REDUCTION OF AMMONIA EMISSIONS FROM TREATED ANAEROBIC SWINE LAGOONS

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ABSTRACT. *There is a need for treatment technologies that can effectively address environmental concerns associated with manure in confined animal production, including accurate assessment of their environmental benefit. These technologies must be able to capture nutrients, kill pathogens, and reduce emissions of ammonia (NH_3) and nuisance odors from manure. To meet these needs, a wastewater treatment plant was demonstrated at full-scale in one of two 4,360-pig production units on a finishing farm in Duplin County, North Carolina. The treatment system installed in the first production unit combined new treatment technologies consisting of solid-liquid separation with removal of nitrogen and phosphorus from the liquid phase. The second production unit was used as a control to evaluate emission improvements using traditional anaerobic lagoon technology. Both production units had similar production management and lagoons with similar surface area (about 0.9 ha each) and wastewater design. As the treatment system recovered the manure solids and replaced the anaerobic lagoon liquid with cleaner water, it converted the anaerobic lagoon into a treated water pond. Our objective was to study changes in NH_3 emissions as a result of improved water quality. The study was done one year after lagoon conversion into aerated pond as a result of manure treatment and included cold and warm weather conditions. Passive flux samplers were used to measure NH_3 gas fluxes from both lagoon systems. Average total ammoniacal N (TAN) concentrations in lagoon liquid were 31 and 388 mg/L for the treated and traditional systems, respectively. We found that free NH_3 (FA) concentration, which integrates TAN, temperature, and pH into a single factor, explained 90% of the variation in NH_3 emissions from these lagoon systems. Lower N concentrations in the converted lagoon substantially reduced annual NH_3 emissions by 90% with respect to those found in the traditional anaerobic lagoon. Ammonia emissions from the converted lagoon totaled 1,210 kg N/lagoon/year (or 1,311 kg N/ha/year). This annual rate compares with NH_3 emissions of 12,540 kg N/lagoon/year (13,633 kg N/ha/year) from the traditional lagoon. Overall, these results demonstrate that production of clean water using new wastewater technologies can accelerate lagoon cleanup and substantially reduce ammonia emissions from confined animal production.*

Keywords. *Ammonia emissions, Ammonia flux, Ammonia volatilization, Anaerobic swine lagoons, Free ammonia, Manure, Nitrogen.*

Anaerobic lagoons are widely used to treat and store liquid manure from confined swine production facilities. During lagoon treatment, urea and other organic N compounds contained in urine and feces are converted into ammoniacal N that can contribute to emission of ammonia gas (NH_3) into the atmosphere. The intensification of confined swine production since 1990 across North Carolina's Coastal Plain region has been correlated with increasing atmospheric ammonium deposition and air pollution (Battye et al., 2003; Cowling et al., 1998; Walker et al., 2000a, 2000b). Ammonia emission is a major environmental concern because of potential air pollution and the detrimental effect of nitrogen (N) deposition on nutrient-sensitive ecosystems (Cowling et al., 1998; Mallin, 2000). Thus, it is critical to develop alternative methods of N management in swine operations that can reduce NH_3 emissions (Erisman and Monteny, 1998). In particular, there is major interest in North Carolina in developing swine manure treatment systems that can eliminate environmental problems associated with anaerobic lagoons.

Several studies described methods to reduce NH_3 emissions from swine lagoons, including use of additives (McCrory and Hobbs, 2001), mechanical aeration of the lagoon (Heber et al., 2002; Westerman and Zhang, 1997), and lagoon covers (Funk et al., 2004 a, 2004b; Miner et al., 2003; Zahn et al., 2001). However, widespread objection to the use of anaerobic lagoons for swine manure treatment in North Carolina prompted a state government-industry framework to give preference to alternative technologies that directly eliminate anaerobic lagoons as a method of treatment. This framework established an agreement between government and the swine industry to develop and demonstrate environmentally superior waste management technologies (EST) that would capture nutrients, kill pathogens, reduce nuisance odors and NH_3 emissions, and eliminate discharge (Williams, 2001). In July 2005, only one on-farm technology out of 18 diverse technologies evaluated was determined to meet

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the environmental performance criteria necessary for EST. This on-farm treatment technology treated the entire waste stream from a swine production unit using a solids separation, nitrification/denitrification, and soluble phosphorus removal system (Vanotti et al., 2005; Williams, 2004). It effectively replaced anaerobic lagoon treatment by discontinuing the loading of liquid raw manure into the lagoon. In turn, the recycled clean water promoted the conversion of the old anaerobic lagoon into an aerobic water storage pond in less than a year (Vanotti, 2004). The remarkable changes in water quality, as a result of storing treated effluent in the old lagoon, led to this investigation of the quantification of NH_3 emission reductions from the cleaned operation.

A number of techniques have been developed to quantify NH_3 emissions from agricultural sources (Phillips et al., 2001). Among these techniques, the method of NH_3 measurement by passive flux samplers described by Schjoerring et al. (1992) has been widely used to measure NH_3 emissions from field plots (Marshall et al., 2001; Sherlock et al., 2002; Wu et al., 2003) and manure storage structures (Rhode and Karlsson, 2002; Sommer, 1997) because of its simplicity, precision, reliable direct measurement of NH_3 fluxes, and relative lower cost with respect to other NH_3 measurement methods (Phillips et al., 2001). For these reasons, the passive flux sampler method was used for measuring NH_3 emissions in our study.

