AIR QUALITY AND EMISSIONS FROM LIVESTOCK AND POULTRY PRODUCTION/WASTE MANAGEMENT SYSTEMS

Corresponding author: José R. Bicudo

Co-authors: David R. Schmidt, Susan W. Gay, Richard S. Gates, Larry D. Jacobson, and Steven J. Hoff

Reviewers: Dr. Dennis Schulte, Dept. of Biological Systems Engineering, University of Nebraska - Lincoln; Dr. Philip Westerman, Dept. of Biological and Agricultural Engineering, North Carolina State University; Dr. Jeffery Lorimor, Iowa State University; and Dr. James Lindley, Curry-Wille & Associates, formerly at Dept. of Agricultural and Biosystems Engineering, North Dakota State University.

Prepared as a White Paper for:

National Center for Manure and Animal Waste Management,
North Carolina State University, Raleigh, N.C.

August 12, 2002

AIR QUALITY AND EMISSIONS FROM LIVESTOCK AND POULTRY PRODUCTION/WASTE MANAGEMENT SYSTEMS

José R. Bicudo[1], David R. Schmidt[2], Susan W. Gay[3], Richard S. Gates[4], Larry D. Jacobson[5], and Steven J. Hoff[6]

TABLE OF CONTENTS

[1] Assistant Professor, Dept. of Biosystems and Agricultural Engineering, University of Kentucky, Lexington, KY;
[2] Extension Engineer, Dept. of Biosystems and Agricultural Engineering, University of Minnesota, St. Paul, MN;
[3] Assistant Professor, Dept. of Biological Systems Engineering, Virginia Tech, Blacksburg, VA;
[4] Professor and Chair, Dept. of Biosystems and Agricultural Engineering, University of Kentucky, Lexington, KY;
[5] Professor, Dept. of Biosystems and Agricultural Engineering, University of Minnesota, St. Paul, MN;
[6] Associate Professor, Dept. of Agricultural and Biosystems Engineering, Iowa State University, IA;
Corresponding author: José R. Bicudo, 128 C.E. Barnhart Building, Lexington, KY 40546, e-mail: jbicudo@bae.uky.edu.

ACKNOWLEDGEMENT

The authors would like to extend their appreciation to the USDA – Cooperative States Research, Economics, and Extension Service (CSREES) for providing funding to the 14-state National Center for Manure and Animal Waste Management who commissioned this white paper. We also would like to thank the reviewers of this paper: Dr. Dennis Schulte, Dept. of Biological Systems Engineering, University of Nebraska - Lincoln; Dr. Philip Westerman, Dept. of Biological and Agricultural Engineering, North Carolina State University; Dr. Jeffery Lorimor, Iowa State University; and Dr. James Lindley, Curry-Wille & Associates, formerly at Dept. of Agricultural and Biosystems Engineering, North Dakota State University.

AIR QUALITY AND EMISSIONS FROM LIVESTOCK AND POULTRY PRODUCTION/WASTE MANAGEMENT SYSTEMS

Executive Summary

The objective of this paper is to summarize the available literature on the concentrations and emissions of odor, ammonia, nitrous oxide, hydrogen sulfide, methane, non-methane volatile organic carbon, dust, and microbial and endotoxin aerosols from livestock and poultry buildings and manure management systems (storage and treatment units).

Animal production operations are a source of numerous airborne contaminants including gases, odor, dust, and microorganisms. Gases and odors are generated from livestock and poultry manure decomposition (i) shortly after it is produced, (ii) during storage and treatment, and (iii) during land application. Particulate matter and dust are primarily composed of feed and animal matter including hair, feathers, and feces. Microorganisms that populate the gastro-intestinal systems of animals are present in freshly excreted manure. Other types of microorganisms colonize the manure during the storage and treatment processes. The generation rates of odor, manure gases, microorganisms, particulates, and other constituents vary with weather, time, species, housing, manure handling system, feed type, and management system. Therefore, predicting the concentrations and emissions of these constituents is extremely difficult.

Numerous control strategies are being investigated to reduce the generation of airborne materials. However, airborne contaminants will continue to be generated from livestock and poultry operations even when best management systems and/or mitigation techniques are employed.

Livestock and poultry buildings may contain concentrations of contaminants that negatively affect human and animal health. Most of these health concerns are associated with chronic or long-term exposure to gases, dust, or microorganisms. However, acute or short-term exposures to high concentrations of certain constituents can also have a negative effect on both human and animal health. For example, the agitation and pumping of liquid manure inside a livestock building can generate concentrations of hydrogen sulfide that are lethal to humans and animals.

Once airborne contaminants are generated they can be emitted from the sources (building, manure storage, manure treatment unit, or cropland) through ventilation systems or by natural (weather) forces. The quantification of emissions or emission rates for gases, odor, dust, and microorganisms from both point sources (buildings) and area sources (beef and dairy cattle feedlot surfaces, manure storage and treatment units and manure applied on cropland) is being intensely researched in the U.S., in many European countries, Japan, and Australia. However, the accurate quantification of emissions is difficult since so many factors (time of year and day, temperature, humidity, wind speed, solar intensity and other weather conditions, ventilation rates, housing type, manure properties or characteristics, and animal species, stocking density, and age) are involved in the generation and dispersion of airborne materials. Furthermore, there

are no standardized methods for the collection, measurement and calculation of such constituents, resulting significant variability and extreme range in the published literature. In fact, emission rates of only a few airborne contaminants have been investigated. Ammonia, hydrogen sulfide, and methane emissions have been more thoroughly studied than other gases and compounds because of the negative environmental impacts or human health concerns associated with them. Unfortunately, there is very little emission data for other contaminants such as odor, nitrous oxide, non-methane volatile organic compounds, dust, and endotoxins. The long-term impacts of these constituents on the environment and on human health are also not known.

INTRODUCTION

Quantifying air emissions from animal production sites is a complex process. The complexity arises from the multitude and variety of individual sources responsible for these emissions, the extreme variability of these emissions, and the variety of components being emitted. Numerous gaseous compounds and living organisms are generated from livestock and poultry manure decomposition shortly after it is produced or during storage and treatment prior to use as a fertilizer on cropland.

Emission sources include barns, feedlot surfaces, manure storage and treatment units, silage piles, dead animal compost structures, and a variety of other smaller emissions sources. Each of these sources will have a different emission profile (i.e., different odor, gases, dusts, and microorganisms emitted) with rates that fluctuate throughout the day and throughout the year. Therefore, quantifying airborne emissions and their impact on the surrounding environment is extremely difficult. This paper compiles information on emission measurement and published data on odor, gas, and particulate emissions from two major sources of agricultural air emissions: animal housing and waste management systems. The research findings reported in this paper are organized by specific compound (odor, ammonia, nitrous oxide, hydrogen sulfide, methane, non-methane volatile organic compounds, dust, and endotoxins). Published emission values from animal housing and waste management systems are reported for each compound.

EMISSION MEASUREMENT

Definitions

Emission refers to the rate at which gases or particulates are being released into ambient air. It is also a mass flux per unit area and time from a particular surface. This is in contrast to concentration-only measurements. Emission rates are determined by multiplying the concentration of a component by the volumetric flow rate at which a component at a given concentration is being emitted. Surprisingly, while accurately measuring gas and odor concentrations within facilities is feasible, the determination of building or manure management system emissions is not straightforward. For example, it is not sufficient to count the number of fans, multiply by some average fan ventilation rate, and then multiply by the gas concentration. Likewise, it is not sufficient to estimate mass flux of a specific gas from the surface of litter on a floor, or manure within the facility, and then assume the building emission is constant regardless of the number of fans running; nor would it be appropriate to assume all similar facilities exhibit similar emissions. While these aforementioned ***crude estimates*** might be suitable for a rough "ball-park" estimate of building emission, at best they would be only useful for that point in time and they completely neglect the effect of daily husbandry activities (feeding, lights, etc) and disturbances to the thermal control systems (especially weather systems).

Odor and gas emission rates are often normalized 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. Emission expressed in terms of AU is often referred to as the emission factor. Area-specific emission, or flux rate, is determined by dividing the total emission rate by the emitting surface area.

The comparison of emissions from various studies is difficult because emissions are reported on numerous bases, including AU, animal live weight, animal place, area, or volume or weight of manure. Furthermore, the definitions of AU and animal place are not standardized. Therefore, conversion of emissions reported in one study to the units used in another study is not always possible; and when done, may lead to misleading interpretations. Also, data collection periods vary widely, ranging from a few hours to several days. In some cases units from original data sources were converted to grams of compound per AU and per day for comparison purposes, but this may not fully correspond to actual emission measurements. Conversion of daily to annual emission values is not encouraged as emission rates vary widely during the year depending on season, air temperature, humidity, etc.

Ventilation rates

A major impediment to determining emissions is the difficulty in knowing how much air is being exchanged. Mechanically ventilated facilities typically use a large number of fans and if the interior airspace is not well-mixed then gas concentration and hence emission rate may differ at each fan. Accurate measurement of airflow is difficult, and a number of factors commonly found in poultry and livestock facilities make this especially so, including dust accumulation on shutters and blades, loose belts, loss of building static pressure which results in variable ventilation effectiveness, and poor mixing, etc.

Basically, three methods can be used for determining building ventilation rates. One method, used for *in situ* ventilation measurement, has been developed by Simmons et al. (1998a) and has been used in poultry facilities (Simmons et al., 1998b). The device is a motorized anemometer array controlled and monitored with a computer. It uses five propeller-driven DC generators mounted on a horizontal bar or rack. The bar travels vertically and the instruments perform an equal area traverse. Volumetric flow determinations can be made in either vertical direction (i.e. going up or down). Following the traverse, the total fan output is calculated as a function of the area of the opening of the anemometer array. Its accuracy has been shown to be within 1% when used with 122 cm diameter fans. The second method uses heat production data and its relation to animal carbon dioxide (CO_2) production (van Ouwerkerk and Pedersen, 1994, Phillips et al., 1998). This latter quantity is measured and the building ventilation rate is obtained by inverse solution of a building CO_2 balance. In addition to these two techniques, measurement of building static pressure may be used if fan manufacturer's performance data are available and if the fans are in a condition similar to the standard test fans used in the performance tests.

European studies on gas emissions from livestock and poultry facilities (e.g., Groot Koerkamp et al., 1998a), often estimate building ventilation rates derived from the relationship between metabolic heat production and the CO_2 production of the animals and manure (if stored in a deep pit, underneath the animals). The validity of this method is based on two factors: a) valid heat production values for the animals, and b) CO_2 production is solely from respiration of the animals. The use of certain literature heat production data, mostly dating 20 to 50 years, has been questioned because of the drastic advancement in animal genetics and nutrition. Moreover, depending upon the manure handling systems, the measured CO_2 production can contain considerable contribution by microbial activities of the manure (e.g., manure storage in a high-

rise building or deep-pit system). Therefore, building ventilation rates derived with the latest heat production data from intensive laboratory measurements should be more reflective of the modern genetics, nutrition, and manure management practices (Xin et al., 2001). Although this technique is less accurate than ventilation flow rate measurement, it has the advantage of being applicable in principle to both mechanically and naturally ventilated buildings (Phillips et al., 1998).

Flux chamber

The development of a method to determine emissions from area sources has remained elusive for many years. Phillips et al. (2000) reviewed four approaches for determining ammonia emissions from livestock buildings and manure storages; a) conducting a mass balance, b) using a direct measurement technique for ammonia concentration and ventilation rate, c) using remote sensing via micrometeorology and ambient sampling, and d) using direct measurements of concentrations along with a tracer gas to determine airflow rates. The first three approaches are most applicable to area source measurements. The challenge with the first method is the biochemical transformations of the compounds of interest. Certainly a mass balance can be done on nitrogen in manure storages, but obtaining representative samples is not an easy task and determining the gaseous form of the emitted nitrogen requires extensive knowledge of the biochemical reactions in the storage.

Several researchers have attempted to use direct measurement techniques. These techniques involve covering some portion of the emitting surfaces, ventilating this covered area at some known rate, and measuring the concentration of the gas being studied in the exhaust air. The design of these enclosures varies significantly in the amount of area covered, the geometry of the enclosure, the ventilation rate of the enclosure, the design of the ventilating system, and the handling of air coming into the enclosure.

One type of flux chamber and design criteria was described in detail by Eklund (1992). These flux chambers are typically dome shaped chambers that cover 0.13 m^2 of emitting surface with a volume of 0.030 m^3. It is assumed that the air inside the chamber is well mixed and there is no excess/deficit pressure inside the chamber. "Zero" concentration air is pumped into the chamber at a flow rate of approximately 0.005 $m^3 min^{-1}$ giving six air exchanges per minute. The change in concentration within the chamber determines the emission rate. Ecklund (1992) sites several key operational parameters for the flux chamber but the most critical is the sweep airflow rate, with typically increasing sweep airflow rates resulting in increasing emission rates. However, studies were also cited in which emission rate decreased as airflow rate increased. Thus it was stated "air flow rate affects the flux being measured and the optimal flow rate depends on the design and operating factors of the specific flux chamber used, as well as the strength of the emission source." Arogo et al. (2002) pointed out the fact that there is a strong dependence of gas volatilization on air and source temperature, and on airflow, all of which can be changed due to the presence of the chamber. Potential limitations, such as microclimate modification and negative feedback between accumulated gases and surface emission rates restrict the sampling duration. Care is required when extrapolating results from chamber measurements, since the chamber environment is not representative of actual field conditions. Despite the limitations of the method, several researchers have used flux chambers to measure emission from area sources

(Gholson et. al., 1991; Ecklund and Lacosse, 1998; Reinhart and Cooper 1992; Jeppsson, 1999, Aneja et al., 2000, etc.).

Ferguson et al. (1998a,b) and Gates et al. (2000) used an equilibrium chamber to compare the effect of dietary manipulation on ammonia volatilization. This technique does not directly measure emission; rather the concentration of a gas in the headspace above a sealed sampling volume that covers an area is measured. A tight container is placed over the emitting surface and continuous concentration readings are recorded until quasi steady-state conditions are achieved. This concentration is the driving potential for mass flux from the emitting surface. To determine emission rates, the surface mass transfer resistance is necessary. Because of the dependency between surface resistance, surface velocity, and mass transfer resistance at the solid-gas or liquid-gas interface, the method is not directly useful for emission estimates without additional calibration.

Wind tunnel

Small portable wind tunnels have also been used to measure emissions from area sources. As described by Smith and Watts (1994a), a wind tunnel is an enclosure with an open bottom that is placed over (on) the emitting source while ambient or filtered air is blown or drawn through the tunnel to mix with and transport the emissions away from the emitting surface. Concentrations in the exhaust air stream and airflow in the tunnel are used to estimate the flux rate (typically mass/time-area). The volume flow rate is the product of the bulk wind speed of the air passing through the tunnel and the cross sectional area of the tunnel. In actuality, these small wind tunnels may not meet all the criteria for wind tunnels since the length of the tunnels does not typically allow for fully developed airflow in the tunnel and thus may be more accurately referred to as flux chambers. However the airflow rate in the wind tunnel is typically much higher than in a flux chamber and the wind direction in the tunnel is more defined than in a flux chamber.

A number of researchers have described the methodology of using wind tunnels on a variety of emitting sources (Lockyer, 1984; Ryden and Lockyer, 1985; Meisinger et al., 2001; Schmidt et al, 1999; Schmidt and Bicudo, 2002; Witherspoon et al., 2002; Wang et. al., 2001; Jiang et al., 1995; Loubet et al., 1999a; Loubet et al., 1999b; Pain et al., 1988; Smith et al. 1994a; Smith et al., 1994b and Watts et al., 1994). Unfortunately, little effort has been made to standardize either wind tunnel design or the measurement protocol. Basic mass transfer principles from surfaces suggest emissions are dependent on surface velocity. Transfer rates have been measured that doubled when air velocity was increased from 1 to 5 m s^{-1} (Phillips et al., 2000). Other factors such as tunnel geometry and materials used to construct the tunnel are also expected to influence measurements (Smith and Watts, 1994a).

Benefits of the wind tunnel over the traditional flux ^{chamber} are the larger surface area covered and the air exchange rates or air speed in the tunnel being more similar to ambient conditions. One of the key parameters that is not usually evaluated in wind tunnel research is the composition of the inlet air. In most cases, the inlet air was ambient air with provisions made to take samples on the upwind side of the odor source. This limitation brings into question the measurement of actual flux rates. This problem is not seen in the traditional flux chamber because of the use of “zero”

air from compressed cylinders. On the other hand, the use of "zero" air may also result in higher measured flux rates due to an increased concentration gradient on the emitting source. The much higher exchange rates in wind tunnels, as required in high emitting sources, make this source of clean air impractical. One possibility to measure actual flux rates is to measure concentrations in both the inlet and outlet air. Another option that is becoming popular is to incorporate an efficient air filtration system into the wind tunnel.

Micrometeorological method

This is basically a mass balance method used to calculate spatial average emissions. Micrometeorological techniques integrate fluxes over large areas, do not disturb the sample area or its microclimate, and allow studies of the changes in fluxes with changing atmospheric and surface conditions (Fowler and Duyzer, 1989). Harper et al. (2000), Thompson and Meisinger (2001), and Zhan et al. (2001) are some of the few researchers who have used this methodology to estimate emissions from lagoons and land application areas. The technique involves the simultaneous measurement of vertical profiles of wind speed and concentration at one or more points within the emitting area. Gas concentrations and wind speed measurements are usually taken at the center of the source with a circular shape to ensure that wind is always perpendicular to the source and that the fetch over the source is constant and equal to the radius of the circle. Measurement height is calculated by trajectory simulation models and based on system surface area and roughness length (Wilson et al., 1982). Gases released from the treated area flows past the center of the circle where it is sampled by drawing air through specific gas detection equipment (acid traps or annular denuders for ammonia - NH_3, chemical sensors for CO_2, laser spectroscopy for methane - CH_4, tunable diode laser trace gas analyzer for nitrous oxide - N_2O, etc.) mounted at several heights. Background air samples are also collected at corresponding heights at the upwind edge of the circle. The emission rate is calculated from the product of the increase in concentration over background levels and wind speed profiles integrated over the height of the profiles (Smith and Watts, 1994).

For both livestock buildings and manure storages or lagoons a downwind flux frame can be used to intercept the plume of ammonia leaving the source. The background ammonia flux must also be determined. According to Phillips et al. (2000), this approach has serious drawbacks in practice: (i) setting up of an array of stationary masts in a position that pre-suppose a particular wind direction is not a straight forward operation; (ii) the prevailing wind direction may not manifest itself for days; and (iii) the fluxes available for measurement may be reduced by dilution due to the height of the flux frame in order to avoid the highly turbulent wake immediately downwind of a building, manure storage, or lagoon.

The micrometeorological method to determine fluxes is generally considered to be more accurate than wind tunnel methods because it minimizes changes of environmental conditions and allows farming equipment and practices to be fully used (Thompson and Meisinger, 2001). Harper (1988) discussed the errors associated with the micrometeorological method, and error attributed to this technique is ±15%. Thompson and Meisinger (2001) pointed out that this method requires relatively large field sites to meet wind-fetch, topographic, and cropping requirements, and it also requires considerable equipment, labor, and analytical support.

Fourier transform infrared spectrometry (FTIR)

This method utilizes a computed tomography algorithm using a smoothed basis function that converts the measured plane-integrated concentrations into a plume profile. Wind data are simultaneously integrated across the plume to yield the flux through a plane.

