

# Emissions Approaches and Uncertainties - Animal



## Global Inventory of Ammonia Emissions from Global Livestock Production and Fertilizer Use

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### Abstract

Ammonia is an important atmospheric pollutant with a wide variety of impacts ranging from aerosol formation, soil acidification, eutrophication, and biodiversity loss in ecosystems. One of the major global anthropogenic sources of atmospheric ammonia is animal waste generated in the production of meat, milk and other livestock products. We present a spatially explicit inventory of ammonia emission from global pastoral and mixed livestock production systems. Emissions are presented for grazing, animal houses and manure storage systems, and spreading of animal manure. We also discuss the major uncertainties in our inventory.

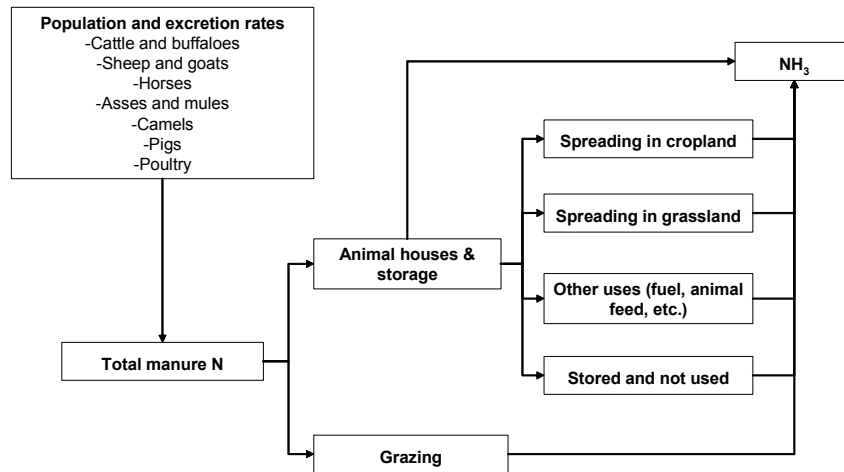
### Introduction

At present the global use of synthetic fertilizer nitrogen (N) is about 80 billion kg yr<sup>-1</sup>, and an even greater amount of animal manure N is generated in livestock production systems. The use of N fertilizer and the production of animal manure are expected to increase in the coming decades, particularly in developing countries (Bruinsma, 2003).

One of the major pathways of loss of N from agricultural systems is ammonia (NH<sub>3</sub>) volatilization. This mechanism may be responsible for the loss of 10-30% of the N fertilizer applied or the N excreted by animals, and is the major anthropogenic source of atmospheric NH<sub>3</sub>. Apart from NH<sub>3</sub>, livestock production facilities, manure storage areas, and manure field-application sites are sources of particulate matter (PM<sub>10</sub>, PM<sub>2.5</sub>), volatile organic compounds (associated with odor or precursors for ozone formation), hydrogen sulfide, greenhouse gases (methane, nitrous oxides), and pathogens. In this paper we concentrate on emissions of NH<sub>3</sub> from global livestock production systems.

Any gaseous NH<sub>3</sub> present in manure, soil, water or fertilizer can volatilize to the atmosphere. Ammonia volatilization is driven by the difference in NH<sub>3</sub> partial pressure between the air and manure, soil, crops or floodwater, which is determined by many interacting factors (Bouwman et al., 2002a). Ammonia is an important atmospheric pollutant with a wide variety of impacts. In the atmosphere NH<sub>3</sub> neutralizes a great portion of the acids produced by oxides of sulfur and nitrogen. An important part of atmospheric aerosols, acting as cloud condensation nuclei, consist of sulfate and nitrate neutralized to various extents by NH<sub>3</sub>. Essentially all emitted NH<sub>3</sub> is returned to the surface by deposition, which is known to be one of the causes of soil acidification (Van Breemen et al., 1982), eutrophication of natural ecosystems and loss of biodiversity (Bouwman et al., 2002b; Dise and Stevens, 2005).

In this paper we present a global inventory of emissions of NH<sub>3</sub> associated with livestock production and fertilizer use. This inventory is an update of the global emission inventory prepared in the framework of the Global Emission Inventory Activity (GEIA) project of IGBP-IGAC (Bouwman et al., 1997) and is a continuation of the work presented by Bouwman et al. (2005a,b,c). In this extended abstract we present global results, and briefly discuss uncertainties and research needs.



**Figure 1. Distribution of manure over different management systems. This is done for both pastoral and mixed/industrial systems.**

### Data and Methods

We update the GEIA 1 by 1 degree resolution (~110 by 110 km at the equator)  $\text{NH}_3$  emission inventory for the year 1990 (Bouwman et al., 1997) with data for the year 2000. In addition, the inventory is improved by using more detailed spatial information, by distinguishing different production systems and improved emission calculations for spreading of manure.

The global land cover map used in this study is based on a combination of two datasets from different satellite sensors, i.e. the IGBP-DisCover and the GLC2000 dataset (Klein Goldewijk et al., 2006). This land cover map includes cropland and grassland areas that are consistent with statistical information from FAO at the country scale and for some large countries at the state (U.S.A.) or provincial (China, Western Europe) level. The resolution is 5 by 5 minutes, which is about 8 by 8 km at the equator.

Within the areas of cropland and grassland we distinguish pastoral and mixed/industrial livestock production systems. In the pastoral production systems grazing is dominant and not integrated with cropping systems. Mixed/industrial systems have integrated cropping and livestock production, in which livestock production relies on a mix of food crops, crop by-products and roughage, consisting of grass, fodder crops, crop residues, and other sources of feedstuffs. In mixed systems the by-products of one activity (crop by-products, crop residues, and manure) often serve as inputs for another. We assume that the production system within a grid cell is mixed/industrial when cropland covers more than 15%. Otherwise the grid cell is considered to be dominated by pastoral production systems. This approach is similar to those of Kruska et al. (2003) and Bouwman et al. (2005c) and it yields a rough distinction between pastoral and mixed/industrial systems, although there may be large variation between countries.

With this land cover distribution as a basis, livestock manure is distributed over different systems (Figure 1). Ten animal categories are distinguished, including nondairy cattle, dairy cattle, buffaloes, pigs, poultry, sheep, goats, asses and mules, horses, and camels. We use data from FAO (2005) for animal populations and from national statistics for the states of the U.S.A., provinces of China and some regions in Western Europe. Associated N excretion rates are taken from Van der Hoek (1998). Excretion rates are assumed to be the same in mixed/industrial and pastoral systems.

First the livestock production for each animal category is divided into that in mixed/industrial and pastoral systems on the basis of data for world regions from Seré and Steinfeld (1996) and Bouwman et al. (2005c). On the basis of a-priori fractions for world regions manure is distributed within each country (or state or province) over grazing, animal housing and storage systems, other uses and stored but unused manure (Figure 1). The fraction grazing is derived from the ratio of grass to total feed in the ration of each animal category presented by Bouwman et al. (2005c). The fraction other uses and stored but unused manure is

based on data presented by Mosier et al. (1998). In industrialized countries we assume that half of the stored manure is applied to cropland and the other half to grassland. In developing countries 95% is applied to cropland. In a number of countries the a-priori estimates for the manure distribution lead to unrealistic annual input rates. In such cases, the manure is re-distributed until annual N input rates are less or close to pre-defined maximum rates, which are  $250 \text{ kg ha}^{-1} \text{ yr}^{-1}$  in mixed/industrial systems and  $125 \text{ kg ha}^{-1} \text{ yr}^{-1}$  in pastoral systems.

Data on total N fertilizer use are from FAO (2005), fertilizer N use by fertilizer type from IFA (2003), and fertilizer use by crop from IFA/IFDC/FAO (2003). The difference between total use and application to grassland is assumed to be applied to cropland and evenly spread over the area of total cropland from Klein Goldewijk et al. (2006).

Ammonia volatilization rates for animal housing and grazing systems are taken from Bouwman et al. (1997). Volatilization from spreading of animal manure and fertilizer is calculated with the empirical model presented by Bouwman et al. (2002a) based on crop type, manure or fertilizer application mode, soil CEC, soil pH, and climate. Fertilizer type is used as a model factor for calculating  $\text{NH}_3$  volatilization from fertilizers. All manure applied to cropland is assumed to be incorporated, while manure is applied to grassland by broadcasting. In the model incorporation leads to considerable reductions of  $\text{NH}_3$  loss of up to 50% compared to broadcasting. We tentatively use an emission factor of 20% for the categories other uses and stored but unused manure.

### Results and Discussion

The global N excretion by the ten animal categories amounts to  $112 \text{ Tg yr}^{-1}$ , of which 73% is generated in mixed/industrial systems, and 18% in pastoral systems (Table 1). The distribution over mixed/industrial and pastoral systems and the use of detailed spatial information allows for portraying the spatial variation of manure N inputs (Figure 2).

Within the mixed/industrial systems more than half of the manure N is collected in animal houses and storage systems, and 40% is excreted in pastures. For the pastoral systems 97% is excreted in pastures. According to our results about 10% of the global quantity of animal manure ends outside the agricultural system. This includes animal manure used as fuel, building material, animal feed or for other purposes.

The global  $\text{NH}_3$ -N emission from animal manure is about  $24 \text{ Tg yr}^{-1}$  (Table 1). More than 40% of this amount is from animal housing and storage systems, 30% is from grazing animals, and about 30% from spreading of animal manure. Emission of  $\text{NH}_3$  from N fertilizer use is about  $11 \text{ Tg yr}^{-1}$ .

We compared our results with the emission inventory presented by Bouwman et al. (1997). For this comparison we applied the updated N excretion rates used in this study. Our new estimate of  $\text{NH}_3$ -N emission from global livestock production for the year 2000 ( $24 \text{ Tg yr}^{-1}$ ) is similar to the one based on the old methodology.