Ammonia emissions from traditional anaerobic swine lagoons depend on several factors, such as total ammoniacal N concentration, pH, temperature, wind speed, chemical and microbiological activities, and material transport processes (Arogo et al., 2003). In particular, NH_3 emissions from anaerobic swine lagoons have been shown to increase with total ammoniacal N concentrations and temperatures (Aneja et al., 2000). Therefore, it appears obvious that improved water quality, such as lower nitrogen (N) levels in a converted lagoon, will substantially reduce ammonia emissions. The purpose of this research was to quantify the magnitude of this reduction in a converted lagoon compared with a traditional anaerobic lagoon, both with similar construction design and under similar animal production management. In addition, we determined the influence of lagoon N levels and climatic factors on NH_3 losses from both converted and traditional anaerobic lagoon systems.

Materials and Methods

Site Description

The study was conducted on Goshen Ridge Farm near Mount Olive, Duplin County, North Carolina. Two years before we started the emission measurement study, in 2002, the operation had three units under identical animal production and waste treatment management. Each unit had six barns with 4,360 finishing pigs and a traditional anaerobic lagoon for treatment and storage of manure, but only two units were used in this study. Manure was collected under the barns using slatted floors and a pit-recharge system typical of many farms in North Carolina (Barker, 1996). In each production unit, liquid manure contained in the pits was drained weekly by gravity to the traditional anaerobic lagoons. Lagoon supernatant effluent was then used to recharge the pits of both production units and facilitate flushing of the newly accumulated manure. Lagoon dimensions, monthly average live animal weight (LAW) computed from farm production records, and N loads are presented in table 1. The relationship between N production by pigs and their weight was 0.29 kg N/1000 kg LAW/day (Vanotti, 2004).

One year before this study was conducted, a full-scale wastewater treatment system was started in 2003. It treated all raw manure produced in unit 1 (fig. 1). Even though waste treatment methods in both production units were substantially different, animal production management remained the same. The treatment system installed in unit 1 combined solid-liquid separation with removal of N and phosphorus (P) from the liquid phase. The system treated an average of 39 m^3 per day of raw manure flushed from the barns in three sequential steps (Vanotti, 2004). The first step flocculated solids in raw flushed manure using polyacrylamide and separated solids from liquid. This step produced 596 Mg (metric tons) of separated solids per year that were transported off-site and converted to organic plant fertilizer, soil amendments, or energy. In the second step, N management to reduce NH_3 emissions was accomplished by passing the liquid through a module where immobilized nitrifying bacteria transformed total ammoniacal N into nitrate, and denitrifying sludge further transformed nitrate into N_2 gas. Subsequent alkaline treatment of the wastewater in a P module precipitated P as calcium phosphate and killed pathogens. This sequence of treatment produced an effluent with low levels of solids, nitrogen, and phosphorus. The treated water was recycled in a closed loop to refill the barn pit recharge system (13 m^3/d), and excess treated water (26 m^3/d) was stored in the lagoon and later used for crop irrigation. As the treatment system recovered the manure solids and replaced the anaerobic lagoon liquid with clean water, it transformed the anaerobic lagoon into a treated water pond. Changes in water quality before and after full-scale plant treatment was implemented are summarized in table 2. One year after the treatment system was started and water quality improvements were evident, we measured NH_3 emissions in both lagoons (fig. 1). The converted lagoon after the treatment plant started operation is referred to throughout this article as lagoon 1, or the treated lagoon. The control lagoon using traditional management in 2004 is referred to as lagoon 2, or the traditional lagoon.

Table 1. Main characteristics of the two production units.

Production Unit	Lagoon Surface (ha)	Lagoon Volume (m^3)	Live Animal Weight (kg)		Total N Load	
			Steady state[a]	Range	kg/day[b]	kg/year
1	0.90	24,145	224,581	13,205 - 377,851	65.1	23,762
2	0.92	22,356	196,636	0 - 367,769	57.0	20,805

[a] Monthly mean of six barns (2003-2004, n = 24).
[b] Total N load = (kg steady-state LAW × 0.29 kg N/1000 kg LAW/day)/1000.

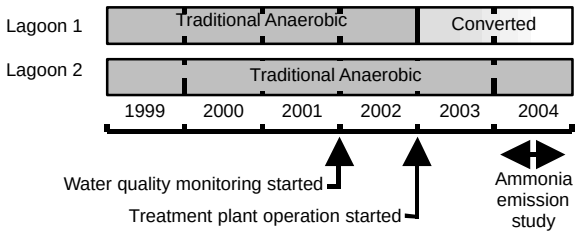

Figure 1. Management of lagoons 1 and 2, Duplin County, North Carolina. Units 1 and 2 had the same animal production management throughout the life of the operation.

