Agricultural Emissions

Ammonia Emissions from Agriculture and Other Sources

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Abstract

Emissions of ammonia play an important role in the formation of fine particles in the atmosphere. Under typical atmospheric conditions, ammonia reacts with gaseous emissions of sulfur dioxide and oxides of nitrogen to form sulfate and nitrate fractions of fine particles (defined as particles less than 2.5 microns in aerodynamic diameter). In many parts of the United States, this fraction is as much as 50% of the total fine particle mass.

Thus, there is an urgent need for better quantification of ammonia emissions. There is also a need for high spatial and temporal resolution of ammonia emissions at scales that are compatible with input requirements for sophisticated chemical transport models. While inventories for many regulated species such as sulfur dioxide and nitrogen oxides are generally available with high resolution, such inventories for ammonia are much less common. Yet knowledge of ammonia emissions at high spatial and temporal resolution is necessary to reliably predict atmospheric concentrations of fine particles.

Many inventories in use today have been derived from the NAPAP inventory of 1985. The NAPAP numbers are often updated using economic factors to artificially "grow" the inventory rather than using real data, even when such data are readily available. This is partially because inventories can consist of thousands of spreadsheets, and it is a great deal of work to update all of these spreadsheets when new data become available. Thus, a key problem with existing ammonia inventories is that they are typically constructed for a specific year and location and do not have the capacity to be easily revised and updated. Nevertheless, there are several reasons why an inventory should be flexible enough to accommodate new information as it becomes available:

- An inventory is typically developed for a specific year. However, end users often need to modify the inventory to reflect a different year.
- Existing activity levels and emission factors for sources of ammonia are uncertain. When better values are released, it is important to include them in the inventory.
- It is sometimes difficult to perform sensitivity analyses using existing inventories. Such analyses are critical in quantifying the contribution of various sources; for example, when considering alternative control strategies.

To remedy this situation, a national ammonia emission inventory has been created in a format that will easily accommodate new information. Instead of a static file, the inventory is in the form of a model that is designed to allow the user to easily update emission factor or activity level data and then regenerate the desired output. Although there are currently many research groups working to improve emission factor and activity level data, some of these data are still of poor quality. Once new data are available, it will be easy to replace the outdated factors in the inventory. This structure also makes it possible to perform sensitivity analyses, an important tool for both regulators and modelers. Furthermore, the output is compatible with ARCVIEW, a GIS program. This is important both because the user is then able to view the inventory graphically, and because ARCVIEW files are compatible with EMS95, a common preprocessor to many atmospheric chemistry models. Thus one of the main contributions of this work is not simply a new national inventory but a framework that makes it possible to easily adjust and modify this inventory for use in different applications.

Measuring and Modeling Gaseous NH_3 and Aerosol NH_4 at the Regional Scale: How Does Ambient Concentration Respond to Emission Controls?

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Abstract

Controlling emissions of NH3 from agriculture is notoriously difficult, as it combines an inadequately known science covering a very disparate range of activities in a complex industry with social and political issues. Thus the modest reductions in many countries could be viewed as significant progress. With this background, it is helpful to show from a combination of monitoring and modeling approaches, the responses of both ambient concentrations and the gas to aerosol partitioning to changes in emissions. In this paper, changes in the chemistry of the atmosphere over the United Kingdom and other countries in Europe during a period in which emissions of SO2, NOx and NH3 have changed appreciably are explored from monitoring data and using a long range transport models. Sulphur emissions in the UK declined by 80% between 1986 and 2005, from 2300 kT S annually in 1986 to 400 kT S in 2005. During the same period emissions of NH3 declined by just 15% to 300kT. NH3-N and emissions of NOx declined by about 45%, to 450 kT NOx-N. The effect of these emission reductions changed the acidic aerosol mixture in the 1980s dominated by (NH4)2SO4 and NH4HSO4 to a more neutral aerosol dominated by NH4NO3 in recent years. Concentrations of HNO3 also increased during this period and the very different fractionation of both oxidized and reduced nitrogen species with different atmospheric lifetimes leads to substantially different deposition footprints for each of the pollutants and different patterns of deposition relative to those of the 1980s. To secure the greatest reduction in deposition in the near field for NH3 (<10km), it is necessary to reduce the emission directly. At the larger (regional) scale (10km to 200km) changes in atmospheric composition have increased the transport distance of NH3 and in principle partitioned more of the deposition as dry deposition. Long term monitoring data show a small decline in wet deposition of NH4+, consistent with the changes in chemistry, but overall, the response are small.

Odor Emissions and Chemical Analysis of Odorous Compounds from Animal Buildings

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Abstract

Odor emission from animal production buildings is a critical local issue according to the National Research Council report to the livestock and poultry industries (NRC, 2003) and people living and working near these operations. Even though federal and some state agencies do not regulate odors, emission of odorous compounds remains a high priority for animal producers (for siting new and expanding existing operations) and for neighbors living near livestock and poultry operations. There is an urgent need for odor emission factors from animal confinement buildings since very limited data is presently available. Odor emission factors are needed as inputs into dispersion models so odor concentrations can be calculated for existing and future livestock and poultry operations and utilized for the addition of control technologies on current operations and the siting and design of new facilities. This study will supplement the National Air Emissions Monitoring Study (NAEMS), which will soon be announced, with comprehensive measurements of odor emissions from eight separate animal buildings in the U.S. The NAEMS will monitor air emissions at swine and layer sites, and potentially at turkey, broiler, and dairy production facilities. The NAEMS will help livestock and poultry producers to comply with EPA regulations concerning regulated gases and particulate matter by monitoring these pollutants continuously for 24 months to fulfill the requirements of a consent agreement. Although odor is the air pollutant that plagues the animal industry with the greatest overall challenge, it is not included in the NAEMS because the EPA does not regulate it. One of the specific goals of the NAEMS is to provide the basic infrastructure for add-on projects that conduct additional measurements. Investigators in this project will add odor emission measurements to the NAEMS. The measurements will include both standard human sensory measurements using dynamic forced-choice olfactometry and a novel chemical analysis technique for odorous compounds found in these emissions. The sensory and chemical methods will be correlated to gain both quantitative and qualitative understanding of odor emissions from animal buildings.

Introduction

Odor emission from animal production buildings is a critical local issue, according to the National Research Council report to the livestock and poultry industries (NRC, 2003) and people living and working near these operations. Even though federal and some state agencies do not regulate odors, emission of odorous compounds remains a high priority for animal producers (for siting new and expanding existing operations) and for neighbors living near livestock and poultry operations. There is an urgent need for odor emission factors from animal confinement buildings since very limited data is presently available.

This study proposes to supplement the air emissions monitoring 2006-2009 National Air Emissions Monitoring Study (NAEMS) with comprehensive measurements of odor emissions. The NAEMS will monitor barn emissions at swine, dairy, broiler and egg layer facilities. The NAEMS will help livestock and poultry producers to comply with the Environmental Protection Agency (EPA) regulations concerning regulated gases and particulate matter by monitoring these pollutants continuously for 24 months to fulfill the requirements of a consent agreement. Although odor is the air pollutant that plagues the animal industry, it is not included in the NAEMS because it is not regulated by the EPA.

The odor measurements to be made in this study includes both standard human sensory measurements using dynamic forced-choice olfactometry and a novel chemical analysis technique for odorous compounds found in these emissions. The sensory and chemical methods would be correlated to gain both quantitative and qualitative understanding of odor emissions from animal buildings. The specific objectives of this

study include: (1) determine odor emission factors of the NAEMS sites using common protocol and standardized olfactometry for use in air dispersion models and evaluation of controls, (2) develop a comprehensive chemical library that delineates the most significant odorants and correlate this library with olfactometry results for the NAEMS sites, and (3) disseminate information to stakeholders, including producers, agencies, regulators, researchers, local government officials, consultants, and neighbors of animal operations.

Background

Livestock and poultry producers in the United States are becoming increasingly concerned over the odors and gases that are generated and emitted from their animal operations. Odors and gas emissions from animal production sites are impacting producers in a variety of ways. Complaints from neighbors are increasing. Local units of government (counties and townships) have or are considering the establishment of setback requirements from rural residences and livestock operations to prevent odor and other nuisance complaints. State and federal regulatory agencies have begun to enforce existing air standards or enact new standards. These enforcements are being addressed during environmental review procedures of state or federal permitting processes.

Because of these growing concerns, there is a need to determine odor and gas emissions levels from animal production sites, such as the buildings, associated manure storage units, and on-farm outdoor feed storages. Emissions need to be known so producers and others can determine which sources are the major contributors. Individuals can then develop an air emission strategy for their operation. Unfortunately, quantifying air emissions from animal agriculture is a complex process. First, the complexity arises from the multitude and variety of individual sources responsible for emissions, the extreme variability of these emissions, and the variety of gaseous components being emitted. Secondly, robust method(s) are necessary to collect emission data from these sources, which involve the measurement of both the concentrations of the contaminant and the airflow rate from the source. Few researchers and engineers have taken on the task of measuring odor and/or gas emission rates because of these and other difficulties.

Most livestock and poultry odors are generated by the anaerobic decomposition of livestock wastes such as manure (feces and urine), spilled feed, bedding materials, and wash water. This decomposition of organic compounds results in the production of odorous volatile compounds that are metabolic intermediates or end products of microbial processes (Zhu et al., 1999). Many of these compounds are then carried by airborne particulates (NCARS, 1995) and dispersed into the atmosphere.