The sensitivity of our calculations to variation in parameter values was investigated for cattle, i.e. nondairy and dairy cattle. Total N excretion by cattle is about  $64 \text{ Tg yr}^{-1}$  which is about 57% of total N excretion by all animal categories in our analysis. Hence, variation of parameters related to cattle are expected to have an important influence of calculated  $\text{NH}_3$  emissions. We selected the parameters N excretion per head, total N excretion in meadows relative to animal houses, and the  $\text{NH}_3$  emission factors for animal housing (Table 2).

The results of this analysis indicate that N excretion per head has the largest influence ( $\pm 15\%$ ) on global  $\text{NH}_3$  emissions from livestock (Table 2). Varying total manure N generation in meadows relative to animal houses has less influence on global  $\text{NH}_3$  emissions from livestock, because the importance of confinement of cattle in animal houses is much less important than for other animals such as pigs and poultry. Variation of the  $\text{NH}_3$  emission factor for animal housing requires some explanation. With a higher  $\text{NH}_3$  emission factor, less N is retained in the manure stored in animal houses and other storage facilities, and less is available for spreading. Since  $\text{NH}_3$  loss rates from animal houses and other storage systems in our inventory are higher than for spreading, the overall emission rates also increase. In contrast, with a lower  $\text{NH}_3$  emission rate more N is retained and more is available for application to cropland and grassland. The overall  $\text{NH}_3$  emission thus decreases when the emission factor for animal houses is reduced.

**Table 1. Global estimates of manure-N and NH<sub>3</sub> emission for animal housing and storage systems and grazing for mixed/industrial and pastoral systems, spreading of stored manure in cropland and grassland, and N fertilizer use for the year 2000.**

Animal manure management system/ fertilizer use	Manure N production/use		NH <sub>3</sub> -N emission	
	Gg yr <sup>-1</sup>	% of total manureN	Gg yr <sup>-1</sup>	Mean emission factor (%)
Mixed/industrial systems				
Storage	48677	43	9735	20
Grazing	33091	29	3474	10
Total	81767	73	13210	16
Pastoral systems				
Storage	581	1	116	20
Grazing	19327	17	2379	12
Total	19908	18	2495	13
Other uses	10757	10	2151	20
Spreading of stored manure <sup>1</sup>				
Cropland	33517	30	4835	14
Grassland	5889	5	1311	22
Total	39406	35	6145	16
Total animal manure	112432		24001	21
Total N fertilizer use	83024		10826	13

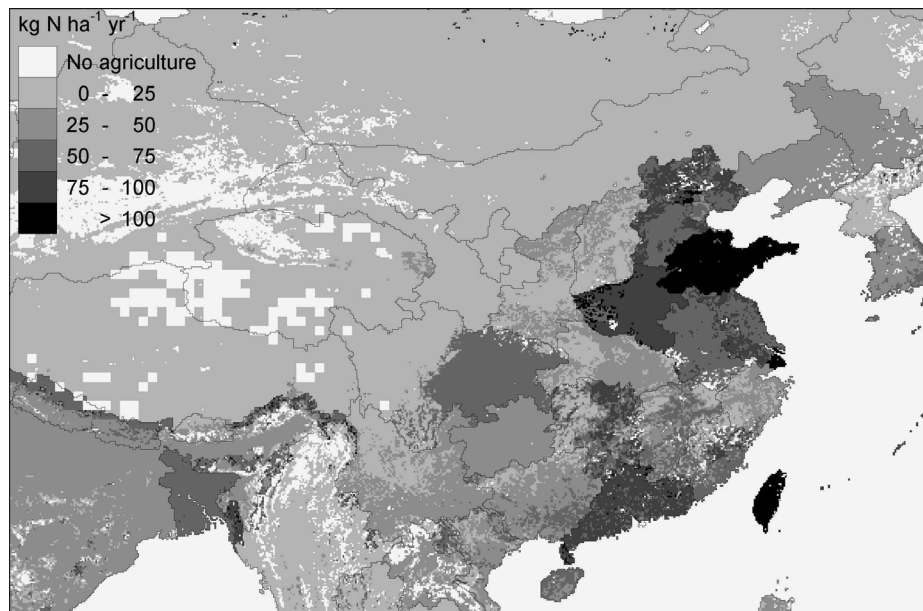
<sup>1</sup> Excluding 9852 Gg yr<sup>-1</sup> NH<sub>3</sub>-N loss from the total of 49258 Gg yr<sup>-1</sup> in animal housing and storage systems.

**Table 2. Effect<sup>1</sup> of variation of three important parameters for cattle on global NH<sub>3</sub> emission from livestock production.**

Parameter	Variation	
	-25%	+25%
N excretion per head	86	115
N excretion in meadow <sup>2</sup>	104	96
NH <sub>3</sub> emission rate for animal housing	95	111

<sup>1</sup> The effect is expressed as an index value, with standard case = 100.

<sup>2</sup> Variation of N excretion in meadows also influences the N excretion in animal houses. More grazing gives less manure production by confined animals, and less grazing causes the manure collected in animal houses to increase.



**Figure 2. Distribution of manure N in Eastern Asia.**

### Concluding Remarks

In this paper we present an update of the GEIA global inventory of  $\text{NH}_3$  emissions. Our inventory is largely based on country data (except for Western Europe, USA and China where more detailed information was included) on animal populations, areas of cropland and grassland, and information available for world regions on production systems (pastoral versus mixed/industrial), and the relative importance of grazing versus confinement of animals. New information can thus be used to modify any aspect of the inventory at the scale of countries or finer.

We recognize that there are several parts where further research and data collection is needed. We discuss four major issues. Firstly, the distribution of animal manure from ruminants over animal housing and other storage facilities and grazing may not be very important at the global scale, but locally it may have a very important influence on total emissions. In our inventory  $\text{NH}_3$  loss from grazing systems is 10-12%, while that from animal housing and storage systems and spreading is much higher. The emission factor from animal housing and storage systems is rather uncertain. There are many different types of animal housing and storage systems worldwide, and  $\text{NH}_3$  loss probably varies widely among them.

Secondly, in this inventory N excretion rates by animal category are average values for world regions, without a distinction between pastoral and mixed/industrial systems. N excretion rates may vary as a result of the productivity of the animal and the energy requirements. Meat and milk production per animal (higher in mixed/industrial systems than in pastoral ones) and the energy required for grazing and labor (lower in mixed/industrial systems) may result in N excretion rates that are not much different between the broad production systems. However, further research is needed to verify this assumption.

Thirdly, the distribution of animals over pastoral and mixed/industrial systems is derived from information for different levels ranging from world regions to countries or finer. Also the distribution of the grassland area in pastoral and mixed systems is based on the assumption that cropland covers at least 15% of the area in mixed agricultural systems. Although similar approaches have been used earlier for the developing countries, it is not clear if this method leads to realistic patterns in all parts of the world.

Fourthly, there is increasing concern about emissions of particulate matter (PM). An important research question is therefore if the data presented in this paper could be a starting point for an inventory of PM emissions.

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## Hydrogen Sulfide Emissions from Southern Great Plains Beef Feedlots: A Review

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### Abstract

Hydrogen Sulfide (H<sub>2</sub>S) is emitted from animal feeding operations as a product of anaerobic breakdown of organic materials. There has been limited research efforts towards quantifying the H<sub>2</sub>S emissions from open lot feedyards in the high plains of Texas, New Mexico, Oklahoma, Kansas and Colorado where more than 40% of U.S. beef cattle are fed and finished. As aerobic conditions are primarily observed in the feedyard manure packs, the ambient and property-line H<sub>2</sub>S concentrations recorded are not as high as those associated with those intensive animal feeding operations where anaerobic conditions are primarily employed in treatment and storage systems. Exposure to high levels of H<sub>2</sub>S can be fatal, while elevated levels can contribute to human health effects. Most states have regulations that set limits for ambient and/or property-line H<sub>2</sub>S concentrations to protect public health. The Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and the Emergency Planning & Community Right to Know Act (EPCRA) set reporting requirements for industries which exceed 100 lbs/day of long list of compounds including H<sub>2</sub>S.

There have been a number of studies looking at ambient H<sub>2</sub>S concentrations near open-lot beef cattle feedyards, Nebraska (Koelsch et al., 2004) and Texas (Rhoades et al., 2003; Koziel et al., 2004; See et al., 2003). Koelsch et al. (2004) monitored H<sub>2</sub>S as total reduced sulfur (TRS) concentrations at three open-lot beef cattle feedyards in Nebraska reporting mean H<sub>2</sub>S concentrations downwind of pens ranging from 0.006 to 0.013 ppm, with 19 of 2,067 total observations greater than 0.100 ppm. Mean concentrations downwind of ponds were 0.002 to 0.014 ppm, with 11 out of 1,888 total observations greater than 0.10 ppm and two greater than 10 ppm. Koelsch et al. (2004) concluded that “TRS levels in the vicinity of beef cattle feedlots are not likely to exceed current regulatory thresholds used by midwestern states”. Rhoades et al. (2003) measured H<sub>2</sub>S (TRS) concentrations upwind and immediately downwind of pens and ponds at three feedyards over a 12-month period in 2002-2003. The authors used a Jerome meter to measure short duration (i.e. over a two-minute interval) concentrations. Three to four readings were made at each location and averaged, thus each mean reading would be representative of about a 10 minute time span. The H<sub>2</sub>S readings were taken during the day, usually between 9 AM and 3 PM. Koelsch et al. (2004) showed a diurnal pattern in H<sub>2</sub>S emissions with higher emissions in the later afternoon when the temperatures were warmer. See (2003) also reported higher H<sub>2</sub>S concentrations between 2 PM and 4 PM. Because all of Rhoades et al. (2003) data was taken in the daytime, it is unknown if this data is representative of true 24-hr emissions. The measurements of TRS made by Rhoades et al. (2003) showed average concentrations of 0.026 ppm at the feedyard pen fence and 0.037 ppm immediately downwind of feedyard retention ponds. See (2003) reported H<sub>2</sub>S concentrations downwind of pens and pond for data collected during summer from a beef cattle feedyard in the Texas Panhandle. Hydrogen sulfide concentrations were measured using a Jerome meter and datalogger every 15 minutes for 44 hours downwind of pens and 22 hours downwind of the pond. See reported mean downwind H<sub>2</sub>S concentrations of 0.005 and 0.005 ppm for the pens and ponds, respectively. Koziel et al. (2004) measured ambient H<sub>2</sub>S concentrations at an open-lot beef cattle feedyard over three seasons (fall, winter, and spring) using a TEI 45C pulsed fluorescence analyzer housed in a instrument trailer. The trailer was located on the western side of the feedyard, immediately adjacent to the pens. Because the trailer was stationary, the wind direction variable, and its location upwind of the feedlot for one of the dominant wind directions, the instrument was not always recording downwind concentrations, the mean values presented in their research are likely skewed on the low side and not representative of true downwind mean concentrations. Mean H<sub>2</sub>S concentrations for fall, winter, and spring seasons were 0.008, 0.001, and 0.002 ppm, while maximum H<sub>2</sub>S concentrations were 0.030, 0.003, and



0.035 ppm, respectively. Koziel et al. (2004) concluded that “measured H<sub>2</sub>S concentrations were always lower than the ambient air ground level concentration maximums for the State of Texas.”