Water Quality Analysis

All water quality analyses were performed according to Standard Methods (APHA, 1998). Chemical analyses consisted of pH, total ammoniacal N (TAN), total Kjeldahl N (TKN), nitrite plus nitrate-N (NO_2 + NO_3-N), chemical oxygen demand (COD), and 5-day biochemical oxygen demand (BOD). Water pH was determined electrometrically (Standard Method 4500-H^+ B). Nitrite plus nitrate-N was determined using Standard Method 4500-NO_3^- F after filtration of samples through a 0.45 μm membrane filter (Gelman type Supor-450, Pall Corp., Ann Arbor, Mich.). The same filtrate was used to measure TAN using Standard Method 4500-NH_3 G. This method determined both ionized and un-ionized ammonia forms (TAN = NH_4^+-N + NH_3-N). The concentration of un-ionized form (NH_3) or free ammonia (FA) is in equilibrium with the ionized form (NH_4^+):

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

The concentration of FA (mg NH_3/L) in lagoon liquid was calculated with equations 2 and 3 (Anthonisen et al., 1976) using measured water temperature (˚C), pH, and TAN concentration (mg N/L):

$$FA = \left(\frac{17}{14}\right) \times \left(\frac{TAN \times 10^{pH}}{\frac{K_b}{K_w} \times 10^{pH}}\right) \quad (2)$$

where K_b and K_w are ionization constants for NH_3 and H_2O, respectively. The K_b/K_w ratio is related to water temperature (T) according to:

$$\frac{K_b}{K_w} = \exp\left[\frac{6344}{273+T}\right] \quad (3)$$

Table 2. Typical wastewater characteristics before and after full-scale plant treatment, Duplin County, North Carolina (Vanotti, 2004).

Constituent[a]	Influent (mg/L)[b]	Effluent (mg/L)[c]
TAN	872	11
TKN	1584	23
NO_2 + NO_3-N	1	224
TSS	11051	264
COD	16138	445
BOD	3132	10
pH[d]	7.6	10.5

[a] TAN = total ammoniacal N, TKN = total Kjeldahl N, NO_2 + NO_3-N = nitrate plus nitrate N, TSS = total suspended solids, COD = chemical oxygen demand, and BOD = biochemical oxygen demand.
[b] Raw wastewater flushed from the hog house. Data are means, n = 121.
[c] After sequential treatment: solid/liquid separation − biological N removal − lime precipitation.
[d] Measured in pH units.

TKN was determined using acid digestion and the same method to determine TAN adapted to digested extracts (Technicon, 1977). For COD determination, we used the closed reflux colorimetric method (Standard Method 5520 D), and BOD was determined using the 5-day BOD test (Standard Method 5210 B). Total suspended solids (TSS) were determined by retaining solids on a glass-fiber filter dried to 105˚C (Standard Method 2540 D). Total solids (TS) were determined after drying the water samples at 105˚C (Standard Method 2540 B).

Environmental Parameters

Environmental parameters were measured separately for each lagoon. Meteorological measurements consisted of air temperature, wind speed and direction, and relative humidity at about 2 m above the lagoon liquid surface. Air temperature, wind speed, and wind direction were measured and stored in a Weather Wizard III instrument (Davis Instruments Corp., Hayward, Mass.). Relative humidity (RH) was measured and stored in a HOBO H8 Pro RH/Temp data logger (Onset Computer Corp., Bourne, Mass.). The environmental parameter dataset was completed with temperature of the lagoon liquid measured at 0.15 m depth and stored in a StowAway TidbiT data logger (Onset Computer Corp., Bourne, Mass.). All environmental parameters were recorded at 5 min intervals throughout each emission sampling period and averaged every 23 h.

Emission Sampling

Ammonia emissions were determined with passive flux samplers (Ferm tubes) using the method described by Schjoerring et al. (1992). A passive flux sampler consisted of two parallel sampling units in opposite directions (fig. 2). Each sampler unit had three 7 mm i.d. glass sampling tubes (length 100, 100, and 23 mm, Mikrolab Aarhus A/S, Hoejbjerg, Denmark) connected by silicon rubber tubing. The two 100 mm tubes were coated with oxalic acid on the inner surface at about 70 mm of the tube length. The 23 mm tube had a nozzle glued onto the free end. The nozzle was made of a 0.05 mm thick stainless steel disc with a 0.5 mm radius hole in its center. The purpose of the nozzle is to decrease the air speed inside the tubes to achieve a low friction resistance and high collection efficiency (Schjoerring et al., 1992).

The passive samplers were mounted evenly separated (0.75 m) at four heights (0.90, 1.65, 2.40, and 3.15 m) on four masts, positioned at 0˚, 90˚, 180˚, and 270˚ angles around the circumference of the circular plots shown in figure 3. This layout maximized the lagoon surface enclosed within a circular plot (fig. 3). This circular plot design was required for the mass balance method used to estimate NH_3-N vertical fluxes using passive samplers (Schjoerring et al., 1992).

Lagoon water levels were recorded to determine the height of samplers with respect to the surface of the lagoons. Due to fluctuation in lagoon levels, the average height of the first sampler on each mast was 0.9 ±0.2 m and 0.9 ±0.3 m with respect to the liquid surface, for lagoon 1 and 2, respectively. This fluctuation produced changes of <2% on the lagoon area enclosed within the circular plot.

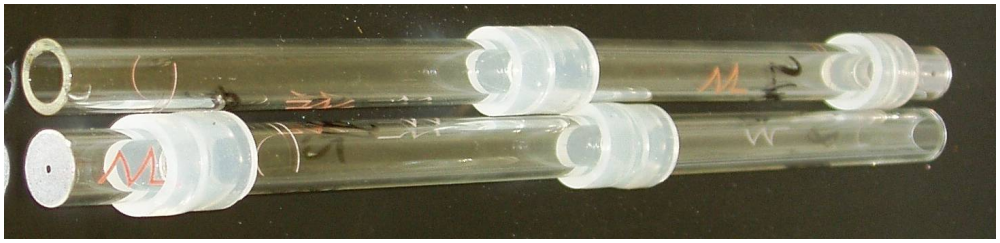

Figure 2. Passive ammonia flux sampler consisting of two parallel sampling units (Sommer et al., 1996). Each sampling unit has a nozzle at the end. One end of the sampler is exposed to the emission source, and the other end faces the surroundings.