Most odors are a mixture of many different gases at extremely low concentrations. The composition and concentrations of the gas mixtures affects the perceived odor. To completely measure an odor, each gas would need to be measured. Most odorous gases can be detected (smelled) by humans at very low concentrations (Table 1). The fact that most odors are made up of many different gases at extremely low concentrations makes it very difficult and expensive to determine the exact composition of an odor.

Odor Measurements

There are two general approaches used to measure odor, either measure the concentrations of each important odorant gas or use olfactometry to evaluate the entire mixture using the human nose. Both approaches have strengths and weaknesses. Future developments will hopefully close the gap between the two approaches.

The specific individual gaseous compounds in an air sample can be identified and measured using a variety of sensors and techniques. The results can be used to compare different air samples. With good sensors and proper techniques, valuable information about the gases that emanate from a source can be collected and evaluated. Gas emission rates and control techniques can be compared rigorously. Regulations can be established to limit individual gas concentrations.

However, the gas measurement approach has some weaknesses when used to measure and control odors. The greatest weakness of the gas measurement approach is that the relationship between specific gas concentrations in a mixture and their perceived odor are unknown (Ostojic and O'Brien, 1996). As a result,

regulations based on gas concentrations may reduce specific gas emissions and concentrations but not adequately address the odors sensed downwind by the neighbors.

Table 1. Odor threshold for selected chemicals found in livestock odors (Kreis, 1978)

Chemical	Odor Detection (ppm)
Aldehydes	
Acetaldehyde	0.21
Propionaldehyde	0.0095
Volatile Fatty Acids	
Acetic acid	1.0
Propionic acid	20.0
Butyric acid	0.001
Nitrogen containing	
Methylamine	0.021
Dimethylamine	0.047
Trimethylamine	0.00021
Skatole	0.019
Ammonia	46.8*
Sulfur containing	
Methanethiol	0.0021
Ethanethiol	0.001
Propanethiol	0.00074
t-Butythiol	0.00009
Dimethyl sulfide	0.001
Hydrogen sulfide	0.0072

*More recent values are 1 to 5 ppm (Reynolds, et al. 1998)

People have proposed using "indicator" gases to quantify livestock odors. Hydrogen sulfide and ammonia are among the most common chemicals proposed. Unfortunately, hydrogen sulfide and ammonia concentrations are not well correlated to livestock odor (Spoelstra, 1980; Pain and Misselbrook, 1990; Jacobson et al., 1997; Zahn et al., 1997). Yashuhara (1980) found that a mixture of 11 compounds strongly resembled the quality and character of solid swine manure. Livestock odors consist of many gases at extremely low concentrations, which are very difficult and expensive to measure. Measuring some of the gases may not be enough to describe the odor. Research and development of new, better, and lower cost sensors is ongoing. Electronic noses, which use electronic sensors to measure a select number of chemical compounds, are being used in some industries for quality control of various odor-producing processes. Most studies indicate that the outputs of electronic noses do not correlate with livestock odors (Watts, 1992; McFarland and Sweeten, 1994); however, one study suggests that technological developments may make it possible in the future (Misselbrook et al., 1997).

Malodor characterization is among the most demanding of all analytical challenges. This occurs because it is usually the case that aroma or odor critical components are present at very trace levels in a complex matrix of odor-insignificant volatiles (Wright et al, 1986). A large body of excellent analytical work has been reported during the past three decades relative to the volatile compounds emitted by high-density livestock operations. Scores of volatile compounds have been identified in these environments utilizing various concentrating and analytical techniques (Mosier et al. 1973; Hutchison et al. 1982; Oehrl et al. 2001; Keener et al. 2002; McGinn et al. 2003; Nielsen et al. 2004). Included among these volatiles are a large number of compounds that are known to be potent individual odorants (Devos et al. 1990). The challenge relative to the odor issue is to extract from this large field of "potential" odorants, the compounds that constitute the primary odor impact relative to these environments. Given sufficiently comprehensive and accurate reference and analytical data regarding the volatile compounds present in these environments, it would seem possible to accurately predict and rank the primary odor impact compounds. However, from a practical standpoint, this does not produce satisfactory results in most cases. The factors working against such success are incomplete or imprecise odor threshold data in concert with the extremely low odor thresholds of many if not most of the key odorants present.

A practical alternative is to carry out GC-olfactometry (i.e. GC-O)-based odor profile ranking studies relative to in-situ headspace volatiles collections taken directly from the target environment. This is the approach that is routinely taken in investigating odor issues surrounding matrices for which limited volatiles compositional data are available. The general experimental approach is to develop a detailed odorant ranking profile for a sensory graded "worst" case sample. Performing equivalent comparative odorant ranking profile analysis for equivalent sensory graded "best" case samples will typically indicate which of the "potential" odorant(s) present in the field account for the odor character differences between the two samples.

The necessity of prioritizing the individual odor carrying volatiles relative to a particular malodor issue is often overlooked in odor-focused investigations. Over the past decades, such prioritization is essential to the resolution of the typical crisis-driven malodor problems. Scores of these investigations have been successfully affected during this period; ranging from aroma and flavor complaints in foods and beverages to malodors in packaging, consumer products, and work environments. Collaborative efforts undertaken with Texas A&M - Texas Agricultural Experimental Station, Amarillo (Koziel) and West Texas A&M, Canyon (Parker) are directed at applying to high-density livestock operations some of the lessons learned in addressing these past, highly-diverse odor-focused investigations (Wright et al., 2004). Odor profiling by GC-O has proven to be an essential element required for defining, prioritizing. and tracking the basic building blocks of odor character in complex matrices (Wright et al., 1986; Nielsen et al. 2001; Willers et al. 2003).

Figure 1 is an "aromagram" that was generated by the GC-O investigator (shown in use in Figure 2) monitoring the odor impact of the individual compounds as they elute from the GC column. The retention time span of the peaks reflects the start and end time for the individual odor responses while the peak height reflects the perceived intensity of these responses. By overlaying these sensory responses with the MS signal, it is possible to correlate the sensory responses with corresponding electronic signals and odorant identification, respectively. At least 66 discrete odor notes were detected under the conditions of collection and many of these reflected intense to overwhelming odor intensities. The full range of previously reported swine farm odorants were detected, including: H_2S and its organic homologs; trimethylamine; VFA's, ranging from acetic to octanoic; phenolics, including phenol, p-cresol and p-ethyl phenol; indole, skatole and a wide variety of ketones, diones, and aldehydes, among others. A summary of a few of the major odorant compounds from this odor profile analysis is presented in Table 2.

Olfactometry, the most common sensory method, uses trained individuals and standardized procedures to measure odor levels and describe odors (see figure 3). The key advantage of olfactometry is the direct correlation with odor and its use of the human's highly sensitive sense of smell. Olfactometry also has the advantage that it analyzes the complete gas mixture so that contribution of each compound in the sample is included in the analysis. There are different olfactometry techniques. Data collected by different techniques can be neither combined nor directly compared.

McFarland (1995) reviewed many of the current olfactometry techniques being used for odor measurement and concluded that dynamic forced-choice olfactometry appears to be the most accepted method. Olfactometry suffers from a lack of precision compared to some of the sophisticated chemical sensors available. The lack of precision in olfactometry is due in part to the variability in each person's sense of smell and their reaction to an odor. Also, olfactometry does not identify the individual compounds that make up the odor. Even though olfactometry has limitations, it still is the best technique available for directly measuring odors at this time.

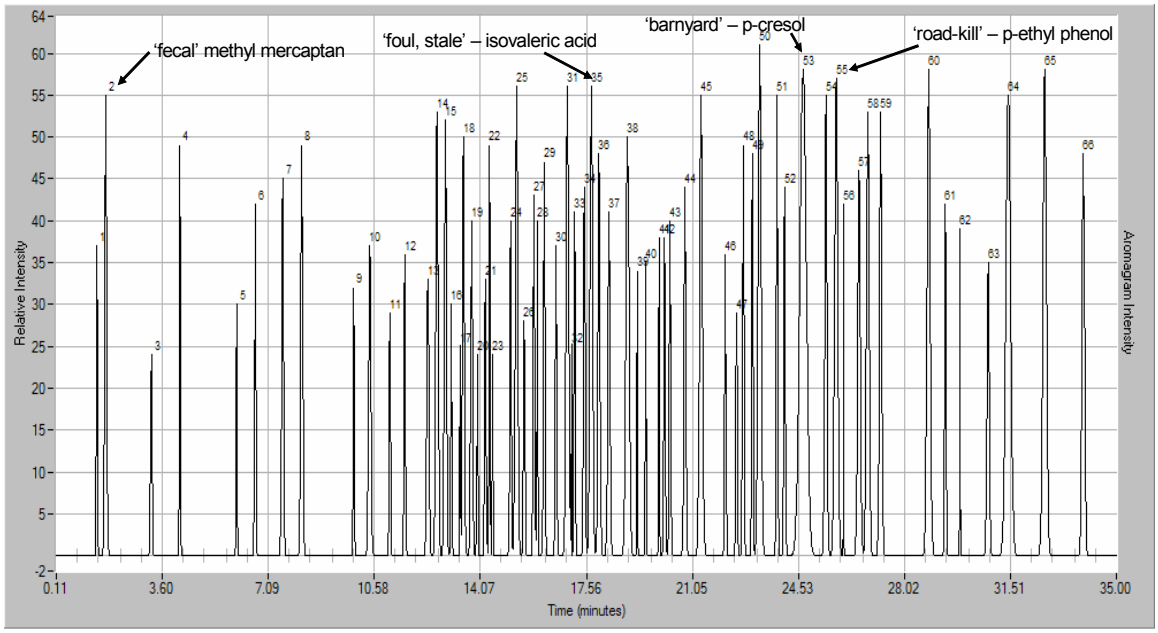

Figure 1. Aromagram from SPME odor collection at the exhaust fan of a swine finish barn.

