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# Air Issues Associated with Animal Agriculture: A North American Perspective


## Abstract

The purpose of this CAST Issue Paper is to go beyond the generalizations and accusations often associated with the air quality topic. Experts from six universities examine a large amount of data and focus their information and conclusions around the key livestock areas: swine, poultry, dairy, and beef. Their critical analyses look at a wide scope of issues, from greenhouse gas (GHG) emissions to the logistics of manure storage facilities. The U.S. Environmental Protection Agency (EPA) is increasing efforts to monitor emissions from agriculture, so further research is important for all parties involved, and this paper provides solid, science-based information.

Studies indicate that large livestock production facilities lower the value of residences within 4.8 kilometers (km; three miles) of the facility. Other economic studies also indicate, however, that the businesses increase economic activity in the county and state. For the greater good of the rural community, a compromise needs to be reached between the positive and negative impacts of livestock production facilities using a common-sense approach that considers both regulatory and market forces.

Air emissions attributed to animal agriculture consist of odorous and gaseous compounds as well as particulate matter associated with manure and animal management. While localized problems associated with odor tend to get highlighted, gaseous compounds having localized or regional impacts, such as ammonia, and global concerns, such as those attributed to GHG, are becoming huge regulatory issues. This paper looks at some of the mitigation techniques being employed to decrease the effects of the aerial pollutants. The authors also examine the disparity between the results of the EPA's estimations of GHG emissions and the findings of the Food and Agriculture Organization (FAO) of


**In the past 15 to 20 years, air quality issues associated with the livestock and poultry industries have become a growing concern for the public. (Photo from iStock.)**

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the United Nations.

A few of the many specific findings include: diet composition has a significant impact on emissions; mitigation methods, such as covering the manure storage surface, can greatly decrease odor emissions; and aeration of the storage basin or employing anaerobic digestion of the manure will also reduce the odor, but with higher costs. The much-quoted study entitled *Livestock's Long Shadow* distinguishes between intensive and extensive livestock production; U.S. production is intensive so it does not have the GHG emissions associated with poor feed quality and deforestation.

## Introduction

Historically, environmental concerns and regulations of animal agriculture have focused on water quality. In the past 15 to 20 years, air quality issues associated with the livestock and poultry industries have become a growing concern for the public, leading to increased attention on enforcing air quality regulations for animal agriculture and new multimedia regulatory efforts.

In 2003, a report by the National Research Council (National Academy of Sciences), *Air Emissions from Animal Feeding Operations: Current Knowledge, Future Needs*, acknowledged the lack of baseline air emissions data for U.S. animal feeding operations (AFOs) and recommended the development of credible methodologies for estimating air emissions. The American Public Health Association (2003) called for a moratorium on new AFOs until their impact on the environment and health was better understood. In 2001, the U.S. Department of Agriculture–Initiative for Future Agricultural and Food Systems program funded two multistate studies that monitored ammonia ($NH_3$) emissions from poultry (broiler and layer) buildings (Iowa, Kentucky, Pennsylvania) and $NH_3$, hydrogen sulfide ($H_2S$), carbon dioxide ($CO_2$), *particulate matter*[1] (PM), and odor emissions from swine and poultry buildings (Illinois, Indiana, Iowa, Minnesota, North Carolina, Texas).

In 2005, the United States Environmental Protection Agency (EPA) settled on an Air Compliance Agreement (ACA) (USEPA 2005) with various livestock and poultry industry groups with the goals of (1) monitoring and evaluating air emissions from AFOs, (2) decreasing air pollution, and (3) ensuring compliance by AFOs with applicable legislation and regulation. Another ACA study was conducted in 2005–2006 in the southeastern United States with a major poultry producer that measured $NH_3$, $H_2S$, greenhouse gases (GHGs—$CH_4$ [methane], $CO_2$, and $N_2O$ [nitrous oxide]), *volatile organic compounds* (VOCs), and PM emissions (ISU 2006).

The EPA recently (January 2011) released air emission data on their website, http://www.epa.gov/airquality/agmonitoring/index.html, from the National Air Emissions Monitoring Study. This study, which was the main portion of the EPA ACA studies, includes emissions data from barns and manure storage sites across the United States for dairy (nine farms in California, Indiana, New York, Texas, Washington, and Wisconsin), swine (eleven farms in Indiana, Iowa, North Carolina, and Oklahoma), broiler (one farm in California), and layer (three farms in California, Indiana, and North Carolina) operations. Air pollutants monitored include PM, $H_2S$, $NH_3$, and VOCs (USEPA 2006a). The EPA cites the Clean Air Act (CAA) goal of protecting public health and environmental quality as the legislation directing it to potentially regulate air quality from animal agriculture, if emissions are found to exceed threshold levels triggering regulation.

Air emissions associated with animal agriculture consist of odorous and gaseous compounds and PM

[1] Italicized terms (except genus/species names and published material titles) are defined in the Glossary.

associated with manure and animal management. Odor problems tend to be localized to within a few kilometers of a production facility. Most gaseous compounds are also more intense locally, but some have regional or even national importance when redeposited on land or in water, or when they react to form secondary particles that can be transported long distances. For example, $NH_3$ loss into the atmosphere can have negative impacts on the environment, such as soil acidification and *eutrophication* of surface waters. When combined with nitric or sulfuric acid, atmospheric $NH_3$ can form fine particles that may impair human health and decrease visibility. Animal agriculture is also a significant source of GHGs, which are of global importance (IPCC 2007).

Air quality associated with animal agriculture has historically been unregulated or minimally regulated. That, however, has been changing in many states with regulation at several levels of political jurisdiction. The EPA and other federal and state agencies, such as the Occupational Safety and Health Administration, are responsible for air quality as it affects the health of workers and nearby residents. State and local governments have begun to enact regulations to minimize the impact of air emissions from AFOs on nearby neighbors, businesses, and public spaces.

## Regulatory Issues

The EPA is increasing its efforts to ensure that air emissions from agriculture, including AFOs, meet the EPA's environmental standards and their recent statement on human health risks from GHGs. The most notable efforts were the two ACAs, mentioned previously, between the EPA and livestock and poultry groups. Other EPA efforts directed toward agriculture entail (1) including sustainable production agriculture in the 2009–2014 EPA Strategic Plan Change Document with a goal of decreasing GHG emissions/contaminants (USEPA 2008), and (2) including manure management in their list of GHG emission sources that must annually report their emissions to the EPA (USEPA 2009a).

The EPA *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2008* estimates that of the 6,957 million metric tons of carbon dioxide equivalent (Mt $CO_2$ Eq) produced in the United States in 2008, 532 Mt $CO_2$ Eq (7.6%) are from agricultural activities. Of the agricultural emissions, 203 Mt $CO_2$ Eq (2.9%) are directly emitted by animal production (USEPA 2010). The EPA follows the UN Intergovernmental Panel on Climate Change (IPCC) 2007 report, *Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories*, recommendation of setting priorities among GHG sources and sinks within the national inventory. A source is designated a key category when it has a "significant influence on the country's total inventory of GHGs in terms of the absolute level of emissions, the trend in emissions, or both." The EPA listed 19 key categories based on 2008 emission levels and trends. Key categories 7, 15, and 18 on the list are $CH_4$ emissions from enteric fermentation, $CH_4$ emissions from manure management, and $N_2O$ emissions from manure management, respectively (USEPA 2010).

Using a Life Cycle Assessment (LCA) methodology, the United Nations FAO report titled *Livestock's Long Shadow: Environmental Issues and Options* estimated that global livestock account for 9%, 35 to 40%, and 65% of the total global anthropogenic-emitted $CO_2$, $CH_4$, and $N_2O$, respectively, which equated to 7,100 Mt $CO_2$ Eq/yr (Eq/year), or 18% of global anthropogenic GHG emissions (Steinfeld et al. 2006). A recent paper by Pitesky, Stackhouse, and Mitloehner (2009) questioned the validity of the FAO (2006) comparison between global livestock and transportation GHGs. The 2006 FAO report used a comprehensive LCA for livestock versus only direct emissions for transportation, thereby elevating livestock's relative contribution to climate change.

As mentioned earlier, the EPA's annual GHG inventory calculated a total of 203 Mt $CO_2$ Eq/yr from U.S. animal agriculture, which is 2.9% of the total U.S. anthropogenic GHG emissions (USEPA 2010). The disparity between the EPA and FAO estimations of GHG emissions from livestock agriculture occurs primarily because of the weight both place on livestock-related land use changes. Specifically, the FAO identifies land use change (principally deforestation to establish new pasture lands) as the number one global source of livestock-related emissions (34% of the FAO's total livestock-related GHG emissions) (Steinfeld et al. 2006). But land use change to supply new pasture lands is not occurring in the United States, and the amount of forestland in the nation has increased during recent years because of more sustainable timbering and farming practices and the abandonment of former farmlands (Pitesky, Stackhouse, and Mitloehner 2009). Furthermore, transportation and other industries dwarf the relative contributions livestock has made on total carbon (C) portfolios in developed countries. For example, the U.S. transportation sector accounts for 26% of all GHG versus 3 to 4% for livestock (Pitesky, Stackhouse, and Mitloehner 2009; USEPA 2009b).

In *Waterkeeper Alliance et al. v. EPA* (2004), the Second Circuit Court of Appeals affirmed that the EPA can regulate point sources of emissions in agriculture. In *American Farm Bureau Federation and National Pork Producers Council et al. v. EPA* (2009), the Court decided that the EPA's decision to regulate emissions of PM in rural areas was legal. This case is the first to allow regulation of PM air emissions in rural areas (except for engine emissions from off-road equipment) and accepts the use of the "precautionary principle" in justifying regulations. The precautionary principle argues that an action is to be avoided if the consequences of an action are unknown but judged to have some potential for major consequences. In this particular case, the Court concluded "the agency need not wait for conclusive findings before regulating a pollutant it reasonably believes may pose a significant risk to public health" (*American Farm Bureau Federation and National Pork Producers Council et al. v. EPA* 2009).

Many states (Iowa, Missouri, Oklahoma, etc.) regulate the siting of AFOs via setback distances between AFOs and residences, businesses, and public use areas (parks, churches, schools, etc.). These regulations are justified on a nuisance or air quality/odor rationale. Other states (Minnesota, Nebraska, etc.) have county or even township (local) control over siting of AFOs. The setback distances vary by state, type of animal building, and AFO size. The assumption is that concentrations of odor and other air pollutants diminish with distance and the setback distance keeps the AFOs from negatively impacting nearby residents.

State courts also have ruled that the odor from AFOs can create a liability. A Missouri court in 2010 ruled that odor from one of Premium Standard Farm's swine operations created an odor nuisance (*Owens et al. v. ContiGroup Companies et al*. not yet on file). A Nebraska court ruled that residences could not build within 0.8 kilometer (a half mile) of an AFO because of odor and dust (*Larry Coffey v. County of Otoe* 2008). Some counties and townships are beginning to regulate AFOs because of human health concerns. In Missouri, 21 counties or townships have instituted zoning or public health department regulations citing air quality as a human health concern (Milhollin 2010). Nebraska courts have ruled that municipalities can regulate AFOs, though primarily for water quality concerns (*Community of Alma v. Furnas County Farms* 2003).

Economic theory would suggest that odors from animal agriculture would decrease the value of real estate near the feeding facility. A summary of economic literature (Ulmer and Massey 2006) on the impact of AFOs on residential land values found the following: (1) five of eight studies did find that AFOs lowered residential property values; and (2) any decrease in property values was localized, or limited to properties less than 4.8 km (three miles) from the AFO.