The FTIR spectroscopy method has been used to estimate emission from fugitive and area sources such as landfills and coal mines (Piccot et al., 1996, Kirchgessner et al., 1993). In these studies downwind Path-Integrated Concentration (PIC) data, wind measurements, and plume dispersion modeling are combined to estimate the total emission rate.

Natschke et al. (2001) have recently used this methodology to estimate NH_3 and CH_4 emissions from a covered swine anaerobic lagoon. The beam geometry, which consists of five beam paths, is positioned in a vertical plane downwind from the lagoon area source. Three beam paths scan the open path FTIR device to ground level reflectors. Two slanted beam paths scan the retroreflectors mounted on a tower at 5 and 9 m height. Establishing such a plane across the plume allows measurement of the flux through it. Each path is sampled for one minute per scan with the total sampling period of at least twenty minutes to minimize data variations and to allow the build up of an approximate Gaussian plume (Natschke et al., 2001).

The instrument does not give direct compound concentrations. The signal is a function of both the optical path and the weighted average concentration. For non-uniform concentration, the observed signal is the sum of individual path segments multiplied by the localized concentration. A smooth basis function method (SBFM) is applied to the beam data in conjunction with measured wind data, to estimate the total flux from the area source (Hashmonay et al., 2001). The SBMF method uses a bivariate Gaussian function. Using the integrated form of the function allows calculations to be performed directly from the PIC data.

ODOR

Livestock and poultry housing

Odor emissions from animal production sites are one of the most important factors to consider when determining setback distances from neighbors since the human nose can readily detect odors. Furthermore, odors are often perceived as indicators of airborne pollutants.

Livestock and poultry odors originate from four primary sources: animal buildings, feedlot surfaces, manure storage units, and land application of manure. Of these four sources, land application of manure is probably the biggest source of odor emissions and complaints. Although not typical, daily land application of manure is still practiced by some producers. Irrigation of manure is still also practiced throughout the United States, in spite of the significant emissions of odor and gases this practice generates. It should be noted that irrigation of anaerobic lagoon liquid generates fewer odors than irrigation of liquid manure, but odor intensity can be high when liquid from heavily loaded lagoons is irrigated as compared to lightly loaded lagoons. Unfortunately, very little scientific information is available on odor emission from manure

irrigation. In the Midwest, particularly in the corn belt area, land application typically occurs during specific periods of the year (usually in the fall, but spring application is also practiced) and known odor control management practices, such as injection of liquid manure into the soil, are available to minimize odor emissions. Therefore, emissions from land application are concentrated in short periods of time and may not be such a nuisance as compared to continuous and long duration emissions from other sources such as animal housing, feedlot surfaces, manure storage, and treatment units. This may help partially explain the fact that odor emission rate measurements have been and continue to be primarily measured from animal housing facilities and manure storage units.

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. The organic matter in these wastes is microbially transformed into non-odorous end products under aerobic conditions (Westerman and Zhang, 1997). However, in anaerobic environments, the decomposition of organic compounds results in the production of odorous volatile compounds that are metabolic intermediates or end products of microbial processes (Zhu, 2000). Many of these compounds are then carried by ventilation air, airborne dust, and other particles and dispersed into the atmosphere.

Odor must first be quantified to determine odor emission values. Air samples are diluted with a known amount of odor-free air. The dilutions are presented to a specially trained panel of test personnel using an olfactometer, which is an air dilution device. The odor detection threshold (ODT) is the number of dilutions with odor-free air required for an odor to be perceived by 50% of the panel members. One odor unit (OU) is defined as the amount of odorant at the panel ODT and is dimensionless. However, the ODT of a sample is often expressed as odor units per cubic meter (OU m^{-3}) for calculation convenience of odor emission (CEN, 1999). If this convention is followed, then odor emission rates (OU s^{-1}) from a livestock building or manure storage unit are the product of the ventilation airflow rate (m^3 s^{-1}) through the barn or over the storage and the odor concentration (OU m^{-3}) in the exhaust air.

Few researchers have attempted to quantify odor and gas emission rates from animal housing and results are widely variable. Table 1 lists odor flux rates measured from buildings for various animal species. This variation likely stems from the lack of standardized methods used to measure both odor and emissions. For example, air samples are often collected and stored in Tedlar™ bags until evaluation by dynamic olfactometry can be performed. However, Zhang et al. (2001) reported that these bags emitted significant levels of acetic acid and phenol, which are common odorants found in livestock and poultry manure. In addition, the Tedlar™ bag was found to have an absorptive selectivity for certain odorants such as indole and skatole. The white paper on odor mitigation for concentrated animal feeding operations (Sweeten et al., 2002) gives a detailed description and discussion of odor sampling and measurement.

Lim et al. (2002) evaluated odor emission and characteristics at two commercial swine nurseries during the spring. Five sampling visits were made to each nursery and nine or ten 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 five-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 1. Odor flux rates from animal housing

Species	Production unit	Location	Odor Flux Rate OU m^{-2} s^{-1}	Reference
Pigs	Nursery (deep pit)	Indiana	1.1-2.7	Lim et al. (2002)
	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.

Waste management systems

Odor emissions from waste treatment systems are more likely to occur when retention times are too short. Increased organic loading rates due to expanding animal numbers, slug loading, concentrated waste streams, and/or inadequate amounts of dilution water may also increase the potential for odor emissions from waste management systems. Odor emissions from manure storages and anaerobic lagoons tend to occur when the liquid surface is disturbed during windy conditions or during agitation and pumping prior to land application. Spring turnover, defined as the vigorous bacterial activity that occurs during spring due to incomplete metabolism during winter, also increases the potential for odor emissions from storages and lagoons.

Information on odor emission from anaerobic digestion systems, including anaerobic lagoons, is fairly limited. An anaerobic digestion system will produce minimum odors when acid-forming and methane-forming anaerobic bacteria are in balance. Provided adequate retention time and specific temperatures exist, a well-controlled anaerobic digestion process will degrade the vast majority of compounds that contribute to odors. For example, Powers et al. (1997) reported that

odor intensity from dairy manure decreased linearly with increased hydraulic retention time (HRT) in a set of laboratory experiments. Wilkie (2000) obtained a 94% decrease in flushed dairy manure odor after fixed-film anaerobic digestion at three-day hydraulic retention time (380 m^3 digester). Welsh et al. (1977) studied the effect of anaerobic digestion on swine manure odors. Their results indicated that anaerobic digestion was effective in reducing odors, but some negative quality in the odor remained after treatment.

Aerobic treatment, including composting, helps remove most of the organic compounds that give off odors in manure. Burton et al. (1998) quantified the effect of the duration of treatment on odor abatement. No odor regeneration was discerned over the first 28 days after anaerobic storage of pig slurry treated aerobically for 2.4 days. Several researchers have evaluated the benefit of aerobic treatment in reducing manure odor concentration and intensity (e.g. Sweeten et al., 1991, Bicudo et al. 1999, Westerman et al., 2000) but information related to actual emission rates is missing.

In fact, considerably fewer studies have measured odor emission from outdoor manure storage and treatment units or open feedlots as compared to animal housing. It should also be noted that most of the available data are published in terms of flux rather than emission rates. The odor flux rate data from manure storage, treatment and land application are summarized in Table 2.

Table 2. Odor flux rates from waste management systems

System	**Odor flux rate (OU m^{-2} s^{-1})**	**Reference**
Swine manure storage	2.5 to 55	Gay et al. (2002)
Swine manure storage	8 to 21	Bicudo et al. (2002)
Dairy manure storage	5.1 to 32	Gay et al. (2002)
Beef manure storage	7.2	Gay et al. (2002)
Swine manure anaerobic lagoons	5 to 30	Smith et al. (1999), McGahan et al. (2001)
Cattle feedlot	12.5 to 725	Watts et al. (1994)
Layer manure composting	42.5	Gay et al. (2002)
Land application	1.5 to 90	Pain and Misselbrook (1991)

Hobbs et al. (1999) conducted a laboratory study on odor emissions from swine manure that had been stored between 0 and 112 days. Approximately 200L of liquid manure were exposed to an enclosed atmosphere for up to four hours (Hobbs et al., 1999). The controlled environment provided an air temperature of 20 ^oC and an air circulation speed of 1 m s^{-1}. Slurry was maintained at 15 ^oC and stirred at a constant rate. Odor emissions peaked at 20,417 OU m s^{-1} after 71 days (Hobbs et al., 1999). In this case, authors reported odor concentration in terms of OU and not OU m^{-3}.

Gay et al. (2002) summarized odor flux rates from livestock and poultry manure storage units in more than 40 farms in Minnesota using a wind tunnel. Mean odor flux rates from swine manure storages varied from 2.5 to 55.1 OU m^{-2} s^{-1}. Dairy manure storages had mean odor flux rates from 5.1 to 32.2 OU m^{-2} s^{-1}. The mean odor flux from stored beef manure was 7.2 OU m^{-2} s^{-1}. Composted layer manure had an estimated odor flux rate of 42.5 OU m^{-2} s^{-1}.

Bicudo et al. (2002) measured odor flux rates from three swine manure storages in Minnesota during two consecutive years using a wind tunnel. Mean odor flux rate was about 14 OU m^{-2} s^{-1}.

The presence of a natural crust on the surface of manure storages had a significant effect on odor flux rates at the 5% level, especially in the first year: odor flux rates from naturally crusted storages were between 7 and 9 OU m^{-2} s^{-1}.

Smith et al. (1999) measured odor flux rates from swine anaerobic lagoons in Queensland, Australia, using a wind tunnel. Measured flux rates were standardized to the odor flux rate with a wind speed of 1 m/s measured 1 m above the pond surface. The "standard" odor flux rates used were averaged as 30 OU m^{-2} s^{-1} for primary lagoons, and 5 OU m^{-2} s^{-1} for secondary (facultative) lagoons.

McGahan et al. (2001) measured odor emissions from five Australian anaerobic lagoons containing swine waste. The "standard" odor flux rates from four lagoons were approximately 10 OU m^{-2} s^{-1}. One lagoon was heavily loaded and the odor flux rate was about 17 OU m^{-2} s^{-1} (McGahan et al., 2001). Odor measurements were made in accordance with a Draft Australian Standard (equivalent to CEN TC 364). If a conversion of 3 is used to convert these numbers to the old NVN 2820 Standard, the odor flux rates compares favorably with the previously suggested value of 30 OU m^{-2} s^{-1} obtained by Smith et al. (1999).

There is clearly a need for more odor emission data from waste management systems. A small number of anaerobic digesters have been recently installed in both swine and dairy farms in the U.S., but there is currently no or very limited information on odor emissions from such systems (e.g. Wilkie, 2000). The use of anaerobic lagoons for the storage and treatment of livestock wastes is widespread in many parts of the U.S., but no odor emission baseline has been established so far. The influence of purple sulfur bacteria in minimizing odor emissions from anaerobic lagoons is still to be determined (e.g. Gilley et al., 2000). Similarly, information on odor emission from stacked livestock and poultry manure and composting operations livestock are also missing.

Information on odor emissions from land application is limited to the studies conducted by British researchers. Most studies have been undertaken at the Institute of Grassland and Environmental research in the UK (e.g. Pain et al., 1990, Pain and Misselbrook, 1991). This work has included a variety of manure types and application methods. Reports from all studies indicated that odor emissions peaked soon after spreading the manure and then declined rapidly with time.

Pain et al. (1991) reported that typical odor flux rates from land application of cattle manure were from 1.5 to 90 OU m^{-2} s^{-1}. Odor emissions during the first hour of pig slurry application were 504 OU s^{-1}. Peak odor emissions during all Pain et al. (1991) experiments occurred within one hour after manure application. Six hours after spreading, odor emissions decayed exponentially to approximately 10% of the initial emissions.

Misselbrook et al. (1993) investigated the relationship between odor emissions and intensities of land applied swine manure. Odor intensity at or below an intensity of two (equivalent to faint odor in a scale that goes from zero to five) may be considered acceptable. Misselbrook et al. (1993) concluded that odor emissions from pig slurry should be less than or equal to 4.5 OU m^{-3} on average.

AMMONIA

Livestock and poultry housing

Ammonia is colorless, lighter than air, highly water-soluble, and has a sharp, pungent odor with detection threshold between 5 and 18 ppm. Gaseous NH_3 has a mean life of about 14 – 36 hours depending on weather. NH_3 is classified as a particulate precursor, i.e. in the vapor phase it will react with other compounds to form particulates. NH_3 and chemical combinations (NH_x) are important components responsible for acidification in addition to sulfur compounds (SO_x), nitrogen oxides, and volatile organic components (Groot Koerkamp, 1994).

Ammonia is deposited downwind of sources by both "dry" and "wet" methods, with dry deposition generally occurring locally. In fact, the amount of ammonia deposited locally is shown to be quite dependent on downwind land-cover with transport and deposition being quite variable across the landscape (Sutton et al., 1998). Other research has shown that local deposition is concentrated in the first 500 meters from the source (Fowler et al., 1998, Pitcairn et al., 1998, Nihlgard, 1985).

Ammonia may cause several ecological problems in the environment. First, excess inputs of nitrogen may lead to considerable changes in plant communities with the result that plants which prefer low nitrogen soils disappear and there is an increase in nitrogen indicator plants (Ellenberg, 1988). Second, acidification in soils with low buffer capacity may occur after nitrification of the added nitrogen. A falling pH leads to the dissolution of toxic soil constituents such as aluminum ions, and to the leaching of nutrients and aluminum into the groundwater (Van Breemen et al., 1982, Speirs and Frost, 1987, Roelofs et al., 1985, Speirs and Frost, 1987). Third, the natural capability of forest soil to take up methane (CH_4) is decreased by NH_3 deposition, thus increasing the concentration of CH_4 in the atmosphere (Steudler et al., 1989). Fourth, surface waters may be affected by eutrophication and acidification (Dillon and Molot, 1989). Finally, NH_3 depositions on buildings will promote bacterial growth, which contributes substantially to weathering and corrosion damage of the buildings (Spiek et al., 1990). The white paper on ammonia emissions from animal feeding operations (Arogo et al, 2002) gives a more detailed description of the environmental impacts of ammonia from animal production.

Ammonia release from animal sources is prevalent due to the inefficient conversion of feed nitrogen to animal product. Livestock and poultry are often fed surplus nitrogen with high protein feeds to ensure nutritional requirements are met. Nitrogen that is not metabolized into animal protein is excreted in the urine of swine and cattle and in the uric acid excreted by poultry. Further microbial action releases NH_3 to the atmosphere.

Ammonia levels of 5 to 10 ppm are typical in well-ventilated swine confinement buildings where slatted floors allow manure to fall into underground manure storage pits. Concentrations of NH_3 tend to be slightly higher (10 to 20 ppm) in buildings where manure is deposited on solid floors. NH_3 levels in animal housing can exceed 25 ppm when lower winter ventilation rates are used and can reach 40 ppm in poorly ventilated buildings (Groot Koerkamp et al., 1998b) or in the manure storage area of high rise layer houses (Wathes et al., 1997). Very high levels of NH_3

concentrations, such as 2,500 ppm may be fatal. The U.S. Occupational Safety and Health Administration (OSHA) indoor 8-h NH_3 exposure threshold is 25 ppm, which is similar to NH_3 threshold limits in many other countries (ACGIH, 1992).

A recent ammonia emission inventory from UK agriculture estimated emission as 197 kt NH_3-N $year^{-1}$ (Misselbrook et al., 2000, Pain et al., 1998). Emissions from livestock and poultry housing accounted for 7%, 12%, and 19% for pigs, poultry, and cattle, respectively.

Table 3 lists published ammonia emissions from livestock and poultry housing.

Table 3. Ammonia emission factors from livestock and poultry housing

Species	**Production unit**	**Notes**	**Emission Factor g NH_3 AU^{-1} day^{-1}**	**Reference**
Pig	Finish	Partly slatted	42	Aarnink et al. (1995)
	Finish	Litter	34-90	Groot Koerkamp et al. (1998a)
	Finish	Litter	50-62	Groot Koerkamp et al. (1998a)
	Finish	Fully slatted	72	Hinz and Linke, (1998)
	Finish	Fully slatted	128	Demmers et al. (1999)
	Finish	Fully slatted – no pigs	5-8	Ni et al. (2000)
	Finish	Slurry removed weekly	30	Osada et al. (1998)
	Finish	Fully-slatted	32	Osada et al. (1998)
	Finish	Fully slatted	40-50	Ni et al. (2000)
	Finish	Fully slatted (warm weather)	68-274	Ni et al. (2000)
	Finish	Fully slatted	10-80	Zhu et al. (2000a)
	Finish	Fully slatted	310	Zahn et al. (2001)
	Gestation	Litter	18-78	Groot Koerkamp et al. (1998a)
	Gestation	Slats	25-40	Groot Koerkamp et al. (1998a)
	Gestation	Fully slatted	2.2	Zhu et al. (2000a)
	Nursery	Slats	15.6-37.4	Groot Koerkamp et al. (1998a)
	Nursery	Fully slatted	23-160	Zhu et al. (2000a)
Poultry	Layer	Winter	190	Wathes et al. (1997)
	Layer	Summer	300	Wathes et al. (1997)
	Layer	Deep litter	177-261	Groot Koerkamp et al. (1998a)
	Layer	Battery	14-224	Groot Koerkamp et al. (1998a)
	Broiler	Winter and Summer	216	Wathes et al. (1997)
	Broiler	Litter	53-200	Groot Koerkamp et al. (1998a)
	Broiler	Litter	45	Demmers et al. (1999)
	Broiler	Litter	5.8-8.4	Zhu et al. (2000a)
Beef		Straw bedding	8.9-21.6	Groot Koerkamp et al. (1998a)
		Slats	8.4-16.6	Groot Koerkamp et al. (1998a)
		Straw bedding	19.4	Demmers et al. (1998)
		Feedlot	18.3-67.7	Hutchinson et al. (1982)
Dairy		Straw bedding	6.2-21.4	Groot Koerkamp et al. (1998a)
		Free stall	20.2-42.5	Groot Koerkamp et al. (1998a)
		Free stall with straw	31.7	Demmers et al. (1998)

Ammonia emissions from beef feedlots and dairy facilities appear to be less variable and lower than NH_3 emissions from swine and poultry housing. However, the limited number of data from beef and dairy operations may account for the low range in values.

Currently, there is wide disparity between the few published tabulations of both swine and poultry emission factors. Ammonia emission factors from swine housing units vary from 0.09 to 12.9 g NH_3 AU^{-1} hr^{-1}, where AU is an animal unit corresponding to 500 kg body mass. Numbers from pig finishing units appear to be higher than both gestation and nursery facilities. Measurements from poultry facilities indicate that ammonia emission factors vary 50-fold, from 0.24 to 12.5 g NH_3 AU^{-1} hr^{-1}. Emission factors from layer facilities seem to be consistently higher than those from broiler facilities.

