Auditing and Assessing Air Quality in Concentrated Feeding Operations[1,2,3]

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ABSTRACT

The potential adverse effects of concentrated animal feeding operations (CAFO) on the environment are a growing concern. The air quality issues of most concern to CAFO vary but generally include ammonia, hydrogen sulfide, particulate matter, volatile organic compounds, greenhouse gases, and odors. Air pollutants may be regulated by federal and state laws or by nuisance complaints. The United States Environmental Protection Agency, and poultry, swine, and dairy industries recently agreed to the National Air Emissions Monitoring System to fund research on atmospheric emissions from production farms in the United States. Air quality regulations may be based on actual emissions, atmospheric concentrations, or human perception, or via limiting the size or location of CAFO. Measuring the concentrations or emissions of most air pollutants is expensive and complex. Because of spatial and temporal variability, concentrations and emissions must be measured continuously over an extended period of time. Because different methods or models can give different results with the same data set, a multitude of methods should be used simultaneously to assure emissions are reasonable. The "best" method to measure concentrations and emissions will depend upon atmospheric concentrations, cost, facility characteristics, objectives, and other factors. In the future, requirements for monitoring of air emissions from CAFO will probably increase. Reliable process-based models need to be developed so that emissions of air pollutants can be estimated from readily obtained diet, animal, facility, and environmental variables. Auditors will need to be trained in a variety of disciplines including animal sciences, chemistry, engineering, micrometeorology, instrumentation, modeling, and logic.

Key words: air quality, concentrated animal feeding operations, dispersion, modeling, regulation

INTRODUCTION

The potential adverse effects of animal feeding operations (**AFO**) on the environment are a growing concern. The effects of concentrated animal feeding operations (**CAFO**) on water quality have been regulated under the Clean Water Act for many years. However, the effects of intensive and extensive livestock operations on air quality have received less attention at least until recently. Even in rural communities, the general public has become less tolerant of the odors and dust emitted from agriculture because of concerns about health, quality of life, property values, and the environment.

In general, the air pollutants of most concern to livestock operations include ammonia, hydrogen sulfide, particulate matter (**PM**), volatile organic compounds (**VOC**), greenhouse gases (methane, nitrous oxide, and carbon dioxide), and odors or odorants. However, the predominant air quality concerns of livestock and poultry feeding operations will vary with the location of the operation, the species reared, type of operation (enclosed or open-lot), and other factors.

To successfully audit and assess air quality at AFO, it may be necessary to quantify gaseous and PM emis-

sions. Measuring atmospheric emissions is difficult and entails 2 major challenges: 1) measuring the atmospheric concentration, and 2) estimating the flux to the atmosphere based on direct measurement or on a flux model that describes or simulates the turbulent dispersion of gases and particulates. Thus development of process-based models will be needed to adequately monitor atmospheric emissions from CAFO.

CURRENT RULES AND REGULATIONS

The ultimate responsibility for air quality regulations resides with the federal government. However, state and local governments can also regulate pollutants in some cases. In addition, based on real, perceived, or potential quality-of-life issues, many air pollutants are "regulated" through the court system via injunctions and law suits. The permitting process is, in itself, a potential regulatory mechanism for CAFO. Some pollutants (odor, PM) have short travel distances and affect relatively small geographic areas. Therefore, local or regional, rather than federal, regulations may be most appropriate. In other cases, for example ammonia, the ultimate negative effects may be both local (dry deposition on sensitive ecosystems) and longer range (formation of respirable particulates near urban areas; wet deposition); thus Federal regulations may be more appropriate.

A small group of "criteria" pollutants {PM [both larger (PM_{10}) and smaller particulates ($PM_{2.5}$)], ozone, carbon monoxide, nitrogen dioxide, sulfur dioxide, and lead} are regulated under The Clean Air Act of 1970 and its amendments (EPA, 1987). [The PM_{10} and $PM_{2.5}$ are particles having an aerodynamic equivalent diameter (**AED**) of less than 10 and 2.5 μ, respectively.] The Environmental Protection Agency (**EPA**) established a set of primary and secondary National Ambient Air Quality Standards (**NAAQS**) designed to

protect the public against adverse health effects and to protect the environment. Recent court cases have established that ambient air quality standards apply not only to large, heterogeneous air sheds but also at the property line of an individual source (EPA, 2007a). As monitoring methods improve and as the weight of scientific evidence increases, ambient air quality criteria are to be modified to accommodate the latest science.

Although agriculture has received some exclusions to the NAAQS in the past, the EPA recently issued a final rule amending the NAAQS regulation of PM_{10} to include agricultural sources (EPA, 2004b). In addition, some states have initiated their own air quality regulations that affect agriculture. For example, California is currently developing regulations for ozone precursors such as reactive VOC and for ammonia emitted by livestock, their manure, or both, and several states have adopted regulations on ammonia, hydrogen sulfide, or odors. In general, current air quality regulations that relate to CAFO are based on actual emissions or atmospheric concentrations of pollutants (EPA, 1988, 1995). More complex pollutants, such as odors, may be regulated based on human perception (olfactometry or scentometry). In some states, regulations limiting the size or location of CAFO, or establishing separation distances from CAFO to businesses or residences, have been used as a regulatory mechanism to decrease water and air pollution from CAFO.

At the time of this writing, ammonia and hydrogen sulfide emissions are not regulated by the EPA. However, it is possible that in the future they may be regulated under the Comprehensive Environmental Response, Compensation and Liability Act (also called "Superfund"; EPA, 2007b) or Emergency Planning and Community Right-to-Know Act (also called "Right to Know"; EPA, 2007c). Under these regulations, all facilities that emit more than 45.4 kg (100 lb)

of ammonia or hydrogen sulfide per day must report the release to a Federal National Response Center as well as to local and state emergency planning committees (EPA, 2006). Until recently, it had been generally assumed these regulations did not apply to agriculture, primarily based on portions of the regulations that limit the response to a release of "a naturally occurring substance in its unaltered form or altered solely through naturally occurring processes or phenomena from a location when it is naturally found are exempt" (US Code, 2005). In 2006 the EPA concurred with a legal assessment by the National Cattlemen's Association stating that Clean Air Act Title V permitting requirements and the Comprehensive Environmental Response, Compensation and Liability Act or Emergency Planning and Community Right-to-Know Act emergency release reporting requirements do not apply to open-lot cattle operations (Drovers Journal, 2006).

In the late 1990s the EPA determined that it did not have adequate air emissions data to determine potential air quality regulatory requirements for AFO. Therefore, the EPA, poultry industry, swine industry, and dairy industry developed the Animal Feeding Operations Air Compliance Agreement (Consent Agreement or National Air Emissions Monitoring System; EPA, 2005). Under the agreement, the industries will fund a 2-yr research project to measure emissions of ammonia, hydrogen sulfide, PM [total suspended particles (**TSP**), PM_{10}, and $PM_{2.5}$], and VOC at designated production farms across the United States to establish scientifically based measures of these emissions (Federal Register, 2005).

The monitoring and regulating of atmospheric emissions is complex. Because emissions of different pollutants have varying spatial and temporal effects, Halberg et al. (2005) recommended that emissions linked to environmental effects with a local or

regional target be area-based (i.e., emission per hectare), whereas those linked to global effects should be product-based (e.g., emission per kilogram milk produced). Similarly, NRC (2003) recommended that air emissions with national or global scope (ammonia, greenhouse gases) should be evaluated based on emissions per unit of production, whereas air emissions of local concern (hydrogen sulfide, PM, odor) should be based on the farm and on concentrations at the farm boundary or nearest occupied dwelling.

European Regulations

Because of adverse effects on the environment, many parts of Europe have established strict regulations on some air pollutants, most notably ammonia (de Vries, et al., 2001; Oenema, 2004). Within the European Union, targets for ammonia emission have been established that require a 0 to 43% (depending upon the member state) decrease in ammonia emissions between the 1990 benchmark and 2010. This has necessitated reduction in animal numbers, diet modifications, and modifications in manure handling. Because the relationship between agricultural activities and their environmental impact is complex and difficult to measure directly, the European Union has developed a series of indicators to provide information regarding the relationship between an agricultural activity and its impact on the environment. This is based on a need to simplify complex phenomena and quantify their significance. Policy measures are designed and evaluated on a conceptual framework named DPSIR, which stands for Driving forces (societal, market, and technology causes), Pressures (emissions of pollutants), State of the environment or recipient (i.e., status or quality of the environment), Impacts (effects on health, ecosystems, and agriculture) and Responses (effect of governmental policy on items above; Oenema, 2004; Halberg et al., 2005). Pressures in-

cluded in at least 1 of the some 35 models used include emissions of greenhouse gases (methane, nitrous oxide), ozone-depleting gases, acidifying gases, and nutrifying substances (ammonia; Halberg et al., 2005; EEA, 2006).