## Animal Production Operations' Air Emissions and Air Quality

### Swine

#### Introduction

In pork production, gaseous compounds are primarily generated from anaerobic microbial decomposition of organic matter in animal manure and spoiled feedstuff. Gaseous compounds originate from the breakdown of various specific components of the pig's diet or normal excretion of compounds from metabolism; therefore, diet composition has a significant impact on the concentrations and emissions of gaseous compounds. Greenhouse gases ($CH_4$, $N_2O$, and $CO_2$) are derived primarily from microbial decomposition of manure in addition to normal animal respiration ($CO_2$). Particulate matter generates from feed systems, dried manure, and animal dander. Airborne microbes, otherwise classified as *bioaerosols*, generate from animals and manure and may be attached to PM. Bioaerosols need not be alive and viable to impair human health, as in the case of bacterial *endotoxin*.

Sources of gaseous emissions in pork production include the buildings that house the animals, manure storage structures, and manure during and after land application. Most swine are housed in either mechanically or naturally ventilated enclosed buildings. Manure is typically stored in liquid form within these buildings in a pit beneath the animal space for a period of only several days to as long as a year. Deep pits (8–10 feet deep) are commonly used to facilitate long-term storage of manure within these buildings, but long-term storage allows manure to decompose and release resulting *biogases*. In another common system, often referred to as a pull-plug system, manure is stored for only about a week within shallow pits and then removed from the buildings routinely to outside storage facilities to keep gas levels low within the animal building. If not covered, these outside storage facilities (earthen, above- or in-ground concrete or steel tanks) can emit considerable amounts of gas. Swine also may be housed in buildings (such as hoop barns) that use added bedding—typically straw, cornstalks, or sawdust—and produce solid manure. Swine manure is almost always applied to cropland as fertilizer.

Following is a discussion of both the concentrations of air pollutants inside or near swine buildings and manure storages and the mass flow or emissions of the airborne pollutants from pork production sites, in order of their perceived importance.

#### Ammonia

Ammonia concentrations inside studied swine confinement buildings have been shown to vary widely, from as low as 1.9 parts per million (ppm) to as high as 25.9 ppm, depending on the cleanliness condition of the building (Duchaine, Grimard, and Cormier 2000) and on the time of year and/or barn ventilation rate (Heber et al. 1997, 2000, 2004, 2005). Generally, farrowing rooms and nurseries had lower $NH_3$ concentrations than swine gestation or grow-finishing facilities (Jacobson et al. 2006; Zhu et al. 2000). As one might expect, ambient $NH_3$ concentrations diminished rapidly with the downwind distances from swine buildings (Stowell et al. 2000).

Arogo, Westerman, and Heber

(2003) conducted a literature review on research studies that reported $NH_3$ emissions from swine operations including buildings, lagoons, and field applications of swine wastes. They concluded that there are many factors affecting $NH_3$ emission rates including housing type; animal size, age, and type; manure management, storage, and treatment; and climatic variables. Because of these many factors, it is difficult to determine specific $NH_3$ emission values for all types of swine operations across the country. In a multistate research project involving different life-cycle aspects of pork production (Jacobson et al. 2004, 2006), average $NH_3$ emissions were 48 and 30 grams/day/animal unit (g/day/*AU*) for gestating and farrowing sows, respectively, and average $NH_3$ emissions for the finishing barns ranged from 102 to 130 g/day/AU in a deep-pit system and 77 to 81 g/day/AU from a pull-plug barn. An observation by Heber and colleagues (2005) was that hourly $NH_3$ emission was positively correlated with indoor and outdoor temperatures, ventilation rate, and total live pig weight. Pig activity and $NH_3$ emission rate displayed similar diurnal patterns. Carter, Lachmann, and Bundy (2008) showed that changing the diet of pigs can have a major impact on $NH_3$ emissions from swine production buildings. They reported that the use of a low protein and synthetic amino acid diet compared to a control diet decreased $NH_3$ emissions by almost half.

## Odor

Because gaseous compounds associated with odor vary widely in molecular weight and odorant strength, the concept of an "odor unit" (OU) was developed as a means of normalizing the specifically odor-related effect of an arbitrarily selected odorant or mixture of odorants. In general, an OU is defined as the mass of a reference odorant (e.g., n-butanol) that, when dispersed in the gas phase into odor-free air, results in an odor precisely at the human detection threshold for that reference odorant. This definition allows scientists to evaluate the strength of an arbitrary mixture of odorants using dilution olfactometry and translate the mixture's "dilutions to threshold" value into the pseudomass quantity OU. Refer to Table 1 for swine building odor emission and concentration studies.

**Table 1. Odor emissions/concentrations levels in sample studies.**

| Source | Odor Emissions | Odor Concentrations | Reference |
|---|---|---|---|
| 1,200-head grow-finish pig buildings | 10,000–12,000 OU/s[a] | 650–1,600 $OU/m^3$ [b] | Jacobson, Hetchler, and Schmidt (2007) |
| Fan-ventilated swine finishing buildings flushed daily with lagoon effluent | Mean of 23.5 OU/s/AU | Mean of 519 $OU/m^3$ | Heber et al. (2004) |
| Swine nursery buildings with under-floor liquid manure storage pits | 51 OU/s/pig–2.1 $OU/s/m^2$ | 190 $OU/m^3$ in exhaust air; 18 $OU/m^3$ outside building | Lim, Heber, and Ni (1999) |
| Four 1,000-head finishing buildings | Average of 96 OU/s/pig or 5.0 $OU/s/m^2$ | Average of 294 $OU/m^3$ (ranged from 12 to 1,586 $OU/m^3$) | Heber et al. (1998) |

[a]Per second
[b]Per cubic meter

## Volatile Fatty Acids and Volatile Organic Compounds

Volatile organic compounds measured at three swine farms (8,000-head nursery; 2,000- and 3,000-head finishing buildings) showed mean total VOC emissions of 204, 291, and 258 $\mu g/s/m^2$ (micrograms/second/square meter) for the nursery and two finishing buildings, respectively (Bicudo et al. 2002). A wean-to-finish pig growth study conducted by Radcliffe and colleagues (2008) reported *volatile fatty acid* (VFA) emissions with a range of 30 to 70 and 40 to 100 millimoles/day/pig from pigs fed a low-nutrient excretion diet and a control commercial diet, respectively.

## Hydrogen Sulfide

Indoor mean $H_2S$ concentrations have been found to be higher in gestating buildings (600 parts per billion [ppb]) than in farrowing barns (300 ppb), according to a study by Jacobson and colleagues (2006) involving different phases of pork production facilities. Hydrogen sulfide concentration levels spiked briefly to 3,000 ppb or 3 ppm in the barn when liquid manure was drained (pull-plug system) in the gestation sow building. Swine finishing buildings that are mechanically ventilated with deep-pit manure storage had mean indoor concentrations of $H_2S$ from 38 to 536 ppb over a period of six months (Ni et al. 2002a,b).

Farrowing (lactating sow) barns give off less $H_2S$ emissions than gestating sow buildings, as do finishing pigs in pull-plug versus deep-pit barns (Jacobson et al. 2006). Hydrogen sulfide emissions from a 1,200-head grow-finish pig building studied by Jacobson, Hetchler, and Schmidt (2007) ranged from 400 to 775 g/d (0.3 to 0.65 g/d/pig) over three seasons (winter, spring, and summer). Also, mean $H_2S$ emissions for two 1,000-head finishing buildings were 590 g/d/building or 6.3 g/d/AU (Ni et al. 2002c). The average $H_2S$ emission for the entire study was 0.72 g/d/pig. These rates were directly proportional to room temperatures and air flow rates. Pig size was not a significant parameter.

## Greenhouse Gases

Greenhouse gas emissions specifically related to swine production systems in Canada were estimated by Laguë (2003) as a total of 1,835 kilotons (kt) $CO_2$. This corresponds to approximately 3% of the total Canadian GHG agricultural emissions, 0.3% of the total Canadian anthropogenic GHG emissions, or 0.006% of the total world GHG emissions. A majority of $CO_2$ generation in pig operations is due to $CO_2$ expiration from animals. The amount of $CO_2$ released from the manure in a pig production building (partially slatted floor and shallow pits), as determined by Ni and colleagues (1999), represents 37.5% of the quantity of $CO_2$ exhaled by the animals, whereas Lim and colleagues (1998) reported $CO_2$ emissions from an 880-head grow-finish swine building with total slotted floors and tunnel ventilation as 3.0 kilograms/day/pig (kg/d/pig), ranging from 1.2 to 9.5.

Methane and $N_2O$ emissions varied from 48 to 54 g/d/AU and from 0.8 to 2.1 g/d/AU, respectively, for a slatted-floor pig finishing barn studied by Osada, Ram, and Dahl (1998). Methane emissions of 160 g/d/AU were measured from deep-pit and pull-plug pig finishing facilities (Zahn et al. 2001), while $N_2O$ emissions from the liquid manure management system of swine production in North America were estimated to be 20 g/yr/animal (Laguë 2003).

### Particulate Matter

Particulate matter ($PM_{10}$—PM with aerodynamic diameter of less than or equal to 10 micrometers [µm]) emission from a 1,200-head grow-finish pig building studied by Jacobson, Hetchler, and Schmidt (2007) ranged from 180 to 900 g/d (0.15 to 0.75 g/d/pig) and the $PM_{10}$ concentrations varied from 200 to 650 µg/m$^3$ over three seasons (winter, spring, and summer). Total suspended particulates (TSP) and $PM_{10}$ from fan-ventilated swine finishing buildings that were flushed daily with lagoon effluent showed a mean $PM_{10}$ emission of 1.6 g/d/AU and a mean $PM_{10}$ concentration of 334 µg/m$^3$ (Heber et al. 2004). Total suspended particulate concentrations measured by Duchaine, Grimard, and Cormier (2000) in eight swine confinement buildings with a range of cleanliness conditions resulted in a mean value of 3.54 mg/m$^3$ (3.54 milligrams/m$^3$; ranging from 2.15 to 5.60 mg/m$^3$). Concentrations of TSP in swine farrowing buildings were significantly lower than mean TSP concentrations in pig finishing buildings (Cormier et al. 1990).

### Bioaerosols

Microbial concentrations in eight swine confinement buildings with a range of cleanliness conditions were determined by Duchaine, Grimard, and Cormier (2000). The average concentrations were: molds, 883 colony forming units per cubic meter (cfu/m$^3$); total bacteria, 4.25 x 10$^5$ cfu/m$^3$; and endotoxin, 4.9 x 10$^3$ endotoxin units/m$^3$. Airborne microorganisms isolated by Cormier and colleagues (1990) in two swine farrowing and two pig finishing buildings showed total bacteria concentrations averaging 1.51 x 10$^5$ cfu/m$^3$ and 1.83 x 10$^5$ cfu/m$^3$ for each of the farrowing units and 4.92 x 10$^5$ cfu/m$^3$ and 5.44 x 10$^5$ cfu/m$^3$ for each of the two finishing barns. Bioaerosols assayed in 24 swine confinement buildings showed mean concentrations of bacteria and fungi ranging from 7.32 to 9.64 x 10$^4$ cfu/m$^3$ and 1.97 to 5.85 x 10$^3$ cfu/m$^3$, respectively. Other mean total bacterial concentrations were 5.9 x 10$^5$ cfu/m$^3$ (Haglind and Rylander 1987) and 1.08 x 10$^5$ cfu/m$^3$ (Heederick et al. 1991) for swine confinement buildings.