There is very limited data available on H<sub>2</sub>S emissions from open-lot beef cattle feedyard pens with the only reported emission rates from open-lot beef cattle feedyards being collected as part of the Federal Air Quality Initiative project. Wood et al. (2001) reported a mean TRS emission rate of 103 µg/m<sup>2</sup>/min from naturally ventilated, loose housed, beef steer housing facilities in Minnesota. Duyson et al. (2003) attempted to measure H<sub>2</sub>S emissions using a wind tunnel and Jerome meter, however the concentrations were too low to quantify. Baek et al. (2003a,b) and Koziel et al. (2005) measured H<sub>2</sub>S emission rates using a flux chamber (NC State design) and TEI 45C pulsed fluorescence analyzer. The data of Baek et al. (2003a,b) and Koziel et al. (2005) are summarized in Table 1. Based on these H<sub>2</sub>S emission rate estimates, this equates to an extrapolated emission rate of 0.065-0.088 lb/d (0.029-0.040 kg/d) per 1,000 head using a stocking rate of 14.7 m<sup>2</sup>/head. For a typical 50,000 head feedyard, this equates to an emission rate of 3.2-4.4 lb/day from the pens only.

**Table 1. Hydrogen sulfide emission rates from two beef cattle feedyards measured using a flux chamber**

Reference	CAFO Type	Method	Emission Rate (µg/m <sup>2</sup> /min)
Baek et al. 2003a,b	Beef Open-Lot	Flux Chamber	1.88
Koziel et al. 2005	Beef Open-Lot	Flux Chamber	1.39

No published estimates of H<sub>2</sub>S emission rate from the runoff retention structures at open lot beef feedyards have been found in the published literature. Conditions in these runoff retention structures are usually slightly acidic and anaerobic with accumulating, decomposing organic matter. These conditions are conducive to H<sub>2</sub>S generation and emission. While the area of the runoff retention structures is less than the pen area in a typical feedyard, it could be safely assumed that the emission rate is greater on a per area basis and these emissions may dominate the overall emissions from the facility.

### Conclusions

Based on the measurements of downwind H<sub>2</sub>S concentrations available from the published literature, it appears that there is a low probability that the average H<sub>2</sub>S concentration downwind of a feedyard will exceed the ambient downwind H<sub>2</sub>S regulatory values for Texas of 80 ppb (30 minute average), however that it is possible during critical atmospheric conditions. Assessment of the potential of a feedyard to exceed the CERCLA/EPCRA reporting requirement of 100 lbs/day is not currently possible given the lack of any published data regarding emissions from the potentially significant runoff retention structures. Measurements of the emission rate from these runoff retention structures is urgently needed to complete this assessment.

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## Emission of Nitrous Oxide from NE Dairy Farms and Agricultural Fields: Laboratory and Field Studies

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### Abstract

Agriculture significantly contributes to overall anthropogenic N<sub>2</sub>O emissions. The input from stored animal waste can be as great as 3% of total agricultural emission. Dairy farms are ubiquitous in New York State and their contribution to nitrous oxide emissions is unknown. The goal of this study is to quantify N<sub>2</sub>O losses from large dairy farms and agricultural fields as influenced by interactions between tillage practices, soil conditions and manure application. We summarize here both laboratory and a portion of field experiments. At the conference we will report N<sub>2</sub>O emissions measured using Tunable Diode Laser Trace Gas Analyzer (TDL TGA). We will discuss nitrous oxide emission rates from the various agricultural fields and recommend improvements in manure management schemes to reduce greenhouse losses.

### Introduction

Global agricultural N inputs to the atmosphere now exceed those from natural sources. One of the main sources of gaseous N losses is spreading of animal waste on agricultural fields, amounting to 35% of the global annual emission (Kroeze et al., 1999). The Kyoto protocol commits signatories to reducing greenhouse gas emissions to 1990 levels.

A number of experimental studies have measured nitrous oxide emissions from agricultural fields (e.g. Gregorich et al., 2005). The results were interesting but also indicate large uncertainties and variability in both time and space arising from soil heterogeneity and complex interactions between chemical, physical and biological variables (e.g Muller et al., 1997). More research is needed to obtain a quantitative understanding of how farm management practices can reduce N<sub>2</sub>O emissions. Of special interest is learning and solving some of the apparent complexities involving the interactions between tillage practices and soil conditions and how they affect N<sub>2</sub>O emissions shortly after manure application when potential losses are the greatest. A good example of the complexities involved in designing manure management strategies is the practice of manure injection into soil to reduce odor emissions following spreading of both liquid slurries and farmyard manures (Webb et al., 2004). Although this is known to reduce ammonia emissions, there are concerns that these direct applications into soil may significantly increase N<sub>2</sub>O emissions by increasing the pool of mineral N in soil (Bouwman, 1996).

To improve our understanding on the interactions between tillage effects, moisture content, manure application and chemical, physical and biological variables we are carrying out laboratory and field experiments. We will summarize here on the laboratory and some of the field experiments. At the conference itself we will report on current determinations of nitrous oxide emissions measured using Tunable Diode Laser Trace Gas Analyzer (TDLTGA), which have been delayed due to technical breakdowns.

### Experimental

#### Laboratory Experiments

**Materials:** Two sand textures – coarse and fine – were used consisting of washed organic-free sand with particle diameters of 1 and 0.25 mm, respectively. Total porosity was ~36% in both cases. The synthetic urine mixture contained urea-N concentrations to simulate levels of dietary N intake (11.5 g/L), as well as glycine (2.9 g/L), KHCO<sub>3</sub> (13.8 g/L), KCl (2.5 g/L), KBr (4.2 g/L), and K<sub>2</sub>SO<sub>4</sub> (1.4 g/L), with 0.5% of cow urine added. The experimental chambers were sealed plastic desiccators (14-cm diameter, Bel-Art No.F42010) packed with 1 cm or 5 cm sand depths. All experiments were repeated in triplicates and carried at constant temperature of 25°C.

Incubation procedure:

(1) Static headspace experiments: synthetic urine was added to each sand to bring the sand moisture content to 10, 15, 30, 50, 60, 70, 85, and 90%. The wetted sand samples were sealed in the chambers and incubated in aerobic conditions at the constant temperature. Samples of headspace air (6 mL) were taken twice a day. After each sampling, the sample units were opened to room atmosphere and then sealed again. Every second day the chambers were weighed to monitor the moisture loss by evaporation. The total duration of each treatment was 30 days.

(2) Flow-through experiments. Nitrous oxide-forming processes were linked to the changes of urine distribution with depth caused by evaporation. Both sand textures were mixed with urine to reach 80% water-filled pore space. The experiments were carried out with constant air flow rates using an air pump. Air was injected at the flow rate of 1250 mL/min during the first 24 hours and thereafter reduced to 125 mL/min; air samples were taken from tubing near input and output ports with syringes every 4 hours during the first two days and then every 8 hours during 14 days. The air samples were analyzed for N<sub>2</sub>O concentrations.

**Field Experiments**

Field site: The research site is located on a large dairy farm central New York (42°49'N, 76°47'W), 15 miles from city of Ithaca. The site is a cornfield, which receives dairy manure fertilization once a year, in spring or fall. Manure is injected into the field with a tractor and a draghose (Wright and Bossard, 2003). On 10 October 2004 all the equipment for flux measurements and soil sampling was installed in the field (after corn harvesting and about 6 months after the last fertilization). The experiment was continued till December 5, 2005. On November 7 one half of the field was tilled.

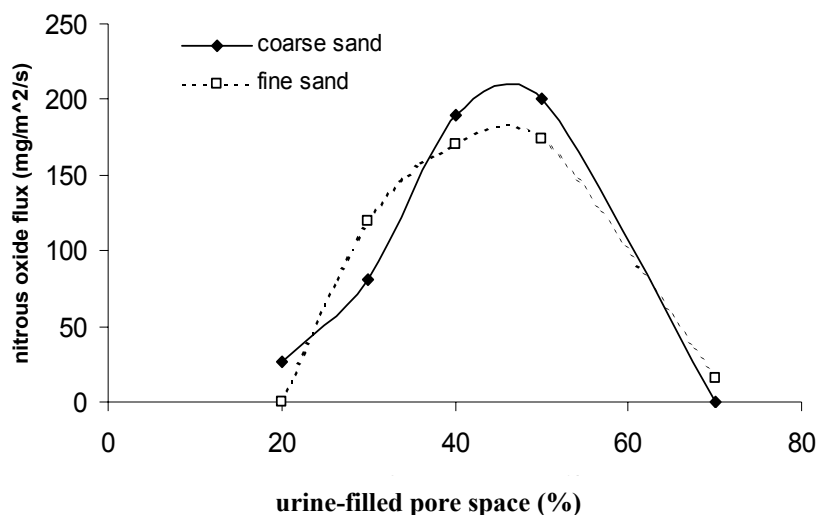
Measurements of nitrous oxide flux using chambers: Accumulation of nitrous oxide was determined using polyvinylchloride chambers covering a rectangular area of 0.89 m<sup>2</sup>. Each chamber consisted of a frame permanently located in the ground which could be sealed for 3 hours with removable plastic lid and additional plastic film isolation to prevent air exfiltration from the chambers. Each chamber lid was equipped with an aperture tightly closed by rubber septa to allow gas sampling with a syringe. Gas samples were collected at the start and termination of sealing in evacuated minivials. The nitrous oxide concentrations were measured by gas chromatography (GC) with an electron capture detector. The instrument used was Varian 3700 GC with a manual injection system and Ni<sup>63</sup> ECD operated at 350°C. The carrier gas was Ar:CH<sub>4</sub> (95:5) at a flow rate of 30mL/min.