ENVIRONMENTAL AUDITS

The EPA (CFR, 2000) defines an environmental audit as a "systematic, documented, periodic, and objective review of facility operations and practices related to meeting environmental requirements." Formal environmental audits and assessments are parts of an organization's environmental management and pollution prevention plans. The objectives of an audit include the following: 1) verifying compliance with environmental requirements, 2) evaluating the effectiveness of in-place environmental management systems, and 3) assessing risks from regulated and unregulated materials and practices. In short, the objectives of an audit are to identify problems, analyze the underlying cause(s), and develop action plans to correct those causes. The EPA (2000) suggests that by conducting audits, an operation gains a better understanding of where its facilities stand compared with specified criteria, such as regulations, management goals, or other legal requirements, and provides the organization with a list of what needs to be done to meet specific goals.

Environmental audits may be as simple as a tour of facilities and review of records or as complicated as intensive air sampling and monitoring. Audit criteria may be management practices that benefit the environment or may be compliance requirements such as regulations. For example, if the goal is to test for compliance with regulations, the audit may provide information on whether compliance has been achieved or not, and if not, what specific measures are needed to comply with regulations. Environmental

audits should be performed by persons that possess a working knowledge of the regulations and have a familiarity with the operations and practices of the facility being audited. These 2 basic skills are a prerequisite for adequately identifying areas at the facility subject to environmental regulations and potential regulatory violations (EPA, 2001).

The US EPA (EPA, 1996b, 1997; 2000, 2001), several states (IWRC, 2000), and the International Organization for Standardization (ISO, 1999, 2001, 2002) have developed resources to assist small businesses in designing audit programs, including audit checklists, protocols, and software. A number of businesses and agencies also provide training courses and certification for environmental auditing (BEAC, 1999; EPA, 2000), most designed around the ISO 14000 standards (ISO, 1999; EPA, 2004a). In 1993 the ISO began work on the "ISO 14001 — Standards for Environmental Management Systems" which was subsequently supplemented with "ISO 19011 — Guidance for Quality and/or Environmental Management Systems Auditing" (ISO, 2002). These ISO standards are the basic framework around which an auditing program may be developed.

To encourage environmental audits, the EPA and several states have developed incentive programs that encourage regulated entities to voluntarily "police" themselves. Facilities that voluntarily discover, promptly disclose, and expeditiously correct violations of federal environmental regulations can obtain elimination of, or substantial reduction of civil penalties, criminal prosecution, or both (CFR, 2000). The policy was designed to encourage greater compliance with federal laws and regulations by promoting a higher standard of self-policing. The potential cost savings associated with waste and emission management and liability risks are factors in motivating industries to establish proactive environmental programs that encourage auditing and pollution prevention.

Pollution prevention opportunity assessments (**PPOA**) can be used by environmental managers to identify opportunities to change facility operations in order to save money, increase worker safety and morale, and decrease regulatory liability (EPA, 1992). The PPOA can be used as a tool for identifying and eliminating the underlying causes of compliance problems. Compliance problems can be resolved via a combination of best management practices (EPA, 2007d), management changes, or technical modifications.

Accurately defining the objectives and scope of an environmental audit is critical to ensure that the audit achieves the desired results. The scope of an audit usually defines a specific procedure or area of investigation and can be influenced by factors such as facility conditions, cost, staff availability, or other resource constraints. An initial air quality audit at an AFO might include a listing of air emissions of concern, their source(s), estimates of emissions or concentrations within and downwind of the facility, and potential effects of management practices on the emissions. It should also identify potential environmental or human health problems and develop schedules for remedial actions. The audit should include management audits, PPOA, and auditing standards (EPA, 1997). Other factors that should be evaluated include organizational structure, environmental commitment, formality of environmental programs, internal and external communication programs, staff training and development, program evaluation, reporting and corrective action, environmental planning and risk management, and the environmental protection program (EPA, 1996b).

The purpose of any pollution prevention program is to prevent pollution, not to collect data. Therefore the simplest auditing system that fits a facility's needs is the best. However, because the facility can not manage what it does not measure,

emission inventories and air emission analyses may both be important components of an initial pollution prevention plan audit (EPA, 1992). For the purposes of this manuscript, we assumed that future auditing and assessment of air quality at AFO will revolve around development of nutritional and management practices to comply with federal, state, and local air quality regulations and to avoid nuisance law suits. Preliminary assessments and site inspections will be required to determine what, if any, regulations are broken, to describe the release, to develop remediation practices, and to evaluate the remediation practices (EPA, 2001).

AIR QUALITY COMPONENTS OF LIVESTOCK OPERATIONS

Air quality components from AFO are formed by a complex set of microbial, physical, and chemical processes that occur within the animal and the manure storage and processing system. Air quality components associated with livestock are generally more complex than those from industrial sources because of the numerous biological processes involved. Unfortunately, there is little information on air quality around livestock operations. However, it is imperative that a better understanding of the effects of AFO on air quality be developed to fairly regulate them and to develop potential solutions to air quality concerns.

Significant variation occurs among AFO in the air pollutants of most concern. These variations are a result of differences in animals' digestive systems, the diets fed, the production systems, and manure storage and handling systems. For example, ammonia and odor emissions can be changed 20 to 50% via modifications of the diet of feedlot cattle (Cole et al., 2005, 2006; Archibeque et al., 2006; Todd et al., 2006), dairy cattle (James et al., 1999; Frank et al., 2002), and swine (Colina et al.,

2001; Powers et al., 2006) with only small effects on animal performance.

Animal production facilities also vary in size, construction, operation, and location. In many cases, for example, litter in broiler houses, deep-pit storage in swine facilities, and the pen surface and mounds in open-lot beef cattle and dairies facilities, the animal facilities serve as both the production unit and manure storage site. Because of differences in bacterial populations and nutrient content, each manure storage system produces different air quality components.

Ammonia

Ammonia is formed in manure storage systems through the fermentation of nitrogenous compounds. A number of forms of nitrogen are excreted by animals, ranging from complete proteins to urea. The major source in most situations is the urea in urine (or uric acid in poultry excreta), which is rapidly converted to ammonia via the bacterial urease enzyme in soil and feces. Ammonia loss appears to begin almost immediately after urea is excreted, and it continues through manure handling, storage, and land application (Arogo et al., 2001). Ammonia emissions from retention ponds and lagoons appear to be a continuous process that occurs primarily from the pond surface (Ni, 1999). Therefore, factors that affect the surface affect ammonia emissions.

In its gaseous form, atmospheric ammonia can travel from rural to urban areas and neutralize acid gases such as sulfates and nitrates (products of fossil fuel burning) in the atmosphere, converting these gases into small particulates ($PM_{2.5}$) that may pose a potential health risk to some individuals (Watson et al., 1998; Neas, 2000; McCubbin et al., 2002). Ammonia that travels downwind may also be deposited via wet or dry deposition onto the soil or water bodies and become a nutrient source. In ecologically sensitive areas, ammonia deposition may pro-

vide an oversupply of N for the native flora, resulting in modifications of the native ecosystem (Hutchinson and Viets, 1969; Wolfe et al., 2003; Todd et al., 2004). From an economic perspective, ammonia losses also represent a loss of potentially valuable N fertilizer.

Atmospheric ammonia concentrations at AFO vary greatly and there appears to be a notable diel pattern with highest concentrations during the day and lowest concentrations at night (Omland, 2002; Todd et al., 2005, 2007). Ammonia concentrations in open-lot feedyards rarely exceed 3 ppm (Todd et al., 2005, 2007); however, concentrations within animal houses can frequently exceed 25 ppm, the threshold limit value for worker safety in Denmark (Omland, 2002). Ammonia emissions from AFO may be affected by many factors including diet (protein quantity and degradability, carbohydrate degradability, acid-base balance), pen surface, retention pond, or lagoon conditions (total ammonium concentration, pH, temperature, moisture, solids), weather, ventilation rate, manure storage method, and animal age (Dewes, 1996; Ni, 1999; Ni et al., 1999; Cole et al., 2005, 2006; Todd et al., 2006, 2007).

Some current ammonia emission factors used by the EPA are in doubt because incorrect assumptions were made and because many values are based on European data in which different managing systems were used (Asman, 1992; Battye et al., 1994) that may not be applicable to American production systems. In addition, a single emission factor for ammonia and many other pollutants is difficult to justify because so many environmental factors can affect emissions.

Hydrogen Sulfide

Hydrogen sulfide forms in livestock operations primarily from anaerobic fermentation by sulfate-reducing bacteria. In solution, sulfide ions develop an equilibrium with hydrogen ions (Shurson et al., 2000). Under basic conditions (pH > 8), most reduced S exists in solution as HS^- and S^{2-} ions, and the quantity of free H_2S is small. At pH < 8 the equilibrium shifts rapidly toward formation of unionized H_2S: this is about 80% complete at pH 7. Thus, in contrast to ammonia, H_2S emissions from ponds tend to be greater at lower pH. Typical pH for swine lagoons and feedlot retention ponds are in the range of 7 to 8.5. Hydrogen sulfide emissions from lagoons and retention ponds appear to occur episodically when sufficient hydrogen sulfide gas, produced from nutrients or sludge on the bottom of the pond, accumulates to overcome the surface tension of the water and rise to the pond surface. Hydrogen sulfide emissions from AFO are related to diet, surface and pond pH, temperature, and biological oxygen demand.

