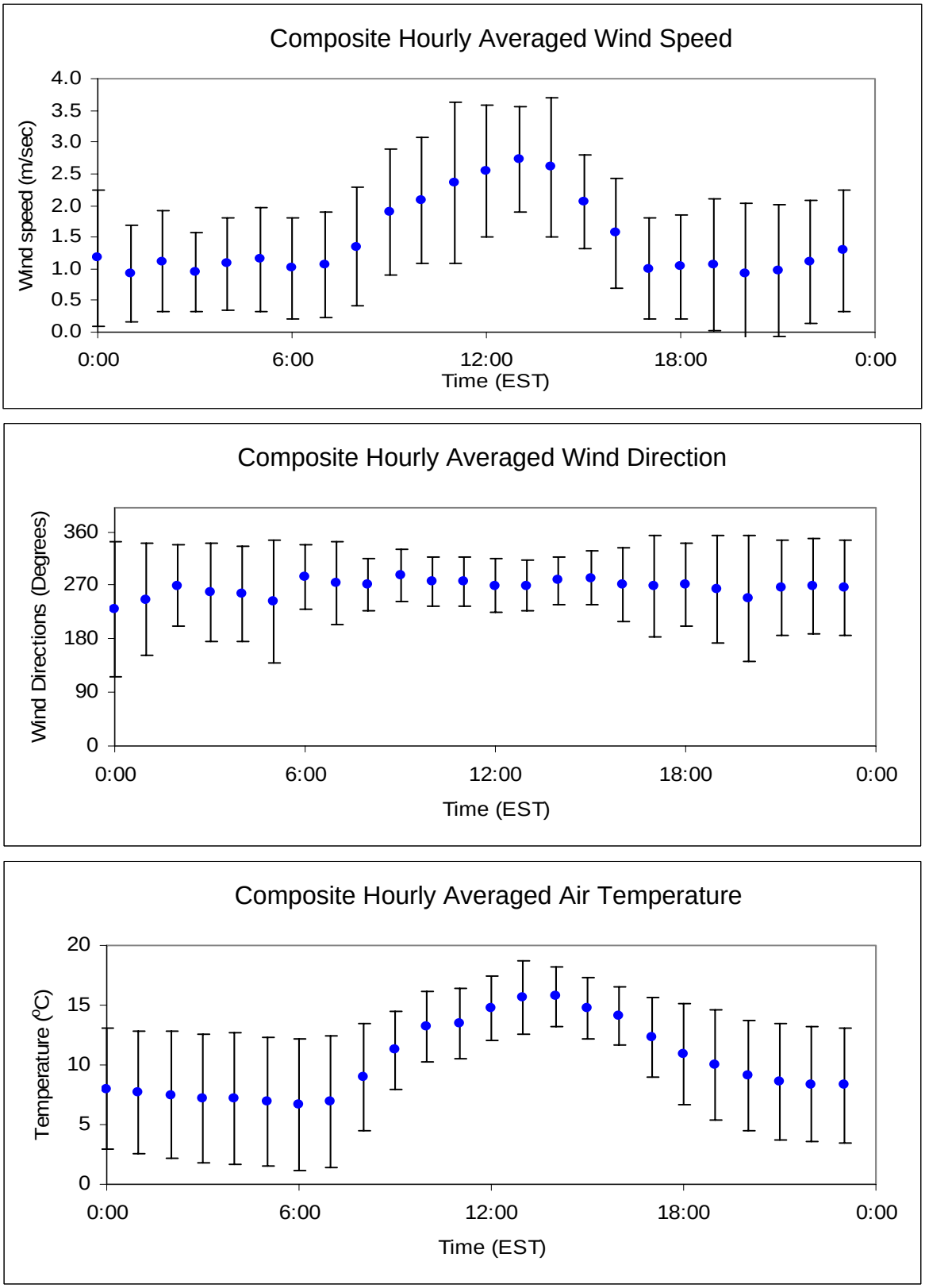


Figure 3.3b Site measurement data during 2nd Barham Farm measurement period (November 11-22, 2002)