Soil sampling and analysis: Soil samples were collected from the depth of 15-20 cm near each chamber when the air samples were collected. Samples were transported for the laboratory where were analyzed for NO<sub>3</sub><sup>-</sup>, NH<sub>4</sub><sup>+</sup>, pH, and gravimetric moisture content (Franson,1985).

**Results and Discussions: a Summary of our Findings**

**The Effect of the Soil Moisture-filled Pore Space (MFPS) and Temperature on N<sub>2</sub>O Emission**

Laboratory experiments with urine. Figure 1 demonstrates the mean N<sub>2</sub>O flux during the laboratory static headspace experiment from sand surface as a function of urine-filled pore space. For both sand types N<sub>2</sub>O emission was observed between 20 and 70 % of urine-filled pore space and reached maxima at ~50%. The general shape of the curves corresponds to the shape of relationship between N<sub>2</sub>O emission and moisture-filled porosity presented by Davidson et al. (2000) for different soil types.



**Figure 1. A generalized relationship between urine-filled pore space and N<sub>2</sub>O flux in two sand types (controlled conditions)**

Figure 1 shows the relationship between urine-filled pore space and nitrous oxide flux in two sand types (controlled conditions). The graph shows that the most gaseous emission of N<sub>2</sub>O was found within the ~30-60% range of urine-filled pore space in both sand types, which corresponds very well to Figure 1. The thickness of the horizon containing ~30-60% urine-filled pore space was greater in the fine sand, therefore the production of N<sub>2</sub>O was more intensive in this sand type. In fine sand (Figure 2a), the thickness of this zone with this porosity range increased from 1 cm to 3 cm from during days 4 to 8 of the experiment, and then decreased to 1 cm after 16 days. In contrast, in coarse sand (Figure 2b), the width of this horizon was a constant ~1 cm throughout the experiment. In both sands the N<sub>2</sub>O production horizons moved deeper with time because of the sand drying process. A more detailed description can be found in Singurindy et al. (2006).

**Field data.** Nitrous oxide flux measured in the cornfield soils is presented in Figure 3 as a function of soil water-filled pore space and soil temperature. The nitrous oxide emissions are represented by the colored spectrum. The maximal flux of nitrous oxide was found when at the water-filled pore space ~ 55%, which corresponds well to the results obtained in the laboratory as presented in Figure 1. Generally, the optimal conditions for nitrous oxide production were found for the temperatures greater than 5°C and at water-filled porosities between 40 and 70 %. Temperatures lower than 5°C reduced microbial activity in the soil that reduced emissions.

(a)

(b)

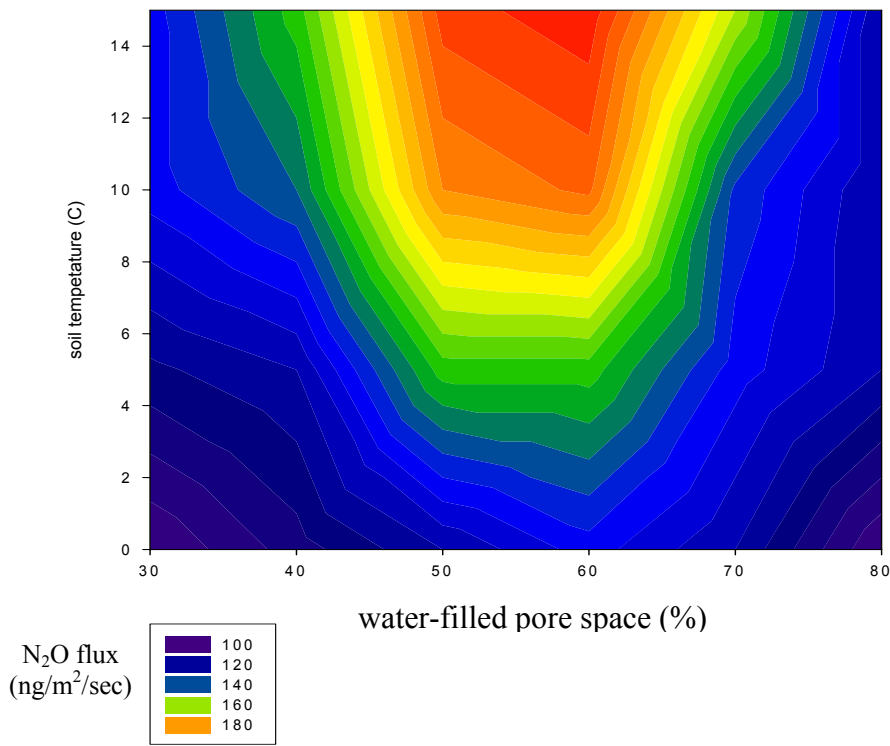
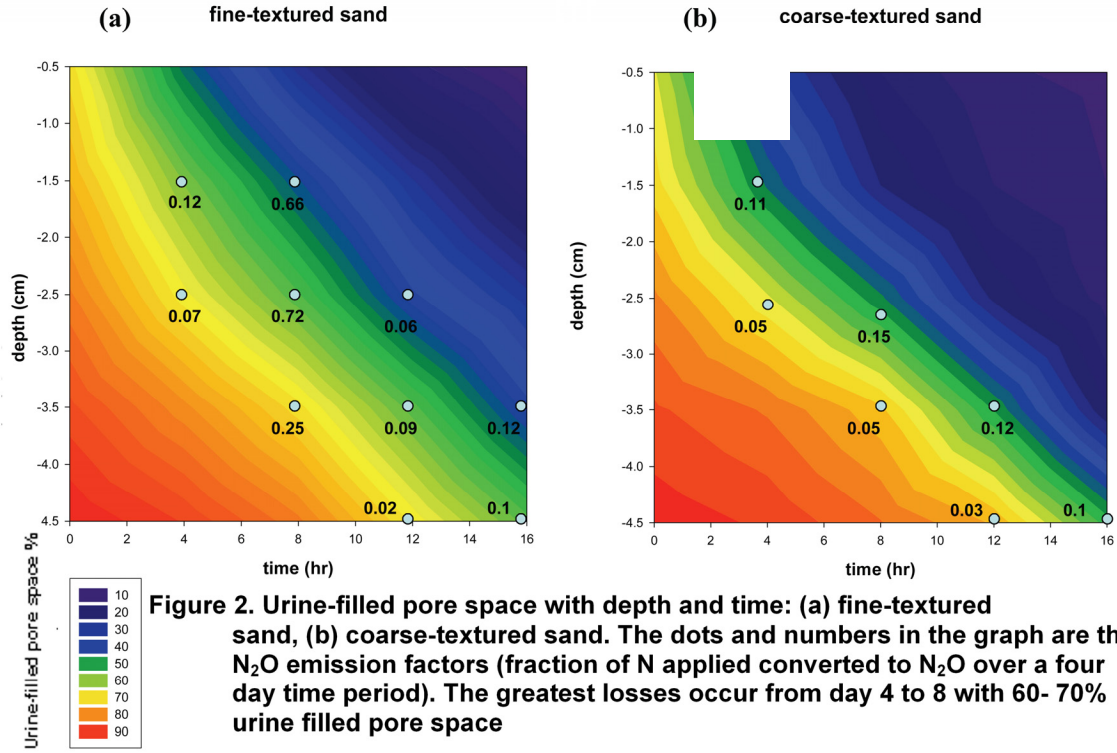
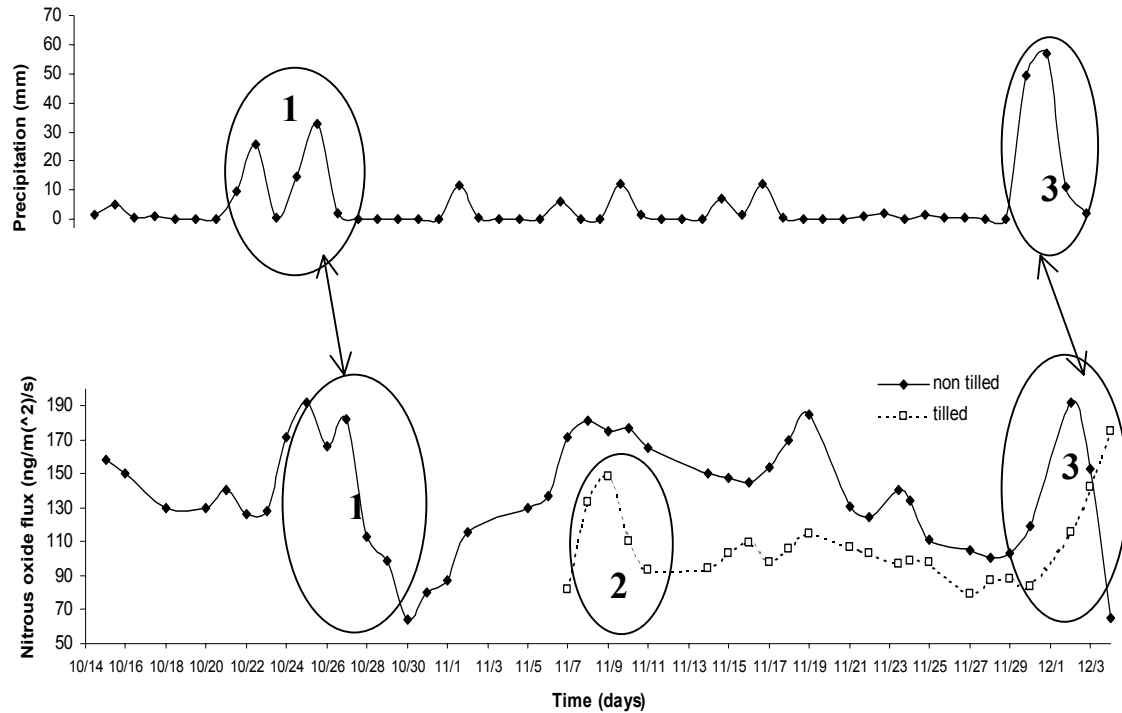


Figure 4 presents the nitrous oxide flux from agricultural field. In non-tilled soils, the significant increase in the flux began one day after intensive precipitation (events 1 and 3), then the flux decreased by 2.5 times after ~70 mm of precipitation in both events. In contrast, the intensive precipitation caused increases in the flux in tilled soils during the whole precipitation event (event 3). Tillage resulted in immediate decrease for ~50% of nitrous oxide emission. These results suggested that denitrification was the dominant process in soils before tillage. Plowing improved aeration conditions in soil and reduced the soil WFPS which was reflected in short-term increase of nitrous oxide (event 2) two days after plowing. This peak corresponds to the increase on nitrate concentration in tilled soils (data are not presented).