Reported hydrogen sulfide emission rates from swine and dairy manure storage tanks and anaerobic lagoons are highly variable (Parker et al., 2005a) ranging from 146 (Zahn et al., 2001) to 46,260 (Hobbs et al., 1999) $\mu g/m^2$ per min. Hydrogen sulfide concentrations downwind of feedyard pens in Nebraska (Koelsch et al., 2004) and Texas (Rhoades et al., 2003) ranged from 0.003 to 0.13 ppm; however, fewer than 1% of measurements exceeded the state regulated value of 0.1 ppm. Hydrogen sulfide emissions from feedyard pens averaged approximately 3.7 kg per 1,000 head daily, and emissions from retention ponds ranged from 102 to 1,348 $\mu g/m^2$ per min (0.54 to 11.2 kg per 1,000 head daily; Rhoades et al., 2003).

At elevated concentrations, hydrogen sulfide is a toxic gas, and thus, is regulated primarily by state and federal regulations designed for worker safety and based on atmospheric concentrations, not total emissions. Concentrations of 100 ppm are considered immediately dangerous to health (ATSDR, 2004). The Occupational Safety and Health Administration Permissible Exposure Limit for hydrogen sulfide is 20 ppm for a 10-min ceiling value (ATSDR, 2004).

Odors and Volatile Organic Compounds (VOC)

Odor is not regulated at the federal level but is often a high-profile issue at the local level where it generally falls under the purview of state nuisance laws (nuisance is generally defined as any condition that unreasonably interferes with another's enjoyment of life or property). Odor is the result of human perception of the hundreds of gaseous odorants released into the atmosphere.

Most odors from livestock operations are the result of formation of VOC via anaerobic fermentation of organic wastes (Mackie et al., 1998). These VOC can be grouped into 5 general classes of compounds (indoles, phenols, acids, cresols, and disulfides) that produce a wide range of odors, from cleaning materials (phenols) to vinegar (acids) to fecal smell (indoles, cresols, and disulfides). When mixed, these compounds create odors that are unique. This class of compounds is very elusive because each behaves differently in the air, reacts with other chemicals in the air, and thus may travel different distances. The major drivers for odor formation include pH, surplus moisture (which excludes oxygen), and warm temperatures. Odor emissions are affected by diet, pen conditions, retention pond or lagoon chemistry, and other factors (Sweeten et al., 1983, 1995).

The concentrations of VOC generally decrease with distance from the source due to dispersion and interaction with other compounds in the air. For example, VFA may react with ammonia close to the source to form ammonium salts that are less odiferous than the original VFA. Thus, few VFA are isolated in air further downwind (i.e., 800 m) of open feedlots, whereas some phenolic and indolic compounds can be measured up to several kilometers downwind

of the lot (Parker et al., 2007). Some odorous chemicals may also be carried by dust particulates. Reactive VOC have been implicated as precursors in ground level ozone formation and thus are coming under increased scrutiny in some ozone nonattainment areas such as California (Rabaud et al., 2003).

Particulates and Dust

Particulates and dust can have adverse effects on visibility, quality of life, and possibly human and animal health (Donham, 1986; MacVean et al., 1986; Barnes, 1994; Guarino et al., 1999; Cole et al., 2000; Neas, 2000; McCubbin et al., 2002; Omland, 2002). Dust particles also can carry other potential pollutants. The chemical composition of PM generated from livestock operations has not been well characterized, but it is generally organic in nature, originating from manure, feed, bedding or litter, and animal dander. Some inorganic dust from roads and pen surfaces is also present. Dusts from AFO fall into a number of classes based on their size and characteristics. Until 1987 EPA regulated TSP. However, in 1987 the EPA replaced its TSP standards with ambient standards for PM_{10} (EPA, 1987). In 2001, EPA added additional new ambient standards for $PM_{2.5}$ based on epidemiological studies that suggested an effect of respirable particulate concentrations on hospitalization and morbidity rates in cities (EPA, 1996a; Neas, 2000). Currently the 24-h average NAAQS for ambient PM_{10} is 150 $\mu g/m^3$ and for ambient $PM_{2.5}$ is 65 $\mu g/m^3$.

In an effort to more appropriately regulate the coarse fraction of particles, the EPA has recently proposed establishing a "coarse fraction" PM ($PM_{10-2.5}$) NAAQS (EPA, 2004b), which theoretically refers to the mass fraction of particles between 2.5 and 10 μ AED. The intended goal of this proposed standard is to regulate the size fraction of particles for which the PM_{10} standard was initially intended — i.e., inhalable coarse particles. The proposed concentration limit for the new $PM_{10-2.5}$ standard is 75 $\mu g/m^3$ on a 24-h-average basis.

Typical agricultural emissions such as those from feed mills and AFO are composed of particles with mass-median diameter (**MMD**) of about 10 to 20 μ or greater; moreover, less than 5% of particle mass is smaller than 2.5 μ (Sweeten et al., 1988; 1998). (Mass median diameter refers to the AED at which 50% by mass of the aerosol consists of particles with AED > MMD and 50% by mass consists of particles with AED < MMD.) The concentrations of PM in open-lot feedyards vary greatly with highest dust concentrations at dawn and dusk, when animal activity is greatest and air movement is the most stable (Sweeten et al., 1988). Similarly, PM emissions in swine facilities are greatest during the day and during the summer months when ventilation rates are highest. Dust emissions from feed mills, a lesser source, also vary greatly, depending on the commodities used, how they are handled and processed, and how the feeds are mixed.

Dust emissions are affected by the environment, pen surface conditions (moisture, manure depth), ventilation, time of day, and dietary factors (Van Wicklen and Yoder, 1988; Van Wicklen et al., 1988; Phillips and Thompson, 1989; Pearson and Sharples, 1995; Auvermann, 2006). Emissions from open lots may be decreased by increasing animal density, collecting manure more frequently, sprinkling, and using windbreaks (Auvermann et al., 2000, Auvermann and Romanillos, 2000), whereas emissions from confinement buildings can be decreased via oil sprays (Pearson and Sharples, 1995) or by impaction air dams downstream of ventilation exhaust fans (Bottcher et al., 1998).

Dust from AFO may also give rise to nuisance complaints, which are typically regulated at the state or local level. Visibility may be signifi-cantly reduced on roadways near open-lot AFO during peak PM emission events or when the atmosphere is thermally stable, creating a traffic hazard and associated civil liability. Because of potentially confounding factors, the effects of PM on the employee health, animal health and performance, or both are still unclear (Curtis et al., 1975; Blanc, 1999). However, Wyatt et al. (2007) recently identified a cellular mechanism by which feedyard and swine dust extracts are thought to induce inflammatory responses in lung cells.

Pathogens and Endotoxins

Airborne pathogens or endotoxins can potentially affect the health of livestock, employees, and neighbors (Cole et al., 2000; Omland, 2002). The pathogens in AFO with the greatest potential to infect humans include some species of *Escherichia coli*, *Salmonella*, *Listeria*, *Campylobacter*, *Staphylococcus*, *Clostridia*, and *Cryptosporidium*. The concentrations and predominant genera of airborne bacteria are affected by housing system, ventilation, feeding practices, animal species, animal age, and management practices (Chang et al., 2001). Endotoxins are highly resistant to radiation and temperatures, and thus are often present in the air of confinement buildings (Eduard, 1997a,b; Zhang et al., 1998) and open feedlots (Purdy et al., 2004). Pathogens and endotoxins may be free in the air or carried on dust particles. Few living gram-negative pathogenic bacteria have been cultured in open-lot feedyard air (Purdy et al., 2002; Wilson et al., 2002). This may be due to rapid killing of gram-negative organisms by radiation and desiccation. However, pathogens have been cultured from air within swine and poultry buildings (Eduard, 1997a,b; Predicala et al., 2000; Zucker et al., 2000; Chang et al., 2001). Although not well studied, it is highly probable that AFO atmospheres also contain fungal anti-

gens and mycotoxins (Eduard, 1997a,b).

Greenhouse Gases

Carbon dioxide, methane, and nitrous oxide are the 3 primary greenhouse gases associated with livestock production and manure handling. Greenhouse gases trap long-wave radiation near the earth's surface, creating a "greenhouse" effect that warms the atmosphere; thus they pose an environmental impact, rather than a direct health or nuisance impact. However, there are few direct measurements of these gases from livestock facilities to help define emission rates.

Carbon dioxide, methane, and nitrous oxide gases are formed through numerous processes including fermentation, aerobic and anaerobic respiration, or enteric methanogenesis. Most manure storage systems use anaerobic storage and thus release greenhouse gases (Sharpe and Harper, 1999). Greenhouse gas losses from manure and enteric fermentation are highly variable and highly dependent upon temperature, diet, and management (Johnson and Johnson, 1995; NRC, 2003).

There is little information on nitrous oxide emissions from AFO, and the data that are available vary greatly (Singurindy et al., 2007). However, research shows that the amount of nitrous oxide generated and emitted from soil depends on soil temperature and on the quantity of nitrate, carbon, water, and oxygen in the soil (Berges and Crutzen, 1996). Nitrous oxide emissions from dairy farms originate about equally from 3 categories: manure management, feed production, and the redeposited volatilized N and leached N (Berges and Crutzen, 1996; Brown et al., 2002).

