

# **Ammonia Emissions from Animal Feeding Operations**

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## 1 EXECUTIVE SUMMARY

The purpose of this paper is to summarize the state of knowledge regarding ammonia ( $\text{NH}_3$ ) emissions from animal feeding operations. Based on the information in published literature, the paper summarizes:

- The effects of  $\text{NH}_3$  emitted from animal production to the environment.
- Emission rates and quantities from animal buildings, storage and treatment facilities, and land application sites.
- Ammonia emission measurement methods.
- Models for  $\text{NH}_3$  emissions, transport, and deposition.
- Possible control strategies and technologies.

Atmospheric  $\text{NH}_3$  is produced by the decomposition of organic materials, biomass burning, and fertilizer production and utilization. Ammonia is involved in plant metabolism and can be exchanged between vegetation and the atmosphere. Ammonia emissions abatement has had high environmental priority in parts of Europe in recent years, and it is receiving more attention in the U.S. as a potential air and water quality concern. The primary concerns about ammonia emissions into the atmosphere are: (1) nutrient deposition in nutrient sensitive ecosystems and (2) formation of aerosol particles that may cause haze and impair visibility and are also a concern for potential health effects from respirable particulate matter.

Agricultural activities, livestock production in particular, have been reported to be the largest contributor of  $\text{NH}_3$  emissions into the atmosphere. Farm animals consume a considerable amount of protein and other nitrogen (N) containing substances with their feed. The conversion of dietary N to animal product is relatively inefficient and 50 to 80% of the N consumed is excreted. Ammonia is produced as a consequence of bacterial activity involving the excreted organic N substrates.

Ammonia can be emitted from animal housing, manure storage and treatment facilities, and manure land application in animal production operations. Factors that affect  $\text{NH}_3$  volatilization include source characteristics (manure, building type, storage and treatment methods, and land application method), pH, temperature, wind speed, and surface characteristics. At this time, the majority of data for emissions from animal feeding operations are from Europe where buildings, manure management, and climate are often different than in the U.S. Previously, little research on ammonia emissions has occurred in the U.S., but research is increasing. Progress is being made in development of measurement equipment and methods, but the expense of measurement and lack of continuous measurement capability has hindered the development of reliable annual emission factors. Typically, data are collected over short durations, and extrapolations beyond the sampling periods and conditions are prone to error.

Emission rates are usually expressed in terms of mass of  $\text{NH}_3$  or ammonia nitrogen ( $\text{NH}_3\text{-N}$ ) per unit time and per animal (or liveweight units) or per unit area (surface sources). Although air quality literature often uses units of  $\text{NH}_3$  mass in reporting emission data, this paper uses  $\text{NH}_3\text{-N}$  mass because it simplifies its use in N accounting for confined animal production. To convert  $\text{NH}_3\text{-N}$  mass to  $\text{NH}_3$  mass, multiply by 1.214.

The ranges of measured emission rates can be large among the European and U.S. data. Building emissions range from 0.2 to 5, 0.12 to 1.48, 0.28 to 0.74, and 0.5 to 10 g  $\text{NH}_3\text{-N/h-AU}$  (1 AU=500 kg live weight) for pigs, dairy cattle, beef cattle, and poultry, respectively. Storage/treatment losses reported are 0.25 to 156 and 3 to 90 kg  $\text{NH}_3\text{-N/ha-day}$  for lagoons and storage tanks, respectively. Land application losses range from 14 to 83%, 6 to 47%, and 0 to 7% of total  $\text{NH}_3\text{-N}$  applied for surface spread, band spread and injected manure, respectively. Data on  $\text{NH}_3$  emission

measurements from beef cattle feedlots and large dairies with open housing in the U.S. is limited. The  $\text{NH}_3$  losses from the various sources at animal production operations are often expressed in different units or on different basis, making it more difficult to calculate  $\text{NH}_3$  loss per animal. Some of the reported emissions are derived from direct assumptions that a certain percentage of N excreted by the animal is lost due to  $\text{NH}_3$  volatilization. Without measurements, these assumed emissions should be used with extreme caution. Additionally, N excretion data for all animal species need better documentation. Published information indicates  $\pm 30\%$  variation and it is even complicated further with the current efforts in dietary manipulation to reduce N excretion.

Ammonia emission rates from different sources in animal feeding operations have been used to develop emission factors. The ammonia emission factor for animals in an animal production operation represents the sum of the annual mean emission rates from housing, manure storage/treatment, and land application. Emissions factors are based on average annual conditions, and typically a composite of various animal sizes and types for a particular animal species. Emission factors currently used in the U.S. are based on those developed for Europe. Composite emission factors in Europe are 14.8 to 23.5, 2.3 to 5.2 and 0.20 to 0.23 kg  $\text{NH}_3$ -N/yr-animal for dairy, swine, and poultry, respectively. The corresponding emission factors estimated for the U.S. based mainly on European data are 18.7 to 18.9, 4.7 to 6.0 and 0.18 to 0.24 kg  $\text{NH}_3$ -N/yr-animal for dairy, swine and poultry, respectively. Emission factors need further determination, especially for livestock and poultry production in the U.S. Also, use of a composite emission factor should be discouraged and emission factors for different production management systems and subsets of animal species (e.g., for pigs: sows/piglets, growing pigs, gestating sows, boars, etc.) should be developed and used instead. This would allow more accurate determination of ammonia emission for specific animal feeding operations. Another deficiency with the emission factors is that they are usually developed from measurements taken over short periods of time, during which, the weather, operating conditions, and animal sizes and numbers may not represent the annual average conditions. This leads to under or over estimation of ammonia emission factors when the value obtained during the short period of measurement is extrapolated annually. Thus, it is important to develop reliable and accurate measurement methods, and to develop capability to have continuous monitoring for long periods if accurate annual emission factors are to be determined. Also, increased emphasis on changing diets to reduce N excretion and other management changes to reduce ammonia emission have the potential to significantly change  $\text{NH}_3$  emissions. This makes it necessary to reevaluate the emission factors developed earlier to incorporate changing trends in animal production operations.

Measurements of  $\text{NH}_3$  concentration and flux provide a basis for formulating emission factors for the different sources at an animal production facility. They are necessary for estimating inputs for models and determining the effects of management changes for controlling emissions. Measurement methods currently used include chemiluminescence analyzers, denuders, detector tubes, optical absorption techniques, wet chemistry and gas chromatography. Ammonia fluxes are estimated using N mass balance, micrometeorological, chamber and wind tunnel, and tracer gas methods. Comparisons of various methods for measuring  $\text{NH}_3$  fluxes can yield differences of greater than 200%. Agreement within 20-30% for different methods is generally considered good. Lack of a proven "ground-truth" method makes it difficult to calculate absolute errors. Mass balance on N should be considered with every measurement as a check on reasonability of  $\text{NH}_3$  emission measurements even though it may be difficult to determine the fate of all the N that is consumed by an animal.

Models are important for predicting emissions for different situations and the effects of changes of the factors that affect  $\text{NH}_3$  emissions. Some empirical and mechanistic models have been published for  $\text{NH}_3$  emission in buildings and from storage/treatment facilities and land application.

Most of the empirical models use statistics to obtain correlations and relationships between factors that affect NH<sub>3</sub> emission. Mechanistic models are built based on the emission processes for the NH<sub>3</sub> source and NH<sub>3</sub> transfer to the atmosphere. Transport and deposition models usually are based on Gaussian dispersion. However, the errors of the models are usually difficult to determine.

Reducing NH<sub>3</sub> loss from an animal feeding operation requires a whole farm systems approach, which shows how intervening in one aspect of the farm may affect NH<sub>3</sub> losses in other parts of the operation. Strategies for reducing NH<sub>3</sub> losses should be directed towards reducing: (1) NH<sub>3</sub> formation, (2) NH<sub>3</sub> losses immediately after it has been formed, or (3) the NH<sub>3</sub> loss potential. Some of the control practices that are potentially useful for reducing NH<sub>3</sub> loss from animal production facilities are summarized in Table 1.

### Research Needs and Issues

Until recent years, most concern for ammonia lost from manure was because of the influence on reducing fertilizer value. Some of the previous research on N losses during storage, treatment and land application was also useful for determining overall ammonia losses to the atmosphere. However, much additional research is needed to specifically address ammonia losses from animal feeding operations and the nature and extent of environmental and health effects resulting from ammonia emissions. Specific research needs are:

- Determination of environmental impacts of NH<sub>3</sub> deposition on land, crops, and water.
- Determination of on-farm and off-farm health effects of NH<sub>3</sub>.
- Evaluation and standardization of NH<sub>3</sub> concentration measurement methods and NH<sub>3</sub> emission or flux methods.
- Improved determination of emission factors for various animal types and sizes and for various animal and manure management facilities and practices.
- Improvement and validation of models for NH<sub>3</sub> emission, transport, and deposition.
- Evaluation of the effectiveness of technologies and control strategies.
- Economic evaluation of control strategies.

TABLE 1. Potentially useful ammonia control practices for animal production.

	Source or Location			
	Excreted Manure and Urine	Confinement Facilities	Treatment & Storage	Land Application
Control Practice	<ul style="list-style-type: none"> <li>• Reduce N excreted by reduced protein diets or improved balance of amino acids.</li> <li>• Dietary electrolyte balance, affecting urinary pH.</li> </ul>	<ul style="list-style-type: none"> <li>• Minimize emitting surface area.</li> <li>• Remove manure frequently (belt transport, scrape and/or flush).</li> <li>• Filter exhaust air (bioscrubbers, biofilters, or chemical scrubbers).</li> <li>• Manure amendments (acidifying compounds, organic materials, enzymes, and biological additives).</li> </ul>	<ul style="list-style-type: none"> <li>• Cover to reduce emissions or collect gas.</li> <li>• NH<sub>3</sub> stripping, absorption and recovery.</li> <li>• Chemical precipitation e.g. struvite.</li> <li>• Biological nitrification (aerobic treatment).</li> <li>• Acidifying manure.</li> </ul>	<ul style="list-style-type: none"> <li>• Injection or incorporation into soil soon after application.</li> <li>• Application method to reduce exposure to air (e.g. low-pressure irrigation near surface, drag or trail hoses).</li> <li>• Acidifying manure.</li> </ul>

## 2 PURPOSE

The purpose of this paper is to summarize the state of knowledge regarding ammonia emissions from confined animal feeding operations. Based on the information in published literature, the paper summarizes:

1. Environmental effects of ammonia emitted from animal production.
2. Emission rates and quantities from animal buildings, storage and treatment facilities, and land application sites.
3. Ammonia emission measurement methods.
4. Models for ammonia emissions, transport, and deposition.
5. Possible control strategies and technologies.

## 3 BACKGROUND

Ammonia (NH<sub>3</sub>) is a colorless gas with a pungent smell and has an odor threshold of approximately 5 ppm (Devos et al., 1990). Ammonia can cause irritation of the eyes and respiratory tract. The threshold limit values (TLV) established by ACGIH (American Conference of Governmental Industrial Hygienists) for humans as reported by De Boer et al. (1991) are: (1) 25 ppm (17.4 µg/m<sup>3</sup>) for the time weighted average (TLV-TWA) (concentration for a normal 8-h work day and a 40-h work week to which nearly all workers may be repeatedly exposed daily without adverse effect), and (2) 35 ppm (24.4 µg/m<sup>3</sup>) for short-term exposure limit (TLV-STEL) (the concentration to which workers can be exposed continuously for a short period of time without suffering adverse effects; the exposure should be no longer than 15 minutes, four times a day, with at least 60 minutes between successive exposure).

Atmospheric NH<sub>3</sub> is produced by biogenic decomposition of organic materials, biomass burning, and fertilizer production and utilization. Ammonia is involved in plant metabolism and can be exchanged between vegetation and the atmosphere. Agricultural activities, livestock production in particular, have been reported to be the largest contributor to NH<sub>3</sub> emissions (ApSimon et al., 1987; Allen et al., 1988; Battye et al., 1994; Sommer and Hutchings, 1995; Kurvits and Marta, 1998; Koerkamp et al., 1998; Hobbs et al., 1999; Aneja et al., 2000).

Ammonia emissions abatement has had high environmental priority in parts of Europe in recent years, and it is receiving more attention in the U.S. as a potential air and water quality concern. There is much published information about the quantities and impact of NH<sub>3</sub> volatilization into the environment in recent years and there is a need to summarize this information.

### 3.1 Ammonia Production and Volatilization

Ammonia production and volatilization processes that occur in animal operations are well understood (Zhang et al., 1994; Genermont and Cellier, 1997; Monteny et al, 1998; Aarnink and Elzing, 1998) and are summarized in Figure 1. These processes may occur in animal buildings, manure storage or treatment structures, and during/after land application of manure.

#### 3.1.1 Ammonia Production

Ammonia is produced as a consequence of bacterial activity involving organic N substrates. Farm animals consume a considerable amount of protein and other nitrogen (N) containing substances with their feed. The dietary N consumed by the animal is partitioned between products such as meat, milk, eggs, urine, and feces (Tamminga, 1992; Sommer and Hutchings, 1997). The conversion of the dietary N to animal product is often inefficient and 50 to 80% of the N consumed is excreted (Tamminga, 1992). More than 50 to 60% of the excreted N by pigs and cattle is in the

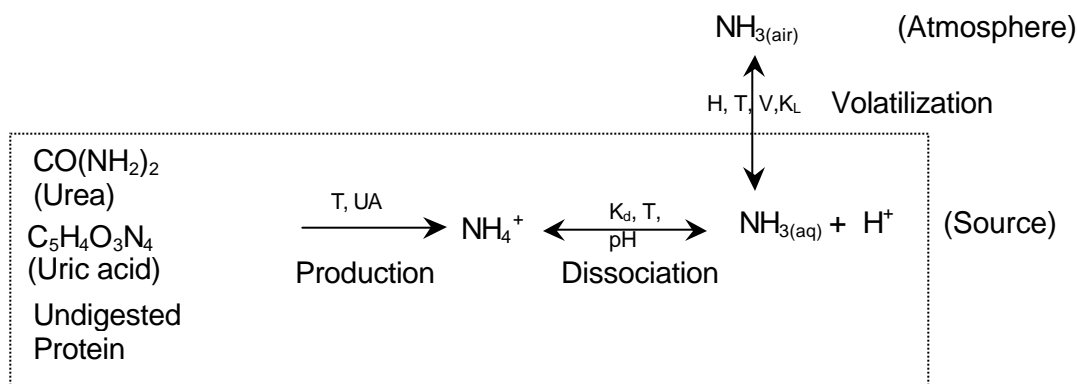
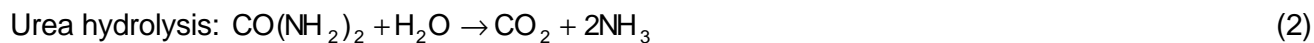
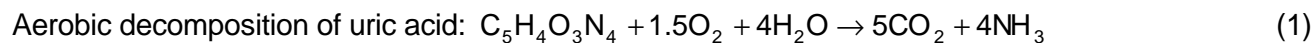


Figure 1. Ammonia Production and Volatilization Equilibria (H - Henry's Law constant,  $K_d$  - dissociation constant,  $K_L$  - mass transfer coefficient, pH, T - Temperature, UA - Urease activity, V- wind speed).

urine and over 70% of the N in the urine is urea-N (Bristow et al., 1992; Tamminga, 1992, Aarnink et al., 1995). In poultry, more than 70% of the total N excreted is uric acid (Koerkamp, 1994).

The primary sources of  $\text{NH}_3$  in livestock and poultry production are urea and uric acid, respectively. Hydrolysis of urea and uric acid to produce total ammoniacal N ( $\text{TAN} = \text{NH}_4^+\text{-N} + \text{NH}_3\text{-N}$ ) occurs very rapidly, requiring only hours for substantial conversion and days for complete conversion (Muck and Steenhuis, 1981; Olesen and Sommer, 1993; Beline et al., 1998; Bussink and Oenema, 1998). Other organic N compounds in feces is a secondary source of TAN, which in this time frame (hours to few days) can account for up to 35% of the production. In total, rapid processes convert about 35% of the total organic N initially in manure to TAN. Over longer time periods, mainly during storage, a total of 50 to 70% of the organic N can be converted to TAN (USDA, 2000).

The biochemical degradation processes of uric acid, urea and undigested proteins to produce TAN are complex but can be simplified as shown in equations 1 to 3 (Koerkamp et al., 1998).



The degradation of uric acid and undigested protein is influenced by temperature, pH, and moisture content (Elliot and Collins, 1982; Whitehead and Raistrick, 1993; Bussink and Oenema, 1998). Urea hydrolysis is influenced by urease activity, pH, and temperature (Elzing and Monteny, 1997).

### 3.1.2 Ammonia Volatilization

Ammonia volatilization is controlled by several factors including TAN concentration, pH, temperature, wind speed, chemical and microbiological activities, diffusive and convective transport in the manure, and gas phase resistance in the boundary layer between the source and the atmosphere. High urease activity, warm temperatures, large emission surface area, and high pH and air velocity (e.g. windy conditions) enhance  $\text{NH}_3$  volatilization. Ammonia volatilization increases linearly with TAN concentration, and curvilinearly with temperature, wind speed, and solution pH (Sommer et al., 1991; Olesen and Sommer, 1993).