**Figure 4. Fluxes of nitrous oxide measured from tilled and non-tilled parts of the agricultural field.**

### Conclusions

The presented results demonstrate that tillage has a significant effect on fluxes of nitrous oxide. Tillage increased nitrate concentration for short-term that was not observed in non-tilled soils. The MFPS determined during each nitrous oxide sampling related to nitrous oxide fluxes indicated that the differences in the fluxes were mainly due to moisture content and temperature fluctuations in the field that controlled the intensity of microbial activity. Generally, tillage reduced nitrous oxide emission before manure application.

Since short-term pulses nitrous oxide occurring after tillage and wetting events are a significant component of annual nitrous oxide flux in the NE dairy farming and vary spatially and temporally it is necessary to determine nitrous oxide fluxes in agroecosystem using accurate measurements using Tunable Diode Laser Trace Gas Analyzer (TDLTGA). Further research is under way to examine the influence of other soil quality indicators and depths of manure injection on nitrous oxide emission from agricultural soils.

### Acknowledgments

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## Quantification of Gas, Odor and Dust Emissions from Swine Wean-Finish Facilities

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### Abstract

A total of 1,920 pigs (equal barrows and gilts) are being used in a 2 x 2 factorial, wean to finish experiment to determine the effects of diet (control, CTL vs. low nutrient excretion, LNE) and manure pit management strategy (deep pit, DP vs. monthly pull plug, PP) on excretion of nutrients and gaseous and particulate emissions. Pigs are being housed in a 12 room environmental building, which allows for real-time monitoring of air quality, and quantitative manure collection from 24 pits (2/room). Each room contains 30 barrows (3 pens) and 30 gilts (3 pens), which are being split-sex and phase fed to meet or exceed the nutrient requirements of pigs (NRC, 1998) at different stages of growth. Dietary treatments (CTL and LNE) are being maintained throughout the trial. Individual pig weights and pen feed consumption data are collected every two weeks. Four pigs from each pen are being scanned ultrasonically for determination of loin eye area and backfat thickness at two months of age and every four weeks thereafter during the study. At the end of the experiment, carcass data is being collected at harvest on all pigs. Air temperature, relative humidity, total suspended particulates, ammonia, hydrogen sulfide, carbon dioxide, and methane concentrations are being recorded every fourth week during the experiment. In addition, odor samples are being collected at months 1, 3 and 5 of each wean-finish replicate in this experiment. A dynamic dilution venturi olfactometer is being used, with trained panelists, to evaluate each bag sample of air for olfactometry. Odor and gaseous emission rates are being calculated by multiplying air flow rate by the difference between inside and outside concentrations. Preliminary data indicates that pigs fed the LNE diet grow faster than control fed pigs while consuming less feed, resulting in an improved feed efficiency throughout most phases of the trial.

### Introduction

In the past two decades, the pork industry has undergone rapid technological and structural change. The most significant changes have been a decrease in farm numbers, an increase in production facility size, and the movement of large production operations to more rural areas of the country. The number of farms raising hogs declined by 83% from 1965 to 1995 (USDA Report, 1996). Additionally, from 1997 to 2002, the number of farms with swine decreased by 45.7% or an average of over 9% per year (USDA report, 2002). Even though the swine industry has seen record losses of farms, relatively little change in the annual number of pigs raised in the US has occurred between 1965 and 2002. Unfortunately, animal feeding operations can affect air quality through emissions of odor, odorous gases (odorants), particulates (including biologic particulate matter), volatile organic compounds, and some greenhouse gases (Arogo et al., 2001; Bicudo, et al., 2001; Sweeten, et al., 2001; USDA AAQTF, 2001; and NAS, 2003). Much of the emitted gases come from the anaerobic decomposition of manure during storage, the release of volatile organic compounds and ammonia immediately after excretion from the animal and dust generated in the building facilities from feed delivery systems, animal movement, and hair and sloughed skin from the animal. New regulatory pressures to meet water and air quality standards for CAFO's (EPA, 2003) and NPDES permit regulations, including the possibility of meeting total maximum daily load (TMDL) of contaminants in the water supply and stricter air quality regulations are placing additional economic and management burdens on pork producers which may lead to further consolidation of the industry.

Much of the public awareness of the potential threat of swine manure to water pollution has been due to a few large operation's having spills. Media attention and activity groups have applied pressure on producers, legislators and regulators for management changes in livestock operations. In many cases, odors, dust and gas emissions from swine units have resulted in nuisance lawsuits and unrealistic regulations not necessarily based on scientific evidence. Residents near operations are concerned about the potential

devaluation of their property and the impact of manure and odors on their health and lifestyle. State and local governments are struggling to develop long term land use plans to maintain sufficient land areas for both pork operations with land application of manure and the influx of urban residents into rural areas. Therefore the objectives of this trial are to determine the amount of gases, odors and dust emitted from buildings when swine are fed different diets and two manure storage strategies are utilized.

### Materials and Methods

#### Animal Design

To date, 960 (avg initial BW = 5.16 kg; avg final BW = 128.16 kg) wean-finish pigs have been utilized. Pigs were housed in an environmentally controlled building with identical and independent ventilation, feeding systems, water, and manure storage pits. Each room housed 10 pigs per pen with 60 pigs per room. Pigs were blocked by BW and sex (10 pigs/pen; 60 pigs/room) and randomly allotted to 1 of 4 treatments arranged in a 2 X 2 factorial design with 2 diet formulations (standard commercial corn-SBM control, CTL; or a low nutrient excretion diet, LNE) and 2 manure storage strategies (6 month deep pit collection, DP; or a monthly pull plug/recharge collection, PP). Pigs were split-sex and phase fed to meet or exceed nutrient requirements (NRC, 1998). This trial consisted of five nursery phases and four grow-finish phases. The nursery phases included: 1) Pellets, d 0-7; 2) Phase 1, d 7-14; 3) Phase 2, d 14-28; 4) Phase 3, d 28-42; and 5) Phase 4, d 42-56. The grow-finish phases included: 1) Grower 1, d 56-84; 2) Grower 2, d 84-112; 3) Finisher 1, d 112-140; and 4) Finisher 2, d 140-152. Individual pig weights and pen feed consumption data were collected once a week during the nursery pellet phase and phase 1, then every two weeks thereafter. Four pigs from each pen were scanned ultrasonically for determination of loin eye area and 10<sup>th</sup> rib backfat thickness starting at two months of age and every four weeks thereafter during the study. At the end of the experiment, carcass data were collected at harvest on all pigs by a commercial slaughter facility.

#### Dietary Treatments

Pigs were fed either a commercial corn-soybean meal control (CTL) diet or a low nutrient excretion (LNE) diet (Table 1). The LNE diets had a reduced crude protein level compared to the CTL diets, and included synthetic amino acids, phytase (Natuphos, BASF, New Jersey, USA), added fat, and a non-sulfur trace mineral premix. Diets were formulated based on NRC (1998) requirements for available phosphorus and true ileal digestible amino acids, while also maintaining similar lysine:calorie ratios.

#### Air Concentration Monitoring

Continuous real-time instruments monitored ammonia (NH<sub>3</sub>), hydrogen sulfide (H<sub>2</sub>S), carbon dioxide (CO<sub>2</sub>), and methane (CH<sub>4</sub>) every fourth week during the trial. Real-time monitoring was conducted for 5 days during the fourth week before the release of manure from the pit in the PP system, during pit sampling and for 2 days after pit emptying to determine any effect of emptying a manure pit on gas emissions.

Thirty nine odor samples were collected at months 1, 3, and 5 of each replicate with three samples obtained from each room exhaust and three from the fresh air plenum that is common to all rooms. The three odor samples obtained from each room were collected at each measurement location simultaneously. Air samples were collected into 10 L Tedlar bags. A dynamic dilution venturi olfactometer (AC'SCENT Internations, St. Croix Sensory, Inc., St. Paul, MN) was used to evaluate each bag sample of air for olfactometry (data not presented). All evaluations were performed by trained human panelists. Sample evaluation occurred the same day as sampling to minimize bag losses.

#### Statistical Design

All data were analyzed using the GLM procedure of SAS (2006; SAS Institute Inc., Cary, NC). Pen was the experimental unit for animal performance and carcass characteristics; data represents 48 of 72 planned observations for diet and sex. Manure pit was the experimental unit for manure storage strategy and data presented represents 16 of 24 planned observations. Finally, room was the experimental unit for aerial gaseous compound concentrations; data represents 8 of 12 planned observations for diet and storage and 16 of 24 planned observations for wk of production. Animal performance and carcass characteristics were analyzed for the main effects of dietary treatment, manure storage type, and sex. Gaseous and particulate

emissions were analyzed for the main effects of dietary treatment, manure storage type, and week of production.

