

International Perspectives



The Study and Regulation of Agricultural Air Quality in the U.S.

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Abstract

The livestock industry in the United States today faces two main classes of legal challenges. First, livestock odor is increasingly precipitating expensive nuisance lawsuits from annoyed citizens. Second, emissions of ammonia (NH₃), hydrogen sulfide (H₂S), and particulate matter (PM) from animal feeding operations sometimes exceed regulatory limits (either state or federal) on emissions or property-line concentrations. Both of these problems are exacerbated by increasing consolidation of livestock production.

Nuisance suits by neighbors are usually the first legal action to hit producers, and nuisance cases have been heard in nearly every livestock-producing state (Miner 1995). One key example has been the case of Iowa, where, in 1998, the State Supreme Court ruled that nuisance defense through “Right to Farm” laws was unconstitutional. By 2004, there were at least 14 separate lawsuits pending against Iowa pork producers, the most at any one time in the state’s history (Lee 2004). Six years ago, a Missouri jury awarded \$5.2 million to 52 neighbors for nuisance claims against a major pork-production facility owned by Continental Grain. The facility has since been purchased by Premium Standard Farms, which was itself the target of over 60 individual nuisance suits in one recent 18-month period.

Most state governments have some type of odor regulation, which goes above and beyond the nuisance concept. To regulate odor, some states use property line limits of hydrogen sulfide or odor dilutions to threshold. For example, at least 10 states have standards limiting H₂S concentrations at the property line; these are generally based on 30-minute and 24-hour averages (Mahin 2001). The 30-minute limits can range anywhere from 30 to 120 parts per billion (ppb), depending upon the state. Some state statutes limit the number of times that violations may occur before a legal action is triggered; for example, any facility which exceeds Minnesota’s 30-minute 50-ppb H₂S limit more than twice in a given year can be subject to legal action. Other states have enacted limits that are dependent upon land usage patterns in the vicinity of the facility; Texas, for example, has 30-minute H₂S limits of 80 and 120 ppb, depending on land use. Several states have developed more stringent regulations to control odor and gas emissions. A prime example of this is Colorado’s Amendment 14, passed by voters in 1998; it requires owners of large swine operations to obtain permits, install covers on effluent ponds, adhere to setbacks and bans on land application of wastes, and implement odor-control technologies and work practices (Sweeten et al 2001). North Carolina placed a moratorium on new hog facilities in 1997 and implemented odor rules for existing facilities in 2000. California’s SB-700 legislation removed the previous agricultural exemption from state air regulations; California now will require permitting for dust, ammonia, and volatile organic compounds (VOCs).

The U.S. Department of Justice and the U.S. EPA followed Missouri and Ohio consent decrees related to odor by initiating federal actions against the same swine and layer operations for alleged excessive emissions of PM and NH₃. Both of these cases were eventually settled (in 2001 and 2004, respectively), and resulted in consent decrees with the respective producers. In the first, a Missouri pork producer with 1.25 million pigs was required, among other stipulations, to conduct long-term air emissions monitoring at two of their more than 1,000 barns, and at one of their 163 anaerobic lagoons. The producer was also required to test an approach (sprinkling of soybean oil) for controlling dust emissions. The second case involved an Ohio egg producer with approximately 12 million chickens. This producer was required by a state-issued consent decree to convert many of their deep pit barns to belt battery manure handling systems. The subsequent federal consent decree required that controls for PM and ammonia be instituted at those barns which retained the deep pit configuration, as well as long-term testing of emissions from those barns with a pair of mobile laboratories.

On the horizon are two new proposed federal rules, each of which could pose significant new challenges to the agriculture industry and that heighten the need for high-quality research into agricultural air emissions. The first of these, which was proposed by EPA on January 17, 2006 (USEPA 2006), redefines what size

fractions of particulate matter would be regulated. Existing PM₁₀ standards would be eliminated, and replaced with a standard for “inhalable coarse particles”, or PM_{10-2.5}, defined as particulate matter between 10 microns and 2.5 microns in diameter. There would be no annual standard for PM_{10-2.5} in terms of concentration. The 24-hour standard would be approximately 50% lower than the current 24-hour PM₁₀ standard. Significantly, any case in which the PM_{10-2.5} is “dominated by windblown dust and soils and PM generated by agricultural and mining sources” would be exempted. However, the proposed rule would also lower the 24-hour PM_{2.5} standard by roughly 50%, from 65 µg/m³ to somewhere in the range of 30 to 35 µg/m³, and there would be no agricultural exemption for PM_{2.5}. The second new rule, proposed November 1, 2005 (USEPA 2005), is significant in that it addresses ammonia as a precursor of PM, particularly PM_{2.5}. Because of this, states with nonattainment areas for PM_{2.5} would be advised to assess whether reductions of ammonia emissions would assist in meeting the PM_{2.5} standards. No state would be required to make this assessment unless it, or EPA, could demonstrate that ammonia emissions, in their particular case, “significantly contribute to the PM_{2.5} problem in a given nonattainment area or to other downwind air quality concerns.” Livestock groups, including the National Pork Producers Council (Buhl 2006) have criticized this proposed rule on the grounds that the science behind it is incomplete in two major areas: 1) understanding which fraction of ammonia emissions can be attributed to livestock agriculture, and 2) understanding fully the role of ammonia in PM_{2.5} formation.

State and federal agencies have primarily relied on universities and federal laboratories to increase scientific knowledge about livestock air emissions through laboratory and field experimentation, and computer model development. They have also developed mechanisms to bring this knowledge and expertise to bear in developing sound, science-based policy. In 1997, the federal Agricultural Air Quality Task Force was established, with members from EPA, USDA, industry, and universities. The AAQTF reports directly to the Secretary of Agriculture on issues relating to the nexus between science and policy. The AAQTF issued a white paper on the livestock odor issue in 2000 (Sweeten et al 2000). AAQTF encouragement was one factor that spurred EPA to request a National Academy of Sciences (NAS) study of air emissions from livestock facilities. The report of this study, issued in 2003, highlighted possible air pollution problems arising from animal feeding operations and called for more research to address data gaps.

Methods of on-farm emission measurements have improved significantly over the past 10 years, and real-time monitoring approaches have been used to collect over 500 barn-months of air emission data since 1997, including a recently completed six-state USDA study (Heber et al 2002 a & b). The experience gained in the various monitoring studies over the last decade will now pave the way for the most comprehensive air emissions study ever conducted at agricultural facilities.

National Air Emission Monitoring Study

The National Air Emission Monitoring Study (NAEMS) is required by an innovative and voluntary Consent Agreement between the U.S. EPA and participating livestock industries. Livestock producers agreed to collect air emission data, via the NAEMS, in exchange for temporary protection from further government lawsuits and forgiveness of possible past offenses. Never before has such an air compliance program been implemented with an entire industry.

The specific objectives of the NAEMS are to:

- Determine whether individual farms are likely to emit particulate matter (PM) and volatile organic compounds (VOCs) in excess of applicable Clean Air Act thresholds.
- Determine whether individual farms are likely to emit NH₃ and/or H₂S in excess of applicable Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and Emergency Planning and Community Right-to-Know Act (EPCRA) reporting requirements.
- Form a database with which additional studies of air emissions and effectiveness of control technologies can be compared, and from which emission factors can be developed.