Nine data collection periods lasting about 23 h each were scheduled from February to November 2004 for the two lagoons. After exposure, the samplers were retrieved from the masts and transported to the laboratory in closed plastic containers. In the laboratory, the sampler tubes were disconnected and NH_3 was extracted from individual 100 mm tubes by dissolving the oxalic acid coating with 3.0 mL of ultrapure (MilliQ) water per tube. The liquid extracts were analyzed for NH_3-N using the salicylate method (Method US-696C-82W; Bran+Luebbe, 1999) on a Bran+Luebbe Autoanalyzer III (Bran+Luebbe Analyzing, Inc., Roselle, Ill.). Analyses included blanks that consisted of two unexposed sampler sets assembled, stored, transported, and extracted in the same way as the exposed samplers. Each NH_3 emission measurement involved the analysis of 72 liquid extracts from the tubes.

Ammonia Flux Determination

The vertical NH_3 flux for each lagoon was determined from net horizontal fluxes by application of a mass balance method (Schjoerring et al., 1992). The average horizontal flux of NH_3 through two glass tubes facing the same direction was estimated according to the following equation (Sommer et al., 1996):

$$\text{Horizontal flux } (F_{hz}) = \frac{(C_1 + C_2)V}{2 \times \pi \times r^2 \times K \times \Delta t} \qquad (4)$$

where C_1 and C_2 are the NH_3-N concentrations (μg NH_3-N/L) in two 100 mm tubes facing in the same direction (either background or emission source), V is the volume of water used to dissolve NH_3-N sorbed on the tubes, r is the radius of the hole in the stainless steel disc (0.5 mm), Δt is the time between the start and conclusion of the measurements, and K is an empirical correction factor for wind speed and direction effects on the sampler (K = 0.77; Schjoerring et al., 1992). Daily emission rates were estimated by calculating the net vertical fluxes of NH_3 at each lagoon by stepwise summation of the net horizontal flux over the height intervals covered by the passive flux samplers using the following equation (Schjoerring et al., 1992):

$$\text{Vertical flux } (F_v) = \frac{1}{2r} \sum_{h=1}^{h=4} \sum_{m=1}^{m=4} (F_{hz,e} - F_{hz,s}) \Delta h \qquad (5)$$

where $F_{hz,e}$ is the horizontal flux from the emission source area (calculated from NH_3 concentrations in tubes facing to the inside of the circular plot, fig. 3), $F_{hz,s}$ is the horizontal flux from the surroundings or background (calculated from NH_3 concentrations in tubes facing to the outside of the circular plot, fig. 3) at each height of measurement (h) with each mast (m), Δh is the height interval between flux samplers, and r is the radius of the circular plot enclosing the emission source. Net vertical NH_3 fluxes were reported as emission rates in kg NH_3-N/ha/d.

Data management, descriptive statistics (PROC MEANS), regression (PROC REG), and mean comparison (PROC TTEST and PROC UNIVARIATE) analyses for water quality, NH_3 emission, and environmental parameters data were performed with version 8.0 of SAS (SAS, 1999). Total annual NH_3 emissions were obtained by fitting a Gaussian curve to daily NH_3-N emission data versus day of the year using Prism 4 software (Miller, 2003). The parameter AREA of this curve represents the total annual NH_3 emissions, as follows:

$$y = \left(\text{AREA} / \left[\text{SD} \times (2\pi)^{0.5}\right]\right) \times \exp\left(-0.5 \times \left[(x - \text{MEAN})/\text{SD}\right]^2\right) \qquad (6)$$

where y is the daily NH_3 emission, x is the day of the year, and AREA, SD, and MEAN are parameter estimates. This curve was selected because it provided a good fit to the changes in daily NH_3 emissions throughout the year both in terms of R^2 and normality of residuals (residuals = emissions observed − emissions predicted) using the Shapiro-Wilk test statistic (Delong and Yuan, 1988).

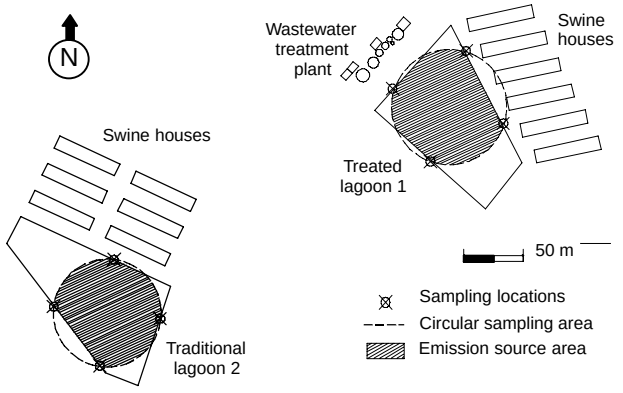

Figure 3. Schematic diagram of monitoring setup for the ammonia emission study for treated lagoon 1 and traditional lagoon 2, Duplin County, North Carolina.