Figure 2. Real-life photo of Aroma Trax™ MDGC-olfactometry system.

Table 2. Representative odorants from inside a tunnel-ventilated swine finish barn collected with SPME near the exhaust fan.

Peak Number	GC Column Retention Time (min)	Odor Descriptor	Preliminary Odorant Identification
1	1.42	foul, fecal	hydrogen sulfide
2	1.68	fecal	methyl mercaptan
3	1.70	fishy	trimethylamine
4	4.15	buttery	diacetyl
6	6.60	amine	unknown amine or diamine
7	7.60	grassy	hexanal
10	10.30	buttery	pentanedione
14	12.60	savory, nutty	dimethyl pyrazine
18	13.45	musty, vinegar	acetic acid
20	13.85	fecal	dimethyl trisulfide
25	15.20	vomitus, body odor	propionic acid
27	15.85	cardboard, musty	? nonenal
31	16.80	vomitus, body odor	butyric acid
35	17.60	body odor, foul	isovaleric acid
38	18.80	foul, characteristic	valeric acid
45	21.30	medicinal	guiacol
50	23.14	medicinal, floral	phenol
52	24.10	beet, vegetable	geosmin
53	24.40	barnyard, characteristic	para-cresol
54	25.80	roadkill, decay, foul	para-ethyl phenol
58	27.15	taco shell, bat cave	2'-aminoacetophenone
60	28.70	outhouse	para-vinyl phenol
62	29.83	outhouse	indole
63	30.70	outhouse, naphthalenic	skatole
64	31.26	floral, honey	phenyl acetic acid
65	32.50	taco shell, bat cave	1-(2-aminophenyl)-1-butanone

Figure 3. AC'SCENT® International Olfactometer in use by sniffer and panel leader.

Odor must first be quantified to determine odor emission values. The most common and frequently reported measure of odor is dilutions-to-threshold. Diluting air samples with a known amount of odor-free

air and presenting the dilutions to a panel of people using an olfactometer, which is an air dilution device, determines this value. Dilutions-to-threshold is the volume of odor-free air required to dilute a unit volume of odorous sample air to the point where it can be detected by 50% of a trained group of panel members (Nòren, 1987). Odor units are defined by CEN (2001) as the mass of odorants in one cubic meter of air at the odor detection threshold (1 D/T). The odor concentration of a sample is therefore expressed as odor units per cubic meter (OU/m^3) for calculation conveyance of odor emission rates (CEN, 2001). Odor emission rates (OU/s) from a livestock building or manure storage unit are the product of the ventilation airflow rate (m^3/s) through the barn or over the storage and the odor concentration (OU/m^3) in the exhaust air (Lim et al., 2001). To allow comparison with other research results, odor and gas emission rates are often specified to the number and weight of animals by dividing the total emission rate by the number of animal units (AU), where one AU is equal to 500 kg of animal live weight (Ni et al., 2000a; Wathes et al., 1997). Area-specific emission rates are determined by dividing the total emission rate by the emitting surface area (Gay et al., 2002; Groot Koerkamp et al., 1998).

Few researchers have attempted to quantify odor and gas emission rates from animal housing, and results are widely variable. Table 3 lists odor emission rates measured from buildings for various animal species. This variation likely stems from the lack of standardized methods used to measure both odor concentration and emissions. Lim et al. (2001) evaluated odor emission and characteristics at two commercial swine nurseries during the spring. Five sampling visits were made to each nursery and nine or 10 air samples were collected during each visit. Zhu et al. (2000b) measured odor at seven different facilities to determine daily variations. Air samples were collected every two hours over a 12-hour period during the day. Watts et al. (1994) measured odor emissions from a feedlot pen using a portable wind tunnel over a 5-day period following 64 mm of rain. The highest emission occurred about 48 hours after the last rainfall. The peak odor concentration was about 60 times higher than odors from the dry pen.

Table 3. Odor emission rates from animal housing

Species	**Production unit**	**Location**	Odor Emission Rate **$OU\ m^{-2}\ s^{-1}$**	**Reference**
Pigs	Nursery (deep pit)	Indiana	1.1-2.7	Lim et al. (2001)
	Nursery	Minnesota	7.3-47.7	Zhu et al. (2000b)
	Finishing	Minnesota	3.4-11.9	Zhu et al. (2000b)
	Farrowing	Minnesota	3.2-7.9	Zhu et al. (2000b)
	Gestation	Minnesota	4.8-21.3	Zhu et al. (2000b)
	All types	Minnesota	0.25-12.6	Gay et al. (2002)
Poultry	Broiler	Minnesota	0.1-0.3	Zhu et al. (2000b)
	All types	Minnesota	0.3-3.5	Gay et al. (2002)
Dairy	Free-stall	Minnesota	0.3-1.8	Zhu et al. (2000b)
	All types	Minnesota	1.3-3.0	Gay et al. (2002)
Beef	Feedlot	Minnesota	4.4-16.5	Gay et al. (2002)
	Feedlot	Australia	12.5-725	Watts et al. (1994)

Gay et al. (2002) have recently summarized odor emission rates from over 80 farms in Minnesota. Mean values for swine housing varied from 0.25 to 12.6 OU $m^{-2}s^{-1}$, poultry housing from 0.32 to 3.54 OU $m^{-2}s^{-1}$, dairy housing from 1.3 to 3.0 OU $m^{-2}s^{-1}$, and beef feedlots from 4.4 to 16.5 OU $m^{-2}s^{-1}$. Ventilation rates for mechanically ventilated buildings were calculated as the sum of the airflow rates for each fan. Fan airflow rates were determined by measuring static pressure across the fan using a manometer and referring to fan rating tables for the corresponding airflow values. For naturally ventilated barns, rates were estimated using mass exchange rates based on the carbon dioxide (CO_2) level between the inside and outside of the buildings. Although there is reasonably high variability, this data set suggests that odor emissions from swine housing and beef feedlots are higher than emissions from poultry and dairy housing.

The USDA-IFAFS funded a six-state project entitled "Aerial Pollutant Emissions from Animal Confinement Buildings" or APECAB, which is nearing completion and has quantified and characterized baseline emissions of odor, NH_3, H_2S, PM_{10}, and TSP from four types of swine buildings and two types of poultry buildings. The APECAB study (Jacobson and Heber, PIs) is a collaboration of land-grant

universities in Minnesota, Indiana, Iowa (Hoff, Co-PI), Illinois, North Carolina, and Texas (Koziel, co-PI, Parker collaborator). The study is utilizing common instrumentation and protocol. At each measurement site, a mobile instrument trailer is stationed between two identical or nearly identical mechanically-ventilated, confined animal production buildings, and emission measurements are quasi-continuous. An instrument trailer houses a gas sampling system (GSS), gas analyzers, environmental instrumentation, a computer, data acquisition system, controller units for the real-time PM monitors, calibration gas cylinders, and supplies and equipment needed for the study. Gas concentrations are measured at the air inlets and outlets of each building while simultaneously monitoring total building airflow rates. Odor samples were taken biweekly to determine odor emissions. Emission rates are calculated by multiplying concentration differences between inlet and outlet air by building airflow rates. The 15-month sampling duration for the APECAB project assures that long-term emissions and annual emission factors can be fully characterized. Long-term measurements allow the recording of variations in emissions due to seasonal effects, animal growth cycles, and diurnal variations.

As mentioned above, odor measurements during the APECAB study were made every two weeks at the sampling site. Air samples are taken at the exhaust and inlet locations in each of the two barns from each site and analyzed by similar olfactometry laboratories at each cooperating university. An example of the preliminary findings for the odor emission data from two gestation barns in Minnesota is shown in Figure 4.

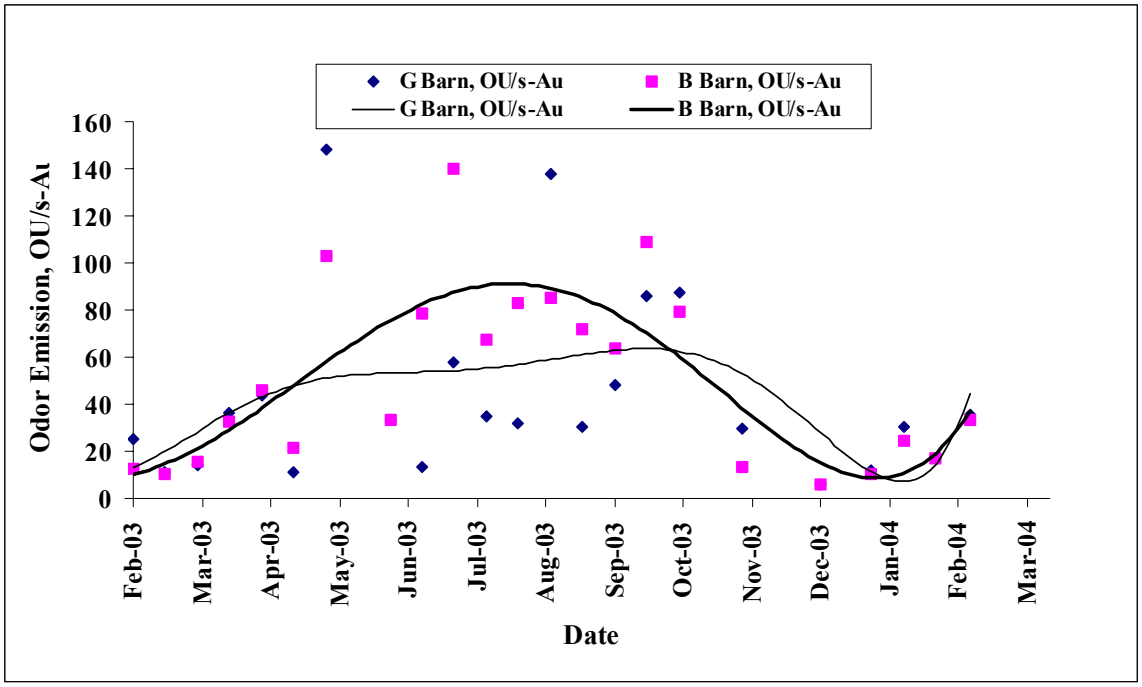

Figure 4. Odor emissions for two Minnesota sow gestation barns from Oct. 8, 2002 to Mar. 4, 2003.

