

Nitrogen Cycling through Swine Production Systems: Ammonia, Dinitrogen, and Nitrous Oxide Emissions

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ABSTRACT

Ammonia (NH_3) emissions from animal systems have become a primary concern for all of livestock production. The purpose of this research was to establish the relationship of nitrogen (N) emissions to specific components of swine production systems and to determine accurate NH_3 emission factors appropriate for the regional climate, geography, and production systems. Micrometeorological instrumentation and gas sensors were placed over two lagoons in North Carolina during 1997–1999 to obtain information for determining ammonia emissions over extended periods and without interfering with the surrounding climate. Ammonia emissions varied diurnally and seasonally and were related to lagoon ammonium concentration, acidity, temperature, and wind turbulence. Conversion of significant quantities of ammonium (NH_4^+) to dinitrogen gas (N_2) were measured in all lagoons with the emission rate largely dependent on NH_4^+ concentration. Lagoon NH_4^+ conversion to N_2 accounted for the largest loss component of the N entering the farm (43% as N_2); however, small amounts of N_2O were emitted from the lagoon (0.1%) and from field applications (0.05%) when effluent was applied nearby. In disagreement with previous and current estimates of NH_3 emissions from confined animal feeding operation (CAFO) systems, and invalidating current assumptions that most or all emissions are in the form of NH_3 , we found much smaller NH_3 emissions from animal housing (7%), lagoons (8%), and fields (2%) using independent measurements of N transformation and transport. Nitrogen input and output in the production system were evaluated, and 95% of input N was accounted for as output N from the system.

THE ARRIVAL of the era in which animal production enterprises must be fully accountable for all nutrient cycling through a production system signals the need to devise mechanisms and strategies to capture or recapture nutrient streams or, failing to do this, limit and/or shift emissions from environmentally compromising to environmentally neutral entities. Systems analysis or Life Cycle Assessment tools must be employed in animal agriculture, as in other industries, to assess and limit the environmental effects of animal production enterprises. Current production, storage, and disposal techniques present a challenge to manage wastes to minimize ammonia (NH_3) losses to the environment that, depending

on the local geography and ambient levels of acid gases (SO_x and NO_x), can have potential short- and long-term impact in compromising surrounding and distant environments and ecosystems. With significant atmospheric concentrations of available acid gases, their neutralization with NH_3 forms particulates ($\text{PM}_{2.5}$) creating transport aerosols. The result is that animal production enterprises may now become part of an airshed impacting very distant ecosystems where a significant percentage of N loading occurs through atmospheric deposition, much of which is from distant sources.

Early estimates (Hatfield et al., 1993) suggested that 89 to 90% of the input N to anaerobic lagoons is lost to the atmosphere through NH_3 volatilization. This estimate represents about 60% of the total feed N input into the farm operation. Current estimates (Doorn et al., 2002) in North Carolina suggest that about 36% of N going into confined animal feeding operations (CAFOs) in the state is volatilized as NH_3 gas. However, other studies in the North Carolina and Georgia Coastal Plains region of the USA (Harper and Sharpe, 1998; Harper et al., 2000) have shown that lagoons emit significantly less NH_3 than previously thought. Much of the N input into lagoons was found to be denitrified (Harper and Sharpe, 1998; Harper et al., 2000) by microbial and/or chemical (Van Cleemput, 1998) denitrification with mean dinitrogen (N_2) emissions about three times larger than NH_3 emissions.

To evaluate the effect of animal concentrations on the region's ecosystems, emissions must be accurately evaluated from these systems. Emission factors currently in use, developed mainly from data of Northern Europe (Battye et al., 1994), are variable and questionable for use in the southern Coastal Plains of the United States. [There is some question that the northern European emission factors (Asman, 1992) were used improperly and the USA factors developed may be too large by about a factor of two (Asman, personal communication, 2000).] Seasonal variations are also inconsistent and must be properly considered when calculating annual emissions. If accurate estimates of emissions are to be determined, systems analysis techniques of N transport within production systems must be made using independent measurements of each N flow. These type analyses can be difficult and expensive and few such studies have been made (Harper et al., 1987); however, they may lead to much greater understanding of systems.

Reduction of N losses may be economically significant both in savings of feed input and from possible economic regulatory judgments against CAFO operations, providing strong motivation from the animal producer's viewpoint to promote reduction or prevention of these losses,

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Abbreviations: FF, farrow-to-finish; FW, farrow-to-wean.

particularly if N can be captured, processed, and utilized competitively to other nutrient sources. The purpose of this research was to (i) evaluate climatic influence on NH_3 emissions from concentrated swine production on a seasonal and annual basis; (ii) evaluate other N flows, emissions, and mechanisms using independent transport measurements; and (iii) obtain appropriate emissions from which factors for the southeastern Coastal Plains region of the USA can be developed.

MATERIALS AND METHODS

The swine production units studied are located in the Coastal Plains of North Carolina. Their waste disposal systems are anaerobic lagoons with the animal house waste emptied directly into the lagoon. The farrow-to-finish (FF) production facility is a 1200-sow operation and the farrow-to-wean facility (FW) is a 2000-sow operation, both under the same management. The lagoons are 2.7 (FF) and 2.4 (FW) ha on the commercial sites. The operating depths of the lagoons were 3.1 and 2.2 m with dimensions of 256×85 m and 190×135 m, respectively, with the long axes in a north–south direction. Occasional reduction in freeboard by about 0.65 m due to irrigation and/or water-level management reduced the surface by about 5%, although no emissions measurements were made during low freeboard conditions. Specific diets included gestation, lactation, starters (three each), and finishers (four each). Nutrient density profiles of these specific diets with quantities fed to each class and stage of animal were used to generate nutrient influx budgets for each enterprise. Nitrogen budgets for respective components of systems analysis budgeting were based on the following assumptions: N retention, 30% of N fed; N excretion, 70% of N fed; N in feces, 15% of N fed; and N in urine, 55% of N fed, based on Hall et al. (1988), Jongbloed (1991), and Jongbloed and Lenis (1992). The FF houses were “flush” type with recycled water from the lagoon used for flushing the pits beneath the slatted floors. Effluent for flushing was pumped from the lagoon to holding tanks and every 4 h, half the pits were flushed from the holding tanks. Complete flushing of the houses was accomplished each 8 h. The FW houses were “pull–plug” systems beneath slatted floors with a cycle time of about 1 wk. Safley et al. (1992) reported that about 25% of swine facilities in the USA use some type of anaerobic lagoon flush system. Other water entering the lagoon is from normal cleaning and washing, loss from the drinking system, urine, and rainfall. Effluent from the lagoons is applied to crops via a “traveling gun” sprinkler system.

For NH_3 emissions measurement from the waste lagoons, micrometeorological instruments were located in the center of the lagoon to obtain a minimum fetch of at least 50:1 (upwind lagoon distance to measurement height) in any direction for wind, temperature, and gas concentration profile development. Instruments were affixed to a platform “barge” with flotation tanks. The barge with micrometeorological equipment, already mounted and tested, was floated to the center of the lagoon and secured into place with adjustable legs extending to the bottom and guylines attached to the shore. The barge was then sunk to about 0.05 m below the water surface to minimize structural interference of windflow patterns by the platform itself. Wind speed (cup anemometers; Thornthwaite Associates, Centerton, NJ), air temperatures (aspirated thermocouples; Campbell Scientific, Logan, UT), and NH_3 concentration profiles were measured above the lagoon at vertical heights of 20, 31.5, 50.5, 79.5, 126, and 200 cm above the water surface. Water temperature was measured with thermocouples at two vertical depths, at the surface

(about -3 cm) and in the sludge layer (about -300 cm). Ammonia concentrations were obtained by drawing unfiltered air through gas washing bottles containing 80 mL of 0.1 M H_2SO_4 at a known rate (6 L min^{-1}) for 4 h, then transferring it to storage bottles for refrigeration until analysis of NH_4^+ concentration by a colorimetric technique similar to that of Weier et al. (1980). For additional description of this measurement technique, see Harper et al. (2000).