## Poultry

### Poultry Housing and Manure Management Systems

Ammonia is the major noxious gas in poultry operations, resulting from biological breakdown of uric acid in the feces. Prolonged exposure to aerial environment with elevated ammonia concentrations adversely affects bird health (e.g., respiratory system) and productivity (e.g., feed intake, body weight gain, egg production, feed conversion). The recommended indoor $NH_3$ level for poultry housing has been less than 25 ppm

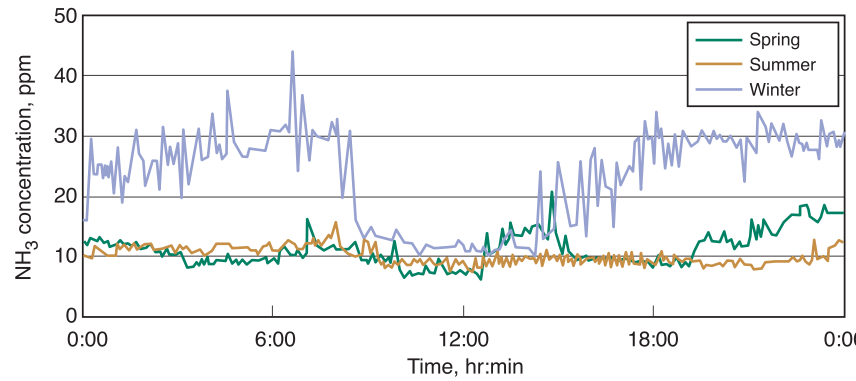


**Figure 1. Diurnal ammonia concentrations in a southeastern United States broiler house during different seasons (Li, H. 2009. Personal communication).**

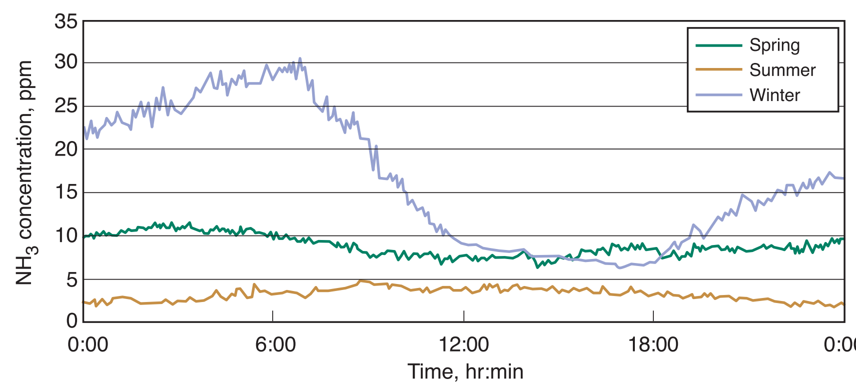


**Figure 2. Diurnal ammonia concentrations in a central United States turkey house during different seasons (Li et al. 2008a).**

(MidWest Plan Service 1990; United Egg Producers 2010). Indoor $NH_3$ generation and concentrations are largely affected by housing and manure management practices.

Broilers and turkeys (referred to as meat-type birds) are produced in littered floor systems. Most of the litter is typically reused (known as built-up litter) over multiple flocks of production, whereas the caked litter (wet, hardened surface layer) along the feed and water lines is removed with special equipment between the flocks. Some poultry producers may apply new bedding (top-dressing) following removal of the caked litter, whereas others opt not to, depending on the amount of litter in the house and availability and cost of bedding materials. An exception is with turkey brooding (less than five weeks of age), where old litter is completely removed and new bedding introduced for each flock. The removed litter from turkey brooder barns is placed in the grow-out barns used to raise the birds to market age. Depending on geographical location, the bedding materials used in broiler and turkey housing include rice hulls, wheat or rye hulls, sawdust or wood shavings, peanut shells, sand, and, in some cases, chopped straw or corn stalks.

More than 95% of the commercial egg production in North America uses cage systems, either high-rise (HR) or manure-belt (MB) housing systems. Hen manure in HR houses generally is stored in the lower level of the house for one year and removed as solid manure in the fall and applied to cropland, whereas manure in MB houses is removed daily to weekly into a separate storage or composting facility. As a result of in-house manure storage for the HR system vs. frequent removal of manure in the MB system, the MB houses generally have much better indoor air quality and lower $NH_3$ emissions than the HR houses. In addition to the two predominant housing types (HR and MB), a small fraction of eggs (< 5%) is produced in cage-free (CF) systems. Due to the significantly lower stocking densities of birds, thus lower body heat generation in the CF systems, maintaining a balance of good air quality (low $NH_3$ concentration) and comfortable barn temperature has proved to be challenging for the CF houses during cold weather (Green et al. 2009).

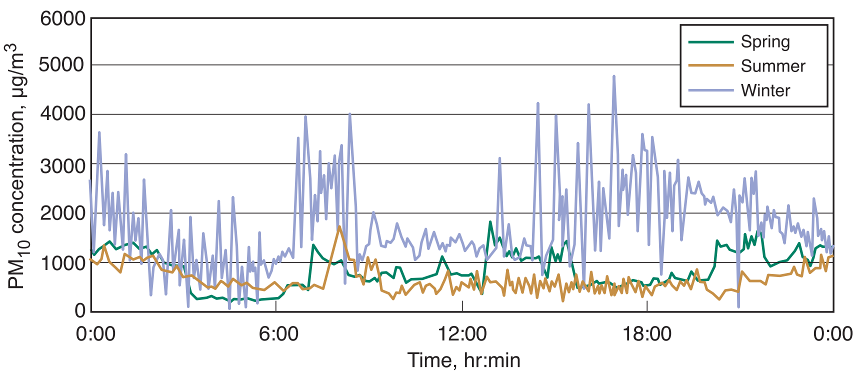


**Figure 3. Diurnal $PM_{10}$ concentrations in a southeastern United States broiler house during different seasons (Burns et al. 2008a).**

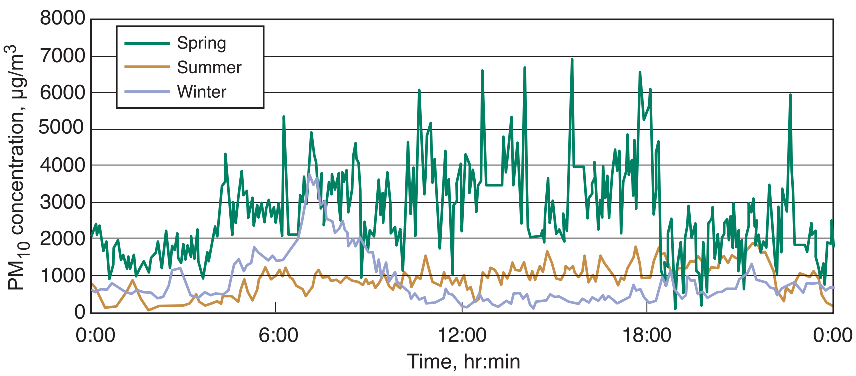


**Figure 4. Diurnal $PM_{10}$ concentrations in a central United States turkey house during different seasons (Li et al. 2008a).**

## Typical Indoor Ammonia and Particulate Matter Concentrations

Figures 1 and 2 show examples of indoor $NH_3$ concentrations found in commercial poultry production facilities during different production seasons. Compared to $NH_3$ data, PM concentration data are more limited. Nevertheless, Figures 3 and 4 illustrate diurnal variations of PM in commercial broiler and turkey buildings in different production seasons. As shown by the data, $NH_3$ and PM concentrations vary considerably throughout the day, especially during cool or cold weather, when there exist the diurnal variation of outside temperatures and hence building ventilation rates to maintain the desired indoor temperature.

The ventilation rate in a poultry barn typically is controlled by indoor temperature, but ventilation is the most commonly used method to decrease indoor $NH_3$ concentration. Increased ventilation, however, usually comes with the price of increased energy costs for ventilation fan operation and fuel usage (primarily LP gas) when supplemental heating is required. Moreover, the decreased $NH_3$ concentration does not represent decreased $NH_3$ emission rates—the amount of aerial $NH_3$ exhausted to the atmosphere. To the contrary, increased building ventilation rate likely leads to elevated emissions.

## Outdoor (Downwind) Concentrations

Information concerning downwind gas or PM concentrations is meager for poultry operations.

**Table 2. Summary of $NH_3$ emission rates (ER, g $NH_3$/d/AU) of laying hen houses with different housing and management schemes in different countries (1 AU=500 kg live weight) (Liang et al. 2006)**

| Country | House Type (season) | Manure Removal | $NH_3$ ER | Reference (Year) |
|---|---|---|---|---|
| England | Deep-pit (winter) | Info not available | 192 | Wathes et al. (1997) |
| England | Deep-pit (summer) | Info not available | 290 | Wathes et al. (1997) |
| England | Deep-pit (N/A) | Info not available | 239 | Nicholson, Chambers, and Walker (2004) |
| USA (Ohio) | High-rise (March) | Annual | 523 | Keener, Elwell, and Grande (2002) |
| USA (Ohio) | High-rise (July) | Annual | 417 | Keener, Elwell and Grande (2002) |
| USA (Iowa) | High-rise (all year) | Annual | 299 | Yang, Lorimor, and Xin (2000) |
| USA (Iowa and Pennsylvania) | High-rise (all year) *–standard diet* | Annual | 298 | Liang et al. (2006) |
| USA (Iowa) | High-rise (all year) *–1% lower CP diet* | Annual | 268 | Liang et al. (2006) |
| The Netherlands | Manure-belt (N/A) | Twice a week with no manure drying | 31 | Kroodsma, Scholten, and Huis in't Veld (1988) |
| The Netherlands | Manure-belt (N/A) | Once a week with manure drying | 28 | Kroodsma, Scholten, and Huis in't Veld (1988) |
| Denmark | Manure-belt (all year) | Info not available | 52 | Groot Koerkamp et al. (1998) |
| Germany | Manure-belt (all year) | Info not available | 14 | Groot Koerkamp et al. (1998) |
| The Netherlands | Manure-belt (all year) | Info not available | 39 | Groot Koerkamp et al. (1998) |
| England | Manure-belt (all year) | Weekly | 96 | Nicholson, Chambers, and Walker (2004) |
| England | Manure-belt (all year) | Daily | 38 | Nicholson, Chambers, and Walker (2004) |
| USA (Iowa) | Manure-belt (all year) | Daily with no manure drying | 17.5 | Liang et al. (2006) |
| USA (Pennsylvania) | Manure-belt (all year) | Twice a week with manure drying | 30.8 | Liang et al. (2006) |

**Table 3. Ammonia emissions of U.S. commercial broiler and turkey houses**

| State, USA | Species | Growth Period, d | Litter | g $NH_3$/Bird Marketed | Reference (Year) |
|---|---|---|---|---|---|
| Tennessee | Broiler | 42 | Built-up | 38.6 | Burns et al. (2003) |
| Texas | Broiler | 49 | Built-up | 30.9 | Lacey, Redwine, and Parnell (2003) |
| Delaware | Broiler | 42 | Built-up | 49.6 | Seifert et al. (2004) |
| Kentucky and Pennsylvania | Broiler | 42 | New | 19.7 | Wheeler et al. (2006) |
| | | 42 | Built-up | 27.3 | |
| | | 49 | Built-up | 37.2 | |
| | | 63 | Built-up | 61.7 | |
| Kentucky | Broiler | 52 | New | 25.5 | Burns et al. (2008a) |
| | | 52 | Built-up | 32.2 | |
| Iowa | Tom turkey | 35–140 | New and built-up | 144±12 | Li et al. (2008a) |
| Minnesota | Hen turkey | 35–84 | New and built-up | 104±10 | Li et al. (2009) |