DISPERSION PROCESSES

Emission and dispersion of gases within the atmosphere are controlled by the concentration of the gas, wind speed, type of surface, and atmospheric stability. Dispersion models can be used to assess the impact of livestock operations on downwind areas and may also be used to establish setback distances. However, there is no general consensus on the best dispersion model to use for AFO, and comparisons of different models have reported large (5- to 200-fold) differences among the available models used to establish set-back distances (Piringer and Schauberger, 1999; Chaoui and Brugger, 2007).

Gases or particulates in the air are transported by the wind. As with other items, the faster the wind velocity, the greater the ability of the air to transport gases and PM within the atmosphere. The velocity of the wind increases with height above the earth's surface. In addition to gradients in wind speed, there are also gradients in air temperature with height. Variations in wind speed profiles and air temperature gradients combine to affect the mixing and diffusion of gases and PM in the air; therefore these 2 forces impact how gases and particulates are emitted, dispersed, and transported in the air (Chen et al., 1998). The pattern of air movement is affected by terrain, buildings, obstructions, plants, and other objects. Thus, dispersion around agricultural areas is normally complex. As the surface roughness increases, the rate of mixing increases and the rate at which materials are moved in the air is enhanced. Typical wind directions and velocity vary throughout the year and from specific locations.

CURRENT PROCEDURES TO ASSESS, MEASURE, OR AUDIT AIR POLLUTANTS

A number of factors complicate air monitoring at CAFO. In some cases, air quality regulations are based on atmospheric concentrations, and in other cases they are based on actual emissions. Because environmental conditions vary greatly, atmospheric concentrations and ambient emissions are usually not highly correlated, especially from open-lot facilities.

Determination of the concentrations of ammonia, hydrogen sulfide, PM, VOC, and odorants from AFO often requires highly sophisticated and expensive equipment and considerable labor. Because of large spatial and temporal variability, concentration measurements should be taken over extended periods of time and include all the annual seasons. For data regarding atmospheric concentrations and emissions to have maximum value, the facility, animals, diets, management, and weather should be adequately described.

There is considerable controversy about the best method(s) to estimate emissions of different air pollutants from AFO, and these estimates can be difficult and prone to errors. One method normally used by regulators to estimate emissions from a specific location is to multiply an emission factor (normally presented as the quantity of gas or particulate load per unit animal per unit time: EPA, 1986, 1988, 1995, 1996a) by the number of animals at the site. In general, emission factors are only crude averages and do not apply to specific locations or environmental conditions (Misselbrook et al., 2000). The National Research Council (NRC, 2003) criticized the emission-factor approach for estimating AFO emissions and recommended a process-based modeling approach to replace it so that weather conditions, management practices, and manure handling technologies could be explicitly acknowledged on a site-specific basis.

Other methods for estimating emission rates include mass balance, micrometeorology, flux chambers, and models. These methods require several components, including a good technique for accurately measuring the concentration of the gas or PM in the atmosphere, measure-

ments of atmospheric flow and stability, and thorough documentation of the livestock operation (e.g., number of animals, area, management, diet, age, type, health, and housing type) and environmental conditions. Preferably, a multitude of methods should be used to estimate emissions and, when feasible, for example with ammonia-N, a complete nutrient balance should be calculated for the facility to assure the values are reasonable.

Air quality observations that have been made in research studies are generally very short in duration and represent a specific site at a specific time, rather than a source area over an extended time period. Some long-term data that can be used to help guide agricultural management and air quality can be taken from the National Atmospheric Deposition Program (NADP, 2007) and the Clean Air Status and Trends Network (EPA, 2007e), which use a number of stations across the United States to document wet and dry deposition of nitrates, sulfates, ammonium, and other nutrients (Demerjoian, 2000).

MEASURING ATMOSPHERIC CONCENTRATIONS OF POLLUTANTS

Ammonia

Atmospheric ammonia concentrations in AFO can range from < 1 ppm (open-lot feedlots and dairy farms) to > 100 ppm (poultry and swine houses). A number of active methods are currently available to measure atmospheric ammonia concentrations including gas washing (Figure 1), annular denuders, open-path or tuned diode lasers (Figure 2), Fourier-transformed infra-red spectroscopy (**FTIR**), ultraviolet differential optical absorbance spectrometry (**UVDOAS**), and chemiluminescence (Phillips et al., 2000, 2001; Mount et al., 2002; Todd et al., 2005, 2006; Marti et al., 2007). A number of passive samplers are also available (Rabaud et al., 2001;
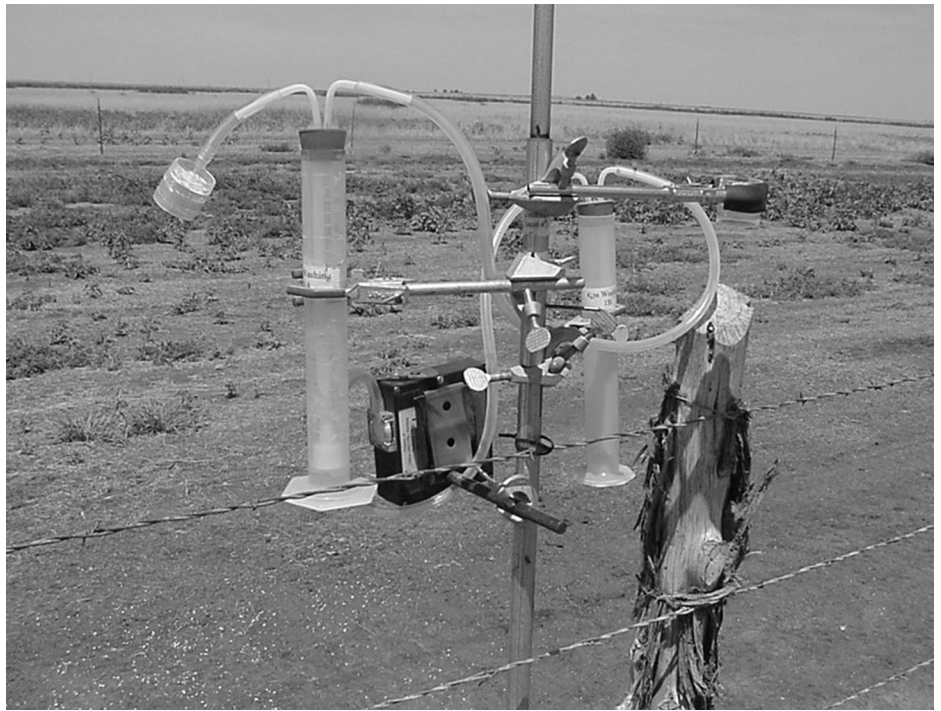

Figure 1. Impingers with portable pump used to sample air for ammonia downwind of a 50,000 head feedyard. A 2-μ Teflon prefilter is used to remove dust particles. The first impinger contains 0.1 *M* sulfuric acid and the second contains deionized water to trap acid fumes to prevent damage to the portable pump. A passive ammonia sampler (Rabaud et al., 2001) is also attached.

Welch et al., 2001; Scholtens et al., 2003). Each method has its own advantages and disadvantages. For example, compared with many other methods, gas washing is relatively inexpensive, but very labor intensive. In addition, gas washing, denuders, and passive samplers provide average ammonia concentrations over an extended sampling period, normally 1 to 4 h; whereas FTIR, UVDOAS, lasers, and chemiluminescence can give near-real-time concentrations and almost continuous (i.e., 5-s average) readings. Open-path lasers, UVDOAS, and FTIR also have the advantage of providing an average concentration over an extended sampling path of 50 to 500 m. To assure that values are accurate, in general, the more mechanized methods and passive samplers should be calibrated against gas washing or standardized gases or both. In our experience, the ammonia concentration in stan-

dardized gases can differ considerably from the designated concentration. Therefore, we believe even standardized gases should be calibrated using gas washing.