Conclusions

This project will start collecting data later this year (2006) when the EPA's National Air Emissions Monitoring Study (NAEMS) begins. Selection of sites will be made during the spring/summer of 2006 and will include swine, poultry, and dairy sites. Measurements during the study will include both standard human sensory measurements using dynamic forced-choice olfactometry and a novel chemical analysis technique for odorous compounds found in these emissions.

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Effects of Reduced Crude Protein on Gaseous Emissions and Swine Performance

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Abstract

The effect of feeding reduced crude protein (CP) diets on air emissions was evaluated using groups of barrows housed in rooms with continuously measured gas concentrations and airflows. Pig weights and feed intake were recorded weekly over the course of four feeding phases: G1 (beginning at 24.5 kg body weight, BW), G2 (55.3 kg BW), F1 (87.2 kg BW), and F2 (111.4 kg BW). Pigs were offered one of three pelleted isocaloric, isolysinic diets: a control diet (C), a low crude protein diet (LCP) and an ultra low crude protein diet (ULCP), each supplemented with varying amounts of amino acids. Formulated CP of G1 was 22.5, 20.0, and 18.4% for the C, LCP, and ULCP diets. As feeding phases progressed there was a decrease in the formulated CP such that F2 was formulated to contain 16.6, 15.4, and 13.8% CP in the C, LCP and ULCP diet. Dietary treatment had no significant effect on animal performance ($P<0.05$). Pigs fed the LCP diet had greater intakes than pigs fed the C or ULCP diets during the grower phases but reduced intake during both finisher phases ($P=0.0287$). A diet effect was observed for average daily ammonia concentrations ($P<0.0001$). Exhaust ammonia (NH_3) concentration in rooms where pigs were fed the LCP diet were 16% less than the C diet (3.86 vs. 4.57 ppm). Ammonia concentrations were reduced 25% (2.93 ppm) in the ULCP diets compared to the LCP diet and 36% compared to the C. Airflow-corrected daily average NH_3 emission rates were 26.8, 21.0, and 14.5 mg min^{-1} for the C, LCP, and ULCP diets, respectively, corresponding to a daily mass of NH_3 emitted per kg of animal live weight, of 88.0, 68.9, and 46.0 mg kg^{-1}. Feeding phase effects were observed for NH_3 concentration, NH_3 emission rate, daily mass emitted and daily mass per unit of live weight with increases from G1 to G2 followed by a decrease from F1 to F2. Similarly, feeding phase effects were observed for hydrogen sulfide concentration and daily emitted mass of hydrogen sulfide per unit of live weight. Hydrogen sulfide concentration and emissions were not different between rooms offered the different dietary treatments ($P>0.05$). Diet had no effect on wet or dry mass of manure produced however TKN and NH_3-N concentration decreased with decreasing diet CP (7.9, 6.7, 5.7% and 5.4, 4.4, and 3.5%, respectively for C, LCP and ULCP diets).

Introduction

A new facility was constructed at Iowa State University specifically for the purpose of evaluating dietary effects on gaseous emissions. The facility became operational in September 2004, and a swine study was begun that same month. The objective of this paper is to describe the facility's capabilities and present the findings of the first study. The study evaluated pigs that were fed either a typical industry diet or one of two diets that contained decreasing crude protein levels by supplementing with amino acids. Animal performance and gaseous emissions were measured.

Material and Methods

Forty-eight crossbred barrows (initial bodyweight = 18 kg) were allocated to one of eight environmental rooms (six pigs per room). A 2-wk acclimation period occurred prior to the start of the feeding trial. Pigs were offered one of three pelleted isocaloric, isolysinic diets: a control diet (C), a low crude protein diet (LCP) and an ultra low crude protein diet (ULCP), each supplemented with varying amounts of amino acids. The C and the LCP diets were offered in three, each, of the eight rooms and the ULCP diet was offered in the remaining two rooms. Formulated CP of G1 was 22.5, 20.0, and 18.4% for the C, LCP, and ULCP diets. As feeding phases progressed there was a decrease in the formulated CP such that F2 was formulated to contain 16.6, 15.4, and 13.8% CP in the C, LCP and ULCP diet. Pig weights and feed intake

were recorded weekly over the course of four feeding phases: G1 (beginning at 24.5 kg BW), G2 (55.3 kg BW), F1 (87.2 kg BW), and F2 (111.4 kg BW). At the start of each feeding phase, diet assignments were randomly assigned to the eight rooms. Pigs were allowed ad libitum access to feed and water. A light schedule was programmed to provide light in each room from 0700 h to 1800 h daily. During each feeding phase, excreted manure was sub-sampled and weighed in order to provide estimates of volume and nutrient content excreted from each treatment.

Exhaust air from each room was sampled in a consecutive manner; 15 min of sampling from a room followed by 15 min of sampling in the next room, and so on. Background measures of the incoming (outdoor) air was sampled in between each full round of room sampling. This provided a total of 10-11 daily observations in each room. All sampling was automated. Analyzers employed for sample analyses included a TEI Model 17C ammonia/NO_x chemiluminescence analyzer and a TEI Model 45C H_2S/SO_x pulsed fluorescence analyzer (Thermo Electron Corp., Franklin, MA). Airflow through each room was measured every 30 sec using differential pressure transducers calibrated for the pressure difference across orifice plates.

Results

Dietary treatment had no significant effect on ADG, ADFI or F:G ($P<0.05$). Pigs fed the LCP diet had greater intakes than pigs fed the C or ULCP diets during the grower phases but reduced intake during both finisher phases ($P=0.0287$).

A diet effect was observed for average daily ammonia concentrations ($P<0.0001$). Exhaust ammonia (NH_3) concentration where pigs were fed the LCP diet were 16% less than the C diet (3.86 vs. 4.57 ppm). Ammonia concentrations were reduced 25% (2.93 ppm) in the ULCP diets compared to the LCP diet and 36% compared to the C. Airflow-corrected daily average NH_3 emission rates were 26.8, 21.0, and 14.5 mg min^{-1} for the C, LCP, and ULCP diets, respectively, corresponding to a daily mass of NH_3 emitted per kg of animal liveweight, of 88.0, 68.9, and 46.0 mg kg^{-1}. Feeding phase effects were observed for NH_3 concentration, NH_3 emission rate, daily mass emitted and daily mass per unit of liveweight with increases from G1 to G2 followed by a decrease from F1 to F2. Feeding phase effects were observed for ammonia concentration, ammonia emission rate and the calculated variables (daily mass emitted and daily mass per unit of liveweight). Concentration and emission of ammonia generally increased through the grower phases then decreased during the finisher phases.

Similarly, feeding phase effects were observed for hydrogen sulfide concentration and daily emitted mass of hydrogen sulfide per unit of liveweight. Hydrogen sulfide concentration and emissions were not different as a result of dietary treatments ($P>0.05$). Though not statistically significant, this was unexpected because a sulfur-containing amino acid (methionine) was included in the LCP and ULCP diet, resulting in what was expected to be a reduced total dietary sulfur content. Diet analyses support no differences in total dietary sulfur content, leaving no apparent explanation for the observation, though non-significant. Feeding phase effects were observed for hydrogen sulfide concentration and daily emitted mass of hydrogen sulfide per unit of liveweight (calculated from concentration, airflow, and bodyweight measures), following the same trend as observed for ammonia, whereby there were increases from G1 to G2 followed by a decrease from F1 to F2. The interaction of diet and feeding phase was not significant for any variable tested.

Diet had no effect on wet or dry mass of manure produced, however; TKN and NH_3-N concentration decreased with decreasing diet CP (7.9, 6.7, 5.7% and 5.4, 4.4, and 3.5%, respectively for C, LCP and ULCP diets).

Discussion

Average daily ammonia concentrations reported in this study from swine fed the C diets are within the range of those reported by Zhu et al. (2000) who measured ammonia concentrations at the exhaust fan. Values reported here fall within the low end of values reported in a review by Arogo et al. (2003). Average daily hydrogen sulfide concentrations reported in this study are approximately 100-fold less than those reported by Zhu et al. (2000), who measured hydrogen sulfide concentrations at the exhaust fan.

Conclusion

The findings from this study demonstrate that reduced crude protein diets can be fed throughout the grow-finish phase with no detrimental effects on animal performance. The result from feeding such diets is substantial reduction in ammonia emissions. The facility demonstrated that it has the sensitivity to detect and quantify such differences, rendering this a feasible approach to establishing diet modification effects on mass of emissions from the farm.