The NAEMS, scheduled to begin in late 2006, is designed, through the use of sound scientific principles and using proven instrumentation and methods, to provide quality-assured air emission data from representative swine, egg layer, dairy, and broiler farms. The NAEMS will employ continuous monitoring from on-farm instrument shelters to determine emissions from barns and micrometeorological methods to

do the same for dairy corrals, lagoons and manure basins. In addition to collecting new data from several farms, the NAEMS will assemble and evaluate existing emissions data from other studies.

At each of one broiler site, three egg production facilities, five swine farms, and six dairies, an on-farm instrumentation shelter (OFIS) will house equipment for measuring pollutant concentrations at representative air inlets and outlets, barn airflows, operational processes and environmental variables. A multipoint gas sampling system will draw air sequentially from representative locations and deliver selected streams to a manifold from which on-line gas monitors draw sub samples. Mass concentrations of PM₁₀ will be measured at a representative exhaust location in each barn using real-time monitors. Total suspended particulate (TSP) and PM_{2.5} will be measured gravimetrically. Sampling of all of these parameters will be conducted for 24 months, with data logged every 60 seconds. Data will be retrieved with network-connected PCs, formatted, validated, and delivered to EPA for subsequent calculations of emission factors.

Micrometeorological techniques will be used to estimate emissions of NH₃ and H₂S from a swine manure basin, five swine lagoons, an egg layer lagoon, and three dairy lagoons. This approach will use optical remote sensing, both downwind and upwind of the storage, coupled with 3D and 2D wind velocity measurements. The concentrations of NH₃ will be determined using scanning tunable diode laser absorption spectroscopy. Measurements of H₂S and NH₃ will be conducted using UV differential optical absorption spectroscopy. A monitoring team will conduct an 11-day test at each farm each quarter for two years.

The benchmark NAEMS data, and accompanying analysis and interpretation, will allow EPA and producers to reasonably determine which farms — based on type, size, and/or geographic location — are subject to federal regulations. The database will aid environmental consultants and air dispersion modelers to assess and reduce the impact these sources have on neighbors and the environment. Policymakers, regulators, environmental groups, and producers will have increased understanding of livestock air pollution.

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Agricultural Air Quality in Europe and the Future Perspectives

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Abstract

Agricultural emissions in Europe are important to several atmospheric transport related environmental issues. These include local and regional air quality problems, such as PM exposure, eutrophication and acidification, toxics and contribution to greenhouse gas emissions, resulting in a number of environmental impacts. Over Europe, agricultural emissions are variable in space and time, and the contribution to the different issues are variable. Policies have been developed to combat some these emissions with some success. However, future, national and European policies are necessary to successfully decrease emissions and its related problems. Currently there is a move towards more integrated policies. A clear example is the nitrogen issue. Nitrogen does not only contribute to air quality, but also to water quality and to the greenhouse effect in a cascade. The efficiency of nitrogen use, especially in animal agriculture, can be increased by best management practices and/or technological innovation, resulting in much lower losses to the environment.

Introduction

European agriculture is extremely diverse, ranging from large, highly intensive and specialized commercial holdings to subsistence farming using mainly traditional practices. Consequently, impacts on the environment vary in scale and intensity and may be positive or negative. There is a legacy of significant environmental damage associated with agriculture in Central and Eastern Europe, the Caucasus and Central Asia, often associated with unique ecosystems, where exploitation of resources (such as freshwater for irrigation) was excessive. The dramatic decline in resource use in these countries, largely due to economic restructuring rather than policy, consumer or technological developments, has scaled back many environmental pressures.

A common policy objective throughout Europe for several decades was to increase food production. Farmers increased agricultural output significantly between the 1940s and the 1990s in response to such policies. Supported by public investment, this resulted in mechanization combined with the abandonment of traditional practices, reliance on non-renewable inputs such as inorganic fertilizers and pesticides, the cultivation of marginal land and improvements in production efficiency. In Western Europe (WE), the common agricultural policy (CAP) and several national policies encouraged intensification. This took various forms, including the sustained use of chemical inputs, increasing field size and higher stocking densities. Intensified farm management led to discontinuation of traditional fallowing practices and crop rotations resulting in a displacement of leguminous fodder crops with increased use of silage and maize. Specialization and intensification have resulted in a decrease in the number of farm holdings and numbers employed, as well as a rationalization of production leading to less diversity of local agricultural habitats.

Agriculture is an important sector contributing to environmental effects and more specifically air quality related issues. Air quality contributes to human health through exposure of ammonia, toxic organic compounds, pesticides, and particulates. Air quality also contributes to climate change in the form of greenhouse gases and as cooling aerosols. After deposition, eutrophication and acidification might occur and, in combination with climate change biodiversity is endangered, and the net-greenhouse gas exchange is affected. There are two ways to assess the contribution of agriculture to air quality; that is the share of agricultural emissions to the total emissions in Europe, or through the contribution of agriculture to the observed effects in Europe. The latter is less uncertain and not followed here.

Contribution of agricultural emissions

In 2002 agriculture contributed 10.1% to the total greenhouse gas emissions in CO₂ equivalents in the EU15 (EEA, 2005). The greenhouse gases emitted by agriculture are nitrous oxide and methane, both of

which have a far greater global warming potential than carbon dioxide. Agriculture also consumes fossil fuels for farm operations, thus emitting carbon dioxide. CO₂ comprises only a small part of these emissions (0.05%) and N₂O and CH₄ contributed equally 4.9%. For N₂O the main source is fertilizer use and for CH₄ enteric fermentation of mainly cattle. The Kyoto target for the EU is 8% reduction in 2008-2012 relative to 1990. Emissions of greenhouse gases by the agriculture sector — methane and nitrous oxide — fell by 8.7 % between 1990 and 2002. This is due mainly to a 9.4% reduction in methane from reduced livestock numbers and an 8.2% reduction in nitrous oxide from decreased nitrogenous fertilizer use and changed farm management practices. These are the emissions from sources, but agriculture also produces indirect greenhouse gas emission, e.g., through nitrates, which are leached and run-off to surface waters and, eventually to the sea. During the transport N₂O can be emitted. The same holds for the deposition of nitrogen to nature areas, which to some extent can be de-nitrified, leading to N₂O emissions. Additional CO₂ can be sequestered as the result of nitrogen deposition and due to agricultural practices (e.g., de Vries et al., 2006). The net-emissions of greenhouse gases are, however, not yet quantified with some accuracy.

The contribution of agriculture to the total acidifying emissions in Western Europe is 31%, and 13% in Eastern Europe, for eutrophication the contribution is 24% and 20% for Western Europe and Eastern Europe, respectively. Figure 1 shows the total and agricultural emission of different components in EU15 as estimated by EMEP. By far the most important component is ammonia, which for more than 90% is determined by agricultural emissions. Within the EU-15, emissions of ammonia to the atmosphere from agriculture decreased by 9% between 1990 and 2002. The majority of this reduction is likely to derive from a reduction of livestock numbers across Europe (especially cattle), and the lower use of nitrogenous fertilizers across the EU-15 (Erisman et al., 2003; EEA, 2005).