Results and Discussion

Lagoon Water Quality

Monitoring of water quality in the lagoon was initiated in 2002, one year before treatment started in production unit 1 (fig. 1). The criterion to determine water quality improvement was the reduction in concentration of various water

Table 3. Changes in water quality in three consecutive years for lagoon 1 before and after treatment plant was operational, and traditional anaerobic lagoon 2 (control), Duplin County, North Carolina.[a]

Sampling Period	Lagoon	Treatment Plant	pH	TAN[b] (mg/L)	TKN (mg/L)	$NO_2 + NO_3$ (mg/L)
Jan.-Dec. 2002	1	N	8.0 (0.1)	464 (98)	506 (108)	0.08 (0.20)
	2	N	8.0 (0.2)	467 (118)	521 (122)	0.07 (0.21)
Jan.-Dec. 2003	1	Y	8.1 (0.1)	186 (129)	230 (138)	4.1 (5.8)
	2	N	7.9 (0.1)	446 (102)	522 (127)	0.43 (1.4)
Jan.-Dec. 2004	1	Y	8.1 (0.3)	37 (32)	76 (34)	20 (16)
	2	N	8.0 (0.2)	364 (88)	406 (79)	n.d.[c]

[a] Data are annual means (standard deviation) of duplicate monthly composite samples.
[b] TAN = total ammoniacal N, TKN = total Kjeldahl N, and $NO_2 + NO_3$-N = nitrate plus nitrate N.
[c] n.d. = not detected.

Table 4. Water quality characteristics in treated lagoon 1 and traditional lagoon 2 during the ammonia emissions monitoring period (Feb. to Nov. 2004).[a]

Lagoon	Treated	pH	TAN[b] (mg/L)	TKN (mg/L)	TS (mg/L)	COD (mg/L)	BOD (mg/L)
1	Y	8.1 (0.3)	31 (26)	73 (31)	2312 (180)	545 (202)	50 (30)
2	N	8.0 (0.2)	388 (109)	431 (103)	2931 (218)	1399 (406)	186 (115)
Level of Significance (P)							
Paired *t*-test		0.33	0.0001	0.0001	0.0001	0.0015	0.0148
Wilcoxon sign test		0.40	0.0039	0.0039	0.0039	0.0039	0.078

[a] Data are means (standard deviation) of duplicate monthly composite samples.
[b] TAN = total ammoniacal N, TKN = total Kjeldahl N, TS = total solids, COD = chemical oxygen demand, and BOD = biochemical oxygen demand.

quality indicators (TAN, TKN, TS, COD, and BOD). In 2002, both lagoons received flushed raw manure from the barns. Thus, the two anaerobic lagoons had similar annual mean pH, TAN, TKN, and $NO_2 + NO_3$-N concentrations (Jan. to Dec. 2002; table 3). Beginning in February 2003, manure flush to lagoon 1 was halted and 100% of the liquid manure generated in the adjacent six barns was processed through the wastewater treatment plant (fig. 3). The quality of the liquid in lagoon 1 rapidly improved during 2003 as cleaner effluent from the treatment plant replaced anaerobic lagoon liquid, while water quality in lagoon 2 remained unchanged. In lagoon 1, the transition from anaerobic to aerobic water storage was noticeable in the first year of treatment. Dissolved oxygen (DO) concentrations in fall 2003 and winter 2004 (Oct. 2003 to March 2004, $n = 5$) at 0.15 m below the liquid surface averaged 3.45 mg/L in lagoon 1 and 0.52 mg/L in lagoon 2 (Vanotti, 2004). In 2003, annual average TAN and TKN levels in lagoon 1 declined 58% and 56%, respectively, with respect to lagoon 2. By 2004, differences in TAN and TKN concentrations in lagoon 1 with respect to lagoon 2 were even larger; on average, TAN declined 90% and TKN declined 81% (table 3).

Table 4 shows water quality data for lagoons 1 and 2 corresponding to the NH_3 emission monitoring period (Feb. to Nov. 2004). Statistical tests showed significant differences between lagoons 1 and 2 in TAN, TKN, TS, COD, and BOD concentrations. These differences in water quality characteristics between lagoons produced remarkable differences in NH_3 emissions, as shown in the following section.

Effects of Environment on Ammonia Emissions

Environmental parameters (air and water temperature, relative humidity, and wind speed and direction) were similar for both lagoons on same sampling dates (table 5). This similarity in environmental conditions, plus the fact that animal production management in both units was also similar, reduced experimental error and made interpretation of NH_3 emissions simpler. For lagoon 1, NH_3 emission rates were in the range of 0.9 to 12.5 kg NH_3-N/ha/d. For lagoon 2, NH_3 emission rates were in the range of 2.5 to 73.4 kg NH_3-N/ha/d. Ammonia emission rates from the traditional lagoon 2 were within the range of 0.6 to 104 kg NH_3-N/ha/d reported for North Carolina's anaerobic lagoons (Arogo et al., 2003).

Wind Speed

We considered the effect of wind speed on NH_3 emissions since it strongly influences volatilization in general (Schjoerring et al., 1992). Several researchers have reported the same effect in anaerobic lagoons (Harper et al., 2000; Liang et al., 2002; Zahn et al., 2001). However, in our study, wind speed poorly explained the variation of NH_3 emissions. Using data from table 5, average wind speed had a poor correlation with NH_3 emission rates ($R^2 = 0.38$, $y = 48.5 - 19.6x$, $P < 0.01$). Sommer (1997) also found poor correlation between wind speed and NH_3 emission in a study that measured emissions from slurry tanks using passive flux samplers. It is important to note that this poor correlation was due to the use of average ammonia emission rates and wind speed measured over periods of 24 to 48 h instead of instantaneous measurements. Since we also used average wind speeds to correlate with daily emissions provided by the passive flux method, we conclude that the effect of wind speed on emissions cannot be properly evaluated in this study.