Twenty data collection periods, 24-h each, were scheduled throughout two years over three measurement seasons (during winter, spring, autumn, and summer, 1997–1999) to evaluate differences in losses related to microclimate and lagoon seasonal variation. Ammonia flux densities were determined above the lagoon surface from profile gradients of NH_3 concentrations and the momentum balance transport coefficient (Harper et al., 1973; Thom et al., 1975; Denmead et al., 1978). The relationship for NH_3 flux density is:

$$N = -K_{\text{mb}} \frac{\partial n}{\partial z} \quad [1]$$

where N is the NH_3 flux density ($\text{kg NH}_3\text{-N ha}^{-1} \text{ d}^{-1}$), K_{mb} is the eddy diffusivity for the gas of interest ($\text{m}^2 \text{ s}^{-1}$), n is the atmospheric NH_3 concentration ($\mu\text{g NH}_3\text{-N m}^{-3}$), and z is the height above the ground surface. The magnitude of K_{mb} depends on the level of turbulence and the factors of wind speed, surface roughness, height above the ground, and the thermal stability of the atmosphere. The value of K_{mb} can be calculated from micrometeorological theory as outlined by a number of authors (Thom, 1975; Wright and Lemon, 1966; Denmead et al., 2000) and can be written in finite difference form as:

$$K_{\text{mb}} = \frac{k^2(u_2 - u_1)}{[\ln[(z_2 - z_d)/(z_1 - z_d)] - [\Psi(z_2 - z_d) - \Psi(z_1 - z_d)]]^2} \quad [2]$$

where k is the von Karman constant (0.41), u is the wind speed (m s^{-1}), z is the gradient measurement height (m), z_d is effective vegetation height [equal to approximately 0 (m) for a water surface], and Ψ is the stability correction factor (Dyer and Hicks, 1970; Denmead et al., 2000). This technique was used since it does not interfere with the convective or turbulent transport processes involved in trace gas emissions (Denmead and Raupach, 1993). Errors associated with the micrometeorological technique have been discussed by Harper (1988, 2004) and Denmead and Raupach (1993) to be about $\pm 15\%$.

Recent studies by Flesch et al. (2002) suggest that while the eddy diffusivity for momentum (K_m) in the atmosphere is reasonably well known, the eddy diffusivity for mass (K_c) is more uncertain. This uncertainty can be conveniently expressed as uncertainty in the turbulent Schmidt number:

$$\text{Schmidt number} = K_m/K_c \quad [3]$$

Most flux–gradient relationships assume that Schmidt number is 1.0. However, some authors have observed this underestimation of flux–gradient emissions and have scaled their estimates of emissions by 30% (Simpson et al., 1995; Wagner-Riddle et al., 1996). This adjustment is justified because it gives a better agreement with energy balance observations and the derivation of the energy balance eddy diffusivity, K_{eb} . For these emissions, we assumed a Schmidt number of 0.70, which is in line with the corrections by Simpson et al. (1995), Wagner-Riddle et al. (1996), and Flesch et al. (2002).

Measurement of ammonia emissions from the confinement houses was accomplished by mass-balance techniques. Further

discussions of housing measurements can be found in Sharpe et al. (2001) and Harper et al. (2004).

Spatial sampling of lagoon nutrient content, both horizontally (in three locations) and vertically (at the surface and in the sludge layer), was accomplished using a remotely actuated, closed sampler to obtain samples representative of each of the vertical layers. The sample containers were lowered from a boat to the appropriate depths, opened for sample collection, then closed before bringing them to the surface for sample retrieval and storage. The samples were frozen immediately and shipped to a laboratory for analysis of ammonium (NH_4^+) and pH (for a description of analysis procedures see Harper et al., 2000). All lagoons were sampled similarly. On a yearly basis the lagoons were sampled monthly. During trace-gas transport measurements periods the lagoons were sampled weekly.

Harper et al. (2000) have shown significant amounts of chemical (chemodenitrification) and/or biological denitrification in lagoons in the Coastal Plains of Georgia and similar measurements were made at the FF and FW sites. Gas bubbles emitted from below the surface of the lagoons (mass-flux gases) were trapped in three to six collectors randomly located in each lagoon below the water surface. The collectors were made of 20-L open-bottom carboys (0.275-m diameter) with flotation collars, tethered to the lagoon bottom, for collection of the mass-flow gases before they reached the water-air interface. Displacement of water by the gases in the collectors was visually measured every few days (as necessary and depending on lagoon water temperature) to determine the mass-flux emissions of dinitrogen (N_2), methane (CH_4), nitrous oxide (N_2O), carbon dioxide (CO_2), and oxygen (O_2). Gases were collected from the samplers using evacuated sample lines (to avoid contamination from atmospheric air) and evacuated SUMMA canisters (B.R.C. Rasmussen, Portland, OR). Periodic samples of helium (He) injected into the collectors showed a sampling procedure contamination of about 1%. Eight-milliliter gas samples were transferred to evacuated glass auto-sampler vials capped with butyl rubber stoppers. The samples were analyzed for N_2 and O_2 using a thermal conductivity detector (Model 540; Tracor, Austin TX) at 200°C and a molecular sieve column; for CH_4 using a flame ionization detector (Tracor Model 540) with the detector at 200°C; for N_2O using electron capture detector (Mini-2; Shimadzu, Koyoto, Japan) with the detector at 300°C, column temperature of 80°C, a Porapak Q column, and N_2 as the carrier gas at 30 mL min^{-1} ; and for CO_2 using an infrared gas analyzer (Model 880A; Rosemont Analytical, LaHabra, CA) by passing the effluent from the electron capture detector. Gas fluxes were determined by measuring the amount of gases collected over time (collection volumes were measured as ebullition necessitated, normally from two to three times in summer and weekly in winter) and then multiplying the emissions by the measured concentrations. Further description of this measurement technique may be found in Harper et al. (2000).

Field emissions for NH_3 and N_2O were determined using micrometeorological techniques (Sharpe and Harper, 1997). Average annual N storage accumulation was based on the design life capacity (15 yr and a maximum sludge depth of 1.22 m) (American Society of Agricultural Engineers, 1996) and the average sludge N content during the study periods. A nitrogen systems analysis for the FF production site was made using each of the independent measurements plus the average annual design sludge retention of the lagoon. Insufficient data were available to do a similar analysis for the FW site.

RESULTS AND DISCUSSION

Ammonia Emissions

Measured lagoon NH_3 emissions were 14 824 and 7759 kg N, respectively, for the FF and FW operations. This represented 8 and 28% of N entering (fed) these respective animal production enterprises. In contrast to lagoon N_2 emissions (see below), which comprised a similar fraction of the input N to each system (average of 44%), lagoon NH_3 emissions were markedly different between systems, with about a fourfold greater fraction of feed N appearing as lagoon NH_3 emissions from the FW (28%) vs. the FF (8%) system.

Ammonia concentration gradients varied considerably, depending on climatic and lagoon conditions. Figure 1 gives some gradients measured during different seasons at both lagoons. Although a large difference in above-surface concentration is observed between winter and summer and between the FF and FW sites, factors other than temperature influence the concentration gradients including wind speed, water pH, and water nutrient concentration (next sections). Normal background atmospheric NH_3 concentrations varied between 0 and 20 $\mu\text{g m}^{-3}$ (Harper and Sharpe, unpublished data, 1999).

Figure 2 illustrates a selected period showing a typical relationship between wind speed (at a height of 200 cm) and NH_3 flux density during summer 1997 when water temperatures (three sites at -5 cm depth) were reasonably constant. During this summer season, the noontime 1-h average wind speed from day of year (DOY) 203.5 to 204.5 decreased 49% and was followed by a 77% decrease in the NH_3 flux density. However, a slight decrease in surface water temperature (from 30.2 to 29.8°C) may have contributed to a slight decrease in flux density since colder water results in a reduced gas vapor pressure in the water.