The contribution of agriculture to the PM_{2.5} emissions is about 5%, and 25% for the PM₁₀ emissions. Current investigations show that PM emissions from agriculture in intensive emission areas might contribute more than currently estimated. The gap between modeled and measured PM concentrations might for a large part be explained by an underestimate of the agricultural sources.

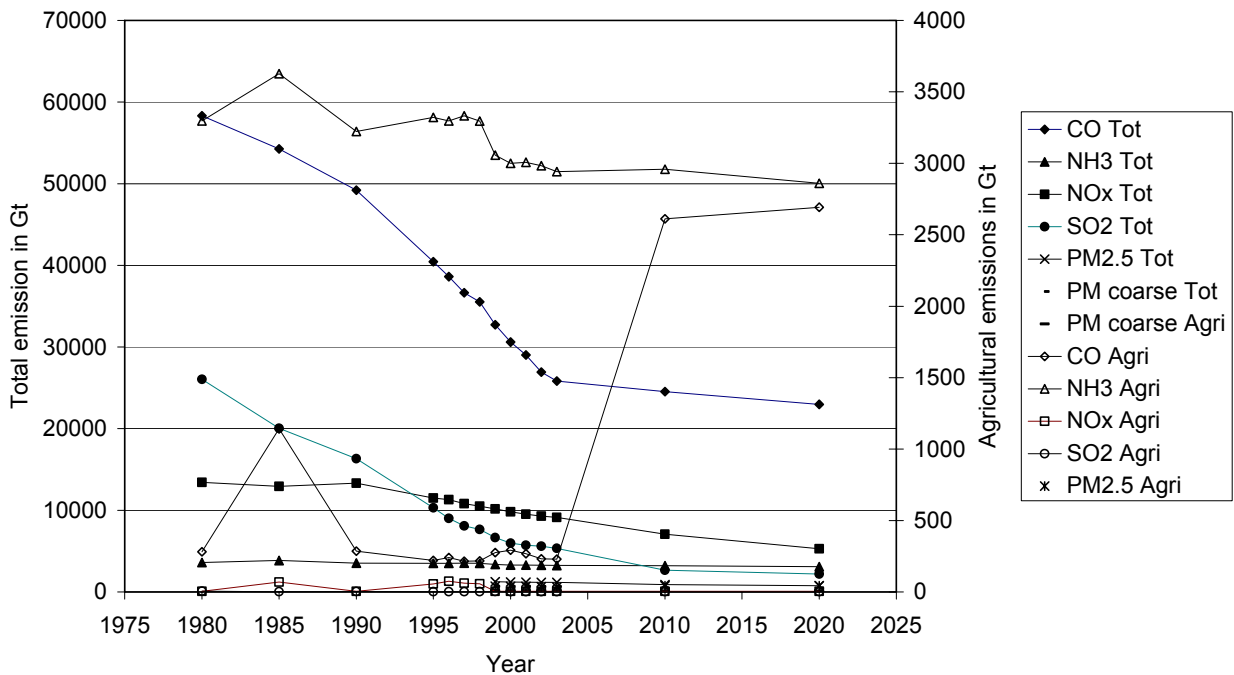


Figure 1. Trend in best estimate emissions from all sources and from agriculture in EU15 (Gt), source: <http://webdab.emep.int/scaled.html>

All EU-15 Member States have action plans for climate change and air quality. Most plans and programs under the National Emission Ceilings (NEC) Directive include measures to reduce ammonia emissions from agriculture due to their negative health and environmental effects. According to current projections

(which exclude potential effects of the 2003 CAP reform) many member states are likely to miss their 2010 ammonia reduction target under the NEC directive. The predictions are shown in Figure 1. The agriculture sector can make a positive contribution to reducing greenhouse gases through the production of bio-energy, thus substituting for fossil fuels. Agriculture at present contributes 3.6% of total renewable energy produced and 0.3% of total primary energy produced in the EU (EEA, 2005).

The common agricultural policy has been one of the important drivers of farm intensification and specialization in the EU. Market pressures and technological development have also contributed to these trends, which are very strong in some sectors that benefit from little public support (e.g., pigs, poultry, potatoes). Intensive farming has had significant impacts on the environment. Public concerns related to production methods and some reorientation of the common agricultural policy has created new opportunities, for example through labeling and agri-environment schemes, for farmers to reduce pressures on the environment. Agriculture in the Central and Eastern Europe countries is likely to intensify when they have full access to the common agricultural policy, although there is an evolving agri-environmental policy framework and some opportunities under the special accession program for agriculture and rural development to address this risk. The common agricultural policy will apply to new member states in a modified form, which may reduce incentives for increasing production.

Future Focus

Erismann et al. (2005) made an overview of the effectiveness of policies in the Netherlands and Europe to reduce nitrogen emissions. The main conclusion is that policies should not be focused on increasing production alone rather than including the farm nutrient efficiencies. Through the focus on production, combined with the low cost of e.g., concentrates and fertilizers, the efficiency of nutrient cycling at the farms has been neglected. Measures include taxes or financial grants, and the targets setting for N losses. Furthermore, the closing of cycles should be done at different scales at the same time. We are accustomed to seek our improvements in technological options. There are potential technologies that might lead to substantial emission reduction (catalytic converters, hydrogen economy, nitrification inhibitors, fermentation of manure, etc.). These are important, but the reduction in environmental load they cause should not lead to increased import of raw materials, leading to changes in cycles at supranational scales. Furthermore, other components (such as carbon) and issues (access to freshwater) should be taken into account to prevent trade-offs. If financial incentives are given, it is important to secure the period that these incentives will last, in order to guarantee a return of investment.

The largest uncertainty in emission estimates is due to the diffuse sources and the generalisation of the different sources, varying in space and time. New methods are developed to measure accurately and with enough temporal resolution the ambient concentrations. This is necessary to improve the quantification of individual sources and the regional sources. The landscape scale is so far not well quantified. Major uncertainties rest in the local and regional ammonia emissions and its temporal variation and in the PM emissions from agricultural sources.

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Air Quality and Agriculture: The Role of Global Drylands and Arid Zones

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Abstract

The available information on trace gas exchange of CH₄, CO, N₂O, NO_x, and VOCs arising from agriculture and natural sources in the world's drylands and arid zones due to soil and plant processes is reviewed.

Together, the semi-arid and semi-humid zones, called the drylands, cover about 30% of the global land area, and a further 17% of the global land area are in the hyper-arid and arid zones. More than a third of the world's human population live in these arid and dryland regions. The vegetation and soils of these arid and dryland regions are fragile and are prone to desertification, particularly through the loss of soil nutrients, soil organic matter and the invasion of woody shrubs.

The contribution of these drylands and arid zones to the global cycles and budgets of trace gases, and the changes to these contributions effected by land use modifications in these regions, have not been extensively researched. Between 10 and 40% of the global soil-atmosphere exchange for these gases (CH₄, CO, N₂O, NO_x, and VOCs) may occur in these zones, but for most of these gases there are less than a dozen studies to support the individual estimates. Significant differences are identified in the biophysical and chemical processes (production and diffusion) controlling these trace gas exchanges between dryland and arid zones and other land regions, and are not well represented in the current global models of trace gas emissions. Therefore, we have a poorly quantified understanding of the contribution of these regions to the global trace gas cycles and atmospheric chemistry. More importantly, we have a poor understanding of the feedbacks between these exchanges and global change and regional land use and air pollution issues. A challenge exists to acquire more data on trace gas exchange from these zones and for models to be extended to incorporate those processes that are special for trace gas exchange to the dryland and arid zones.