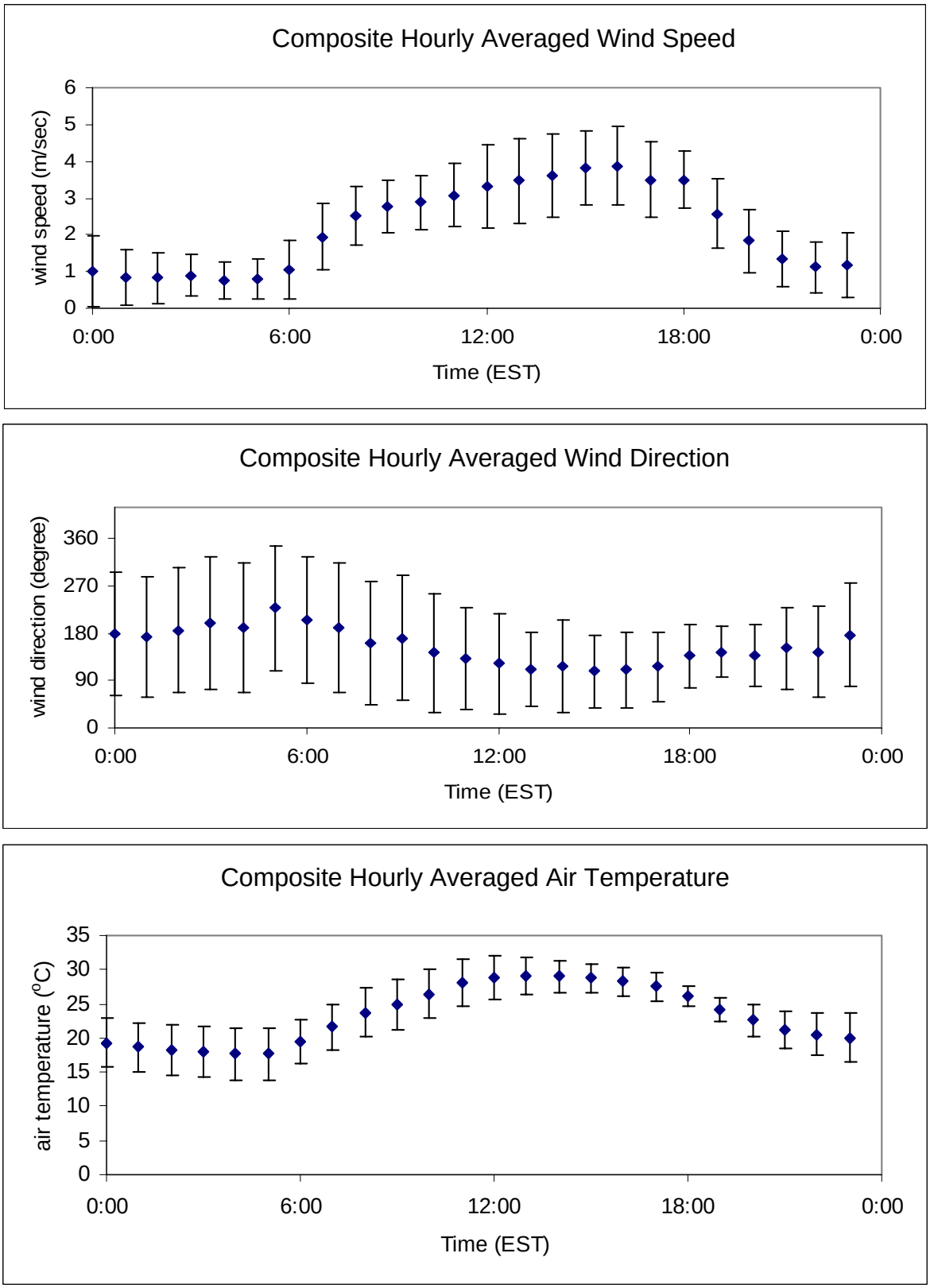
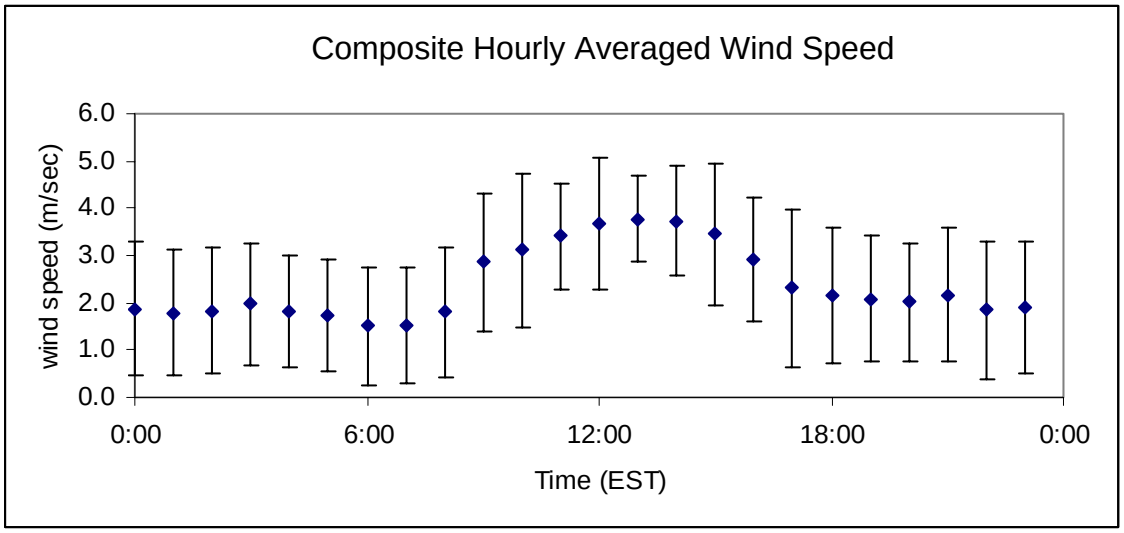
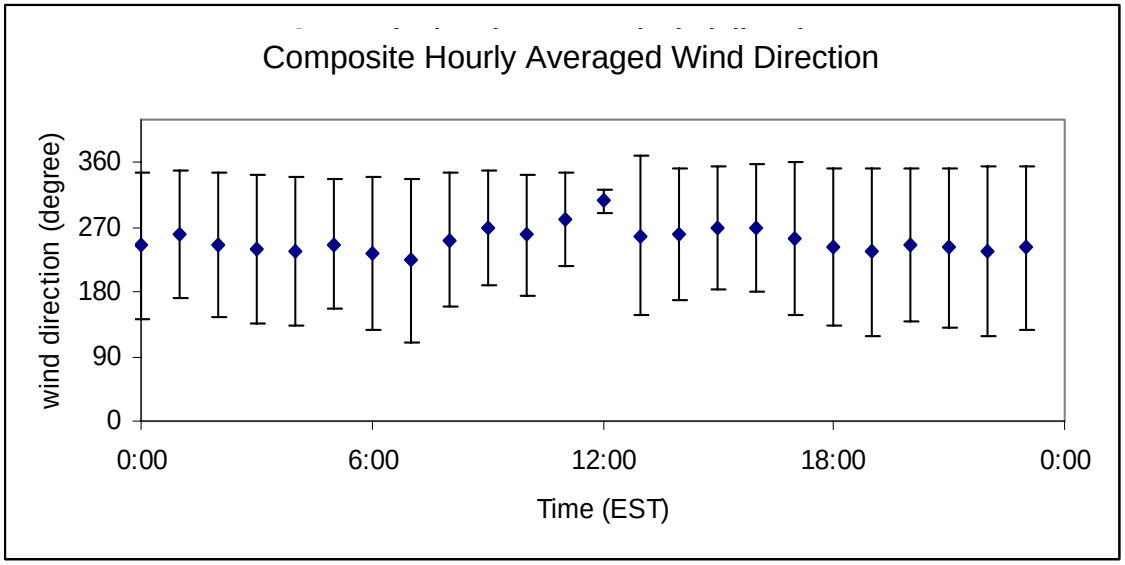


Figure 3.4a Site Meteorological data during the 1st Howard Farm measurement period (June 3 – 14, 2002).





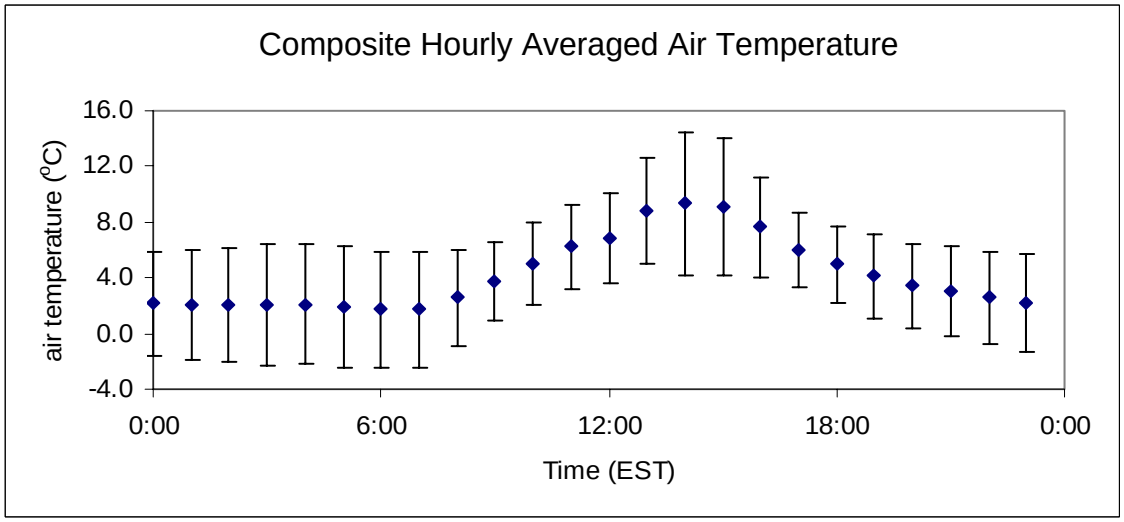


Figure 3.4b Site Measurement data during 2nd Howard Farm measurement period (December 2-13, 2002).