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Detection and Enumeration of Airborne Microbial Pathogens Associated with Swine Farms in Eastern North Carolina

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Abstract

Animal agriculture has evolved so that large concentrations of animals are housed on relatively small parcels of commercial properties. These large concentrations of animals may have adverse impacts on environmental quality, including air quality, in close proximity to sites. Considerable work has focused on air quality associated with particulate, ammonia, and odor (odorant) emissions associated with farms. However, until recently, little research has addressed airborne microbial pathogens on farms. Of the current research, most has focused on air quality in the animal housing units. The focus of this research was to enumerate outdoor airborne pathogens and indicator microbes on farm properties, including at the upper and lower property boundaries. The objectives were to determine airborne microbial impacts both in close proximity to the waste management systems on the farms and at other locations, including at farm boundaries. Air samples were collected for microbial indicator and pathogens using standard all-glass impingers (AGI-30) and open-face filter cassettes for bacterial endotoxin analyses. Microbial concentrations of a suite of fecal and general indicator organisms, representative of bacterial, viral, and parasitic human pathogens, as well as the frank bacterial pathogen, *Salmonella*, were measured. The indicator organisms included: fecal coliform bacteria, *E. coli*, spores of *Cl. perfringens*, and coliphage as fecal indicators and total aerobic bacteria and total fungi as general air quality indicators. Bacterial endotoxins, generally associated with barn dust, also were measured throughout the farm properties. Microbial levels ranged from below detection limits to 1.7×10^6 cfu/m^3. Fecal indicator organisms were readily detected at most of the farm sites; however, *Salmonella* were not detected in air samples at any of the sites during the course of this study. Bacterial endotoxins, shown to have adverse health impacts to workers on the farms, were detected close to the barns, as well as at lower property boundaries at levels higher than those in upwind boundary samples for some of the study sites. Microbial concentrations were generally highest associated with barns that utilized tunnel ventilation, as compared to those that are naturally ventilated, and were generally higher at the lower property boundaries when compared to the upper property boundaries. These results suggest that there are airborne microbial impacts associated with swine farms in North Carolina and that these impacts may extend off of the property boundaries. These results have direct implications for future regulations concerning airborne microbial pathogens associated with confined animal feeding operations. The extent to which these increased airborne microbial levels pose actual human health risks is unknown and deserves further investigation.

Introduction

In the United States, confined animal feeding operations (CAFOs) have become the standard for large-scale animal production to supply the demands of a growing population. Because of the scale at which animals are produced, there are large quantities of fecal waste that are traditionally recycled for their nutrient value to crop agriculture not used for human consumption(Humenik, Rice et al. 2004). Typically, the system utilizes anaerobic treatment in lagoons followed by land application for manure that is handled as slurry, or direct land application following removal from the housing units for wastes that are traditionally handled dry. Because of the magnitude of fecal wastes produced by CAFOs, it is important to consider the

environmental impacts that may be associated with these farming practices. These impacts may include fecal contamination of ground and surface waters in close proximity to the farm sites, airborne fecal contamination, and off-farm contamination associated with vectors, such as rodents and houseflies.

To date, there have been few environmental studies with the goal of determining airborne fecal contamination on agricultural sites outside of the housing units and at property boundaries. Several studies have focused on air quality inside the housing facilities, concluding that there is airborne microbial contamination within the barns that can lead to animal and human exposures(Cole, Hill et al. 1999; Bilic, Habrun et al. 2000; Chang, Chung et al. 2001; Allen, Hinton et al. 2003; Richardson, Mitchell et al. 2003). Another concern for airborne microbial exposures associated with CAFOs is antimicrobial resistance traits carried by bacteria, which may lead to the ineffectiveness of certain drugs for treating animal or human infectious diseases(2003; Gibbs, Green et al. 2004; Chapin, Rule et al. 2005). Finally, studies have shown that bacterial endotoxins, associated with dust from agricultural barns, can have adverse health effects for workers on farms, as well as for more susceptible populations living in close proximity to the farm properties(Clapp, Becker et al. 1994; Schwartz, Thorne et al. 1995; Thorne, Reynolds et al. 1997; Chrischilles, Ahrens et al. 2004; Kim, Ko et al. 2005; Merchant, Naleway et al. 2005).

Concerns about human and animal health risks from animal fecal wastes released by CAFOs have led to the development and evaluation of alternative manure treatment technologies for better management of animal wastes and to better control environmental impacts(Humenik, Rice et al. 2004). Airborne releases of microorganisms from CAFOs may be important pathways for pathogen movement off farms and has been poorly studied for conventional and advanced animal waste management technologies. The objective of these trials was to investigate and quantify the release of airborne microorganisms and endotoxins associated with swine CAFOs in North Carolina on the farm properties and at the property boundaries.

Methods and Materials

During the course of this study, air samples were collected at 17 different commercial swine farms throughout Eastern North Carolina. Waste management systems on the farms consisted of both conventional anaerobic lagoon-sprayfield systems as well as those that utilize advanced biological treatment and other alternative processes. For all of the sites, air samples were collected at the upper and lower farm boundaries, as well as at locations on the properties where airborne releases were expected, such as at the housing units, at locations where open air handling of fecal wastes occurred, and in areas on the farms where there was land application of treated waste residuals.

Two different types of air samples were collected to enumerate a suite of microbial indicator and pathogenic organisms. All-glass impingers (AGI-30) containing 1% peptone-water with 0.01% Tween 80 and 0.005% antifoam A were used in duplicate to collect air samples for the fecal indicator organisms: fecal coliform bacteria, *E. coli*, spores of *Cl. perfringens*, total coliphage, and the frank pathogen, *Salmonella*. Samples from the AGI-30s were also assayed for total aerobic bacteria and total fungi as general microbial indicators of air quality. The AGI-30s were operated for 30-minute intervals at a flow rate of 12.5 L/minute. In addition to the AGI-30s that were used to collect air samples for detection of airborne microbial and fecal indicator organisms, SKC personal samplers were used, with open-faced filter cassettes, to collect bacterial endotoxins at each of the sites on the farms where the AGI-30 were utilized(Thorne, Reynolds et al. 1997). These samplers were operated for 4-hour intervals at a flow rate of 4 L/minute (equivalent to roughly a cubic meter of air).

Total bacteria and fungi were analyzed using a spread-plate technique on R2A agar (bacteria) or MEA agar (fungi) respectively and were cultured aerobically at room temperature for 5 to 7 days. Fecal coliform and *E. coli* were enumerated using a most-probable number (MPN) format and a defined substrate medium (IDEXX Colilert, Portland, ME), with incubation at 37°C for 4 hours, followed by subsequent incubation at 44.5°C for 18 hours. *E. coli* from this system was further cultured on EC-MUG agar at 37°C overnight and then colonies were isolated for further characterization (BBL Enterotube II, Beckton Dickson, Sparks, MD) and antibiotic resistance testing by a micro-dilution assay system (TREK Diagnostic Systems, Cleveland, OH)(Sullivan 2004). *Cl. perfringens* spores were detected and quantified by an MPN assay using an iron milk medium method (IMM) with incubation for 24 hours at 41°C, following an initial heat treatment at 65^{o}C for 20 minutes to inactivate the vegetative bacteria in the sample(St. John, Matches et al. 1982). Total coliphage were detected and quantified by two-step enrichment-spot plate lysis methods using *E.coli*

C3000 host bacteria (modified from EPA method 1601)(USEPA 2001). *Salmonella* was detected and quantified by an MPN method with pre-enrichment in buffered peptone water, enrichment in Rappaport-Vassiliadis broth, followed by streaking on *Salmonella-Shigella* agar for distinctive colonies. Colonies were isolated, biochemically confirmed, and subjected to antibiotic resistance testing using a micro-dilution method, as previously described(Vassiliadis 1983). Bacterial endotoxins were assayed by a *Limulus* Amebocyte Lysate assay in Peter Thorne's laboratory, University of Iowa(Thorne, Reynolds et al. 1997).

Environmental conditions were measured at the sites on farms where air samples were collected at the time of sample collection. The environmental parameters that were measured included: temperature, relative humidity, wind velocity, and solar irradiation (1.5 m height from the ground during the microbial air sampling).

Results

Results for mean bacteria and fungi concentrations are summarized in Figure 1. Bacteria concentrations ranged from 161 to 296,999 cfu/m^3. Fungi concentrations ranged from 118 to 5057 cfu/m^3. Farm 2 had the highest mean bacteria concentrations and Farm 16 had the lowest mean bacteria concentrations. Mean bacteria concentrations were more variable than mean fungi concentrations among the farm sites tested. Farm 3 had the highest mean fungi concentrations and Farm 10 had the lowest mean fungi concentrations.