Introduction

Human changes to the Earth's biosphere have profoundly changed the concentrations of climatically-active and other trace gases in the Earth's atmosphere. The atmospheric effects of deforestation and the expansion of ruminant animal numbers, as well as some other changes to the biosphere, are well quantified. In other cases, such as the loss of vegetation and soil organic matter from unsustainable use of drylands, the atmospheric consequences are poorly understood. The purpose of this paper is to review and analyse the contribution of agriculture and natural processes in the world's drylands and arid zones to global and local air pollution problems via soil-atmosphere exchange.

There are two ways of classifying arid, semi-arid and dryland areas. One is based solely on precepitation (P), and the other on the ratio of precipitation to potential evapotranspiration (PET). Under the first approach, arid and semi-arid regions have an annual precipitation of less than 300-400 mm. Under the second approach, the ratio of P/PET which is called the "degree of aridity" (UNESCO 1977) is used. It should be noted that this is a climatological land classification. Within any of these zones, the full range of land cover types from barren to forest can potentially occur, although, naturally, shrub land and sparsely vegetated land cover prevail in the arid zone. Similarly, there is a separate range of land uses.

The definitions of these areas, the areas covered as a fraction of the global land area and the populations of these areas are summarised in Table 1. Together, the semi-arid and semi-humid zones, called the drylands, cover about 3.8×10^9 ha globally, or about 30% of the global land area (Leemans and Kleidon 2002), and a further 17% of the global land area is in the hyper-arid and arid zones, subsequently described as the arid zones. More than a third of the world's human population live in these arid and dryland regions.

The vegetation and soils of these arid and dryland regions are fragile and are prone to desertification, particularly through the loss of soil nutrients, soil organic matter and the invasion of woody shrubs (Leemans and Kleidon 2002). The contribution of these dryland and arid zones to the global cycles and budgets of trace gases and the changes effected by land use modifications in these regions have not been extensively researched and are poorly quantified and not adequately represented in models of these exchanges.

Table 1. Areas and populations of the world's arid and drylands (UNDP/UNSO 1997)

Zone	Hyper-arid	Arid	Semi-arid	Dry sub-humid
P/PET	≤0.03	0.03 – 0.2	0.2 – 0.5	0.5 – 0.75
Area 10 ⁶ km	9.4	15.9	23.7	13.9
% of global land area	7	12	18	10
% of global population	-	4	16	18

Trace Gas Exchange in the Drylands and Arid Zones

A summary of observations and models of trace gas exchange from dryland and arid land systems is reviewed and discussed, each gas separately. Data from an unpublished year-long field study of mature Mallee vegetation and a nearby wheat crop in the semi-arid zone in Australia, (34.5°S, 141.5°E), conducted by the authors (Galbally et al. 2006) is included in the review. In preparing this review, the authors have attempted to identify all the relevant literature, although no doubt some papers have been overlooked. When considering the semi-arid zone, studies from sites with precipitation less than 400 mm per annum are included. Studies with less than 2 weeks field data have been omitted. Where data from dryland and semi-arid zone studies have been used in summaries in the review and not explicitly acknowledged in the text, the reference is included and followed by an asterisk.

Methane

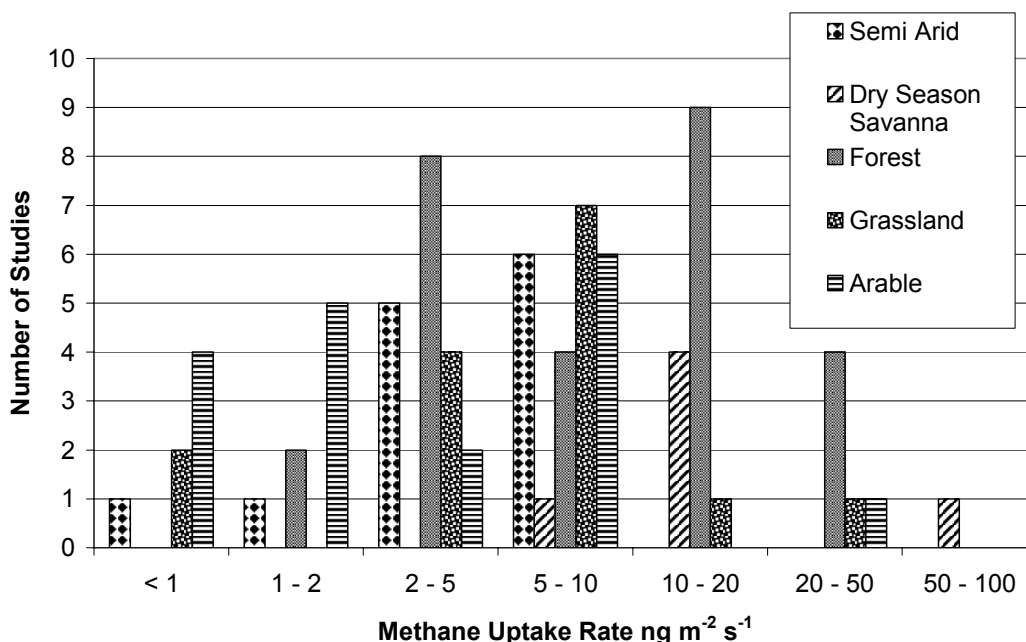
Methane (CH₄) is a potent greenhouse gas, and its abundance in the atmosphere makes it a significant component of the global carbon cycle. Globally, soils are a significant sink for methane and account for a global annual uptake of 30 Tg y⁻¹ (IPCC 2001). The factors affecting methane exchange in desert, semi-arid and seasonally dry lands are somewhat different from those controlling methane exchange in temperate systems, including the complicating factor of the significant activity of soil invertebrates, particularly termites, which produce methane as a byproduct of cellulose digestion. Under very dry conditions, with soil moisture below 5% water filled pore space (WFPS), methane uptake is near zero (Striegl et al. 1992, Otter and Scholes 2000). At moderate soil moisture levels, both uptake and emission of methane occur (Poth et al. 1995, Anderson and Poth 1998, Otter and Scholes 2000, Verchot et al. 2000). The switch between the methane uptake and emission can be associated with the burning of above-ground vegetation (Poth et al. 1995, Anderson and Poth 1998). The explanation of this phenomenon is that there is extensive methane production by foraging subterranean termites and that these emissions counterbalance the soil methane uptake. Burning of the above-ground biomass removes the food source for these termites, reduces emissions and tips the balance in favour of methane uptake. Other processes, such as cultivation or harvesting or overgrazing the above-ground vegetation, will presumably have the same effect in reducing the termite component of methane emissions.

In the semi-arid shortgrass steppe of southwestern United States, cultivation and nitrogen fertilization reduced methane uptake (Mosier et al. 1991, Mosier et al. 1996). In the swamps and wetlands of seasonally wet savannas, the transition to higher WFPS and higher root or microbial respiration can promote anaerobic conditions in the soil and increase methane emissions from fermentation (Otter and Scholes 2000, Verchot et al. 2000).