The potential for NH_3 emissions from the lagoon surface is generally a function of physical and chemical factors. The chemical effect, the partial pressure of NH_3 at the lagoon surface [$p(\text{NH}_3)$], may be explained as a function of Henry's law and the dissociation relationship of NH_3 and NH_4^+ :

$$p_{(\text{NH}_3)} = RT \left(10^{1.6 - \frac{4207.6}{T}} \right) \frac{[\text{NH}_4^+]}{[\text{H}^+]} \quad [4]$$

where $[\text{NH}_4^+]$ is the lagoon NH_4^+ concentration ($\mu\text{g g}^{-1}$), $[\text{H}^+]$ is the hydrogen (H) ion concentration ($[\text{H}^+] = 1 \times 10^{-\text{pH}}$ of the lagoon solution), R is the gas constant, and T is the solution temperature (K). The physical effect is the ability of the wind turbulence to remove NH_3 from the water surface-atmospheric boundary layer. For a soluble gas, such as NH_3 , the rate-limiting boundary layer is the diffusive air boundary layer and the removal of NH_3 from this boundary layer is a function of the atmospheric turbulence (wind speed) and atmospheric stability (measured by Richardson's number [Richardson, 1929]). In lieu of developing a physical model, this relationship suggests the physical and chemical factors that affect emissions from the lagoon surface including the ammoniacal N concentration of the lagoon, pH, water temperature, and wind speed. A physi-

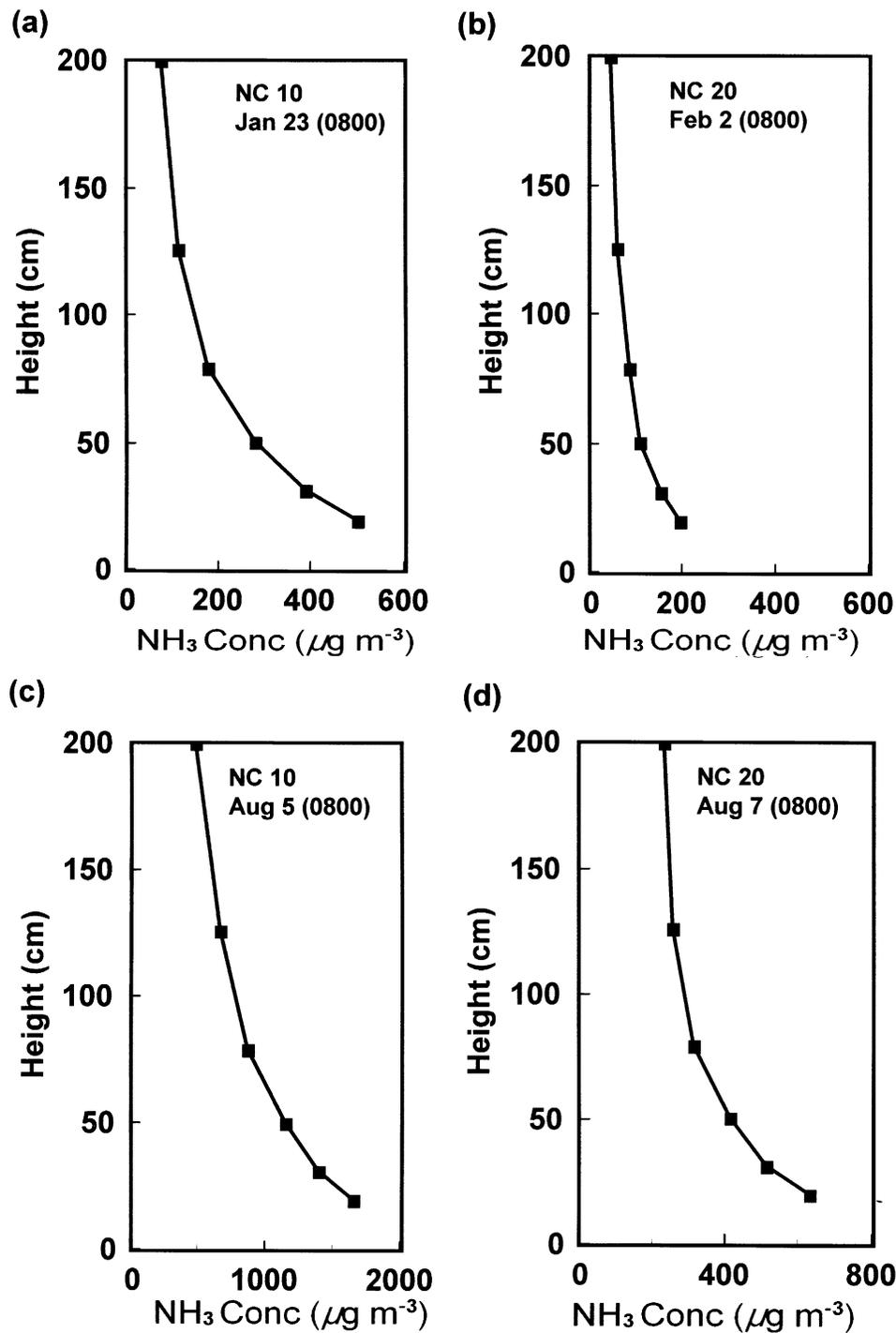


Fig. 1. Ammonia concentration gradients with height during winter and summer and over lagoons of different ammonium concentrations: (a) North Carolina farrow-to-finish farm number 10 at 0800 h on 22 Jan. 1998, (b) North Carolina farrow-to-finish farm number 20 at 0800 h on 2 Feb. 1998, (c) North Carolina farrow-to-finish farm number 10 at 0800 on 5 Aug. 1997, and (d) North Carolina farrow-to-finish farm number 20 at 0800 on 7 Aug. 1997.

cal model was developed by DeVisscher et al. (2002) using these data to predict NH₃ emissions from a lagoon or other water surface.

Ammonia emissions were quite variable among seasons (Table 1) with dependence on the interrelationships of physical and chemical factors. Emissions were higher in the more nutrient-concentrated lagoon (FF)

and generally higher during summer with highest effluent temperature. However, during spring, NH₃ emissions in the FF lagoon occasionally equaled some of the summertime emissions because of periodic high wind speeds. The interrelationship of physical (turbulence and water temperature) and chemical (ammoniacal N concentration and pH) factors was most apparent in the