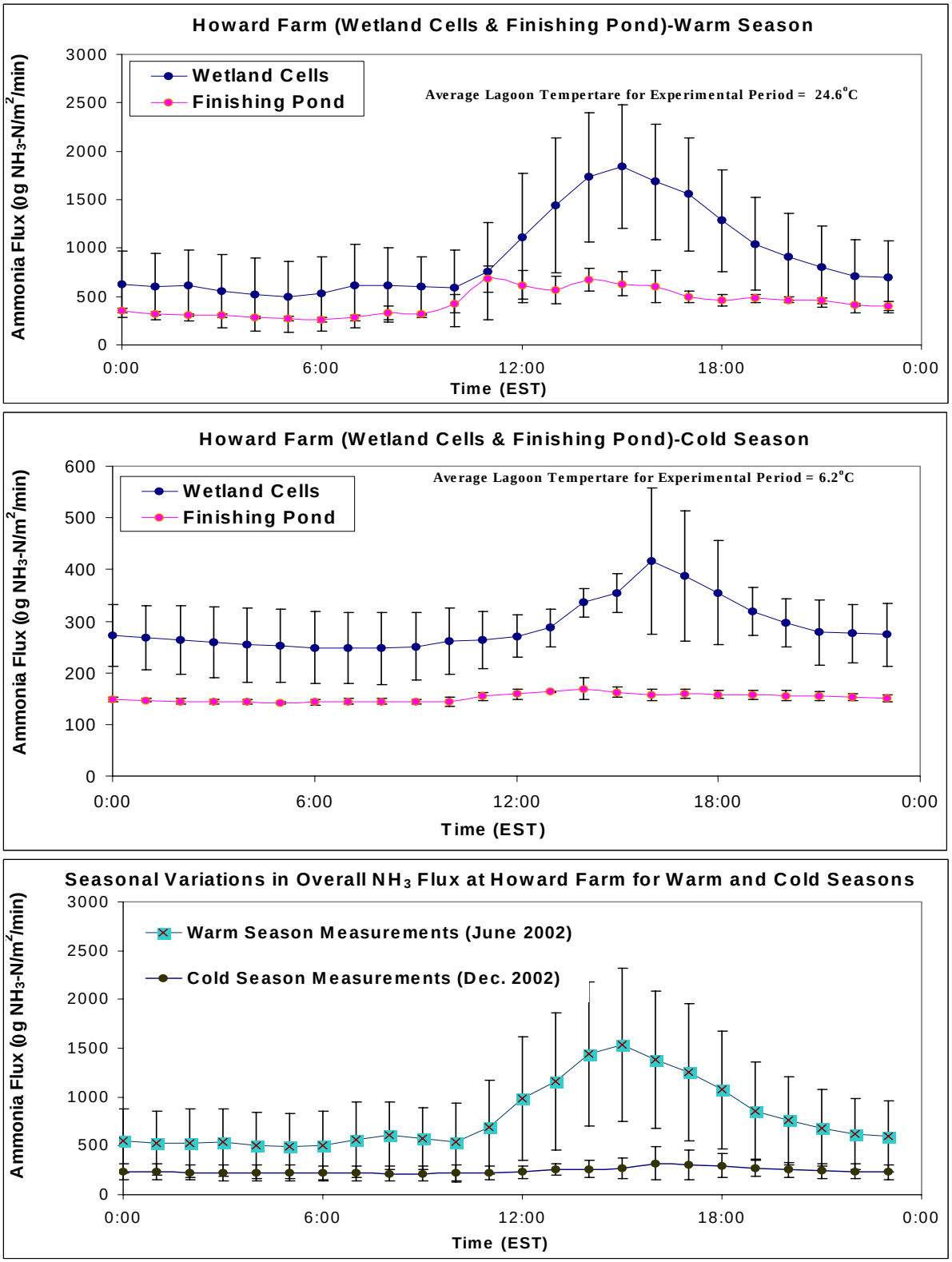


Figure 3.6 Diurnal pattern of ammonia flux at Howard Farm from the wetland cells, finishing pond, and overall measurements for warm and cold seasons. Each data point consists of an hourly averaged ammonia flux (µg NH_3-N/m^2/min).

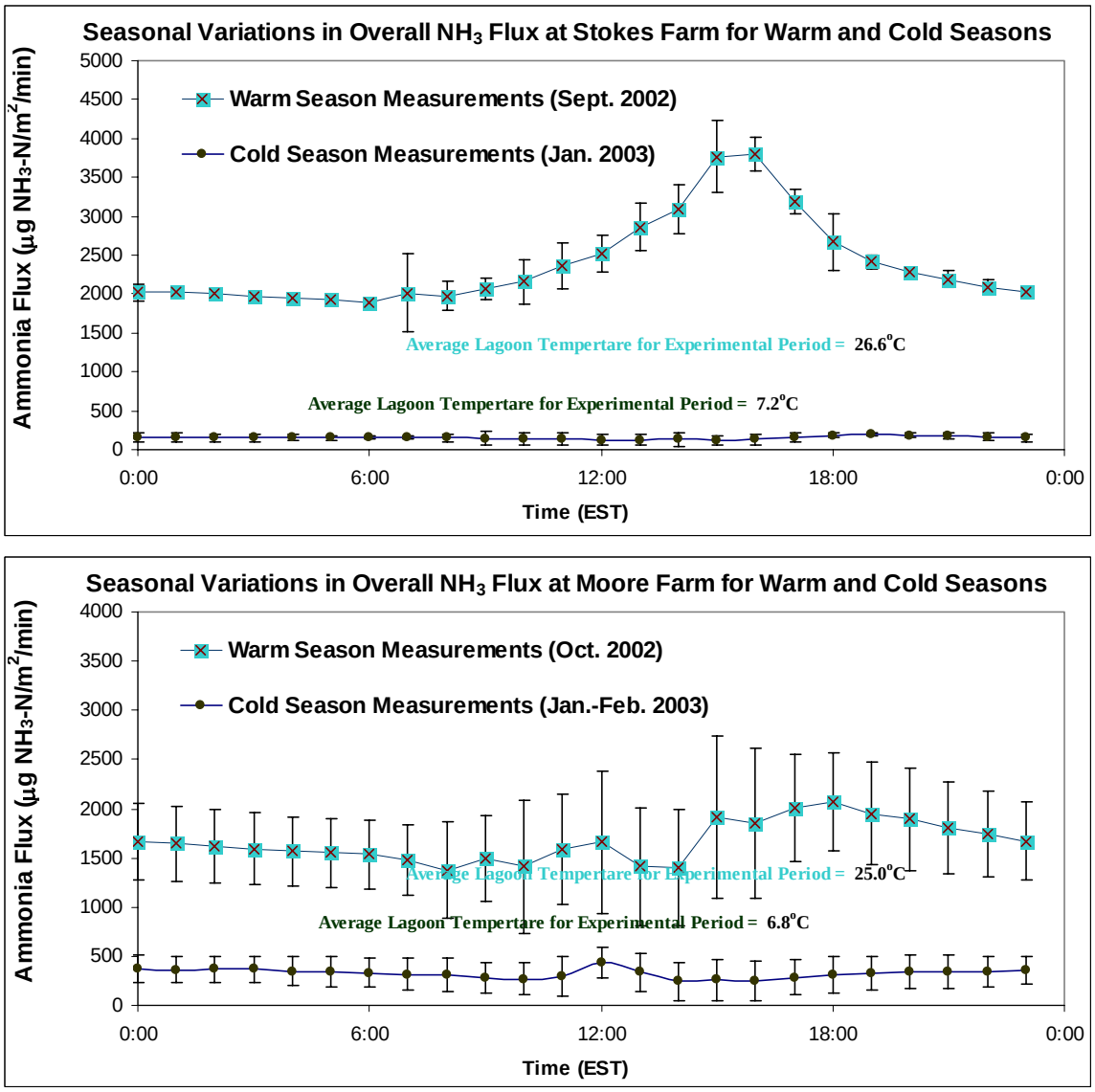


Figure 3.7 Diurnal Pattern of Ammonia Flux at Stokes and Moore Farms for Warm and Cold Season Measurements. Each data point consists of an hourly averaged ammonia flux ($\mu gN\text{-}NH_3/m^2/min$).