We have gathered the methane uptake rates from the five available extended studies in semi-arid regions, and find an average methane uptake of $5.6 \pm 2.9 \text{ ng (C) m}^{-2} \text{ s}^{-1}$, (range 1-10). Similarly we have gathered the methane uptake rates for the eight available extended studies from the dry season in seasonally wet/dry savanna sites. During the dry season, when only 5-15% of the annual precipitation occurs, the average methane uptake rates are $12 \pm 20 \text{ ng (C) m}^{-2} \text{ s}^{-1}$, (range -10 to 60) where the negative uptake is an emission. In the wet season for seasonal wet/dry savanna, methane emission frequently dominates methane exchange. These uptake rates for the dry season in savanna are higher than those of the semi-arid region, perhaps because the wet/dry seasonal cycle keeps the soil more moist, and the methanotrophic bacteria more active.

We present the methane uptake rates for dryland and arid systems in Figure 1, along with existing estimates for other regions (Smith et al. 2000; Boeckx and van Cleemput 2001). It is clear from Figure 1 that the methane uptake rates in semi-arid regions are small compared to those in other systems throughout the world, but because of the large area of the arid and dryland zones, see Table 1, as much as 40% of the soil sink for methane may occur in drylands and arid zones where little data exists.

Figure 1. The rates of methane uptake in different systems Units $\text{ng (C) m}^{-2} \text{ s}^{-1}$. (This study, Smith et al. 2000, Boeckx and van Cleemput 2001).



The paucity of data for this significant uptake of methane in drylands and arid zones has been commented on also by Potter et al. (1996) and Smith et al. (2000). The processes controlling the methane uptake in drylands and arid zones are soil moisture and soil microbial activity, subterranean termites, above-ground biomass and its loss through burning, drought, grazing and harvesting. It is a challenge for current models that incorporate methane exchange to include these processes. An even greater challenge is to understand how microbial community structure might change in response to other environmental changes and affect global soil-atmosphere trace gas exchange including methane (Schimel and Gulledege 1998). If climatic boundaries change (as is suggested by the IPCC (2001), then there will be a feedback through changes in the global methane uptake rate. More critical experimental studies and developments in techniques for modelling of these soil exchange processes will be necessary to quantify and model that feedback of methane uptake in arid and dryland zones.

Carbon Monoxide

Carbon monoxide (CO) plays important roles in the atmosphere through its reaction with hydroxyl radicals (Levy 1971) and its ozone-forming potential in NO_x-rich environments (Crutzen and Zimmermann 1991). Both of these processes lead to the oxidation of the carbon monoxide to carbon dioxide.

Carbon monoxide is produced in the soil through the thermal and photochemically induced oxidation of humic acids and phenolic materials that are present in the soil and by the decomposition of cellulose, particularly in dead above-ground biomass (Conrad and Seiler 1985a & b, Schade et al. 1999). The production of CO is positively correlated with the amount of dead above-ground biomass and the organic carbon levels in the soil, and its emission increases exponentially with temperature (rising markedly at temperatures above 35°C). Production of CO increases with sunlight, particularly at UV wavelengths, but is unaffected by soil moisture at low to moderate soil moisture contents (Conrad and Seiler 1985a & b, Tarr et al. 1995; Zepp et al. 1996; Schade et al. 1999; Gödde et al. 2000). For these reasons the global drylands and arid zones are potentially the major regions for soil emissions of CO.

Carbon monoxide in the soil can be utilized as a substrate by soil microbes, so that there may be CO uptake. A number of authors have demonstrated CO uptake in the field in drylands (Conrad and Seiler 1982; Conrad and Seiler 1985a & b; Scharffe et al. 1990, Sanhueza et al. 1994). This CO uptake does not vary substantially with temperature and does become more active in the transition from a dry to a moist soil.

These processes of CO production and uptake compete. Because the CO production increases with higher temperatures and CO uptake does not, there is an observed strong diurnal variation of CO exchange, with uptake at night and emissions during the daytime (Scharffe et al. 1990, Sanhueza et al. 1994).

Sanhueza et al. (1994) found that there was relatively little variation between the wet and dry seasons, although enhanced emissions were observed after heavy rains. In the Australian semi-arid study (Galbally et al. 2006), where there is a significant seasonal cycle of temperature and solar radiation, low daytime CO emissions were observed in winter and high daytime CO emissions were observed in summer.

Sanhueza et al. (1994) recorded an increase in emissions of up to 18 ng(C) m⁻² s⁻¹ from grassland from which the trees had been removed and which had been regularly mown. Interestingly, when part of the native grassland was cultivated and sown to arable crops, the site became a net sink for CO of -3.2 ng(C) m⁻² s⁻¹ during the first weeks following ploughing. Later in the season, it reverted to a net CO source (Sanhueza et al. 1994). The Southeastern Australian wheat field and the undisturbed Mallee measurements (Galbally et al. 2006) are amongst the highest CO emissions observed with yearly averaged daytime emissions of 11 ±18 ng(C) m⁻² s⁻¹, and these emissions behave according to the same temperature dependence observed by Scharffe et al. (1990). During the daytime, from these observations, the CO emissions are equivalent to 0.1% of the carbon lost from the soil by soil plus root respiration.

Globally, CO emission from soils and above-ground dead plant material is estimated to be 100 (50 – 170) Tg CO yr⁻¹ (Schade and Crutzen 1999) occurring particularly in the tropical regions including savannas, grasslands and rainforests. Soil uptake of CO is estimated to be 190 – 580 Tg CO yr⁻¹ (King 1999), although a much lower estimate is made by Potter et al. (1996). Land use change and climate change affect the amount of above-ground dead biomass, the solar radiation and air temperature, and thereby the CO emission. These processes are essential ingredients for understanding and modelling CO exchange and atmospheric chemistry in the lower atmosphere (Schade and Crutzen 1999), particularly for the lower atmosphere in the drylands and arid zones. Schade and Crutzen (1999) call for more measurements as well as modeling efforts to improve our understanding of CO exchange in tropical savanna, grassland and rainforest ecosystems.

Nitrous Oxide

Nitrous oxide (N₂O) is an important greenhouse gas and a key component of the global nitrogen cycle. In the soil, nitrous oxide is produced predominantly through the conversion of forms of nitrogen, such as ammonium and nitrate, by soil bacteria (Bremner 1997). Nitrous oxide (and nitric oxide) are gaseous intermediates of both processes. The microbial processes of nitrification and denitrification account for about 99% of the nitrous oxide generated in the soil ecosystem (Webster and Hopkins 1996). Because

nitrous oxide is an intermediate in these processes, its emission is not quantitatively linked to the masses of ammonium or nitrate substrates (either available or transformed).