Temperature and pH have been reported to be two of the most important factors that influence  $\text{NH}_3$  volatilization. The influence of temperature and pH on  $\text{NH}_4^+/\text{NH}_3(\text{aq})$  equilibrium in an aqueous solution is shown in Figure 2. Ammonia volatilization rate from an aqueous solution increases with temperature because as temperature increases, the solubility of  $\text{NH}_3$  decreases. Secondly, temperature influences the  $\text{NH}_4^+/\text{NH}_3$  equilibrium. At a given pH, the fraction of unionized  $\text{NH}_3(\text{aq})$  in solution increases with temperature. Thirdly, higher temperatures increase mineralization of organic matter, which may increase  $\text{NH}_3$  production (TAN concentration). Model calculations have shown pH to be one of the most important factors controlling  $\text{NH}_3$  volatilization from lagoon liquid (Westerman et al., 1999), stored manure (Muck and Steenhuis, 1982; Olesen and Sommer, 1993; Sommer and Sherlock, 1996) and land applied manure (van der Molen et al., 1990a; Hutchings et al., 1996). Below a pH of 7.0,  $\text{NH}_3$  volatilization will not occur because the  $\text{NH}_4^+$  form is virtually 100%.

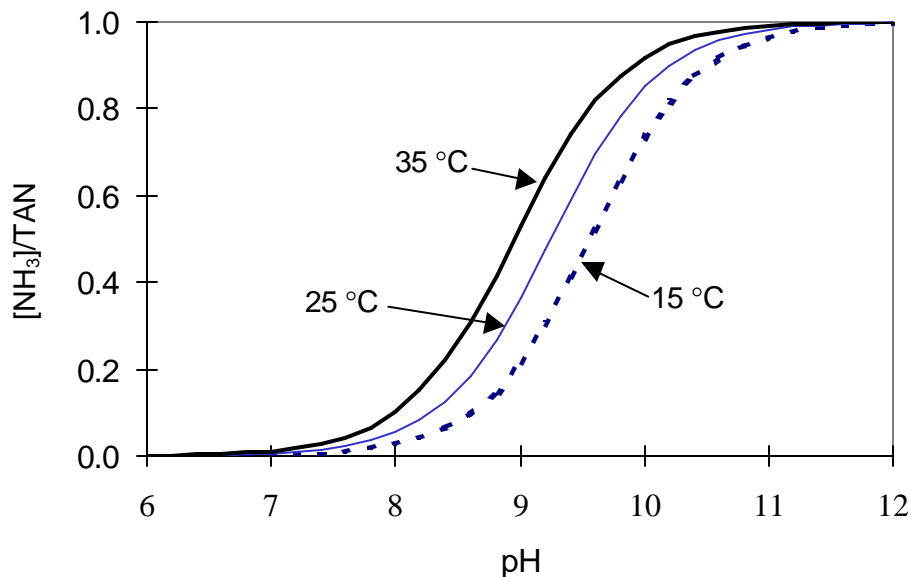


Figure 2. Effect of temperature and pH on the fraction of unionized ammonia ( $\text{NH}_3$ ) concentration compared to TAN concentration present in an aqueous solution.

In addition to temperature, ionic strength of the solution may also affect the  $\text{NH}_4^+/\text{NH}_3$  equilibrium, specifically the dissociation constant ( $K_d$ ), which is defined as

$$K_d = \frac{[\text{NH}_3][\text{H}^+]}{[\text{NH}_4^+]} \quad (4)$$

where:  $[\text{NH}_3]$ ,  $[\text{H}^+]$  and  $[\text{NH}_4^+]$  are molar concentrations of the respective compounds.

The presence of dissolved solutes in the solution will alter the relative equilibrium concentrations of  $\text{NH}_4^+$  and  $\text{NH}_3$  thereby changing  $K_d$  as compared to values in clean water (Hashimoto and Ludington, 1971; Snoeyink, 1980; Zhang, 1994). For example, the  $K_d$  of  $\text{NH}_3$  for concentrated chicken manure is one-sixth the value of  $K_d$  in dilute anhydrous  $\text{NH}_3$  solution (Hashimoto and Ludington, 1971) and the  $K_d$  value of  $\text{NH}_3$  in 1% total solids liquid swine manure is one-fifth of  $\text{NH}_3$   $K_d$  in water (Zhang et al., 1994).

Ammonia volatilization from an aqueous to the gas phase is commonly calculated using Henry's law. One underlying assumption in these calculations is that Henry's law is valid for dilute aqueous

systems, which for NH<sub>3</sub>, are concentrations up to 1000 mg/L (Anderson et al., 1987). The Henry's law constant may be expressed in units of (1) pressure (e.g. atm), (2) pressure-volume/mass concentration (e.g. atm-m<sup>3</sup>/mol), or (3) as a dimensionless (no units) quantity (Staudinger and Roberts, 1996). Equation 5 shows a definition of the dimensionless form of Henry's law constant for the partitioning of NH<sub>3</sub> volatilization from liquid (aqueous) to the gas phase (air).

$$H = \frac{[\text{NH}_{3(\text{air})}]}{[\text{NH}_{3(\text{aq})}]} \quad (5)$$

where: H is the Henry's law constant (dimensionless), [NH<sub>3(air)</sub>] is the NH<sub>3</sub> concentration in the air (g/m<sup>3</sup> or mol/m<sup>3</sup>), and [NH<sub>3(aq)</sub>] is the NH<sub>3</sub> concentration in the liquid phase (g/m<sup>3</sup> or mol/m<sup>3</sup>).

Ammonia volatilization from manure to the atmosphere is normally reported as a mass flux, defined as the product of the difference in NH<sub>3</sub> concentration between the source and the atmosphere and a mass transfer coefficient.

$$\frac{dM}{dt} = K_L ([\text{NH}_{3(\text{aq})}] - [\text{NH}_{3(\text{air})}]) \quad (6)$$

where: K<sub>L</sub>- mass transfer coefficient for NH<sub>3</sub> (m/s), and dM/dt is the mass of NH<sub>3</sub> released in time t (g/s).

#### 4 ENVIRONMENTAL IMPACTS OF AMMONIA FROM ANIMAL PRODUCTION

Ammonia volatilization is considered to be the main pathway for N loss from animal production operations. It is a critical issue because it represents a loss of fertilizer value and can adversely impact the environment (McGinn and Janzen, 1998; Harper et al., 2000). The NH<sub>3</sub> volatilized to the atmosphere at one location and deposited to land at other locations may be beneficial to plants as a nutrient for growth. Conversely, deposition of excess N in environments where the natural N supply is low or in N-sensitive ecosystems may impact these systems negatively.

Although N is a critical nutrient for the survival of microorganisms, plants and animals, it is detrimental at concentrations above certain threshold concentrations (Kirchmann et al., 1998; Kurvits and Marta, 1998; Jongbloed et al., 1999). Potential consequences associated with exceeding threshold concentrations of both oxidized and reduced forms of N include: (1) respiratory diseases caused by exposure to high concentrations of fine particulate aerosols (PM<sub>2.5</sub>); (2) nitrate contamination of drinking water; (3) eutrophication of surface water bodies resulting in harmful algal blooms and decreased water quality; (4) vegetation or ecosystem changes due to higher concentrations of N; (5) climatic changes associated with increases in nitrous oxide (N<sub>2</sub>O); (6) N saturation of forest soils; and (7) soil acidification via nitrification and leaching.

Ammonia is the most prevalent alkaline gas in the atmosphere and it readily combines with acidic species such as sulfur dioxide (SO<sub>2</sub>), nitric acid (HNO<sub>3</sub>) and sulfuric acid (H<sub>2</sub>SO<sub>4</sub>) to form aerosols such as ammonium nitrate (NH<sub>4</sub>NO<sub>3</sub>), ammonium bisulfate (NH<sub>4</sub>HSO<sub>4</sub>), and ammonium sulfate ((NH<sub>4</sub>)<sub>2</sub>SO<sub>4</sub>). These aerosols may cause haze and impair visibility characteristics in the atmosphere (Apsimon and Kruse, 1991; Barthelmie and Pryor, 1998). For example, visibility degradation and impairment in the Lower Fraser Valley, British Columbia, Canada, was associated with fine aerosols formed by NH<sub>3</sub> emitted to the atmosphere (Barthelmie and Pryor, 1998). The atmospheric aerosol, NH<sub>3</sub>, and NH<sub>4</sub><sup>+</sup> concentrations were 1 to 2, 0.3 to 12, and 0.03 to 1 μg/m<sup>3</sup>, respectively. Fine aerosol concentrations and visibility impairment were most severe in the portions of Lower Fraser Valley with the greatest concentrations of poultry and dairy farms. More detail on particulate formation from NH<sub>3</sub> emissions from animal feeding operations are presented in Auvermann et al. (2001).

Transportation of volatilized  $\text{NH}_3$  over long distances depends on the competition between upward diffusion and transformation to  $\text{NH}_4^+$  aerosols, and surface deposition (Kruse et al., 1993). The lifetime of  $\text{NH}_3$  in the atmosphere is short (0.5 h to 5 d) because of rapid gas to particle conversion of  $\text{NH}_3$  to  $\text{NH}_4^+$  and deposition to natural surfaces, particularly wet surfaces and vegetation (Fowler et al. 1997; Walker et al., 2000). Some  $\text{NH}_3$  will disperse into the clouds where it impacts the cloud chemistry (Seinfeld, 1986; Apsimon and Kruse, 1991; Suh et al., 1992; Kruse et al., 1993). If  $\text{NH}_3$  is absorbed in the clouds, it may increase the pH of the cloud droplets and as a result increase the uptake and oxidation of  $\text{SO}_2$  (Apsimon et al., 1987; Kruse et al., 1993). Raising the cloud pH enhances the oxidation of  $\text{SO}_2$  by ozone ( $\text{O}_3$ ) to sulfate ( $\text{SO}_4^{2-}$ ), which is important in facilitating the removal of  $\text{SO}_x$  in precipitation (Barthlemie and Pryor, 1998). The ozone reaction also complements in-cloud oxidation by  $\text{H}_2\text{O}_2$ , which is more effective at higher pH and reduces sharply as pH falls below 4.

Ammonia may also neutralize acidic aerosols, a role which has received considerable attention regarding environmental acidification and health effects of atmospheric aerosols (Seinfeld, 1986; Suh et al., 1992). Although epidemiological studies results are not conclusive, there is evidence that exposure to acid aerosols (defined as having pH less than 4.3) has a more significant deleterious impact on human health than exposure to neutral aerosols of the same size distribution (Suh et al., 1992). However, this apparent health benefit which may be achieved by the neutralizing effect of  $\text{NH}_3$  on acidic aerosols may be offset by the role  $\text{NH}_3$  plays in enhancing aerosol formation.

Ammonia concentration and distribution near animal production facilities and its impact on the environment surrounding these facilities has been documented in several studies (Luebs et al., 1974; Allen et al., 1988; Fangmeier et al., 1994; Barthelmie and Pryor, 1998; Fowler et al., 1998; Pitcairn et al., 1998; Kirchmann et al., 1998). Obviously, the  $\text{NH}_3$  concentrations were higher near the source (Fangmeier et al., 1994; Fowler et al., 1998; Pitcairn et al., 1998) because of its high deposition velocity (van der Eerden et al., 1998). Fangmeier et al. (1994) reported 50 and 70%  $\text{NH}_3$  concentration reductions at 0.6 and 4 km from the source, respectively. Fowler et al. (1998) reported annual mean  $\text{NH}_3$  concentrations at 15 m from a 120,000 broiler chicken as 23 to 63  $\mu\text{g}/\text{m}^3$ , which declined to the background concentration of 1 to 2  $\mu\text{g}/\text{m}^3$  at 270 m. Estimated mass of ammonia nitrogen ( $\text{NH}_3\text{-N}$ ) deposited were 42 and 5 kg N/ha at 15 and 270 m, respectively. Allen et al. (1988) measured the ground level concentrations of  $\text{NH}_3$  and  $\text{NH}_4^+$  at 19 sites over a period of 17 months. Spatial variations in  $\text{NH}_3$  were related to local sources with nearby livestock farming causing pronounced elevation in concentrations. Seasonal variations were also evident with the highest concentrations occurring in the warmer months of the year. Concentrations of  $\text{NH}_4^+$  were more uniform than  $\text{NH}_3$ .

Deposition of  $\text{NH}_3/\text{NH}_4^+$  affect the growth of forests, crops and other types of vegetation (Nihlgard, 1985; Fangmeier et al., 1994; Kurvits and Marta, 1998; Pitcairn et al., 1998). Forests (conifers) near livestock and poultry farms, manure storages or fertilizer fabrication industries have been observed to develop needle necrosis leading to needle losses and eventually, death of the tree because of  $\text{NH}_3$  deposition (Fangmeier et al., 1994). Deposition of N compounds such as  $\text{NH}_3/\text{NH}_4^+$ ,  $\text{NO}_x$  and  $\text{NO}_3^-$  on forests has been reported to be one of the key factors that triggered the severe die back of forests in Europe (Nihlgard, 1985). Pitcairn et al. (1998) reported the relationship between N deposition, species composition and foliar N concentrations in woodland flora in the vicinity of livestock (poultry, pig and dairy) farms by measuring  $\text{NH}_3$  concentration, species composition and tissue N of a range of plant species.  $\text{NH}_3$  concentrations were large at the edges of the woodland close to the livestock buildings (annual means 20-60  $\mu\text{g}/\text{m}^3$ ). The ground flora species composition showed marked changes within 300 m downwind of the buildings. The numbers of the species native to this area were reduced and the "weed species" were abundant closer to the buildings.

Visible injuries to pine and spruce needles were observed immediately downwind of the buildings. Foliar N concentration of a number of the species was large close to the building and declined with distance. The estimated total N deposition at the woodland boundaries ranged from 40-80 kg N/ha-yr which exceeds the critical loads (15-20 kg N/ha-yr) for acidic coniferous forests.

Controlled exposures in greenhouses with acute  $\text{NH}_3$  concentrations (2000-2800  $\mu\text{g}/\text{m}^3$ ) showed that visible symptoms differ with plant species (Fangmeier et al., 1994). For example, in broad-leaved trees black discoloration of the leaves was observed whereas conifers mostly showed brown necrosis. However, in most cases, leaf injury from acute  $\text{NH}_3$  concentrations may not be easily distinguished from symptoms caused by other stresses. Other visible symptoms that have been observed in plant organs due to  $\text{NH}_3$  exposure other than leaves include reduced flowering in *Arnica montana* (53  $\mu\text{g}/\text{m}^3$  for 15 months) and *Petunia hybrida* (> 2000  $\mu\text{g}/\text{m}^3$  for two weeks) and empty ear development in oats (2300-3900  $\mu\text{g}/\text{m}^3$ ).

When  $\text{NH}_3$  and  $\text{NH}_4^+$  are deposited onto the soil,  $\text{NH}_4^+$  nitrification may occur. This oxidation process can result in soil acidification and possible long term plant nutrient imbalances of calcium, potassium and magnesium (Van der Molen et al., 1990a). During the nitrification process, one mole of  $\text{NH}_4^+$  produces 2 moles of  $\text{H}^+$  (Sommer and Jacobsen, 1999). Therefore, nitrification may reduce  $\text{NH}_3$  volatilization by reducing the  $\text{NH}_4^+$  concentration and also reducing the soil pH.

## 5 AMMONIA EMISSION FROM ANIMAL PRODUCTION FACILITIES

Important factors that determine the amount of  $\text{NH}_3$  emitted from animal operations are: (1) number, age, and type of animals; (2) housing design and management; (3) type of manure storage and treatment; (4) land application technique; (5) N excretion rates per animal; and (6) environmental conditions (Leneman et al., 1998). A growing number of animal producers are attracted to the use of large, intensively managed production units due to the benefits of economy of scale, especially in labor, feed, and facility management. These systems produce large amounts of manure making manure storage/treatment and utilization an important management consideration. Animal and manure management can vary substantially within and between regions. For example, dairy cows may be housed in enclosed barns or open sided barns, and spend variable amounts of time outside the barn depending partly on climate. Beef cattle may be in pasture, barns, or open air feedlots. Most of poultry are raised indoors but manure management varies from flushing systems with treatment/storage lagoon to relatively dry litter/manure removed about once a year. In the Midwestern States of the U.S., pig manure is generally collected and stored in pits (inside or outside the barns) and then spread on agricultural land. In South and Southeastern States, lagoons are commonly used for pig manure treatment and storage with the effluent irrigated to nearby cropland or evaporated to the atmosphere (evaporation ponds). The management of the animals and manure can substantially affect  $\text{NH}_3$  emission. The emphasis of this paper is on  $\text{NH}_3$  emission from confined animal feeding operations, and not from animals on pasture. However, some of the composite emission factors presented later for Europe include emissions from animals from pasture.