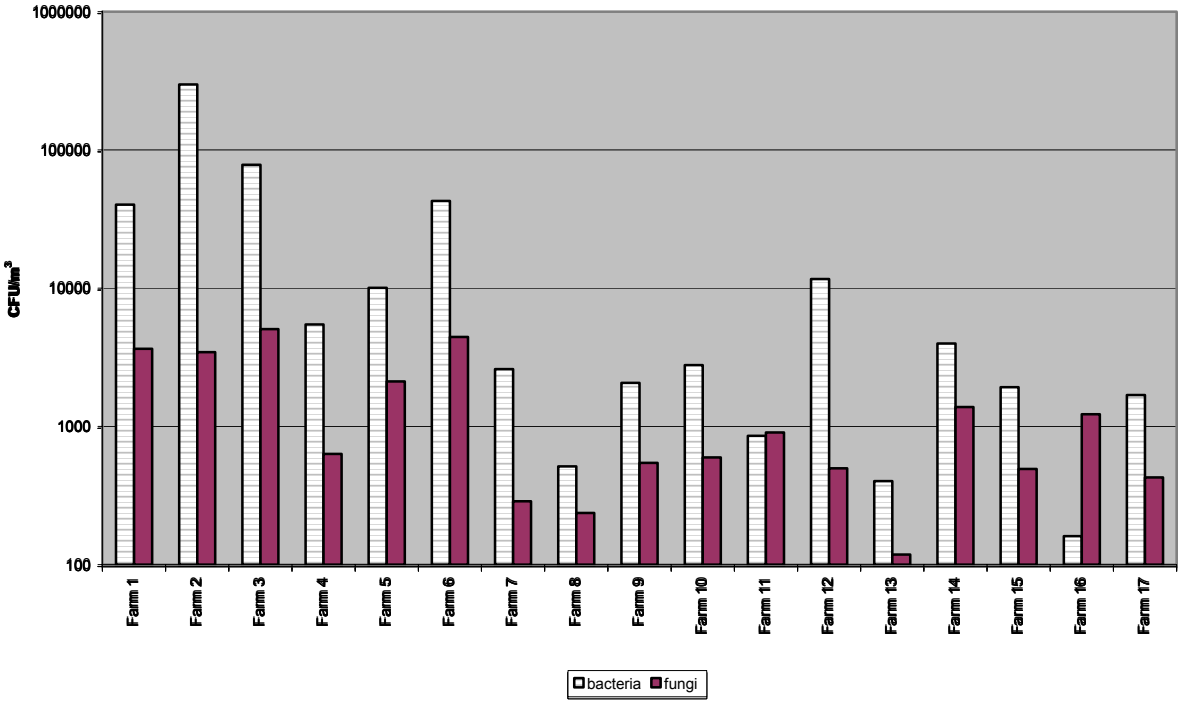

Figure 1. Mean Bacteria and Fungi Concentrations (CFU/m^3) in Air Collected on Farm Sites

Air samples were collected at multiple sites on each of the farms. The sites included, at minimum, the upper and lower boundary, and swine housing units (barns). Bacteria, fungi, and endotoxin concentrations for each of the sites at Farm 12 are shown in Figure 2. These results were typical of results for other farms in the study. Air samples were also collected during spray irrigation/land application events at farm sites when possible. Figure 2 shows bacteria, fungi, and endotoxin concentrations at the upper and lower boundary, at the barns, and upwind and downwind of the land application of treated liquid waste residuals.

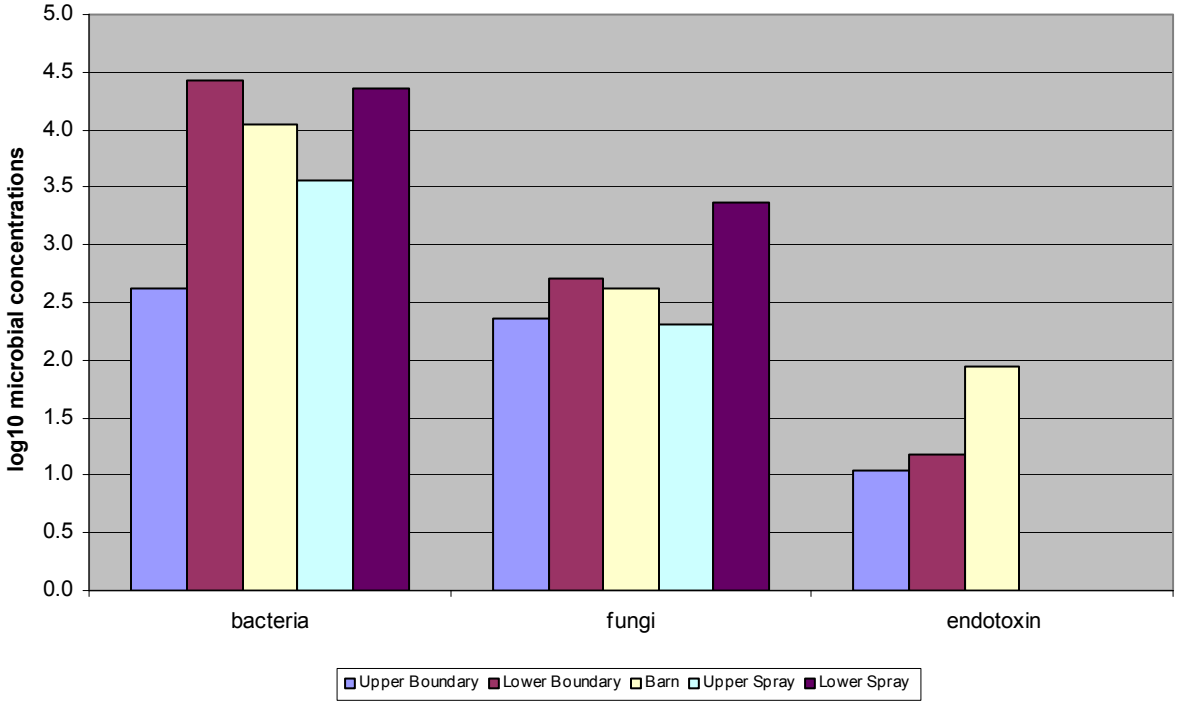

Figure 2. Bacteria, Fungi, and Endotoxin Concentrations at Various Locations on Farm 12

It is also important to take into account the concentrations of fecally associated microbes when measuring air quality on farm sites. Table 1 summarizes the number and percentage of samples that were positive for fecal microbes (fecal coliform bacteria, *E. coli*, spores of *Cl. perfringens*, and total coliphage) for each of the farms and the geometric mean and median concentrations where positive samples were detected. Fecal microbe occurrence ranged from below detection limits to an occurrence rate of 24% (at Farm 6). Geometric mean fecal microbe concentrations ranged from below detection limits to 154 cfu or pfu/m^3. Those farms with tunnel ventilation tended to have the higher percent positive rates and higher geometric mean and median concentrations of fecal microbes (data not shown).

Table 1. Percentage of Air Samples Positive and Geometric Mean and Median Concentrations of Samples Positive for Fecal Microbes

Location	Number Positive/Total Number (%)	Geom. Mean (Median) Concentration (cfu or pfu/cm3)
Farm 1	38/416 (9%)	154 (69)
Farm 2	37/193 (19%)	87 (100)
Farm 3	19/175 (11%)	90.7 (46)
Farm 4	1/85 (1%)	110 (110)
Farm 5	0/200 (0%)	-
Farm 6	29/120 (24%)	88 (70)
Farm 7	2/160 (1%)	32 (33)
Farm 8	3/112 (3%)	29 (32)
Farm 9	6/96 (6%)	96 (48)
Farm 10	6/80 (8%)	37 (36)
Farm 11	1/72 (1%)	10 (10)
Farm 12	10/152 (7%)	43 (53)
Farm 13	5/200 (3%)	43 (31)
Farm 14	5/120 (4%)	24 (31)
Farm 15	19/120 (16%)	63 (31)
Farm 16	0/24 (0%)	-
Farm 17	0/72 (0%)	-

Summary

As shown in Figures 1 and 2, all of the farms had measurable concentrations of total bacteria and fungi. In order to account for upwind, off-farm airborne contamination, upper boundary measurements were made at each of the farm sites. Figure 2 shows bacteria, fungi, and endotoxin concentrations typical of many of the farm sites tested. Microbial concentrations were generally lowest at the upper boundary, higher at the barns, and highest at the lower boundary. Additionally, for land application events, microbial concentrations were generally higher downwind of the practice as compared to the upwind sample. Endotoxin concentrations were generally low at the upper boundary, higher at the lower boundary, and highest near the barns. This is consistent with reports in the literature, as bacterial endotoxins are generally associated with dust from the barns. For each of the farms, it was noted which utilized tunnel ventilation, as opposed to being naturally ventilated (data not shown). Airborne microbial concentrations were generally highest associated with those farms that utilize tunnel ventilation for their housing units. In addition to the detectable levels of bacteria and fungi, there were also generally detectable levels of fecal microbes on these farm sites. The levels of fecal microbes ranged from below detection limits to a geometric mean concentration of 154 cfu or pfu/m^3. These results demonstrate that there is general and fecally associated airborne contamination associated with the sampled farms, as documented by the generally higher microbial concentrations at the lower property boundaries compared to the upper property boundaries. Bacterial endotoxins, associated with adverse health effects in humans, were detected throughout the properties at many of the farm sites. It is unclear the extent to which airborne microbial contamination extends off the farm sites and how this contamination might effect human health, however, these points deserve further study in the future.

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Modeling and Regulating Ammonia Emissions

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Introduction

In most European countries, animal husbandry is the major source of ammonia emissions. The source of ammonia emission within animal husbandry is the animal manure that is exposed in the following situations:

- Animal housing,
- Manure storage,
- Fields to which manure has been applied,
- Fields receiving excreta deposited by grazing animals.

The instantaneous emission from these sources is dependent on the characteristics of the manure (particularly the concentration of ammonium and pH) and the effectiveness of the mechanisms in the air that transport ammonia within the atmosphere. Since these latter processes are also responsible for the dispersion of ammonia in the atmosphere, the emission and dispersion processes are correlated. The processes driving ammonia emissions therefore change over time-scales that vary from the seasonal, reflecting some agricultural practices, to the daily or hourly, reflecting other agricultural practices and changes in atmospheric processes.