**Table 4. Summary of limited data on PM ER from poultry facilities in different countries (1 AU=500 kg live weight)**

| Country | Poultry Species | Ventilation | PM Size | ER (mg/h/AU) | Reference (Year) |
|---|---|---|---|---|---|
| England | Broilers | Mechanical | Inhalable; respirable | 6,218; 706 | Takai et al. (1998) |
| Holland | | | | 4,984; 725 | |
| Denmark | | | | 1,856; 245 | |
| Germany | | | | 2,805; 394 | |
| USA (Minnesota) | Turkey | Natural | $PM_{10}$ | 135~210 (W)<br>431~2,133 (SU) | Schmidt, Jacobson, and Janni (2002) |
| USA (Texas) | Broiler | Mechanical/Mixed | $PM_{10}$ | 536 | Lacey, Redwine, and Parnell (2003) |
| | | | TSP | 10,210 | |
| USA (Indiana) | Layer | Mechanical | $PM_{2.5}$ | 26 | Lim et al. (2003) |
| | | | $PM_{10}$ | 384 | |
| | | | TSP | 1,512 | |
| USA (Ohio) | Layer | Mechanical | $PM_{10}$ | 361~1,319 | Zhao et al. (2005) |
| | | | TSP | 1,292~3,778 | |
| | | | | g/Bird Marketed | |
| USA (Kentucky) | Straight-run broiler grown to 52 d or 2.73 kg | Mechanical | $PM_{2.5}$ | 0.25 (±0.02) | Burns et al. (2008a) |
| | | | $PM_{10}$ | 2.52 (±0.46) | |
| | | | TSP | 6.0 (±0.29) | |
| USA (Iowa) | Tom turkey (35 to 140 d at 19.1 kg) | Mechanical | $PM_{2.5}$ | 3.7 (±0.8) | Li et al. (2008a) |
| | | | $PM_{10}$ | 30 (±4) | |
| USA (Minnesota) | Hen turkey (35 to 84 d at 6.8 kg) | Mechanical | $PM_{10}$ | 5.0 (±2.6) | Li et al. (2009) |

The Iowa Department of Natural Resources conducted downwind monitoring at designated distances from AFOs (cattle, egg, and swine) and compared the results against certain human health effect values (HEVs) (hourly average of 30 ppb $H_2S$) and health effect standards (seven daily exceedances of the HEV per year) (IDNR n.d.). Fairchild (2008) reported ammonia concentrations downwind from tunnel-ventilated broiler houses, indicating that 94% of the ammonia concentration readings were lower than 1 ppm at 150 m downwind from the exhaust fans.

## Air Emissions from Poultry Facilities

Before 2000, air emissions data in the literature were collected mostly from European poultry production facilities. During the past six to eight years, in response to the recommendations from the previously mentioned 2003 National Academy of Sciences report, researchers in the United States have made considerable strides toward collecting baseline air emissions data for U.S. production conditions. The most extensive field studies focusing on poultry air emissions completed to date are those reported by Liang and colleagues (2006) for $NH_3$ and $CO_2$ emissions from 10 laying-hen houses in Iowa and Pennsylvania over one year; by Wheeler and colleagues (2006) for $NH_3$ and $CO_2$ emissions from 12 broiler houses in Kentucky and Pennsylvania over one year; and by Burns and colleagues (2008a,b) for $NH_3$, $CO_2$, $H_2S$, $CH_4$, $N_2O$, TSP, $PM_{10}$, and $PM_{2.5}$ (PM with aerodynamic diameter of less than or equal to 2.5 μm) from two broiler houses in Kentucky over one year.

### Air Emissions from Poultry Houses

Table 2 summarizes $NH_3$ emissions for laying-hen houses in different countries, whereas Table 3 summarizes $NH_3$ emissions from broiler and turkey houses in different parts of the United States. Compared to data on $NH_3$ emissions, data on PM emissions are more limited. Table 4 summarizes the available literature information for poultry. Data on GHG emissions are even more limited. Burns and colleagues (2008b) reported GHG emissions of 3.41 g $CH_4$ (85.3 g $CO_2$ Eq) and 1.72 g $N_2O$ (513 g $CO_2$ Eq) per bird marketed for broiler houses in the southeastern United States. Compared to bird respiration of $CO_2$ production (4.64 kg), $CH_4$ and $N_2O$ emissions account for 9.8% and 1.6% of the total $CO_2$ Eq emission.

## Air Emissions from Poultry Manure Storage

Manure-belt housing systems involve separate manure storage, which also contributes to the overall air emissions of the production system. Li and Xin (2010) reported lab-scale measurements of $NH_3$, $CH_4$, and $N_2O$ emissions from hen manure storage. Factors considered in the evaluation included manure stacking configuration, manure moisture content (MC) as encountered under commercial production conditions, and ambient air temperature in the storage. Hen manure at 77% MC emits more $NH_3$ than that at 50% MC (Figure 5). It should be noted that storage conditions under commercial settings can differ considerably from the controlled environment of the lab studies, and field measurements over extended periods are thus warranted.

## Air Emissions from Poultry Manure Land Application

Poultry manure is typically land applied as fertilizer, although poultry litter has found very limited use in renewable energy generation as a fuel in power plants. Data on air emissions associated with land application of poultry manure are much more limited when compared to emissions from poultry houses or manure storage due to technical difficulties in quantifying such emissions and inherent large variations of application conditions. Ammonia nitrogen loss from manure during land application has been expressed as percentage of manure nitrogen content. These losses have been estimated to be 7% for dry laying-hen manure (Lockyer and Pain 1989), 41.5% for wet laying-hen manure (Lockyer and Pain 1989), and 25.1% for broiler litter (Cabera et al. 1994), as cited by the EPA in development of livestock and poultry manure management training.

## Mitigation Technologies

To improve indoor air quality and decrease air emissions, researchers and others have been actively investigating viable techniques that will decrease the generation and/or emissions of the aerial pollutants. The mitigation techniques that have been studied include dietary manipulation (Roberts et al. 2007; Xin, H. 2010. Personal communication), topical application of chemical or mineral additives on poultry manure (Li et al. 2008b), electrostatic precipitation of particulates (Ritz et al. 2008), treatment of exhaust air via biofilter or wet scrubber (Bandekar, Bajwa, and Liang 2008; Manuzon et al. 2007; Melse and Ogink 2005; Shah et al. 2008), or vegetative environmental buffer (Malone, VanWicklen, and Collier 2008), as well as prompt incorporation of land-applied manure. Although these prospective mitigation strategies have shown appreciable efficacy in emissions reduction, the economic viability of the techniques under production conditions remains to be further evaluated.

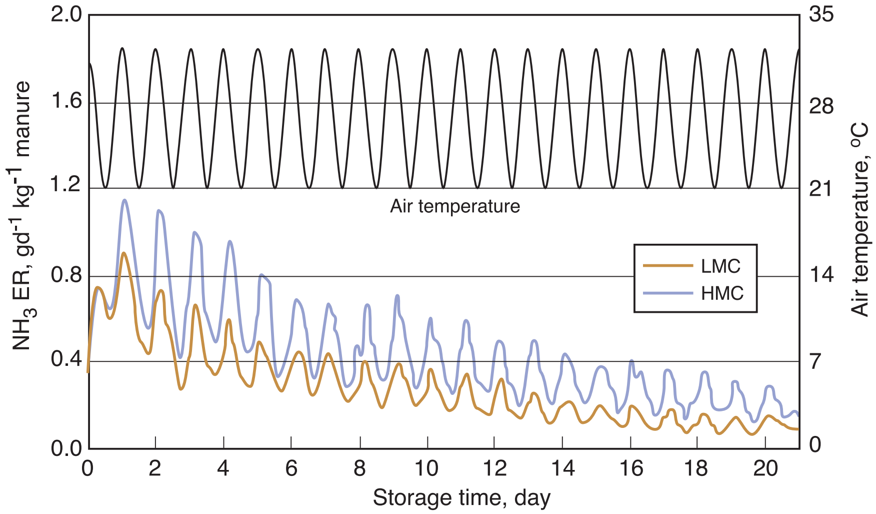


**Figure 5 . Ammonia emission rates from hen manure storage associated with manure-belt housing system as affected by air temperature and moisture content of the manure (LMC=low moisture content, 50%; HMC=high moisture content, 77%) (Li and Xin 2010).**

## Dairy

The effects of different classes of emissions (e.g., VOCs, GHGs, bioaerosols) associated with dairy AFOs in the United States have become a considerable public policy and regulatory issue. Dairies emit compounds identified as pollutants including $NH_3$, $PM_{2.5}$ and $PM_{10}$, VOCs, $H_2S$, other odor-forming compounds, and GHGs. These pollutants have a variety of local, regional, and global environmental effects including odor production, human exposure and health, ozone and PM formation, and production of GHGs that contribute to climate change. Therefore, a better understanding of the nature and significance of each of these pollutants with respect to dairy AFOs in North America is essential for appropriate legislation/regulation and ultimately cost-effective mitigation of pollutants.

Most intensive modern dairies in North America house large numbers of cows in barns or dry-lot corrals (instead of grazing on pasture). Significant emission sources identified with these facilities include barns or corrals associated with housing, manure storage and treatment facilities, silage piles, and cropland to which manure is applied.

Multiple abiotic (e.g., physical) and biotic (e.g., biological) factors influence the type and quantity of air emissions from each of the emissions sources just listed. Specifically, dairy emission rates are influenced by the following:

1. **Diet composition, feed conversion efficiencies, and animal physiology**.

   The makeup of a dairy cow's

diet, lactation status, and efficiency of its conversion into meat and milk products affects the quantity and composition of manure and urine excreted by cows. For example, a lactating cow produces more than twice the amount of methane compared to its dry cow equivalent.

2. **Manure handling practices.** How manure is handled affects its chemical and physical properties, including chemical composition, biodegradability, microbial populations, oxygen content, moisture content, and pH. An adult dairy cow produces approximately 41 to 54 kg (90 to 120 pounds [lbs]) of manure daily, which on dairy AFOs is usually either flushed with water or scraped from the housing area (free-stall) into large manure storage basins or treatment lagoons. The type of liquid manure system determines the potential for gas volatilization and odor production in both dairy housing and land application (ASABE 2006). For example, long-term stored cattle manure produces twice the concentration of VFA compared to fresh manure during anaerobic incubation, reflecting the significance of time and hence management on mitigation of odor-forming compounds (ASABE 2006).
3. **Environmental conditions.** Time of year, temperature, wind speed, and relative humidity greatly influence production and emissions of gaseous and particulate compounds. For example, $NH_3$ emissions from dairies were found to be approximately twice as much during the summer (11.3 to 18.2 g/d/cow) compared to the winter (3.8 to 6.8 g/d/cow) (Bluteau, Masse, and Leduc 2009).
4. **Dairy infrastructure.** Stocking density, barn ventilation rates, housing type, and bedding material can significantly affect air emissions.

As these examples demonstrate, understanding the different emissions profiles of pollutants throughout their "life cycle" and the factors that influence the emissions rates are essential for understanding, regulating, and mitigating dairy air emissions effectively. The following brief synopsis reviews the major classes of emissions from dairy AFOs.

## Odor

Odor production and the mitigation of odor-producing compounds from dairies are of growing concern because of urban encroachment into dairying regions. Mackie, Stroot, and Varel (1998) identified six major groups of odorous compounds, which include VOCs (e.g., VFAs, volatile amines, phenols, and indoles), $NH_3$, and sulfur-containing compounds. Most of these compounds are produced by anaerobic digestion and fermentation of organic matter. On a dairy operation, these compounds can be emitted from silage mounds, barns, waste storage facilities, or manure applied to land (National Research Council 2003).