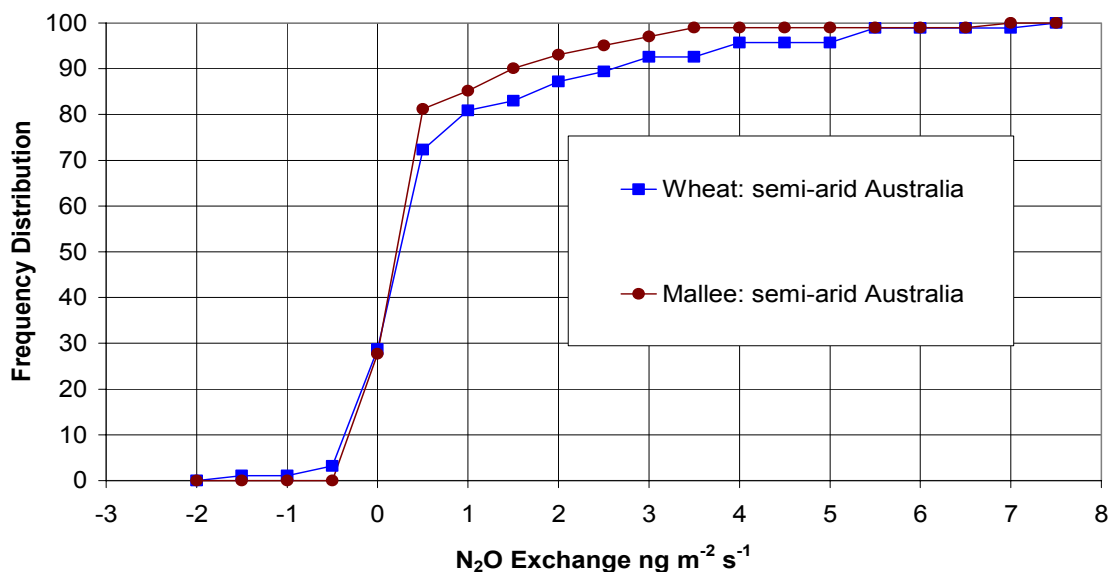
Both ammonium and nitrate are important plant nutrients and are commonly applied to the soil as fertiliser to encourage plant growth. However, these ions are also essential substrates for nitrifying and denitrifying bacteria. It is not surprising, therefore, to find that much of the literature related to nitrous oxide emissions is involved with the relationship between fertiliser application and nitrous oxide (or nitric oxide) emission (Matson et al. 1998). The rates of microbial activity, and hence both nitrification and denitrification, increase with increasing temperatures. Although temperature optima vary with bacterial species, the optimum temperature range for nitrification is 25°C to 35°C, with a somewhat higher temperature optimum for denitrification (Granli and Bøchman 1995).

In hot dry climates, the addition of fertiliser has little impact on the concentration of soluble nitrogen or on emission of nitrous oxide (Matson et al. 1998). However, when the soil is moistened (either through rainfall or irrigation) ammonium levels increase dramatically, and then decrease to nearly zero as the proliferation of nitrifying bacteria under the moist conditions convert the ammonium to nitrate and releases nitric and nitrous oxides (Mummey et al. 1994, Matson et al. 1998, Wulf et al. 1999). The production of nitrous oxide from nitrification upon wetting of dry soil happens with a rapidity and intensity that is not characteristic of temperate agricultural systems. (A similar phenomena is observed with CO₂ respiration in hot dry soils by Xu et al. (2004)) Also, in spite of semi-arid soils being substantially aerobic, there is a persistent pool of denitrifying enzymes that are able to survive the drought and become activated rapidly after the soil becomes wet, provided there are sufficient carbon and nitrogen substrates available in the soil (Peterjohn 1991).

A study of Mexican wheat fields, by Panek et al. (2000), indicated that, following irrigation, denitrification accounted for most of production of nitrous oxide, but as the soil dried out, nitrification became predominant. Over the growing season, the nitrous oxide flux was influenced almost equally by the two processes. In Australian grazed pasture, emission from nitrification accounted for more than 50% of annual total emissions (Wang et al. 1997). In Figure 2, the cumulative frequency distributions of N₂O emissions from the Australian study (Galbally et al. 2006) are presented. As can be seen from the distribution of individual measurements in the Australian study, the highest 5% of the observations (those greater than the 95 percentile) are around 10 times or more than the median emissions. These high emissions correspond to the few rain events but over the year contribute around half of the total emissions.

There are nine comparisons of nitrous oxide emissions from farmed versus natural vegetation in semi-arid regions, and these indicate a net extra contribution from cultivation to N₂O emissions of 0.1 kg N ha⁻¹ y⁻¹ (Parton et al. 1988, Mosier et al. 1991, Mosier et al. 1996, Corre et al. 1996; Mosier et al. 1997; Guilbault and Mattias 1998, Huang et al. 2003, Galbally et al. 2006). Due to the large areas involved, this may represent as much as 10% of the global soil emissions of N₂O from agricultural disturbance may occur in drylands and arid zones where little data exist. There are more than 800 studies of nitrous oxide emissions from agricultural systems (Bouwman et al. 2002) and only nine of these are from the extensive semi-arid zone. (The global emissions are drawn from IPCC (2001). The processes controlling the N₂O emissions in drylands and arid zones that occur on re-wetting of dry soil as well as denitrifying enzymes that are able to survive the drought and become activated rapidly after the soil becomes wet are not characteristic of the temperate zone. Current models of N₂O emissions from soils (Wang et al. 1997, Frolking et al. 1998, Parton et al. 2001) do not have these processes explicitly included. A consequence is that, if climatic boundaries change (as is suggested by the IPCC (2001), then there will be a feedback through changes in the global N₂O emissions; but we cannot quantify or model that feedback because of the lack of understanding of N₂O emission processes in arid and dryland zones.

Figure 2. The cumulative frequency distributions of N₂O emissions from the Australian study (Galbally et al. 2006).



Nitric Oxide and Nitrogen Dioxide

Nitric oxide (NO) and nitrogen dioxide (NO₂) are of importance due to their roles in the nitrogen cycle and in the acidification and ozone levels of the atmosphere (Crutzen 1979). The chief sources of nitric oxide in soils results from the action of denitrifying and nitrifying bacteria and the chemical decomposition of NO₂⁻ under acidic conditions (Firestone and Davidson 1989), although on a global scale, the latter process is of only minor significance (Galbally 1989). While there are many observations of NO emissions from tropical savanna, the database from semi-arid regions is sparse. At the time of the inventory of global emissions studies by Davidson and Kingerlee (1997), there were no published and one unpublished study from desert regions. Subsequently there have been a number of studies in semi-arid zones (Le Roux et al. 1995, Smart et al. 1999, Hartley and Schlesinger 2000, Barger et al. 2005, Galbally et al. 2006).

In dryland and semi-arid systems, where soils are well aerated, nitrification is expected to be much more important for the production of nitric oxide than denitrification. Soil moisture is a critical factor in determining the rate of nitric oxide emission. Following the wetting of dry soils, nitric oxide emissions are rapidly increased by about one order of magnitude, and then decays to original levels over the next few days (Johansson et al. 1988, Rondón et al. 1993, Levine et al. 1996, Scholes et al. 1997, Meixner et al. 1997, Smart et al. 1999, Hartley and Schlesinger 2000, Pinto et al. 2002), probably due to increases in microbial activity and in the rate of diffusion of NO in the moist aerated soils (Davidson et al. 1993, Cárdenas et al. 1993). However, as soil moisture approaches field capacity, NO emissions from nitrification decrease due to the restriction in oxygen diffusion (Bollman and Conrad 1998). The temperature dependence of NO emissions is quite weak in semi-arid and arid regions, where soils are drier (Stocker et al. 1993) or where temperatures may be above the optimal temperature for microbial activity (Cárdenas et al. 1993). Ludwig et al. (2001) suggests that soil temperature is best considered as “a factor that mainly modulates short-term variation of the NO exchange, whereas the magnitude of NO emissions is predominantly controlled by other factors.” The presence of termites in the soil leads to higher than normal emissions of NO, possibly due to enhanced NO₃⁻ concentrations in the termite mounds (Rondón et al. 1993, Le Roux et al. 1995).