Within the framework of the UN Convention on Long-Range Transboundary Pollution (CLTRP), the Co-operative Programme for Monitoring and Evaluation of the Long-range Transmission of Air pollutants in Europe (EMEP) is responsible for assessing the dispersion and deposition of nitrogenous compounds for Europe. The ammonia emissions used are generally reported by signatory countries as total annual emissions. The EMEP model imposes a temporal variation on the NH_3 emission totals derived from the results of GENEMIS project. This currently assumes that the seasonal variation is very similar across Europe, with a main maximum in early spring and a secondary maximum in October-November, despite the large variation in agricultural practices. Recognizing that this was a gross simplification of reality, a project was initiated under the auspices of the Nordic Council of Ministers to develop a model that would improve the temporal resolution of ammonia emission estimates. A second objective was the modeling of a range of ammonia abatement measures that either are or could be a part of the regulation of agricultural ammonia emissions.

The Model

The model is based on the principles previously described in Hutchings et al (1996) and Pinder et al (2004) when modeling ammonia emissions from dairy cattle farms. The two main principles are that the model should follow the fate of N excreted by livestock through the manure handling chain (housing, storage and field application) and should account for the major physical and chemical processes determining emission. The first principle enables simulation of the knock-on effect of changes in emission at the start of the chain on the emission from subsequent links. The second principle enables the model to describe the magnitude and temporal variation of emission from each source.

Ammonia is emitted whenever an aqueous solution of ammonium is exposed to an atmosphere in which the ammonia concentration is lower than in the air that is immediately above the surface of the solution. The formula underlying the instantaneous ammonia emission from the *i*th source (E_i) is:

$$E_i = \frac{A_i N_{A,i}}{H_i V_i r_i}$$

Where A_i is the area of manure exposed, $N_{A,i}$ is the amount of ammoniacal N present, H_i is a coefficient relating the concentration of ammonia at the surface of the solution to the concentration of ammonium, pH and temperature of the solution, V_i is the volume of manure and r_i is the resistance to ammonia transport from the manure to the free atmosphere. The sources of emission are animal housing (i=1), manure storage (i=2), field-applied manure (i=3) and manure deposited during grazing (i=4). The source of the ammoniacal N is the animal excretion and this flows through each of the emission sources, with progressive depletion by the gaseous losses. The area of the source is identified with the following; for animal housing, the floor and manure channels covered or containing excreta; for storage, the surface area of the store; for field-applied manure and manure deposited during grazing, the area of manure exposed. Calculation of the H_i coefficient requires knowledge of the temperature and pH of the manure. The temperatures are calculated for animal housing whereas air temperature is assumed for the other sources. The pH of manure in each source is assumed to be constant. The resistance to ammonia transport is calculated using a ventilation model for the animal housing and using standard micrometeorological principles for the other sources. The time steps used in the model are determined by the user but should normally be one hour or less.

The main changes relative to the previous models are the inclusion of modules to more realistically simulate animal housing with forced or free ventilation and the treatment of emissions from field-applied manure.

Animal house ventilation

The inside temperature and ventilation rate of animal housing with force ventilation are simulated using a simple energy model. This model assumes that the aim of management is to maintain the indoor temperature at a target value suitable for the animal species. The ventilation rate necessary to achieve this objective is calculated from the outside temperature and the sensible heat production of the animals. The sensible heat production is calculated from the total heat production, using the methods described in CIGR (2002). An empirical constant is then used to obtain the resistance to ammonia transport. The ventilation is assumed to be constrained between the lower and upper limits, for welfare and engineering reasons respectively. If the ventilation rate is constrained at the minimum level, it is assumed that supplementary heating is provided to maintain the indoor temperature at the target value. If ventilation is constrained at the maximum level, no mechanical cooling is assumed and the inside temperature is allowed to exceed the target level.

For freely ventilated housing, it is assumed as in Gyldenkærne *et al.* (2005) that farmers can partially control ventilation by opening or closing doors and air vents. At ambient air temperatures above freezing, this enables them to the inside temperature at a constant margin above ambient. The ventilation rate is then a constant that is estimated from. At ambient air temperatures below freezing, the model again follows Gyldenkærne *et al.* (2005) and assumes that the restriction of ventilation, combined with the ground heat flux and deposition of warm excreta, is sufficient to ensure that inside temperature never falls below freezing.

Field-applied manure

The fate of ammoniacal N in manure applied to the field is assumed to be determined by the competition between the processes of infiltration into the soil and volatilization into the atmosphere. The model treats the field-applied manure as if it were a leaky tank, in which the manure is maintained at a constant depth. Processes such as manure application or precipitation that increase the volume of field-applied manure increase the area of this tank whereas processes such as evaporation, infiltration or plowing that decrease the volume, decrease the area. Evaporation is made a function of solar radiation, temperature and wind speed, whilst infiltration is made dependent on the saturated conductivity of the soil and the dry matter

content of the manure. To account for the effect of freezing on the infiltration rate, the infiltration rate was linearly reduced from 100% of the value determined by the soil and slurry properties at +1°C to zero at -1°C.

Simulations

The model was used to simulate an area with 30 000 finishing pigs (average weight 70 kg) with a daily excretion of 41g N, of which 70% is ammoniacal N. In the animal housing, the assumptions were a floor area of $1m^2$ per pig, fully slatted flooring, a volume of washing + spilt drinking water of 1.5 litre per pig per day, a target temperature of 20°C, a minimum temperature of 15°C, minimum and maximum ventilation rates of about 5 and 57 litres per pig per hour respectively and the handling of the manure as slurry. The surface area of manure storage was calculated on the basis of the number of months storage required and a depth of 4 m. For field application, a soil with an infiltration capacity of 72 mm d^{-1} was assumed and the application technique was assumed to be broadcast spreading. The application rate was 30 m^3ha^{-1} and manure was applied between the hours of 08.00 and 16.00, without incorporation by plowing. The maximum daily application rate was set to one fortieth of the maximum capacity of the manure storage. This is intended to reflect the variation in application date between farmers, rather than capacity limitations on any particular farm. The presence of well-developed arable crops was assumed to prevent field application of manure between mid spring and late summer. A synthetic climate was used, with daily mean air temperature and solar radiation varying sinusoidally over the year between -5 and +20°C, and 10 and 40 MJ d^{-1} respectively. Hourly air temperature was varied sinusoidally over the day, with a daily range of 10°C. A day length of 12 hours was used throughout, with solar radiation peaking at midday. The daily precipitation was 2 mm, evenly divided over the day. The wind speed was 2 ms^{-1} and neutral stability was assumed.

Three scenarios were investigated. In the first, no restriction was placed on the timing of the field application of manure, other than that due to the presence of a well-developed crop. This meant that the manure storage capacity could be limited to the equivalent of 3 months production. In the second, an additional restriction imposed by not making field applications during the winter. This is a common restriction in areas that are classified as nitrate sensitive under the EU Nitrates Directive. Manure storage capacity had then to be the equivalent to 6 months production. In the final scenario, a range of abatement methods were applied; the proportion of the floor area dirtied by animals was reduced by about 40%, a porous cover was placed on the manure storage and the field-applied manure was incorporated within 6 hours of application. Manure storage capacity was again equivalent to 6 months production.

The model has yet to be fully parameterised. The empirical constants used in modeling the housing and storage emissions were therefore adjusted to give annual emissions similar to the Danish standard values. The model was run for two years and the results from the second year are shown here.

Results

The results from the first scenario (Fig 1a) show a peak in animal house emissions in the summer, corresponding with the period of highest temperature and ventilation rate. The emissions from storage also peak in mid summer but are much lower. The emissions from field applications show a more complex pattern. During the late autumn, winter and early spring, the rate of application is limited by the rate of production of manure; there is little storage. There is a shallow peak in the mid winter that corresponds with the period when the soil is frozen and infiltration is reduced, giving a longer period during which the ammoniacal N in the manure can volatilize. Emissions fall in early spring as the frequency of sub-zero temperatures decrease but rise again, as the rise in temperature increases the volatilization rate. There is a peak in the early autumn, ending when the manure accumulated during the late spring and summer has been applied. Emissions during this autumn peak gradually fall as the temperature falls. The annual emissions are shown in Table 1.

In the second scenario (Fig 1b), the housing emissions are identical to the first scenario but the emissions from storage are higher, due to the larger surface area of the storage facility (Table 1). The restriction of manure application to the field to spring and autumn creates two large peaks of emission. Total field emissions are higher than in the first scenario because the concentration of manure application in the spring

means that the temperature and radiation during the application period are higher. The higher storage and field emissions mean that the annual emission in the second scenario is higher than in the first. In the third scenario (Fig 1c), the pattern of emission is similar to the second scenario but all emissions are reduced, particularly those from the field applications.

Discussion

The scenarios used here demonstrate the large temporal variation in ammonia emissions from agriculture. This variation is caused partly by the weather but mainly by the farmers' responding to the weather, the cycle of crop production, the seasonal demand on labor and the regulatory environment. It is not yet clear how detailed a representation of these variations is necessary. Work by Sjoth *et al* (2004) suggests that a static estimate of farming activities, as can be obtained from a survey of management practices, may be adequate when comparing model results and measurements. However, if the objective is to predict past or future emissions, assess the likely impact of abatement measures or extrapolate research results to estimate emission factors, functional relationships relating farm management to more readily available data need to be developed.

In a lightly regulated environment, the short-term dynamics of farm management are likely to be driven predominately by the interplay of weather, livestock, crop and soil. At the EU scale, the Farm Structural Survey provides a good source of data on the geographic distribution of different farm types, the livestock species kept and the land holding. More detailed data may be available at the scale of the member state, but the varied nature and format of these data make them difficult to use. Soil type and land use data, including detailed cropping data are also available at the EU scale. Unfortunately, data that would enable livestock species to be distributed between combinations of type of animal housing, manure storage and field application method are either not collected or are not readily accessible.