Temperature

With respect to air temperature, NH_3 emissions in geographic regions with intensive livestock production usually display strong seasonal patterns, with maximums occurring during the summer (Gilliland et al., 2003). This seasonal trend has also been observed in anaerobic swine lagoons where NH_3 fluxes are about 80% to 90% higher during summer as compared to winter and fall conditions (Aneja et al., 2000). In our study, we found similar seasonal patterns for both lagoons (table 5). During cold weather

Table 5. Ammonia emission rates and weather conditions during monitoring period (Feb. to Nov. 2004) and means during cold and warm weather for treated lagoon 1 and traditional lagoon 2 (control), Duplin County, North Carolina.

Lagoon	Treated	Date (2004)	Emission Rate (kg NH_3-N/ha/d)	Mean Daily Temp. (°C) Water	Mean Daily Temp. (°C) Air	Mean Relative Humidity (%)	Mean Wind Speed (m/s)	Predominate Wind Direction
1	Y	18 Feb.	0.9	6.5	6.5	55.2	1.2	W
2	N	18 Feb.	2.5	6.5	5.6	55.2	1.6	WNW
1	Y	23 Feb.	3.2	12.6	8.5	75.9	1.7	NE
2	N	23 Feb.	4.5	12.6	8.4	75.9	2.5	NE
1	Y	10 March	4.1	14.1	5.9	60.8	3.9	N
2	N	10 March	7.1	14.1	5.9	60.7	2.2	N
1	Y	20 April	<0.0[a]	25.0	22.0	67.5	1.3	S
2	N	20 April	49.3	23.6	21.8	69.8	0.3	SSW
1	Y	19 May	6.7	27.7	23.7	79.4	1.1	S
2	N	19 May	69.6	28.8	23.9	81.6	0.6	WSW
1	Y	15 June	11.0	28.6	25.6	79.6	1.5	S
2	N	15 June	73.4	30.2	25.9	81.7	0.5	S
1	Y	28 July	12.5	28.7	25.6	86.4	1.8	SSE
2	N	28 July	68.5	27.6	25.8	87.1	0.6	S
1	Y	22 Sept.	1.1	25.0	21.1	74.3	1.0	N
2	N	22 Sept.	53.1	25.2	20.6	76.2	0.7	N
1	Y	09 Nov.	1.3	14.1	5.4	71.2	1.5	N
2	N	09 Nov.	14.5	15.4	4.8	74.3	1.3	NE
Cold Weather[b]								
1	Y		2.4 (1.5)	11.8 (3.6)	6.6 (1.3)	65.8 (9.5)	2.1 (1.3)	
2	N		7.2 (5.3)	12.1 (3.9)	6.2 (1.5)	66.5 (10.2)	1.8 (0.5)	
		t-test:	NS[c]	NS	NS	NS	NS	
Warm Weather[d]								
1	Y		7.8 (5.1)	27.0 (1.9)	23.6 (2.1)	77.4 (7.0)	1.3 (0.3)	
2	N		62.8 (10.8)	27.1 (2.7)	23.6 (2.4)	79.3 (6.6)	0.5 (0.1)	
		t-test:	0.0001	NS	NS	NS	0.001	

[a] Vertical net flux <0. Net flux measurements using the passive flux sampler method are calculated only when the NH_3 flux from the emission source is greater than that of the surroundings (Sommer et al., 1996).
[b] Means (standard deviation) of February, March, and November 2004, air temperature <10°C.
[c] NS = non-significant differences (P > 0.01).
[d] Means (standard deviation) of April to September 2004, air temperature >10°C.

(Feb., March, and Nov. 2004; air temperature <10˚C), emission rates in both lagoons were low (<7.2 kg NH_3-N/ha/day). During warm weather (April to Sept. 2004; air temperature >10˚C), emissions increased in both lagoons, but emissions rates were markedly higher in the traditional lagoon. A significant eight-fold difference was observed in NH_3 emission rates between treated lagoon 1 (7.8 NH_3-N/ha/d) and traditional lagoon 2 (62.8 NH_3-N/ha/d) (paired t-test, $P < 0.01$, table 5). This seasonal difference in NH_3 emissions was evident in the horizontal flux profiles of each lagoon (fig. 4). These profiles showed that large NH_3 horizontal fluxes (5 to 20 kg N/ha/d) consistently occurred only in the traditional anaerobic lagoon (lagoon 2) when mean daily air temperature was >10˚C (fig. 4b).

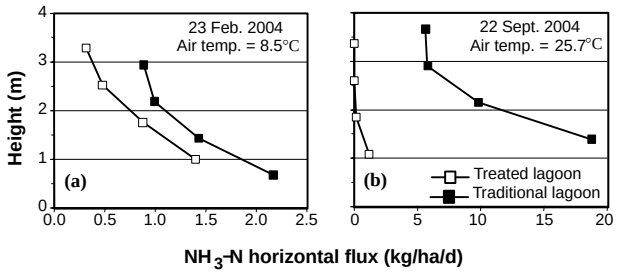

Figure 4. Typical ammonia horizontal flux profiles measured above treated lagoon 1 and traditional lagoon 2 during (a) cold weather and (b) warm weather. Fluxes were calculated using equation 4.