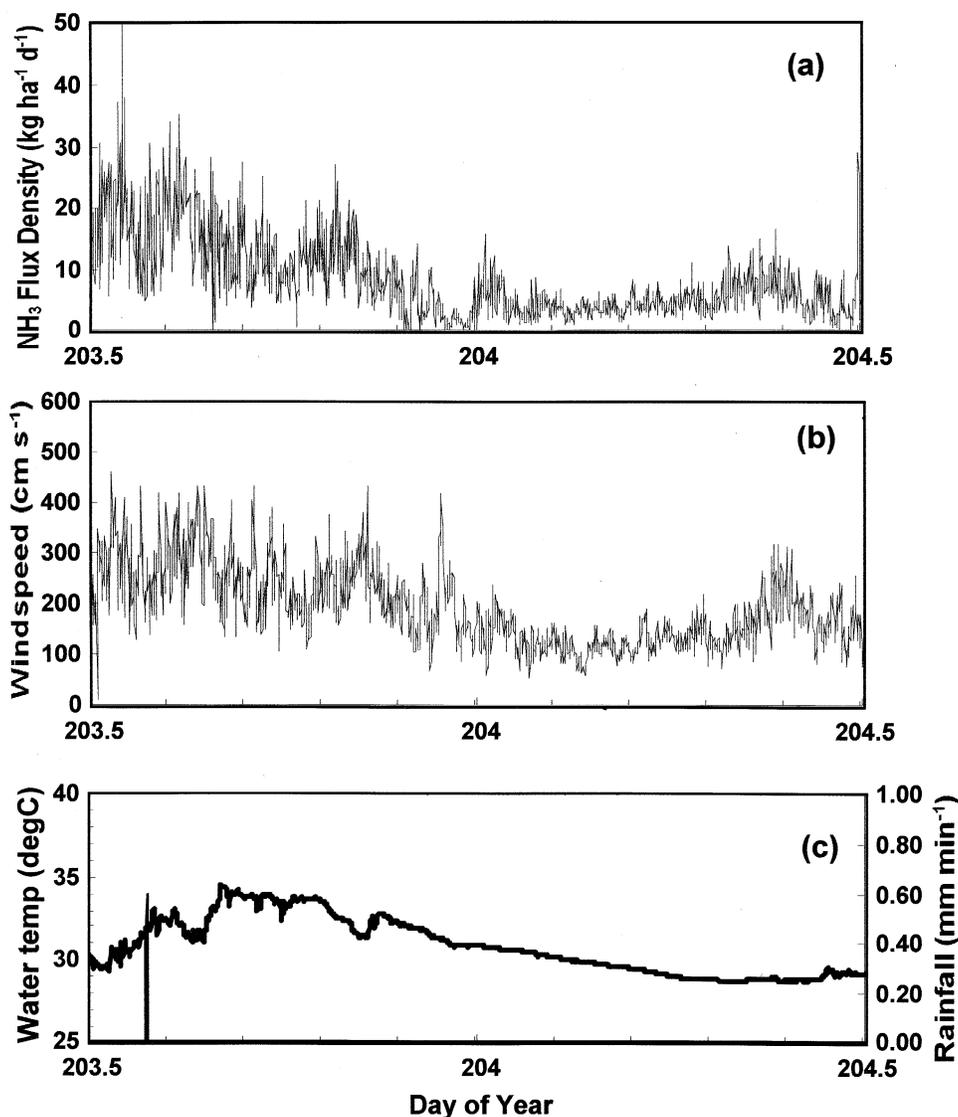


Fig. 2. Ammonia flux density variation compared with wind speed and lagoon water temperature over one day in summer, 1997: (a) ammonia flux density vs. time, (b) wind speed over the lagoon vs. time, and (c) lagoon water temperature and rainfall vs. time.

more dilute lagoon (FW) with emissions during winter and spring being larger in some situations than during summer, mainly due to the turbulence factor. Wind speeds were sometimes almost five times higher in winter. Therefore, it is not useful to estimate emissions by chemical factors of the lagoon nor by wind speed individually. Thus, the information obtained over the three seasons was used to develop a statistical model for the emissions.

Ammonia Emissions Models

Harper et al. (2000) presented seasonal data of NH_3 emissions in a swine system in the Coastal Plains of Georgia. Ammonia flux densities were generally a function of the factors of effluent NH_4^+ concentration, effluent temperature, and acidity (pH) and the physical factor of turbulence, indicated by wind speed. Observations extending over six seasons gave a rather good statistical correlation with flux density (Table 2). However, the

range of lagoon chemical factors was rather limited, making the relationship perhaps not geographically, generally useful. The farms to be studied in North Carolina were selected to extend the chemical and geographical ranges of the statistical model developed from the Georgia Coastal Plains and to include a much broader range of management systems. Table 2 compares the seasonal chemical and physical ranges of the Georgia Coastal Plains lagoon (Georgia FF1) [Harper et al., 2000] with the North Carolina Coastal Plains lagoons, FF and FW. The Georgia FF1 concentrations ranged from 229 to 291 $\mu\text{g g}^{-1} \text{NH}_4^+\text{-N}$. The North Carolina FF lagoon was considerably higher in concentration with seasonal ranges from 538 to 741 $\mu\text{g g}^{-1} \text{NH}_4^+\text{-N}$, whereas the North Carolina FW lagoon had a somewhat lower range of 183 to 227 $\mu\text{g g}^{-1} \text{NH}_4^+\text{-N}$. The pH ranges of all the lagoons were characteristic of typical lagoons. Weather conditions in all the measurement sites in the southern Coastal Plains were similar. Selection of the

Table 1. Ammonia emissions among seasons for North Carolina farrow-to-finish (FF) and farrow-to-wean (FW) farms.

Year	Farm type	Season	Day	Air temperature	Wind speed	Effluent temperature	Rainfall	NH ₃ -N concentration	NH ₃ -N emissions†	Effluent NH ₄ -N	Effluent pH
				(1.26 m)	(1.26 m)	(-0.05 m)		(1.26 m)	kg ha ⁻¹ d ⁻¹	μg mL ⁻¹	
				°C	cm s ⁻¹	°C	mm	μg m ⁻³	kg ha ⁻¹ d ⁻¹	μg mL ⁻¹	
1997	FF	spring	124.5	16.96	352.1	20.58	0.6	329.9	14.1	741	7.8
1997	FF	spring	125.7	14.04	145.7	20.83	0.0	555.3	7.4	741	7.8
1997	FF	spring	126.7	17.81	325.4	21.00	0.0	379.8	14.2	741	7.8
1997	FF	spring	127.7	18.32	206.7	21.25	0.0	527.1	22.0	741	7.8
1997	FF	spring	128.7	17.28	149.8	21.76	0.0	541.8	11.0	741	7.8
1997	FF	summer	218.3	23.00	162.3	28.65	0.7	744.7	31.4	574	8.1
1997	FF	summer	221.3	21.84	123.5	28.07	0.0	1239.8	22.0	574	8.1
1998	FF	winter	23.5	11.13	335.9	8.65	26.0	82.2	12.0	538	8.1
1998	FF	winter	24.5	11.48	183.8	9.64	9.1	142.3	17.3	538	8.1
1998	FF	winter	27.3	6.73	268.8	9.35	0.0	108.6	6.7	538	8.1
1998	FF	winter	29.3	6.15	287.3	8.97	2.3	142.8	7.3	538	8.1
1997	FW	spring	116.7	14.21	158.7	18.34	0.0	257.1	4.3	227	7.7
1997	FW	spring	117.7	15.44	179.1	18.93	1.3	214.9	9.4	227	7.7
1997	FW	spring	118.7	19.81	318.6	17.60	32.3	89.3	7.3	227	7.7
1997	FW	summer	204.5	28.31	184.8	28.50	0.0	262.0	12.0	193	8.3
1997	FW	summer	205.5	26.17	235.7	29.36	0.0	264.4	8.6	193	8.3
1997	FW	summer	207.6	24.32	98.6	28.92	0.0	403.6	4.1	193	8.3
1998	FW	winter	34.0	7.53	163.8	9.20	0.0	82.1	8.6	183	7.9
1998	FW	winter	35.0	10.72	503.0	9.36	0.5	44.9	13.0	183	7.9
1998	FW	winter	36.0	10.61	426.3	8.94	0.4	49.5	12.4	183	7.9

† Corrected for turbulent Schmidt number.

latter two farms gave a significantly greater predictive range to the development of a statistical model. Microclimate data were collected and averaged every 20 min and then averaged over the length of the NH₃ gas-washing (integrated) period, normally 4 h. Because of the true diurnal variation in the weather-driven parameters (wind speed and effluent temperature), flux densities, along with the independent variables were averaged over 24 h, which reduced 882 individual 4-h observations to 11 daily averages at the FF and 829 individual observations to nine daily averages in the FW site (Table 2). Multiple linear regressions were run on the individual farms and on the combined data from all farms measured in the southern Coastal Plains. The individual *R*² values for the FF and FW sites were not significantly different at 0.68 and 0.58. The *R*² value from five seasons during the Georgia study (Harper et al., 2000) was 0.94 for the one lagoon of quite narrow limits. However, the chemical characteristics of the Georgia FF1 were between the ranges of North Carolina FF and FW sites. When all three farms' data were combined, an *R*² for regression of all the measured farms in the southern Coastal Plains of 0.78 was obtained. The predictive relationship is:

$$N = -75.1776 + 0.0391 \times u + 0.4656 \times T_w + 0.0127 \times [\text{NH}_4^+-\text{N}] + 7.5023 \times \text{pH} \quad [5]$$

where *N* is lagoon emissions (kg NH₃-N ha⁻¹ d⁻¹), *u* is the wind speed (cm s⁻¹) at 1.26 m above the lagoon

surface, *T_w* is the lagoon water temperature (°C) at 2.5 cm below the surface, and [NH₄⁺] is the effluent NH₄⁺ concentration. Values for this relationship have a considerably larger range than the relationship by Harper et al. (2000) and are valid for much of the southern Coastal Plains. A process model developed by De Visscher et al. (2002), and tested on this data set of Table 1, was developed that should have generally a wider geographical applicability. The model simulated emissions, as measured by these micrometeorological techniques (uncorrected for Schmidt number), with an accuracy that explained 70% of the variability of the data using average daily emissions and 50% using 4-h average data.