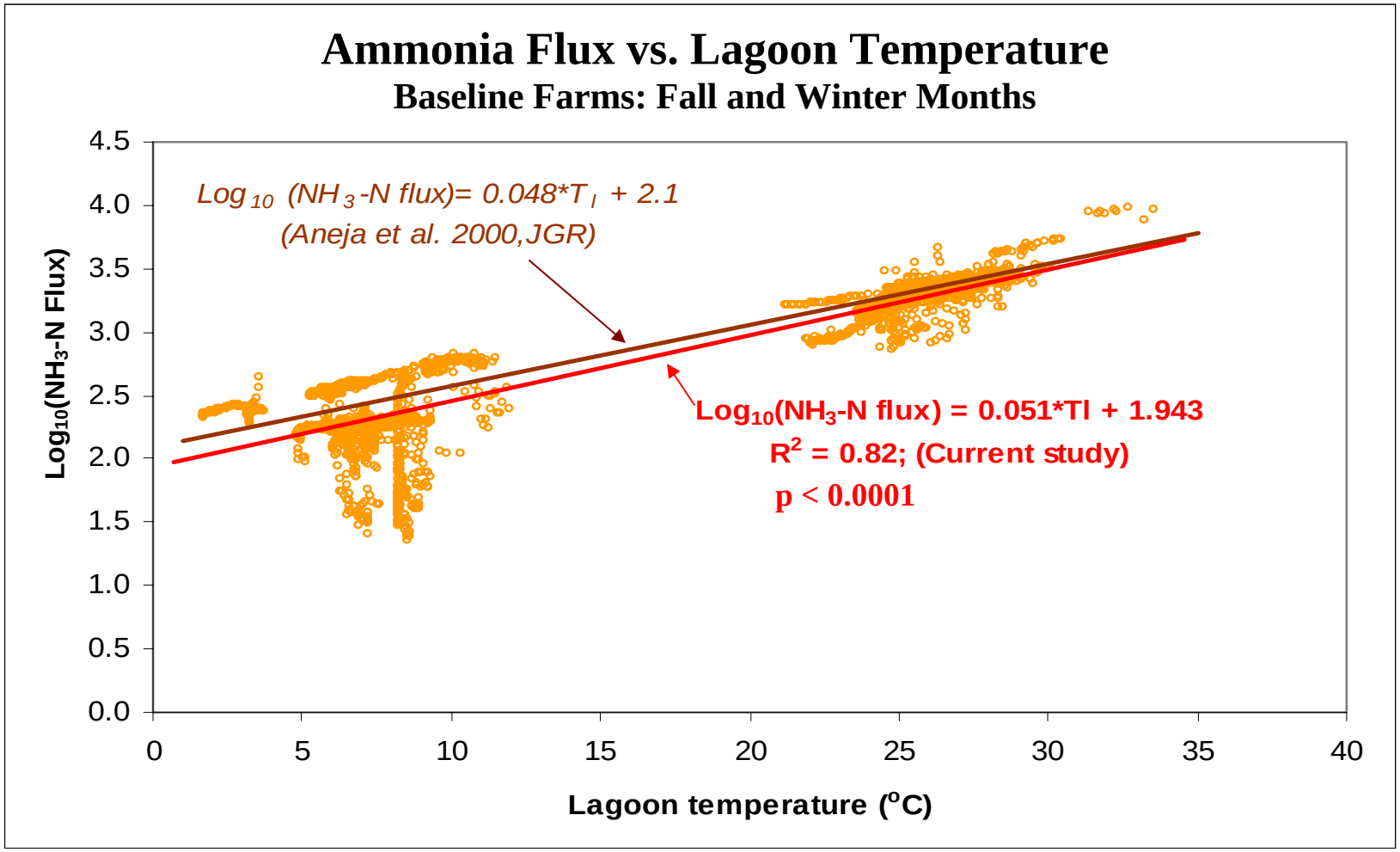


Figure 3.8 Lagoon NH_3-N flux from the baseline farms during 1st (Sep. 9-20; Sep. 30-Oct. 11, 2002) and 2nd (Jan. 6-17; Jan. 27-Feb. 7, 2003) measurement periods for Stokes and Moore farms, respectively.

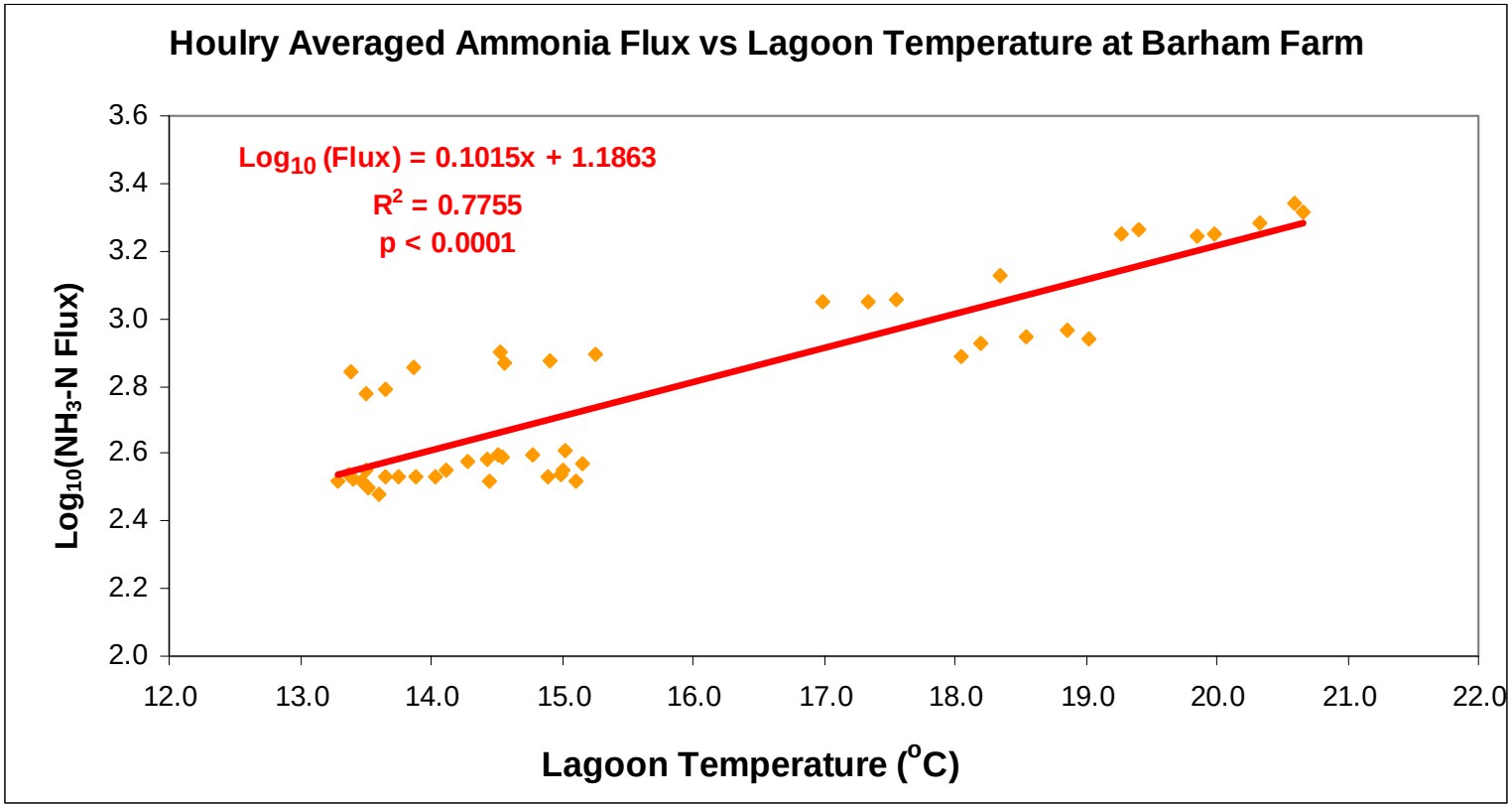


Figure 3.9 Lagoon NH_3-N flux during 1st (April. 11-22; 2002) and 2nd (Nov. 11-22, 2002) measurement periods for Barham Farm (EST).