The four longer term studies from the semi-arid zone indicate average NO emissions in the range 0.1 to 1.8 kg N ha⁻¹ y⁻¹ with the higher value coming from the Sahel ((Le Roux et al. 1995, Smart et al. 1999, Hartley and Schlesinger 2000, Barger et al. 2005). These four studies indicate that the semi-arid lands (as well as the tropical savannas) are a major contributor to soil NO emissions. This has been borne out by the satellite

sensing and inverse modeling study of Jaeglé et al. (2004) who observed strong pulses of soil NO emissions lasting 1-3 weeks after the onset of rain in the semi-arid sub-Saharan savanna. Jaeglé et al. (2004) extrapolated this finding to all of the tropics and estimated a 7.3 Tg N yr⁻¹ biogenic soil NO source, perhaps 20% of the global NO_x emissions. So far neither this NO pulse following re-wetting of dry hot soils nor termite activities are included in process based models of NO emissions. Recently a statistical model of NO emissions (Yan et al. 2005) has developed an empirical methodology to include this NO pulse activity. There is a need for more surface based critical studies of NO emissions in the semi-arid zone to confirm the satellite observations of Jaeglé et al. (2004), and to provide the experimental basis so that process models may be extended to include this re-wetting pulse phenomena that significantly affects the soil atmosphere exchange of NO, N₂O and CO₂.

Volatile Organic Compounds

The volatile organic compounds emitted from plants (BVOCs) play a significant role in atmospheric processes including ozone production and the formation of secondary aerosols (Monson and Holland 2001, Atkinson and Arey 2003). As a consequence of their emission, BVOCs influence the global carbon cycle, the Earth's radiation balance and the distribution of reactive gases such as carbon monoxide and organic acids (Guenther et al. 1999).

Numerous BVOC are emitted from vegetation, but on a global scale, isoprene and monoterpenes are the most commonly investigated because of their high relative abundance and their high chemical reactivity, and trees are the major source. Guenther et al. (1995) estimated a global BVOC annual budget of 1150 Tg(C) y⁻¹, an amount comparable to the global emission of methane. Of this budget, isoprene and monoterpenes contributed 44% and 11%, respectively. Grass species are, in general, low emitters of isoprene and monoterpenes (Hewitt and Street 1992), and instead emit mainly low molecular weight oxygenated compounds such as methanol, acetone, and acetaldehyde (Kirstine et al. 1998).

The emission rates of BVOCs vary markedly with plant species and also with leaf age and past environmental conditions, making the ecosystem emissions of BVOCs quite variable (Guenther et al. 1999). Since the emission of BVOCs increases with temperature (and in some cases with solar radiation), semi-arid biomes in tropical or temperate regions have a high potential for BVOC production (Guenther et al. 1996).

Tropical savannas are semi-arid mixed tree-grass ecosystems that cover about one-half of the areas of Africa, Australia, and South America and cover one-eighth of the Earth's total land area (Scholes and Archer 1997). For savannas (as for other grassland ecosystems) most of the isoprene and monoterpene emissions result from trees and shrubs that occur within these landscapes. For central African savannas, Otter et al. (2003) determined summer time isoprene emissions of 5.1 mg(C) m⁻² h⁻¹ for wooded savannas and 1.7 mg(C) m⁻² h⁻¹ for grass savannas. Winter emissions were about a factor of four lower. Summer time emissions of monoterpenes in wooded savannas were generally in the range of 0.1-0.3 mg(C) m⁻² h⁻¹, although emission rates from mopane woodlands (dominated by *Colophospermum mopane*) were much higher, at approximately 2.0 mg(C) m⁻² h⁻¹ (Otter et al. 2003). These measurements are consistent with those of other investigations of BVOC emissions from African savannas (Guenther et al. 1996, Klinger et al. 1998, Guenther et al. 1999, Otter et al. 2002). In a study of VOC emissions from savannas in southern Africa, Otter et al. (2002) calculated annual emission rates of 1.9 to 9.3 g(C) m⁻² y⁻¹ for isoprene and 0.7 to 1.7 g(C) m⁻² y⁻¹ for monoterpenes. If these estimates are taken to be representative of the drylands of the world, which have a global area of 28 x 10⁶ km², the total production of isoprene from savanna ecosystems is 130 ± 90 Tg (C) y⁻¹, and the total production of monoterpenes is 28 ± 11 Tg (C) y⁻¹.

Grasslands are undoubtedly a significant global source of methanol (Galbally and Kirstine 2002) and acetone (Jacob et al. 2002). A study of VOC emissions from grass pasture in Australia by Kirstine et al. (1998) found that about 0.25% of the carbon fixed by NPP in the pasture was lost as BVOCs. Given a global annual NPP for perennial grasslands and savannas of 42.5 Pg y⁻¹ (Field et al. 1998), this conversion factor predicts a release of over 100 Tg(C) y⁻¹ of BVOCs, of which about 15 Tg y⁻¹ would be methanol. Grasslands are also a significant potential source of a group of C₆ aldehydes and alcohols — so-called hexenyl compounds — that are produced by grass cutting or grazing (Kirstine et al. 1998, de Gouw et al. 1999). Unlike other BVOCs, these hexenyl compounds are emitted only when the plant leaf is wounded, and their emission rate declines rapidly to zero within a few hours of the wounding. Nonetheless, these

compounds are of comparable reactivity to isoprene (Arey et al. 1991) and hence have an equal potential to contribute to tropospheric photochemistry (Kirstine and Galbally 2004). While harvesting of hay crops has been shown to contribute a very minor fraction of the total global emissions of BVOCs (Karl et al. 2001, Warneke et al. 2002), the potential emission of hexenyl compounds through grazing or the trampling of animals has, to date, not been estimated.

The above estimates indicate that 25% of the global BVOC emissions may come from the world's drylands, based on a few studies. The mechanisms of BVOC emissions vary markedly with plant species and also with leaf age and past environmental conditions, and these are not the same in the drylands as in temperate regions. Our ability to understand and model the role of volatile organic compounds emitted from plants (BVOCs) in atmospheric processes including ozone production and the formation of secondary aerosols and the consequences for the global carbon cycle, the Earth's radiation balance and the distribution of reactive gases is limited by this paucity of understanding.

Conclusions

The gases considered in this review, CH₄, CO, N₂O, NO_x, and VOCs arising from agriculture and natural sources due to soil and plant processes in the world's drylands and arid zones and their products through atmospheric chemical transformations, are important contributors to (a) the climate and radiation balance of the atmosphere (through gaseous infrared absorbance and scattering of solar radiation), (b) the chemistry and oxidising power of the troposphere through photochemistry and ozone production, and (c) to regional air quality issues through ozone and aerosol. The exchanges of these gases in the drylands and arid zones have been shown to make up 10 to 40% of the global exchanges of these gases from these land based sources and sink, yet these estimates are based in each case on a handful of studies. More importantly, we have a poor understanding, quantification and modelling capability for some of the processes that affect these soil-atmosphere trace gas exchanges including:

- The interaction of above-ground biomass, soil invertebrate activity and soil microbial activity;
- The change of soil microbial community structure and its effect of trace gas exchange in response to other environmental change;
- The processes of decay of dead above ground biomass under high solar radiation and temperature conditions and the consequent release of trace gases including CO and VOCs;
- The soil microbial processes associated with the pulse phenomena, when on re-wetting a dry hot soil there are significant increases in the soil-atmosphere exchange of NO, N₂O and CO₂;
- The BVOC exchange of dryland and arid zone vegetation.