A recently completed U.S. EPA funded study (Strader et al., 2000, citing a previous study by Battye et al., 1994), stated that livestock (including poultry) contribute 50-70% of the total national ammonia emission inventory, which is about 5,300 kt/year. However, the underlying emission factors for different livestock and poultry types were taken from a systems analysis with limited U.S. agricultural input (Battye et al., 1994) and yet were used to extrapolate to an entire national level. For example, the US-EPA estimated annual emission for layer hens is approximately 435g NH_3 per bird (which can be traced back to the Battye report). By contrast, The Netherlands currently use a range of 10-83 g NH_3 per bird annual emissions (Groot Koerkamp et al., 1998b). Considering that there were on average 322 million layers in the U.S. in 1999 (USDA, 2000), the difference between the 83 and 435 g estimates results in a disparity in annual contribution to the national annual inventory of roughly 113,300 metric tons of NH_3. This example clearly indicates that the lack of quality, scientific-based emission data may result in system models that predict highly inaccurate estimates of NH_3 emission contribution by animal production.

The limited number of NH_3 emission data for beef and dairy facilities show a narrower range and significantly lower values as compared to swine and poultry. Gay et al. (2002) have recently summarized NH_3 flux rates from 66 farms in Minnesota. Swine housing means varied from 0.35 to 13.0 g NH_3 m^{-2} day^{-1}, poultry housing from 2.85 to 8.0 g NH_3 m^{-2} day^{-1}, dairy about 3.7 g NH_3 m^{-2} day^{-1}, and beef feedlots from 2.2 to 4.4 g NH_3 m^{-2} day^{-1}. Ventilation rates from mechanically ventilated buildings were determined by measuring static pressure across the fan. For naturally ventilated buildings a CO_2 mass balance approach was used. It is difficult to compare this data to other studies because it is highly variable and not reported on the basis of animal units. However, the data indicate that NH_3 emissions from swine and poultry housing are consistently higher than NH_3 emissions from dairy and beef housing and open feedlots.

Facility design and management

The effect of animal facility design and management can have a major impact on all types of emissions. Specific research that has investigated these factors has generally determined large variations in airborne emissions of contaminants like ammonia or dust. Unfortunately, all of the management factors and environmental conditions contributing to these changes in emissions are not well understood or documented.

It has been shown that odor and gaseous emissions from buildings are increased if the walls and floors are constantly covered with layers of feces and urine (Voermans et al., 1995). Design modifications are based on reducing the area of the emitting surfaces, frequent removal of slurry from the houses, movement of slurry through slats, temperature control and ventilation rates. Use

of sloped “catch pans”, gutters and narrow collection channels help reduce emitting surfaces under the slats. Reductions in ammonia emission from new buildings varied from 30 to 70% as compared to conventional buildings.

In the United States, hoop structures with straw bedding are being considered as an alternative to large-scale confinement structures for swine production (Brumm et al., 1997). On deep litter systems (6.8 kg straw pig^{-1} day^{-1}), ammonia emission is comparable with emission from a fully slatted floor barn (Valli et al., 1994). Emissions can be kept at low levels by increasing the amount of straw or by allowing partial urine drainage. However, emissions of nitrogen gases in deep litter systems tend to be higher due to the formation of N_2O which contributes to the greenhouse effect and affects the ozone layer (Groenestein and Faassen, 1996).

Traditional methods of NH_3 control in buildings have involved removal of manure, drying of manure to avoid or reduce urease breakdown, and litter amendments to control pH in broiler litter. Groot Koerkamp et al. (1998b) reported on the effects of a litter drying system on the composition of the litter and the emission of ammonia from a tiered wire floor poultry housing system for layers. They concluded that forced air movement (0.5 m^3 hr^{-1} per hen) above the litter enhanced the evaporation of water from the litter substantially as compared to no forced air movement above the litter. Litter dry matter content was kept above 900 g kg^{-1} and the Total Ammoniacal Nitrogen (TAN) concentration (0.7 g kg^{-1}) and pH (7.3) decreased as compared to the composition of litter in poultry houses without drying of litter. The change in litter composition apparently helped lower ammonia emissions. The lowest levels of ammonia emission (about 2.0 mg hen^{-1} hr^{-1}) were recorded when manure was removed more frequently and more ventilation was provided.

Yang et al. (2000) determined nitrogen losses from four high-rise laying hen houses located in Iowa. Nitrogen losses were between 25 and 41% based on Total Kjeldahl Nitrogen (TKN) in feed. They found that the higher the moisture content of manure, the higher the ratio between NH_3 and TKN in manure, and therefore, the higher the percentage of N loss. These findings are in reasonable agreement to the conclusions reached by Dutch researchers in a previous study described above (Groot Koerkamp et al., 1998b).

It is a common broiler industry practice to manipulate minimum ventilation rates continuously to strike a balance between the need for energy conservation (supplemental heat must be provided during cold weather) and indoor air quality (Gates et al., 1996, Xin et al., 1996). Recent advances in water delivery systems have greatly improved poultry environments (Gates et al., 1996), to the point where problems with dust and gases have replaced humidity as a common complaint to extension personnel and consultants. With a tendency for lower litter moisture content, less ammonia is generated and volatilized. However, this may be offset by an industry practice of reusing broiler litter for multiple flocks; if litter moisture is high enough to support urease breakdown then the potential for high ammonia emission exists because total litter N is greater.

Ammonia emissions from cattle housing is usually influenced by the flooring system, type of bedding and manure handling system (slats, scrape, or flush). Kroodsma et al. (1993) determined the effects of different floor types and flushing on ammonia emission rates from free-stall dairies. Scraped or dirty solid floors gave the highest ammonia emission (about 15 g NH_3 m^{-2}

day^{-1}), while flushing gave the lowest (5 g NH_3 m^{-2} day^{-1}). Scraped or dirty slatted floors were found to emit about 9 g g NH_3 m^{-2} day^{-1}.

Braam et al. (1997) looked at practical ways to reduce ammonia emission from double-sloped solid floors with a central urine gutter in dairy houses. They found that ammonia emission from the compartment with the double-sloped solid floor operating with one urine gutter and without water spraying was reduced by 50% when compared to a control (slatted floor with underfloor slurry pit). Ammonia emission was further reduced when water was sprayed after scraping.

Swierstra et al. (2001) have recently reported on a grooved floor system consisting of prefabricated concrete elements with perforations spaced 1.1 m apart to channel urine from the floor. Feces were removed every two hours by a mechanical scraper and were dumped into the pit through a floor opening at the end of the alley. The blade of the scraper was equipped with a tooth-shaped rubber strip to clean the grooves. Ammonia emissions from the grooved floor were found to be 46% less than emissions from a reference floor (concrete slotted floor). Closing of the perforations resulted in an ammonia emission reduction of only 35% compared to the reference floor.

Demmers et al. (1998) showed that ammonia emission from straw bedded beef housing was 40% less than ammonia emission from a slurry-based dairy unit. Jeppsson (1999) concluded that ammonia emission from deep-bedded housing for heifers using a mixture of peat (60%) and chopped straw (40%) was reduced by almost 60% as compared to bedded areas with long straw. Reduction of ammonia emission was attributed to the ability of peat to absorb water and ammonia, lower pH level, and also to its high C/N ratio. Ammonia emissions were 8 and 18 g NH_3/m^2-day, for the peat-straw mixture and long and chopped straw bedding, respectively. In addition, ammonia emission from the manure alley was found to be significantly less than from the bedding area with straw bedding.

Diet manipulation

Use of improved feeding management practices, selective feed ingredient use, precision in diet formulation, and dietary electrolyte balance has been shown to reduce nutrient excretion, and subsequent, odor and gas emissions from livestock manure (Sutton et al., 2002).

Yucca schidigera extract has been shown to reduce ammonia emission from manure by inhibiting urease activity (Ellenberger *et al*., 1985; Gibson *et al*., 1985). Sutton *et al*. (1992) showed that ammonia emission was suppressed by 55.5% in swine manure from pigs fed sarsaponin extract at a rate of 4 oz/ton of feed, but Kemme *et al* (1993) was unable to verify this response, and showed that much higher amounts of the extract (6,000 ppm) was needed for maximal suppression of ammonia from urea.

Reduced crude protein diets containing synthetic amino acids have been shown to reduce nitrogen excretion in pigs, which can lead to potential reduced ammonia emissions (Hartung and Phillips, 1994, Cahn et al., 1997, 1998, etc.). Reductions in ammonia emissions from 28 to 79% through diet modifications in swine have been reported (Sutton *et al*., 1999).

Ferguson et al. (1998a,b) have examined the effects of diet manipulation on the litter equilibrium NH_3 gas concentration in broiler housing. Gas was sampled using an equilibrium chamber. Equilibrium concentrations between 53 and 83 ppm were obtained for different diet treatments. Reducing crude protein caused equilibrium NH_3 gas concentration to decline by about 30%. Gates et al. (2000) also reported on the effect of reduced crude protein on equilibrium NH_3 broiler litter. Equilibrium NH_3 concentrations varied from 0 to 161 ppm, depending on the flock number, ventilation rate, and diet treatment. A low crude protein diet resulted in about 90% reduction in equilibrium NH_3 concentration even for used litter. The differences between the Gates (2000) and Ferguson (1998b) studies were basically litter moisture content and number of flocks. Gates (2000) worked with significantly drier litter (16 – 25%) than Ferguson (1998b) (50 – 60%) and took measurements over a period equivalent to the raising of three flocks using the same litter, while Ferguson (1998b) data is from one flock only.

Reduced crude protein diets also help reduce NH_3 emissions from dairy and beef cattle. James et al. (1999) reported a 28% reduction in NH_3 emission from dairy cows fed a low crude protein ration. Smits et al. (1995) observed further reductions in NH_3 emissions as compared to James et al. (1999) study. Klopfenstein and Erickson (2001) observed reductions in NH_3 emissions from the surface of beef cattle feedlots between 15 and 30% when cattle were fed a lower crude protein diet.

Diet manipulation as well as its effects on manure production and composition is addressed in detail in another white paper (Sutton et al., 2002).

Waste management systems

Ammonia losses from storage tanks and large open treatment/storage areas are reasonably well documented, but there is a wide variation in the numbers. NH_3 flux rate data from waste management systems is summarized in Table 4.

Sommer et al. (1993) reported that an uncovered manure (dairy and swine) storage unit emitted 5.5 g NH_3 m^{-2} day^{-1}. Ammonia emissions were from 0.6 to 1.8 g NH_3 m^{-2} $day^{-1}$when the manure surface was covered by a natural crust, from 0.24 to 1.2 g NH_3 m^{-2} $day^{-1}$when the manure surface was covered with 15 to 23 cm of straw, and from 0 and 0.36 g NH_3 m^{-2} $day^{-1}$when the manure surface was covered with oil and the storage tank was covered with a wood lid fastened by screws to the tank (Sommer et al., 1993). Hobbs et al. (1999) reported that NH_3 emissions from a swine manure storage unit varied from 2.5 to 7.9 g NH_3 m^{-2} day^{-1}during a 112-day period: average NH_3 emissions were 4.4 g NH_3 m^{-2} day^{-1}. Misselbrook et al. (2000) estimated that NH_3 loss from manure storages in the U.K. at 22.8 kt NH_3 per year. Ammonia emissions from cattle manure storages were estimated to contribute more than 80% of total manure storage emissions (Misselbrook et al., 2000).

Pratt et al. (2002) studied the effect of ambient temperature on NH_3 emissions from stored layer manure. One-ton samples of layer manure were stored in four environmentally controlled chambers for 18 weeks. Setpoint temperatures for the four chambers were 12, 15, 20, and 25 ^oC. Ammonia emissions varied from 126 g 1000 kg^{-1} $week^{-1}$ at 12.3 ^oC to 313 g 1000 kg^{-1} $week^{-1}$ at 24.4 ^oC (Pratt et al., 2002).

Table 4. NH_3 flux rates from waste management systems

System	NH_3 flux rate (g NH_3 m^{-2} day^{-1})	Reference
Manure storage (swine and dairy, uncovered)	5.5	Sommer et al. (1993)
Manure storage (112-day storage)	4.4	Hobbs et al. (1999)
Swine manure storage (deep pit, pull-plug)	57	Zahn et al. (2001)
Swine manure storage (above ground tanks)	144	Zahn et al. (2001)
Swine manure storages	4.5 to 40	Gay et al. (2002)
Dairy manure storages	8 to 27	Gay et al. (2002)
Broiler litter	4.2 to 9.1	Brewer and Costello (1999)
Anaerobic lagoons	4.4	Koelliker and Miner (1973)
Anaerobic lagoons	0.4 to 2.7	Harper and Sharpe (1998)
Anaerobic lagoons	0.19 to 6.0	Harper et al. (2000)
Anaerobic lagoons	0.4 to 5.8	Aneja et al. (2000)
Anaerobic lagoons	77 to 94	Zahn et al. (2001a)
Anaerobic lagoons	15.8	Zahn et al. (2001b)
Anaerobic lagoons	9.6	Natschke et al. (2001)
Land application – swine manure at excessive rates (over 100 m^3/ha)	65 to 140	Chadwick et al. (1998)
Land application – dairy manure at 30 m^3/ha	10.2	Chadwick et al. (2000)
Land application – dairy manure at 39 m^3/ha	9.6	Thompson and Meisinger (2001)
Land application – anaerobic lagoon liquid	5.9 to 12.5	Sharpe and Harper (2002)

Zahn et al. (2001a) reported on NH_3 emissions from 24 swine manure storage systems in Iowa. Ammonia emissions were found to be significantly higher than values reported by European researchers (Table 11). Gay et al. (2002) summarized NH_3 emissions from livestock and poultry manure storage units on 25 farms in Minnesota. Mean NH_3 emissions from swine manure storages varied from 4.5 to 40 g NH_3 m^{-2} day^{-1}; emissions from dairy manure storages ranged from 8 to 27 g NH_3 m^{-2} day^{-1} (Gay et al., 2002).

Brewer and Costello (1999) reported on the NH_3 emissions from new and reused broiler litter. Ammonia emissions were 4.2 and 9.1 g NH_3 m^{-2} day^{-1}, respectively, from new and reused litter. Re-use of litter after the second flock appeared to have no effect on NH_3 emissions.

The first attempt to quantify NH_3 emissions from an anaerobic lagoon was based on a nitrogen mass balance (Koelliker and Miner, 1973). Known quantities of nitrogen removed from and accumulated in the lagoon were subtracted from the known quantity of nitrogen added to the lagoon. The difference was assumed to be the result of NH_3 desorption. Koelliker and Miner (1973) determined this value to be 4.4 g NH_3 m^{-2} day^{-1} or 64 % of the total nitrogen added to the lagoon.

Harper and Sharpe (1998) reported average NH_3 emissions estimated from measurements taken from two lagoons in North Carolina and four lagoons in Georgia (micrometeorological mass balance technique). The values range from 0.4 to 2.7 g NH_3 m^{-2} day^{-1}. The same authors reported high N_2 emissions from anaerobic lagoons, ranging from 0.9 to 12 g N_2 m^{-2} day^{-1}. They attributed these large N_2 emissions to chemical denitrification and biological denitrification (or combination of the two) processes. Harper et al. (2000) reported that NH_3 emissions were quite variable among seasons. The largest emissions occurred during periods of high wind speed and effluent temperature, but during relatively low NH_4^+ effluent concentration. Wind speed and effluent temperature had the highest correlation with NH_3 emissions. Aneja et al. (2000) found

NH_3 emissions rates in the same order of magnitude, but about 2.6 times higher than those reported by Harper and Sharpe (1998), using a dynamic chamber system for flux measurements. Zahn et al. (2001a) found significantly higher values while studying ammonia emission rates from anaerobic lagoons in Iowa, Oklahoma and North Carolina, as shown in Table 11. Lower NH_3 flux rates were reported for a phototrophic lagoon with purple, nonsulfur bacteria as compared to other systems. Volatile solids loading rate differed between the purple and the non-purple lagoons by more than 4-fold (0.07 versus 0.3 kg VS m^{-3} day^{-1}) (Zahn et al., 2001a). Fulhage and Hoehne (1999) demonstrated that deep lagoons (between 6 and 8 m deep) exhibited equilibrium nitrogen levels approximately twice that of shallow lagoons (between 2.5 and 4.0 m deep). The surface area to volume ratio also appeared to strongly influence the equilibrium concentration of nitrogen in deep versus shallow lagoons (Fulhage and Hoehne, 1999). These observations could help explain the differences in NH_3 emission values obtained by Zahn et al. (2001a) as compared to the values reported by Harper and Sharpe (1998), Harper et al. (2000), and Aneja et al. (2000). Unfortunately Zahn et al. (2001a) did not provide information on physical lagoon characteristics, making it impossible to determine if the higher emission values were indeed coming from deep lagoons.

Zahn et al. (2001b) studied the efficiency of a polymer biocover to minimize H_2S and NH_3 emissions from a Missouri swine lagoon (surface area of about 7,800 m^2 and depth of 3.8 m). The cover was installed over the south half of the lagoon. The measured NH_3 flux rate from the uncovered north half of the lagoon was about 15.8 g NH_3 m^{-2} day^{-1}. This was 1.7 times higher than the flux obtained in the covered part of the lagoon.

Natschke et al. (2001) measured NH_3 flux over a swine anaerobic lagoon using the open path FTIR spectroscopy technique. A flux rate of about 9.6 g NH_3 m^{-2} day^{-1} was obtained.

There are only a limited number of studies that deal with emissions from stacked livestock and poultry manure and composting operations. Most research on manure composting has concentrated on chemical transformations, particularly nitrogen, that occur as raw manure becomes composted. Kuroda et al. (1996) measured NH_3 losses during composting of swine manure using a laboratory composting apparatus. Emission of NH_3 changed with material's temperature and occurred mostly during the periods of high temperature. Emissions increased remarkably after starting and at every turning. Eghball et al. (1997) reported N losses of 19 to 42% during outdoor composting of beef feedlot manure in Nebraska. Nitrogen losses were estimated based on a mass balance approach. Dewes (1999) reported on NH_3 emissions from liquid and solid (with 2.5 and 15 kg straw per animal unit per day) cattle manure. Emissions increased from 800 μg NH_3 h^{-1} per kg of liquid manure up to 7,650 μg NH_3 h^{-1} per kg of solid manure. These emission levels were attained soon after the maximum temperatures induced by microbial self-heating had been reached. Sommer and Dahl (1999) measured NH_3 emissions from three piles of deep bedding from a dairy cow facility. Emission of NH_3 occurred during the first 10 days after the piles were established, and 2-3 days after turning. Cumulative NH_3 volatilization was 0.2 kg N per 1,000 kg of compost corresponding to about 3% of the total N. Sommer (2001) studied nutrient loss during composting of deep bedding from dairy farms. NH_3 was emitted during the first 20 days after the composting piles were established. NH_3 emissions varied between 1.5 and 4 mg N 1,000 kg^{-1} s^{-1} during the first 5 days of composting for compacted, mixed, and covered treatments.

The removal of excess nitrogen from manure as nitrogen gas (N_2) prior to land application greatly reduces the risk of pollution from NH_3 emissions into the atmosphere and nitrate leaching into groundwater. However, aerobic treatment used to oxidize NH_3 and remove nitrogen from manure can potentially add to problems by stripping out ammonia from slurry before land spreading, particularly if uncontrolled or excessive airflow rates are used. Treating manure to remove nitrogen has been investigated by a number of researchers, but only a few studies addressed the issue of NH_3 emission from manure treatment plants. Vetter et al. (1987) reported that NH_3 losses from 63 aeration trials with cattle and swine manure were about 20% of the incoming total-N. Willers et al. (1996) described gaseous emissions measurements from batch and continuous treatment plants with capacity to treat 75,000 and 180,000 m^3 of veal calf slurry per year, respectively. Ammonia emissions were 0.7% of the total Kjeldahl nitrogen (TKN) in the slurry for the batch system and between 0.1 and 0.2% of TKN for the continuous system.