Emissions and Emission Factors

Table 3 gives the farm average annual effluent NH₄ concentration (μg g⁻¹) of the lagoons along with average annual emissions (kg NH₃-N farm⁻¹) and emission factors (kg NH₃-N animal⁻¹ yr⁻¹). Standard deviations of the emissions are not presented since they are true seasonal variability and are not valid for use in statistical comparisons. The variation range in emissions from the North Carolina farms was much smaller than in Georgia since the average represents three seasons' measurements (winter, spring, and summer) over one year. The Georgia farm measurements had a much larger variation, probably due to measurements taken over a longer

Table 2. North Carolina farrow-to-finish (FF) and farrow-to-wean (FW) and Georgia FF1 statistical regression correlation coefficients and minimum and maximum weather and lagoon parameters.

Farm	Days	<i>R</i> ²	Wind speed, <i>u</i>		Water temperature, <i>T_w</i>		NH ₄ ⁺ concentration		pH	
			Minimum	Maximum	Minimum	Maximum	Minimum	Maximum	Minimum	Maximum
			cm s ⁻¹		°C		μg NH ₄ ⁺ -N mL ⁻¹			
North Carolina FF	11	0.683	123	352	9.0	28.6	538	741	7.8	8.1
North Carolina FW	9	0.583	98	503	6.3	19.5	183	227	7.7	8.3
Georgia FF1	19	0.938	99	1016	6.1	29.5	229	291	7.4	8.0
All	39	0.784	98	1016	6.1	29.5	183	741	7.4	8.3

Table 3. Swine farm lagoon concentrations, emissions per surface of lagoon, total emissions, emission factors per animal, and emission factors per steady-state live weight for farrow-to-finish (FF) and farrow-to-wean (FW) farms.

Farm	Lagoon concentration	Average emissions [†]	Lagoon size	Total emissions [†]	Emission factor per animal [†]	Emission factor per steady-state live weight ^{†‡}
	$\mu\text{g NH}_4^+\text{-N g}^{-1}$	$\text{kg NH}_4^+\text{-N ha}^{-1} \text{d}^{-1}$	ha	$\text{kg NH}_3\text{-N d}^{-1}$	$\text{kg NH}_3\text{-N animal}^{-1} \text{yr}^{-1}$	$\text{kg NH}_3\text{-N kg}^{-1} \text{steady-state live weight yr}^{-1}$
North Carolina FF	636.8	15.04	2.67	40.17	1.07	0.0190
North Carolina FW	201.0	8.86	2.37	21.06	1.79	0.0196
Sum of all Georgia lagoons [§]		4.16	9.76	40.51	0.86	0.0204
FF1	251.2	6.97 \parallel	3.54	24.69	0.76	0.0093
FF2	178.1	4.94 \parallel	1.34	6.63	0.24	0.0066
FF3	78.1	2.17 \parallel	3.54	7.69	0.23	0.0020
FF4	40.9	1.14 \parallel	1.34	1.53	0.03	0.0011

[†] Corrected for turbulent Schmidt number.

[‡] Steady-state weight unit is based on the type animal and design standards established by the USDA-NRCS (USDA Natural Resources Conservation Service, 1997).

[§] Harper and Sharpe (1998).

\parallel Based on North Carolina average climate conditions.

span of time (two and a half years also taken during winter, spring, and summer). It is difficult to directly compare the Georgia and North Carolina studies due to differences in the waste management systems and microclimate. The waste disposal system in Georgia was a series of four lagoons with the animal house waste emptied into the first lagoon (Georgia FF1) and the effluent then gravity-fed successively to and through each of the other three lagoons (Georgia FF2, Georgia FF3, Georgia FF4). In addition, wind speeds averaged about 50% higher during the Georgia study periods than during the North Carolina study periods, making the direct comparison of the average emissions misleading. Consequently, average Georgia emissions were calculated from the Georgia statistical model using average North Carolina wind speed data. Average emissions in $\text{kg NH}_3\text{-N ha}^{-1} \text{d}^{-1}$ for the North Carolina FF and FW lagoons were not proportional (on an emissions ha^{-1} basis) with respect to the lagoon concentrations,

partially because of the size differences. On an emissions per farm basis ($\text{kg NH}_3\text{-N ha}^{-1} \text{d}^{-1}$), the emissions for the North Carolina FF and FW and the Georgia FF1 lagoons varied generally with the ammoniacal N concentration at the water surface, corroborating the validity (or usefulness) of the statistical model (Eq. [5]). It appears that the larger lagoon surface area of all four lagoons at Georgia FF1 provided a larger surface area for volatilization, although the initial ammoniacal N concentration of lagoon Georgia FF1 was only 40% as concentrated as the North Carolina FF site and total emissions were about the same.

Emission factors have been used to calculate NH_3 emissions from animal husbandry all over Europe (Asman, 1992). Table 4 presents a list of U.S. and European emission factors based on a “per animal per year” basis. The factors are quite variable because there are not clear-cut links between countries and their studies (Battye et al., 1994), measurement technologies, components

Table 4. Emission factors in the United States and Europe.

Reference	Emission factor	Region
	$\text{kg NH}_3 \text{ animal}^{-1} \text{ yr}^{-1}$	
Asman (1990)	3.3	Europe
Asman (1992)	4.4	Europe
Buijsman et al. (1987)	1.9	Europe
Cass (1982)	3.7	Europe
Jarvis and Pain (1990)	3.6	Europe
Krause et al. (1989)	2.9	Europe
van der Hoek and Couling (1995)		
Farrow to finish (FF)	0.7 [†]	Europe (composite)
Farrow to wean (FW)	1.8 [†]	Europe (composite)
FF	5.3	Europe (composite)
FW	13.5	Europe (composite)
Battye et al. (1994)	9.2	USA (composite)
Harper et al. (2000)		
FF, primary lagoon	1.8 ^{†‡}	USA (Georgia FF1)
FF, four-stage lagoons	3.3 ^{†‡}	USA (Georgia FF1-4)
Warn et al. (1990)	1.3	USA
Aneja et al. (1999)		
FF	1.8 [†]	USA (North Carolina FF, chambers)
McCulloch (1999)		
FF	1.7 [†]	USA (North Carolina FF, Gaussian simulation)
Todd et al. (2001)		
FF	4.3 [†]	USA (North Carolina, tracers)
This study		
FF	1.1 ^{†‡}	USA (North Carolina, flux gradient)
FW	1.7 ^{†‡}	USA (North Carolina, flux gradient)

[†] Storage only.

