# AIR EMISSIONS FROM ANIMAL PRODUCTION BUILDINGS

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**Abstract**

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.

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.

## INTRODUCTION

Air emission sources from animal production sites include buildings, 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. Although there are from two major sources of agricultural air emissions: animal housing and waste management systems, this paper will provide information on emission measurement and published data on odor, gas, and particulate emissions from animal housing only. 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 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. Thus the comparison of emissions from various studies is often difficult if not done on the same basis, such as 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 the building's 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 back 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).

## ODOR

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.

## AMMONIA

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 2 lists published ammonia emissions from livestock and poultry housing.

***Table 2.*** *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. (1998 a,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).

## NITROUS OXIDE

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.

## HYDROGEN SULFIDE

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 ($\pm$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 3). 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 3.*** *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.

## METHANE

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 4 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 4.*** *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 5). 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 5.*** *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).

## NON-METHANE VOLATILE ORGANIC COMPOUND

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 $hour^{-1}$.

## DUST

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, 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 6 summarizes the results obtained by Wathes et al. (1997).

***Table 6.*** *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 7). Emissions were estimated from mean daily dust concentrations near air outlets and the daily mean ventilation rate through the buildings.

***Table 7.*** *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

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 8.

***Table 8.*** *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 ($\mu$g $AU^{-1}$ $h^{-1}$)** | **Mean respirable endotoxin ($\mu$g $AU^{-1}$ $h^{-1}$)** |
|---|---|---|
| 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 9).

***Table 9.*** *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.

## CONCLUSIONS

Substantial research has been conducted to quantify the air quality and emission rates from livestock and poultry facilities. 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 facilities, 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 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.

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