The effect of air temperature on NH_3 emission rates was not clear when data from both lagoons were combined and provided poor understanding of cause and effect. This is because water quality, in particular TAN concentrations, was markedly different between lagoons (table 4). A better understanding of the effect of air temperature on the NH_3 emissions was obtained when data for both lagoons were plotted separately (fig. 5a). The only significant relationship ($R^2 = 0.97$, $P < 0.01$) between NH_3 emission rate and air temperature was observed in traditional lagoon 2 (fig. 5a). Because TAN concentration was uniformly low in lagoon 1, NH_3 emission rates were not related to air temperature in this lagoon (fig. 5a). The same trends were obtained when lagoon water temperature was used as the independent variable (fig. 5b).

Background

Vertical net flux measurements using the passive flux sampler method are calculated only when horizontal NH_3 flux from the emission source is greater than that of the surroundings or background (Sommer et al., 1996). Atmospheric NH_3 re-deposition from nearby spray fields and animal housing adjacent to the lagoon can contribute to significant background NH_3, but this was not a problem in

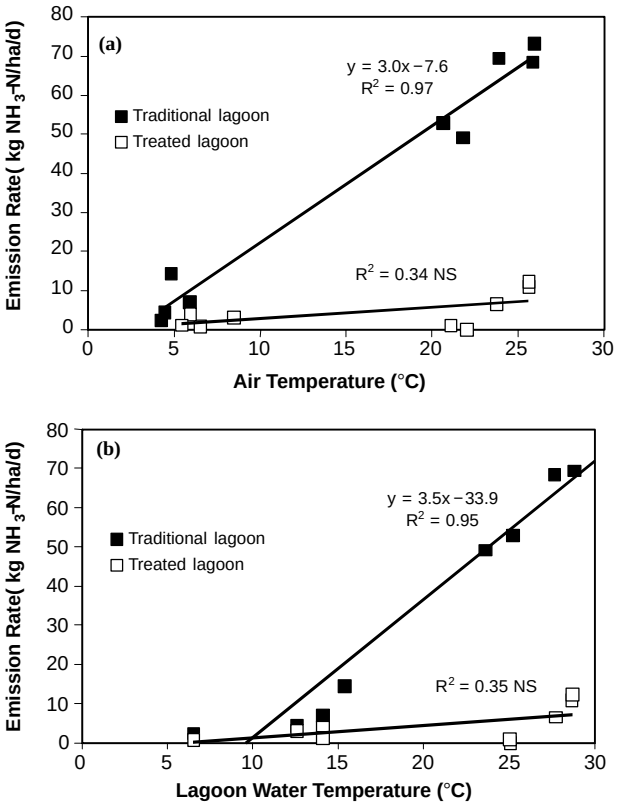

Figure 5. Temperature effect on ammonia emissions rates: (a) air temperature, and (b) lagoon water temperature. NS = non-significant temperature effect.

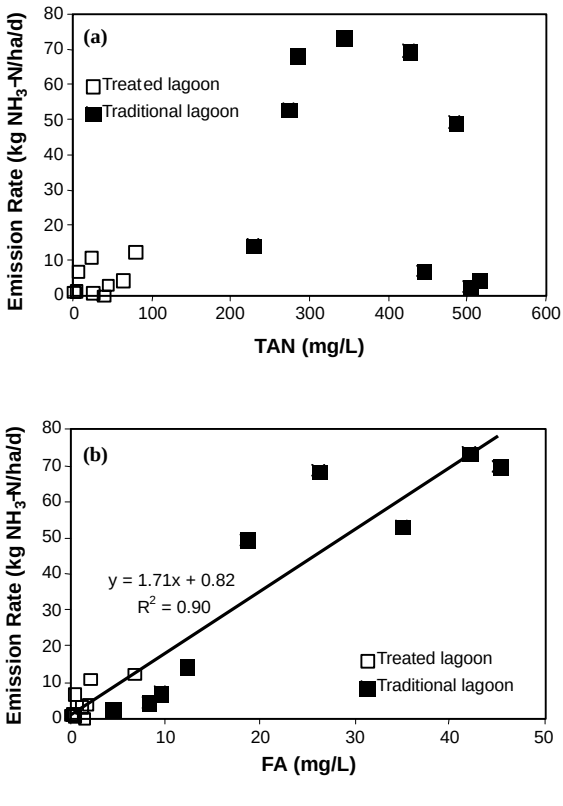

Figure 6. Effect of total ammoniacal N (TAN) and free ammonia N (FA) concentrations on ammonia emission rates using combined data from both lagoons. FA calculated according to equations 2 and 3.

our study. Of all 18 measurements, only one vertical net NH_3 flux was not possible to determine because horizontal fluxes in the surroundings were higher than those of the lagoon emission source (20 April 2004, <0 kg/ha/d, table 5). Overall, NH_3 fluxes from the surroundings of both lagoons were much lower than those from the emission source (table 6). Most important to the paired lagoon comparison in this study, fluxes from the surroundings were not significantly different between lagoons during cold or warm weather (paired *t*-test, P > 0.01, table 6).