Land application of manure constitutes a large source of NH_3 emissions. An inventory for the UK animal agriculture sector indicated that land application of manure represented 30% of the total NH_3 loss (Pain et al., 1998). The major factors influencing NH_3 emission from animal agriculture were (Pain et al., 1998): (i) weather, (ii) soil and manure characteristics, and (iii) application technique. The study by Pain et al. (1998) concluded that the adoption of new land application technology offered the greatest potential for reduction of NH_3 emissions from animal manures.

Burton (1997) compiled European literature (1992 to 1997) on the effects of various land application techniques on NH_3 emissions (Table 5).

Table 5. Effect of land application technique on the reduction of ammonia emissions after spreading cattle and pig slurry on grassland and arable land

Spreading technique	**Application on grassland**				**Application on arable land**			
	Trials	**Handling rate (m^3/ha)**	**% NH_3 loss[1]**	**% Reduction on NH_3 emission[2]**	**Trials**	**Handling rate (m^3/ha)**	**% NH_3 loss[1]**	**% Reduction on NH_3 emission[2]**
Deep injection (30 cm deep)	6	37.4	0.9	98	4	31.8	1.0	98
Shallow injection (7.6 cm deep)	32	21.5	9.4	87	2	18.7	2.8	90
Drag shoe	27	14.0	20	63	5	20.6	9.5	73
Band spreader	3	12.2	43	41	2	18.7	33	31

1 – as a percentage of the NH_3-N in the slurry.
2 – compared to the emission from broadcasting application.

Misselbrook et al. (1997) studied the effects of swine dietary manipulation on NH_3 emissions after land application of pig slurry. Two groups of finishing pigs were fed either a standard commercial diet (CD) or a reduced crude protein diet (RD). Slurries were collected from two groups of finishing pigs and spread on grass/clover swards at 50 m^3/ha in early spring. Slurry from the RD fed pigs had a lower ammoniacal-N, total-N, and volatile fatty acid content, lower pH, and a higher dry matter than slurry from the pigs fed CD. Following land application, NH_3 volatilization from the RD slurry over the first five days was 60% less than NH_3volatilization from the CD slurry.

Chadwick et al. (1998) measured nitrogen losses after overapplication of swine manure in a soil filter system (Solepur process – Martinez, 1997). Approximately 227 m^3 ha^{-1} and 95 m^3 ha^{-1} of swine manure was applied in the fall and summer, repectively, using a tow hose system connected to a 40-m wide spray boom. Total nitrogen application was 656 kg N ha^{-1} and 1,180 kg in the summer and fall, respectively (Chadwick et al., 1998). Total applied nitrogen losses through NH_3 volatilization were 6% following application in the fall and 31% after summer application. The nitrogen losses reported by Chadwick et al. (1998) were within the reported range following surface broadcasting of pig slurries at agronomic rates (see Table 5 for comparison). Nearly 95% of the total nitrogen loss occurred during the first four days after manure application. Ammonia volatilization rates immediately following the application were about 66 g NH_3 m^{-2} day^{-1} and 137 g NH_3 m^{-2} day^{-1} for the summer and fall applications, respectively Chadwick et al., 1998).

Chadwick et al. (2000) described a series of experiments where NH_3 emissions were compared following land application of manure at 30 m^3 ha^{-1} with broadcast, low trajectory spreaders, and shallow injection equipment. Ammonia emissions were 85% less using shallow injection rather than broadcasting. Low trajectory spreaders reduced NH_3 emissions by 40 to 75% as compared to broadcast spreading. Ammonia emissions were high (up to 10.2 g NH_3 m^{-2} day^{-1}) during the first six hours after application (Chadwick et al., 2000). Smith et al. (2000) found similar results during a manure application study.

Little information is available on NH_3 emissions from land application of manure in the U.S. Safley et al. (1992) reported on nitrogen loss during irrigation of anaerobic lagoon effluent from swine operations. Total Kjeldahl Nitrogen (TKN) losses ranged from 15 to 43% during sprinkler irrigation using a center pivot. Fifty-four to 100% of TKN losses was from volumetric losses such as evaporation and drift (Safley et al., 1992). Ammonia losses ranged from 14 to 37% and are comparable to losses from untreated pig slurry applied with band spreaders or drag shoes (Table 5) (Safley et al., 1992).

Thompson and Meisinger (2001) measured NH_3 emission following land application of dairy manure using micrometeorological and wind tunnel techniques. The average rate of loss was about 10 g NH_3 m^{-2} day^{-1} in the six-hour period following application. Subsequently, the rate of loss declined, although diurnal variations were observed over the next several days. The total NH_3 loss over eight days was 36 kg N ha^{-1} or 71% of the total applied NH_3-N (Thompson and Meisinger, 2001). Fifty-eight percent of the total NH_3 loss occurred within six hours of application (Thompson and Meisinger, 2001).

Sharpe and Harper (2002) measured NH_3 flux from a soybean field irrigated with effluent from an anaerobic swine lagoon. Large NH_3 fluxes occurred immediately after the effluent was irrigated on the field. However, fluxes decreased to background level within 24 to 48 hours after irrigation (Sharpe and Harper, 2002). Maximum flux rates, obtained with wind speeds ranging from 1.5 to 2.0 m s^{-1}, were between 11.1 and 12.5 g NH_3 m^{-2} day^{-1}. Ammonia flux rates of 5.9 g NH_3 m^{-2} day^{-1} were obtained at wind speeds lower than 1 m s^{-1} (Sharpe and Harper, 2002).

Thompson and Meisinger (2001) stated that numerous methods of land application that minimize NH_3 emissions are available. However, reliable field data on NH_3 losses under the soil, climate, and application regimes of the individual states or regions are needed to evaluate the contribution of land application of manure to the nation NH_3 emissions inventory.

NITROUS OXIDE

Livestock and poultry housing

Nitrous oxide is a product of both nitrification and denitrification. Pahl et al. (2001) demonstrated that there was a large variation in the split between nitrification and denitrification processes as the source of N_2O production. Their results showed that specific conditions could favor nitrification or denitrification to be the principal source of N_2O emissions: (i) through denitrification under oxygen inhibition; or (ii) through nitrification in aerobic systems, in combination with the presence of nitrification products. Therefore, N_2O can be released at any stage of livestock production where conditions favor these processes (Chadwick et al., 1999). Leaching, absorption by plants, or utilization by microorganisms indirectly influences the production of N_2O.

Nitrous oxide emissions are an environmental concern. Houghton et al. (1992) stated that N_2O is approximately 200 times more efficient than CO_2 in absorbing infrared radiation. Methane, another strong greenhouse gas, is only 26 times more efficient than CO_2 in absorbing infrared radiation. Furthermore, N_2O contributes to the reduction of ozone in the stratosphere through the photochemical decomposition of N_2O to NO.

Data on N_2O emissions from animal housing is limited. Osada et al. (1998) measured N_2O emissions from an experimental swine finishing unit with a slatted floor during an 8-week period. Nitrous oxide emissions varied from 0.8 to 2.1 g N_2O AU^{-1} day^{-1}. Emissions were reduced when underground manure pits were discharged weekly (Osada et al., 1998).

Chadwick et al. (1999) summarized N_2O emissions from animal housing in the U.K. Nitrous oxide emissions varied from 0.4 to 26 g N_2O AU^{-1} day^{-1}. The lowest emissions values were from swine housing and the highest were from poultry housing. Chadwick et al. (1999) also noted that dairy housing with slurry-based systems had significantly lower N_2O emissions than dairy housing that used straw bedding.

Waste management systems

Intensively managed grasslands for beef and dairy cattle are a significant source of N_2O emissions. In the U.K. a recent inventory of N_2O emissions indicated that approximately 10% of all N_2O losses are related to livestock on pasture, especially dairy and beef cattle (Chadwick et al., 1999). Velthof et al. (1998) modelled N_2O emissions from intensively managed dairy farms on sandy soils: results indicated that direct (grazing, manure storage, land application, silage, etc.) N_2O emissions were from 1.4 to 4.1 mg N_2O-N m^{-2} day^{-1} and indirect emission (purchased

N fertilizer, roughage and concentrate) were from 0.27 to 1.1 mg N_2O-N m^{-2} day^{-1}. Yamulki et al. (1998) found N_2O fluxes from cattle dung and urine that were significantly lower than the values reported by Velthof et al. (1998). Maximum N_2O emissions were 0.17 mg N_2O-N m^{-2} day^{-1} from urine and up to 0.05 mg N_2O-N m^{-2} day^{-1} from dung, based on a stocking rate of two animals per hectare and 180 day grazing period per year (Yamulki et al., 1998). Total nitrous oxide emissions from excreta were significantly higher during fall than during summer. Average N_2O emissions from urine patches during the experimental period was more than five times greater than average N_2O emissions from dung (Yamulki et al., 1998).

Chadwick et al. (1999) estimated N_2O emissions from stockpiled manure to be 5.6 kt N_2O $year^{-1}$. Stockpiled cattle and poultry manure contributed 3.58 kt N_2O $year^{-1}$ and 1.86 kt N_2O $year^{-1}$, respectively. Sommer et al. (2000) measured N_2O emissions from covered cattle slurry. Maximum N_2O emissions of 25 mg N m^{-2} day^{-1} were recorded during summer. Total N_2O emission was highest from digested slurry (Chadwick et al., 1999). Brown et al. (2000) measured N_2O flux from stored solid dairy manure using a flow-through flux chamber and a tunable diode laser trace gas analyzer (Edwards et al., 1994) for N_2O detection. The mean daily flux was between 0 and 330 mg N m^{-2} day^{-1} for samples collected within 30 cm of the surface of the pile. This variability was attributed to variations in water content and redox potential. N_2O fluxes were highest at water content between 55 and 70% and redox potentials between 150 and 250 mV.

Kuroda et al. (1996) conducted a laboratory experiment on emissions from composting swine wastes. Nitrous oxide emissions accounted for only 10% of the total-N content in the initial material (Kuroda et al., 1996). Sommer (2001) obtained similar results during a study on composted bedding from cattle facilities: less than 0.3% of the total-N was emitted as N_2O.

Sommer and Moller (2000) studied N_2O emissions from composting litter from swine housing. A closed chamber technique was used for sampling. Nitrous oxide emissions were high when only small amounts of straw had been used in the bedding (high density material). Nitrous oxide was produce at the start of composting and after compost temperatures decreased of the compost. Total N_2O emissions from the high-density compost heap were 58 g N ton^{-1}.

Hao et al. (2001) measured greenhouse gas emissions during cattle feedlot manure composting. Emissions were measured using a vented chamber. Compared with CO_2, N_2O emissions were relatively low during composting. N_2O emission values were equivalent to 0.62% (passive composting) and 1.07% (windrow composting) of the total initial N in manure. Daily emissions varied between 0 and 600 mg N/m^2-day. Several mechanisms were thought to contribute to N_2O production during composting, including nitrification, denitrification, and chemo-denitrification.

Significant N_2O emissions are released from manure treatment plants that use a combination of aerobic and anoxic treatments. Until recently the use of intermittent aeration to produce aerobic, anoxic and anaerobic cycles during the treatment of manure was regarded as a working solution to the problem of excess nutrients because the treated effluents from such plants were virtually free of NH_3-N and NO_3, and had reduced amounts of total-P. However, recent research has shown that this type of treatment generates not only N_2 as the end product, but also N_2O. Burton et al. (1993), Willers et al. (1996), and more recently Beline and Martinez (2002), have shown

that up to 20% of the nitrogen removed from manure can be released as N_2O. Pahl et al. (2001) measured N_2O production from continuous aeration and cyclic aeration schemes. The percentage of TAN removed that resulted in N_2O production varied between 2 and 46%. It was suggested that a process with sequential aeration, which creates a cycling of nitrifying and completely denitrifying phases, reduces N_2O emissions.

Pahl et al. (2001) measured N_2O production from continuous aeration and cyclic aeration during manure treatment. The percentage of TAN removed that resulted in N_2O production varied between 2 and 46%. Pahl et al. (2001) suggested that treatment processes using sequential aeration, which create a cycling of nitrifying and completely denitrifying phases, reduce N_2O emissions.

Harper et al. (2000) did not detect N_2O emissions from swine anaerobic lagoons using micrometeorological and laser spectrometry techniques. No N_2O emissions were detected from the sludge layer of the lagoon (Harper et al., 2000).

Nitrous oxide emissions during land application of manure are relatively low. Chadwick et al. (1999) reported that net N_2O emissions following land application of swine and dairy manure represented 0.4 and 0.3%, respectively, of the nitrogen added by the manure. Sharpe and Harper (2002) observed slightly higher N_2O emissions after irrigation of swine lagoon liquid to soybean fields: N_2O flux was 0.0016 mg N_2O-N m^{-2} day^{-1} prior to irrigation and varied from 2.5 to 3.8 mg N_2O-N m^{-2} day^{-1} after irrigation. Total N_2O emissions during the measurement period were about 1.5% of total-N applied (Sharpe and Harper, 2002).

Chadwick et al. (1998) studied N_2O emissions from grasslands spread with excessive volumes of manure (average annual loading of 5,000 kg N ha^{-1}) in summer and fall. Manure was applied using a towed hose connected to a 40-m wide spray boom to ensure uniform application across the field. A total of 272 kg N ha^{-1} (23% of the total-N applied to the field) was emitted as N_2O in the 111 days following fall application: maximum N_2O flux rates were 400 g N_2O-N ha^{-1} hr^{-1}. In the summer, total N_2O emissions were 1.1 kg N ha^{-1} (0.17% of the total-N applied to the field) eight days after manure application (Chadwick et al., 1998).

HYDROGEN SULFIDE

Livestock and poultry housing

Hydrogen sulfide is formed by bacterial sulfate reduction and the decomposition of sulfur-containing organic compounds in manure under anaerobic conditions (Arogo et al., 2000). H_2S gas is colorless, heavier than air, highly soluble in water and has the characteristic odor of rotten eggs at low concentrations. At concentrations around 30 ppb the H_2S odor can be detected by over 80% of the population (Schiffman et al., 2002). The U.S. OSHA has implemented a 10 ppm limit for indoor 8-hour H_2S exposures to protect human worker health (ACGIH, 1992). Most human health problems associated with hydrogen sulfide emissions are related to emissions from paper mills, refineries, and meat packing plants (Schiffman et al., 2002). Currently, there is only

circumstantial evidence relating emission of hydrogen sulfide from livestock and poultry to human health.

Although there are health risks associated with high concentrations of H_2S, concentrations are usually very low in and around animal housing as compared to concentrations of CO_2 and NH_3. Ni et al. (2000) and Ni et al. (2002) measured H_2S concentrations between 65 and 536 ppb in swine finishing facilities in Indiana. Bicudo et al. (2000) measured hydrogen sulfide concentrations continuously during 30-day periods around swine buildings in Minnesota. A maximum of 450 ppb of H_2S was recorded at 5 m downwind from a naturally ventilated finishing barn. Mean H_2S concentrations around a nursery (mechanically ventilated) and wean-to-finish (naturally ventilated) barns were between 4.5 and 10.9 (±0.3) ppb. H_2S levels around a hoop barn were lower than 2 ppb. Zhu et al. (2000b) studied the daily variations in H_2S emissions from various mechanically and naturally ventilated swine housing systems in Minnesota. H_2S concentrations varying between 200 and 3,400 ppb were reported.

Koelsch et al. (2001) measured total reduced sulfur levels in a beef cattle feedlot using a Jerome meter. This instrument measures total reduced sulfur (TRS) compounds, including alkyl sulfides, disulfides, mercaptans, and cyclic sulfur compounds. Concentrations in the center of the feedlot varied between 1 and 14 ppb. Clark and McQuitty (1987b) recorded a maximum H_2S level of 145 ppb in four of six commercial free-stall dairy barns in Alberta. McQuitty et al. (1985) reported on H_2S concentrations in three commercial laying barns under winter conditions. No detectable traces of H_2S were found in two barns and a maximum H_2S concentration of 30 ppb was measured in the third barn.

Several researchers have studied the effects of swine dietary sulfur intake on H_2S levels in pig housing. Shurson et al. (1998) reported a reduction in H_2S emissions from nursery pigs fed a low sulfur diet as compared to a traditional diet. Donham et al. (1988) documented a positive, but not significant, correlation between sulfate levels in drinking and cleaning water and the sulfide content in swine manure. A slightly positive relationship between total sulfides in manure and hydrogen sulfide concentration in the building exhaust air was also reported.

A limited amount of research has focused on H_2S emissions from animal housing. Most of this data is from swine facilities (Table 6). Measurements obtained by Zhu et al. (2000a) were reported for a 12-hour period, and values shown in Table 6 were not converted to a 24-hour period.

Table 6. Hydrogen sulfide emission factors from livestock and poultry housing

Species	Production unit	Notes	Emission Factor (g H_2S AU^{-1} day^{-1})	Reference
Pig	Finish	Fully slatted	2.4-22.6	Ni et al. (2002)
	Finish	Fully slatted – no pigs	0.22-0.49	Ni et al. (2000)
	Finish	Fully slatted	1.25	Ni et al. (2000)
	Finish	Fully slatted (mechanically ventilated)	5	Zhu et al. (2000a)
	Finish	Fully slatted (naturally ventilated)	2-7	Zhu et al. (2000a)
	Farrowing	Fully slatted	4	Zhu et al. (2000a)
	Gestation	Fully slatted	1	Zhu et al. (2000a)
	Nursery	Fully slatted	23-160	Zhu et al. (2000a)
Poultry	Broiler	Litter	3.3	Zhu et al. (2000a)

Hydrogen sulfide emissions from swine and poultry housing tend to be under 5 g H_2S AU^{-1} day^{-1}. Ni et al. (2002) found that diurnal fluctuations and differences between daily H_2S mean concentrations were relatively large and that spatial differences were not significant when averaged over long durations.

Gay et al. (2002) reported on H_2S emissions rates from 80 farms in Minnesota. Mean H_2S emissions varied from 0.02 to 1.5 g H_2S m^{-2} day^{-1} from swine housing, from 0.03 to 0.35 g H_2S m^{-2} day^{-1} from poultry housing, from 0.09 to 0.25 g H_2S m^{-2} day^{-1} from dairy housing, and were about 0.15 g H_2S m^{-2} day^{-1} from beef feedlots. Ventilation rates were measured as explained before. This data set was subject to large variability and it is difficult to compare it to other data reported in terms of AU. However, this data indicates that H_2S emissions are consistently higher for swine housing as compared to poultry and dairy housing and beef feedlots. More data is needed to identify baseline H_2S emissions from livestock and poultry housing.

Waste management systems

Limited information is available on H_2S emissions from manure management systems. Most data are reported in terms of flux rates (Table 7).