The greatest odors on dairy AFOs are associated with land application of manure (1.5 to 90 OU/s/$m^2$) followed by dairy manure storage (5.1 to 32 OU/s/$m^2$) and housing (1.3 to 3.0 OU/s/$m^2$) (ASABE 2006; Pain et al. 1991). Although dairy manure that is applied to land has the greatest odor flux rates, the rates decrease rapidly over time (Mackie, Stroot, and Varel 1998). Furthermore, it should be noted that a 52% decrease in total odor units was achieved over 48 hours by immediately incorporating dairy slurry into the soil (Pain et al. 1991).

The class of compound with the greatest impact on odor production in dairy operations seems to be VOCs. Filipy and colleagues (2006) identified 82 different VOCs at a dairy lactating-cow pen stall and 73 VOCs from the manure lagoon, many of which are known to be odorous. Emissions rates of ethanol and dimethyl sulfide were highest and estimated at 1,026±513 and 13.8±10.3 µg/s/cow, respectively. Emissions rates from the manure lagoon for acetone, 2-butanone, methyl isobutyl ketone, 2-methyl-3-pentanone, dimethyl sulfide, and dimethyl disulfide were at much lower rates at 3.03±0.85, 145±35, 3.46±1.11, 25.1±8.0, 2.19±0.92, and 16.1±3.9 nanograms/second/cow, respectively (Filipy et al. 2006). Although the concentration of most of the detected compounds in this study was below published odor detection thresholds, the cumulative effects (additive or multiplicative) of different VOCs still require further investigation.

## Particulate Matter and Bioaerosols

The available literature is not extensive with respect to PM emissions from dairy AFOs in part because the emissions of particulates from different sources are dependent on several factors including animal density, manure management, flooring systems, and weather. A considerable and growing body of evidence, however, shows an association between adverse health effects and exposure to ambient airborne particulates in livestock operations (Aneja, Schlesinger, and Erisman 2009). Particulate matter emissions from dairies are considered a human health risk because of the present and potential inhalation of $PM_{2.5}$ and $PM_{10}$. Specifically, exposure to $PM_{2.5}$ has been associated with various types of pulmonary disease (Pope, Ezzati, and Dockery 2009), whereas exposure to $PM_{10}$ has been associated with decreased lung function, cardiac arrhythmia, heart attacks, and premature death (Madden, Southard, and Mitchell 2008).

In addition to the health effects, both $PM_{2.5}$ and $PM_{10}$ have local and regional environmental effects. On a dairy and at downwind locations, $NH_3$ reacts with atmospheric nitric and sulfuric acids to form $PM_{2.5}$. These fine particles contribute to smog or haze formation, decrease visibility, and may contribute to eutrophication in surface water following deposition downwind.

Feed storage and processing, cropping, composting, and manure storage can produce $PM_{10}$, including those derived from soil matter, feed, dried feces, bacteria, fungi, and endotoxins (Cambra-López et al. 2009). Although the relative significance of $PM_{10}$ is dependent on the type of dairy and the surrounding environmental conditions, soil-derived material often is the most significant source of $PM_{10}$ (Madden, Southard, and Mitchell 2008). Madden, Southard, and Mitchell (2008) noted a reduction of $PM_{10}$ from cropland between 52% and 93% with the use of conservation tillage.

With respect to bioaerosols, Seedorf and colleagues (1998) measured bacterial, enterobacteriaceael, and fungal emissions of 6.8, 6.2, and 6.0 cfu/hour/AU, respectively, which were similar to measurements in other livestock housing operations. The significance of these data for public health and worker safety has not been determined.

### Volatile Organic Compounds and Ammonia

Emissions of VOCs and $NH_3$ from dairy operations occur when bacteria decompose organic matter in manure and silage under anaerobic conditions (Filipy et al. 2006). Both of these emissions have local effects including odor production, particulate formation, troposphere ozone formation, ecosystem degradation, and health effects including ear, nose, and throat damage (Cambra-López et al. 2009; Mitloehner and Calvo 2008; Pinder, Adams, and Pandis 2007).

Volatile organic compounds include thousands of individual gases that vaporize at room temperature. They contribute to ozone formation when combined with oxides of N ($NO_x$) in the presence of sunlight (Shaw et al. 2007), although VOCs have differing reactivity and potential to form ozone (Carter 1994). Volatile organic compounds typically are divided into the following classes: VFAs, indoles and phenols, amines, and sulfur-containing compounds (ASABE 2006).

Among the major VOC emissions identified in dairy operations, five compounds (2-butanone, p-cresol, phenol, methyl isobutyl ketone or hexone, and methanol) have been identified as hazardous air pollutants by the EPA. To attain "major-source" emissions levels, however, herd sizes will have to be of considerable size (Ndegwa 2009).

In addition to the cows and their manure, recent work has identified fermented feeds as a significant source of VOC emissions on dairy AFOs (Alanis et al. 2008; Howard et al. 2010a). But more research is needed and is currently being conducted to better identify VOC sources and emission rates from dairy production.

Volatile organic compound production, emissions, and flux rates are determined by biological (i.e., chemical reactions catalyzed by bacteria) and environmental (i.e., wind speed, volatility) factors. Consequently, there is considerable variability in the type and concentration of VOCs from dairy AFOs, leading to much controversy regarding the impact of VOCs (Filipy et al. 2006; Hobbs, Misselbrook, and Cumby 1999). For example, in 2004, California regulatory agencies estimated dairies to contribute as many VOCs to the atmosphere as light/medium-duty trucks or light passenger vehicles. But two subsequent studies concluded that the sum of reactive VOC fluxes measured was 6 to 10 times less than previous estimates and that the VOC emissions from dairy cattle and their fresh waste have a relatively small impact on ozone formation per VOC mass emitted (Howard et al. 2010b; Shaw et. al. 2007). On dairies, fermented feed is a much larger contributor of smog-forming VOCs than manure or any other source (Howard et al. 2010a).

Ammonia is emitted from AFOs when the nitrogen (mainly urea) in animal waste is hydrolyzed, mineralized, and/or volatilized. Once volatilized, the $NH_3$ can be harmful in several ways including acidification of ecosystems, eutrophication, and formation of aerosols and particulates, including $PM_{2.5}$ (National Research Council 2003). As discussed previously, $NH_3$ is derived from N that originates in the excreta. The N excretion (urine and feces) estimate for an average dairy cow in a modern western U.S. dairy is 169 kg/head/yr (Sunesson, Gullberg, and Blomquist 2001). Urine urea N, when combined with urease contained in feces, is readily hydrolyzed to ammonia; however, the content of urea N in a dairy animal's urine varies greatly with diet (James et al. 1999).

Ammonia volatilization is the main loss pathway for N from dairy AFOs (Kirchmann et al. 1998) with total $NH_3$ losses ranging from 17 to 46 kg N/yr/cow (Bussink and Oenema 1998) or between 20 and 40 g $NH_3$/d/AU in the free-stall area (Groot Koerkamp et al. 1998; Snell, Seipelt, and Van den Weghe 2003). The wide ranges in $NH_3$ losses and volatilization reflect the variability in amount and composition of animal excreta (urine + feces), management of the slurry, and soil and environmental conditions. Sources of $NH_3$ emissions on dairy AFOs in decreasing relative importance are slurry application, housing, slurry storage, and fertilizer application (Bussink and Oenema 1998).

Manure and facilities management (i.e., flooring system, type of bedding, and manure handling system) have a significant effect on volatilization rates of $NH_3$. Kroodsma, Huis in't Veld, and Scholtens (1993) determined that scraped or dirty solid floors gave $NH_3$ emissions of 15 g $NH_3$d/m$^2$, versus 5 g $NH_3$/d/m$^2$ for a flushed system. Thompson and Meisinger (2002) measured $NH_3$ emissions following surface land application of liquid dairy manure and found when manure was left unincorporated into the soil for a period of five days, 39 to 52% of the ammonium-N ($NH_4^+$-N) was lost. Mitigation of $NH_3$ emissions from land application via land injection of manure (i.e., anaerobic conditions) versus traditional land application is an area of active research. Although immediately incorporating manure into the soil can decrease $NH_3$ emissions (Thompson and Meisinger 2002), incorporating manure can greatly increase emissions of the potent GHG $N_2O$ (Wulf, Maeting, and Clemens 2002). Likewise, mitigation of $N_2O$ emissions via composting of manure often results in increased $NH_3$ (Amon et al. 2001).

### Greenhouse Gases

Major sources of $CH_4$ on dairies include enteric fermentation within the rumen and stored manure. Methane emissions from dairy cows raised in intensive systems range from between 55 and 70 kg $CH_4$/yr/animal (ASABE 2006; Moiser et al. 1998).

Dairy AFOs produce significant amounts of $CH_4$ because of the widespread use of anaerobic manure lagoons and storage basins. Specifically, manure stored under aerobic conditions produces less $CH_4$ than the anaerobic systems used in dairy AFOs (ASABE 2006). Safley and Casada (1992) measured $CH_4$ conversion factors (MCF) between 20 and 90 from anaerobic liquid slurry systems compared to MCFs of 10 for solid and pasture-applied manure (ASABE 2006).

Gaseous emissions of $N_2O$ from soils associated with dairy waste or dairy-owned cropland are a major source of indirect GHG emissions associated with dairy manure. Nitrous oxide formation results from denitrification processes reducing nitrate ($NO_3$) to $N_2O$ and from nitrification oxidizing ammonium ($NH_4$) to $N_2O$. Therefore, when excess $NO_3$ or $NH_4$ is present in the soil (i.e., from fertilizer or manure application), $N_2O$ emissions can be significant (Menneer et al. 2005). In addition, higher $N_2O$ emissions have been noted at sites with long-term N applications, suggesting that long-term accumulation of N in soils affects emissions. Consequently, effective manure and land management is essential to the mitigation of $N_2O$. Enteric fermentation has been found to produce small amounts of $N_2O$ (Kaspar and Tiedje 1981); however, little research has been conducted to quantify $N_2O$ emissions derived directly from dairy cattle. Decreasing $N_2O$ emissions could have a significant impact on total GHG emissions from dairies because of its high global warming potential (IPCC 2007).

## Beef

Although most of the confined beef production in the United States now takes place in the Great Plains and the West in outdoor facilities known as “feedlots” or “feedyards,” some cattle feeding still occurs in open-sided or fully enclosed barns. Most of the beef production that takes place under roof in the United States is found in the higher-rainfall areas east of the Mississippi River, where the abundant rainfall is captured as roof drainage and directed away from the production areas without being contaminated by manure, feed, or other kinds of waste.

In the Great Plains and West, open feedlots are the norm because most or all of the precipitation that falls on the production area can be evaporated or irrigated (during the warmer months) or cheaply stored in runoff ponds through the colder months until it can be evaporated or used for irrigation. As a result, air emissions from beef production areas in the Great Plains and West tend to be driven and modified by precipitation (or its extended absence). Air emissions from beef production areas in the eastern United States tend to be driven and modified by ventilation, either passive or active (forced). Where beef animals are fed under roof, additional gas-phase emissions result from uncovered manure storages, holding ponds, and anaerobic lagoons, similar to most of the confined swine- and poultry-feeding operations across the nation.

In the semi-arid and temperate climates of the Great Plains and the western United States, open-lot feedyards are the norm and consist of unvegetated, usually earthen corrals in which beef animals receive a grain-based, mixed feed in concrete feed troughs two or three times per day. Depending on liveweight or growth stage, animal sex, bunk space, climatic factors, management schemes, and topography, each animal may be allocated from 7 to 28 $m^2$/head (7 to 28 $m^2$/hd; 75 to 300 $ft^2$/hd) of corral area. The production area of a cattle feedyard with a one-time capacity of 20,000 hd may occupy a footprint between 14 and 57 hectares (ha; 35 and 140 acres). In the most concentrated cattle-feeding region in the United States—the southern High Plains, comprising parts of Texas, Oklahoma, and southwest Kansas—the typical summertime cattle stocking density is about 14 $m^2$/hd (150 $ft^2$/hd).