## Results

### Nursery Performance

During nursery phase 1 (Table 2), CTL fed pigs had a 7.8% increase (2.96 vs. 2.73 kg/d;  $P<0.02$ ) in average daily feed intake (ADFI) compared to LNE fed pigs, however diet did not affect average daily gain (ADG) or feed efficiency (Gain:Feed). During this phase gilts tended to have increased feed efficiency compared to barrows (0.72 vs. 0.67, respectively,  $P<0.08$ ). In phase 2 of the nursery period there were no differences in ADG or ADFI, however numerical differences in these parameters resulted in an increased feed efficiency for LNE fed pigs compared to CTL fed pigs (0.77 vs. 0.73, respectively;  $P<0.05$ ). Gilts tended to have increased ADFI compared to barrows ( $P<0.07$ ). During phase 3, LNE fed pigs had a 6.3% increase ( $P<0.001$ ) in feed efficiency and tended to have an increased ADG ( $P<0.08$ ) compared to CTL fed pigs. During phase 4 (d 42 to 56) of production, growth performance of LNE fed pigs was superior to CTL fed pigs. The LNE diet increased ADG by 4.5% ( $P<0.003$ ) and decreased ADFI by 8.6% ( $P<0.002$ ) which resulted in a 12.9% increase ( $P<0.001$ ) in feed efficiency. Although growth performance was improved with the LNE diet throughout the nursery phase, there was only a tendency for LNE fed pigs to be heavier on d 56 than CTL fed pigs (35.53 vs. 34.58 kg;  $P<0.06$ ). However since this data remains preliminary, we expect as our sample size increases there will be a separation in BW between the two dietary treatments resulting in LNE fed pigs being heavier at the end of the nursery period.

### Grow-finish Performance

Manure storage type did not have any effect on grower 1 growth performance (Table 3). However, ADFI was increased by 7.6% ( $P<0.003$ ) and feed efficiency was decreased by 8.0% ( $P<0.003$ ) for CTL fed pigs compared to LNE fed pigs. During grower 1, barrows had greater ADG than gilts ( $P<0.006$ ), but d 84 BW was not different between sexes (61.38 vs. 60.39;  $P=0.15$ ). Pigs fed the LNE diet during grower 2 had a 10.2% increase in ADG, a 14.6% increase in G:F, and a 4.5% decrease in ADFI ( $P<0.02$ ) compared to CTL fed pigs. LNE fed pigs also weighed approximately 3.4 kg more than CTL pigs on d 112 (91.36 vs. 88.01 kg;  $P<0.002$ ). These improvements in growth performance continued throughout the two finisher 2 phases. LNE fed pigs were 4.3 and 5.0 kg heavier at d 140 and at market (d 152) compared to CTL fed pigs ( $P<0.001$ ). Additionally, barrows were heavier ( $P<0.04$ ) following the end of grower 2, finisher 1, and finisher 2 compared to gilts. Manure storage type only had a significant affect on grower 2 ADFI, where pigs reared under the DP system consumed 4.3% more feed per day than pigs reared under the PP system.

### Carcass Characteristics

Live ultrasonic measures of loin eye area and backfat thickness were unaffected by sex during the grower 1 and 2 phases (Table 4). However, LNE fed pigs had greater backfat thickness compared to CTL fed pigs during grower 1 and 2 (8.85 vs. 8.01 mm and 11.07 vs. 9.50 mm, respectively;  $P<0.004$ ). Pigs fed LNE diets also showed a tendency for a greater loin eye area for grower 1 (14.52 vs. 13.82 cm<sup>2</sup>;  $P<0.08$ ). Dietary effects on backfat thickness were observed throughout the experiment. Backfat depths for LNE fed pigs were 9.5, 14.2, 10.4, 12.3, and 13.8% higher than CTL fed pigs for grower 1, grower 2, finisher 1, finisher 2, and market, respectively ( $P<0.004$ ). These results are not surprising since LNE fed pigs were heavier than CTL fed pigs at the end of grower 2, finisher 1, and finisher 2 (Table 3). Backfat thickness was also 13.1, 20.9, and 19.5% greater in barrows for finisher 1, finisher 2, and market, respectively, compared to the gilts. Again this result can be explained by the fact that barrows were heavier than the gilts during the grower 2 and finisher phases.

Similar to live ultrasonic measures, carcass backfat thickness, lean percent, carcass grade premium, and hot carcass wt were all affected by dietary treatment (Table 5). Carcass backfat thickness was 8.6% greater ( $P<0.000$ ) in LNE fed pigs, compared to CTL fed pigs. Pigs fed LNE diets also had a decreased lean percentage compared to CTL fed pigs (53.24 vs. 53.80%;  $P<0.001$ ). Although the live ultrasonic measurements indicate a larger loin eye area for LNE fed pigs, carcass loin depth measurements were not different among dietary treatments. Decreased lean percentages and increased backfat resulted in a reduced carcass grade premium for LNE fed pigs compared to CTL fed pigs (\$0.10 vs. \$0.11 per kg;  $P<0.003$ ).

However, hot carcass wt was increased by 3.7 kg (96.72 vs. 93.13 kg;  $P < 0.003$ ) for LNE fed pigs compared to CTL pigs. Therefore, overall carcass value was not different for LNE and CTL fed pigs.

Barrows had 16.3% more backfat ( $P < 0.001$ ) and a heavier hot carcass wt ( $P < 0.002$ ) compared to gilts, which were leaner than barrows (54.06 vs. 52.98%;  $P < 0.001$ ). Carcass grade premium, carcass value, and live value for gilts were increased ( $P < 0.05$ ) by 25, 2.9, and 2.9%, respectively compared to barrows.

### Gas Concentrations

Data presented in this section is preliminary and consists of gaseous concentrations only. Air flow data is being calculated to determine gaseous emission rates. Gas concentration data collected from May 2005 to January 2006 reveals that diet, manure storage type, and week of production have significant effects on the concentration of various gaseous compounds (Table 6 and 7). Based on preliminary data, LNE diets reduced aerial  $\text{NH}_3$  concentration over the wean-finish period by 13.6% ( $P < 0.001$ ) compared to CTL diets. The PP system significantly reduced aerial  $\text{NH}_3$  concentrations by 7.3% ( $P < 0.005$ ) compared to the DP system. The PP system also reduced aerial  $\text{CH}_4$  concentration by 17.7% (9.44 vs. 11.47 ppb;  $P < 0.001$ ) compared to the DP system. Aerial  $\text{H}_2\text{S}$  and  $\text{SO}_2$  concentration were not different ( $P > 0.10$ ) among dietary treatments even though LNE diets were formulated with a non-sulfur trace mineral premix. Additionally there was no effect of manure management system on  $\text{H}_2\text{S}$  concentration (139.38 vs. 138.76, respectively;  $P > 0.10$ ). However, the PP system tended to increase aerial  $\text{SO}_2$  concentration compared to the DP system (12.71 vs. 9.90 ppb, respectively;  $P < 0.07$ ).

Air concentration data was also affected by wk of production, except for aerial  $\text{SO}_2$  concentration (Table 7). Aerial  $\text{NH}_3$ ,  $\text{H}_2\text{S}$ , and  $\text{CH}_4$  concentrations were increased by 43.4, 68.3, and 29.0%, respectively, from wk 4 to wk 16 ( $P < 0.001$ ). Conversely, the concentration of  $\text{CO}_2$  was reduced by 13.6% during wk 20 compared to wk 4 ( $P < 0.001$ ).

### Implications

The most significant change seen in the swine industry has occurred over the last sixty years. We have seen a shift from many farms producing a limited number of pigs to a small number of large confinement production facilities. New regulatory pressures to meet water and air quality standards for CAFO's and NPDES permit regulations are placing additional economic and management burdens on pork producers, which may lead to further consolidation of the swine industry. Preliminary data presented in this proceedings paper illustrates that feeding low nutrient excretion diets does not have to result in poor animal performance or carcass characteristics to yield reductions in gaseous compounds. Pigs fed the low nutrient excretion diets had improvements in average daily gain, feed efficiency, and were approximately 5.0 kg heavier at market than pigs fed control diets. Although backfat thickness was greater for low nutrient excretion fed pigs, there was no difference in percent carcass yield or total carcass value. More data needs to be analyzed to determine accurate air emission data from this trial, however by reducing emissions from swine facilities; there can be less neighborly concern and more acceptance of the swine industry. Moreover, this data will serve as a modeling tool for producers, extension educators, regulators, consultants, and legislators to plan environmentally sound pork production systems throughout the United States.