### 5.1 Units

Emission rates are usually expressed in terms of mass of  $\text{NH}_3$  or ammonia nitrogen ( $\text{NH}_3\text{-N}$ ) per unit time and per animal (or live weight units) or per unit area (surface sources). Although air quality literature often uses units of  $\text{NH}_3$  mass in reporting emission data, this paper uses  $\text{NH}_3\text{-N}$  mass because it simplifies its use in N accounting for confined animal production. To convert  $\text{NH}_3\text{-N}$  mass to  $\text{NH}_3$  mass, multiply by 1.214.

## 5.2 Emission Factors

Ammonia emission factors for animals in an animal production operation represent the sum of the annual mean emission rates from housing, manure storage/treatment, and land application of manure. The emission factors are commonly expressed as mass/year-animal, e.g. for pigs, kg NH<sub>3</sub>-N/yr-pig. Ammonia emission factors can be used to develop emission inventories for countries or regions (Hutchings et al., 2001; Misselbrook et al., 2000; Pain et al., 1998; Battye et al., 1994). The total NH<sub>3</sub> emitted by an animal species is usually the product of a composite emission factor and the total number of animals (pigs, dairy cattle, poultry, or beef cattle) in production. The composite emission factor for an animal species is developed to represent the mix of animal type, size, and production conditions. The composite emission factor is a weighted average across several sizes and type, e.g. pigs would include sows, gilts, boars, piglets, and growing (finishing) pigs.

Ammonia emission factors commonly used in the U.S. were developed by Battye et al. (1994) for the U.S. Environmental Protection Agency. These emission factors are based on studies done in Europe, specifically, the Netherlands (Asman, 1992). One problem of using these emission factors is that they are not specific to styles of management in farms in the U.S. Animal production systems and management in the U.S. are often different from those in Europe. Therefore, it is important that emission factors for the U.S. systems be developed from data taken in the U.S. Previously, little research on ammonia emissions has occurred in the U.S., but research is increasing. Progress is being made in development of measurement equipment and methods, but the expense of measurement and lack of continuous measurement capability has hindered the development of reliable annual emission factors. Comparisons of composite emission factors for cattle, pigs, and poultry for the U.S., Europe, and some countries in Europe are presented in Table 2.

The composite emission factor for pigs in the U.S. presented in Table 2 (Battye et al., 1994) is almost double that in Europe from which it was derived. Based on the emission factors for pigs reported by Asman (1992), a possible reason for this discrepancy is in the way Battye et al. (1994) assigned emission factors to different animal classifications and weight for U.S. pigs. Battye et al. (1994) classifies 18.6 million (33% of the total hog population) market hogs in three weight categories (54.1 to 81.2, 81.3 to 99.3, and greater than 99.4 kg) as mature boars with an emission factor of 11 kg NH<sub>3</sub>/yr-pig. This is an overestimation because in the U.S., these weight categories of market hogs should be finishing (fattening) pigs with an emission factor of 6.98 kg NH<sub>3</sub>/yr-pig. Also, the classification of hogs by Battye et al. (1994) suggests a boar:sow ratio of 3:1, which is not correct for hog production. Finally, Battye et al. (1994) assigns pigs less than 27.2 kg (18.7 million - also 33% of the total hog population) an emission factor of 6.98 kg NH<sub>3</sub>/yr-pig similar to fattening pigs while Asman (1992) uses zero for this weight (< 20 kg) category and states that emissions from these small pigs are included in those of the fattening pigs.

We recalculated the emission factor for pigs in the U.S. using the pig classification and population (for December 1991 inventory) given by Battye et al. (1994) (assuming the boar designation was incorrect) and emission factors from Asman (1992). The results are presented in Table 3. In our calculations, we added a classification for boars and used a boar:sow ratio of 1:15 to determine the boar population. Using this ratio, we also split the classification 'Other' into boars and sows not farrowing. The recalculation indicates NH<sub>3</sub> emission factor of 5.68 kg NH<sub>3</sub>/yr-pig (4.7 kg NH<sub>3</sub>-N/yr-pig) compared to 9.20 kg NH<sub>3</sub>/yr-pig (7.6 kg NH<sub>3</sub>-N/yr-pig) from Battye et al., 1994. This recalculation represents a 38% reduction in the NH<sub>3</sub> emission factor for pigs in the U.S. based on 1991 inventory of pigs. Preferably, the emission factor for pigs in the U.S. should be based on current pig inventory and emission rates determined in the U.S. However, emissions data from U.S. facilities are not adequate to make calculations of emission factors with confidence at this time.

TABLE 2. Composite ammonia emission factors (kg NH<sub>3</sub>-N/yr-animal).

Animal	Emission factor <sup>a</sup>	Country/Region	Reference
<b>Cattle <sup>b</sup></b>			
Beef and Dairy	14.8	Europe	Buijsman et al., 1987
Beef and Dairy	15.4	Scotland	Sutton et al., 1995
Beef and Dairy	16.6	North Ireland	Sutton et al., 1995
Beef and Dairy	17.0	U.K.	Sutton et al., 1995
Beef and Dairy	17.4	England & Wales	Sutton et al., 1995
Beef and Dairy	18.7	U.S.	Bowen and Valiela, 2001
Beef and Dairy	18.9	U.S.	Battye et al., 1994
Beef and Dairy	19.0	Europe	Asman, 1992
Beef and Dairy	22.1	German Democratic Rep.	Moller & Schieferdecker, 1989
Dairy	18.8	Denmark	Hutchings et al., 2001
Dairy	19.4	U.K.	Pain et al., 1998
Dairy	21.8	U.K.	Misselbrook et al., 2000
Dairy	23.5	Europe	Van der Hoek, 1998
Other Cattle	5.61	U.K.	Misselbrook et al., 2000
Other Cattle	5.92	Denmark	Hutchings et al., 2001
Other Cattle	11.8	Europe	Van der Hoek, 1998
<b>Pigs</b>			
	2.3	Europe	Buijsman et al., 1987
	4.3	U.K.	Sutton et al., 1995
	4.3	England & Wales	Sutton et al., 1995
	4.4	Europe	Asman, 1992
	4.4	Scotland	Sutton et al., 1995
	4.5	North Ireland	Sutton et al., 1995
	5.2	German Democratic Rep.	Moller & Schieferdecker, 1989
	6.0	U.S.	Bowen and Valiela, 2001
	7.6	U.S.	Battye et al., 1994
	4.7 <sup>c</sup>	U.S.	This paper – modified Battye et al., 1994
<b>Poultry</b>			
	0.18	U.S.	Battye et al., 1994
	0.20	Europe	Asman, 1992
	0.22	U.K.	Sutton et al., 1995
	0.22	England & Wales	Sutton et al., 1995
	0.23	Scotland	Sutton et al., 1995
	0.23	North Ireland	Sutton et al., 1995
	0.24	U.S.	Bowen and Valiela, 2001

<sup>a</sup> Composite emission factors. <sup>b</sup> Beef and Dairy – includes all cattle; Dairy – includes Dairy cows only; Other Cattle – Cattle other than dairy cows. <sup>c</sup> Recalculated emission factor for swine. 1 g NH<sub>3</sub> = 1.214 g NH<sub>3</sub>-N.

Ammonia emissions within an animal species vary by age, size, and type (e.g. for pigs, Table 3). Therefore, using composite emission factors to determine local or regional NH<sub>3</sub> emission may be under- or overestimated, depending on the predominant animal type and size in that region. For example, based on emission factors listed in Table 3 for different pig sizes, using the composite emission factor (5.69 kg NH<sub>3</sub>/yr-pig) to calculate NH<sub>3</sub> emission for a region that predominantly produces finishing pigs (emission factor = 6.98 kg NH<sub>3</sub>/yr-pig) would under estimate emissions. Therefore, to allow more accurate determination of ammonia emission from animal feeding operations, using composite emission factor should be discouraged. Emission factors for different

TABLE 3. Recalculated ammonia emission factors for the pigs in the U.S.

Classification	Population × 10 <sup>6</sup>		Emission Factor (kg NH <sub>3</sub> /yr-pig) (Asman, 1992)	
			Battye et al., 1994	This Paper
<u>Kept for breeding</u>				
Sows farrowing	6.02		16.13	16.13
Other	1.25	0.796 <sup>a</sup>	5.22	5.22
Boars	-	0.454 <sup>a</sup>		11.0
<u>Market hogs by weight group</u>				
Under 27.2 kg	18.7		6.98	0
27.3 – 54 kg	13		6.98	6.98
54.1 – 81.2 kg	10.4		11	6.98
> 81.3 kg	8.4		11	6.98
Total	57.77			
Emission Factor (kg NH <sub>3</sub> /yr-pig)			9.20	5.69
Emission Factor (kg NH <sub>3</sub> -N/yr-pig)			7.6	4.7

<sup>a</sup> For this paper, assumed boar:sow ratio of 1:15. 1 g NH<sub>3</sub> = 1.214 g NH<sub>3</sub>-N.

production management systems and subsets of animal species (e.g., for pigs: sows/piglets, growing pigs, gestating sows, boars, etc.) should be developed and used instead. Beef and dairy cattle and poultry also have different emission factors for depending on the type, age, and size of the animal or bird. Also, increased emphasis on changing diets to reduce N excretion and other management changes to reduce NH<sub>3</sub> emission have the potential to significantly change NH<sub>3</sub> emissions. This makes it necessary to reevaluate the emission factors frequently to incorporate changing trends in animal production operations.

Another deficiency with the emission factors is that they are usually developed from measurements taken over short periods of time, during which, the weather, operating conditions, and animal sizes and numbers may not represent the annual average conditions (Sutton et al., 1995; Leneman et al., 1998; Oudendag and Luesink, 1998). This leads to under or over estimation of ammonia emission factors when the value obtained during the short period of measurement is extrapolated annually. Thus, it is important to develop reliable and accurate measurement methods, and to develop capability to have continuous monitoring for long periods if accurate annual emission factors are to be determined.

Substantial judgement is necessary in selecting and using emission factors to develop NH<sub>3</sub> inventories. It is important to evaluate the assumptions and techniques that were used to develop the emission factors to determine their suitability to the conditions or region for which the inventory is being developed. For example, for dairy cattle, did housing include whole or part year? Did it also include emissions from manure piles, storage ponds and other site emissions? Were the animals grazing or on feed? What is the N content of their diet? How many cows were present in the barn per unit area? What was the manure collection, removal, storage/treatment, and land application technique? Some emission factor reports do not provide the level of detail needed to answer these questions.

One way of checking whether emission factors are reasonable, is to compare them to N excreted by the animal. Estimated annual N excreted by animals in the U.S. (MWPS, 2000) and U.K. (Smith and

Frost, 2000; Smith et al., 2000a) are presented in Table 4. For animal production systems that have more than one group or production cycle per year and some periods without animals in a particular “animal place”, an occupancy time (% of year) needs to be estimated. Then, annual excretion can be calculated per “animal place” if an average daily excretion rate for the animal is known. Due to differences in diet, genetics, production management and animal efficiency, the N excretion can vary  $\pm 30\%$  (MWPS, 2000). Thus, when comparing  $\text{NH}_3\text{-N}$  emission to N excretion, it is important that both the  $\text{NH}_3\text{-N}$  emission and N excretion be accurately determined.

Emission factors can be used to evaluate the contribution of different livestock species and the various stages of manure handling in an animal operation to the total  $\text{NH}_3$  inventory. An example using  $\text{NH}_3$  emission factors to estimate  $\text{NH}_3$  production by different livestock types and the contribution of different manure handling stages in the U.K. is shown in Table 5 (Misselbrook et al., 2000). Housing was the major source of  $\text{NH}_3$  volatilized followed by land application, and storage. For the U.S., Batty et al. (1994) estimated the relative contribution of animal agriculture to the total  $\text{NH}_3$  emission inventory as follows: cattle-43.4%; swine-10.7%; poultry-26.7%; sheep-0.7%. All other sources constituted 18.5% of total estimated  $\text{NH}_3$  emissions. The relative contribution to the  $\text{NH}_3$  inventory depends on livestock numbers and manure management practices.

### **5.3 Emissions from Stages of Production**

#### **5.3.1 Buildings**

Ammonia emission from buildings depends on many factors including: (1) N content of the feed; (2) efficiency of the animal to convert N into products such as meat, milk, and eggs; (3) species, age and weight of the animal; (4) housing type and manure management system in the building; and (5) environmental conditions in the building.

The building emission rate is the sum of the net  $\text{NH}_3$  mass flows through all outlets. From mechanically ventilated buildings  $\text{NH}_3$  emission is generally obtained by measuring  $\text{NH}_3$  concentration in the inlet and exhaust air streams and multiplying these concentrations by the ventilation flow rates (Burton and Beauchamp, 1986; Kroodsma et al., 1993; Swiestra et al., 1995; Ogink and Kroodsma, 1996; Braam et al., 1997). Some studies have assumed zero concentration at the inlet in calculating emissions which results in reported emission rates that are higher than actual rates (Ni et al., 2000b). The inlet concentration can be significantly greater than zero because of exhaust air reentry and from other sources on the farm. For naturally ventilated buildings,  $\text{NH}_3$  emission can be estimated using passive  $\text{NH}_3$  flux samplers (Phillips et al., 1998a) or tracer gas method to determine the ventilation rate to use with  $\text{NH}_3$  concentration measurements (Demmers et al., 1998; 2001).

Various units are reported in literature for expressing emission rate from livestock buildings such as: (1) mass per unit time per animal (e.g. mg/h-pig); (2) mass per unit time per 500 kg live weight or animal unit (AU) (e.g. mg/h-AU); (3) mass per time per animal place or space (kg/yr-pig place); and (4) mass per unit time per unit area (mg/h- $\text{m}^2$ ). Expressing the emission rate per 500 kg live weight or AU implies a linear relationship between weight and emission, which may not be correct because of differences in diets and N excretion for different animal classes and sizes. Estimating emission rate per animal is a composite for all animal classifications and requires that the number and the specific description of the animal, e.g. sow, boars or finisher for pig be stated clearly. The average number of animals during the measurement period is typically used to determine the emission rate per animal. An emission rate per unit time per animal place requires the knowledge of emission rate during the production and the downtime cycles and the number of animals present. Emission rate



TABLE 4. Estimates of total N excreted per animal or animal place for different livestock types in the U.S. and U.K.

Animal	U.S.					U.K.				
	Type	Size <sup>a</sup> (kg LW)	Occupancy (%) <sup>b</sup>	N excreted (kg) Day <sup>c</sup> Year <sup>c</sup>		Type	Size (kg LW)	Occupancy (%)	N excreted (kg) Day    Year	
Dairy Cattle	Lactating cow	450	100 <sup>1</sup>	0.263	96	Heifer (> 2 yr)	500	100	0.16	58
	Lactating cow	635	100 <sup>1</sup>	0.372	135.8	Cow	450	100	0.21	76
	Dry cow	450	100 <sup>1</sup>	0.163	59.5	Cow	550	100	0.27	96
	Dry cow	635	100 <sup>1</sup>	0.227	82.85	Cow	650	100	0.32	116
	Heifer	340	100	0.104	37.96					
	Calf	68		0.023						
	Calf	113		0.036						
Beef Cattle	Feedlot	340	88	0.172	55.24	Calf (0-0.5 yr)	100	50	0.04	7
	Feedlot	500	88	0.245	78.69	Fattener (0.5-1 yr)	180	50	0.07	12
	High forage	340		0.186		Fattener (1-2 yr)	400	100	0.13	47
	High forage	500		0.277		Fattener (> 2 yr)	500	100	0.16	58
	Cow	450		0.141		Suckler cow (> 2 yr)	500	100	0.16	58
	Calf	204		0.064						
Pigs	Nursery	11	90 <sup>2</sup>	0.009	2.96	Weaners	7-18	90	0.009	3
	Grow/Finish	68	95 <sup>3</sup>	0.036	12.48	Growers	18-35	90	0.019	6.1
	Gestating	125	72 <sup>4</sup>	0.023	7.22	Finishers	35-105	90	0.032	10.5
	Lactating	170	84 <sup>5</sup>	0.082	25.14	Gilts	90-130	100	0.036	13
	Boar	160	100	0.023	8.28	Sow + Litter (< 7kg) <sup>6</sup>	130-225	100	0.055	19.5
Poultry	Layers	1.8	96	0.0016	0.56	Layers <sup>7</sup>	2.2	97	0.0018	0.66
	Broiler	0.9	76 <sup>7</sup>	0.0010	0.29	Broiler <sup>7</sup>	2.2	76	0.0018	0.50

U.S. values adapted from MWPS-18 (2000) and UK values from Smith and Frost (2000) and Smith et al. (2000a). The values reported with % occupancy refer to excretion per animal place.

<sup>a</sup> Weights represent the average size of the animal size during the stage of production. <sup>b</sup> Occupancy is for confined animals only. <sup>c</sup> Nitrogen excretion values can vary  $\pm$  30%.

<sup>1</sup> Excretion per animal place assuming 100% occupancy (constant number in milking herd). For a particular cow, there is typically 305 days lactation and 60 days dry period per year. <sup>2</sup> Assumes 9 wks per cycle and 1 wk between cycles. <sup>3</sup> Assumes 2.5 grow outs per year and 1 wk between grow outs. <sup>4</sup> Assumes 114 days gestation period (approximately 2.3 per year). <sup>5</sup> Excretion per sow place includes baby pigs; assumes 21-day lactation period and 4 days between each group. <sup>6</sup> Excretion per sow place based on 2.3 lactations covering 23% of yr and dry period of 77%. <sup>7</sup> Broilers output per 6.6 crops/yr, 42 day cycle.