Ammonia readily adsorbs to many surfaces; therefore, any sampling lines must be as short as possible to avoid loss of ammonia from the sample. Because AFO tend to have relatively high dust concentrations, methods to measure ammonia and other gaseous contaminants must be either unaffected by the dust, or a method to remove the dust, such as a cyclone or prefilter, must precede the detector. When filters are used, Teflon is preferred over more adsorbent materials; however, tentative data at the USDA-ARS Conservation and Production Research Laboratory (N. A. Cole, unpublished data) suggest that the filter material has little, if any, effect on measured ammonia concentrations when using gas washing and short (< 10 cm) sampling lines.
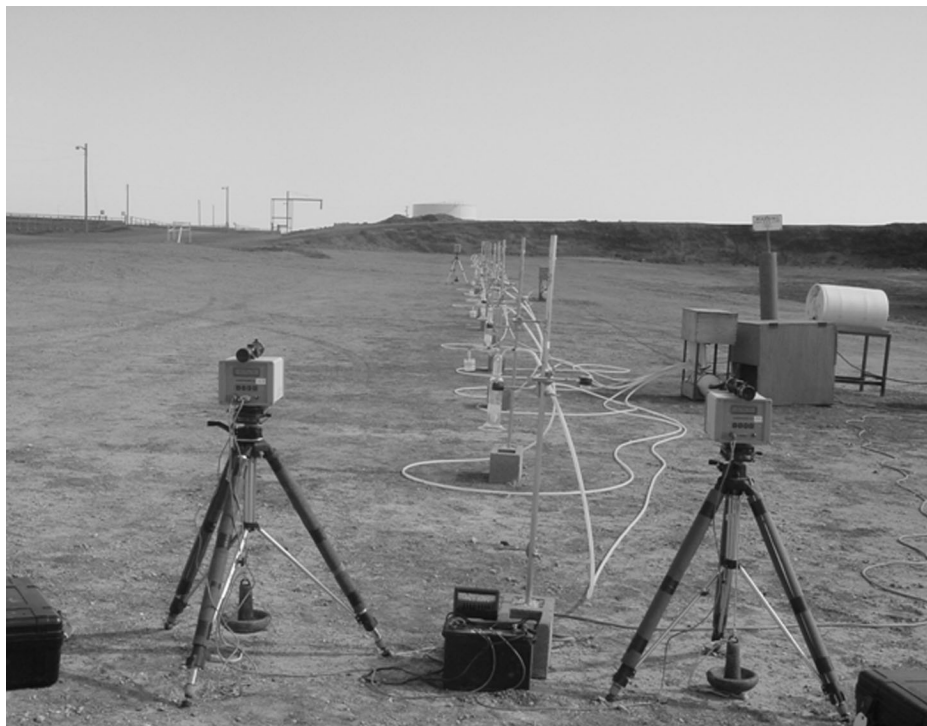
Figure 2. Open-path ammonia lasers measuring ammonia concentrations over a 100-m path at a 50,000 head feedyard. Gas washing bottles are set up at 20-m intervals to calibrate the lasers.

Hydrogen Sulfide

A number of instruments are available for measuring hydrogen sulfide concentrations. One frequently used method is the Jerome meter (Figure 3; Arizona Instruments LLC, Tempe, AZ). The instrument uses a gold film sensor to adsorb reduced sulfur; then the change in resistivity of the film surface is measured and converted to hydrogen sulfide concentration. The instrument detects reduced S concentrations from 2 ppb by volume to 50 ppm by volume. Because the instrument measures total reduced S rather than hydrogen sulfide specifically, it will also detect other S-bearing compounds such as dimethyl sulfide and methyl mercaptan (Winegar and Schmidt. 1998). High concentrations of ammonia, sulfur dioxide, and water vapor do not appear to interfere with Jerome meter readings.

A frequently used laboratory method to determine atmospheric hydrogen sulfide is the TEI Model 45C hydrogen sulfide monitor (Thermo Electron Corp, Franklin,

Figure 3. A Jerome meter with inlet air sampling port to the top.

MA.). For use in the field it must be placed inside a protective housing. The instrument catalytically converts hydrogen sulfide to sulfur dioxide and then analyzes the sulfur dioxide concentration using a pulsed fluorescence analyzer. The instrument has a range of about 0.1 to 100 ppm by volume. The open-path UVDOAS (Marti et al., 2007) can also be used to measure hydrogen sulfide.

Particulate Matter

Agricultural dusts have much larger particle sizes than urban or industrial emissions. Because large particles tend to dominate in agricultural dusts, the validity of PM_{10} and $PM_{2.5}$ concentrations determined at AFO using EPA-approved samplers (known as Federal Reference Methods, **FRM**) developed for urban air sampling has been questioned (Auvermann et al., 2000; Wang et al., 2005; Buser et al., 2007a,b). The FRM for PM_{10} and $PM_{2.5}$ are filter-based, gravimetric samplers that were designed for use in urban and industrial settings in which ambient PM tends to be dominated by fine particles (i.e., less than 10 μ AED). The FRM do not measure particle sizes directly, but rely on particle aerodynamics as well as sampler-specific geometrics, airflow paths, and airflow rates to separate particles into 2 size fractions. Theoretically, particles larger than the sampler's "cutpoint" are captured in an inertial pre-separator, and particles smaller than the cutpoint pass through the pre-separator and are collected on a filter (micro quartz for PM_{10} and Teflon for $PM_{2.5}$). The increase in the weight of the filters from before to after the sampling period is divided by the volume of air sampled to obtain the PM concentration. However, PM samplers that rely on inertial pre-separation do not perform perfectly. Some larger particles will penetrate the pre-separator to the filter (over-sampling), and the pre-separator will also collect some of the particles

that are smaller than the cutpoint (under-sampling). When FRM samplers are deployed in areas in which ambient PM tends to be dominated by fine particles, the errors associated with over-sampling and under-sampling offset one another, and the resulting measurements are very close to reality. However, when the samplers are deployed in agricultural settings in which the ambient PM is much larger than the sampler's design cutpoint, the effect of the over-sampling error is magnified and the under-sampling error does not compensate for that bias. The result is an upward bias of the measured concentration that increases in magnitude as the difference between sampler cutpoint and ambient particle size increases (Auvermann et al., 2000; Buser et al., 2007a,b). Because of the over-sampling bias of FRM samplers, Buser et al. (2007a) recommended that when used in agricultural settings, FRM monitoring for PM_{10} or $PM_{2.5}$ needs to be augmented by concurrent and collocated TSP monitoring with ensuing particle size distribution analysis to determine the true fractions of the populations of PM caught on the TSP sampler filters.

Odors and VOC

Standardized methods for measurement of odors from AFO have not been developed (Hobbs et al., 1995). In addition, measurement of odorants and VOC is difficult because the concentrations are normally very low (ppb or ppt in air), the compounds may adhere to some surfaces, and because odorants will react with other chemicals in the air to produce new chemicals (Parker et al., 2005a,b, 2007; Miller and Woodbury, 2006). When studying odor and its effects on people living near CAFO, 4 characteristics are typically used: 1) frequency or how often the odor occurs, 2) intensity or concentration of the odor, 3) duration or how long the odor is present, and 4) offensiveness or hedonic tone

(Sweeten, 1995; Redwine and Lacey, 2000). (Not all authors use these terms to mean precisely the same thing, therefore the reader should be cautious in interpreting others' data, analysis, and conclusions.)

Olfactometry using human panelists has long been used as a method to quantify odors, as the human nose can often detect odors below the detection levels of current analytical equipment (CEN, 1999; ASTM, 2001; Parker et al., 2007), and continues to be one of the primary methods of odor assessment for animal feeding operations (Jones et al., 1992; Zhu et al., 1999; Lim et al., 2001; Galvin et al., 2003; Gay et al., 2003; Bicudo et al., 2004). One of the difficulties with olfactometry is the inherent variability between odor panelists (Sweeten et al., 1983; Sweeten, 1995; Clanton et al., 1999).

Olfactometry can be conducted in the laboratory or in the field, and there are specialized instruments for both. In field olfactometry, the human panelist uses a portable scentometer for diluting the ambient air prior to presentation to the nose. By opening and closing holes of different sizes, the human panelist can control the dilutions of clean air to odorous air.

For laboratory olfactometry, the air sample must be transported from the field to the laboratory for presentation to a human panel typically consisting of 4 to 8 people. Samples are collected in the field using a vacuum apparatus in bags constructed of relatively nonadsorbent materials such as Tedlar, Teflon, or Melinex. One difficulty with sampling in plastic bags is that odorants can adsorb to the sides of the bags and greatly affect the odor concentration (Koziel et al., 2005; Perschbacher-Buser et al., 2006). The bag is connected to a laboratory olfactometer, which dilutes the odorous air with clean air scrubbed through an activated carbon filter. The odor "detection

threshold" (**DT**) is usually determined using triangular forced-choice olfactometry, where the odor is compared with 2 other clean air presentations, and the panelist is asked to pick which one is different. The DT is a measure of the ratio of dilutions of clean air to ambient (odorous) air at which 50% of the human panelists can just detect the presence of an unrecognized odor (Sweeten, 1995; ASTM, 2001). Detection threshold is dimensionless and commonly reported as odor units. In addition to the DT, the "recognition threshold" (**RT**) can be used to characterize odor strength. The RT is the concentration at which a panelist can describe the odor. For a given panelist, the RT will always have a greater concentration than the DT. There is often a poor correlation between field olfactometry and laboratory olfactometry (Sheffield et al., 2004), probably because of the inherent variability in sample collection methods and among panelists.

Because human olfactometry alone does not provide the scientific information needed to research odor abatement methods, the scientific community is now using measurement technologies, such as gas chromatography-mass spectrometry (**GC/MS**) and proton-transfer-reaction mass spectrometry to quantify chemical compounds present in odor samples (Sunesson et al., 2001; Wright et al., 2005a,b; Filipy et al., 2006; Shaw et al., 2007). With GC/MS, VOC present in ambient air samples are typically concentrated by adsorption using solid phase microextraction fibers or sorbent tubes. In the laboratory, the fiber or tube is heated to volatilize the VOC into the GC column. The VOC separate based on their molecular weight and polarity and are quantified by a detector. Although flame ionization detectors are sometimes used, MS detectors are often used to better identify the various chemicals.