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**Table 1. Dietary Treatments for Finisher 1.**

<i>Ingredients, %</i>	Control		LNE	
	Barrows	Gilts	Barrows	Gilts
Corn	81.05	79.27	81.66	79.68
Soybean meal	17.00	18.79	12.03	14.01
Choice white grease	-----	-----	4.00	4.00
Calcium carbonate	0.66	0.65	0.90	0.90
Dicalcium phosphate	0.70	0.69	0.34	0.33
Vitamin premix	0.10	0.10	0.10	0.10
TM premix	0.05	0.05	-----	-----
Non-sulfur TM premix	-----	-----	0.05	0.05
Phytase	-----	-----	0.083	0.083
Salt	0.25	0.25	0.25	0.25
Lysine-HCl	0.10	0.10	0.32	0.32
DL-methionine	-----	-----	0.05	0.06
L-threonine	0.01	0.02	0.12	0.12
L-tryptophan	-----	-----	0.02	0.02
Tylan 40	0.025	0.025	0.025	0.025
Se 600	0.05	0.05	0.05	0.05
<i>Calculated Analysis</i>				
ME, kcal/kg	3347	3346	3517	3517
Lysine:calorie ratio	2.101	2.235	2.101	2.235
Calcium, %	0.50	0.50	0.50	0.50
Avail. Phosphorus. %	0.19	0.19	0.19	0.19

**Table 2. Effects of diet, manure storage type, and sex on nursery pig performance (preliminary data)<sup>a</sup>.**

Main Effects	Diet		Storage		Sex		MSE	P Values		
	Control	LNE	Deep Pit	Pull Plug	Barrows	Gilts		Diet	Storage	Sex
Initial wt, kg	5.15	5.17	5.15	5.16	5.22	5.10	0.800	0.94	0.95	0.46
d 7 wt, kg	6.11	6.12	6.11	6.11	6.16	6.06	0.758	0.98	0.99	0.50
<i>Phase 1 (d 7 to d 14)</i>										
ADG, kg/d	0.21	0.19	0.19	0.21	0.19	0.21	0.052	0.25	0.03	0.11
ADFI, kg/d	2.96	2.73	2.83	2.86	2.82	2.87	0.483	0.02	0.78	0.60
Gain:Feed	0.69	0.70	0.65	0.74	0.67	0.72	0.135	0.53	0.002	0.08
d 14 wt, kg	7.54	7.45	7.41	7.59	7.49	7.51	0.923	0.63	0.33	0.89
<i>Phase 2 (d 14 to d 28)</i>										
ADG, kg/d	0.45	0.46	0.46	0.46	0.45	0.47	0.048	0.26	0.52	0.12
ADFI, kg/d	6.25	6.12	6.18	6.19	6.07	6.30	0.623	0.33	0.90	0.07
Gain:Feed	0.73	0.77	0.74	0.76	0.76	0.74	0.110	0.05	0.29	0.40
d 28 wt, kg	13.89	13.96	13.79	14.06	13.80	14.04	1.40	0.82	0.34	0.40
<i>Phase 3 (d 28 to d 42)</i>										
ADG, kg/d	0.63	0.65	0.64	0.64	0.65	0.64	0.058	0.08	0.86	0.73
ADFI, kg/d	10.72	10.36	10.51	10.57	10.52	10.56	1.156	0.15	0.78	0.85
Gain:Feed	0.59	0.63	0.61	0.60	0.61	0.61	0.034	0.001	0.19	0.92
d 42 wt, kg	22.75	23.15	22.81	23.09	22.84	23.06	1.948	0.32	0.49	0.59
<i>Phase 4 (d 42 to d 56)</i>										
ADG, kg/d	0.84	0.88	0.87	0.86	0.87	0.86	0.063	0.003	0.25	0.58
ADFI, kg/d	15.75	14.40	15.21	14.94	14.94	15.21	1.683	0.002	0.45	0.43
Gain:Feed	0.54	0.62	0.58	0.58	0.59	0.57	0.059	0.001	0.95	0.13
d 56 wt, kg	34.58	35.53	35.02	35.08	34.99	35.11	2.416	0.06	0.90	0.82
<i>Overall (d 0 to d 56)</i>										
ADG, kg/d	0.45	0.47	0.46	0.46	0.46	0.46	0.133	0.58	0.98	0.82
ADFI, kg/d	9.44	8.93	9.19	9.17	9.10	9.27	0.818	0.003	0.91	0.31
Gain:Feed	0.47	0.52	0.49	0.50	0.49	0.50	0.141	0.13	0.98	0.85

<sup>a</sup>Data represents 48 of 72 planned observations for diet, 16 of 24 planned observations for storage, and 48 of 72 planned observations for sex.

**Table 3. Effects of diet, manure storage type, and sex on grow-finish performance (preliminary data)<sup>a</sup>.**

Main Effects	Diet		Storage		Sex		MSE	P Values		
	Control	LNE	Deep Pit	Pull Plug	Barrows	Gilts		Diet	Storage	Sex
d 56 wt, kg	34.58	35.53	35.02	35.08	34.99	35.11	2.416	0.06	0.90	0.82
<i>Grower 1 (d 56 – 84)</i>										
ADG, kg/d	0.93	0.91	0.92	0.92	0.94	0.90	0.069	0.30	0.63	0.006
ADFI, kg/d	20.39	18.42	19.30	19.51	19.46	19.36	2.521	0.003	0.69	0.87
Gain:Feed	0.46	0.50	0.48	0.47	0.48	0.47	0.071	0.003	0.44	0.35
d 84 wt, kg	60.68	61.09	60.82	60.95	61.38	60.39	3.377	0.55	0.85	0.15
<i>Grower 2 (d 84 -112)</i>										
ADG, kg/d	0.97	1.08	1.03	1.02	1.04	1.01	0.145	0.002	0.67	0.28
ADFI, kg/d	27.19	26.01	27.18	26.01	27.16	26.04	2.359	0.02	0.02	0.03
Gain:Feed	0.35	0.41	0.38	0.39	0.38	0.39	0.066	0.001	0.37	0.58
d 112 wt, kg	88.01	91.36	89.69	89.69	91.07	88.30	4.213	0.002	1.00	0.002
<i>Finisher 1 (d 112 – 140)</i>										
ADG, kg/d	1.05	1.08	1.06	1.07	1.09	1.04	0.231	0.29	0.78	0.24
ADFI, kg/d	29.88	28.60	29.40	29.08	29.60	28.88	3.546	0.08	0.67	0.33
Gain:Feed	0.34	0.37	0.35	0.35	0.35	0.35	0.045	0.006	0.80	0.62
d 140 wt, kg	114.75	119.02	116.69	117.09	118.85	114.93	5.262	0.001	0.71	0.004
<i>Finisher 2 (d 140 – 152)</i>										
ADG, kg/d	0.81	0.81	0.83	0.79	0.80	0.82	0.265	0.93	0.52	0.84
ADFI, kg/d	32.44	29.27	31.02	30.69	31.39	30.32	4.530	0.009	0.72	0.25
Gain:Feed	0.25	0.27	0.26	0.25	0.25	0.26	0.079	0.15	0.51	0.32
d 152 wt, kg <sup>b</sup>	125.68	130.64	128.44	127.88	129.73	126.59	6.237	0.001	0.71	0.04

<sup>a</sup>Data represents 48 of 72 planned observations for diet, 16 of 24 planned observations for storage, and 48 of 72 planned observations for sex.

<sup>b</sup>Data represents 36 of 72 planned observations for diet, 12 of 24 planned observations for storage, and 48 of 72 planned observations for sex.

**Table 4. Effects of diet, manure storage type, and sex on ultrasound backfat and loin eye area scans of grow-finish pigs (preliminary data)<sup>a</sup>.**

Main Effects	Diet		Storage		Sex		MSE	P Values		
	Control	LNE	Deep Pit	Pull Plug	Barrows	Gilts		Diet	Storage	Sex
<i>Grower 1, d 56</i>										
Loin eye area, cm <sup>2</sup>	13.82	14.52	14.17	14.17	13.86	14.48	1.907	0.08	0.99	0.12
Backfat, mm	8.01	8.85	8.29	8.57	8.59	8.26	1.089	0.003	0.20	0.14
<i>Grower 2, d 84</i>										
Loin eye area, cm <sup>2</sup>	21.81	22.19	22.28	21.72	21.47	22.52	3.276	0.57	0.41	0.12
Backfat, mm	9.50	11.07	10.16	10.41	10.47	10.10	2.617	0.004	0.64	0.49
<i>Finisher 1, d 112</i>										
Loin eye area, cm <sup>2</sup>	31.29	31.73	31.95	31.08	31.32	31.70	3.372	0.52	0.21	0.58
Backfat, mm	13.40	14.96	13.90	14.45	15.16	13.19	1.844	0.001	0.15	0.001
<i>Finisher 2, d 140</i>										
Loin eye area, cm <sup>2</sup>	34.80	36.16	36.01	34.95	35.61	35.34	2.733	0.04	0.11	0.68
Backfat, mm	16.33	18.63	17.50	17.47	19.44	15.52	2.709	0.006	0.96	0.001
<i>Market, d 152<sup>b</sup></i>										
Loin eye area, cm <sup>2</sup>	37.68	39.42	39.11	37.99	38.15	38.95	2.550	0.005	0.07	0.19
Backfat, mm	18.06	20.95	19.59	19.41	21.49	17.51	2.843	0.001	0.79	0.001

<sup>a</sup>Data represents 48 of 72 planned observations for diet, 16 of 24 planned observations for storage, and 48 of 72 planned observations for sex.

<sup>b</sup>Data represents 36 of 72 planned observations for diet, 12 of 24 planned observations for storage, and 48 of 72 planned observations for sex.



**Table 5. Effects of diet, manure storage type, and sex on slaughtered carcass characteristics of finishing pigs (preliminary data)<sup>a</sup>.**

Main Effects	Diet		Storage		Sex		MSE	P Values		
	Control	LNE	Deep Pit	Pull Plug	Barrows	Gilts		Diet	Storage	Sex
Backfat depth, mm	21.66	23.69	22.57	22.78	24.69	20.66	2.308	0.001	0.66	0.001
Loin depth, cm	6.55	6.52	6.57	6.50	6.51	6.56	0.242	0.57	0.16	0.32
Lean, %	53.80	53.24	53.61	53.43	52.98	54.06	0.629	0.001	0.18	0.001
Base meat price, \$/kg	1.27	1.25	1.26	1.26	1.26	1.27	0.108	0.29	0.92	0.58
Carcass grade premium, \$/kg	0.11	0.10	0.11	0.10	0.09	0.12	0.022	0.003	0.09	0.001
Carcass value, \$/kg	1.39	1.35	1.37	1.37	1.35	1.39	0.111	0.11	0.79	0.05
Hot carcass wt, kg	93.13	96.72	95.13	94.72	96.40	93.45	4.612	0.003	0.66	0.002
Live value, \$/kg	1.03	1.01	1.02	1.01	1.00	1.03	0.083	0.16	0.47	0.05
Yield, %	74.00	74.31	74.61	73.69	73.97	74.34	1.833	0.41	0.02	0.32
Total carcass value, \$ <sup>b</sup>	129.03	130.36	130.24	129.15	129.54	129.86	9.855	0.51	0.59	0.87

<sup>a</sup>Data represents 48 of 72 planned observations for diet, 16 of 24 planned observations for storage, and 48 of 72 planned observations for sex.