[‡] Corrected for turbulent Schmidt number.

included (housing, storage, application), and animal categories. Consequently, emission factors must be used with caution because of variability induced by geography and meteorology, methodology for measurement, type and weight of animals, N content of feedstuffs, housing and management, and other factors. Asman (1992) developed a composite factor for Europe of 4.4 kg NH₃-N animal⁻¹ yr⁻¹ with later evaluations by van der Hoek and Couling (1995) giving slightly higher emissions for FF animals of 5.3 and a considerably higher factor of 13.5 for FW animals. Battye et al. (1994), using emissions from Europe and USDA Agricultural Statistics Service animal classifications, developed a composite factor for the USA of 9.2 kg NH₃-N animal⁻¹ yr⁻¹, which is considerably higher than the European factors or the earlier NAPAP factors (Warn et al., 1990). Comparisons of lagoon emission factors for FF management systems were similar for van der Hoek and Couling (1995) in Europe, for Harper et al. (2000) in Georgia (the primary lagoon only, Georgia FF1), and for this study. Annualized emission factors for these studies gave a factor for the North Carolina FF farm of 1.1 kg NH₃ animal⁻¹ yr⁻¹ [using Schmidt corrections, Flesch et al. (2002)]. Emission evaluation for the same FF lagoon by Aneja et al. (1999) using chamber technology and McCulloch (1999) using Gaussian plume simulations gave emissions about 1.7 and 1.5 times larger, respectively, than those of this study, for the same lagoon and time periods. It is interesting that the emissions of Aneja et al. (1999) and McCulloch (1999) are similar but larger than those of this study. Another study on the same lagoon using tracers (Todd et al., 2001) gave an emission factor of 4.3 kg NH₃ animal⁻¹ yr⁻¹ or about 4.0 times larger than these studies. The authors are not aware of error analyses for the chamber, Gaussian, or tracer studies based on other measurements, on mass-balance studies, or on other tracer studies. However, error analyses of the flux-gradient techniques have shown accuracies within about ±15% (Denmead, personal communication, 1993; Harper, 1988, 2004; Denmead and Raupach, 1993).

Table 3 shows examples of variability between the animal categories of the North Carolina FF and FW production units. Although the lagoon concentrations for the FF operation were more than three times higher than the FW production unit, possibly due to the different diet, animal type and size, animal number, and waste management, the emission factor per animal on the FF unit was 60% of the emission factor of the FW unit on a per animal per year basis. This comparison shows the inherent dangers of basing emissions on animal numbers. Yet, general emission factors are requested by planners, designers, and regulators to estimate NH₃ and other trace gas emissions.

Because of the large and perhaps inappropriate variability in emissions based on an animal number basis, we suggest a more universal emission factor based on the “steady-state animal live weight” used by the USDA-NRCS (USDA Natural Resources Conservation Service, 1997) to design and permit the construction of confined animal production operations. Waste treatment lagoons

are designed on the maximum steady-state live weight of animals using the lagoon and are based on the 5-d biochemical oxygen demand (BOD₅) or volatile solids (VS) loading. In lieu of local data for reliable waste production, Table 1 of the Code 359, Waste Treatment Lagoon, Conservation Practice Standard (USDA Natural Resources Conservation Service, 1997) provides values for waste production. It is based on data from the North Carolina Cooperative Extension Service. Table 3 in this report compares three production operations for NH₃ emissions per steady-state live weight basis (kg NH₃-N kg⁻¹ steady-state live weight yr⁻¹), showing little difference in steady-state live weight “unit” emissions for each of the operations.

Dinitrogen Gas Emissions

Earlier studies of N transport in a swine system (Harper et al., 2000) initially suggested that NH₃ emissions provided only a fraction of the N losses from lagoons, instead of the 89 to 90% suggested losses (Hatfield et al., 1993). Other segments of nitrogen transport were evaluated, including mass-flux gases emitted from the sludge layer, and we determined that a significant quantity of N₂ gas was being emitted. In the cases of FF and FW farms, we also found that only a small fraction of the N entering the lagoon was lost as NH₃. Interestingly, although the FF lagoon NH₄⁺ concentration was three times greater than the FW lagoon, the total average lagoon NH₃ emissions were only two times larger (Table 3). Also, emission factors (kg NH₃-N animal⁻¹ yr⁻¹) for the FF farm were smaller than the FW farm, both on a per-animal and steady-state live weight basis, corroborating the similar findings of van der Hoek and Couling (1995), although smaller in magnitude. Much of the proportionally smaller NH₃ emissions from the FF lagoon may be explained by N losses to the atmosphere as N₂, similar to the losses recently reported by Harper et al. (2000) in a Georgia Coastal Plains’ series of swine lagoons. These studies have shown much larger emissions of N₂ emitted from the FF than the FW lagoon (Table 5 and Fig. 3). The 1998–1999 average N₂ emission rates were 85.6 and 14.4 kg N₂-N ha⁻¹ d⁻¹, respectively. We think that the processes may be chemical conversion of NH₄⁺ to N₂ (Van Cleemput, 1998) (often called chemical [nonbiological] denitrification or chemodenitrification) in the concentrated FF lagoon (with no measured dissolved O₂ and with little nitrification [production of NO₃⁻]), and biological and/or chemodenitrification in the less-concentrated FW lagoon.

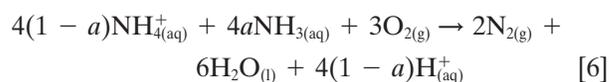
Chemical conversion is known to occur in anoxic systems by several different mechanisms, most of which

Table 5. Selected standard Gibbs free formation energies.

Species	ΔG_f° kJ mol ⁻¹
NH _{3(aq)}	-26.65
NH _{4(aq)} ⁺	-79.50
H _{2O(l)}	-237.19
O _{2(g)}	0
N _{2(g)}	0
H _(aq) ⁺	0

require the presence of nitrous acid (HNO_2 , $\text{p}K = 3.3$), which would not be present in anaerobic lagoons ($\text{pH} = 7.5\text{--}8.5$; Table 1). In trying to evaluate N_2 emissions, we assessed the theoretical possibility of oxidation of NH_4^+ to N_2 at very low O_2 pressures. This interpretation and subsequent calculation were based on the use of the Gibbs free reaction energy change (ΔG_r). The sign of this value indicates to what direction the equilibrium of a particular reaction is reached. A negative ΔG_r value indicates that the reaction equilibrium is situated to the right side of the equation and spontaneous change occurs. A positive ΔG_r value indicates that the reactants remain stable. However, it should be recognized that thermodynamic data do not give information on the velocity of reaching the equilibrium for a certain reaction. For the calculation of the ΔG_r of the oxidation reaction of ($\text{NH}_4^+ + \text{NH}_3$) or ammoniacal N concentration to N_2 , the concentrations (activities) and partial

pressures (fugacities) of the participating reaction compounds, as they were measured in the different lagoons, were used. We performed calculations using a minimum and maximum value of concentrations and partial pressures found under our field conditions. As such, all conditions were covered. The Gibbs free change of the oxidation reaction of NH_4^+ to N_2 is written:



and is calculated from:

$$\Delta G_r = \Delta G_r^0 + RT \frac{(\text{N}_2)^2 (\text{H}_2\text{O})^6 (\text{H}^+)^{4(1-a)}}{(\text{NH}_4^+)^{4(1-a)} (\text{NH}_3)^{4a} (\text{O}_2)^3} \quad [7]$$

where a (the fractional contribution of the alkaline form of NH_4^+) and $(1 - a)$ (the fractional contribution of the acid form) are given by $K/[K + (\text{H}^+)]^{-1}$ and $(\text{H}^+)(K +$