A recent technology improvement in odor science is a combina-

Figure 4. A gas chromatograph-mass spectrophotometer-olfactometer.

tion of the GC/MS with an olfactometry port (GC/MS-O; Microanalytics Inc., Round Rock, TX; Figure 4). Between the separation column and the GC/MS unit, a portion of the column exhaust is diverted to a sniffing port at which the operator continually observes the relative intensity and hedonic tone of the odor associated with each volatile component exiting the column over time using a touch-screen monitor. This results in a so-called "aromagram" (Wright et al., 2005a,b; Figure 5).

Microbes and Endotoxins

Bioaerosols of AFO are a complex mixture of live and dead microorganisms, their products, and other aeroallergens. Unfortunately, no standardized methods have been developed to sample or analyze bioaerosols at AFO, making comparison across experiments or locations difficult.

Microorganisms in the atmosphere are often sampled using 2-stage or 6-stage Andersen biological cascade impactors (Andersen Sampler Inc., Atlanta, GA). Petri dishes containing the desired selective media are placed in each stage to "trap" viable microorganisms. Each stage of the Andersen sampler contains various size holes designed to mimic portions of the human respiratory system in which inhaled particles may deposit (6-stage sampler: stage 1 = nasal passages and sinuses, stage 2 = pharynx, stage 3 = trachea and primary bronchi, stage 4 = secondary bronchi, stage 5 = terminal bronchi, and stage 6 = alveoli; 2-stage sampler: stage 1 = nonrespirable, stage 2 = respirable). However, the sampling efficiency is affected by airflow rate (Stewart et al., 1995) and length of sampling time (Folmsbee et al., 2000). Other methods used to assess airborne concentrations of microorganisms include all-glass impingers and Nuclepore filtration and elution (Thorne et al., 1992; Chang et al., 2001). The optimal sampling method appears to be dependent upon the purpose of the sampling, expected bioaerosol concentrations, organism of interest, and en-

vironmental conditions (Thorne et al., 1992).

Culture methods have routinely been used for measurement of airborne microorganisms (Eduard, 1997a,b; Purdy et al., 2002, 2004, 2007). However, nonculturable microorganisms, which can potentially cause adverse health effects, can not be measured via culture methods. In addition, aerosolization of gram-negative microorganisms can affect their culturability (Heidelberg et al., 1997). Thus, plate culturing and counting techniques may not provide an accurate description of atmospheric bacterial burdens. In addition, sampling times are frequently short (5 min or less). More "hardy" organisms can be collected using filters and quantified by nonculturable methods (Eduard, 1997b). The use of molecular techniques that can give more accurate estimates of possible exposure to pathogens is increasing.

Bacterial endotoxins (the heat-stable lipopolysaccharide in the cell envelopes of gram-negative bacteria) are usually measured by biological assays based on the reaction of *Limulus* amoebocyte lysate with the lipopolysaccharide (Eduard, 1997b). Although the precision of assays performed within individual labs is good, there can be significant differences in values reported from different labs on the same samples (Reynolds et al., 2002). Cole et al. (2000) reported that a difference of up to 17-fold in apparent airborne endotoxin concentrations could be obtained using different sampling, storage, extraction, and analysis methods.

Greenhouse Gases

Methane is routinely measured using a gas chromatograph equipped with a flame ionization detector with tunable diode lasers (Sharpe and Harper, 1999; McGinn et al., 2006), or with FTIR (Amon et al., 2001). Nitrous oxide is normally measured via gas chromatog-
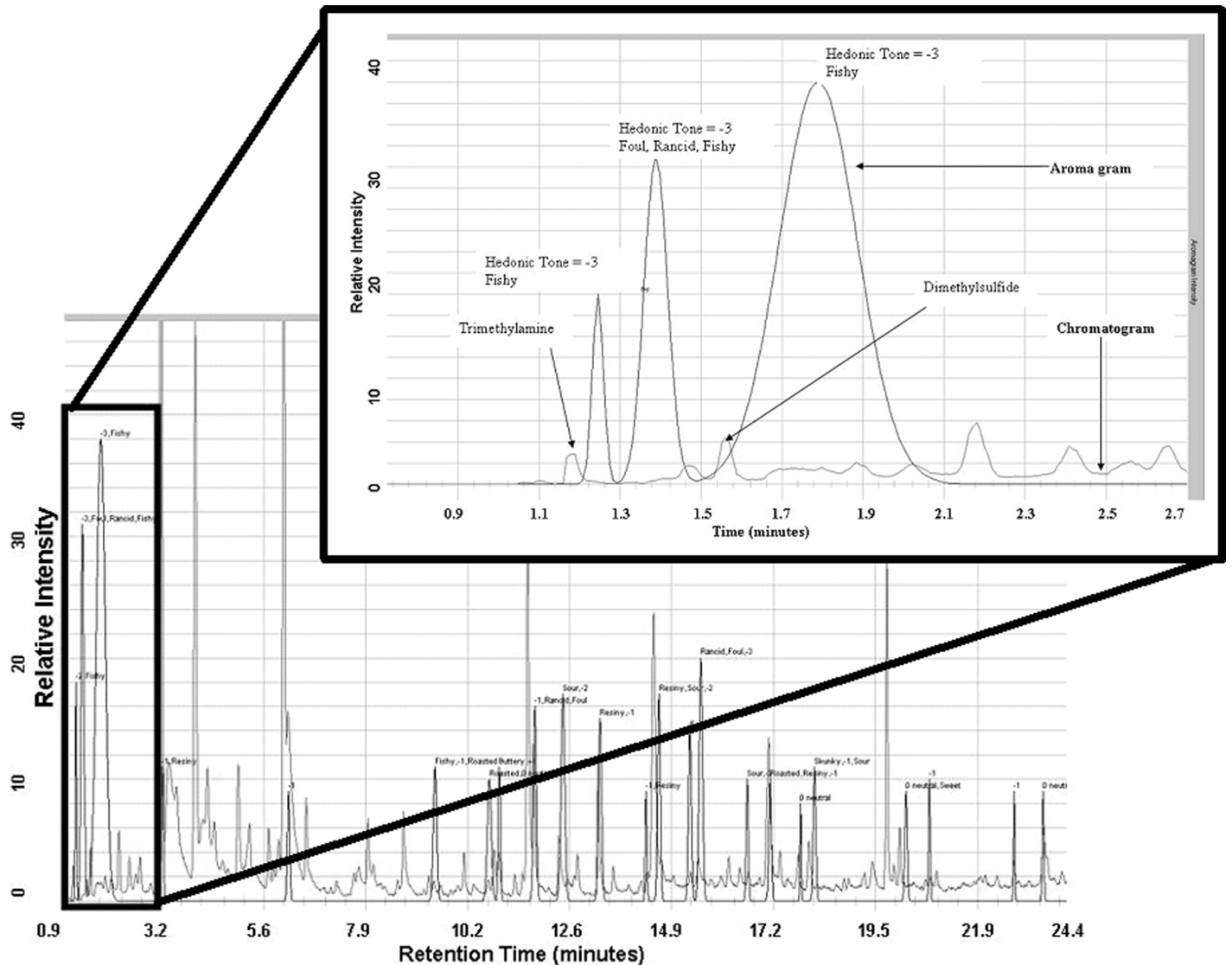
Figure 5. An example chromatogram and the corresponding aromagram produced with the gas chromatograph-mass spectrometer with an olfactometry port. This sample was taken downwind of a fish meal plant and was used to identify the primary odorants trimethylamine and dimethylsulfide. Note the small odor peaks associated with the large chromatogram peaks from 5.6 to 24.4 min, in contrast to the enlarged image which shows large odor peaks associated with the very small chromatogram peaks in the first 2.7 min.

raphy using an electron capture detector (Berges and Crutzen, 1996), but has also been measured using tunable diode lasers (Brown et al., 2000, 2002; Kulling et al., 2001) or FTIR (Griffith and Galle, 2000; Amon et al., 2001). These techniques have been used for ambient air samples, for grab samples captured in canisters, and for samples from flux chambers. Because ruminants may produce a considerable quantity of methane via ruminal fermentation, methane emissions

from cattle facilities need to be partitioned between the enteric and manure or lagoon fermentation.

General

In general there is no one "best" method to measure most air pollutants. The optimal method will be determined by cost, labor availability, objectives, the environment, the facility characteristics, concentrations of the pollutant to be measured, and other factors.

MEASURING EMISSIONS OF POLLUTANTS

Quantifying gaseous emissions from AFO entails 2 major challenges: 1) measuring the concentration of the gas of interest, and 2) obtaining an estimate of flux from the surface to the atmosphere based either on direct measurement or on a flux model that describes or simulates the turbulent dispersion of gases. The method used for measur-

ing emissions will vary depending upon the type of emission and housing system (open-lot vs. lagoon; area source vs. ventilated building or point source).