Beef production in the United States includes a substantial herd grown on pasture and rangeland, either all the way to slaughter or for a period of time before being transported to feedyards to be finished. Although such operations are beyond the scope of this report, they also exchange gases with the atmosphere that may represent significant proportions of the North American budgets of ecologically important elements such as C, N, and sulfur (S). A life-cycle understanding of emissions attributable to beef production would need to include pasture operations, where applicable.

### Emissions from Beef Feedyards

The primary emissions of local concern to cattle feeders and their neighbors are fugitive particulate matter ($PM_{10}$ and coarse particulate or $PM_{10-2.5}$), which arise mostly from uncovered corrals, unpaved roads, or exhaust fans, and nuisance odor, which consists mainly of VOCs and, to a lesser extent, $NH_3$ and $H_2S$. At the regional and national scales, $NH_3$, $H_2S$, and VOCs may contribute to secondary fine particulate matter ($PM_{2.5}$), acid rain, and ground-level ozone ($O_3$).

As with other livestock and poultry species, most of the airborne emissions from cattle feedyards derive from what the animals eat. The nutrient composition of so-called "finishing" rations—mixed feed given to animals within a few weeks of slaughter, with a dry-matter digestibility of about 83%—includes about 2.2% N and 0.32% S on a dry-matter basis (Heflin, K., L. McDonald, and B. W. Auvermann. 2000. Unpublished data). When distillers' grains are included at up to 30% of feed dry matter, N and S may be as high as 2.6 and 0.43% of dry matter, respectively, with dry-matter digestibility decreasing to around 78% (MacDonald, J. Personal communication). The more N, S, and dry matter excreted by the animals, the greater the potential for those constituents to be emitted as $NH_3$, $H_2S$, and PM, respectively.

Feed efficiency of cattle (i.e., dry matter intake per unit of body weight gain) ranges from 5:1 to 7:1 depending on breed, sex, diet composition, days on feed, dietary supplements, animal health, climate, and anabolic implant status. Nutrition technologies have been quite successful in decreasing the feed-to-gain ratio (increasing feed efficiency) industry-wide, which increases production efficiency and decreases emissions of macronutrient gases per unit of beef produced.

## Particulate Matter

Particulate matter is the primary air pollutant of concern to cattle feeders, their neighbors, and their local communities. Airborne PM emitted by cattle-feeding operations includes dust from unpaved roads, dander, hair, and dry, pulverized manure and soil resuspended in the air by hoof action on the corral surface. Under typical conditions, hoof action is the primary mechanism for fugitive PM emissions.

In the semi-arid and arid regions of the western United States, mass concentrations of ground-level PM immediately downwind of cattle feedyards vary significantly and predictably with time of day, with the peak occurring shortly after dusk and reaching a short-term (e.g., five-minute averaged) concentration 5 to 15 times greater than the corresponding 24-hour average concentration (see Figure 6). This so-called "evening dust peak," during which five-minute average $PM_{10}$ concentrations may reach into the thousands of micrograms per cubic meter (µg/m$^3$), results from the confluence of three quasi-independent influences: (1) increasing atmospheric stability, (2) increasing animal activity, and (3) daily minimum moisture content in the uncompacted manure on the corral surface. The absolute values of the short-term concentrations at the downwind corral edge vary widely from day to day and from feedyard to feedyard. The PM concentrations are quite sensitive to variations in atmospheric stability, feedyard geometry, and animal inventory.

Emission rates of fugitive dust from feedyard corrals are notoriously difficult to estimate. The typical method involves measuring mass concentrations of dust at fixed, downwind receptors and using meteorological data in a dispersion model to infer the flux of dust from the source area. The original emission factor for fugitive TSP from cattle feedlots was 127 kg (280 lb) TSP/d/1,000 hd (USEPA 1985). Subsequent work established that the $PM_{10}$/TSP ratio in feedyard dust was between 0.19 and 0.40 (Sweeten et al. 1988), meaning that the federally recommended emission factor was equivalent to 31.75 kg (70 lb) $PM_{10}$/d/1,000 hd.

Since 1994, as new modeling algorithms were developed for ground-level area sources, many new estimates of the *AP-42 emission factor* have been proposed, ranging from 1.8 to 11.3 kg (4 to 25 lb) $PM_{10}$/d/1,000 hd and converging at an annualized value of 7.7 kg/d/1,000 hd (17 lb/d/1,000 hd) for feedyards in the southern High Plains (Parnell, Shaw, and Auvermann 1999). A more rational approach to emission factors would account for the vast regional diversity in climatic settings; an appropriate emission factor for cattle feedyards is likely to be higher in winter-rainfall areas like California and lower in the more humid regions of the Great Plains and eastern United States as compared to that of the southern High Plains.

Road-dust emissions are ex-

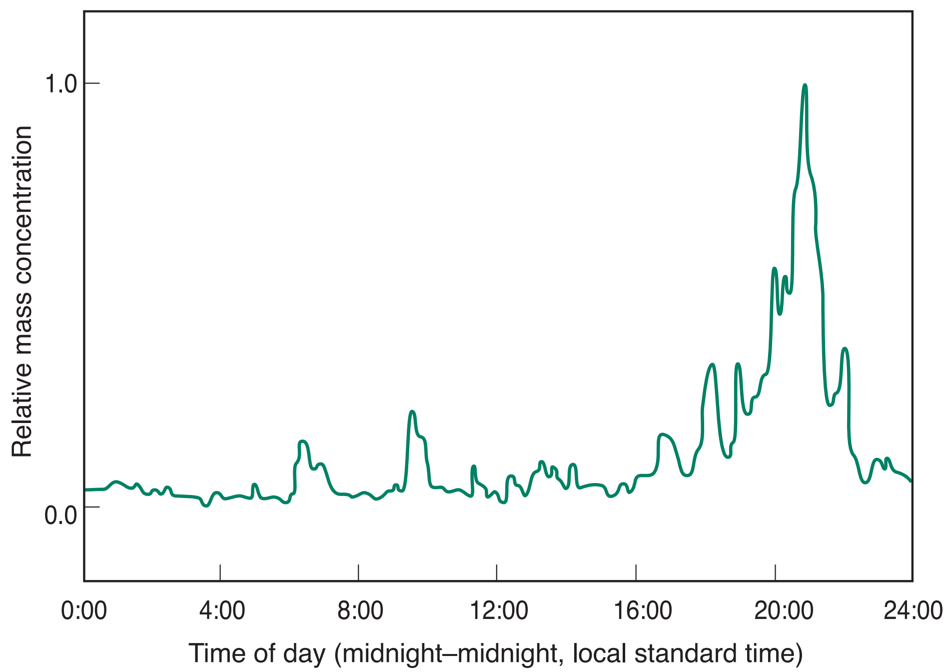


**Figure 6. Typical daily variation in mass concentrations of $PM_{10}$ downwind of cattle feedyards (Auvermann, B. Unpublished data).**

pressed in units of mass per distance traveled (e.g., kg/vehicle miles traveled). The EPA has developed emission estimates of $PM_{10}$ from unpaved roads (USEPA 2006b) that are a function of the silt content (%) of the road surface, the gross vehicle weight, the vehicle ground speed, and the moisture content (%) of the road surface. An estimate of the total annual $PM_{10}$ emissions from a feedyard's vehicle fleet may be obtained by estimating the average daily emissions for each vehicle used in the feedyard, summing those emissions over all vehicles used in the facility, and multiplying by 365.

Emission factors, which are estimates of the mass emitted per unit of industrial throughput or capacity, are often criticized as one-size-fits-all estimates that do not account for differences in feeding regimes, climate, corral design, or management factors that vary from feedyard to feedyard. A report commissioned by the National Academy of Sciences recommended that the emission-factor approach to estimating emissions of all kinds of air pollutants be replaced by a process-based modeling approach that accounts for site-specific feedyard attributes, climate, and management tactics (National Research Council 2003). A mathematical framework for process-based modeling of PM emissions from cattle feedyards has been proposed (Auvermann 2003); although that framework has been used to develop screening methods for various dust-control techniques (Razote et al. 2006), it has not been comprehensively and conclusively validated.

## Ammonia

Ammonia emissions from cattle feedyards arise principally from the metabolism of proteins and amino acids. Metabolites of those feed constituents are excreted as urea, which is subsequently hydrolyzed to aqueous-phase $NH_3$ in the presence of the enzyme urease. Once aqueous $NH_3$ is present in the manure on the feedyard surface, it may be volatilized to the atmosphere as gaseous $NH_3$. Because the chemical pathways from feed N to gas-phase N emissions are well defined, gaseous $NH_3$ emissions are probably best expressed in mass-balance terms as a percentage of the N fed to the animals.

Ammonia emissions from open-lot cattle feedyards vary seasonally, with summertime emission rates approximately twice the wintertime emission rates (Todd et al. 2008). Using a variety of independent methods to estimate emissions, annualized $NH_3$-N emissions from commercial cattle feedyards in Texas were between 36 and 53% of the N fed to the animals (Todd and Cole 2007; Todd et al. 2008). Thus, a 20,000-hd yard feeding a corn-based ration at about 10.2 kg (22.5 lb) dry matter per day per head would be expected to emit a total of about 2,050 kg (4,519 lb) of N per day (2,050 kg N/d). Other researchers have estimated $NH_3$ emissions from open-lot feedyards at rates between 9 and 56% of the N fed to the animals (Faulkner and Shaw 2008). Although some of that N is emitted from runoff holding ponds, most of it is emitted by the corral surfaces.

Concentrations of $NH_3$ at ground level near cattle feedyards are highly variable and, as is the case with nearly all air pollutants, depend strongly on the averaging time over which the concentrations were measured (National Research Council 2003).[2] Moreover, because $NH_3$ is so reactive, changing the location at which $NH_3$ is measured may change the measured concentrations significantly. Finally, $NH_3$ plumes downwind of open-lot feedyards are spatially nonuniform, so path-averaged and point monitors generally yield different measured concentrations.

Flesch and colleagues (2007) used open-path lasers to measure 15-minute average $NH_3$ concentrations within and downwind of a cattle feedyard. The long-term mean of their measurements within the feedyard was between 1.2 and 1.7 ppm, with daily maximum 15-minute concentrations approaching 5 ppm. At their downwind measurement path nearest the feedyard, long-term mean concentrations were between 0.8 and 1.0 ppm, with daily maximum 15-minute concentrations approaching 2.5 ppm. Long-term averages and daily maximum 15-minute concentrations decreased substantially at the more distant monitoring sites downwind. They assumed that the long-term average background concentration was on the order of 0.015 ppm (15 ppb) or two orders of magnitude lower than the in-source and downwind concentrations (Flesch et al. 2007).

McGinn, Janzen, and Coates (2003) measured 48- to 72-hour average concentrations of $NH_3$-N downwind of three cattle feedyards in Alberta. Concentrations immediately downwind occasionally approached 1.5 ppm (850 $\mu g/m^3$) but averaged about 1 ppm. Short-duration (five-minute) concentrations measured with portable analyzers were on the order of 3 ppm or less at the downwind perimeter. At 200 m downwind, $NH_3$-N concentrations routinely decreased by 60 to 80% as compared to the concentrations measured at the feedyards' downwind perimeter.