TABLE 5. Ammonia emission from pig, cattle and poultry production in U.K. in kt NH<sub>3</sub>-N/yr

Source	Cattle	Pigs	Poultry	Total
Housing	42	16.2	27.5	85.7
Storage	15.7	2.8	0.3	18.8
Land spreading	45.1	7.0	14.1	66.2
Grazing/outdoors	15.2	0.9	1.0	17.1
Total	118	26.9	42.9	187.8

Adapted from Misselbrook et al. (2000). 1 g NH<sub>3</sub> = 1.214 g NH<sub>3</sub>-N.

per year per animal place can be estimated as follows (Koerkamp et al., 1998):

$$E_{\text{NH}_3} = (24E_X X + 24E_Y Y) \frac{365}{(X + Y)N_A} \quad (7)$$

Where:  $E_{\text{NH}_3}$  – emission rate mass/yr-animal place; X – production period (days); Y – non production period (days);  $N_A$  – number of animals present in a production period;  $E_X$  and  $E_Y$  – average emission rates (mass/h) during the production and non production periods, respectively.

### 5.3.1.1 Swine Housing

Many swine buildings in the U.S. have concrete floors which are partially or fully slatted. Swine manure is either stored in under-floor pits or removed from the building at various frequencies to an outdoor storage or treatment facility. In swine buildings, NH<sub>3</sub> can be emitted from floors, walls, pen partitions, animals wetted with urine and water leakage, and slurry surface in under floor pit. Manure is removed from buildings using flushing liquid or scrapers.

Ammonia emission increases with age of pigs and fouling of pens (Hoeksma et al., 1992; Aarnink et al., 1995). Feed and water intakes increase with pig age. In general, higher feed intake leads to more N excreted in the urine, which enhances NH<sub>3</sub> emission. Pen fouling increases towards the end of the growing period as the pigs become heavier (Randall et al., 1983; Hoeksma et al., 1992; Krieger et al., 1993; Hacker et al., 1994; Aarnink et al., 1995; Aarnink et al., 1997; Ni et al., 1999). Sometimes, owing to lack of space, especially in slatted floor buildings with under-floor manure storage, the pigs will lie on the slatted floor area, which may cause convective airflow in pits and increase NH<sub>3</sub> emission.

Some of the NH<sub>3</sub> concentration and emission rates from swine buildings are presented in Table 6. It is not possible to directly compare all the emission rates because of the differences in housing, diet, management, periods of measurement, and the units used to report the emission rates. Few data are reported for U.S. Much of the data are reported on the basis of time per hour or day, and thus not appropriate to convert to annual emissions.

### 5.3.1.2 Poultry Housing

Traditionally, layers are raised in battery cage systems where manure can be collected either on a conveyor belt or dropped into a storage pit or pile below the cages. Layers are also raised in deep litter and perchery houses. Broilers and turkeys are almost exclusively raised in deep litter systems. In deep litter systems it is a common practice for birds to be raised on old litter for one year or longer before cleaning out the entire house. Between clean-outs, only caked litter is removed at the end of the grow-out and fresh bedding may be added to the litter (Xin and Berry, 1995).

TABLE 6. Ammonia concentration inside and emissions from swine buildings

	Floor	Manure Storage/ Management	Indoor Temp °C	Concentration ppm	Emission rate (NH <sub>3</sub> -N)	Reference	Country or Region
Farrow	SF	Flushed, 1×daily	21		4.0 kg/yr-pig place	Gastel et al., 1995	Netherlands
Finish	PSF	UFP, 3 wks	22		5.45 g/d-pig place	Aarnink et al., 1997	Netherlands
Finish	FSF	Deep pit	21.1-25.3		2.0-5.1 g/h-AU	Heber et al., 2000	U.S.
Finish	PSF			10-35	2.5 g/h-AU	Hinz and Linke, 1998	Europe
Finish	Straw		8.4-10.5	4.3-9.1	1.2-3.1 g/h-AU	Koerkamp et al., 1998	Europe
Finish	SF		8.4-10.5	12.1-18.2	1.7-2.1 g/h-AU	Koerkamp et al., 1998	Europe
Finish	SF	High-Rise™, UF	14-36	0-19	1.9-26.6 kg/yr-pig	Stowel and Foster, 2000	U.S.
Finish	FSF	Deep pit	19.9		1.1-1.4 g/h-AU	Ni et al., 2000a	U.S.
Finish	PSF	UFP, 1 wk	21		2.45 kg/yr-pig place	Oosthoek et al., 1991	Netherlands
Finish	PSF	Deep pit	20		2.44 kg/yr-pig place	Oosthoek et al., 1991	Netherlands
Finish	PSF	Outside	18		2.53 kg/yr-pig place	Oosthoek et al., 1991	Netherlands
Grow	PSF	UFP, 3 wks	24-26		0.71 g/d-pig	Aarnink et al. 1995	Netherlands
Grow/Finish	SF	Flushed, daily	23		0.48 g/h-AU	Heber et al., 2001	U.S.
Grow/Finish	SF	Drained, 1 wk	25		0.79 g/h-AU	Heber et al., 2001	U.S.
Grow/Finish	SF	Drained, 2 wks	25		0.76 g/h-AU	Heber et al., 2001	U.S.
Grow/Finish	SF	Pit recharge, 1 wk	21		0.20 g/h-AU	Heber et al., 2001	U.S.
Grow/Finish	SF	Pit recharge, 2 wks	21		0.23 g/h-AU	Heber et al., 2001	U.S.
Grow/Finish	SF	Pit recharge, 6 wks	22		0.25 g/h-AU	Heber et al., 2001	U.S.
Grow/Finish	FSF	UFP			4.4 g/h-AU	Demmers et al., 1999	U.K.
Grow/Finish	PSF	UFP, 3 wks	19-23		4.7 g/d-pig	Aarnink et al. 1995	Netherlands
Grow/Finish	PSF	UFP, 3 wks	20-24		5.0 g/d-pig	Aarnink et al., 1996	Netherlands
Grow/Finish	SF	Flushed, 1×daily	19		0.9 kg/yr-pig place	Gastel et al., 1995	Netherlands
Grow/Finish	PSF	Deep pit	14-22	11-14.7	1.65-4.94 g/h-AU	Hendriks et al., 1998	Belgium
Grow/finish		Deep pit	22-31		4.96 g/h-AU	Ni et al., 2000b	U.S.
Nursery	SF	Flushed, 1×daily	23		0.16 kg/yr-pig place	Gastel et al., 1995	Netherlands
Sow	Litter		8.4-10.5	5.1-12.5	0.62-2.68 g/h-AU	Koerkamp et al., 1998	Europe
Sow	SF		8.4-10.5	11-22.1	0.86-1.41 g/h-AU	Koerkamp et al., 1998	Europe
Weaner	PSF	UFP, 6 wks	24-26.5		0.70 g/d-pig	Aarnink et al., 1996	Netherlands
Weaner	PSF	UFP, 3 wks	24-25.4		0.72 g/d-pig	Aarnink et al. 1995	Netherlands
Weaner	SF		8.4-10.5	4.6-7.8	0.55-1.27 g /h-AU	Koerkamp et al., 1998	Europe

AU – Animal unit = 500 kg live weight, FSF-Fully Slatted floor, PSF-Partly slatted floor, SF-Slatted floor, UF-under floor; UFP-under floor pit;

Blanks – Not reported. 1 g NH<sub>3</sub> = 1.214 g NH<sub>3</sub>-N.

Ammonia production by poultry increases with age (Amon et al., 1997). The most important parameters in  $\text{NH}_3$  emission from poultry houses are temperature, pH, air velocity, relative humidity, and water activity, i.e. the availability of water for microorganisms in the litter. High  $\text{NH}_3$  concentrations (50-200 ppm) have been reported in poultry houses due to reusing old litter, reduced ventilation, and excessive moisture in cold weather due to condensation and waterer leakage (Carlile, 1984; Wathes et al., 1997). Condensation is greater in poorly insulated houses in winter and results in wet litter which favors  $\text{NH}_3$  release. Exposing poultry to gaseous  $\text{NH}_3$  for a prolonged period of time can cause reduced body weight and feed consumption as well as an increased susceptibility to keratoconjunctivitis, diseases and respiratory ailments (Elliot and Collins, 1982; Carlile, 1984; Hinz and Linke, 1998b).

Ammonia concentrations and emission rates from poultry buildings are presented in Table 7. There is little data from the U.S. Most of the emission rates are expressed as g/h-AU and varies from 0.5 to 10 g/h-AU. The variation in the emission rates can be due to differences in housing, ventilation, bird sizes, diets, manure management, and other factors.

### **5.3.1.3 Cattle Housing and Feedlots**

Dairy cattle are commonly housed in free or tie stall barns. There are variations in these housing systems due to the use of bedding and manure storage. The barns may have (1) straw bedding, (2) slatted floors with manure storage below the slats, and (3) solid floors and a scraper system to remove the manure to storage. There are also open systems with no roof and walls. Combinations of these housing variations also exist. Beef cattle are mostly raised on pastures and fed (finished) for slaughter in open feedlots. There is little confined beef housing in the U.S.

Ammonia concentration in dairy buildings depends on urine production and the urea-N concentration in the urine (Muck and Steenhuis, 1981). Feces mainly contain organic N that mineralizes very slowly to produce  $\text{NH}_3$  suggesting that  $\text{NH}_3$  volatilization potential of fresh feces is relatively low in buildings (Bussink and Oenema, 1998). However, during long-term storage for buildings with under floor storage, organic N compounds are degraded anaerobically, leading to an increase in ammonium concentration in the manure. Up to 50% of this organic N may become mineralized to  $\text{NH}_3$ -N during storage for six months. The mineralization rate increases with temperature (Whitehead and Raistrick, 1993; Bussink and Oenema, 1998).

For dairy buildings with under-floor manure storage pits,  $\text{NH}_3$  lost per unit floor and manure storage surface area are equal (Voorburg and Kroodsmas, 1992). However, Monteny (2000) indicates that slurry pits accounts for an average 25-40% of the total  $\text{NH}_3$  emission from cubicle (confined housing with animal walking area) dairy houses with slatted floors, which may go up to 80% depending on the temperature difference between the outside and air in the slurry pit.

Ammonia losses from dairy buildings vary during the year. The losses are higher in the summer because of increased indoor temperature and ventilation airflow in the livestock barns (Smits, 1995; Sommer and Hutchings, 1997). The changes in airflow rates influence the internal air distribution patterns and may increase the air speeds above the manure surface.

Calculation of ammonia emissions from dairy and beef cattle operations have been based largely on estimates (Battye et al., 1994; Meyer et al., 1997). The California Air Resources Board (CARB, 1999) published preliminary  $\text{NH}_3$  emissions estimates from typical open-lot dairies and beef cattle

TABLE 7. Ammonia concentration and emission from poultry houses

Poultry type	House/Manure system	NH <sub>3</sub> conc. (ppm)	NH <sub>3</sub> -N emission rate	Reference	Country or Region
Broiler	Litter	24.2	7.6 g/h-AU	Wathes et al., 1997	U.K
Layers	Battery cage/Deep pit	13.5	7.6 g/h-AU	Wathes et al., 1997	U.K.
Layers	Perchery/Deep pit	12.3	7.6 g/h-AU	Wathes et al., 1997	U.K
Layers	Perchery		6.6-8.24 g/h-AU	Phillips et al., 1995	U.K
Layers	Battery cage		5.8-10.1 g/h-AU	Phillips et al., 1995	U.K.
Broiler	Litter		7.0-7.7 g/h-AU	Phillips et al., 1995	U.K.
Broiler	Litter		0.5-6.7 g/h-AU	Amon et al., 1997	Slovenia
Broiler	Litter	8-27.1	1.8-6.8 g/h-AU	Koerkamp et al., 1998	Europe
Broiler	Litter		4.45 g/h-AU	Demers et al., 1999	U.K.
Layers	Battery cage	1.6-11.9	0.5-7.7 g/h-AU	Koerkamp et al., 1998	Europe
Layers	Perchery/Litter	8.3-29.6	6.0-9.0 g/h-AU	Koerkamp et al., 1998	Europe
Layers	Liquid manure		3.6 g/h-AU	Hartung and Phillips, 1994	Germany
Broiler	Litter		74 mg/h-m <sup>2</sup>	Brewer and Costello, 1999	U.S.
Layer	Cage/Deep pit		0.376-0.472 kg/yr-bird	Keener et al., 2001	U.S.
Layer	Cage/Belt		0.125-0.136 kg/yr-bird	Keener et al., 2001	U.S.
Layers	Litter		1.65 g/h-AU	Hartung and Phillips, 1994	Germany
Layers	cage		10 g/h-AU	Yang et al, 2000	U.S.

AU – Animal Unit = 500 kg live weight.

1 g NH<sub>3</sub> = 1.214 g NH<sub>3</sub>-N.

feedlots in California based on the Battye et al. (1994) report. The board concluded that it is not possible to produce precise estimates of livestock emissions as can be done for factories or motor vehicles because of uncertainties in the number of animals and the  $\text{NH}_3$  emission per animal.

Some of the reported emissions for cattle are derived from direct assumptions that a certain percentage of N excreted by the animal is lost due to N volatilization. Without measurements these assumed emissions should be used with much caution. Some of the measurements found were done by Luebs et al. (1974), Hutchinson et al. (1982), and Ashbaugh et al. (1998). Luebs et al. (1974) reported only concentrations of  $\text{NH}_3$  and not emission rates. Hutchinson et al. (1982) reported  $\text{NH}_3$  flux from a beef cattle feedlot to vary from 0.64 to 2.37 kg  $\text{NH}_3$ -N/h-ha. Ashbaugh et al. (1998) reported emissions from a dairy cattle operation to be between 11 and 107 kg/yr-cow, based on low and high emissions that occurred at night and late morning during the measurement period, respectively.

A summary of  $\text{NH}_3$  emission data from cattle buildings and feedlots are reported in Table 8. Very little data is from the U.S. Units for reporting emission rates vary, but most are in g/h-AU or g/h-cow. Feedlot emission can also be reported as g/h-ha.

### 5.3.2 Storage and Treatment Structures

Common manure storage structures on farms include earthen basins, under floor pits, storage tanks, and stockpiles. Storage facilities primarily contain manure for some period of time before utilization. Treatment structures include lagoons and digesters (aerobic and anaerobic) and composting operations. Factors that can influence  $\text{NH}_3$  volatilization from manure storage and treatment structures include wind velocity above the manure surface, temperature, surface cover and the type of treatment process (aerobic or anaerobic) (Voorburg and Kroodsma, 1992; Olesen and Sommer, 1993; Bussink and Onema, 1998; Monteny and Erisman, 1998). Ammonia losses are much higher from manure stored in open tanks and lagoons than manure stored in covered tanks and lagoons (Bussink and Onema, 1998; Hornig et al., 1999). Losses from liquid and solid manure stored outdoors increase with temperature, surface area, and storage duration. An increase in temperature increases the mineralization of organic N in the slurry thereby increasing the production and emission of  $\text{NH}_3$ .

Many livestock production facilities in the U.S. use anaerobic lagoons for liquid manure treatment and temporary storage. Nitrogen concentration in properly operating lagoons should be fairly low compared to raw manure because of dilution water, settling, and volatilization of N-based gases. However, management and climatic variations have a great influence on the lagoon liquid concentrations; for example, average TAN concentrations in lagoons were 280-570  $\text{NH}_3$ -N mg/L in North Carolina (Bicudo et al., 1999) and 613-1143  $\text{NH}_3$ -N mg/L in Missouri (Fulhage and Hoehne, 1999; Zahn et al., 2001). Ammonia volatilization from lagoons has been reported in few studies (Tables 9 and 10) and measured fluxes range by an order of magnitude from 0.25 to 156 kg  $\text{NH}_3$ -N/ha-d (Aneja et al., 2000; Harper and Sharpe, 1998; Harper et al., 2000; Heber et al., 2001; Zahn et al., 2001). Ammonia flux from lagoons displays a daily and seasonal diurnal variation with maximum values occurring during the day and in the summer, respectively (Aneja et al., 2000; Harper et al., 2000).