Table 7. H_2S flux rates from waste management systems

System	H_2S flux rate (g H_2S m^{-2} day^{-1})	Reference
Stored and agitated manure	28 to 100	Hobbs et al. (1999)
Manure storages	0.95	Zahn et al. (2001a)
Anaerobic lagoons	0.21 to 0.28	Zahn et al. (2001a)
Anaerobic lagoons	0.63 to 1.82	Zahn et al. (2001b)
Swine manure earthen basins	0.65 to 5.1	Gay et al. (2002)
Dairy manure earthen basins	0.37	Gay et al. (2002)
Swine manure above ground storages	0.8 to 12.5	Gay et al. (2002)
Dairy manure above ground storages	70	Gay et al. (2002)

Hobbs et al. (1999) measured H_2S fluxes from swine manure stored from 0 and 112 days and constantly stirred. The average daily H_2S flux rate was 66.6 g H_2S m^{-2} day^{-1}; the flux rate decreased from 100 to about 28 g H_2S m^{-2} day^{-1} at the end of the 112-day period.

Arogo et al. (2000) investigated the effects of manure settling characteristics and initial sulfate concentrations on H_2S production in stored liquid swine manure. After 30 days of storage, the molecular H_2S concentration was higher in the bottom layer of the manure than in either the middle or top layers. The cumulative H_2S concentration was between 300 and 400 mg L^{-1}. Arogo et al. (2000) observed that higher initial sulfate concentration in the manure resulted in higher sulfide concentration during the storage period.

Clanton and Schmidt (2000) measured various sulfur-containing compound concentrations in collected air and liquid samples from stored swine and dairy manure and correlated these sulfur compound concentrations with each other. Of the 20 sulfur-containing compounds analyzed, six were detected in the air samples and seven were detected in liquid samples. Carbonyl sulfide and carbon disulfide gave the highest mean concentrations in air samples (10.9 and 32.3 ppb,

respectively). The compounds that gave the highest mean concentrations in liquid samples were methyl mercaptan and carbonyl sulfide (52.5 and 47.2 ppb, respectively). It was also observed that H_2S, dimethyl sulfide and carbon disulfide changed with time. A strong correlation was obtained between odor and H_2S concentrations.

Zahn et al. (2001a) reported on H_2S emission from 29 swine manure storages and lagoons. The mean flux for manure storages (earthen basins, concrete and steel above ground tanks) was 0.95 g H_2S m^{-2} day^{-1}. The mean H_2S flux for anaerobic lagoons was between 0.21 (purple lagoons) and 0.28 g H_2S m^{-2} day^{-1} (non-purple lagoons). No information on physical characteristics and dimensions of lagoons were given. Zahn et al. (2001b) obtained H_2S fluxes between 0.63 and 1.82 g H_2S m^{-2} day^{-1} from an anaerobic swine lagoon in Missouri (about 7,800 m^2 of surface area and 3.8 m of depth).

Gay et al. (2002) have summarized H_2S flux rates from livestock and poultry manure storage units from about 40 farms in Minnesota. Mean H_2S flux rates from swine storages varied from 0.4 (manure stack) to 12.5 g H_2S m^{-2} day^{-1} (concrete storage tank). Fluxes from swine earthen basin storages varied from 0.65 to 5.1 g H_2S m^{-2} day^{-1}. Dairy manure storage units had mean H_2S flux rates that varied from 0.37 (earthen basin) to 70 g H_2S m^{-2} day^{-1} (concrete storage tank).

Large quantities of H_2S can be released during agitation of stored liquid manure. Patni and Clarke (1991) measured peak H_2S concentrations of 70 and 100 ppm during agitation in a dairy and swine barn, respectively. A maximum H_2S concentration of 220 ppm was reported for the exhaust air from the pit fan of the deep-pitted swine facility. Tengman et al. (2001) measured H_2S concentrations at various distances from six different swine manure storages during agitation and pumping. Average H_2S concentrations peaked at 400 ppb at 15 m downwind from the storage, 200 ppb at 30 m downwind from the storages, and about 120 ppb at 60 m downwind from the storages. The highest H_2S concentration recorded was 671 ppb. High H_2S levels were recorded 4 to 6 hours after agitation was initiated. Concentrations decreased to about 30 ppb 30 to 60 minutes after agitation ended.

METHANE

Livestock and poultry housing

Methane (CH_4) is produced by the microbial degradation of soluble lipids, carbohydrates, organic acids, proteins, and other organic components. CH_4 is another strong greenhouse gas. The presence of atmospheric CH_4 has been associated with climatic changes: Sommer and Moller (2000) reported that CH_4 contributes between 9 and 20% to the total global warming potential.

Table 8 lists estimated CH_4 contributions from various livestock and poultry species. These CH_4 emission estimates were based on standard methane conversion factors (MCF) applied to a global scale rather than actual measurements (Safley and Casada, 1992).

Table 8. Estimated methane emissions from livestock and poultry waste (Safley and Casada, 1992)

Animal type	CH_4 Emission Factor (kg CH_4 $animal^{-1}$ $year^{-1}$)
Cattle in feedlots	23
Dairy	70
Swine	20
Caged Layer	0.3
Broiler	0.09
Turkey and ducks	0.16

The MCFs used by Safley and Casada (1992) were based on manure handling method, temperature, and the amount of volatile solids in manure. Steed and Hashimoto (1994) conducted a laboratory experiment to verify the estimated MCF values for dairy cows used by Safley and Casada (1992) (Table 9). This research indicated that the MCF was less for dry manure under aerobic conditions, such as that found on feedlots and pastures, than for liquid or solid manure storage systems.

Table 9. Measured methane emission factors (MCF) for dairy cows.

System Type	MCF estimates by Safley and Casada (1992)	MCF measured at 20°C Steed and Hashimoto (1994)
Pasture/Feedlot	10	0.3
Liquid slurry	20-90	55.3
Solid	10	45.7

Kaharabata and Schuepp (2000) used an atmospheric tracer (SF_6) to estimate CH_4 emissions from dairy cattle housed in a barn and feedlot. The tracer gas was released from sixteen point sources distributed within the barn or feedlot to simulate the CH_4 release from cows. Predicted CH_4 emissions from the barn and feedlot were 542 L CH_4 cow^{-1} day^{-1} and 631 L CH_4 cow^{-1} day^{-1}, respectively. Overall uncertainty of the results was approximately 30%.

Osada et al. (1998) measured CH_4 emissions from an experimental swine finishing unit with a slatted floor during an 8-week period. Methane emissions varied from 48 to 54 g CH_4 AU^{-1} day^{-1}. Zahn et al. (2001) measured CH_4 emissions from deep-pit and pull-plug swine finishing facilities during August and September of 1997. Methane emissions of 160 g CH_4 AU^{-1} day^{-1} were reported (Zahn et al., 2001).

Waste management systems

Substantial amounts of CH_4 are produced by the anaerobic degradation of organic compounds and proteins contained in manure. Table 10 summarizes the information related to CH_4 flux rates obtained from waste management systems.

Anaerobic digestion systems that are overloaded with organic waste have high volatile organic compound (VOC) emissions and low CH_4 emissions. However, optimum loading rates increase the efficiency of converting complex organic matter into CH_4 (Hill and Bolte, 1989, Zahn et al., 2001a).

Table 10. CH_4 flux rates from waste management systems

System	CH_4 flux rate (g CH_4 AU^{-1} day^{-1})	Reference
Stored and agitated manure	6.9 to 36.6	Hobbs et al. (1999)
Manure storages	154	Zahn et al. (2001a)
Anaerobic lagoons	0.1 to 50	Sharpe and Harper (1999)
Anaerobic lagoons	173 to 188	Zahn et al. (2001a)
Anaerobic lagoons	69 to 140	Zahn et al. (2001b)
Anaerobic lagoons	19.2	Natschke et al. (2001)
Anaerobic lagoons	0.5 to 11.5	Sharpe et al. (2002)

Several researchers have studied biogas production from anaerobic swine and dairy manure lagoons (Chandler et al., 1983; Safley and Westerman, 1988; Safley and Westerman, 1989). Biogas primarily consists of CH_4 and CO_2; CH_4 content is usually between 60 % and 80 % of the total biogas. Biogas produced from anaerobic lagoons and digesters can be (i) collected and used as alternative energy, (ii) burned, or (iii) discharged to the atmosphere. The rates of biogas production from anaerobic treatment lagoons are usually lower than those from anaerobic digesters. Production rates vary from 0.05 to over 1 m^3 biogas m^{-2} day^{-1}.

Zahn et al. (2001a) measured CH_4 fluxes from manure storages and anaerobic lagoons located in Iowa, Oklahoma and North Carolina. Methane flux was 154 g CH_4 AU^{-1} day^{-1} from manure storages and between 173 and 188 g CH_4 AU^{-1} day^{-1} from anaerobic lagoons. Zahn et al. (2001a) did not include information on physical characteristics and dimensions of storages and lagoons. Total CH_4 emissions were estimated at approximately 2,000 g CH_4 $hour^{-1}$ from manure storages and between 14,000 and 15,000 g CH_4 $hour^{-1}$ from anaerobic lagoons. Zahn et al. (2001b) obtained CH_4 fluxes that ranged from 116 g CH_4 AU^{-1} day^{-1} in summer to 69 g CH_4 AU^{-1} day^{-1} in fall from an anaerobic lagoon located in Missouri (surface area of about 7,800 m^2 and depth of 3.8 m).

Natschke et al. (2001) obtained a flux rate of about 19.2 g CH_4 AU^{-1} day^{-1} from a swine anaerobic lagoon using open path FTIR spectroscopy. More recently, Sharpe et al. (2002) reported on CH_4 emissions from swine anaerobic lagoons located in North Carolina using micrometeorological techniques. Emissions varied from 0.5 to 11.5 g CH_4 AU^{-1} day^{-1} and were reasonably well correlated with wind speed, lagoon temperature and volatile solids content.

Hobbs et al. (1999) measured CH_4 flux rates from agitated manure stored from 0 to 112 days. Average daily CH_4 emissions were 21.4 g CH_4 AU^{-1} day^{-1}. Methane flux rates increased from 6.9 to over 36.6 g CH_4 AU^{-1} day^{-1} during the 112-day period (Hobbs et al., 1999.).

Sommer et al. (2000) measured CH_4 emissions from stored cattle slurry and fermented cattle slurry covered with straw, Leca® rock, and natural crust. Methane emissions from stored fermented slurry and cattle slurry varied between <0.01 and 1.9 or 0.9 g CH_4 m^{-3} $hour^{-1}$, respectively. Methane emissions from covered surfaces were almost 40% less than emissions from uncovered manure (Sommer et al., 2000). The reduced CH_4 emissions from the covered surfaces may have been the result of CH_4 oxidation in the surface covers or in the interface between the cover and liquid in the storage. Sommers et al. (2000) also concluded that fermentation did not reduce CH_4 emissions during storage of the slurry.

Methane can also be emitted during composting processes depending on the method of composting and management of composting piles. Kuroda et al. (1996) reported CH_4 emissions from a laboratory-scale composting experiment. CH_4 emissions began within 24 hours of the beginning of the composting process and were low afterwards. In another trial CH_4 increased considerably when aeration stopped.

Sommer and Dahl (1999) measured low CH_4 emissions from composting of deep bedding used in dairy operations. Dynamic chambers were used to obtain flux rates. CH_4 from the compressed and untreated bedding was observed between 30 and 40 days after the start of the composting process. The highest emission rate obtained was about 53 g CH_4 ton^{-1}.

Sommer and Moller (2000) studied CH_4 emissions from composting deep bedding from swine housing during a 4-month period. A compost pile with a high bulk density (0.44 kg L^{-1}) produced CH_4 at a high rate (254 g CH_4 ton^{-1}) during the thermophilic phase of composting. Methane emissions from the high bulk density pile were high at the beginning of the composting process and between 15 and 30 days after composting started (Sommer and Moller, 2000); emissions increased slightly after 50 to 60 days of composting. Methane emissions from a low bulk density (0.23 kg L^{-1}) compost pile were not significant (Sommer and Moller, 2000).

Hao et al. (2001) measured CH_4 emissions from composting beef feed lot manure using a vented chamber. CH_4 concentration was found to increase with depth of the pile. The highest CH_4 concentrations were always found at the bottom of the composting piles. Emissions from passive composting (8.4 kg CH_4 Mg $manure^{-1}$) were not significantly different from active windrow composting (10.8 kg CH_4 Mg $manure^{-1}$).

Sommer (2001) assessed the potential for the compaction, cutting, and mixing and covering of stored deep bedding used in dairy houses to reduce CH_4 emissions from that source. The highest CH_4 fluxes varied from 2.7 and 9.3 mg CH_4 ton^{-1} min^{-1} from the compacted, cut, and mixed deep bedding piles from 0 to 40 days after the start of composting. Methane emissions were only 0.03% of the total-C in the compost. Emissions of CH_4 from the covered and untreated piles were always smaller than from the compacted, cut, and mixed piles.

Chadwick et al. (1998) measured CH_4 emissions following the application of large amounts of manure on grassland in the summer (95 m^3 ha^{-1}) and fall (227 m^3 ha^{-1}). Methane emissions were negligible prior to manure application but increased significantly after application. Over 98% of CH_4 loss occurred in the first four days following manure application; emissions rapidly decreased to a relatively low and constant level after the initial peak was reached (Chadwick et al., 1998). Summer application resulted in total CH_4 emissions of 2.35 kg CH_4 ha^{-1} after 11 days. Methane emissions more than doubled following fall application at 5.5 kg CH_4 ha^{-1} after 11 days (0.12% of the total-C added in the manure).

NON-METHANE VOLATILE ORGANIC COMPOUND

Livestock and poultry housing

Animal housing and manure handling systems generate a variety of gases. Most of the research conducted to date has not quantified VOC emissions but rather documented the generation of these gases. Kreis (1978) developed one of the earliest lists of volatile compounds associated with decomposition of cattle, poultry, and swine wastes. He listed 32 compounds reported to have come from cattle wastes, 17 from poultry wastes, and more than 50 compounds from swine wastes. Hartung and Phillips (1994) reported quantitative information on concentrations found in the air of animal houses for 23 VOC. O'Neill and Phillips (1992) compiled a list of 168 different compounds identified in swine and poultry wastes. More recently, Schiffman et al. (2001b) identified a total of 331 different VOC and fixed gases from swine facilities in North Carolina.

These odorous compounds are usually produced and accumulated in collection and storage systems where feces and urine are decomposed by bacteria under anaerobic conditions. There are four different chemical classes of VOC: volatile fatty acids (VFA), indoles and phenols, amines, and sulfur-containing compounds. The VFA group consists of acetic, propionic, butyric, iso-butyric, valeric, iso-valeric, caproic, and capric acids. Indole, skatole, cresol, 4-ethylphenol appear to be the major odorants included in the indole and phenol group. Phenolic compounds are produced from the microbial degradation of amino-acids such as tyrosine in the intestinal tract of animals. Volatile amines include compounds such as methylamine, ethylamine, putrescine, etc. The main components of the sulfur-containing group are sulfides as well as methyl- and ethyl- mercaptans. These compounds are produced by the reduction of sulfate and by bacterial degradation of sulfur-containing amino-acids. Zhu (2000) provided a thorough review of the microflora in swine manure and its potential to produce odorous volatile compounds.

Information on VOC emissions from animal housing is limited. Zahn et al. (2001a) measured VOC emissions from pull-plug and deep-pit swine houses during August and September 1997. Twelve different non-methane VOCs were detected at a total concentration of 806 $\mu g\ m^{-3}$. The VOC mixture consisted primarily of acetic, propionic, and butyric acid. Estimated VOC emissions were 90 g VOC $hour^{-1}$.

Waste management systems

The volatilization process of malodorous and other VOCs from waste management systems is a dynamic process that includes both biological and chemical transformation processes that occur not only within the liquid itself, but also at the air-liquid interface. Most of the research conducted on VOCs in manure storage and treatment systems has focused on the generation and characterization of these compounds rather than on emissions.

Zahn et al. (1997) monitored VOC emissions from an above ground concrete storage tank (24.4 m in diameter, and 2.44 m deep) using thermal adsorbent tubes containing a combination of Tenax TA and Carboxen- 569. Twenty-four different compounds were detected. The volatilization rate of VOCs from the stored manure was positively correlated with wind speed

between 0.2 and 9.4 m s^{-1}. Total-VOC concentrations at 0, 25, and 100 m from the storage basin were 27.7, 19.4 and 10.7 mg m^{-3}, respectively. Increased flux rates were associated with higher wind speeds. The total VOC flux rate for the manure storage was estimated to be 173 g hr^{-1} at an average wind speed of 3.6 m s^{-1}, but there was a 1.3 and 1.6-fold increase over flux rates determined at 3.6 m s^{-1} for the measurements taken at wind speeds of 6.3 and 9.4 m s^{-1}, respectively.

Hobbs et al. (1999) measured and recorded VFA, phenols and indole concentrations and flux rates from stored swine manure with storage times between 0 and 112 days. Concentrations of VFA declined to less than 5% of their original value after 100 days of storage. Hobbs et al. (1999) suggested that this was probably due to conversion to CH_4 and CO_2. The VOC found in the greatest concentration within the slurry was 4-methyl phenol with over 150 mg L^{-1} after 8 days. This reduced to below 60 mg L^{-1} after 73 days. The flux rate of phenols declined proportionally to the slurry concentration over the storage period. The average flux rate for 4-methyl phenol was 0.44 g m^{-2} day^{-1}, with a maximum of 0.72 g m^{-2} day^{-1}. Emission rates of indole and 3-methyl indole were less than 0.001 g m^{-2} day^{-1} throughout the storage period.

Zahn et al. (2001a) measured VOC flux rates for different swine manure management systems including manure storages and anaerobic lagoons. The concentration of VOC in air samples was highest for manure storages, which received a high input of volatile solids. VOC flux rates were 30.2, 1.4, and 0.18 g VOC m^{-2} day^{-1} for manure storage, anaerobic lagoon, and purple anaerobic lagoon, respectively. The flux rate obtained for manure storages by Zahn et al. (2001a) was significantly higher than values presented by Hobbs et al. (1999), but the conditions in which measurements were performed were much different from the controlled experiments conducted by Hobbs et al. (1999).

Bicudo et al. (2002) recently reported on odor and VOC emissions from swine manure storages using a wind tunnel and SPME (solid phase micro extraction) field samplers. Total VOC flux rates obtained during a two-year monitoring period averaged 17.7 g VOC m^{-2} day^{-1} in the first year and 26.1 g VOC m^{-2} day^{-1} in the second year. Significant amounts of alkane compounds were found in air samples. There was preliminary indication that some VOC sampled with a wind tunnel could have originated as artifacts. VOC determinations from manure samples indicated the presence of the same alkane compounds in manure, but at a significantly lower concentration than in air samples, thus suggesting rapid volatilization of such compounds. Other VOC emitted from manure included several known odorants such as volatile fatty acids (VFA), phenol, 4-methylphenol, 4-ethylphenol, indole, and skatole.

Only limited information is available on the relationship between VOC and odor emission from manure treatment systems. Sneath (1988) studied the effects of removing solids from aerobically treated piggery slurry on volatile fatty acids (VFA) levels during storage. Stability was measured in terms of the time taken to reach two specific concentrations of VFA, 0.23 and 0.52 kg m^{-3}. Slurries stored until a VFA concentration reaches 0.23 kg m^{-3} was found not to cause odor problems, while those containing above 0.52 kg m^{-3} have shown to release offensive odors. It was found that removal of solids using fine sieves or decanting centrifuge extended the storage times of the liquid portion by one-third before the VFA level indicated that offensive odors had returned to the slurry.