Flux Chambers and Wind Tunnels

Many studies have used flux chambers or wind tunnels or both to directly measure flux of gases from ground-level area sources. However, flux chambers also have significant limitations, the primary problem being their effect(s) on the emitting surface. In agreement with a number of previous studies, Rhoades et al. (2005) and Cole et al. (2007) noted that ammonia emissions from a simulated retention pond and feedlot surface increased with increasing air exchange rate. Air turnover rates of approximately 15 chamber volumes/min were required to obtain flux rates equivalent to undisturbed ammonia sources. Similarly, Rhoades et al. (2003) noted that ammonia and hydrogen sulfide emissions from feedyard pen surfaces, estimated using flux chambers, were as low as 1% of those determined using 2 dispersion models. Sommer et al. (2004a) noted that emissions of methane, carbon dioxide, and nitrous oxide measured from manure stockpiles using a static chamber were only 12 to 22% of flux measured using a micrometeorology method. Many flux chambers routinely used to measure flux have air exchange rates of 0.5 chamber volumes/min or less (Kienbusch, 1986). In addition, emissions of some pollutants, such as ammonia, are concentrated in small areas (i.e., urine spots); therefore many measurements (> 100) must often be made to account for spatial variability (Cole et al., 2007). Thus, most emission estimates based on flux chambers must be viewed with caution. It appears that flux chambers can be used to effectively partition emissions from different segments of the AFO (manure stockpiles, lagoon, or pen surface)

(Sommer et al., 2004a) and to measure treatment differences (Meisinger et al., 2001).

Micrometeorological Methods

Micrometeorological methods to determine gaseous emissions to the atmosphere are advantageous because they do not interfere with the processes of emissions and they can integrate emissions over large areas (McGinn and Janzen, 1998; Fowler et al., 2001; Harper, 2005). Generally speaking, micrometeorological methods rely on concentration measurements in the site of interest and characterization of the atmosphere near the ground. They have been successfully applied to crops (Denmead et al., 1978; Harper and Sharpe, 1995; Rana and Mastrorilli, 1998), open-lot AFO (Hutchinson et al., 1982; McGinn et al., 2003; Todd et al., 2005, 2007; Flesch et al., 2007) and AFO lagoons (Harper et al., 2000; Harper 2005).

The only micrometeorological method that directly measures turbulent transfer is the eddy covariance method (Fowler et al., 2001; Meyers and Baldocchi, 2005). This method requires very fast measurements (10 to 20/s) of the vertical velocity of turbulent eddies and gas concentration. Sonic anemometers are used to measure the vertical eddies and fast-response instruments such as tunable diode lasers are used to simultaneously measure the concentration of the gas of interest of those eddies. Alternatively, in the relaxed eddy accumulation method, air from up-eddies and down-eddies is segregated, usually with denuders, and after a period of accumulation, concentration is measured using ion chromatography or other methods (Ham and Baum, 2007).

Mass balance methods account for the amount of a gas that passes across the upwind edge of an emitting surface and the amount that passes across the downwind edge, so that the difference is the amount emitted. Mass balance methods as-

sume that source strength is homogeneous, airflow is fully turbulent, and that the boundaries of the system (i.e., plume dimensions) are defined. The surface is considered the lower boundary, and the upper boundary (i.e., top of the plume) is defined as the height where gas concentration equals background concentration. The Integrated Horizontal Flux method uses measurements of wind speed and gas concentration profiles to calculate the horizontal flux (the product of wind speed and concentration) at various heights (Wilson et al., 1983; Wilson and Shum, 1992); integrating the horizontal fluxes gives the vertical flux. Typically, circular plots are used to simplify the determination of upwind source area (Yang et al., 2003; Todd et al., 2006), although the method can be used with strip sources (Denmead et al., 1977), irregularly shaped fields (Flesch et al., 2002; Laubach and Kelliher, 2004), or finite volumes (Denmead et al., 1998).

A variant of the mass balance method is the box model. Box models have been used to measure ammonia flux from beef and dairy operations (Freitas et al., 1997; Ashbaugh et al., 1998) and from swine operations (McCulloch et al., 1998). Ammonia flux from a 50,000 head beef cattle feedyard, measured using the box model of Ashbaugh et al. (1998) gave emissions similar to the flux-gradient, backward Lagrangian stochastic (**bLS**; see later text), and N-balance (see later text) methods (N. A. Cole and R. W. Todd, unpublished data).

The aerodynamic flux-gradient (**FG**) method treats turbulent flux as analogous to molecular diffusion (McGinn and Janzen, 1998; Fowler et al., 2001; Harper, 2005). Flux of a gas is the product of the vertical concentration gradient of the gas and an eddy diffusivity, which varies with wind speed, atmospheric stability (Fowler et al., 2001), and distance from the surface (Thom, 1975). The FG method requires pro-
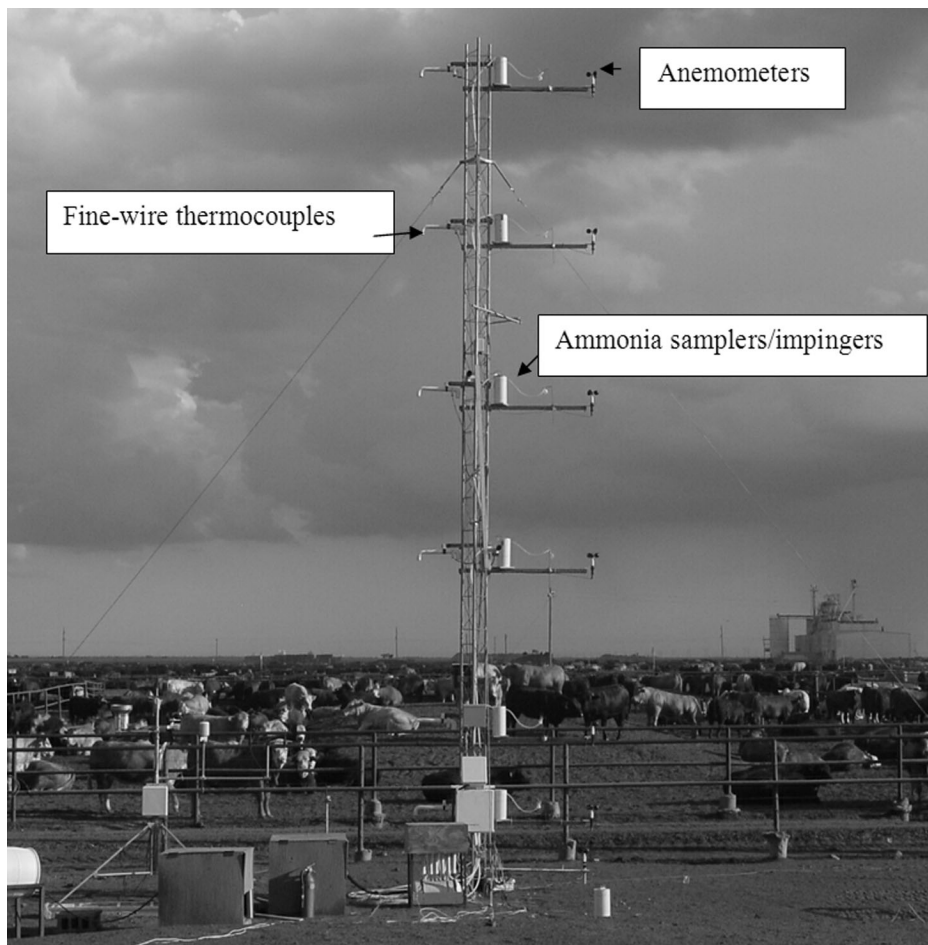
Figure 6. A 10-m micrometeorology tower at a 50,000-head feedyard used to determine ammonia emissions using the flux-gradient method. Temperature, wind speed, and ammonia concentrations are determined at heights of 0.5, 1, 2, 4, 6, 8, and 10 m.

file measurements of gas concentration, wind speed, and air temperature (Figure 6). The FG method also assumes that there is horizontal uniformity of airflow, that horizontal concentration gradients are negligible, and that vertical flux is constant with height (Thom, 1975; Harper, 2005). In situations of disturbed flow, such as those encountered at AFO, these assumptions may be violated and the FG method could underestimate flux (Wilson et al., 2001).

Dispersion Models

More complex dispersion models are based on a description of the relationship between a source of a gas and a downwind receptor or point (Harper, 2005) using assumptions about turbulent flow (Wilson et al., 2001). Sometimes, source strength of a gas is known and the dispersion model is used to predict concentrations at a specified distance downwind, and in other cases the reverse is true. Gaussian plume models are an example of this type of dispersion model, in which empirical parameters describe the 3-dimensional spread of a plume of gas from its source. The bLS model estimates flux of a gas by taking concentration of a gas measured at a point downwind of an emitting source and modeling the trajectories of thousands of gas particles backward to the emitting source (Flesch et al., 1995). Advantages of the bLS model include a small number of required inputs (gas concentration, wind speed and direction, atmospheric stability, and defined source area; Flesch et al., 1995; Laubach and Kelliher, 2005; Sommer et al., 2005).