Farming regulation can affect the management of manure both directly and indirectly. An example of an indirect effect is if farmers change their crop rotation, and therefore manure management, to comply with an obligation to maintain green cover over the winter, to reduce nitrate leaching. Regulations affecting farm structure and management can be enacted at the EU, member state or regional levels. Although much important environmental legislation is now enacted at the EU level, it has to be incorporated into law at the level of the member state. This leads to variations in interpretation, supplementation and enforcement. There is a need to collate and maintain information on the regulation of farming at the member state level. This would enable the development of heuristic rules that could be used for pan-European modeling.

Atmospheric dispersion modelers are taking full advantage of the technical developments in computing to increase the resolution of their models in both time and space. The processes driving agricultural ammonia emissions are largely understood, so models of emission can be constructed that will produce estimates with the time resolution required. However, the improvement in predictive performance will be limited unless such developments are matched by improvement in the modeling of manure management and in the acquisition of data to drive them.

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Table 1. Ammonia emissions by source

Scenario	Housing	Storage	Field	Total
	% of TAN production			
1	18	4	14	37
2	18	9	19	46
3	12	3	3	18

TAN production =315 Mg year^{-1}

Fig 1. Daily ammonia emissions from a. scenario 1, b. scenario 2 and c. scenario 3.

Fig 1a

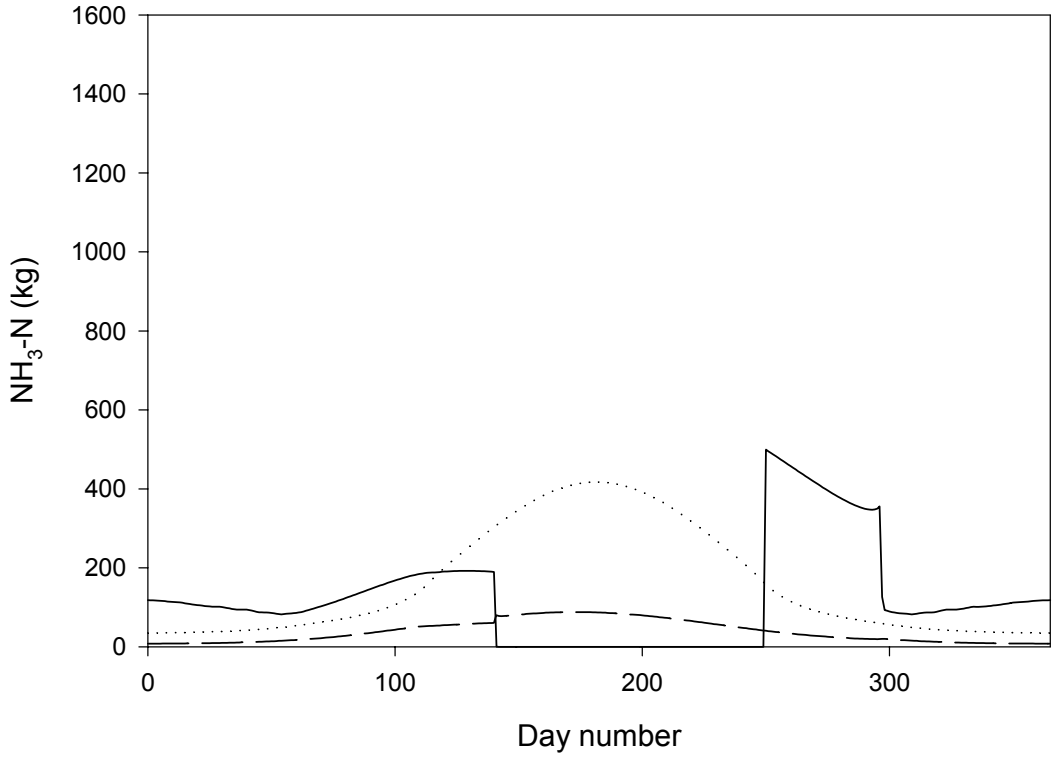


Fig 1b

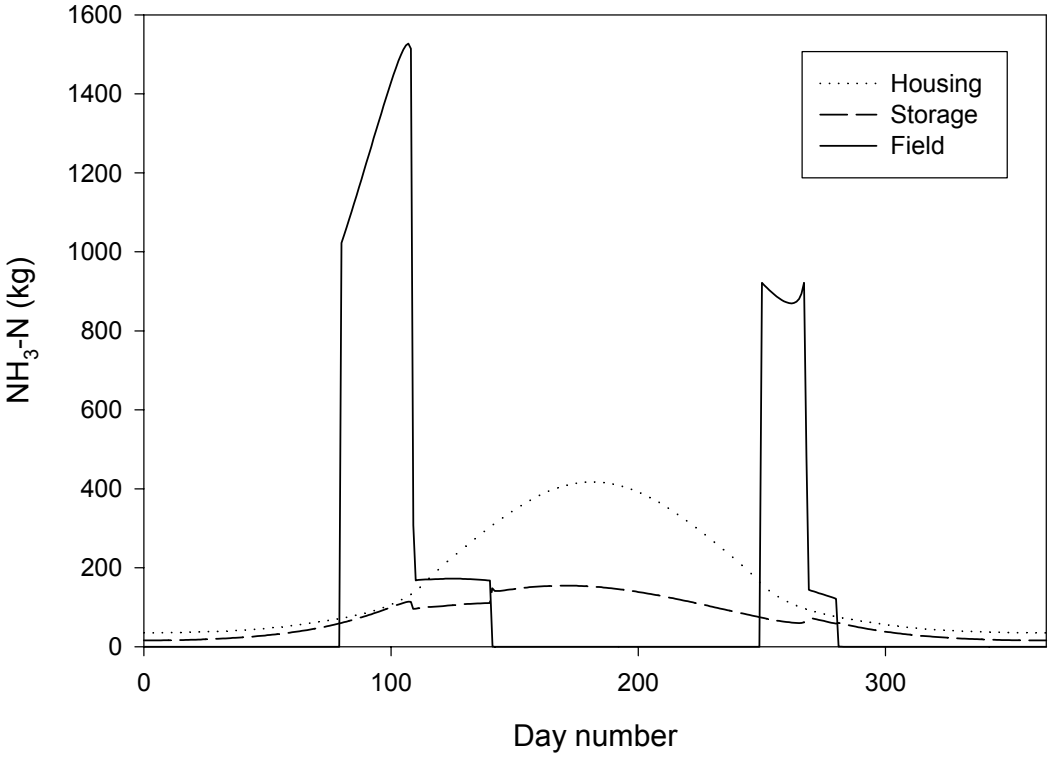
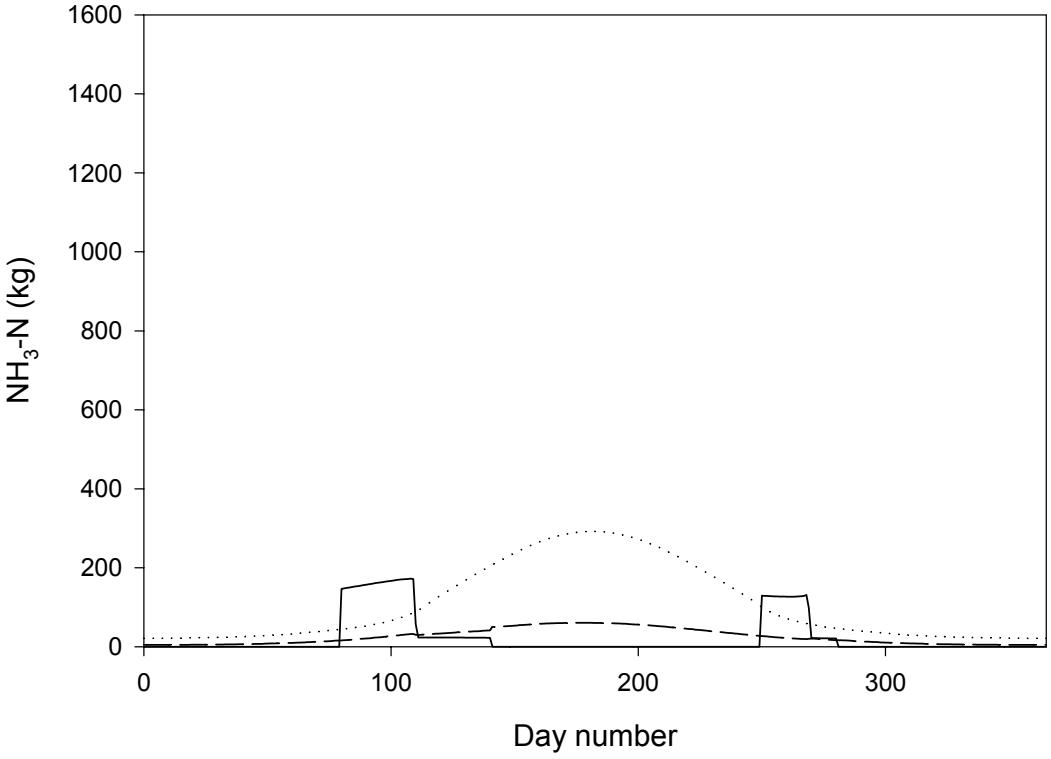

1600
1400
1200
1000
800
600
400
200
0
NH_3-N (kg)
0
100
200
300
Day number
Housing
Storage
Field

Fig 1c


1600
1400
1200
1000
800
600
400
200
0
NH_3-N (kg)
0
100
200
300
Day number