## Hydrogen Sulfide

Hydrogen sulfide is a product of the fermentation of sulfur-containing compounds (primarily the amino acids methionine and cysteine, as well as sulfur salts) ingested by cattle in feed and drinking water and then excreted in urine and feces. It is produced under anaerobic conditions, generally postexcretion, which in the feedyard are found in runoff holding ponds and on corral surfaces that remain wet for an extended time (Parker et al. 2005). Its odor (*he-*

[2] In general, peak concentrations over short durations significantly exceed the peak concentrations over much longer durations. Thus, the highest peak 1-hour $NH_3$ concentration on a given day would be much greater than the corresponding 24-hour average concentration.

*donic tone*) is widely described as "rotten eggs," which accounts for its offensiveness and its reputation as a primary cause of nuisance odor. At high concentrations (> 150 ppm), $H_2S$ can be lethal, but such concentrations are not observed in outdoor livestock facilities.[3] Koelsch and colleagues (2002) measured total reduced S, which includes mainly hydrogen sulfide, concentrations on a 15-minute averaging time near cattle feedyards in Nebraska and reported a maximum 15-minute concentration of 0.03 ppm along feedyard perimeters. Within the feedyard properties (i.e., adjacent to primary sources such as runoff holding ponds) per se, "no situations were observed where three consecutive readings exceeded Nebraska's 0.1 ppm, 30-minute standard" (Koelsch et al. 2002).

Like $NH_3$, $H_2S$ is listed as a "hazardous substance" under the EPA's Emergency Planning and Community Right-to-Know Act (EPCRA), and as of January 2009 livestock facilities releasing more than 45.4 kg/day (100 lb/day) of $H_2S$ are required to report those releases to local emergency-response authorities. Drawing on research conducted in the Texas High Plains, the range of emission factors used in the National Cattlemen Beef Association's 2009 EPCRA guidance document for cattle feedyards[4] is $2.132 \times 10^{-3}$ to $3.856 \times 10^{-3}$ kg/d/hd ($4.7 \times 10^{-3}$ to $8.5 \times 10^{-3}$ lb/d/hd).[5] Because the study area has a semi-arid, summer-rainfall climate, those emission factors would need to be adjusted for use in winter-rainfall, colder, or temperate climates such as the Pacific Coast, the northern Great Plains, and the southeastern United States, respectively.

## Volatile Organic Compounds

Trace gases emitted by cattle feedyards include many of the same metabolites and co-products emitted by other livestock and poultry species. The class of emissions known as VOC is diverse.

Odorous VOCS tend to be, though not exclusively, intermediate products of fermentation and include mercaptans, amines, organic sulfides, VFAs, phenols, alcohols, aldehydes, ketones, and many others. Their odor potentials, as measured by the threshold concentrations at which the human nose can detect them, vary over many orders of magnitude.

Reactive VOCs are known, along with a wide variety of hydrocarbons, to react with other atmospheric gases in the presence of sunlight to form $O_3$, a criteria pollutant[6] and the main component of smog. The reactive VOCs include alkenes, aldehydes, and some industrial chemicals. Of particular regulatory importance are the so-called "highly reactive VOCs," which react more rapidly than other reactive VOCs to form $O_3$. Some of the most notorious of the reactive VOCs, isoprene and monoterpenes, are emitted naturally and in large quantities by deciduous and coniferous forests, respectively (Fuentes et al. 1996).

Emission fluxes of VOCs, like other fugitive emissions from cattle feedlots, are difficult to obtain with good precision, and accuracy is difficult to assess because there are no standard methods that have been validated against known emission fluxes. In most cases, researchers use the inverse-modeling technique of measuring downwind concentrations and inferring source strength via dispersion modeling.

Downwind concentration data are more easily found. For example, researchers in Texas measured *p*-cresol (an odorous VOC) concentrations in the air upwind and at multiple locations downwind of two cattle feedyards in the Texas Panhandle (Perschbacher-Buser et al. 2007). One-hour concentrations of *p*-cresol occasionally approached 0.07 ppb at the downwind property line but decreased rapidly with downwind distance. Researchers in Canada measured VFAs and other odorants over two- to three-day averaging times near the downwind perimeters of three cattle feedyards in Alberta (McGinn, Janzen, and Coates 2003). Total VFA concentrations varied from 10.4 to 177.6 μg/m$^3$, with acetic, butyric, and propionic acids predominating. The *p*-cresol concentrations ranged from 0.003 to 0.018 μg/m$^3$, which are in the low part-per-trillion range. It also seems from their research that VFAs are less prone than $NH_3$ to depletion from the plume as the compounds are carried downwind.

Organic compounds—not all of them volatile—are also likely to be carried along in fugitive aerosols either as (inherently) solid-phase or adsorbed constituents. Rogge, Medeiros, and Simoneit (2006) found more than 100 organic compounds in samples of the soil collected near the surfaces of cattle feedyards and dairies in the San Joaquin Valley of California.[7] Not surprisingly, the compounds they found are associated with feedstuffs, vitamins, and supplements routinely fed to livestock, as well as metabolites of those materials, and the authors proposed that the

[3] Agricultural fatalities associated with $H_2S$ tend to cluster around indoor facilities in which manure is stored below the production floor. When a manure pit is agitated to ensure that it can be emptied efficiently, it releases a burst of $H_2S$ that may exceed the concentration that is lethal to any worker present in that building without respiratory protection. See, for example, Ni et al. (2000).

[4] See Sweeten et al. (1988).

[5] Expressing the upper bound of the $H_2S$ emission factor in terms of mass per animal unit is slightly misleading in the case of cattle feedyards because (1) basal (dry-weather) $H_2S$ emissions come primarily from runoff holding ponds, and (2) elevated (wet-weather) emissions are roughly proportional to wet corral area. In either weather regime, the emitting area is likely to be a more reliable predictor of $H_2S$ emissions than the number of animals on feed at the time. Lower flux estimates were reported by Baek et al. (2006).

[6] "Criteria pollutants" are air pollutants for which health-based, ambient air quality standards have been adopted under the CAA Amendments. They include $PM_{10}$ and $PM_{2.5}$, lead, $O_3$, $NO_x$, sulfur dioxide, and carbon monoxide. For more information, see http://www.epa.gov/air/urbanair/.

[7] Using the term "soil" for the materials they analyzed may be confusing. The samples in question were collected outside the corrals and should be interpreted as mineral soil that has been subjected to continuous dry deposition of dust and gases emitted from the feedyard surfaces.

organic compounds be used as source markers that distinguish feedyard dust from other aerosol sources.

Because there are so many organic, N- and S-based, and aerosol-phase compounds present in cattle excreta on the feedyard surface, no brief account can describe all of them with respect to emissions processes, fate and transport, or downwind concentrations. Researchers generally group them into classes and subclasses, which obscures individual compound behavior but permits useful generalizations for selecting control measures.[8] Still, the scientific literature on VOC emissions from cattle feedyards is thin, and to the extent odors and ground-level ozone continue to be nuisance and public-health threats (respectively) near cattle-feeding operations, efforts to quantify emissions and develop abatement measures should continue.

### Greenhouse Gases

Greenhouse gases of concern to the cattle-feeding industry include $CH_4$, $CO_2$, and $N_2O$. They vary widely in their atmospheric prevalence and in their heat-trapping capacity, with $N_2O>CH_4>CO_2$. Greenhouse gases from beef production units in Ireland were the focus of a life-cycle inventory study by Casey and Holden (2006). Although Irish beef systems differ considerably from cattle feedyards in the United States, their projections serve as an interesting benchmark for future work on U.S. confinement operations. Synthesizing survey responses from 15 suckler-beef operations, they projected that GHG emission from Irish beef production was between 1,500 and 6,000 kg of $CO_2$ equivalents per hectare per year, depending on feed composition, management strategies, and other site-specific descriptors.[9]

[8] A good example of this tendency can be found in Miller and Berry (2005).

[9] Projections like these are difficult, if not impossible, to validate at the national scale. For a full description of the authors' life-cycle procedure, see Casey and Holden (2006).

## Manure Storage and Land Application

Manure is stored at nearly all commercial animal operations in the United States, from a few days to more than one year, primarily to conserve manure nutrients for timely application onto cropland or for value-added processes, such as anaerobic digesters and composting. The practice of production site manure storage creates another emission source of odors, gases, particulates, and bioaerosols. The type of manure storage varies depending on species and region of the country. Earthen storage basins or treatment lagoons are common for dairy farms throughout the country and for swine operations in the southeastern and southern regions of the United States. Concrete storage pits beneath buildings are commonly used in the Midwest by swine producers, especially pig finishing and nursery facilities. Poultry operations may store their manure (semi-dry litter) within the buildings (on floor) for up to two years or as short as a single flock (seven weeks) and then stockpile the litter outside the building or adjacent to cropland for up to six months or more. All of these manure storage options affect the farm's overall air quality and emissions.

Land application of manure has the potential to be the largest emitter of air pollutants from the animal production systems. But in manure handling systems that experience large emission losses during the housing and/or manure storage phases, there may not be much N or S to lose during land application. Also, the magnitude of these emissions (primarily odor and gas) during land application of manure can be greatly decreased with known technologies such as direct incorporation (injection) into the soil and tillage immediately or shortly after surface application. Because manure contains valuable crop nutrients, nearly all manure from animal production systems is eventually applied to cropland or pastures. Besides this strong economic incentive to recover nutrients, land application of manure may be the most environmentally friendly method for using animal manure.

### Odor

Manure storage units are chronic sources of odor emissions for most of the animal species. This is especially the case for uncovered earthen storage or treatment lagoons that store liquid manure, because any disturbance of the liquid surface can result in odor (as well as many gas) emissions. A number of studies, summarized by Casey and colleagues (2006), have measured odor emissions (primarily odor flux rates) with ranges from 5 to 50 $OU/s/m^{-2}$ for swine and dairy liquid manure storage units and also for stockpiled solid poultry manure storage units. Mitigation methods such as covering, either by a natural crust or an artificial membrane, of the storage surface can greatly (70 to 90%) decrease the odor emissions from these sources (Bicudo et al. 2002). Other mitigation techniques, such as aeration of the storage basin (Westerman, Bicudo, and Kantardjieff 2000) or employing anaerobic digestion (Powers et al. 1997) of the manure, will also decrease the odor generated but with considerably higher costs.

Odor emissions during land application of "stored" manure (especially liquid manure) are an acute problem for the animal production industry. Much of the odor released during land application originates during the agitation/mixing process of manure storage units to suspend manure solids so they can be pumped and removed from the storage basin or tank. Also, the spreading of manure, especially liquid manure, onto cropland or grassland can produce high amounts of odor if it is simply broadcast or spread on the land surface. A large decrease in odor (90+%) emissions occurs if manure is directly injected into the soil or tilled into the soil immediately after surface application (Pain et al. 1991). Pain and colleagues (1991) also measured odor emissions from the application of solid cattle manure and found considerably smaller odor flux rates than for pig slurry spreading.

### Ammonia, Hydrogen Sulfide, and Volatile Organic Compounds

Manure storage systems can emit so-called hazardous gases—$NH_3$, $H_2S$, as well as VOCs—into the atmosphere. Uncovered liquid manure storage basins and tanks along with treatment lagoons can release moderate to high levels (5 to 40 g $NH_3$/d/m$^2$) of $NH_3$ and relatively low levels (0.65 to 5 g $H_2S$/d/m$^2$) of $H_2S$ (Gay et al. 2003). Volatile organic compound emission data are much more limited for manure storage sources and have typically identified specific compounds such as VFAs, phenols, and indoles (Bicudo et al. 2004; Hobbs, Misselbrook, and Cumby 1999; Zahn et al 2001). As with odor emissions, covered liquid manure storage and most solid manure storage systems have lower gas emission rates, typically in the lower ranges (5 to 10 g $NH_3$/d/m$^2$) of those seen for uncovered liquid systems (Bicudo et al. 2004; Brewer and Costello 1999) for $NH_3$ and below the range (0.16 g $H_2S$/d/m$^2$) for uncovered liquid systems for $H_2S$ (Bicudo et al. 2004).