There is no information related to VOC emissions from land application of manure. Limited information is available on VOC from land application of sewage sludge. Potential for VOC crop uptake, livestock ingestion, and contamination of ground and surface water should be low under routine, managed applications of manure to agricultural land.

DUST

Livestock and poultry housing

Particulates in and around animal production sites include soil particles, bits of feed, dried skin, hair or feathers, dried feces, bacteria, fungi, and endotoxins (Koon et al., 1963, Anderson et al., 1966, Curtis et al., 1975b, Heber and Stroik, 1988, Anderson et al., 1966, Curtis et al., 1975a, Heber et al., 1988). Sources include animals, feed storage and processing sites, floors, manure storage and handling equipment, open lots, compost sites, and other elements of animal agriculture systems.

Feed was found to be the primary component of the dust in animal housing (Curtis et al., 1975b, Heber and Stroik, 1988, Heber et al., 1988). Soil particles from open unpaved feedlots also contribute to dust levels (Alegro et al., 1972, Sweeten et al., 1988). Dust emissions from feedlots depend on soil texture, rainfall, feedlot surface moisture content, wind speed, season, and other factors. The white paper on particulate matter emissions from confined animal feeding operations – management and control measures (Auvermann et al., 2002) provides more specific information on dust emission from cattle feedlots.

Flooring design has been shown to significantly affect the airborne dust levels; solid floors have much higher levels than open-mesh floors (Carpenter and Fryer, 1990, Dawson, 1990). The latter allow feces and soiled bedding to fall below the floor level and minimize dust generated by animal activities.

There is little research on dust emission factors from animal agriculture facilities and their environmental impact. Most studies have focused on dust concentrations and characterization in swine (Barber et al., 1991, Maghirang et al., 1997) and poultry (Jones et al., 1984, Carpenter et al., 1986) housing rather than emissions. Limited information is available on dust concentrations in dairy (Clark and McQuitty, 1987a, Hillman et al., 1992) and horse facilities (Navarotto et al., 1994, McGorum et al., 1998). Auvermann et al. (2002) summarize information on particulate matter in swine and poultry housing as well as on open cattle feedlots. Other studies have concentrated on the effects of dust in confinement housing on human worker and animal health (Donham and Gustafson, 1982, Donham et al., 1986). Impacts of particulate matter and bioaerosols on human health are discussed in detail in the white paper on health effects of aerial emissions from animal production and waste management systems (Schiffman et al., 2002).

Wathes et al. (1997) measured dust emissions from broiler and layer facilities in the U.K. Table 11 summarizes the results obtained by Wathes et al. (1997).

Table 11. Emission of dust by poultry houses (Wathes et al., 1997)

Type	Season	Inhalable dust ($g\ AU^{-1}\ h^{-1}$)	Respirable dust ($g\ AU^{-1}\ h^{-1}$)
Layers	Winter	0.9	0.24
Broilers	Winter	5.2	0.60
Layers	Summer	1.1	0.09
Broilers	Summer	8.2	0.88

Takai et al. (1998) reported on inhalable (includes all size particles) and respirable (particles that are less than 5 microns) dust emissions from various cattle, swine, and poultry facilities in four European countries (Table 12). Emissions were estimated from mean daily dust concentrations near air outlets and the daily mean ventilation rate through the buildings.

Table 12. Mean inhalable and respirable dust emission factors from English, Dutch, Danish, and German livestock buildings (Takai et al., 1998).

Species	Mean inhalable dust ($g\ AU^{-1}\ h^{-1}$)	Mean respirable dust ($g\ AU^{-1}\ h^{-1}$)
Cattle Housing (dairy and beef)		
England	0.10	0.03
The Netherlands	0.14	0.04
Denmark	0.13	0.01
Germany	0.18	0.02
Overall mean	0.15	0.02
Swine Housing		
England	0.63	0.09
The Netherlands	0.67	0.07
Denmark	1.10	0.12
Germany	0.65	0.05
Overall mean	0.76	0.09
Poultry Housing		
England	3.14	0.37
The Netherlands	3.64	0.72
Denmark	3.51	0.62
Germany	2.12	0.25
Overall mean	3.19	0.50

Statistical analysis indicated that both country and housing type were significantly different for inhalable dust emissions (Takai et al., 1998), although this could be an artifact from measurement system bias. Inhalable dust emissions from cattle buildings were not affected by season. There were significant seasonal effects on inhalable dust emissions from both swine and poultry housing. The highest dust emissions were from percheries (laying hen facilities with litter flooring and perches) in the Netherlands and Denmark, and from broiler houses in England and the Netherlands (Takai et al., 1998). Animal activity level, stocking density, spilled feed, bedding material selection, and humidity levels affected dust emissions. The significance of country, season and other factors suggests that results from Takai et al. (1998) are unlikely to accurately describe dust emissions from animal buildings in the United States.

ENDOTOXIN

Livestock and poultry housing

Endotoxin is a hazardous component of airborne particulates in animal operations. It arises from the degradation of Gram-negative bacterial cell wall and is ubiquitous in the agricultural environment. Endotoxin is a potent inflammatory agent that produces systemic effects and lung obstruction, even at low levels of exposure (Hoff et al., 2002). Despite a clear recognition that inhaled endotoxin is an occupational hazard in livestock and poultry confinement housing (Kullman et al., 1998, Thorne et al., 1997, Donham et al., 1989), cotton processing, vegetable processing, fiberglass manufacturing, and metal machining environments, there are no established occupational exposure limits in the United States or Canada (Duchaine et al., 2001). This is probably due to the fact that endotoxin exposure assessment methods have not been adequately optimized and validated.

Wathes et al. (1997) measured endotoxin emissions from broiler and layer facilities in the U.K. Endotoxin emissions varied between less than 1 and 10 g AU^{-1} h^{-1} in the winter, and between 30 and 45 g AU^{-1} h^{-1} in the summer.

Seedorf et al. (1998) measured concentrations of airborne endotoxins and microorganisms in cattle, swine, and poultry housing in four European countries (England, The Netherlands, Denmark, and Germany). The emission rates were estimated by using the ventilation rate and the indoor concentration. Estimated endotoxin emission rates in the inhalable and respirable dust fractions from various livestock and poultry housing are summarized in Table 13.

Table 13. Mean emission rates of inhalable and respirable endotoxin over 24 hours from different livestock and poultry housing (Seedorf et al., 1998)

Species	Mean inhalable endotoxin (µg AU^{-1} h^{-1})	Mean respirable endotoxin (µg AU^{-1} h^{-1})
Cows	2.9	0.3
Beef	3.7	0.6
Calves	21.4	2.7
Sows	37.4	3.7
Weaners (growing pigs)	66.6	8.9
Fattening pigs	49.8	5.2
Layers	538.3	38.7
Broilers	817.4	46.7

Data from the Seedorf et al. (1998) study indicate that endotoxin emissions were highest from poultry housing and lowest from cattle facilities. Seedorf et al. (1998) concluded that it was not known whether outdoor human exposure to such endotoxin emissions was hazardous to health.

The same study (Seedorf et al., 1998) reported on total airborne microorganism emissions rates from various livestock and poultry housing. Emissions were reported as the logarithm base 10 of the number of colony forming units (cfu) per hour per 500 kg of live-weight animals housed in the building (Table 14).

Table 14. Livestock and poultry housing microorganism emissions (Seedorf et al., 1998)

Species	Total bacteria (Log cfu AU^{-1} h^{-1})	Enterobacteriacae (Log cfu AU^{-1} h^{-1})	Fungi (Log cfu AU^{-1} h^{-1})
Swine			
Sows	7.7	6.0	6.5
Nursery pigs	7.1	6.9	5.8
Finishing pigs	7.6	6.9	6.1
Poultry			
Layers	7.1	7.1	6.0
Broilers	9.5	6.1	7.8
Cattle			
Dairy cows	6.8	6.2	6.0
Beef	6.7	6.2	5.9
Calves	7.3	6.1	6.5

Seedorf et al. (1998) noted that data on the biological half-life period of viable microorganisms under varying environmental conditions was needed in order to predict their dispersion and estimate the risk of airborne disease transmission. Local topography, weather, and ventilation system design also affect potential contaminant transmission.

There is no information on emission of endotoxins related to manure storage, land application and treatment systems.

CONCLUSIONS

Substantial research has been conducted to quantify the air quality and emission rates from livestock and poultry facilities and waste management systems. Much of the work related to emission rates was conducted in Europe over the past decade; more recently, work conducted in the U.S. has begun to be published. Considerable literature to quantify air quality, in terms of odor, dust, and gas emissions exists and has been cited in this paper.

The work summarized in this paper shows substantial variability in some measurements, such as odor and NH_3 emission rates. In part this variability is inherent in the livestock and poultry production systems, and in part is due to external influences including regional climatic differences, housing or storage facility differences, management practices and variable diets. However, a generally unreported contribution to the variability in the literature is from use of differing measurement methods and equipment. Depending on how emission levels are to be used, caution is recommended since even an “average” value may under or over estimate a specific building or manure management system emission. It seems most prudent to develop a database of emission rates or factors for various dependent variables such as housing system, location (by region in U.S. for instance), and species. This would assure that the best estimates for emission of odor, gases, and particulates are obtained for a given situation.

From this review of the literature, there is seen to be a clear need for the development and use of standard methods for measuring emission rates of odor, dust and gases from livestock and poultry facilities. While these methods may exist and may be applied to industrial and municipal

waste management systems, the research communities involved in the work cited here do not generally follow a common method.

As countries move toward regulation of the various gases and compounds emitted from poultry and livestock facilities, and their waste storage systems, it will be increasingly important to have an understanding of what factors can be manipulated to provide cost-effective reductions in emissions.

REFERENCE LIST

Aarnink, A.J.A., Keen, A., Metz, J.H.M., Speelman, L. and Verstegen, M.W.A. 1995. Ammonia emission patterns during the growing periods of pigs housed on partially slatted floors. *Journal of Agricultural Engineering Research* **62**, 105-116.

ACGIH. 1992. Threshold Limit Values for Chemical Substances and Physical Agents and Biological Exposure Indices. American Conference of Governmental Industrial Hygienists, Cincinnati, OH.

Alegro, J.W., C.J. Elam, C.J., Martinez, A. and Westing, T. 1972. Feedlot air, water and soil analysis. Bulletin D. How to control feedlot pollution. Bakersfield, CA: California Cattle Feeders Association.

Anderson, D.P., Beard, C.W. and Hanson, R.P. 1966. Influence of poultry house dust, ammonia, and carbon dioxide on the resistance of chickens to Newcastle disease virus. *Avian Diseases* **10** (2):177-188.

Aneja, V.P., Chauhan, J.P. and Walker, J. 2000. Characterization of atmospheric ammonia emissions from swine waste storage and treatment lagoons. *Journal of Geophysical Research* **105**, 11535-11545.

Arogo, J., Zhang, R.H., Riskowski, G.L. and Day, D.L. 2000. Hydrogen sulfide production from stored liquid swine manure: a laboratory study. *Transactions of the ASAE* **43**, 1241-1245.

Arogo, J., Westerman, P.W., Heber, A.J., Robarge, W.P., and Classen, J.J. 2002. Ammonia emissions from animal feeding operations. *National Center for Manure and Animal Waste Management White Papers*, North Carolina State University, Raleigh, NC (available on CD-ROM from MidWest Plan Service), 67 p.

Auvermann, B.W., Bottcher, R.W., Heber, A.J., Meyer, D. Parnell, C.B., Shaw, B., and Worley, J. 2002. Particulate matter emissions from confined animal operations: management and control measures. *National Center for Manure and Animal Waste Management White Papers*, North Carolina State University, Raleigh, NC (available on CD-ROM from MidWest Plan Service), 41 p.

Barber, E.M., Dawson, J.R., Battams, V.A. and Nicol, R.A. 1991. Spatial variability of airborne and settled dust in a piggery. *Journal of Agricultural Engineering Research* **50**, 107-127.

Battye, R., Battye, W., Overcash, C. and Fudge, S. 1994. Development and selection of ammonia emission factors. EPA 68-D3-0034, U.S. EPA Atmospheric Research and Exposure Assessment Laboratory, Research Triangle Park, NC.

Beline, F. and Martinez, J. 2002. Nitrogen transformations during biological aerobic treatment of pig slurry: effect of intermittent aeration on nitrous oxide emissions. *Biores.Technol.* **83**, 225-228.

Bicudo, J.R., Classen, J.J., Goldsmith, C.D. and Smith, T. 1999. Effects of aeration cycles and hydraulic retention time on the sequencing batch treatment of flushed swine manure. *Advances in Environmental Research* **3**, 58-73.

Bicudo, J.R., Schmidt, D.R., Powers, W., Zahn, J.A., Tengman, C.L., Clanton, C.J., and Jacobson, L.D. 2002. Odor and VOC emissions from swine manure storages. *Procs. of Odor and Toxic Air Emissions 2002*, Albuquerque, NM, April 28 to May 1, WEF, Alexandria, VA (on CD-ROM).

Bicudo, J.R., Tengman, C.L., Jacobson, L.D., and Sullivan, J.E. 2000. Odor, hydrogen sulfide and ammonia emissions from swine farms in Minnesota. *Procs. of Odors and VOC Emissions 2000*, Cincinnati, OH, April 16 to 19, WEF, Alexandria, VA (on CD-ROM).

Braam, C.R., Smits, M.C.J., Gunnink, H. and Swierstra, D. 1997. Ammonia emission from a double-sloped solid floor in a cubicle house for dairy cows. *Journal of Agricultural Engineering Research* **68**, 375-386.

Brewer, S.K. and Costello, T.A. 1999. In situ measurements of ammonia volatilization from broiler litter using an enclosed air chamber. *Transactions of the ASAE* **42**, 1415-1422.

Brown, H.A., Wagner-Riddle, C., and Thurtell, G.W. 2000. Nitrous oxide flux from solid dairy manure in storage as affected by water content and redox potential. *J. Environ. Qual.* **29**(2):630-638.

Brumm, M.C., Harmon, J.D., Honeyman, M.C. and Klibenstein, J.B. 1997. Hoop structures for grow-finished swine. MidWest Plan Service AED-41, Ames, IA.

Burton, C.H. 1997. *Manure management - treatment strategies for sustainable agriculture*. Silsoe Research Institute, Silsoe, Bedford, UK.

Burton, C.H., Sneath, R.W. and Farrent, J.W. 1993. Emissions of nitrogen oxide gases during aerobic treatment of animal slurries. *Biores.Technol.* **45**, 233-235.

Burton, C.H., Sneath, R.W., Misselbrook, T.H. and Pain, B.F. 1998. The effect of farm scale aerobic treatment of piggery slurry on odour concentration, intensity and offensiveness. *J.agric.Engng Res.* **71**, 203-211.

Cahn, T.T., Verstergen, M.W.A., Aarnink, A.J.A., and Schrama, J.W. 1997. Influence of dietary factors on nitrogen partitioning and composition of urine and feces of fattening pigs. *Journal of Animal Science* **75**:700-706.

Cahn, T.T., Aarnink, A.J.A., Schulte, J.B., Sutton, A., Langhout, D.J., and Verstergen, M.W.A. 1998. Dietary protein affects nitrogen excretion and ammonia emission from slurry of growing-finishing pigs. *Livestock Production Science* **56**:181-191.

Carpenter, G.A. and Fryer, J.T. 1990. Air filtration in a piggery: filter design and dust mass balance. *Journal of Agricultural Engineering Research* **46** (3):171-186. ill.

Carpenter, G.A., Smith, E.K., MacLaren, A.P.C. and Spackman, D. 1986. Effect of internal air filtration on the performance of broilers and the aerial concentrations of dust and bacteria. *British Poultry Science* **27** (3):471-480.

CEN. 1999. Draft prEN 13725. CEN Central Secretariat, Brussels, Belgium.

Chadwick, D., Misselbrook, T., and Pain, B.F. 2000. Is Europe reducing its ammonia emissions at the expense of the global environment? *Procs. of the 2nd International Conference on Air Pollution from Agricultural Operations*, Des Moines, IA, October 9-11, ASAE, St. Joseph, MI, 1-9.

Chadwick, D.R., Sneath, R.W., Phillips, V.R. and Pain, B.F. 1999. A UK inventory of nitrous oxide emissions from farmed livestock. *Atmospheric Environment* **33**, 3345-3354.

Chadwick, D.R., van der Weerden, T., Martinez, J. and Pain, B.F. 1998. Nitrogen transformations and losses following pig slurry applications to a natural soil filter system (Solepur process) in Brittany, France. *J.agric.Engng Res.* **69**, 85-93.

Chandler, J.A., Hermes, S.K., and Smith, K.D. 1983. A low cost 75 kW covered lagoon biogas system. *Procs. of Energy from Biomass and Wastes VII*, Lake Buena Vista, FL.

Clanton, C.J. and Schmidt, D.R. 2000. Sulfur compounds in gases emitted from stored manure. *Transactions of the ASAE* **43**, 1229-1239.

Clark, P.C. and McQuitty, J.B. 1987. Air quality in six Alberta commercial free-stall dairy barns. *Canadian Engineering Agriculture* **29**, 77-80.

Curtis, S.E., Drummond, J.G., Grunloh, D.J., Lynch, P.B. and Jensen, A.H. 1975a. Relative and qualitative aspects of aerial bacteria and dust in swine houses. *Journal of Animal Science* **41** (5):1512-1520.

Curtis, S.E., Drummond, J.G., Kelley, K.W., Grunloh, D.J., Meares, V.J., Norton, H.W. and Jensen, A.H. 1975b. Diurnal and annual fluctuations of aerial bacterial and dust levels in enclosed swine houses. *Journal of Animal Science* **41** (5):1502-1511.

Dawson, J.R. 1990. Minimizing dust in livestock buildings: possible alternatives to mechanical separation. *Journal of Agricultural Engineering Research* **47** (4):235-248.

Demmers, T.G.M., Burgess, L.R., Short, J.L., Phillips, V.R., Clark, J.A. and Wathes, C.M. 1998. First experiences with methods to measure ammonia emissions from naturally ventilated cattle buildings in the UK. *Atmospheric Environment* **32**, 285-293.

Demmers, T.G.M., Burgess, L.R., Short, J.L., Phillips, V.R., Clark, J.A. and Wathes, C.M. 1999. Ammonia emission from two mechanically ventilated UK livestock buildings. *Atmospheric Environment* **33**, 217-227.

Dewes, T. 1999. Ammonia emissions during the initial phase of microbial degradation of solid and liquid cattle manure. *Bioresource Technology* **70**, 245-248.

Dillon, P.J. and Molot, L.A. 1989. The role of ammonium and nitrate retention in the acidification of lakes and forested catchments. In: *The role of nitrogen in the acidification of soils and surface waters* (Malanchuk, J.L. and Nilsson, J., eds.), Nordic Council of Ministers, Kopenhagen, DK, Appendix A 1-25.

Donham, K., Haglind, P., Peterson, Y., Rylander, R. and Belin L. 1989. Environmental and health studies of farm workers in Swedish swine confinement buildings. *British Journal of Industrial Medicine* **46** (1):31-37.

Donham, K.J. and Gustafson, K.E. 1982. Human occupational hazards from swine confinement. *Ann. Am. Conf. Ind. Hyg* **2** 137-142.

Donham, K.J., Scallon, L.J., Popendorf, W., Treuhaft, M.W. and Roberts, R.C. 1986. Characterization of dusts collected from swine confinement buildings. *American Industrial Hygiene Association Journal* **47**, 404-410.