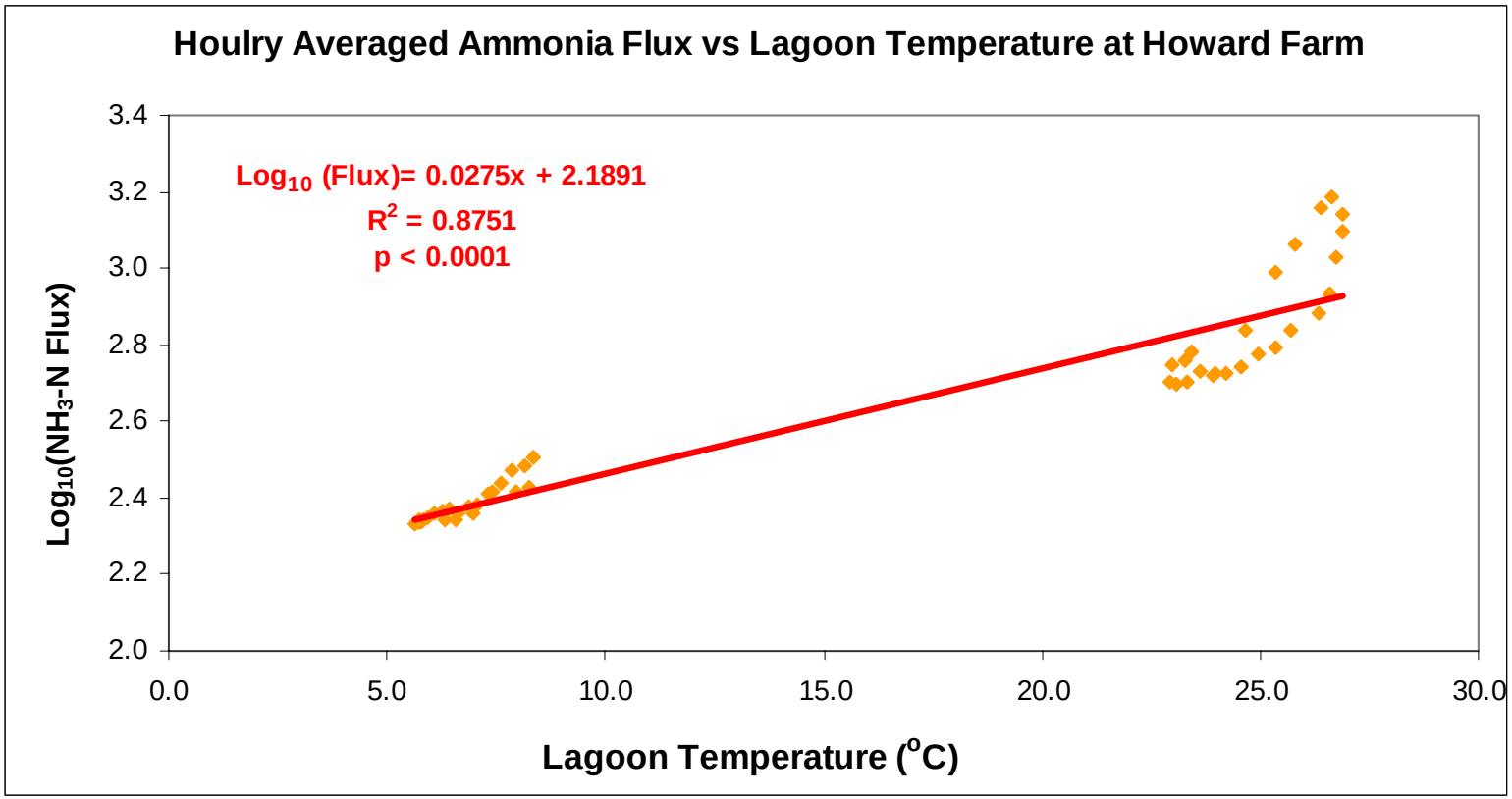


Figure 3.10 Lagoon NH_3-N flux during 1st (June 3-14; 2002) and 2nd (Dec. 2-13, 2002) measurement periods for Howard Farm (EST).

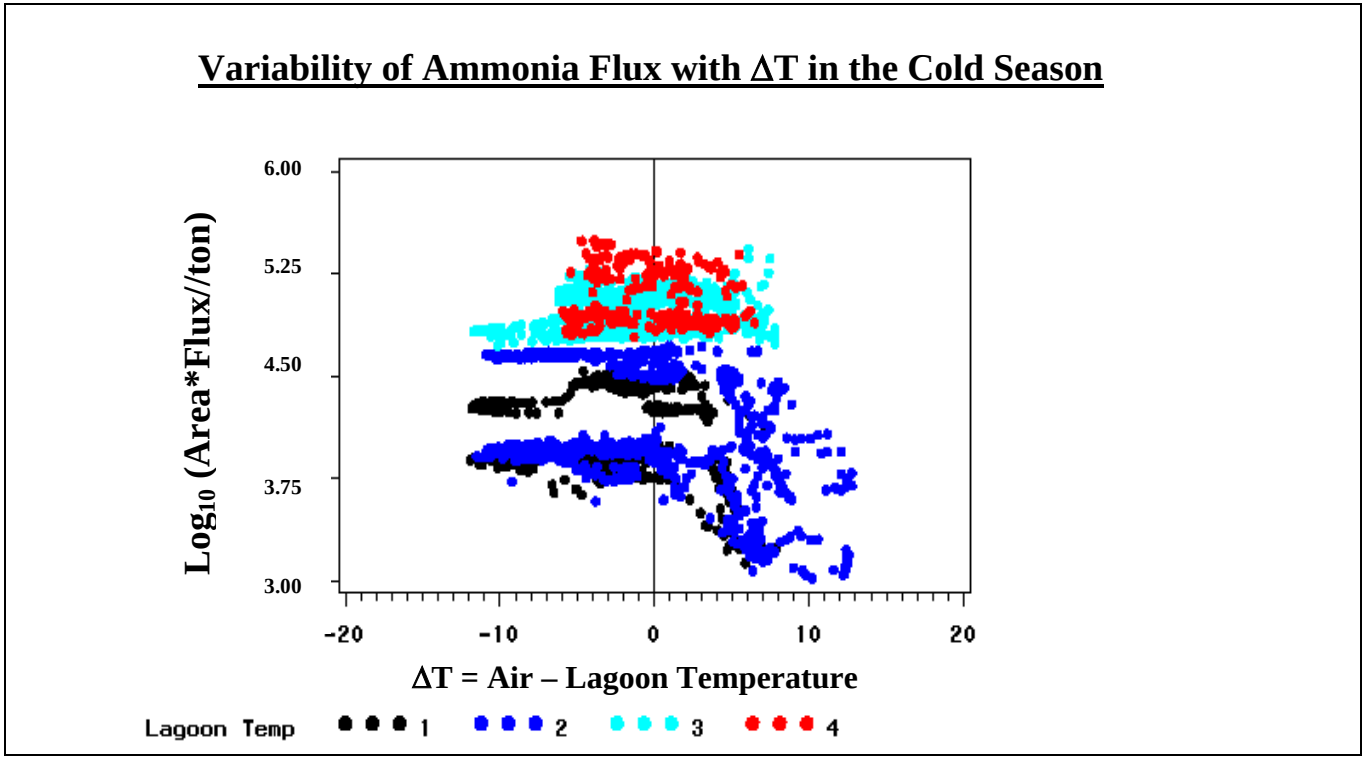


Figure 3.11 Log (*A flux/ton*) versus difference between the air temperature and the lagoon temperature (ΔT) at baseline farms for different ranges of lagoon temperature (1: < 7 ^{o}C; 2: 7 to 20 ^{o}C; 3: 20 to 27 ^{o}C; 4: > 27 ^{o}C).