Land application of manure can constitute large $NH_3$ and other gas emission sources with reported levels above 100 g $NH_3$/d/m$^2$ (Chadwick et al. 1998). As with odor, much of the gas emission occurs during the agitation of the manure storages (liquid) that often produces large spikes of $H_2S$, $NH_3$, and other VOCs (Clarke and McQuitty 1987). Ammonia loss during land application using both deep and shallow injection and drag shoe methods was decreased from 90+, 80+, and 60+%, respectively, in a study conducted by Burton (1997). Other techniques can be used to decrease these gases (especially $NH_3$) but are highly dependent on soil, moisture, and climate conditions during the application process (Thompson and Meisinger 2001).

### Greenhouse Gases

Designs for anaerobic treatment basins or lagoons often result in low VOC and other gas emissions but may produce increased levels of $CH_4$ emissions (Zahn et al. 2001). Measurements of lagoon $CH_4$ emission rates have varied from almost 0 up to as high as 188 g $CH_4$/d/AU (Casey et al. 2006). Average $CH_4$ emissions of 154 g/d/AU and 21.4 g/d/AU were measured from liquid manure storage units by Zahn and colleagues (2001) and Hobbs, Misselbrook, and Cumby (1999), respectively. Covers over cattle slurry storages (straw, Leca, or natural crust) decreased $CH_4$ emissions 40% compared to uncovered storages (Sommer and Moller 2000). Solid manure storages may also produce $CH_4$ depending on the degree of composting (increasing $CH_4$ generation) that occurs. Sommer and Moller (2000) found solid dairy manure produced about 50.8 g $CH_4$/tonne (50 g $CH_4$/ton) whereas composting of deep-bedded housing swine manure with a high bulk density produced up to 254 g $CH_4$/tonne (250 g $CH_4$/ton).

Solid manure storages have been identified as sources of $N_2O$ with emissions of 3.58 kt $N_2O$/yr and 1.86 kt $N_2O$/yr being reported by Chadwick and colleagues (1999) for stockpiled cattle and poultry manure, respectively. Nitrous oxide emissions will be released in any manure treatment system that uses a combination of aerobic and anoxic treatments (Beline and Martinez 2002). Sommer and Moller (2000) measured $N_2O$ emissions from the composting of solid manure with straw from swine housing and found a total of 58.9 g/tonne (58 g/ton) of $N_2O$ emitted during the composting period. Anaerobic lagoons in North Carolina from a farrow-to-finish farm and a farrow-to-wean operation emitted 0.3 and 0.4 kg/d/ha of $N_2O$, respectively (Harper et al. 2004).

Chadwick and colleagues (1998) measured $CH_4$ emissions after land application of pig slurry and found nearly all (98%) occurred within the first four days after spreading. They also found that $CH_4$ emissions during fall application were twice as large as summer spreading (5.5 vs. 2.35 kg $CH_4$/ha). Sharpe and Harper (2002) reported an $N_2O$ flux of 0.0016 mg/d/m$^2$ before irrigation of swine lagoon liquid to cropland and 2.6 to 3.8 mg/d/m$^2$ after irrigation. Akiyama and Tsuruta (2003) measured yearly $N_2O$ fluxes of 184 and 61.3 mg/m$^2$ from the land application of poultry and swine manure, respectively.

### Particulate Matter and Bioaerosols

Airborne emissions of dust and bioaerosol from manure storage units and land application are relatively unknown. Liquid manure would emit little or no PM during the storage or land application phase but could be a source of airborne microorganisms (pathogenic or otherwise). However, little or no data have been collected. Solid manure storage and the associated application of these solids onto cropland could produce some PM if the solid manure is sufficiently dry. Typically only cattle feedyard stockpiles and lots (especially in the dry High Plains area of the southwestern United States) have been identified as a source of PM emissions (Auvermann et al. 2002).

### Manure Use Summary

On-farm manure storage systems and the land application of animal manure are sources of odor, gases (hazardous and greenhouse), particulates, and bioaerosols. The magnitude and importance or priority of each of these pollutants varies depending on species, location in the United States, and type of manure management system. From a producer's perspective, odor is often the main neighborhood concern during siting of a new production operation or expansion of an existing one. From a regulatory standpoint, hazardous gases ($H_2S$ and $NH_3$) may be of the greatest concern, because depending on the state or area of the country, there may be regulatory limits on what can be emitted from the production site or there is a property line concentration threshold for certain gases. Until recently, GHGs, PM, and bioaerosols have been less of a concern from either manure storage or land application sources. But GHGs and PM are a growing concern as more data are becoming available and the concern over climate change and regional air quality in nonattainment sections of the country increases. Bioaerosols are being examined both from public and animal health perspectives but are not high on the list for most producers.

## Air Pollutant Impacts and Policy

Most studies indicate that large livestock production facilities lower the value of residences within 4.8 km (three miles) of the facility (Ulmer and Massey 2006). Studies also indicate that these businesses increase economic activity in the county and state. Political actions try to balance the competing positive and negative impacts of any business. To the extent that pollutants (including odor) have local effects, regulatory actions should be taken into account locally. For example, the risk of nitrogenous gases leaving lagoons and being redeposited within water bodies is greater in Missouri (with average rainfall of 1.02 m/year [40 inches/year]) than in western Oklahoma (with average rainfall less than 0.5 m [20 inches] per year).

A combination of regulatory and market forces is causing a shift in the way manure is stored and land applied. Regulatory forces are emphasizing sufficient storage capacities for spreading windows, setbacks of storage and spreading areas from environmental concerns such as wells and streams, and application limits that reflect crop nutrient needs. Environmental forces are reinforcing some of these ideas. Higher fertilizer prices are causing producers to seek to maximize the value of their manure resource by storing in a manner that conserves nutrients and land applying it to crops that benefit from the nutrients. Livestock production is moving back to the crop-producing regions of the nation as opposed to areas where the nutrients are not as valuable because crop production is minimal.

Greenhouse gases from livestock production are primarily from enteric fermentation and manure management. The much-quoted UN study *Livestock's Long Shadow* distinguishes between intensive and extensive livestock production (Steinfeld et al. 2006). United States livestock production is intensive and does not have the GHG emissions associated with poor feed quality and deforestation. Enteric fermentation is a natural process for ruminant animals, but the GHG emissions/unit of meat produced is very low in the U.S. production system.

The U.S. manure management system is relatively high in GHG emissions because most all pork manure and at least half of the dairy manure is stored wet rather than dry. The $CH_4$ emissions that come from manure storage structures can be captured and destroyed by using covers and digesters on large facilities. The economies of scale associated with capturing and destroying $CH_4$ indicate that attempts to decrease $CH_4$ from livestock production will foster greater concentration in the industry. Currently, the science exists to capture and destroy $CH_4$, but the economic benefit from qualifying and selling C offsets or credits using registries such as the Climate Action Reserve, Regional Greenhouse Gas Initiative, and Chicago Climate Exchange and from electricity generation using methane-powered generators is insufficient in most livestock operations to justify the expense of the equipment. Other benefits must be identified (e.g., odor reduction or increased retention of nutrients) or costs must come down in order for methane digesters to be voluntarily adopted.

## Summary

In the past 10 to 20 years, air quality and emissions inside of and from animal production systems have become environmental concerns equal to those of water quality and emissions to surface and groundwater for animal agriculture in the 1970s and '80s. During the same time, there has been a large increase in air quality/emission research, much of it driven and funded by the animal industry, that has provided air emission rates and information for a variety of airborne parameters from animal housing units, associated manure storages, and land application systems. The airborne parameters measured and the levels of concern include the following:

- Odor and $PM_{10}$—primarily a local issue and nuisance complaints
- Hazardous gases ($NH_3$ and $H_2S$) and $PM_{2.5}$—regional/national issues; primarily environmental impacts, but also health concerns
- Greenhouse gases—global importance on climate change impacts
- Air quality (pollutant concentrations) inside animal buildings or ambient levels near neighbors—animal and human health issues

Swine production systems generate a large variety of significant air emissions. Ammonia, $H_2S$, odor, VOCs, PM, bioaerosols, and GHGs are all emitted in quantities of concern from pork production systems. This has created challenges for pork producers, because a more comprehensive approach (vs. targeting one or two contaminants) is necessary to control or manage several of these air emission parameters simultaneously. But the pork industry has been quite progressive in dealing with air quality problems, funding emissions research projects, and promoting practices and technologies that manage and mitigate air emissions from their production systems.

Poultry producers, on the other hand, have been concerned primarily with one airborne contaminant—ammonia—secondarily, with PM and, potentially, bioaerosols. This is especially the case for layer operations. Meat bird (broiler and turkey) producers have some concern with PM levels in barns and PM emissions because of the litter pack manure-handling systems. Ammonia emissions have been determined for the major housing systems used in the poultry industry, with some housing systems and/or management practices producing substantial $NH_3$ emission decrease.

Dairy producers, like swine producers, have a number of different air emission parameters of concern. Ammonia, odor, and GHG emissions are of interest to the U.S. dairy industry; in addition, dairies in western states such as California are also concerned about VOC emissions. Dairy operations not only have emissions from buildings and manure storages, but they also have large feed (forage) storage sources. Additionally, the dairy industry has become proactive in managing and mitigating emissions from their operations, funding air emission monitoring and C footprint studies.

Beef cattle are primarily finished in large open feedyards. The main air quality concern with these open lots is PM (dust) emissions. Other airborne parameters such as $NH_3$ may be of concern under certain weather (wet) conditions.

Emission of gases from manure storages and land application of animal manure on cropland and pastures can be a significant percentage of the total air emissions from animal operations. Management practices and mitigation technologies have been and are being developed and used to decrease odor, hazardous gases, GHGs, and VOCs from manure storage units and during land application of manure.

Regulation of air emissions from animal production at the federal, state, and even local level has been steadily increasing during the past 20 years, creating uncertainty for producers and the industries. Compliance to existing and new regulations is being met through a combination of new mitigation technologies and management practices depending on the animal species, location of the production operations, and economics of the industry. Many of these mitigation technologies are site and specie specific, but several, such as diet manipulation, are common to all animal species. The goal is to use science-based information to help all stakeholders involved in animal production to protect the environment and public health in a proactive manner and avoid costly litigation to solve nuisance suits or enforce regulations.

## Glossary

**Animal unit**. 1 AU=500 kg or 1,100 lbs of animal weight.

**AP-42 emission factor**. Emission factors listed in the AP-42 series of documents are best viewed as EPA guidance, primary regulatory authority residing with the states.

**Bioaerosols**. Airborne particles that are biological in origin.

**Biogas**. A mixture of methane and carbon dioxide produced by the bacterial decomposition of organic wastes and used as a fuel.

**Endotoxin**. A toxin produced by certain bacteria and released upon destruction of the bacterial cell.

**Eutrophication**. A process where water bodies receive excess nutrients that stimulate excessive plant growth.

**Greenhouse gases**. Gases that trap heat in the atmosphere.

**Hedonic tone**. A property of an odor relating to its pleasantness or unpleasantness.

**Particulate matter**. Minute airborne liquid or solid particles (such as dust, fume, mist, smog, smoke) that cause air pollution.

**Volatile fatty acid**. A short-chain fatty acid—acetic, propionic, and butyric acid—which, apart from its presence in some foods, is produced by intestinal bacteria from undigested starch and dietary fiber.

**Volatile organic compounds**. A large group of carbon-based chemicals that easily evaporate at room temperature.

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