For the same lagoon, ammonia emission varies with temperature and wind velocity. Use of different measurement methods at the same lagoon in similar climatic conditions have shown that estimates of emission rates can vary 200% or more depending on the measurement method (Harper and Sharpe, 1998; Aneja et al., 2000; Todd et al., 2001) (Table 10). Average annual  $\text{NH}_3$

TABLE 8. Ammonia emissions from cattle barns and feedlots

	Floor type	Manure Handling	Manure storage	Ventilation system	Ambient Temp. °C	Emission rates, NH <sub>3</sub> -N	Reference	Country or Region
Beef	Feedlot					0.64-2.37 kg/h-ha	Hutchison et al. 1982	U.S.
Beef	Feedlot					0.77 g/h-cow	CARB, 1999	U.S.
Beef	Litter					0.35-0.4 g/h-AU	Koerkamp et al., 1998	Europe
Beef	Slats					0.30-0.74 g/h-AU	Koerkamp et al., 1998	Europe
Beef	Straw	None		Natural	Feb-May	0.28 g/h-AU	Demmers et al., 2001	U.K
Beef	Straw	Scraped	Outside	Natural	Winter	0.67 g/h-AU	Demmers et al., 1998	U.K
Calves	Litter				8.4-10.5	0.26-0.86 g/h-AU	Koerkamp et al., 1998	Europe
Calves	SF				8.4-10.5	0.95-1.48 g/h-AU	Koerkamp et al., 1998	Europe
Dairy		Scraped	Outside	Natural	Summer	0.12 g/h-AU	Phillips et al., 1998	U.K.
Dairy	Cubicles				8.4-10.5	0.69-1.46 g/h-AU	Koerkamp et al., 1998	Europe
Dairy	Feedlot					1.28 g/h-cow	CARB, 1999	U.S.
Dairy	FSF	None	Deep pit	Natural		0.78 g/h-AU	Van't Ooster. 1994	Netherlands
Dairy	Litter	None			8.4-10.5	0.21-0.73 g/h-AU	Koerkamp et al., 1998	Europe
Dairy	Slat			Mechanical	12-18.4	25.5-40.0 g/d-cow	Kroodsma et al., 1993	Netherlands
Dairy	Straw	Scraped alley	Outside	Natural	Feb-May	0.84 g/h-AU	Demmers et al., 2001	U.K.
Dairy	Straw	Scraped	Outside	Natural	Winter	0.21 g/h-AU	Phillips et al., 1998	U.K.
Dairy	Straw	Scraped	Outside	Natural	Winter	1.09 g/h-AU	Demmers et al., 1998	U.K.

AU-animal unit, 500 kg live weight; FSF-fully slatted floor; SF – slatted floor; 1 g NH<sub>3</sub> = 1.214 g NH<sub>3</sub>-N.

Table 9. Ammonia emission from anaerobic swine lagoons in the U.S.

Measurement Method <sup>a</sup>	Period	Farm Type	Sequence	Area ha	TKN mg/L	TAN mg/L	Temp.	pH	Emission rate kg NH <sub>3</sub> -N/ha-d	Reference
Micromet.	Aug-Oct	FF	Primary	0.39	962-977	917-935	20.6-25.1	8.1-8.2	156	Zahn et al., 2001 <sup>b</sup>
Micromet.	Aug-Oct	FF	Primary	0.39	962-977	917-935	18.4-22.9	8.1-8.2	73-130	Zahn et al., 2001 <sup>c</sup>
Micromet.	Summer	FF	Primary	3.5		230-238	29.5	7.7-8.0	3.2-40	Harper et al., 2000
Micromet.	Winter	FF	Primary	3.5		239-269	10.3-15.4	7.4-7.7	1.3-1.9	Harper et al., 2000
Micromet.	Spring	FF	Primary	3.5		278-298	27.1-27.8	7.5-7.6	3.1-9.8	Harper et al., 2000
Micromet.	Summer	FW	Primary	2.4		193	28.5-29.4	8.34	2.9-8.4	Harper and Sharpe, 1998
Micromet.	Winter	FW	Primary	2.4		183	8.9-9.4	7.9	6.0-9.1	Harper and Sharpe, 1998
Micromet.	Spring	FW	Primary	2.4		227	17.6-18.9	7.7	3.0-6.6	Harper and Sharpe, 1998
Chamber	Sept	FF	Primary	0.78	146-202	101-110	21.4-28.9	7.0-8.1	0.57-3.5	Aneja et al., 2001
Chamber	Oct	FF	Secondary	0.33	82-93	37-44	7.5-29.9	6.8-7.3	0.25-1.87	Aneja et al., 2001
Chamber	Nov	FF	Tertiary	0.12	497-530	288-311	1.9-16.9	6.8-8.0	0.32-1.21	Aneja et al., 2001
Chamber	Nov	FW	Primary	0.36	569	350	11.2-14.1	6.8-7.4	0.46-1.73	Aneja et al., 2001
Chamber	Feb/Mar	FF	Primary	0.49	672-686	543-560	8.1-14.4	6.8-8.1	0.72-5.39	Aneja et al., 2001
Chamber	Mar	F	Primary	0.39	881-2102	709-909	13.3-23.5	6.8-7.1	0.82-2.95	Aneja et al., 2001
Chamber	Apr-Jul	BW	Primary	3.07	1072-1537	978-1143	16.7-27.7	8.0-8.2	104	Heber et al., 2001
Chamber	May-Jul	BW	Primary	1.23	374-628	326-387	22-27.7	7.9-8.1	39	Heber et al., 2001

<sup>a</sup> Methods: Micromet.-Micrometeorological. <sup>b</sup> Uncovered part of the lagoon, <sup>c</sup> Covered half of the lagoon. TKN, TAN, Temp., and pH are for lagoon liquid (supernatant). 1 g NH<sub>3</sub> = 1.214 g NH<sub>3</sub>-N.



TABLE 10. Ammonia emission rate from the same primary anaerobic lagoon (~2.5 ha) for a farrow to finish swine operation using different measurement methods in the U.S.

Measurement Method <sup>a</sup>	Period	TKN mg/L	TAN mg/L	Temp. ° C	pH	Emission rate kg NH <sub>3</sub> -N/ha-d	Reference
Micromet.	Aug. 6-9, 1997		574	28.1-28.6	8.08	15.4-22	Harper and Sharpe, 1998
Micromet.	Jan. 23-29, 1997		538	8.6-9.6	8.13	4.7-12.1	Harper and Sharpe, 1998
Micromet.	May 4-8, 1997		741	20.6-21.6	7.8	5.2-15.4	Harper and Sharpe, 1998
Chamber	Aug. 1-15, 1997	587-695		25.3-39.1	7.1-7.8	34-123	Aneja et al., 2000
Chamber	Dec. 1-17, 1997	599-715		8.4-15.3	7.9-8.1	5.3-28	Aneja et al., 2000
Chamber	Feb. 1-26, 1998	580-727		8.8-15.1	7.7-8.0	1.3-10	Aneja et al., 2000
Chamber	May 16-27, 1998	540-720		20.4-35.9	7.6-7.8	12.3-52	Aneja et al., 2000
TG OP-FTIR	May 15-20, 1997					37-122	Todd et al., 2001
TG OP-FTIR	Nov. 11-19, 1997					7.8-67.6	Todd et al., 2001

<sup>a</sup> Measurement method: Micromet.-Micrometeorological; TG OP-FTIR – Tracer Gas Open Path Fourier Transform Infrared Spectroscopy. 1 g NH<sub>3</sub> = 1.214 g NH<sub>3</sub>-N.

emission factors for anaerobic swine lagoons have been estimated to be 0.75-2.2 kg NH<sub>3</sub>-N/yr-pig place for a farrow to finisher farm (Harper et al., 1998; Aneja et al., 2000). The annual emissions were based on measurements taken for a few days in each season

Storing manure in tanks and in under-floor pits is common in the Midwest and corn belt states of the United States. Estimated liquid pit manure NH<sub>3</sub>-N concentrations are 719-6000, 600-2500, and 600-4400 mg/L, for swine, cattle and poultry, respectively (MWPS, 2000). Some N losses from stored manure with 2500-4200 mg/L TAN concentration are presented in Table 11. The only data from the U.S. is for dairy. The data generally indicate a wide range of emission rates, which could vary with climatic conditions, TAN concentration, crusting (cover), and other variables.

Anaerobically digested manure has higher pH and TAN concentrations compared to untreated manure (Velsen, 1977; Sommer and Husted, 1995; Sommer 1997). Digesting manure anaerobically increases the TAN concentration, reduces VFA concentration, and increases the total inorganic carbon concentration, and pH. Therefore anaerobically digested manure generally will have higher NH<sub>3</sub> volatilization potential compared to untreated manure. If anaerobically digested manure is stored in an open structure, relatively high NH<sub>3</sub> volatilization rates during storage should be expected because surface crusts rarely develop on digested manure (Sommer, 1997).

### 5.3.3 Land Application

Ammonia loss from land applied manure depends on meteorological factors, manure characteristics, application technique as well as soil parameters (Brunke et al., 1988; Morken and Sakshaug, 1998). Meteorological factors include temperature, rainfall, relative humidity, and wind speed. Soil properties include moisture content, infiltration capacity, cation exchange capacity, pH, and buffer capacity. Manure characteristics include TAN concentration, solids content, and pH.

Ammonia volatilization rate is usually highest during the first few hours after application and thereafter declines rapidly for temperatures above 10 °C (Beauchamp et al., 1982; Marshall et al., 1998; Pain et al., 1989; Thompson et al., 1990a; Sommer et al., 1991; Mattila, 1998; Sommer and Jacobsen, 1999; Smith et al., 2000b). More than 50% of the total NH<sub>3</sub> volatilization occurs within 24 hours after application. However, at temperatures close to 0 °C, NH<sub>3</sub> volatilization is slow and fairly uniform and may occur for several days (Sommer et al., 1991; Amberger, 1991; Pain et al., 1989).

Volatilization rate increases with temperature (Hoff et al., 1981; Beauchamp et al., 1982; Marshall et al., 1998; Pain et al., 1989; Thompson et al., 1990a; Sommer et al., 1991; Rubaek et al., 1996; Mattila, 1998; Sommer and Jacobsen, 1999; Smith et al., 2000b), wind speed (Hoff et al., 1981; Pain et al., 1989; Thompson et al., 1990b; Sommer et al., 1991), TAN concentration (Sommer and Olesen, 1991), and pH (Hoff et al., 1981; Sommer and Sherlock, 1996). Ammonia volatilization from slurry increases with wind speeds up to 2.5 m/s (Thompson et al., 1990b; Sommer et al., 1991). However, no consistent increase occurred when wind speeds increased from 2.5 to 4 m/s (Sommer et al., 1991). Higher wind speeds may cause higher water evaporation and induce accelerated crust formation on the applied manure, thereby reducing NH<sub>3</sub> volatilization. The high volatilization rates during the first 24 h have been associated with elevated pH at the manure surface (Sommer and Sherlock, 1996). As NH<sub>3</sub> volatilizes, the pH declines thereby reducing the volatilization rate in subsequent periods.

Diurnal pattern in NH<sub>3</sub> volatilization has been observed with the maximum and minimum occurring around midday and midnight, respectively (Hoff et al., 1981; Beauchamp et al., 1982; Harper et al.,

TABLE 11. Ammonia loss from manure storage structures

Storage type	Cover	Storage Time (days)	Animal species	Total N mg/L	TAN, mg/L	Emission or N lost	Reference	Country/Region
Earthen pit	None	150-365	Dairy	4150-4200	1420-1890	3-39% of Total N	Muck et al., 1984	U.S.
Tank (Stirred)	None	60-180	Dairy/Swine	4100-6100	2500-4200	3.5-5.2 g/m <sup>2</sup> -d	Sommer et al., 1993	Denmark
Tank	Lid	60-180	Dairy/Swine	4100-6100	2500-4200	0-5% of Stirred tank <sup>a</sup>	Sommer et al., 1993	Denmark
Tank	PVC foil	60-180	Dairy/Swine	4100-6100	2500-4200	2-26% of Stirred tank <sup>a</sup>	Sommer et al., 1993	Denmark
Tank	Peat	60-180	Dairy/Swine	4100-6100	2500-4200	1-32% of Stirred tank <sup>a</sup>	Sommer et al., 1993	Denmark
Tank	Leca	60-180	Dairy/Swine	4100-6100	2500-4200	5-17% of Stirred tank <sup>a</sup>	Sommer et al., 1993	Denmark
Tank	Oil	60-180	Dairy/Swine	4100-6100	2500-4200	0-48% of Stirred tank <sup>a</sup>	Sommer et al., 1993	Denmark
Tank	Straw	60-180	Dairy/Swine	4100-6100	2500-4200	3-60% of Stirred tank <sup>a</sup>	Sommer et al., 1993	Denmark
Tank	Crust	60-180	Dairy/Swine	4100-6100	2500-4200	8-24% of Stirred tank <sup>a</sup>	Sommer et al., 1993	Denmark
Tank	Straw	300	Mixed <sup>b</sup>			0.7 g/m <sup>2</sup> -d	Sommer,1997	Denmark
Tank	Clay pebble	365	Mixed <sup>b</sup>			0.3 g/m <sup>2</sup> -d	Sommer,1997	Denmark
Tank	None	365	Mixed <sup>b</sup>			9.0 g/m <sup>2</sup> -d	Sommer,1997	Denmark

<sup>a</sup> Emission expressed as a percentage relative to emission from the stirred storage tank.

<sup>b</sup> Composite mixture of anaerobically digested animal slurry (75%) and slaughterhouse and fish processing plants waste (25%).

1 g NH<sub>3</sub> = 1.214 g NH<sub>3</sub>-N.

1983; Thompson et al., 1990a; Klarenbeek and Bruins, 1991; Nathan and Malzer, 1994; Moal et al., 1995). The diurnal variation has been attributed to temperature, wind speed and TAN concentration changes after the manure is applied. Temperature affects the kinetic energy of  $\text{NH}_3$  molecules and their ability to escape from the aqueous phase and also the equilibrium constants for  $\text{NH}_3$  ionization. At a given TAN concentration, an increase in temperature will cause a higher  $\text{NH}_3$  to  $\text{NH}_4^+$  ratio. Furthermore, the solubility of  $\text{NH}_3$  decreases with rise in temperature, thereby enhancing volatilization. Both temperature and wind speed are generally lowest at night.

Slurry solids content (Sommer and Olesen, 1991, Smith et al., 2000b) and application rate (Thompson et al., 1990a; Frost, 1994) may also influence  $\text{NH}_3$  volatilization rates. Volatilization increases with solids content in the 4-12% range. Volatilization rate per unit volume of applied slurry (specific volatilization) decreases with increase in application rate. However, despite the low specific volatilization rates, higher slurry application rates result in larger quantities of  $\text{NH}_3$  lost compared to low application rates (Frost, 1994). Low application rates result in a thin slurry layer compared to higher application rates. A thin manure cover generally dries faster and results in higher loss rates while higher application rates produce a thicker slurry layer, which has a higher resistance to evaporation. As a result, the release of  $\text{NH}_3$  into the atmosphere slows down. Sometimes, the decline in the rate of volatilization with time after spreading is due partly to the formation of a surface crust. The crust forms more readily at high temperatures and it tends to counteract the direct effect of temperature on  $\text{NH}_3$ -N volatilization.

Soil properties and the type of surface to which manure is applied also affect  $\text{NH}_3$  losses. Volatilization from slurry applied to vegetated surfaces can be higher compared to bare soil (Amberger, 1991; Thompson et al., 1990a; Moal et al., 1995). This is because the presence of herbage and leaves reduces infiltration of manure into the soil.  $\text{NH}_3$  losses from slurry applied to hard, dry grassland soil are higher than moist grassland or arable soils (Smith et al., 2000b). The water holding capacity of manure and soil influences infiltration of liquid manure. Dilute manure infiltrates more rapidly into the soil resulting in lower volatilization rates (Frost et al., 1990; Pain et al., 1989; Pain et al., 1990a; Sommer and Olesen, 1991; Sommer and Jacobsen, 1999). The soil moisture content and porosity mainly affect the water holding capacity of the soil. High soil moisture content will decrease the soil's ability to absorb liquid from the slurry resulting in higher volatilization rates (Amberger et al., 1991; Sommer and Jacobsen, 1999).

Land application technique also influences  $\text{NH}_3$  volatilization. The application techniques in use include surface broadcasting, surface band spreading, plowing or harrowing in after application, and injection (Smith et al., 2000b). Low emission land application techniques include injection and plowing or harrowing in manure after application. Injection is most effective on moist arable soils but its use is limited to suitable soil conditions. Dry, hard grassland, and stony terrain are not conducive to soil injection. Generally, volatilization from injected manure is considerably lower than from surface applied manure because of lower exposed surface area and adsorption of  $\text{NH}_3$  to the soil particles. For all manure and animal types, 14 to 84 and 0 to 2.5% applied TAN can be lost from surface broadcast and injecting manure, respectively. These two land application methods represent high (surface broadcast) and low (injection)  $\text{NH}_3$  volatilization potential.

Ammonia loss from irrigated lagoon effluent during land application has been attributed to both drift and volatilization (Safley et al., 1992; Westerman et al., 1995; Sharpe and Harper, 1997). Safley et al. (1992) reported 4-37% of TAN loss during lagoon effluent irrigation to bare soil using a center pivot system. Of this amount, 62-100% of the loss could be accounted for in the liquid volume loss which occurred as a result of either evaporation or drift. Westerman et al. (1995) reported 10-25% TAN loss during irrigation of lagoon effluent on Coastal Bermuda grass. Sharpe and Harper (1997) quantified  $\text{NH}_3$  emissions after applying swine lagoon effluent using a solid-set sprinkler irrigation

system. During irrigation, about 13% of TAN in the liquid was lost through drift or volatilization before reaching the crop (oats) or soil surface. An additional 69% was volatilized from the soil and vegetation surface within 24 h of application.