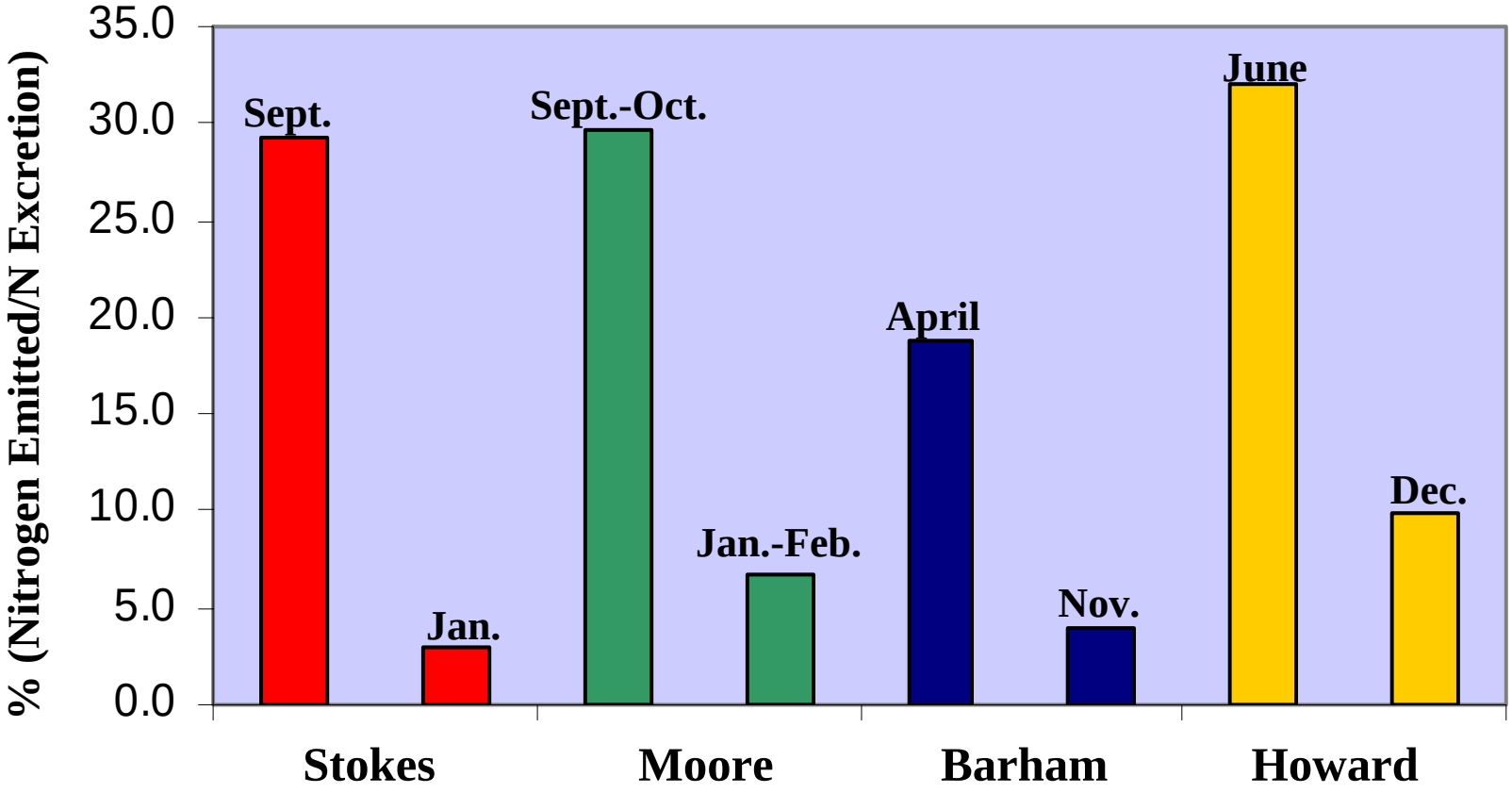


Figure 3.12 Seasonal variation of NH_3-N emissions relative to the total nitrogen available from LST and EST swine liquid waste storage systems.

4.0 SUMMARY AND CONCLUSIONS

This study focused on the emissions of ammonia from different components/processes involved in waste handling and treatment, including waste storage ponds, barnhouses, and sprays fields at two selected potential Environmentally Superior Technology (EST) farms and compared them with emissions from two baseline farms. A multiple regression analysis based on lagoon temperature and the difference of air and lagoon temperatures was employed for baseline farms, so that projected lagoon emissions could be determined at the same environmental conditions as observed at EST farms. Live animal weight (LAW), hog population, %N in feed consumed and hog feed efficiency rates were all considered in the evaluation of ESTs. A special emphasis was placed on quantifying the emissions of ammonia, using a mass balance method, to develop emission factors based on nitrogen losses from major components of the swine waste treatment systems from LSTs and ESTs, and to investigate their seasonal variability.

A multiple regression relationship between NH_3 flux and lagoon temperature and the difference between air and lagoon temperatures was developed over a relatively wide range of lagoon temperatures (~2^{o}C to 35^{o}C) observed during the field campaigns of fall and winter months at both baseline farms. A simple linear regression between $\log_{10}$(Ammonia Flux in $\mu g\ NH_3\text{-}N/m^2/min$) and lagoon temperature (T_L in ^{o}C); $\log_{10}$ Flux = 0.0516 T_L + 1.9433 (R^2 = 0.82 and $p < 0.0001$) was found, which showed very good agreement with the regression relation based on measurements from Aneja et al. (2000) with almost identical fluxes at higher lagoon temperatures. However, large scatter in the data around the best-fitted regression relationship suggested that other environmental parameters also affected NH_3 flux, especially in the colder seasons. The positive difference between air and lagoon temperature

(D) was found to have a strong negative relationship on ammonia flux, when combined with lagoon temperature, which is explained by bulk-transfer relations used for estimating surface fluxes. The relative contributions of these parameters to a multiple regression relationship for both baseline farms are given by $\log_{10}$ (A*flux/ton) = 3.8655 + 0.0449*T_L – 0.05946*D, where A is the surface area of the lagoon (m^2) and emission rate from lagoon is normalized by the live animal weight in metric tons (1000 kg). For this study, no statistical significant correlation was found between ammonia flux and pH or TAN for the two baseline farms.