<sup>b</sup>Total carcass value (\$) = Carcass grade premium (\$/kg) \* Hot carcass wt (kg).

**Table 6. Effects of diet and manure storage type on air concentration data in a wean-finish confinement building (preliminary data)<sup>a</sup>.**

Gas Concentrations	Diet		Storage		MSE	P Values	
	Control	LNE	Deep Pit	Pull Plug		Diet	Storage
NH <sub>4</sub> (PSA), ppm	6.6	5.7	6.4	6.0	3.52	0.001	0.005
H <sub>2</sub> S, ppb	136.1	142.1	139.4	138.8	286.63	0.65	0.96
SO <sub>2</sub> , ppb	11.1	11.5	9.9	12.7	32.89	0.81	0.07
CO <sub>2</sub> (PSA), ppm	1308.3	1312.5	1318.0	1302.8	353.87	0.86	0.33
CH <sub>4</sub> , ppb	10.5	10.4	11.5	9.4	6.37	0.83	0.001

<sup>a</sup>Data represents 8 of 12 planned observations for diet and storage.

**Table 7. Effect of week of production on air concentration data in a wean-finish confinement building (preliminary data)<sup>a</sup>.**

Dietary phase:	Nursery	Grower 1	Grower 2	Finisher 1	Finisher 2	MSE	P Values
Week of production:	4	8	12	16	20		Wk of Production
<i>Gas concentrations</i>							
NH <sub>4</sub> (PSA), ppm	3.4	6.6	8.6	6.1	6.3	3.52	0.001
H <sub>2</sub> S, ppb	56.9	171.2	143.9	179.9	149.5	286.63	0.001
SO <sub>2</sub> , ppb	9.9	14.1	13.3	11.2	8.3	32.89	0.24
CO <sub>2</sub> (PSA), ppm	1424.7	1311.5	1393.7	1157.5	1231.5	353.87	0.001
CH <sub>4</sub> , ppb	7.2	11.4	13.0	10.2	10.7	6.37	0.001

<sup>a</sup>Data represents 16 of 24 planned observations for wk of production.



## Estimating Annual NH<sub>3</sub> Emissions from U.S. Broiler Facilities

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### Abstract

Recently, several U.S.-based research projects have been completed to acquire ammonia emissions baseline data for broiler housing (Wheeler et al., 2004, 2006; Burns et al., 2003) and layer housing (Liang et al., 2005). The issue of estimating these operations' contribution to an annual ammonia emission budget needs to be resolved. For the case of layer houses, ammonia emission is strongly impacted by the nature of manure management within the building, with emissions from high-rise facilities typically an order of magnitude greater than those from manure-belt facilities. Thus, for a layer facility the use of a standard emission factor (EF) is not unreasonable, provided that the factor reasonably takes into account the parameters which influence emission rate, namely temperature, ventilation rate, number, size and age of birds, etc).

Production facilities in which poultry or animals grow rapidly, i.e. "meat-type" animals, present a completely new challenge to standard methods for estimating emissions. For example, broiler operations are uniquely different than layer operations, because the birds are grown from day-old to market weight. While consumer demand drives the specific mature weight of a "broiler (from "Cornish hens" weighing 1 kg to "roasters" weighing 4 kg), the fact remains that building emission rate changes with bird size. Integrating this variable emission rate over an entire year is the ideal means of estimating the annual contribution to an ammonia emission inventory. Conceptually, one could estimate a mean daily emission rate, expressed for example as (g NH<sub>3</sub> bird<sup>-1</sup> day<sup>-1</sup>), or (kg NH<sub>3</sub> house<sup>-1</sup> day<sup>-1</sup>), and then multiply by the number of days per year in which birds are present in a facility. This approach requires knowledge of number of flocks grown per year, mature weight of each flock, and down-time between flocks.

We have developed an alternative emissions estimate (Gates et al., 2005), using results from the recently completed U.S. broiler emission project (Wheeler et al, 2004, 2006). Emission rate increases in a linear relationship with flock age from near zero at the start of the flock to a maximum at the end, 28 to 63 days later.

An estimate of daily NH<sub>3</sub> emissions per bird ( $\pm$ std. dev.) from these data is thus:

$$ER_b = 0.031(\pm 0.0057) \cdot x, \text{ where } x = \begin{cases} \text{age,} & \text{if used litter} \\ 0, & \text{if } 1 < \text{age} < 6; \text{age} - 6, \text{if new litter} \end{cases} \quad (1)$$

where

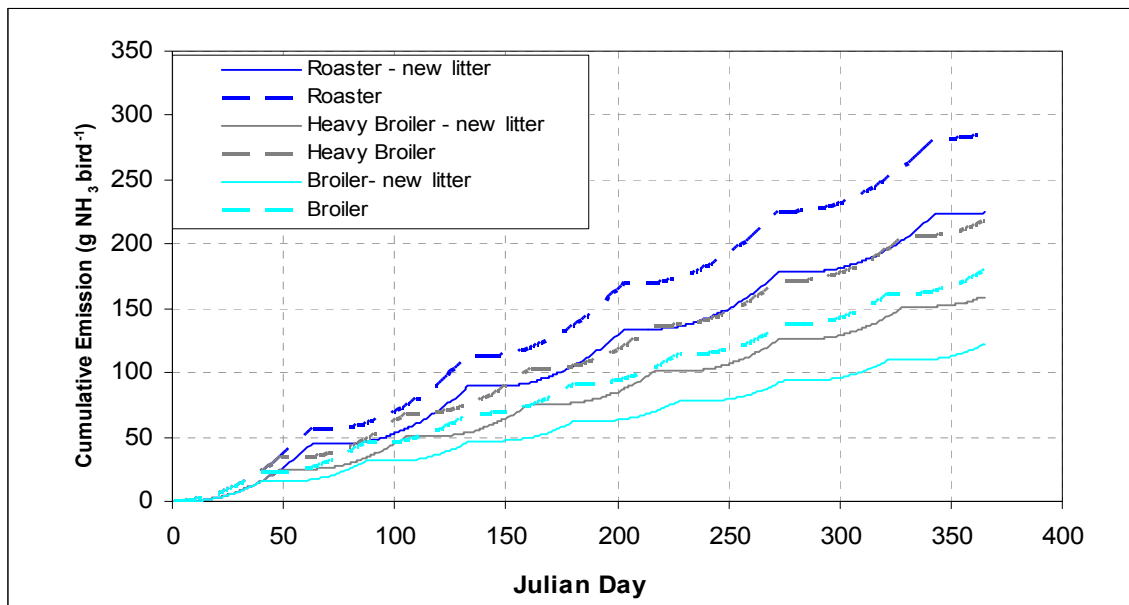
ER<sub>b</sub> = emissions rate, g NH<sub>3</sub> bird<sup>-1</sup> d<sup>-1</sup>  
age = bird age, d

On a typical broiler farm, five to eight flocks (per house) are grown annually depending on finished bird weight and market demand. The houses are empty for seven to fourteen days between flocks while cleaning and maintenance is accomplished. Production houses may be empty for a further fourteen to twenty-one days to allow for annual maintenance and litter removal. We present a model that takes into account; broiler market weight, numbers slaughtered and ammonia emissions to compute annual emissions estimates. The method can be readily applied and should provide for a more accurate annual budget estimate than is currently available. In addition, because the slope in equation (1) has a known standard

error (i.e.  $0.0057 \text{ g NH}_3 \text{ bird}^{-1} \text{ d}^{-2}$ ), we can assess the impacts of a) fresh litter vs. re-used litter, b) number of flocks per year and market weight of birds, which can vary between flocks, and c) uncertainty in emission estimates.

The model was used to evaluate some of these effects, using a typical Kentucky broiler house (12m x 150m). Interestingly, the annual emission rate was less for broilers (28,000 birds per flock, 2.1 kg mature weight @40 d) than for heavy broilers (24,000 birds per flock, 2.45 kg mature weight @49 d) or for roasters (18,000 birds per flock, 3.25 kg mature weight @63 d), 3,786, 4,219 4,473 kg  $\text{NH}_3 \text{ house}^{-1} \text{ yr}^{-1}$  respectively (new litter). This can be explained by the fact that while fewer birds were raised in the houses with larger market weights (5.2, 6.7 and 7 flocks per year), there are more days with older birds. Another interesting finding was that the effect of new versus re-used litter was substantial (27%, 37% and 47% potential reduction for broiler, heavy broiler and roaster birds, respectively). Unfortunately, this potential for reduction occurs at a heavy price, namely a 5 to 10-fold increase in litter costs, a 5 to 7-fold increase in litter volume to handle and appropriately dispose, and a substantially reduced fertilizer value.

Application of a reasonable prediction interval on these estimates was performed by adjusting the slope in equation (1) by  $\pm 3$  standard errors, as depicted graphically in Figure 1. This demonstrates the variable nature of emissions estimates, and with annual emission (the end-point for each curve) for broiler, heavy broiler and roaster facilities ranging from 3,420, 3,811, and 4,040 kg  $\text{NH}_3 \text{ house}^{-1} \text{ yr}^{-1}$  respectively (new litter), with similar differences determined for new versus re-used litter.



**Figure 1. Cumulative ammonia emission over a year, from multiple flocks, for roaster, heavy broiler or broilers, and new vs. re-used litter, assuming mean ER – 3 SE<sub>ER</sub> as per Equation 1**

The broiler emission inventory model was used to estimate annual emission from Commonwealth of Kentucky broiler facilities, using USDA’s National Agricultural Statistics Service data for 2002. The predicted emission ranges from 8,844 to 11,696 metric tons for new and re-used litter, respectively. This compares with the current EPA National Emission Inventory estimate of 664,238 metric tons  $\text{NH}_3$  in 2002. For this time period, Kentucky production was about 3.2% of national production, and if we assume a proportional contribution to national emission inventory, this is 21,255 metric tons. This clear discrepancy between the model predictions and current EPA estimates (a factor of 2 to 3) suggests a need to refine the national inventory estimates in the case of meat-type animals.

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