Ammonia loss following land spreading of solid manure (farm yard manure and poultry litter) resulted in 30-89% loss of TAN applied from farm yard manure and 15-46% of  $\text{NH}_4^+$  uric acid N applied from poultry litter (Chambers et al., 1997). Fifty percent of TAN applied was lost one day after application and 90% loss occurred after 10 days. Emissions increased after rainfall events. Emissions continued even after 3 weeks of application. Ammonia emissions following application of solid manure were generally about 20% higher than from liquid manure with similar application rate and TAN content. Emissions from liquid manure are strongly influenced by climatic factors (mainly temperature and humidity). Emissions from solid manure mainly depend on the rate of TAN applied.

Ammonia emission rates from land application of manure are presented in Table 12. The data from the U.S. are for swine liquid manure, swine lagoon liquid, and broiler litter. The data verifies that injection of liquid greatly reduces ammonia loss (< 3% TAN applied) while surface broadcasting of swine manure or broiler litter and irrigation of lagoon liquid can result in losses generally 10 to 50% of TAN applied. The ammonia losses have a wide range because of differences in manure characteristics, climatic variables, and other factors previously mentioned.

## 6 EMISSION ESTIMATION METHODS

### 6.1 Measuring Atmospheric Ammonia Concentration

Several techniques ranging from simple acid traps to optical methods using Open-Path Fourier Transform Infrared (OP-FTIR) are available for measuring  $\text{NH}_3$  concentration in the atmosphere (Phillips et al., 2000; Ni and Heber, 2001). Concentration measurement methods include: acid scrubbers, gas detection tubes, electrochemical cells, passive diffusion devices, Denuders, optical absorption techniques, chemiluminescence detectors, fluorescence methods, and gas chromatography. Many of the earlier methods give  $\text{NH}_3$  concentrations averaged over long periods of time (typically several hours). More recent spectroscopy based techniques have shortened the response time for analysis in the order of 1 to 10 Hz at < 1 ppb concentrations. It is important to remember that both gaseous ( $\text{NH}_3$ ) and particulate ( $\text{NH}_4^+$ ) forms of  $\text{NH}_3$  exist in the atmosphere. Some measurement methods can discriminate between these two forms of atmospheric N while others give a gross concentration (TAN).

- i) Acid scrubbers – These devices are based on the rapid reaction of  $\text{NH}_3$  (a weak base) and ammonium aerosols with strong acid solutions. Typically, non-volatile acids (sulfuric acid, hydrochloric acid, phosphoric acid, or boric acid) are used to prevent loss of acid when ambient air at a known flow rate is bubbled through the solutions. The mass of N accumulated divided by the product of airflow rate and exposure time yields concentration. Acid scrubbers cannot discriminate between  $\text{NH}_3$  and  $\text{NH}_4^+$ , or N-containing volatile organic compounds that may become entrained in the acid solution. Uncertainties inherent in this technique include assuming 100% capture efficiency, uniform flow rates over long periods of time, and relatively pure acid sources to minimize blank values. In addition, care is needed in handling the acid solution to avoid exposure to the atmosphere before and after the prescribed sampling period. Exposure times must also be set with regard to anticipated ambient atmospheric concentrations.

TABLE 12. Ammonia losses from land applied animal manure

Manure Type	Application Method	Surface Nature	Manure TAN concentration	Ammonia Loss (% of TAN applied)	Country or Region	Reference
Broiler Litter	Surface broadcast	Grass	3.8 – 10 g/kg	26-46	U.S.	Marshall et al. 1998
Cattle – Liquid	Surface broadcast	Plowed land	1.4 – 1.7 g/kg	24-34	Canada	Beauchamp et al. 1982
Cattle - Slurry	Surface broadcast	Stubble	2.6 – 3.0 g/kg	38-63	Denmark	Sommer et al., 1991
	Surface broadcast	Grass		45-83		
	Surface broadcast	Harrowed soil		45-75		
	Surface broadcast	Cultivated stubble		55-80		
Cattle – Slurry Separated Unseparated	Surface broadcast	Grassland	1.0 – 1.5 g/kg	35	U.K.	Thompson et al., 1990a
	Surface broadcast	Grassland		38		
	Surface broadcast	Bare soil		67% of grassland		
Cattle – Slurry	Injected (5 cm) open Band spread	Grass	3.0 – 4.2 g/kg	6-7	Denmark	Rubaek et al., 1996
		Grass		22-47		
Cattle – Slurry No additives Acidified-pH 5.5	Surface broadcast Surface broadcast	Grass Grass	1.1 – 1.8 g/kg	31-84	U.K.	Pain et al., 1990b
				14-57		
Cattle – Slurry Untreated Separated Separated + acid	Surface broadcast Surface broadcast Surface broadcast	Grassland Grassland Grassland	2.8 – 3.6 g/kg	18	U.K.	Frost et al., 1990
				7-15		
				2-3.5		

Table 12 is continued in the next page.

TABLE 12 (cont'd). Ammonia losses from land applied animal manure

Manure Type	Application Method	Surface Nature	Manure TAN concentration	Ammonia Loss (% of TAN applied)	Country or Region	Source
Cattle –Slurry			1.8 – 2.8 g/kg		Finland	Mattila, 1998.
Untreated	Surface broadcast	Grassland		14-30		
Untreated	Band spreading	Grassland		15-28		
Separated	Surface broadcast	Grassland		18-46		
Aerated	Surface broadcast	Grassland		20-63		
Cattle –Slurry	Surface broadcast	Bare soil	2.7 – 3.0 g/kg	32-67	Netherlands	Van der Molen et al., 1990a
	Spread & incorporate	Bare soil		11-16		
Pig – Slurry	Surface broadcast	Grassland		24-62	U.K.	Pain et al. 1989
Pig –Slurry	Surface broadcasting	Growing barley	2.4 g/kg	13	Sweden	Malgeryd 1998
	Band spreading	Growing barley		6-8		
	Injection shallow	Growing barley		0.2		
Swine			2.1 – 4.2 g/kg		U.S.	Hoff et al. 1981
Liquid	Injected	Bare soil		0-2.5		
Storage pit	Surface broadcast	Bare soil		14-65		
Swine – Lagoon Liquid	Big gun irrigation	Bermuda grass	144 – 333 mg/L	10-25	U.S.	Westerman et al., 1995
Swine - Lagoon Liquid	Center pivot irrigation		244 – 334 mg/L	14-37	U.S.	Safley et al. 1992
Swine - Lagoon Liquid	Sprinkler irrigation	Oats	35 – 63 mg/L	82	U.S.	Sharpe and Harper, 1997

- ii) Detector tubes – e.g. Drager™ tubes. These devices afford a simple and convenient way of measuring atmospheric NH<sub>3</sub> concentrations. Tubes exist for both short (10 sec-15 min.) and long (2-8 h) term exposure periods. However, these devices are only semi-quantitative in nature and are primarily designed to assess human exposure at relatively high concentrations.
- iii) Electrochemical cells – These devices consists of electrodes in an electrolyte that has electrochemically active reagents with respect to NH<sub>3</sub>. Ammonia contained in a flowing air stream diffuses through a permeable membrane into the electrolyte solution. NH<sub>3</sub> contained in the resulting electrochemical reaction produces an electric potential that can be measured as a voltage or current. The electric signal is proportional to NH<sub>3</sub> concentration. These sensors are used primarily as toxic gas monitors in alarm systems. The normal response range begins at 5.5 ppm or greater (Fritsche et al., 1991; Phillips et al. 2001), which means these devices are not suited for monitoring ambient concentrations.
- iv) Passive diffusion devices – Passive diffusion devices like acid scrubbers use acid matrix to react with and retain NH<sub>3</sub>. Unlike acid scrubbers, ambient air is not allowed to contact the acid matrix, which is usually an organic acid impregnated in a filter paper. A pre-filter, usually a Teflon membrane with a set pore size, is used as a diffusion boundary, such that NH<sub>3</sub> must diffuse through the Teflon membrane before it can react with acid treated filter paper. The physical design of the passive collector is not random but an integral part of the assumptions used to calculate ambient concentrations. To calculate an average ambient NH<sub>3</sub> concentration requires the mass of N as ammonium captured by the impregnated filter paper, the time of exposure, a resistance factor for the Teflon membrane, and average wind speeds and air temperature during the time of exposure (Moss et al., 1995; Schjoerring, 1995; Sommer et al., 1996; Fowler et al., 1998; Genfa et al., 1998; Kirchner et al., 1999; Brown, 2000; Genfa and Dasgupta, 2000).

Passive diffusion devices are inherently selective for only atmospheric NH<sub>3</sub> and the resulting ammonium formed upon reaction with the acid matrix is easily extracted from the filter paper with distilled or de-ionized water. However, as with the acid scrubbers, exposure times must be set with regards to anticipated ambient concentrations. At relatively low ambient atmospheric concentrations (1-3 µg NH<sub>3</sub> m<sup>-3</sup>) exposure times of several days to weeks are not atypical. Attempting to use passive diffusion devices to measure ambient atmospheric NH<sub>3</sub> concentrations requires an NH<sub>3</sub> free source of organic acid (low analytical blanks), careful attention to detail in handling the devices before and after the exposure period (and during extraction of the acid impregnated filter paper), and a sensitive colorimetric or similar analytical technique for determining relatively low concentrations of ammonium in the water extract. Replicate devices (usually triplicate) should be used to assess the degree of uncertainty in the mass of NH<sub>3</sub> collected during the exposure period. Another source of uncertainty is related to the estimate of wind speeds and air temperatures during the exposure period. Ideally, these measurements should be nearby and at the same height as the passive diffusion device. This is sometimes not the case and the added uncertainty introduced by using wind speeds or ambient temperatures not measured directly adjacent to the diffusion device should be duly noted in the resulting calculations.

A range in physical sizes and configurations exists for the various types of passive diffusion samplers. Samplers with larger diffusion surfaces can accumulate NH<sub>3</sub> faster than those with small surfaces. In addition, most passive diffusion devices are designed and installed for protection from rainfall and are not exposed directly to the wind. Some designs,



however, are aerodynamically configured to turn into the wind to enhance the flow of air through the device thereby increasing the exposure to ambient  $\text{NH}_3$  concentrations. It is important that the inherent assumptions associated with their design are met for the sampling location where they are deployed. It is recommended that due to the variation in designs for passive diffusion devices, that their use to measure ambient  $\text{NH}_3$  concentrations be compared to other devices e.g. acid scrubbers or denuders (see below). Comparing to other devices may not ensure they are used properly.

- v) Denuders – These devices collect  $\text{NH}_3$  without interference from ammonium aerosol. The method is based on the difference in diffusion velocity of  $\text{NH}_3$  gas and ammonium particles (Ferm, 1979). The air is drawn at a known and controlled flow rate through a tube with an absorbing surface for  $\text{NH}_3$ . Ammonia diffuses to the tube wall and is absorbed while the particulates pass through with air and are impacted on a filter. The collected ammonia may be extracted for analysis in the laboratory or measured as nitric oxide in the instrument after catalytic conversion. Coating denuder surfaces with boric, citric, oxalic, and sulfuric acids (O'Halloran, 1993; Andersen et al., 1993; Andersen and Hovmand, 1994; Jaeschke et al., 1998; McCulloch et al., 1998; Zhu et al., 2000) and solutions such as  $\text{NaHSO}_4$  (Wyers et al., 1993) have been used to collect  $\text{NH}_3$  from air streams. Other materials such as molybdenum oxide and tungsten oxide have been used in denuders (Braman et al., 1982; Braman et al., 1986; Langford et al., 1989; Williams et al.; 1992).
- vi) Optical absorption techniques – These systems use a narrow band width of either infrared or ultraviolet radiation and determine the concentration of specific molecules based on absorption of the corresponding radiation (infrared or ultraviolet). The optical techniques are designed to measure mean concentrations along an open path (Klarenbeek et al., 1993; Sommer et al., 1995; Drescher et al., 1997; Todd et al., 2001) or in closed cells (Lachish et al., 1987; Galle et al., 2000; Griffith and Galle, 2000). Fourier transform infrared spectroscopy (Carter et al., 1993; Todd and Bhattacharyya, 1997; Hashmonay et al., 1999; Galle et al., 2000; Samanta and Todd, 2000) methods have detection limit of about  $5 \mu\text{g}/\text{m}^3$  while the UV fluorescence (Schendel et al., 1990) technique has limit of approximately  $0.005 \mu\text{g}/\text{m}^3$ . Most of these technologies combine meteorological measurements and plume dispersion modeling techniques or the tracer gas ratio methods to calculate  $\text{NH}_3$  emission rates. Open-path measurement techniques are non-invasive and there is no adsorption of  $\text{NH}_3$  or other gases by parts of the measurement device. The disadvantages of this method lie in the determination of emission rates. It is also difficult to distinguish emissions from sources lying close to each other.
- vii) Chemiluminescence – Detectors can be used to measure  $\text{NH}_3$  concentration provided that  $\text{NH}_3$  is first converted to NO. Ammonia analyzers that use this technology are combinations of a chemiluminescence NO analyzer and an upstream thermal  $\text{NH}_3$  converter (Breitenbach and Shelef, 1973; Aneja et al., 1978, Williams et al., 1992; Phillips et al., 1998b; Aneja et al., 2000). The analyzer records all gases that are converted to NO in the upstream converter (which operates at  $600^\circ\text{C}$  or higher) as Total N. Compounds that would be converted to NO include organic N, nitric acid ( $\text{HNO}_3$ ), nitrogen dioxide ( $\text{NO}_2$ ), and ammonium containing aerosols (Phillips et al. 1998b). A second converter ( $350^\circ\text{C}$ ) converts  $\text{NO}_x$  to NO. Ammonia concentration is determined as difference between total N and  $\text{NO}_x$ .
- viii) Fluorescence – Fluorescence methods are based on absorptive paper impregnated with solutions that absorb  $\text{NH}_3$  from the air. The fluorescence intensity of the tape changes with

increasing pH as  $\text{NH}_3$  is absorbed. The degree of change in fluorescence intensity is proportional to the concentration of  $\text{NH}_3$  gas at constant sampling time and flow rate (Nakano et al., 1995). Detection of 0.1 ppm of  $\text{NH}_3$  gas can be determined at a sampling time of 40 s and a 400 ml/min flow rate.

- ix) Gas Chromatography – Gas chromatography procedure has been reported in which  $\text{NH}_3$  is automatically collected from the air over a period of a few minutes and passed through a GC with a flame thermionic detector and photo-ionic detectors (Phillips et al., 2001). Detection as low as  $0.02 \mu\text{g}/\text{m}^3$  has been claimed.

## 6.2 Determining Ammonia Flux

Several techniques exist to calculate the exchange rate of  $\text{NH}_3$  between a source and the atmosphere. Many integrate the atmospheric  $\text{NH}_3$  concentration and relate the mean concentration to surface emission.

### 6.2.1 Nutrient Mass Balance Method

A N mass balance may be attempted for a single component or source of  $\text{NH}_3$  or to a total animal production system. For a single source, the N mass balance method involves determining the changes in the N content of the source, and estimating how much of the loss is due to  $\text{NH}_3$  volatilization. This approach can be manageable for small sources or laboratory measurements, but is difficult to apply to a large source of  $\text{NH}_3$  emission.

The N balance approach for an animal production system involves constructing a budget that accounts for N inputs and outputs. N input sources include animal feed (concentrates and forage), fresh bedding, animal protein (animals imported to the facility), atmospheric deposition, N-fixation, and N-containing fertilizers. Possible N outputs (exports) from a farm include sale of animals, animal products (e.g. milk, eggs, and meat) and harvested crops; surface runoff and leaching from land applied manure and manure containment structures, and gaseous emissions to the atmosphere. Typically,  $\text{NH}_3$  emissions are calculated as the difference between inputs and outputs considered in the budget or components of a given production facility, such as buildings (Burton and Beauchamp, 1986; Naser et al., 1997; Koerkamp et al., 1998; Pollet et al., 1998), storage and treatment structures (Muck et al., 1984), and land application. Since  $\text{NH}_3$  emission estimate derived using N balance is essentially a difference, the uncertainty associated with the derived emission value will be greater than those associated with the various components used to construct the budget. Thus while this method avoids direct measurements of  $\text{NH}_3$  emissions, the uncertainty in the estimated emission depends on the complexity of the constructed N-balance. There is also the inherent assumption that the difference between inputs and outputs can be attributed totally to a single gas species (e.g.  $\text{NH}_3$ ) ignoring the possibility that considerable amounts of  $\text{N}_2$ ,  $\text{NO}_2$ , and  $\text{N}_2\text{O}$  may also be lost. Therefore, the N balance method can result in higher values especially if two or more distinct N containing gases are emitted.