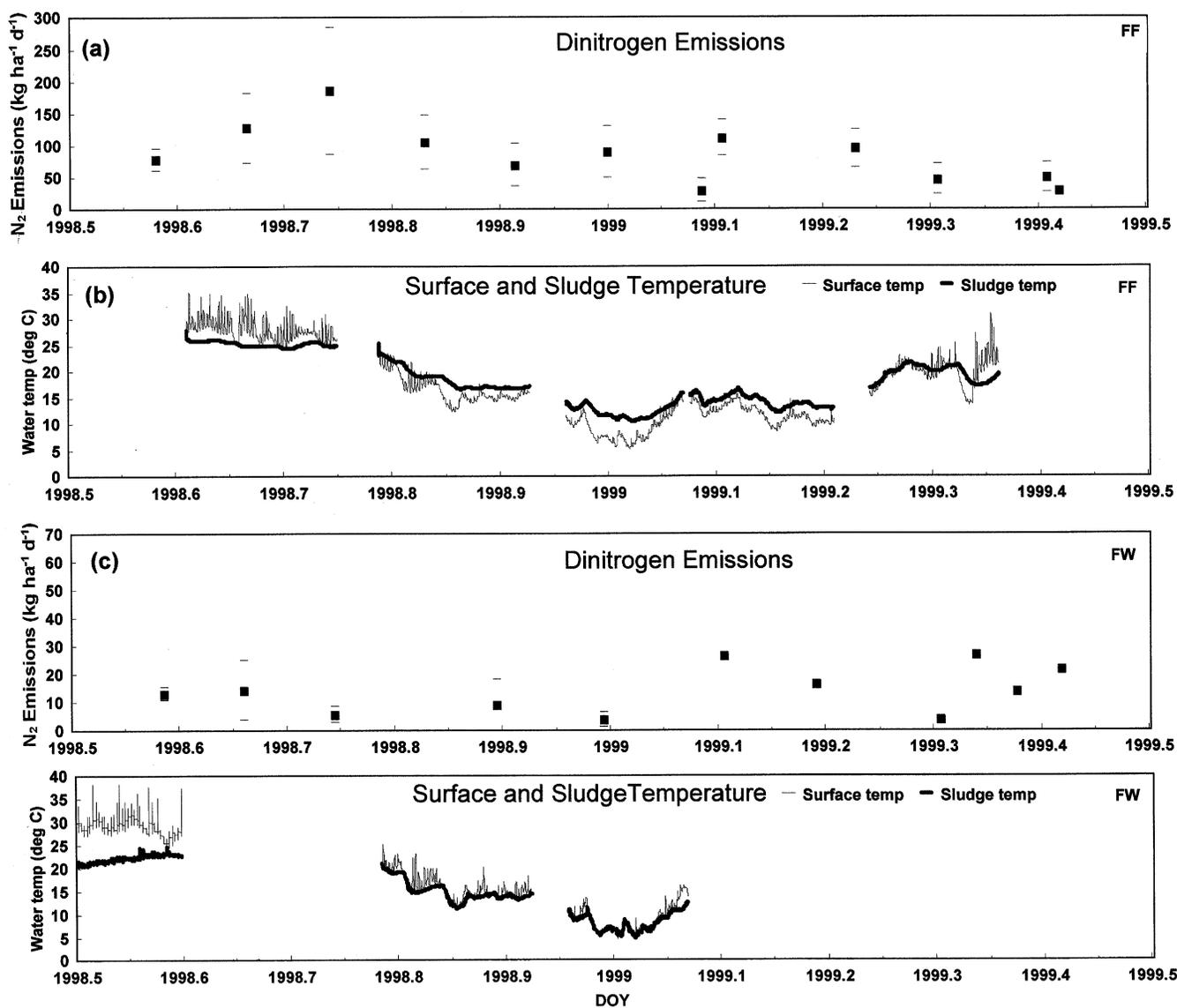


Fig. 3. Dinitrogen emissions from farrow-to-finish (FF) and farrow-to-wean (FW) production systems: (a) FF dinitrogen emissions vs. time, (b) FF surface and sludge temperatures vs. time, (c) FW dinitrogen emissions vs. time, and (d) FW surface and sludge temperatures vs. time. Error bars include measurement error and the unequal emissions spatial distribution of anaerobic decomposition in the lagoons.

Table 6. Minimum and maximum values of the participating compounds in Eq. [7], used in Eq. [8].

Species	Minimum value	Maximum value
NH ₄ ⁺ , mg L ⁻¹	500	1500
pH	6	8
N ₂ , MPa	0.01	0.05
O ₂ , MPa	10 ⁻²	10 ⁻¹⁷

$[H^+]^{-1}$, with K being the protolytic constant (5.5×10^{-10}), T the absolute temperature (298 K), R the gas constant ($8.314 \times 10^{-3} \text{ kJ mol}^{-1} \text{ K}^{-1}$), and ΔG_r^0 the standard free reaction energy. The standard free reaction energy can be calculated from the following, taking into account the ΔG_f^0 values in Table 5:

$$\Delta G_r^0 = \Sigma \Delta G_f^0 (\text{products}) - \Sigma \Delta G_f^0 (\text{reactants}) \quad [8]$$

where ΔG_f^0 is the formation energy.

In considering measured values for the concentration of the reactants and products of Eq. [6], the protolysis of NH₄⁺ must be considered. Values of the different concentrations and partial pressures for the participating compounds are given in Table 6. For partial pressure values of O₂, a range of 1.0 mPa was used to a very low arbitrary value of 10⁻¹⁵ mPa. This low value cannot be measured by an O₂ electrode, but negative redox potential values reflect these low O₂ pressures. It should be noted that although a partial pressure of 10⁻¹⁶ is very low, it still corresponds to 2.5×10^6 molecules L⁻¹. Gibbs free energies were calculated for the range of conditions that might be found in these lagoons in Table 7. The ΔG_r^0 values used (calculated from Eq. [4]) were 1105, 1106, and 1116 kJ mol⁻¹ for pH 6, 7, and 8, respectively. From Table 7, it may be seen that all ΔG_r values for a range of NH₄⁺ concentrations and a range of N₂ and O₂ partial pressures at pH 6, 7, and 8 were negative. This means that the equilibrium of Eq. [6] is negative and spontaneous oxidation may occur. Increasing the concentration of NH₄⁺ makes the ΔG_r values even more negative. Similarly, increasing the pH from 6 to 8 produces even lower negative values. Consequently, the oxidation of NH₄⁺ to N₂ is spontaneously possible, even at very low O₂ pressures. Indeed, almost every N transformation that is thermodynamically favorable is

Table 7. Gibbs free reaction energies of Eq. [6] using Eq. [7] and [8] for a set of activities or fugacities of the participating reactants and products of Eq. [6] found in North Carolina farrow-to-finish (FF) and farrow-to-wean (FW) farms.

NH ₄ ⁺ -N	N ₂	O ₂	Gibbs free reaction energy (ΔG_r)		
			pH 6	pH 7	pH 8
mg kg ⁻¹	MPa	MPa	KJ mol ⁻¹		
500	0.01	0.01	-1201	-1224	-1246
500	0.01	10 ⁻¹⁷	-944	-967	-989
500	0.05	0.01	-1193	-1216	-1238
500	0.05	10 ⁻¹⁷	-936	-959	-981
1000	0.01	0.01	-1208	-1231	-1253
1000	0.01	10 ⁻¹⁷	-951	-974	-996
1000	0.05	0.01	-1200	-1223	-1245
1000	0.05	10 ⁻¹⁷	-943	-966	-988
1500	0.01	0.01	-1212	-1235	-1257
1500	0.01	10 ⁻¹⁷	-955	-978	-100
1500	0.05	0.01	-1204	-1227	-1249
1500	0.05	10 ⁻¹⁷	-947	-970	-992

found in some ecosystem somewhere; therefore, it could be expected that conversion of NH₃ to N₂ should exist.

When gas bubbles are produced in the sediment of a swine lagoon, two exchange mechanisms of gas between the lagoon and the atmosphere exist: bubble transport and gas exchange at the surface of the lagoon. These processes can be modeled and gas exchange with the atmosphere can be calculated (unpublished data, 2004). About 80% of the N₂ in the bubbles were found to originate from the lagoon with the balance potentially from the atmosphere. This contribution is probably an underestimate since the calculations did not account for hydrostatic pressure, which increases the gas solubility and, in turn, decreases the driving force for N₂ buildup in the bubbles. However, if we assume that 20% of the N₂ comes from the atmosphere, then our measurements would provide an uncertainty of percent N fed (due to N₂ emissions) by about 8% of N fed. Contamination measurements using He injected as a replacement inert gas suggested that the combined carboy contamination and sample-transfer uncertainty was about 1%. Average annual spatial emissions variability was 43 and 17% for the FF and FW lagoons (Fig. 3) over the two years of measurement, respectively.