Table 6. Vertical ammonia fluxes components (emission source and surroundings) for cold and warm weather in treated lagoon 1 and traditional lagoon 2, Duplin County, North Carolina.[a]

Weather Conditions	Lagoon	Treated	Vertical Fluxes (kg NH_3-N/ha/d)	
			Source	Surroundings
Cold[b]	1	Y	4.9 (1.8)	2.5 (1.0)
	2	N	9.7 (6.4)	2.5 (1.4)
		t-test:	NS[c]	NS
Warm[d]	1	Y	12.5 (7.2)	8.9 (7.1)
	2	N	78.9 (14.2)	16.1 (6.3)
		t-test:	0.0001	NS

[a] Calculated using equation 5.
[b] Means (standard deviation) of February, March, and November 2004, air temperature <10°C.
[c] NS = non-significant differences (P > 0.01), paired *t*-test.
[d] Means (standard deviation) of April to September 2004, air temperature >10°C.

TAN vs. Free Ammonia

Lagoon liquid pH and TAN concentration may play an important role in the potential for NH_3 volatilization (Sommer, 1997; Aneja et al., 2001; Liang et al., 2002). In our study, NH_3 emission rates did not relate to pH (R^2 = 0.03) because pH of the lagoon liquid remained mostly unchanged in the lagoons studied (table 4). Ammonia-N emission rates were also not related to TAN when TAN was used as a single independent variable (R^2 = 0.22; fig. 6a). Nevertheless, we found that free NH_3 (FA) concentration in lagoon liquid was an excellent predictor of NH_3 emissions in these lagoons. Free NH_3 accounts for the joint effect of TAN, pH, and temperature (eqs. 2 and 3) and constitutes the pool of ammoniacal N readily available to loss by volatilization. Thus, NH_3 losses had a significant linear response (P < 0.01) to increasing FA levels in lagoon liquid (fig. 6b). Concentration of FA in lagoon liquid explained 90% of the variation in NH_3 emissions observed in the study (fig. 6b).

Ammonia Emissions Reduction by Treatment

Total annual NH_3 emissions in both lagoon 1 and lagoon 2 were calculated by fitting a Gaussian curve to measured daily NH_3-N emission values (fig. 7). The total annual NH_3 emission for each lagoon is represented by the area under the curves in figure 7. On an annual basis (year 2004), NH_3 emissions from the treated lagoon totaled 1,210 kg N/lagoon/year (or 1,311 kg N/ha/year), compared to 12,540 kg N/lagoon/year (13,633 kg N/ha/year) from the traditional lagoon. Compared with the traditional lagoon, annual NH_3 emissions from the treated lagoon were reduced 90%. These

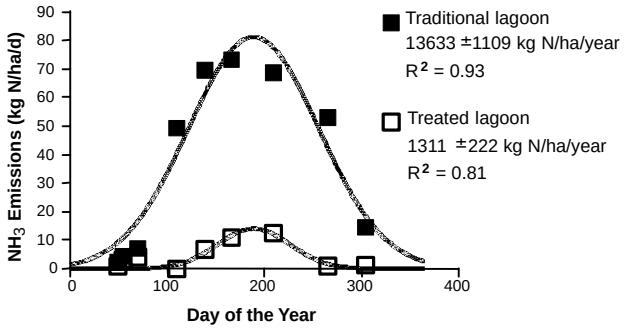

Figure 7. Change in daily rate of ammonia emissions throughout the year for the traditional lagoon and the treated lagoon. The total annual ammonia emissions are represented by the area under each regression curve (eq. 6) and indicated in the legend. Equation parameters are: AREA = 13,633, SD = 67.04, and MEAN = 188.9 for the traditional lagoon; and AREA = 1311, SD = 37.19, and MEAN = 189.1 for the treated lagoon.

emissions represent a 65% loss of the total annual N load of 20,805 kg produced by unit 2 versus just 5.5% of the total annual N load of 23,762 kg produced by unit 1 (table 1).

CONCLUSIONS

We conducted a study to determine the effects of water quality improvement in swine lagoons on NH_3 emission rates. This study was conducted on a facility with two adjacent swine production units that had similar animal production management and lagoon design, but their waste treatment and lagoon liquid characteristics were substantially different. In one production unit, a full-scale wastewater treatment plant produced clean effluent that in turn transformed the old lagoon into a water storage pond. In the other production unit, the traditional anaerobic lagoon treatment method was maintained as a control. We were therefore able to quantify environmental benefits of reducing NH_3 emissions by improving water quality with the use of a new technology that produces clean water.

In summary, our findings indicate:

- Ammonia-N losses were greatly influenced by average daily temperature. During cold weather, average emissions in both lagoons were <7.2 kg NH_3-N/ha/day. However, during warm weather, there was a significant eight-fold difference in mean daily NH_3 emissions between the treated lagoon (7.8 NH_3-N/ha/d) and traditional lagoon (62.8 NH_3-N/ha/d).
- Concentration of FA in lagoon liquid was a much better indicator of NH_3 emissions than TAN concentrations and explained approximately 90% of the variation in NH_3 emissions from both lagoons.
- Lower N concentrations in the converted lagoon substantially reduced annual NH_3 emissions by 90% with respect to emissions found in the traditional anaerobic lagoon. The NH_3 emissions from the treated lagoon totaled 1,210 kg N/lagoon/year (or 1,300 kg N/ha/year), compared to NH_3 emissions of 12,540 kg N/lagoon/year (13,600 kg N/ha/year) from the traditional lagoon.

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