### 6.2.2 Micrometeorological Techniques

Micrometeorological methods are used to determine mass flux from a surface with minimal disruption of the measurement environment. These techniques require that the surface be horizontally homogenous to ensure that the concentration measurements made above the surface reflect the source/sink strength at the surface. The height of the adapted layer responds not only to the distance from the leading edge, but also to atmospheric stability. (McGinn and Hanzen, 1998). Some of the techniques used with this method are:

- i. Aerodynamic approach. This technique requires measurements of  $\text{NH}_3$  concentration, wind speed and temperature at several heights (Beauchamp, et al., 1982; Harper et al., 1983;

- Brunke et al., 1988; van der Molen et al., 1990a; Genermont and Cellier, 1997; Genermont et al., 1998; Hansen et al., 1999; Sharpe and Harper, 1997; Harper et al., 2000). Ammonia flux is calculated as a product of friction velocity and the turbulent concentration.
- ii. Bowen ratio-Energy Balance. This technique generally requires measurements of gradients of temperature, water vapor and the gases of interest and fluxes of net radiation and soil heat.
  - iii. Eddy correlation and relaxed eddy accumulation. Eddy correlation is based on the fact that turbulence has frequencies less than the current sampling rate of modern sonic anemometers. Thus it is relatively easy to sample turbulence. There is no equipment to sample  $\text{NH}_3$  at the same frequency (approximately 1 to 10 Hz). Thus eddy correlation technique is not directly applicable to measuring  $\text{NH}_3$  flux (McGinn and Janzen, 1998; Phillips et al., 2000). Relaxed eddy accumulation protocols try to avoid the need for rapid ("instantaneous") sampling of  $\text{NH}_3$  concentration. It is essentially a gradient approach where the flux is proportional to the difference in mass collected in updrafts and down drafts (Zhu et al., 2000).
  - iv. Passive samplers in a gradient approach. Passive samplers have been used to determine  $\text{NH}_3$  concentration and horizontal flux. Vertical flux of  $\text{NH}_3$  can be calculated by integrating the horizontal flux with height and dividing by the upwind distance to the leading edge (Pain et al., 1989; Moss et al., 1995; Genermont et al., 1998; Phillips et al., 1997; Sommer and Olesen, 2000). Alternatively, the flux can be calculated by dividing the horizontal flux by the wind speed at each height (Schjoerring, 1995, Sommer et al., 1996).

### 6.2.3 Chamber Methods

Chamber methods are also referred to as wind tunnel (Lockyer, 1984; Thompson et al., 1990a,b; Sommer and Olesen, 1991; Sommer et al., 1991; Olesen and Sommer, 1993; Sommer et al., 1993; Rubaek et al., 1996; van der Weerden et al., 1996; Menzi et al., 1998; Misselbrook et al., 1998; Loubet et al., 1999a,b), convective flux chamber, or dynamic chamber (Hoff et al., 1981; Svensson, 1994; Jiang et al., 1995; Mattila, 1998; Morken and Sakshaug, 1998; Jeppson, 1999; Peu et al., 1999; Sommer and Jacobsen, 1999; Aneja et al., 2000). The chamber typically has a controlled air flow rate, and possibly additional mixing inside the chamber. Concentrations of the air into and out of the chamber are measured, or the inlet air has zero concentration. Generally, the surface flux is obtained by the product of change in concentration and the volume flow rate divided by the enclosed surface area. Chamber methods can modify the microclimate inside the chamber relative to the surroundings (Svensson, 1994). There is a strong dependence of  $\text{NH}_3$  volatilization on air and source temperature, and on airflow, all of which can be changed due to the presence of the chamber. Potential limitations, such as microclimate modification and negative feedback between accumulated gases and surface emission rates can restrict the sampling duration. Chamber methods provide specific conditions and allows good control for frequent or continuous measurement of concentrations, but caution is required when extrapolating results from chamber studies to the field, because measurement environments may not represent actual field conditions.

### 6.2.4 Emission Factors

Estimating ammonia emissions from animal production operations can be done using an average emission factor for the animal type. The overall ammonia emission factor for an animal production operation can be estimated by summing appropriate emission rates per animal for various sources (buildings, storage and treatment, and land application) on the farm. Total  $\text{NH}_3$  emissions from the farm is obtained by multiplying the overall emission factor by total number of animals present on the farm (Misselbrook et al., 2000; Hutchings et al., 2001). Providing a sound estimate of total  $\text{NH}_3$

emission therefore depends on the accuracy in quantifying losses from each of these stages, which may not be the same for different management systems. Currently, the complexities of the effects of different environmental conditions have not been fully integrated into emission estimates and much uncertainty surrounds average emission factors.

### 6.2.5 Dispersion Models

There is a growing interest in using dispersion models to estimate emissions from whole production facilities. This approach bypasses the multiple sources of uncertainty associated with calculating emissions based on N balance or using non-site specific emission factors. It involves measuring the concentrations of the emitted gas (e.g. NH<sub>3</sub>) and a number of meteorological variables. The uncertainty in measuring these variables can be estimated but the major uncertainty is mainly due to the assumptions from which the model is constructed and the applicability to a given location.

Dispersion models used to determine emissions are those based on Gaussian diffusion theory (Phillips et al., 2000; Asman, 2001). Gaussian diffusion models are used in regulatory applications to estimate impacts of sources on air quality. An advantage of such models is that the plume dispersion parameters are based on theory and are well characterized by experimental data (Arya, 1999). The shortcomings and limitations of these models arise from the many simplifying assumptions implicit in the mathematical solutions of these models (such as conditions of steady, uniform flow and homogenous turbulence), and the assumption of vertical Gaussian concentration distribution which is often not realized in the boundary layer. A few applications using dispersion models to estimate emissions from animal production facilities have been published and it is apparent that their use must be, for the foreseeable future, correlated with the more traditional approaches of using a N balance or emission factors (Rege and Tock, 1996). However, the overall appeal of dispersion models in terms of their relative analytical simplicity and application to direct estimates of emissions from an entire production facility means that research into their development and deployment will continue.

## 7 MODELING

Quantifying NH<sub>3</sub> emission is critical in formulating and comparing possible control strategies. Quantifying NH<sub>3</sub> emission using measurement methods is expensive, and also contains a relatively high degree of uncertainty when extrapolated across a larger population of potential emission sources. One possible approach to reduce this uncertainty is to develop suitable models that successfully account for variation at the sources that are associated with NH<sub>3</sub> volatilization. Such models would enhance the ability to compare and discern differences between different NH<sub>3</sub> emission sources, and would also allow estimating NH<sub>3</sub> emissions from physical locations which are not amenable to direct measurements, either because of site characteristics (physical limitations), and/or cost. Models are also more flexible than direct measurements in that they can be scaled to fit the need either on a small scale (covering a few meters to a few kilometers) or on a large scale covering geographical or economic regions, multiple states or countries (Buijsman et al., 1998).

Models that have been developed can be classified into three categories: statistical, empirical, and mechanistic; however, often models are a combination of two or more of these categories. Statistical models are generally based on experimental data derived from monitoring NH<sub>3</sub> emissions from a specific animal production facility, i.e., no attempt is made to control the factors that can influence NH<sub>3</sub> volatilization. Statistical models to describe NH<sub>3</sub> release based on observational data have been published for anaerobic swine lagoons (Aneja et al., 2000; Harper and Sharpe, 1998; Harper et al., 2000) and land applied manure (Menzi et al., 1998; Sommer and Olesen, 2000). Ammonia flux models based on statistically derived mathematical equations

provide an index of the relative importance of measured environmental variables on flux (at least for the conditions during the measurement period at a specific site or sites). Field studies' monitoring data should provide a range of environmental variables likely to be encountered in a region.

Empirical and mechanistic models that describe  $\text{NH}_3$  release from some types of animal buildings (Elzing and Monteny, 1997; Aarnink and Elzong, 1998; Monteny et al., 1998), stored liquid manure (Muck and Stenhius, 1982; Olesen and Sommer, 1993; Zhang et al., 1994; Ruxton, 1995) anaerobic lagoons (Westerman et al., 1999; Harper et al., 2000), land-applied manure (Lockyer, 1984; Van der Molen et al., 1990b; Sommer et al., 1991; Sommer and Olesen, 1991; Moal et al., 1995; Genermont and Cellier, 1997; Huijsman and Mol, 1999), and transport of  $\text{NH}_3$  in the atmosphere (Bartnicki and Alcamo, 1989; Asman, 1998; Singles et al., 1998) have been published.

Some empirical models have been derived from experiments conducted under semi controlled conditions. Each model includes the influence of only a restricted number of factors and predicts  $\text{NH}_3$  losses limited to conditions under which they were established. Some are statistical correlations of measured parameters. However, they can be used to assess the accuracy and application of mechanistic models.

Mechanistic models, on the other hand, describe the volatilization process through  $\text{NH}_3$  transformation, equilibria, and transfer within the system (Ni, 1999). While mechanistic models account for most of the factors involved in  $\text{NH}_3$  volatilization, sometimes they require a large number of parameters that are difficult to obtain under field conditions. Dispersion models are also utilized in combination with measurements of aerial  $\text{NH}_3$  concentration and meteorological measurements to estimate  $\text{NH}_3$  emissions. It is usually difficult to determine the uncertainty or error when using this approach.

Models are useful in preparing and evaluating environmental policies. Therefore, a practical model should have a realistic description of all the implied mechanisms/processes so that it can be used under a wide range of environmental/field conditions. The emission model should therefore be able to use readily available input data and parameters. Estimates of error or uncertainties for models should be calculated, especially for use in making policies.

## **8 CONTROL STRATEGIES AND TECHNOLOGIES**

Strategies for reducing  $\text{NH}_3$  losses should be directed towards reducing: (1)  $\text{NH}_3$  formation, (2)  $\text{NH}_3$  losses immediately after it has been formed, (3) the  $\text{NH}_3$  loss potential and (4) policies and regulations. Some of the potential control strategies for  $\text{NH}_3$  control from animal production facilities include changes in diet, barn design or retrofits to reduce  $\text{NH}_3$  emissions, cleaning building exhaust air, manure treatment methods, land application techniques, and policies and regulations. In summary, reducing  $\text{NH}_3$  loss requires a whole farm systems approach. This approach shows how intervening in one aspect of the farm may affect  $\text{NH}_3$  losses in other parts. For example, draining pull-plug manure pits frequently to reduce swine building emissions could increase overall emission from the operation if the drained manure is not properly handled, treated, and stored. Therefore, to obtain adequate  $\text{NH}_3$  volatilization abatement in an animal production operation, a combination of these control strategies should be used.

### **8.1 Buildings**

Building emissions can be significantly reduced through proper diet, manure management, ventilation, and building hygiene. Minimizing emitting surface area, length of time manure is exposed to the air, TAN concentration, and air velocity above the manure surface can reduce  $\text{NH}_3$

volatilization in buildings. Additionally, transforming or fixing  $\text{NH}_3$  physically or chemically and reducing particulate matter and gases in the ventilation exhaust air can be used to reduce  $\text{NH}_3$  emission from animal buildings.

### **8.1.1 Diet**

One method of reducing  $\text{NH}_3$  volatilization from livestock and poultry, is to reduce the excretion of N by feeding reduced protein diet (Tamminga, 1992; Smits et al., 1995; Kay and Lee, 1997; Canh et al., 1997; Canh et al., 1998a,b,c; Sutton et al., 1999; James et al., 1999; Erickson et al., 2000; Jacob et al., 2000a,b). Reduced N excretion reduces TAN that impacts  $\text{NH}_3$  emission from animal housing, storage, land applied manure, and grazing livestock. Matching feed carefully to the nutritional requirements of the animal can reduce N excretion without affecting yields (Jongbloed and Lenis, 1992; Hobbs et al., 1996). Improving feed composition by supplementing with limiting essential amino acids, enhances N uptake from feed. Feeds can be combined to create the desired balance of amino acids, or by supplementing feeds with other artificial sources of limiting amino acids. Feeding can be tailored to the age, size, or sex of the animals. More details about dietary manipulation to affect manure production and composition are addressed in another white paper (Sutton et al., 2001).

Ammonia emissions can also be reduced by decreasing the pH of urine and/or manure (Aarnink et al., 1998). In swine buildings where a large portion of  $\text{NH}_3$  emission is derived from urine deposited in the floor rather than from the manure below the slats, changing the pH of urine is most effective (Aarnink et al., 1998). The pH of the urine/feces can be reduced by replacing  $\text{CaCO}_3$  in the diet by  $\text{CaSO}_4$  or  $\text{CaCl}_2$  (Mroz et al., 1996) or by adding adipic or phosphoric acid to the feed (Van Kempen, 2001). Another possibility to reduce  $\text{NH}_3$  emissions in swine buildings would be to alter the ratio of N excretion in the urine and feces by addition of fermentable carbohydrates (Canh et al., 1997; Sutton et al., 1997; Canh et al., 1998a,b,c; Mrotz et al., 2000). However, shifting N excretion from urine to feces may not reduce the overall  $\text{NH}_3$  emission from an animal feeding operation.

In poultry husbandry, the composition of the feed and the efficiency of feed conversion influence the N content of the feces. For broiler and layer chickens, reduced protein diets have been observed to reduce excreted N (Jacob et al., 2000a,b).

Feeding a reduced crude protein balanced diet reduces  $\text{NH}_3$  emissions from dairy cattle (James et al., 1999; Smits et al., 1995). A 14% reduction in dietary N intake resulted in a 28% decrease in ammonia emission and 29.6 and 19.8% decrease in urea-N and total N excreted by dairy animals, respectively (James et al., 1999). Manure produced by cows fed reduced protein diets had lower pH, high dry matter content, and 39% less  $\text{NH}_3$  emissions compared to manure from high protein diets (Smits et al., 1995). Using lower crude protein diets and phase feeding can reduce N excretion in yearlings and fattening beef cattle, thereby, reducing runoff and volatilized N from feedlots (Erickson et al., 2000).

### **8.1.2 Building Design**

In dairy barns reducing the emitting area surface by tying the animals (tie stalls) reduces the floor area to which excreta is spread, and nearly 80%  $\text{NH}_3$  emission reduction relative to unrestricted animals (free stalls) has been reported (Groenestein, 1993). However, the current move to discourage tie stall barns because of animal welfare violation may lead to discontinuing their use in Europe (Monteny and Erisman, 1998). Sloped concrete floors can reduce  $\text{NH}_3$  volatilization by up to 50% compared to slatted floors in dairy barns (Swierstra et al., 1995; Braam et al., 1997).

Additional reductions, up to 65%, is possible using a combination of scraping and spraying the floor with water (Braam et al., 1997).

In pig barns, emitting surface area can be reduced using different floor systems (Aarnink et al., 1995; Voermans et al., 1995). For example, using a floor made of triangular cross-section (1 cm wide with 1-cm gaps) metal slats in partially slatted pens for grow/finish pigs reduced  $\text{NH}_3$  emissions (27%) compared to concrete slatted floors (10 cm wide with 2-cm gaps) (Aarnink et al., 1995).

### **8.1.3 Manure Management**

Removing manure from animal occupied areas is done by scraping, flushing, slatted flooring, or combination of any of these systems (Voermans, et al., 1995; Gastel et al., 1995; Verdoes et al., 1996; Aarnink et al., 1995; Monteny and Erisman, 1998) and by using conveyer belts in some caged layer houses (Hartung and Phillips, 1994). In poultry buildings (cage) removing manure twice a week using belts or weekly with drying manure on belts has reduced  $\text{NH}_3$  emission from battery cage houses by 60% or more. Flushing with water removes and dilutes urine and TAN concentration in the manure. Flushing floors with water reduced  $\text{NH}_3$  volatilization by 14-70% compared to slatted floors in dairy barns (Voorburg and Kroodsma, 1992; Kroodsma et al., 1993; Ogink and Kroodsma, 1996). Adding formaldehyde to flushing water reduced volatilization from dairy barns by 50% compared to 14% in barns flushed with water only (Ogink and Kroodsma, 1996). Formaldehyde reduced the pH of the flushing liquid. Flushing frequency and the quality and amount of water determine the amount of reduction that can be achieved (Voorburg and Kroodsma, 1992; Hoeksma et al., 1993; Monteny, 1996; Monteny and Erisman, 1998). Frequent flushing gives the best results because hardly any degradation then takes place inside the house (Monteny, 1996; Cowell and Apsimon, 1998). However, using more water increases the volume of the slurry if fresh water is used for flushing. Reducing  $\text{NH}_3$  emission up to 70% in flushed buildings is possible using aerated or acidified liquid effluent after separation for flushing (Hoeksma et al., 1993). However, treating flushing liquid may not be economically feasible (Hoeksma et al., 1993; Gastel et al., 1995; Monteny, 1996).

### **8.1.4 Ventilation**

Ventilation has been used to improve air quality in animal housing. However, ventilation can be expected to increase  $\text{NH}_3$  losses. The most common remedy for elevated  $\text{NH}_3$  concentration in poultry houses is to increase ventilation rates above the values needed for proper litter moisture control. The increased ventilation rates reduce  $\text{NH}_3$  concentration in the house but translate directly into higher  $\text{NH}_3$  emissions and energy costs associated with the house during the cold season (Elliot and Collins, 1982).

Associated with ventilation, temperature control to encourage excretion over a minimal area is sometimes used in combination with scraping or flushing systems to control  $\text{NH}_3$  emission from pig houses (Aarnink et al., 1995). Unfortunately, owing to lack of space, especially in slatted floor buildings with under-floor manure storage, the pigs will lie on the slatted floor area, which may cause convective airflow in pits and increase  $\text{NH}_3$  emission. This could be due to poor management of ventilation and cooling systems. Fouling of solid floors is worse in the summer than in the winter.