The EPA provides a list of preferred air quality models (CFR, 2007). Until recently, the EPA and most state pollution regulatory agencies used the Industrial Source Complex Short Term Version 3 model as their regulatory model. However, this has recently been replaced with the American Meteorological Society-EPA Regulatory Model with Plume Rise Model Enhancements (AERMOD-PRIME). Both are Gaussian models, the accuracy of which in agricultural situations has been debated (Beychok, 1996). Lagrangian stochastic models are often used by researchers as alternatives to Gaussian models. Using the same data set, these models give feedyard ammonia emission rates that differ by as much as 50% (Faulkner et al., 2006). Thus, in general, flux estimates obtained using one model should not be used to predict downwind concentrations using a different model.

Nutrient Balance

For some gaseous emissions, such as ammonia from dry-lot pen surfaces, it appears that a total N-balance for the AFO (simply N in feed and water minus N retained by animals and in manure) can give reasonable estimates of ammonia emissions because most of the gaseous N losses are as ammonia, rather than N_2O, N_2, or NO_x (Todd et al., 2005, 2007). Using the ratio of N to a nonvolatilizable mineral (usually P) in the diet and in "aged" manure (combination of feces and urine) from the pen surface can also give reasonable estimates of ammonia losses from dry-lots, as long as fresh urine spots are not sampled. Harper et al. (2000) reported that a considerable portion of the volatile N losses from swine lagoons were as N_2; thus, a N-balance of N entering

and leaving lagoons may over-estimate ammonia losses.

Flux Measurements from Enclosed Housing (Point Sources)

A number of differing methods have been used to estimate emissions from enclosed animal houses (Monteny and Erisman, 1998; Dore et al., 2004; Scholtens et al., 2004). In general, emissions are determined by multiplying the measured pollutant concentrations within the house or in the air stream leaving the building by the volumetric flow rate (Zhu et al., 1999; Xin et al., 2003). Livestock buildings may be ventilated in a number of mechanical (negative pressure, positive pressure, or neutral) and natural ways. Measurement of ventilation rates, and thus flow rates, is difficult and potentially prone to errors (Bottcher et al., 1996). Ventilation rate is affected by a number of factors including fan performance, weather conditions, and building environment. An alternative approach is to measure the ambient concentrations upwind and downwind of the building and back-calculate the emission rate using dispersion models (Flesch et al., 2005).

Tracer Methods

Some studies have used atmospheric tracers to assist in measuring gaseous emissions. The tracer used most frequently is probably sulfur hexafluoride (SF_6; Kaharabata et al., 2000; McGinn et al., 2006). Known quantities of the tracer are released from the area(s) of interest, upwind of the sampler, to mimic flux of the gas of interest. The concentration of the gas of interest and the tracer are determined downwind of the emission site. The emission of the gas of interest is then calculated by multiplying the true emission of the tracer by the ratio of the gas of interest and the tracer gas. For accurate determinations, the tracer must disperse in a man-

ner similar to the gas of interest. Tracer methods are usually limited to use in relatively small source areas (Harper, 2005). Tracers can also be used to determining ventilation rates from enclosed buildings.

Statistical, Empirical, and Process-Based Models

A number of empirical, statistical, and process-based (mechanistic) models have been developed to estimate emissions from AFO and manured fields. Most of these models have been concerned with ammonia emissions, possibly because the chemical and physical factors affecting ammonia formation and emission are better understood than other emissions. In general, statistical models are based on experimental data derived from monitoring emissions at a specific facility; thus they are often site-specific, may contain a high degree of uncertainty, and a potentially large error may occur when applied to other sites. Empirical models are frequently derived from more controlled experiments and are often limited to conditions under which they were developed. Mechanistic models describe emission processes via chemical and physical transformations, transfer, and equilibria within the modeled system.

As a first step in developing models to predict atmospheric emissions of some pollutants, models describing the effects of diet and management factors on nutrient excretion (Baldwin et al., 1987; de Boer et al., 2002; Monteny et al., 2002; Fox et al., 2004; Kebreab et al., 2004; Burgos et al., 2005) or enteric fermentation (Wilkerson et al., 1995; Rossi et al., 2001; Mills et al., 2003; Garnsworthy, 2004) have been developed. Once the route (urine vs. feces), chemical form (urea, volatile fatty acids, organic, inorganic, etc.), and quantity of nutrient excretion is modeled, it becomes necessary to partition the nutrients to different storage or treatment locations (pen surface, manure pit, lagoon, reten-

tion pond, etc.). The chemical and physical processes that occur during manure storage and treatment must then be modeled.

Ni (1999) reviewed a number of mechanistic models developed to estimate ammonia losses from swine lagoons. Most of these models require only measurements of pH, total ammonia-N, temperature (water and air) and wind speed to estimate ammonia flux. However, Ni (1999) noted that different ammonium and ammonia dissociation constants, Henry's constant, and mass transfer coefficients were used in the models. The mechanistic model of DeVisscher et al. (2002) developed to estimate ammonia emission from swine lagoons also appears to predict ammonia emissions from beef cattle retention ponds with reasonable accuracy; however, a statistical model they developed from swine lagoon data was very inaccurate when used to estimate ammonia emissions from beef cattle retention ponds in Texas (N. A. Cole and R.W. Todd, unpublished data). Additional models of ammonia emission from anaerobic lagoons (Liang et al., 2002; Rumburg et al., In press-b), manure pits (Olesen and Sommer, 1993; Zhang et al., 1994), swine houses (Aarnink and Elzing, 1998), dairy houses (Elzing and Monteny, 1997; Rumburg et al., In press-a), broiler litter (Carr et al., 1990), compost (Paillat et al., 2005), following field application of manures (Genermont and Cellier, 1997; Menzi et al., 1998; Sommer and Olesen, 2000; Guo et al., 2001; Wu et al., 2003), and from whole farms (Pinder et al., 2004; Zhang et al., 2004) have also been developed.

Several models have been developed to predict odor emissions and odor dispersion from AFO and manured fields, as well as downwind odor concentrations (Janni, 1982; Carney and Dodd, 1989; Smith, 1993; Guo et al., 2001; Schulte et al., 2007). Most are empirical and are based primarily on meteorological conditions or manure handling

methods. Henry et al. (2007) developed an empirical model to predict odor emissions from feedyard retention ponds that was based on air temperature, effluent inflow volume, and days since last inflow event.

To our knowledge, few models have been developed to estimate other AFO emissions such as dust, hydrogen sulfide, and greenhouse gases. Auvermann (2003) developed a preliminary mechanistic model to predict fugitive PM emissions from beef cattle feedyard surfaces, and Razote et al. (2006) reported preliminary values of some parameters in that model using a benchtop hoof-action simulator. Sommer et al. (2004b) developed a simple model to predict methane and nitrous oxide emissions from livestock manures and the EPA (1990) has used the rumen model of Baldwin et al. (1987) to estimate enteric methane emissions from ruminants.

THE FUTURE

While an awareness of air quality issues is a good start, it is not enough. It is imperative that everyone associated with livestock production and its potential impact on air quality begin to address the issue. The effects of air pollution from AFO on quality of life, land values, and the ability of communities to attract and maintain educational, industrial, and medical facilities also need to be determined (Cole et al., 2000). The USDA-Natural Resources Conservation Service is in the process of developing an Agricultural Air Quality and Atmospheric Change Planning Tool (Johnson et al., 2007) to assist Natural Resources Conservation Service employees and producers in identifying important air quality concerns and provide management information to help mitigate air quality impacts and emissions.

Training Auditors

In the future, requirements for monitoring of air emissions from livestock and poultry operations will vary from state to state and will need to be adapted for the specific type of operation. For example, in typical open-lot beef cattle feedyards, ammonia emissions from retention ponds are less than 5% of total ammonia emissions (Flesch et al., 2007), whereas in swine operations, ammonia losses from the lagoon may be 30% or more of total ammonia losses (Doorn et al., 2002). Based on the National Air Emissions Monitoring System agreement, it is probable that producers will be responsible for at least some of the costs of any air quality auditing and monitoring program. However, government incentives, similar to the Environmental Quality Improvement Program, may be developed to help livestock producers reduce air emissions. Obviously, the monitoring of emissions, air quality, or both at every AFO is not practical. Therefore, process-based or empirical models will need to be developed so that emissions or concentrations of air pollutants can be estimated from readily obtainable variables such as diet characteristics, animal characteristics, facility design, and environmental conditions. When monitoring is required, it should be conducted over extended periods so that affected times and places can be determined. To be accurate and fair, auditors (or preferably teams of auditors) will need to be trained in a variety of disciplines including animal sciences, chemistry, engineering, micrometeorology, instrumentation, mathematical modeling, and logic. If air quality auditing and monitoring becomes required in the future, it is highly probable that consulting firms will develop to fill this void.

IMPLICATIONS

Air quality concerns relating to animal feeding operations will continue to grow in the future. Future requirements for auditing and monitoring of air emissions from livestock and poultry operations will vary from state to state and will need to be adapted for specific operations. Producers will probably be responsible for at least some of the costs of any air quality monitoring program. However, government incentives may be developed to provide technical assistance to help livestock producers reduce air emissions. Because the monitoring of air quality at every livestock operation is not practical, process-based models will need to be developed so that emissions and concentrations of air pollutants can be easily estimated. When monitoring is required, it should be conducted over extended periods of time with proper methods. Teams of auditors will need to be trained in a variety of disciplines to assure accurate results.

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