The comparison of total NH_3-N emissions (kg NH_3-N/yr/1000 kg-LAW) and relative emissions, as a percentage of the nitrogen excretion rate (%E), from the two ESTs with those from the appropriate baseline farms was used as the evaluation criteria for this study. Soil NH_3-N emissions or from spray fields were found to be below the detectable limit of the chamber or NH_3 detection instruments used for measuring NH_3 concentrations in the chamber. The total lagoon and barn emissions, as % of E, at Barham Farm were 39.4 ± 6.8% compared to a near equivalent 40.8 ± 7.0% at baseline farms during the April 2002 measurement period. During this spring measurement campaign at Barham Farm, it is likely that the EST did not attain steady-state operation. For the cold season measurement period in November 2002, the total lagoon and barn emissions as % of E for Barham Farm was lower at 31.8 ± 3.9% compared to 37.1 ± 6.4% for baseline farms. The projected relative lagoon emission from the Baseline Model was lower at 11.3% compared to 18.8% from Barham farm for the spring experimental period. However, during the cold season measurement period, the measured lagoon NH_3-N emission relative to N-excretion (%E) was 3.9% at Barham Farm, as compared to 9.6% relative emission projected from the Baseline Model. The EST at Barham Farm was effective in reducing NH_3-N from lagoon (storage and

overflowing ponds) surfaces only during the second (cold season) experimental period. Barnhouse emissions of NH_3-N relative to N-excretion at Barham Farm were about the same or higher than those from baseline farms for both seasons.

The total lagoon and barn relative emissions from June 2002 measurements at Howard Farm were significantly higher (70.1 ± 12.2%) compared to 52.2 ± 8.3% for baseline emissions. For December 2002, the total relative emission were reduced at both EST and baseline sites, but emissions at Howard Farm were still higher at 43.3 ± 7.2% compared to 31.4 ± 6.0% for baseline farms. Thus, the EST at Howard Farm was not effective in reducing ammonia emissions from any of the components/processes involved in waste handling and treatment; but it appeared to enhance them.

The storage ponds at Stokes and Moore (baseline) farms had surface areas of 15,170 m^2 and 30,630 m^2. Stokes Farm had 4,391 finishing hogs for the September 2002 measurement period and 3,726 finishing hogs for the January 2003 measurement period. Moore Farm had 7,617 finishing hogs for the October 2002 measurement period and 5,784 finishing hogs for the January-February 2003 measurement period. For the fall season measurement campaigns, the average ammonia fluxes for the storage pond was 2,385 µg NH_3-N/m^2/min for Stokes Farm during the September 2002 campaign and 1,657 µg NH_3-N/m^2/min for Moore Farm during the October 2002 measurement period. For winter season measurement campaigns, average ammonia flux measurements were 153 and 325 µg NH_3-N/m^2/min for the storage ponds at Stokes and Moore farms, respectively. Stokes Farm was measured in January 2003, while Moore Farm was measured in Jan-Feb 2003.

The storage and overflow ponds at Barham Farm had surface areas of 4,459 m^2 and 19,398 m^2, respectively. For the April and November 2002 measurement campaigns,

respectively, the average ammonia flux was 1,169 μg NH_3-N/m^2/min and 352 μg NH_3-N/m^2/min from both storage and overflowing ponds. The wetland cells and finishing pond at Howard Farm had surface areas of 29,591 and 7,428 m^2, respectively. Average measured ammonia fluxes were 790 μg NH_3-N/m^2/min and 243 μg NH_3-N/m^2/min from both wetland cells and finishing pond for the June and December 2002 field campaigns.

The estimated emission factors for the early and middle fall season were 41.1 and 67.5 kg NH_3-N/1000 kg-LAW/year for Stokes and Moore farms, respectively. The estimated emission factors for the winter season were 3.8 and 13.8 kg NH_3-N/1000 kg-LAW/year for Stokes and Moore farms, respectively. Normalizing by the rate of nitrogen excretion for each farm in different seasons, NH_3-N losses from storage ponds at Stokes Farm were 29.2% and 2.9% for fall and winter seasons measurements, respectively. For Moore farm, %NH_3-N losses from storage ponds were 29.6% and 6.8% of the total nitrogen excretion (N waste available) for the fall and winter seasons experimental periods, respectively. For the two baseline farms, winter season relative losses of NH_3-N (%E) from lagoons were 26% and 23% less than those during the warmer season measurement periods for Stokes and Moore farms, respectively.

For Barham Farm, the estimated emission factors of the liquid storage systems for the spring and fall seasons were 16.1 and 3.6 kg NH_3-N/1000 kg-LAW/yr, respectively. Relative NH_3-N losses from storage and overflow ponds were 18.8% and 3.9% of the total nitrogen excretion (N waste available) for the spring and fall experimental periods, respectively. For Barham farm, November 2002 relative emissions of NH_3-N (%E) from the storage and overflow ponds were around 15% less than those during the warmer April 2002 experimental period. The estimated emission factors of the liquid storage systems for the summer and

winter seasons at Howard Farm were 62.2 and 13.0 kg NH_3-N/1000 kg-LAW/yr, respectively. This accounted for 32.0% and 9.0% of NH_3-N emissions from the wetland cells and finishing pond relative to the nitrogen excretion (N waste available) for the spring and winter measurement periods, respectively. For Howard farm, December 2002 relative losses of NH_3-N (%E) from the wetland cells and finishing pond were about 22% less compared to the warm June 2002 experimental period. Thus, NH_3-N emissions from all liquid waste storage systems strongly depended on the season, with highest values expected to occur in the summer and lowest in winter.

Variations in emission factors between swine agricultural farm operations can be attributed to differing waste treatment technologies, animal management, and feeding practices, environmental parameters and experimental error. Further investigation of ammonia flux from spray fields should be made shortly before or during lagoon spray from waste treatment systems to further explore the contribution of these sources of atmospheric ammonia. Potential sources of error in calculating ammonia emissions based on mass balance approach include the estimation of nitrogen excretion from feed analysis, steady-state assumption of liquid waste storage systems, and the contribution of other nitrogen gases such as dinitrogen. Future evaluations of other alternative waste treatment technologies by the OPEN Project Team may determine more effective technologies that may be employed by North Carolina hog farmers in the near future.

REFERENCES

Aneja, V.P., B. Bunton, J.T. Walker, and B.P. Malik, Measurement and Analysis of Atmospheric Ammonia Emissions from Anaerobic Lagoons, Atmospheric Environment, 35, 1949-1958, 2001a.

Aneja, V.P., B.P. Malik, Q. Tong, D. Kang, and J.H. Overton, Measurement and Modeling of Ammonia Emissions at Waste Treatment Lagoon-Atmospheric Interface, Water, Air, and Soil Pollution: Focus, 1, 177-188, 2001b.

Aneja, V.P., J.P. Chauhan, and J.T. Walker, Characterization of Atmospheric Ammonia Emissions from Swine Waste Storage and Treatment Lagoons, Journal of Geophysical Research, 105, 11,535-11,545, 2000.

Aneja, V.P., G. Murray, and J. Southerland, Proceedings of the Workshop on Atmospheric Nitrogen Compounds: Emissions, Transport, Transformation, Deposition, and Assessment, North Carolina State University, Raleigh, NC, pp. 299, 1998.

Arkinson, H.L., Measurements, Modeling, and Analysis of Ammonia Flux from Hog Waste Treatment Technologies, M.S. thesis p. 141, North Carolina State University, Raleigh, NC, 2003.

Arya, S.P., Personal Communication, North Carolina State University, Raleigh, NC, 2003.

Arya, S.P., Introduction to Micrometeorology, Academic Press, Inc., New York, NY, pp. 141-143, 2001.

Arya, S.P., Air Pollution Meteorology and Dispersion, Oxford University Press, New York, NY, pp. 98-100, 1999.

Bouwman, A.F., D.S. Lee, W.A.H. Asman, F.J. Dentener, K.W. Van der Hoek, and J.G.J. Oliver, A Global High-Resolution Emission Inventory for Ammonia, Global Biogeochemical Cycles, 11 , 561-587, 1997.

Boyles R.P., and S. Raman., Analysis of Climate Trends in North Carolina (1949-1998), Environmental International, 29, 263-275, 2003.