Removing  $\text{NH}_3$  from vented air using filters or scrubbers (water and acid) is feasible where barns are mechanically ventilated (Sommer and Hutchings, 1995). In poultry buildings, exhaust air can be cleaned using bioscrubbers, biofilters or chemical scrubbers. However, the practical applications of these cleaning devices may be limited by cost and technical problems due to dust in

poultry and swine houses. Moreover, only mechanically ventilated buildings can be equipped with air cleaning devices, and air cleaning does not reduce the  $\text{NH}_3$  inside the poultry and swine houses (Koerkamp, 1994).

### 8.1.5 Additives

Ammonia loss from animal/poultry barns can be reduced by amending manure/litter with: (1) acidifying chemical compounds such as alum, ferric chloride, ferrous sulfate, inorganic acids and organic acids, (2) organic materials such as straw and sphagnum peat, and biological additives (Carlile, 1984; Al-Kanani et al., 1992; Moore et al., 1995; Subair et al., 1999; Hendriks et al., 1998; Heber et al., 2000). Amendments are effective if they: (1) directly adsorb  $\text{NH}_4^+$  and  $\text{NH}_3$ ; (2) reduce the manure pH; (3) promote microbial production of organic acids that reduces the manure pH; (4) increase microbial N immobilization; or (5) inhibit microbial growth. However, problems such as non-uniform mixing and increased moisture content in the manure might arise. Decreasing the pH lowers the  $\text{NH}_3$  emission potential. However, large amounts of acid/acidic salts are generally required and precautions must be taken to safeguard the safety of animals/birds and workers. Adding acids leads to an undesirable increase in the mineral content of the manure/litter and corrosion problems of equipment and the structure. Ammonia emission from slurry stored in under-floor pits can be reduced by adding a layer of oil to the slurry (Aarnink and Wagemans, 1997). More information on amending manure/litter is presented by Moore et al (2001).

## 8.2 Manure Storage and Treatment

Ammonia volatilization from manure storage and treatment facilities can be controlled using physical, chemical and biological methods.

### 8.2.1 Physical and Chemical Methods

For storage and treatment facilities open to the atmosphere, the simplest physical control method is to use a cover. Several cover materials have been shown to reduce  $\text{NH}_3$  emissions compared to uncovered storage or treatment structures. Sommer et al. (1993) reported surface covers reduced  $\text{NH}_3$  emissions by less than 60% (Crust, 8-24%; Peat, 1-32%; Straw, 3-60%; PVC foil, 2-26%; Leca®, 5-14%; Lid, 0-5%) compared to uncovered storage tanks. Zahn et al. (2001) reported a 17-54% reduction of  $\text{NH}_3$  emission from a lagoon covered using a biocover. Hornig et al. (1998) reported the following  $\text{NH}_3$  emission reduction compared to uncovered controls – 80% for straw covered storage tanks, 63-91% for Pegulit (white buoyant, natural mineral buoyant material) covered slurry tanks, 99.7% for a floating film (two 2-mm thick polyethylene film layers glued together) for a lagoon, and 99.5% for a tent covered lagoon.

Drying can also be used to reduce  $\text{NH}_3$  volatilization from stored manure. Drying poultry manure to less than 40% water content reduces  $\text{NH}_3$  volatilization by more than 50% (Koerkamp, 1994). At low water content, transformation of uric acid and urea are reduced and little  $\text{NH}_3$  is produced.

Additives to reduce  $\text{NH}_3$  concentration and emission from stored manure, animal houses, and slurry tanks are available. However, reliable data to prove/show the efficacy of these products are rare (Miner et al., 1995; Jungbluth and Busher, 1996; Hendricks et al., 1998.). The basic principles of these additives are discussed in the section above. Hendriks et al. (1998) reported a 45-59% reduction in  $\text{NH}_3$  emissions from swine buildings using an additive that contained a mixture of enzymes, bacteria, yeast and molds.

Combined physical and chemical processes can be used to control  $\text{NH}_3$  volatilization by binding and/or recovering  $\text{NH}_3$ . Some of the combined processes include (1)  $\text{NH}_3$  stripping, absorption and recovery; (2) membrane filtration, chemical absorption and recovery; and (3) chemical precipitation



of ammonium salts and recovery.  $\text{NH}_3$  stripping process involves raising the pH of the slurry using lime or sodium hydroxide to promote the transfer of  $\text{NH}_3$  to the gaseous phase. The slurry is stripped with air or steam at a high temperature.  $\text{NH}_3$  resulting from the gas-stripping phase is absorbed in water or in acid solution. With membrane separation technology, only  $\text{NH}_3$  is transferred through a membrane into an absorbing liquid (e.g. acid) while other slurry components are retained. In chemical precipitation, ammonium salts that are moderately or barely soluble in water are normally used. Struvite ( $\text{Mg NH}_4\text{PO}_4 \cdot 6\text{H}_2\text{O}$ ) is the most commonly precipitated mineral (Buchanan et al., 1994; Maekawa et al., 1995). The  $\text{NH}_3$  content of the slurry can be increased by pretreatment processes such as wet oxidation, hydrothermolysis or anaerobic digestion of the manure (Rulkens et al., 1998).

### **8.2.2 Biological Methods**

Biological removal methods of  $\text{NH}_3$  from animal manure is generally achieved by a combination of two processes - nitrification and denitrification. Nitrification/denitrification processes convert ammonium-N to nitrate-N then to dinitrogen ( $\text{N}_2$ ) gas or nitrous oxide. Removal efficiencies ranging from 47- 100% have been reported (Osada et al., 1991; Svoboda, 1995; Vanotti et al., 1999; Westerman et al.; 2000). An obstacle to farmers adopting this process is the relatively high energy cost for aerating the manure for nitrification.

Biological conversion processes can also reduce  $\text{NH}_3$  volatilization by recovering N products from liquid animal waste. Three categories can be distinguished by the production of: (1) single cell proteins (Rulkens et al., 1998); (2) amino acids - lysine - (Sanders, 1993); and (3) protein rich aquaculture plants such as duckweed and algae.

### **8.3 Land Application**

Land application measures to reduce  $\text{NH}_3$  emissions aim to minimize the amount of manure and time of exposure on the ground. This can be achieved by mechanical means such as injection or prompt plowing-in, increased infiltration, washing manure in after application, and lowering manure pH.

#### **8.3.1 Application Method**

Injection or immediate incorporation of manure into the soil reduces  $\text{NH}_3$  losses compared to other surface application methods. Injections to 3-30 cm depths have been reported to reduce  $\text{NH}_3$  volatilization by 75-96% (Hoff et al., 1981; Thompson et al., 1987; Phillips et al., 1990; Sommer and Thomsen, 1993; Smith et al., 2000b). Surface placement methods such as band spreading using trailing hoses, shoes, and shallow slot injection (up to 50 mm depth) reduced  $\text{NH}_3$  loss by 40-60% of the TAN applied compared with surface broadcasting (Smith et al., 2000b). Losses from incorporation immediately after surface broadcasting manure were 11 to 16% of the applied TAN (van der Molen et al., 1990a)

Ammonia losses from manure applied during the growth period may be reduced using trail hoses which apply the slurry onto the soil between rows of plants (Bless et al. 1991; Holtan-Hartwig and Bockman, 1994). The reduction in  $\text{NH}_3$  loss can be attributed to absorption of  $\text{NH}_3$  by plant leaves and to some extent by reduced slurry surface. The  $\text{NH}_3$  loss is reduced due to small surface area, increased infiltration and reduced wind speed above the slurry. Living plants can also reduce  $\text{NH}_3$  volatilized from land application by: (1) absorption by plant leaves of  $\text{NH}_3$  volatilized from the underlying soil (Holtan-Hartwig and Bockman, 1994), (2) absorption of ammonium through the roots, and (3) microclimatic effects due to the canopy.

### 8.3.2 Manure Dilution

Ammonia losses from surface applied slurry are inversely related to infiltration. Increasing manure infiltration into the soil can be achieved by diluting with water. Adding water in the proportions 1-3:1 reduced NH<sub>3</sub> losses by 44-91% compared to untreated slurry (Stevens et al., 1992; Frost, 1994; Morken and Sakshaug, 1998). Additionally, diluting the manure with water reduces the TAN concentration thereby reducing NH<sub>3</sub> loss potential. Cultivating the soil surface or increasing the surface roughness can also increase infiltration rates (Sommer and Thomsen, 1993). Cultivating the soil surface before surface application of slurry reduces NH<sub>3</sub> losses (40-90%) compared to uncultivated soils. The reduction is caused by higher infiltration rates into the soil and increased surface roughness (Sommer and Thomsen, 1993).

### 8.3.3 Acidifying Manure

Ammonia volatilization from manure may be reduced (50-80%) by acidification (Gordon et al., 1988; Stevens et al., 1989; Pain et al., 1990b; Frost et al., 1990; Husted et al., 1991; Al-Kanani et al., 1992; Stevens et al., 1992; Vandre and Clemens, 1997). In general, the most effective strategy for preventing NH<sub>3</sub> volatilization is a combination of separation, dilution, and acidification (Stevens et al., 1992). However, manure generally has a high buffer capacity and requires significant amount of acid to reduce pH.

## 8.4 Policy

There are three types of policy instruments that can be used to address NH<sub>3</sub> volatilization from animal production facilities: (1) education and technical assistance through Cooperative Extension Service, USDA-NRCS or soil and water conservation districts, (2) financial incentives through cost-sharing or subsidies, and (3) legal regulations. Most measures to reduce NH<sub>3</sub> release from animal production are not (yet) economically attractive. Thus, utilization of some of these policy instruments may be required to induce producers to reduce NH<sub>3</sub> release.

An example of how policy has been used is in the Netherlands where low emission NH<sub>3</sub> housing systems were introduced to reduce NH<sub>3</sub> volatilization from animal buildings. The Dutch government introduced a "Green Label Award" system to stimulate farmers, research and development institutions, and equipment suppliers to develop and implement buildings with low NH<sub>3</sub> emissions. The advantages to the farmer who invests in the Green Label Houses are tax incentives and assurance that the facility would not have to be rebuilt for 15 years. NH<sub>3</sub> emission must be reduced by 50-70% in the year 2000 compared to measurements in the year 1980. Housing systems considered for green label awards include flushing and scraping systems, manure cooling and narrow manure channels (Gastel, et al., 1995; Voermans et al., 1995; Voermans et al, 1996; Verdoes et al., 1996). Ammonia emission reductions ranging from 30-85% by the new designs over the standard building designs have been reported (Gastel et al., 1995; Voermans et al., 1995).

## 9 SUMMARY OF MAJOR FINDINGS

### Environmental concerns

- Nutrient deposition in N-sensitive ecosystem may change the vegetation or the ecosystem due to high N concentrations.
- Formation of aerosol particles may cause haze and impair visibility and also have potential health effects from respirable 2.5 µm particulate matter (PM<sub>2.5</sub>).
- There is relatively little data for verifying environmental and health effects from NH<sub>3</sub> emissions in the U.S. at present.

### Emissions

- Emissions vary greatly depending upon source; environmental conditions; animal diet, type size, and age; and management.
- Emissions are not usually reported in the same units for buildings, storage/treatment, and land application, making it difficult to calculate emission factor per animal unit.

### Emission Factors

- Use of composite factor for each animal species is common. The composite factor can over or under estimate the NH<sub>3</sub> inventory depending on the predominant animal class, size, and numbers present in a region.
- The emission factors need to be updated frequently to reflect the developments and changes in dietary and facility management directed towards reducing N loss.
- Factors used in the U.S. are often based on assumptions that a certain percentage of N excreted is volatilized, or European data because there is relatively little measurement data for the U.S.
- Caution should be used when applying emission data from Europe to the U.S. because operations in the U.S. may have different management and climate conditions.
- Emission factors are developed with data collected over short duration, and extrapolations beyond these sampling periods are prone to error.
- Data collection on NH<sub>3</sub> emissions in the U.S. has increased in the recent years, but much more emission data is needed in the U.S. in order to make better estimates of composite emission factors.

### Estimation and Measurement methods

- Measurement equipment and methods are improving, but are generally expensive and not very amenable to long-term continuous monitoring.
- Different measurement methods can yield results which often vary by 30% to even more than 200%.
- Errors in measurements and emission estimation methods are usually difficult to determine.
- Each measurement method should include a nitrogen mass balance to check reasonableness of data.

### Emission Models

- Emission and transport models are often a combination of empirical, statistical, and mechanistic (process) components.
- Models are useful for planning new facilities and evaluating the effects of changing climatic conditions and management on NH<sub>3</sub> emissions from an animal feeding operation.
- Errors or uncertainties of model prediction are difficult to determine.
- Additional development and field verification are needed for models.

### Control Strategies

- Several control strategies have potential to reduce ammonia emission, but their effectiveness and economic cost - benefit analysis need evaluation.
- Strategies should stress a whole farm systems approach. Intervening in one aspect may affect NH<sub>3</sub> losses in other parts.
- Policy or regulations should be based on science and provide incentives.

## 10 RESEARCH NEEDS AND ISSUES

1. Environmental impacts of NH<sub>3</sub> deposition on land, crops, and water.
2. Health effects (human and animal) of NH<sub>3</sub> on-farm and off-farm, including epidemiological studies.
3. Evaluation and standardization of measurement methods for NH<sub>3</sub> concentration and NH<sub>3</sub> emission flux.
4. Improved determination of emission factors for various animal types and sizes and for various animal and manure management facilities and practices.
5. Improvement and validation of models for NH<sub>3</sub> emission, transport, and deposition.
6. Evaluation of the effectiveness of technologies and control strategies.
7. Economic evaluation of control strategies.

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**APPENDIX**  
**LIST OF UNITS**

Symbol	Definition
Mass	
μg	microgram
mg	milligram
g	gram
kg	kilogram
Length	
μm	micrometer
mm	millimeter
cm	centimeter
m	meter
km	kilometer
Time	
s	second
min.	minute
h	hour
d	day
wk	week
wks	weeks
yr	year
Area	
m <sup>2</sup>	square meter
ha	hectare
Volume	
L	liter
m <sup>3</sup>	cubic meter



## LIST OF SYMBOLS AND ACRONYMS

Acronym/Symbol	Definition/Description
[ ]	Concentration
Apr	April
atm	Atmosphere
AU	Animal Unit (=500 kg live weight)
Aug	August
BW	Breed to wean
$C_5H_4O_3N_4$	Uric acid
$CO(NH_2)_2$	Urea
$CO_2$	Carbon dioxide
Dec	December
$E_{NH_3}$	Annual emission rate, mass/yr-animal place
$E_x$	Mean emission rate during a production cycle, mass/h
$E_y$	Mean emission rate in a non productive cycle, mass/h
F	Finish (fattening facility for hogs)
Feb	February
FF	Farrow to finish
FSF	Fully slatted floor
FW	Farrow to wean
H	Henry's Law constant
$H^+$	Hydrogen ion
$H_2O_2$	Hydrogen peroxide
$H_2SO_4$	Sulfuric acid
$HNO_3$	Nitric acid
Hz	Hertz
Jan	January
Jul	July
$K_d$	Dissociation constant
$K_L$	Mass transfer coefficient
LW	Live weight
Mar	March
$MgNH_4PO_4 \cdot 6H_2O$	Struvite
Micromet.	Micrometeorological
mol	mole
MWPS	Midwest Plan Service
N	Nitrogen
$N_2$	Dinitrogen gas
$N_2O$	Nitrous oxide
$N_A$	Number of animals present in a production period
$NH_3$	Ammonia
$NH_{3(air)}$	Molecular ammonia in air (atmosphere)
$NH_{3(aq)}$	Molecular ammonia in solution
$NH_3-N$	Ammonia-Nitrogen
$NH_4^+$	Ammonium ion
$NH_4^+-N$	Ammonium-Nitrogen
$(NH_4)_2SO_4$	Ammonium sulfate

Acronym/Symbol	Definition/Description
$\text{NH}_4\text{HSO}_4$	Ammonium bisulfate
$\text{NH}_4\text{NO}_3$	Ammonium nitrate
NO	Nitric oxide
$\text{NO}_3^-$	Nitrate ion
Nov	November
$\text{NO}_x$	Nitrogen oxides
$\text{O}_2$	Oxygen
$\text{O}_3$	Ozone
Oct	October
$\text{PM}_{2.5}$	2.5 $\mu\text{m}$ size particulate matter
ppb	Parts per billion
ppm	Parts per million
PSF	Partially slatted floor
Rep.	Republic
SF	Slatted floor
$\text{SO}_2$	Sulfur dioxide
$\text{SO}_4^{2-}$	Sulfate ion
$\text{SO}_x$	Sulfur oxides
STEL	Short term exposure limit
T	Temperature, °C, °K
TAN	Total Ammoniacal Nitrogen
Temp.	Temperature
TG OP-FTIR	Tracer Gas Open Path Fourier Transform Infrared
TKN	Total Kjeldahl Nitrogen
TLV	Threshold limit values
TWA	Time weighted average
U.K.	United Kingdom
U.S.	United States of America
UA	Urease activity
UFP	Under floor pit
USDA	United States Department of Agriculture
UV	Ultra violet
V	Wind speed
VFA	Volatile fatty acids
X	Production period, d
Y	Non production period, d