With comparison of N₂-N emissions to NH₃-N emissions in all the lagoons tested in Georgia and North Carolina, the current studies show a range of about twice (FW) to more than eight times (FF) as much N₂ emissions to the atmosphere as NH₃-N emissions. The lower the NH₄⁺ concentration of the lagoons (and, interestingly, lower methanogenesis [Table 8]), the smaller the amount of N₂ that was emitted, along with a reduction in the ratio of N₂ to NH₃ emissions. There was also some relationship to effluent and sludge temperature. These findings suggest that lagoons emit much less NH₃ than had previously been estimated.

Systems Nitrogen Balance

Because of N reactivity (NH₃), leachability (NO₃), plant relations (NH₃ use and emissions and N fixation), bacterial modification (nitrification and denitrification), the number of N forms that can be found in the soil (urea, NH₄, NO, NO₂, N₂O, etc.), and the large number of N forms that can be transported with the atmosphere (use and release; NH₃, NH₄ aerosols, N₂, NO, NO₂, N₂O, and various amines), it is very difficult to obtain a measured systems analysis for all the forms of N transport. The N balances in only a few cropping systems have been successfully measured (Harper et al., 1989) and we are aware of no animal production systems where a nitrogen balance has been made. Nitrogen budgets for the two different swine production enterprises are detailed in Table 9. Total N fed to the 1200-sow FF unit was 197 732 kg yr⁻¹. Of this amount entering the animal system, 138 412 kg is expected to leave as waste N, with 29 660 kg expected to appear as fecal N, and the majority, 108 752 kg, as urine N. For the 2000-sow FW enterprise, N entering the system as N feed was 27 458 kg. Waste N is expected to be 19 211 kg, of which 4119

Table 8. Comparison of lagoons including effluent concentration and gas emissions for farrow-to-finish (FF) and farrow-to-wean (FW) farms.

Location	Farm	Lagoon concentration	Ammonia gas average emissions†	Dinitrogen gas average emissions	N ₂ to NH ₃ ratio	Methane gas average emissions	Nitrous oxide gas average emissions	Total biological gas flux‡
		mg NH ₄ -N kg ⁻¹	kg NH ₃ -N ha ⁻¹ d ⁻¹	kg N ₂ ha ⁻¹ d ⁻¹		kg CH ₄ ha ⁻¹ d ⁻¹	kg N ₂ O-N ha ⁻¹ d ⁻¹	kg gas ha ⁻¹ d ⁻¹
North Carolina	FF	636.8	10.5	85.6	5.7	189.4	0.3	301.4
North Carolina	FW	201.0	6.2	14.4	1.6	5.6	0.4	21.3
Georgia	FF	238.6§	4.9§¶	23.1§	3.3	125.8§	0.0§	159.0§

† Corrected for turbulent Schmidt number.

‡ Total gas production excluding NH₃ and water vapor.

§ Harper et al. (2000).

¶ Based on North Carolina meteorological conditions.

kg is fecal N and the remainder, 15 102 kg, leaves the animal as urine N.

Nitrogen emissions from the lagoon component of the enterprise N cycle are delineated in Table 9. Annual measured N₂ emissions were 84 359 kg N yr⁻¹ for the FF and 12 483 kg N yr⁻¹ for the FW operations. These represented 43 vs. 45% of N fed for the management systems, respectively. Measured lagoon NH₃ emissions were 10 377 and 5431 kg NH₃-N, respectively, for the FF and FW operations, representing 5 and 20% of N entering (fed) these respective animal production operations. Clearly, a very major fraction of all N entering these swine production enterprises is emitted as environmentally neutral N₂ gas emissions through the lagoon biogeochemistry occurring in these systems.

Sharpe and Harper (2002) measured NH₃ and N₂O emissions from the sprayfields for the FF site during 1998 to 1999. Of the NH₄-N in the effluent, they found that about 35% of the effluent was lost as NH₃-N during application, drift, and secondary volatilization within 48 h of application. This amount represented 1.6% of N entering the farm with an additional 0.05% leaving as N₂O gas emissions. The balance of the N applied to the field was either taken off as plant seed or residue, denitrified as N₂ emissions, leached as NO₃ below the sampling zone, or stored as residual N in the soil. Housing losses were measured by Harper et al. (2004) showing that about 7.3% of the N entering the farm was lost as NH₃. Average annual N storage as sludge accumulation over the 15-yr life of the lagoon represented 3.5% of annual N fed to the animals. Using the studies of

housing (Harper et al., 2004) and field emissions (Sharpe and Harper, 2002), along with these studies and production data provided by the producer for the study site, we were able to do a nitrogen balance for FF and a partial N balance for FW farm (Table 9). Consequently, these independent measurements plus the annual design storage for the lagoon accounted for approximately 95% of N brought onto the FF production site.

CONCLUSIONS

Micrometeorological techniques were used to noninvasively measure trace-gas emissions from two types of confined animal farming operations. Measurements were made over an annual basis to obtain diurnal, seasonal, and annual emission rates. Daytime rates were generally higher than nighttime rates and warmer season rates were higher than cooler season emissions. Emissions are related to physical and chemical factors of the lagoons and a statistical model was developed to predict emissions in the southern Coastal Plains using the lagoon NH₄⁺ concentration, acidity, and temperature along with ambient wind speed above the lagoon. The model explained 78% of the variability in emissions. Lagoon emission factors were similar between our USA studies and selected European studies. Emission factors were compared between the production units, and even though the FF operation had about three times as many animals, had three times the lagoon concentration of NH₄⁺, and about twice the total lagoon farm emissions as the FW farm, the FF production unit lagoon emission

Table 9. Systems analysis of nitrogen transport in North Carolina farrow-to-finish (FF) and farrow-to-wean (FW) swine production systems. All components are independently measured.

Item	FF	FF ratio of N fed	FW	FW ratio of N fed
	kg N yr ⁻¹		kg N yr ⁻¹	
N fed†	197 732.0	—	27 458.0	—
N leaving in animals	59 320.0	0.3000	NA‡	NA
Lagoon NH ₃ emissions§	14 824.7	0.0750	8 355.7	0.2826
Lagoon denitrification N ₂ emissions	84 358.8	0.4266	12 483.1	0.4546
Lagoon denitrification N ₂ O emissions	271.6	0.0014	251.5	0.0092
Housing NH ₃ emissions	14 347.4	0.0726	NA	NA
Field NH ₃ emissions¶	3 183.9	0.0161	NA	NA
Field N ₂ O emissions¶	94.4	0.0005	NA	NA
Field N use and denitrification N ₂ loss	3 972.7	0.0201	NA	NA
Design annual N storage for the lagoon#	6 939.6	0.0351	NA	NA
Ratio sum		0.9473		0.7464

† From the producer.

‡ Not available.

§ Corrected for turbulent Schmidt number.

¶ After Sharpe and Harper (2002).

Design criteria provided by the producers.

factor was 60% of that from the FW operation, partly because of the larger N being removed from the systems as finished animals and the greater percentage of N₂ emissions. These comparisons exemplify the danger of basing emissions on animal numbers. Emissions based on a suggested emission factor related to steady-state live weight showed similar emission factors for the three production units measured. In disagreement with previous and current estimates of NH₃ emissions from confined animal feeding operation production systems, we found much smaller emissions of NH₃ from the housing, lagoons, and fields using independent measurements for a systems analysis of N in the farming operations, accounting for approximately 95% of the N entering the operation as feed. Although gaseous N losses from lagoon-based manure storage systems might not be as deleterious to air quality as has been previously thought, in terms of energy efficiency (N conservation and recycling), this type of waste management system is highly inefficient with 48% (in the FF operation) and 65% (in the FW operation) of N fed lost as NH₃ and N₂ (with a small amount as N₂O) during storage and land application with no further reuse of it in the agroecosystem.

Clearly, there is a critical need for improvement in N conservation from this type of waste management system. Systems analysis of a farming operation, using independent component measurements, provides a more realistic picture of nutrient dynamics. Continued studies are in progress to evaluate different housing and waste-application emissions, establish mechanisms for denitrification in these anaerobic lagoon systems, develop new technologies for evaluating trace gas emissions from sources (such as farming systems), and develop new concepts for conserving nutrients in the production systems.

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