Donham, K.J., Yeggy, J. and Dague, R.R. 1988. Production rates of toxic gases from liquid swine manure: health implication for workers and animals in swine confinement buildings. *Biological Wastes* **24** (3):161-173.

Duchaine, C., Thorne, P.S., Meriaux, A., Grimard, Y., Whitten, P. and Cormien, Y. 2001. Comparison of endotoxin exposure assessment by bioaerosol impinger and filter sampling methods. *Applied and Environmental Microbiology* **67** , 2775-2780.

Edwards, G.C., Neumann, H.H., den Hartog, G., Thurtell, G.W., and Kidd, G.E. 1994. Eddy correlation measurements of methane fluxes using a tunable diode laser at the Kinosheo Lake Tower site during the Northern Wetland Study (NOWES). *J. Geophys. Res*. **99**:1511-1517.

Eghball, B., Power, J.F., Gilley, J.E. and Doran, J.W. 1997. Nutrient, carbon, and mass loss during composting beef cattle feedlot manure. *Journal of Environmental Quality* **26**, 189-193.

Eklund, B, LaCosse, J. 1998. Field Measurement of Greenhouse Gas Emissions Rates and Development of Emission Factors for Wastewater. EPA/600/SR-97/094.

Eklund, B. 1992. Practical guidance for flux chamber measurements of fugitive volatile organic emission rates. *Journal of Air & Waste Management Association* 42:1583-1591.

Ellenberg, H. 1988. Eutrophication-changes in woodland vegetation-effects of browsing behaviour of roe deer and consequences for vegetation. *Schweizer Zeitschrift fur Forstwesen* **139** 261-282.

Ellenberger, M.A., Rumpler, W.V., Johnson, D.E., and Goodall, S.R. 1985. Evaluation of the extent of ruminal urease inhibition by sarsaponin and sarsaponin fractions. *J. Anim. Sci.* **61** (Suppl. 1):491-498.

Ferguson, N.S., Gates, R.S., Taraba, J.L., Cantor, A.H., Pescatore, A.J., Straw, M.L., Ford, M.J., and Burnham, D.J. 1998a. The effect of dietary protein and phosphorus on ammonia concentration and litter composition in broilers. *Poultry Science* **77**:1085-1093.

Ferguson, N.S., Gates, R.S., Taraba, J.L., Cantor, A.H., Pescatore, A.J., Ford, M.J., and Burnham, D.J. 1998b. The effect of dietary crude protein on growth, ammonia concentration, and litter composition in broilers. *Poultry Science* **77**:1481-1487.

Fowler, D., Pitcairn, C.E.R., Sutton, M.A., Flechard, C., Loubet, B., Coyle, M. and Munro, R.C. 1998. The mass budget of atmospheric ammonia in woodland within 1 km of livestock buildings. *Environmental Pollution* **102** (S1):343-348.

Fowler, D. and Duyzer, J.H. 1989. Micrometeorological techniques for the measurement of trace gas exchange. In *Exchange of trace gases between terrestrial ecosystems and the atmosphere*, Dahlem Workshop Proceedings, John Wiley and Sons, Berlin, 198-208.

Fulhage, C. and Hoehne, J. 1999. A manure nutrient profile of 100 Missouri swine lagoons. Paper presented at the *1999 ASAE/CSAE Annual International Meeting*, July 18-21, Toronto, Canada, Paper No. 994026, ASAE, St. Joseph, MI.

Gates, R.S., Overhults, D.G. and Zhang, S.H. 1996. Minimum ventilation for modern broiler facilities. *Transactions of the ASAE* **39**, 1135-1144.

Gates, R.S., Taraba, J.L., Liberty, K., Pescatore, A.J., Cantor, A.H., Ford, M.J., and Burnham,D.J. 2000. Dietary manipulation for reduced ammonia emission and TAN in broiler litter. *Procs. of the 2nd International Conference on Air Pollution from Agricultural Operations*, October 9-11, Des Moines, IA, 147-156.

Gay, S.W., Clanton, C.J., Schmidt, D.R., Janni, K.A., Jacobson, L.D. and Weisberg, S. 2002. Odor, total reduced sulfur, and ammonia emissions from livestock and poultry buildings and manure storage units. *ASAE Applied Engineering in Agriculture* (Submitted).

Gholson, A.R., J.R. Albritton, R.K.M. Jayanty. 1991. Evaluation of an enclosure method for measuring emissions of volatile organic compounds from quiescent liquid surfaces. Environ. Sci. Technol. 25:519-524.

Gibson, M.L., Preston, R.L., Pritchard, R.H., and Goodall, S.R. 1985. Effect of sarsaponin and monensin on ruminal ammonia levels and *in vitro* dry matter digestibilities. *J. Anim. Sci.* **61**(Suppl. 1):492.

Gilley, J.E., Spare, D.P., Koelsch, R.K., Schulte, D.D., Miller, P.S., and Parkhurst, A.M. 2000. Phototrophic anaerobic lagoons as affected by copper and zinc in swine diets. *Transactions of the ASAE*, 43(6):1853-1859.

Groenestein, C.M. and Faassen, H.G.v. 1996. Volatilization of ammonia, nitrous oxide and nitric oxide in deep-litter systems for fattening pigs. *Journal of Agricultural Engineering Research* **65** (4):269-274.

Groot Koerkamp, P.W.G. 1994. Review on emissions of ammonia from housing systems for laying hens in relation to sources, processes, building design and manure handling. *Journal of Agricultural Engineering Research* **59** (2):73-87.

Groot Koerkamp, P.W.G., Metz, J.H.M., Uenk, G.H., Phillips, V.R., Holden, M.R., Sneath, R.W., Short, J.L., White, R.P., Hartung, J. and Seedorf, J. 1998a. Concentrations and emissions of ammonia in livestock buildings in Northern Europe. *Journal of Agricultural Engineering Research* **70** (1):79-95.

Groot Koerkamp, P.W.G., Speelman, L. and Metz, J.H.M. 1998b. Litter composition and ammonia emission in aviary houses for laying hens. 1. Performance of a litter drying system. *Journal of Agricultural Engineering Research* **70** (4):375-382.

Hanna, H.M., Bundy, D.S., Lorimor, J.C., Mickelson, S.K., Melvin, S.W. and Erbach, D.C. 2000. Manure incorporation equipment effects on odor, residue cover, and crop yield. *ASAE Applied Engineering in Agriculture* **16**, 621-627.

Hao, X., Chang, C., Larney, F.J. and Travis, G.R. 2001. Greenhouse gas emissions during cattle feedlot manure composting. *Journal of Environmental Quality* **30**, 376-386.

Harper, L.A. 1988. Comparisons of methods to measure ammonia volatilization in the field. In Ammonia volatilization from urea fertilizers (B.R. Bock and D.E. Kissel, eds.), Tennessee Valley Authority, Muscle Shoals, AL, 93-109.

Harper, L.A. and Sharpe, R.R. 1998. Ammonia emissions from swine waste lagoons in the southeastern U.S. coastal plains. Final Report, Div. of Air Quality, N.C. Dept. of Environment and Natural Resources, Raleigh, NC.

Harper, L.A., Sharpe, R.R. and Parkin, T.B. 2000. Gaseous nitrogen emissions from anaerobic swine lagoons: ammonia, nitrous oxide, and dinitrogen gas. *Journal of Environmental Quality* **29**, 1356-1365.

Hashmonay, R.A., Natschke, D.F., Wagoner, K., Harris, D.B., Thompson, E.L., and Yost, M.G. 2001. Field evaluation of a method for estimating gaseous fluxes from area sources using open-path Fourier transform infrared. *Environm. Sci. & Technol.* **35**(11):2309-2313.

Hartung, J. and Phillips, V.R. 1994. Control of gaseous emissions from livestock buildings and manure stores. *J.agric.Engng Res.* **57**, 173-189.

Heber, A.J. and Stroik, M. 1988. Influence of environmental factors on concentrations and inorganic of aerial dust in swine finishing houses. *Transactions of the ASAE* **31** (3):875-881.

Heber, A.J., Stroik, M., Faubion, J.M. and Willard, L.H. 1988. Size distribution and identification of aerial dust particles in swine finishing buildings. *Transactions of the ASAE*

31 (3):882-887.

Hill, D.T. and Bolte, J.P. 1989. Digester stress as related to isobutyric and isovaleric acids. *Biological Wastes* **28**, 33-37.

Hillman, P., Gebremedhin, K. and Warner, R. 1992. Ventilation system to minimize airborne bacteria, dust, humidity, and ammonia in calf nurseries. *Journal of Dairy Science* **75**, 1305-1312.

Hinz, T. and Linke, S. 1998. A comprehensive experimental study of aerial pollutants in and emissions from livestock buildings - part 2: results. *Journal of Agricultural Engineering Research* **70**, 119-129.

Hobbs, P.J., Misselbrook, T.H. and Cumby, T.R. 1999. Production and emission of odours and gases from ageing pig waste. *J.agric.Engng Res.* **72**, 291-298.

Hoff, S.J., Hornbuckle, K.C., Thorne, P.S., Bundy, D.S. and O'Shaughnessy, P.T. 2002. Emissions and community exposures from CAFOs. Final Report, Iowa State University and University of Iowa Study Group, Ames, IA.

Houghton, J.T., Callander, B.A. and Varney, S.K. 1992. The Supplementary Report to the IPPC Scientific Assessment, Climate Change. Cambridge University Press, New York, NY.

Hutchinson, G.L., Mosier, A.R., and Andre, C.E. 1982. Ammonia and amine emissions from a large cattle feedlot. *J. Environ. Qual*. **11**(2):288-293.

James, T., Meyer, D., Esparza, E., Depeters, J., and Perez-Monti, H. 1999. Effects of dietary nitrogen manipulation on ammonia volatilization from manure from Holstein heifers. *Journal of Dairy Science* **82**:2430-2439.

Jeppsson, K.H. 1999. Volatilization of ammonia in deep-litter systems with different bedding material for young cattle. *J. Agric. Eng. Res.* **73**, 49-57.

Jiang K, Bliss PJ, Schulz TJ. 1995. The development of a sampling system for determining odor emission rates from areal surfaces: Part 1. Aerodynamic performance. *Journal of Air and Waste Management Association* **45**:917-22.

Jones, W., Morring, K., Olenchock, S.A., Williams, T. and Hickey, J. 1984. Environmental study of poultry confinement buildings. *American Industrial Hygiene Association Journal* **45** (11):760-766.

Kaharabata, S.K. and Schuepp, P.H. 2000. Estimating methane emissions from dairy cattle housed in a barn and feedlot using an atmospheric tracer. *Environmental Science and Technology* **34**, 3296-3302.

Kemme, P.A., Jongbloed, A.W., Dellart, B.M., and Krol-Kramer, F. 1993. The use of *Yucca schidigera* extract as a "urease inhibitor" in pig slurry. In: *Nitrogen flow in pig production*

and environmental consequences (M.W.A. Verstegen, L.A. den Hartog, G.J.M. van Kempen, and J.H.M. Metz, eds.). EAAP Publ. No. 69, Wageningen, The Netherlands, 330-335.

Kirschgessner, D.A., Piccot, S.D., and Chadha, A. 1993. Estimation of methane emissions from a surface coal mine using open path FTIR spectroscopy and modeling techniques. *Chemosphere* **26** (1-4):23-30.

Klopfenstein, T.J. and Erickson, G.E. 2001. Effects of manipulating protein and phosphorus nutrition of feedlot cattle on nutrient management and the environment. *Journal of Animal Science* **79**:97-103.

Koelliker, J.K. and Miner, J.R. 1973. Desorption of ammonia from anaerobic lagoons. *Transactions of the ASAE* **16**, 148-151.

Koelsch, R. K., Woodbury, B., Stenberg, D., Miller, D. N., and Schulte, D. 2001. Total reduced sulfur concentration in beef cattle feedlots. *Procs. of the International Symposium addressing Animal Production and Environmental Issues*, Raleigh, NC, October 3-5, North Carolina State University, Raleigh, NC (on CD-ROM).

Koon, J., Howes, J.R., Grub, W. and Rollo, C.A. 1963. Poultry dust: origin and composition. *Agricultural Engineering* **44** (11):608-609.

Kreis, R.D. 1978. Control of animal production odors: the state-of-the-art. U.S. Environmental Protection Agency, Office of Research and Development Ada, OK.

Kroodsma, W., Huis in't Veld, J.W.H. and Scholtens, R. 1993. Ammonia emission and its reduction from cubicle houses by flushing. *Livestock production Science* **35**, 293-302.

Kullman, G.J., Thorne, P.S., Waldron, P.F., Marx, J.J., Ault, B., Lewis, D.M., Siegel, P.D., Olenchock, S.A. and Merchant, J.A. 1998. Organic dust exposures from work in dairy barns. *American Industrial Hygiene Association Journal* **48**, 436-445.

Kuroda, K., Osada, T., Yonaga, M., Kanematu, A., Nitta, T., Mouri, S. and Kojima, T. 1996. Emissions of malodorous compounds and greenhouse gases from composting swine feces. *Biores.Technol.* **56**, 265-271.

Lim, T.T., Heber, A.J., Ni, J.-Q., Sutton, A.L. and Kelly, D.T. 2002. Characteristics and emission rates of odor from commercial swine nurseries. *Transactions of the ASAE* (In Press)

Lockyer, D.R. 1984. A system for the measurement in the field of losses of ammonia through volatilisation. *J. Sci. Food Agric.* **35**:837-848.

Loubet, B., P. Cellier, D. Flura and S. Genermont. 1999a. An evaluation of the wind-tunnel technique for estimating ammonia volatilization from land: Part 1. Analysis and improvement of accuracy. *Journal of Agricultural Engineering Research* **72**:71-81.

Loubet, B., P. Cellier, D. Flura and S. Genermont. 1999b. An evaluation of the wind-tunnel technique for estimating ammonia volatilization from land: Part 2. Influence of the tunnel on transfer processes. *Journal of Agricultural Engineering Research* **72**:83-92.

Maghirang, R.G., Puma, M.C., Liu, Y. and Clark, P. 1997. Dust concentrations and particle distribution in an enclosed swine nursery. *Transactions of the ASAE* **40**, 749-754.

Martinez, J. 1997. Solepur: a soil treatment process for pig slurry with subsequent denitrification of drainage water. *J.agric.Engng Res.* **66**, 51-62.

McGahan, E.J., Nicholas, P.J., Watts, P.J., Galvin, G., Lowe, S., Stepnuk, L.M. and Casey, K.D. 2001. The effect of loading rate and spatial variability on pond odor emission. APL Project No. 1628, PRDC Progress Report, FSA Environmental, Toowoomba, Australia.

McGorum, B.C., Ellison, J. and Cullen, R.T. 1998. Total and respirable airborne dust endotoxin concentrations in three equine management systems. *Equine Veterinary* **30**, 430-434.

McQuitty, J.B., Feddes, J.J.R. and .Leonard, J.J. 1985. Air quality in commercial laying barns. *Canadian Agricultural Engineering* **27** (2):13-19.

Meisinger, J.J., A.M. Lefcourt, R.B. Thompson. 2001. Construction and validation of small mobile wind tunnels for studying ammonia volatilization. *Applied Engineering in Agriculture* **17**(3):375-381.

Misselbrook, T.H., Chadwick, D.R., Hobbs, P.J., and Pain, B.F. 1997. Control by dietary manipulation of emissions from pig slurry following land spreading. *Procs. of the International Symposium on Ammonia and Odour Control from Animal Production Facilities*, Vinkeloord, The Netherlands, 261-266.

Misselbrook, T.H., Clarkson, C.R. and Pain, B.F. 1993. Relationship between concentration and intensity of odours for pig slurry and broiler houses. *Journal of Agricultural Engineering Research* **55**, 163-169.

Misselbrook, T.H., Van der Weerden, T.J., Pain, B.F., Jarvis, S.C., Chambers, B.J., Smith, K.A., Phillips, V.R. and Demmers, T.G.M. 2000. Ammonia emission factors for UK agriculture. *Atmospheric Environment* **34**, 871-880.

Natschke, D., Hashmonay, R.A., Wagoner, K., Harris, D.B., Thompson, E.L., and Vogel, C.A. 2001. Seasonal emissions of ammonia and methane from a hog waste lagoon with a bioactive cover. *Procs. of the International Symposium addressing Animal Production and Environmental Issues*, Raleigh, NC, October 3-5, North Carolina State University, Raleigh, NC (on CD-ROM).

Navarotto, P., Guarino, M. and Santambrogio, A. 1994. Evaluation of the environmental dust and mycetes in a thoroughbred stable. *Transactions of the ASAE* **37**, 229-233.

Ni, J.-Q., Heber, A.J., Diehl, C.A. and Lim, T.T. 2000. Ammonia, hydrogen sulphide and carbon dioxide release from pig manure in under-floor deep pits. *J. Agric. Eng. Res.* **77**, 53-

66.

Ni, J.-Q., Heber, A.J., Diehl, C.A., Lim, T.T., Duggirala, R.K. and Haymore, B.L. 2002. Summertime concentrations and emissions of hydrogen sulfide at a mechanically-ventilated swine finishing building. *Transactions of the ASAE* (In Press)

Ni, J.-Q., Heber, A.J., Lim, T.T., Diehl, C.A., Duggirala, R.K., Haymore, B.L. and Sutton, A.L. 2000. Ammonia emission from a large mechanically-ventilated swine building during warm weather. *J. Environ. Qual.* **29**, 751-758.

Nihlgard, B. 1985. The ammonium hypothesis: An additional explanation to the forest dieback in Europe. *Ambio* **14** (1):2-8. ill.

O'Neill, D.H. and Phillips, V.R. (1992) A review of the control of odour nuisance from livestock buildings: Part 3, Properties of the odorous substances which have been identified in livestock wastes or in the air around them. *J.agric.Engng Res.* **53**, 23-50.

Osada, T., Rom, H.B. and Dahl, P. 1998. Continuous measurement of nitrous oxide and methane emission in pig units by infrared photoacoustic detection. *Transactions of the ASAE* **41**, 1109-1114.

Pahl, O., Burton, C.H., Dunn, W. and Biddlestone, A.J. 2001. The source and abatement of nitrous oxide emissions produced from the aerobic treatment of pig slurry to remove surplus nitrogen. *Environmental Technology* **22**, 941-950.

Pain BF, Rees YJ, Lockyer DR. 1988. Odour and ammonia emmision following the application of pig or cattle slurry to land. *In Volatile emissions from livestock farming and sewage operations* (Nielson VC, Voorburg JH, L'Hermite P, Editors), Elsevier Applied Science Publishers, UK, 2-11.

Pain, B.F., Clarkson, C.R., Phillips, V.R., Klareneek, J.V., Misselbrook, T.H. and Bruins, M. 1991. Odour emission arising from application of livestock slurries on land: measurements following spreading using a micrometeorological technique and olfactometry. *Journal of Agricultural Engineering Research* **48** 101-110.

Pain, B.F. and Misselbrook, T.H. 1991. Relationships between odour and ammonia emission during and following application of slurries to land. In: *Odour and ammonia emissions from livestock farming* (Nielsen, V.P., Voorburg, J.H. and L'Hermite, P., eds.), Elsevier Applied Science, New York, NY, 2-9.