Bunton, B.J., Measurements and Analysis of Atmospheric Ammonia Emissions from Anaerobic Lagoons, M.S. thesis, pp. 1-22, North Carolina State University, Raleigh, NC, 1999.

Chauhan, J.P., Characterization of Ammonia Emissions from Swine Waste Storage and Treatment Lagoons, M.S. thesis, p. 36, North Carolina State University, Raleigh, NC, 1999.

Dai A., K.E. Trenberth, T.R. Karl, Effects of Clouds, Soil Moisture, Precipitation, and Water Vapor on Diurnal Temperature Range, Journal of Climate, 12, 2451-2473, 1999.

Dasgupta, P.K., and S. Dong, Solubility of Ammonia in Liquid Water and Generation of Trace Levels of Standard Gaseous Ammonia, Atmospheric Environment, 20, 565-570, 1986.

Dickey, D.A., and L.A. Stefanski, Personal Communication, North Carolina State University, Raleigh, NC, 2003.

Doorn, M.R.J., Natschke, D.F., Thorneloe, S.A., Southerland, J, Development of an Emission Factor for Ammonia Emissions from US Swine Farms Based on Field Tests and Application of a Mass Balance Method, Atmospheric Environment, 36, 5619-5625, 2002.

Easterling, D.R., B. Horton, P.D. Jones, T.C. Peterson, T.R. Karl, D.E. Parker, et al., Maximum and Minimum Temperature Trends for the Globe, Science, 277, 354-367, 1997.

Erisman, J.W., T. Bridges, K. Bull, E. Cowling, P. Grennfelt, L. Nordberg, K. Satake, T. Schneider, S. Smeulders, K.W. Van der Hoek, J.R Wisniewski, and J. Wisniewski, Summary Statement of the Proceedings of the First International Nitrogen Conference, Noordwijkerhout, The Netherlands, 1998.

Finlayson-Pitts, B.J., and J.N. Pitts, Jr., Chemistry of the Upper and Lower Atmosphere: Theory, Experiments, and Applications, Academic Press, San Diego, CA, 2000.

Griffing, E., M. Overcash, and S. Kim, Environmental Analysis of Swine Waste Processing Technologies using the Life-Cycle Method, North Carolina Water Resources Research Institute Submitted Report, pp. 44-50, 2003.

Harper, L.A., R.R. Sharpe, and T.B. Parkin, Gaseous Nitrogen Emissions from Anaerobic Swine Lagoon: Ammonia, Nitrous Oxide, and Dinitrogen Gas, Journal of Environmental Quality, 29, 1356-1365, 2000.

Ibusuki, T., and V.P. Aneja, Mass Transfer of NH_3 into Water at Environmental Concentrations, Chemical Engineering Science, 39, 1143-1155, 1984.

Johansson, C., and L. Granat, Emission of nitric oxide from arable land, Tellus, 36B, 25-37, 1984.

Kaplan, W.A., S.C. Wofsy, M. Keller, and J.M.D. Costa, Emission of Nitric Oxide and Deposition of Ozone in a Tropical Forest System, Journal of Geophysical Research, 93, 1389-1395, 1988.

Kim, D.S., V.P. Aneja, and W.P. Robarge, Characterization of Nitrogen Oxide Fluxes from Soil of a Fallow Field in the Coastal Piedmont of North Carolina, Atmospheric Environment, 28, 1129-1137, 1994.

Kim, D., and V.P. Aneja, Chemical Composition of Clouds at Mt. Mitchell, North Carolina, USA, Tellus, 44BB, 41-53, 1992.

McCulloch, R.B., G. S. Few, G.C. Murray, Jr., and V.P. Aneja, Analysis of Ammonia, Ammonium Aerosols, and Acid Gases in the Atmosphere at a Commercial Hog Farm in Eastern North Carolina, USA, Environmental Pollution, 102, 263-268, 1998.

Muck, R.E., and T.S. Steenhuis, Nitrogen Losses from Manure Storages, Agricultural Wastes, 4, 41-54, 1982.

National Research Council, Air Emissions from Animal Feeding Operations, Current Knowledge, Future Needs, National Acadamy Press, p. 263, 2003.

North Carolina Department of Agriculture & Consumer Services (NCDA&CS), Agricultural Statistics Division, Raleigh, NC, Available at the following Internet site: http://www.agr.state.nc.us/stats, June 10, 2003.

Olesen, J.E., and S.G. Sommer, Modeling Effects of Wind Speed and Surface Cover on Ammonia Volatilization from Stored Pig Slurry, Atmospheric Environment, Part A, 27, 2567-2574, 1993.

Paerl, H.W., Coastal Eutrophication in Relation to Atmospheric Nitrogen Deposition: Current Perspectives, Ophelia, 41, 237-259, 1995.

Roelle, P.A., Oxidized and Reduced Biogenic Nitrogen Compound Emissions into the Rural Troposphere: Characterization and Modeling, Ph.D. thesis, North Carolina State University, Raleigh, NC, 2001.

Roelle, P.A., Chemically Reactive Nitrogen Trace Species in the Planetary Boundary Layer, M.S. thesis, North Carolina State University, Raleigh, NC, 1996.

Schlesinger, W.H., and A.E. Hartley, A Global Budget for Atmospheric Ammonia, Biogeochemistry, 15, 191-211, 1992.

Showers, W., Extended Abstract: Stable Nitrogen Isotopic Tracers of Excess Nitrogen Sources to the Neuse River Basin in Proceedings of the Workshop on Atmospheric Nitrogen Compounds: Emissions, Transport, Transformation, Deposition, and Assessment, North Carolina State University, Raleigh, NC, pp. 259-263, 1997.

Sommer, S.G., J.E Olesen, and B.T. Christensen, Effects of Temperature, Wind Speed, and Air Humidity on Ammonia Volatilization from Surface Applied Cattle Slurry, Journal of Agricultural Sciences, 117, 91-100, 1991.

TEI: Thermo Environmental Instruments, Inc. Model 17c Chemiluminescence NH_3 Analyzer Instruction Manual, Thermo Environmental Instruments Inc., Franklin, Massachusetts, 2000.

Todd, L.A., M. Ramanathan, K. Mottus, R. Katz, A. Dodson, and G. Mihlan, Measuring Chemical Emissions using Open-Path Fourier Transform Infrared (OP-FTIR) Spectroscopy and Computer-Assisted Tomography, Atmospheric Environment, 35, 1937-1947, 2001.

PigCHAMP, National Summary Reports, PigCHAMP Inc., Ames, IA, 1999.

Van Der Hoek, K.W. Estimating Ammonia Emission Factors in Europe: Summary of the Work of the UNECE Ammonia Expert Panel, Atmospheric Environment, 32, 315-316 1998.

Walker, J.T., V.P. Aneja, and D.A. Dickey, Atmospheric Transport and Wet Deposition of Ammonium in North Carolina, Atmospheric Environment, 34, 3407-3418, 2000.

Warneck, P., Chemistry of the Natural Atmosphere, second edition, Academic Press, Inc., New York, NY, pp. 484-485, 511-528, 2000.

Westerman, P.W., J.R.Bicudo, and A. Kantardjieff, Upflow Biological Aerated Filters for the Treatment of Flushed Swine Manure, Bioresource Technology, 74, 181-190, 2000.

Whitall, D., B. Hendrickson, H. Paerl, Importance of Atmospherically Deposited Nitrogen to the Annual Nitrogen Budget of the Neuse River Estuary, North Carolina, Environmental International, 29, 393-399, 2003.