Pain, B.F., Misselbrook, T.H., Clarkson, C.R. and Rees, Y.J. 1990. Odour and ammonia emissions following the spreading of anaerobically-digested pig slurry on grassland. *Biological Wastes* **34** (3):259-267.

Pain, B.F., Van der Weerden, T.J., Chambers, B.J., Phillips, V.R. and Jarvis, S.C. 1998. A new inventory for ammonia emissions from UK agriculture. *Atmospheric Environment* **32**, 309-313.

Patni, N.K. and Clarke, S.P. 1991. Transient hazardous conditions in animal buildings due to manure gas released during slurry mixing. *Applied Engineering in Agriculture* **7** (4):478-484.

Phillips,V.R., Holden, M.R., Sneath,R.W., Short, J.L., White, R.P., Hartung, J., Seedorf, J., Schroder, M., Linkert, K.H., Pedersen, S., Takai, H., Johnsen, J.O., Groot Koerkamp, P.W.G., Uenk, G.H., Scholtens, R., Metz, J.H.M., and Wathes, C.M. 1998. The development of robust methods for measuring concentrations and emission rates of gaseous and particulate air pollutants in livestock buildings. *J. Agric. Eng. Res*. **70**:11-24.

Phillips, F.R., R. Scholotens, D.S. Lee, J.A. Garland, R.W. Sneath. 2000. A review of methods for measuring emission rates of ammonia from livestock buildings and slurry or manure stores, Part 1: Assessment of basic approaches. *J. Agric. Eng. Res*. **77**(4):355-364.

Piccot, S.D., Masemore, S.S., Lewis-Bevan, W., Ringler, E.S., and Harris, D.B. 1996. Field assessment of a new method for estimating emission rates from volume sources using open path FTIR spectroscopy. *J. Air & Waste Mgt. Assoc.* **46**:159-165.

Pitcairn, C.E.R., Leith, I.D., Shappard, L.J., Sutton, M.A., Fowler, D., Munro, R.C., Tang, S. and Wilson, D. 1998. The relationship between nitrogen deposition, species composition and foliar nitrogen concentrations in woodland flora in the vicinity of livestock farms. *Environmental Pollution* **102** (S1):41-48.

Powers, W.J., Wilkie, A.C., Van Horn, H.H. and Nordstedt, R.A. 1997. Effects of hydraulic retention time on performance and effluent odor of conventional and fixed-film anaerobic digesters fed dairy manure wastewaters. *Transactions of the ASAE* **40** (5):1449-1455.

Pratt, E.V., Rose, S.P. and Keeling, A.A. 2002. Effect of ambient temperature on losses of volatile nitrogen compounds from stored laying hen manure. *Biores.Technol.* **84**, 203-205.

Reinhart, D.R. and D.C. Cooper. 1992. Flux chamber design and operation for the measurement of municipal solid waste landfill gas emission rates. *Journal of Air & Waste Management Association* **42**:1067-1070.

Roelofs, J.G.M., Kempers, A.J., Houdijk, A.L.F.M. and Jansen, J. 1985. The effect of airborne ammonium sulfate on pinus nigra var. maritima in the Netherlands. *Plant and Soil* **84** 45-56.

Ryden, J.C., D. R. Lockyer. 1985 Evaluation of a system of wind tunnels for field studies of ammonia loss from grassland through volatilization. *J. Sce. Food Agric*. **36**:781-788.

Safley, L.M., Barker, J.C. and Westerman, P.W. 1992. Loss of nitrogen during sprinkler irrigation of swine lagoon liquid. *Biores.Technol.* **40**, 7-15.

Safley, L.M. and Casada, M.E. 1992. Global Methane Emissions from Livestock and Poultry Manure. U.S. Environmental Protection Agency, Report 400/1-91/048, Washington, DC.

Safley, L.M. and Westerman, P.W. 1988. Biogas production from anaerobic lagoons. *Biological Wastes* **23**, 181-193.

Safley, L.M. and Westerman, P.W. 1989. Anaerobic lagoon biogas recovery system. *Biological Wastes* **27**, 43-62.

Schiffman, S. S., Auvermann, B.W., and Bottcher, R.W. 2002. Health effects of aerial emissions from animal production waste management systems. *National Center for Manure and Animal Waste Management White Papers*, North Carolina State University, Raleigh, NC (available on CD-ROM from MidWest Plan Service), 45 p.

Schiffman, S.S., Bennett, J.L. and Raymer, J.H. 2001b. Quantification of odors and odorants from swine operations in North Carolina. *Agricultural and Forest Meteorology* **108**, 213-240.

Schmidt, D.R., J.R. Bicudo and K.A. Janni. 1999. Determining odor emissions from manure surfaces using a wind tunnel. ASAE Meeting Paper #994033, St. Joseph, MI.

Schmidt, D.R. and J.R. Bicudo. 2002. Using a wind tunnel to determine odor and gas fluxes from manure surfaces. ASAE Meeting Paper #024083, St. Joseph, MI.

Seedorf, J., Hartung, J., Schroder, M., Linkert, K.H., Phillips, V.R., Holden, M.R., Sneath, R.W., Short, J.L., White, R.P. and Pedersen, S. 1998. Concentrations and emissions of airborne endotoxins and microorganisms in livestock buildings in Northern Europe. *Journal of Agricultural Engineering Research* **70** (1):97-109.

Sharpe, R.R. and Harper, L.A. 1999. Methane emissions from an anaerobic swine lagoon. *Atmospheric Environment* **33**:3627-3633.

Sharpe, R.R. and Harper, L.A. 2002. Nitrous oxide and ammonia fluxes in a soybean field irrigated with swine effluent. *Journal of Environmental Quality* **31**, 524-532.

Sharpe, R.R., Harper, L.A., and Byers, F.M. 2002. Methane emissions from swine lagoons in Southeastern US. *Agriculture Ecosystems & Environment* **90**:17-24.

Shurson, J., Whitney, M. and Nicolai, R. 1998. Nutritional Manipulation of Swine Diets to Reduce Hydrogen Sulfide Emissions. Final Report to Minnesota Department of Agriculture, Department of Animal Sciences, University of Minnesota, St. Paul, MN.

Simmons, J.D., Hannigan, T.E., and Lott, B.D. 1998. A portable anemometer to determine the output of large in-place ventilation fans. *Applied Engineering in Agriculture* 14(6):649-653.

Simmons, J.D., Lott, B.D., and Hannigan, T.E. 1998. Minimum distance between ventilation fans in adjacent walls of tunnel ventilated broiler houses. *Applied Engineering in Agriculture* 14(5):533-535.

Smith, K.A., Jackson, D.R., Misselbrook, T.H., Pain, B.F. and Johnson, R.A. 2000. Reduction of ammonia emission by slurry application techniques. *J. Agric. Eng. Res.* **77**(3):277-287.

Smith, R.J., Dalton, P.A. and DeBruyn, J. 1999. Assessment and reduction of the odour impact of piggeries. Pig Research and Development Corporation, Canberra, Australia.

Smith RJ, Watts PJ. 1994a. Determination of odour emission rates from cattle feedlots: Part 1, a review. *Journal of Agricultural Engineering Research* **57**:145-55.

Smith RJ, Watts PJ. 1994b. Determination of odour emission rates from cattle feedlots: Part 2, evaluation of two wind tunnels of different size. *Journal of Agricultural Engineering Research* **58**:231-40.

Smits, M.C.J., Valk, H., Elzing, A., and Keen, A. 1995. Effect of protein nutrition on ammonia emission from a cubicle house for dairy cattle. *Livestock Production Science* **44**:147-156.

Sneath, R.W. 1988. The effects of removing solids from aerobically treated piggery slurry on the VFA levels during storage. *Biological Wastes* **26**, 175-188.

Sommer, S.G. 2001. Effect of composting on nutrient loss and nitrogen availability of cattle deep litter. *European Journal of Agronomy* **14**, 123-133.

Sommer, S.G., Christensen, B.T., Nielsen, N.E. and Schjorring, J.K. 1993. Ammonia volatilization during storage of cattle and pig slurry: effect of surface cover. *Journal of Agricultural Science* **121**, 63-71.

Sommer, S.G. and Dahl, P. 1999. Nutrient and carbon balance during the composting of deep litter. *J.agric.Engng Res.* **74**, 145-153.

Sommer, S.G. and Moller, H.B. 2000. Emission of greenhouse gases during composting of deep litter from pig production - effect of straw content. *Journal of Agricultural Science* **134**, 327-335.

Sommer, S.G., Petersen, S.O. and Sogaard, H.T. 2000. Greenhouse gas emission from stored livestock slurry. *Journal of Environmental Quality* **29**, 744-751.

Speirs, R.B. and Frost, C.A. 1987. The enhanced acidification of a field soil by low concentrations of atmospheric ammonia. *Research and Development in Agriculture* **4** (2):83-86.

Spiek, E., Sand, W. and Bock, E. 1990. Influence of ammonia on buildings. In: Ammoniak in der Umwelt (Hartung, J., Paduch, M., Schirz, S., Dohler, H. and van den Weghe H., eds.), Landwirtschaftsverlag GmbH, Munster, Germany.

Steed, J. and Hashimoto, A.G. 1994. Methane emissions from typical manure management systems. *Bioresource Technology* **50** (2):123-130.

Steudler, P.A., Bowden, R.D., Melillo, J.M. and Alber, J.D. 1989. Influence of nitrogen fertilization on methane uptake in temperate forest soils. *Nature* **341** 314-316.

Strader, R., Anderson, N. and Davidson, C. 2000. Development of an Ammonia Emissions Inventory for the Mid-Atlantic States and New England. Progress Report No. 3, Mid-Atlantic Regional Air Management Association, Baltimore, MD.

Sutton, A.L., Goodall, S.R., Patterson, J.A., Mathew, A.G., Kelly, D.T., and Meyerholtz, K.A. 1992. Effects of odor control compounds on urease activity in swine manure. *J. Anim. Sci.* **70** (Suppl. 1):160.

Sutton, A.L., Kephart, K.B., Verstegen, M.W.A., Canh, T.T., and Hobbs, P.J. 1999. Potential for reduction of odorous compounds in swine manure through diet modification. *J. Anim. Sci.* **77**:430-439.

Sutton, M.A., Milford, C., Dragosits, U., Place, D.J., Singles, R.J., Smith, R.I., Pitcairn, C.E.R., Fowler, D.H.J., ApSimon, H.M., Ross, C., Hill, R., Jarvis, S.C., Pain, B.F., Phillips, V.C., Harrison, R., Moss, D., Webb, J., Espenhahn, S.E., Lee, D.S., Hornung, M., Ullyett, J., Bull, K.R., Emmett, B.A., Lowe, J. and Wyers, G.P. 1998. Dispersion, deposition and impacts of atmospheric ammonia: quantifying local budgets and spatial variability. *Environmental Pollution* **102** (S1):349-361.

Sutton, A., Applegate, T., Hankins, S., Hill, B. Allee, G., Greene, W., Kohn, R., Meyer, D., Powers, W., and van Kempen, T. 2002. Manipulation of animal diets to affect manure production, composition, and odor: state of the science. *National Center for Manure and Animal Waste Management White Papers*, North Carolina State University, Raleigh, NC (available on CD-ROM from MidWest Plan Service), 31 p.

Sweeten, J.B., Parnell, C.B., Etheredge, R.S. and Osborne, D. 1988. Dust emissions in cattle feedlots. *Veterinary Clinics of North America, Food Animal Practice* **4** (3):557-578.

Sweeten, J.M., Childers Jr., R.E., Cochran, J.S. and Bowler, R. 1991. Odor control from poultry manure composting plant using a soil filter. *Applied Engineering in Agriculture* **7**, 439-449.

Sweeten, J.M., Jacobson, L.D., Heber, A.J., Schmidt, D.R., Lorimor, J.C., Westerman, P.W., Miner, J.R., Zhang, R.H., Williams, C.M., and Auverman, B.W. 2002. Odor mitigation for confined animal feeding operations. *National Center for Manure and Animal Waste Management White Papers*, North Carolina State University, Raleigh, NC (available on CD-ROM from MidWest Plan Service), 58 p.

Swierstra, D., Braam, C.R. and Smits, M.C. 2001. Grooved floor system for cattle housing: ammonia emission reduction and good slip resistance. *ASAE Applied Engineering in Agriculture* **17**, 85-90.

Takai, H., Pedersen, S., Johnsen, J.O., Metz, J.H.M., Koerkamp, P.W.G.G., Uenk, G.H., Phillips, V.R., Holden, M.R., Sneath, R.W. and Short, J.L. 1998. Concentrations and emissions of airborne dust in livestock buildings in Northern Europe. *Journal of Agricultural Engineering Research* **70** (1):59-77.

Tengman, C. L., Goodwin, R. N., and Bicudo, J. R.2001. Hydrogen sulfide concentrations around swine farms. *Procs. of the International Symposium addressing Animal Production and Environmental Issues*, Raleigh, NC, October 3-5, North Carolina State University, Raleigh, NC (on CD-ROM).

Thompson, R.B. and Meisinger, J.J. 2001. Measurement and management of ammonia

volatilization from dairy slurries. *Procs. of the International Symposium addressing Animal Production and Environmental Issues*, Raleigh, NC, October 3-5, North Carolina State University, Raleigh, NC (on CD-ROM).

Thorne, P.S., Reynolds, S.J., Milton, D.K., Bloebaum, P.D., Zhang, X., Whitten, P. and Burmeister, L.F. 1997. Field evaluation of endotoxin air sampling assay methods. *American Industrial Hygiene Association Journal* **58**, 792-799.

USDA .2000. Agricultural Statistics. National Agricultural Statistics Service, Washington DC.

Valli, L., Navarotto, P. and Bonazzi, G. 1994. Controlling ammonia emisions in a straw bedded finishing house. In *Animal Waste Management* (Hall, J.E., ed.), FAO, Rome, Italy, 59-63.

Van Breemen, N., Burrough, P.A., Velthorst, E.J., van Dobben, H.F., de Wit, T., Ridder, T.B. and Reijnders, H.F.R. 1982. Soil acidification from atmospheric ammonium sulfate in forest canopy throughfall. *Nature* **299** 548-550.

Van Ouwerkerk, E.N.J. and Pedersen, S. 1994. Application of the carbon dioxide mass balance method to evaluate ventilation rates in livestock houses. *Procs. of the CIGR-AgEng 94 Conference*, August 29 to September 1, Milan, Italy, 516-529.

Velthof, G.L., van Beusichem, M.L. and Oenema, O. 1998. Mitigation of nitrous oxide emission from dairy farming systems. *Environmental Pollution* **102, S1**, 173-178.

Vetter, H., Steffens, G. and Schropel, R. 1987. The influence of different processing methods for slurry upon its fertilizer value on grassland. In: *Animal manure on grassland and fodder crops - fertilizer or waste*, Martinus Nijhoff Publ, Dordrecht, The Netherlands, 73-86.

Voermans, J.A.M., Verdoes, N., and den Brok, G.M. 1995. The effect of pen design and climate control on the emission of ammonia from pig houses. *Procs. of the Seventh International Symposium on Agricultural and Food Processing Wastes*, June 18-20, Chicago, IL, ASAE, St. Joseph, MI, 252-260.

Wang, X., J. Jiang, R. Kaye. 2001. Improvement of wind tunnel sampling system for odour and VOCs. *Procs. of the 1st IWA International Conference on Odour and VOCs: Measurement Regulations and Control Techniques,* 25-28 March, Sydney, Australia.

Wathes, C.M., Holden, M.R., Sneath, R.W., White, R.P. and Phillips, V.R. 1997. Concentrations and emission rates of aerial ammonia, nitrous oxide, methane, carbon dioxide, dust and endotoxin in UK broiler and layer houses. *British Poultry Science* **38**, 14-28.

Watts, P.J., Jones, M., Lott, S.C., Tucker, R.W. and Smith, R.J. 1994. Feedlot odor emissions following heavy rainfall. *Transactions of the ASAE* **37**, 629-636.

Welsh, F.W., Schulte, D.D., Kroeker, E.J., and Lapp, H.M. 1977. The effect of anaerobic digestion upon swine manure odors. *Canadian Agricultural Engineering* **19**(2):122-126.

Westerman, P.W., Bicudo, J.R. and Kantardjieff, A. 2000. Upflow biological aerated filters for

the treatment of flushed swine manure. *Biores.Technol.* **74**, 181-190.

Westerman, P.W. and Zhang, R.H. 1997. Aeration of livestock manure slurry and lagoon liquid for odor control: a review. *ASAE Applied Engineering in Agriculture* **13 (2)**, 245-249.

Wilkie, A.C. (2000) Reducing dairy manure odor and producing energy. *Biocycle* **41**(9):48-50.

Willers, H.C., Derikx, P.J.L., ten Haave, P.J.W. and Vijn, T.K. 1996. Emission of ammonia and nitrous oxide from aerobic treatment of veal calf slurry. *J.agric.Engng Res.* **63**, 345-352.

Wilson, J.D., Thurtell, G.W., Kidd, G.E., and Beauchamp, E.G. 1982. Estimation of the rate of gaseous mass transfer from a surface source plot to the atmosphere. *Atmos. Environ*. **16**:1861-1867.

Xin, H., Berry, I.L. and Tabler, G.T. 1996. Minimum ventilation requirement and associated energy cost for aerial ammonia control in broiler houses. *Transactions of the ASAE* **39**, 645-648.

Xin, H., Berry, I.L. and Tabler, G.T., and Costello, T.A. 2001. Heat and moisture production of poultry and their housing systems: broilers. *Transactions of the ASAE* **44**(6), 1851-1857.

Yamulki, S., Jarvis, S.C. and Owen, P. 1998. Nitrous oxide emissions from excreta applied in a simulated grazing pattern. *Soil Biology and Biochemistry* **30**, 491-500.

Yang, P., Lorimor, J.C. and Xin, H. 2000. Nitrogen losses from laying hen manure in commercial high-rise layer facilities. *Transactions of the ASAE* **43**, 1171-1180.

Zahn, J.A., Hatfield, J.L., Do, Y.S. and DiSpirito, A.A. 2001a. Functional classification of swine manure management systems based on effluent and gas emission characteristics. *J. Environ. Qual.* **30**, 635-647.

Zahn, J.A., Hatfield, J.L., Do, Y.S., DiSpirito, A.A., Laird, D.A. and Pfeiffer, R.L. 1997. Characterization of volatile organic emissions and wastes from a swine production facility. *J. Environ. Qual.* **26**, 1687-1696.

Zahn, J.A., Tung, A.E., Roberts, B.A. and Hatfield, J.L. 2001b. Abatement of ammonia and hydrogen sulfide emissions from a swine lagoon using a polymer biocover. *Journal of the Air & Waste Management Association* **51**, 562-573.

Zhang, J., Keener, K. M., Bottcher, R. W., and Munilla, R. D.2001. Quantification of odorant recoveries from tedlar bags. *Procs. of the International Symposium addressing Animal Production and Environmental Issues*, Raleigh, NC, October 3-5, North Carolina State University, Raleigh, NC (on CD-ROM).

Zhu, J. 2000. A review of microbiology in swine manure odor control. *Agriculture, Ecosystems and Environment* **78**, 93

Zhu, J., Jacobson, L.D., Schmidt, D.R. and Nicolai, R. 2000. Daily variations in odor and gas

emissions from animal facilities. *ASAE Applied Engineering in Agriculture* **16**, 153-158.