Furthermore there are larger scale couplings not addressed in this review, including the effects of rainfall and fire, drought and soil erosion on these dryland and arid zone exchanges that need further understanding.

New knowledge is necessary to understand how human activities and climate change may affect these dryland and arid zone exchanges and to model the feedbacks between these exchanges and global change and regional land use and air pollution issues.

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Role of Agricultural Ammonia Emissions in Formation of Secondary Particulate Matter

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Introduction, Sampling and Study Area

In a broad sense, out of pollutants monitored under National Air Quality Monitoring Program in India, there are two major issues of air quality: (i) consistently high particulate matter (PM) levels and (ii) consistently rising levels of oxides of nitrogen (NO_x). The levels SO_2 in India have dropped considerably after the introduction of low sulfur diesel (less than 0.25% sulfur) in the year 2000 (CPCB 2001). PM has been widely studied in recent years due to its potential health impacts and need for its control (e.g. Schwartz et al 1996). To understand the health effects and formation/emission of PM, it is essential to know physical characteristics and chemical composition of particles. The main components of $\text{PM}_{2.5}$ (particle with aerodynamic diameter of less than 2.5 micron) are organic matter (30-60%), metals (<1%), nitrates and sulfates (25-35%), elemental carbon (5%) and rest others (USEPA, 1995).

As seen, the major contribution to $\text{PM}_{2.5}$ is from nitrates and sulfates which are result of hundreds of reactions that take place in the atmosphere (Seinfeld and Pandis, 1998). The primary precursors for formation of nitrate and sulfate are NH_3 , SO_2 and NO_x . In other words, levels of PM and NH_3 , SO_2 and NO_x are interlinked through atmospheric reactions to a large extent. It has been established that most of the reactions are inorganic in nature (for formation of sulfate and nitrate), complex, and competing with each other. These reactions mostly depend on solar insolation, temperature, humidity, and presence of other constituents in the atmosphere (Utsunomiya and Wakamatsu, 1996). These parameters are location specific and show seasonal and diurnal variations, and one needs to study/measure these parameters at local and regional level.

NO_x is mainly emitted in the atmosphere as NO, which is subsequently chemically transformed into NO_2 and then into gaseous (HNO_3) and/or particulate nitrate. NH_3 , emitted through agriculture and industrial activities, plays an important role in neutralizing the atmospheric acids. This neutralization occurs predominantly in aerosols, a compound system that includes ammonia, sulfuric acid, nitric acid, and water to form nitrates and sulfates.

The primary objective of the study was to understand the role of NO_x , NH_3 , SO_2 , HNO_3 , temperature and humidity in formation of particulate sulfate and nitrate. The study area was city of Kanpur (latitude $26^\circ 26'$ N and Longitude $88^\circ 22'$ E), India. Specifically, the study was designed to measure the atmospheric levels of NH_4^+ , Ca^{2+} , Mg^{2+} , Na^+ , K^+ , NO_3^- , SO_4^{2-} , Cl^- , NH_3 (gas), HNO_3 (gas), NO_2 and PM_{10} covering winter and summer seasons and day and night samplings to capture the diurnal variations at two locations.

Results and Discussion

Results show NO_3^- , SO_4^{2-} , NH_4^+ , K^+ levels are found to be significantly high in winter season compared to the summer season. In summer conditions (high temperatures and low relative humidity), particulate ammonium nitrate is volatile, and there will be less particulate nitrate. On the other hand, wintertime nitrate levels were statistically higher. The role of NH_3 , which is a precursor for formation of NH_4NO_3 , was very critical as nitrates under all conditions correlated with NH_4^+ . The results of sampling suggested that the possible source of NH_3 , at a relatively clean site (upwind of an urban area), could be from agricultural activities in rural areas in the upwind. There are three possible sources of NH_3 that can influence the atmospheric ammonia levels at a site, these include: (i) application of urea in the fields, (ii) decomposition of agricultural waste which convert organic ammonia to gaseous NH_3 , and (iii) decomposition of waste

from cattle population. In rural areas of India, cattle provide basic livelihood and are still employed to plough fields, as a result, India has one of the largest cattle populations in the world.

In summer time, NH_4^+ levels were significantly less (than winter levels) that suggests equilibrium shift towards NH_3 (gas) due to high temperature ($>38^\circ\text{C}$); as a result, the nitrate levels in summer will remain in gaseous form as HNO_3 . As regards diurnal variation in summer, the nighttime levels of NH_4^+ were higher than daytime levels. As the temperature drops in nighttime, there may be equilibrium shift from NH_3 to NH_4^+ in presence of higher humidity resulting in higher levels of SO_4^- and NO_3^- .

In winter, mainly NO_3^- , SO_4^{2-} , NH_4^+ ions show significant diurnal variations, with higher values in daytime. In the daytime, the production of gaseous precursor (HNO_3) is found to be higher due to photochemical oxidation of NO_2 by OH radical than night time levels, when aqueous chemistry appears to dominate.

In addition to role of agricultural ammonia, potassium ions were found significantly high in the winter month ($3.5 \pm 2 \mu\text{g}/\text{m}^3$) compared to summer ($1.8 \pm 0.9 \mu\text{g}/\text{m}^3$). This high level of potassium is attributed to agricultural biomass burning and guttation process, which are dominant in winter season in the agricultural areas.

This study also concluded that daytime nitrogen conversion ratio (total nitrate to NO_2 + total nitrate) was quite high at 45-52% in both the seasons. This suggests the current level of NO_x may not fully explain the problem of NO_x pollution and one must account for NO_3 conversion and examine the health implications of higher NO_3 (in form of HNO_3 and/or particulate nitrate) through a comprehensive analysis.

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Has Livestock Introduction Changed Nitrogen Openness in Southern South America?

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Abstract

In grasslands of Argentina and Uruguay a large fraction of N stored in the vegetation is consumed by domestic herbivores introduced by European settlements 500 years ago, affecting the N cycle. We evaluated the rate of N accumulation in soils after livestock removal, with soil data collected at 13 grazed/non-grazed paired sites across the region. Because exclosures were relatively small and surrounded by actively grazed pastures, N accumulation over time was used as a proxy of the minimum N input to the ungrazed areas. Soils of exclosures with deep soils (>30cm depth, n=9) showed, on average, an increase of 43 kg/ha.yr of total N (in 0-30cm depth) relative to the paired grazed sites. Exclosures in shallow soils (<30cm depth, n=4) did not accumulate N in soils, probably because of high N losses by leaching. Legumes were not present in exclosures or in the grazed pairs. No papers measured rates of non-symbiotic N fixation in the region, but for other regions, it has been estimated as 1 to 5kg/ha.yr. Assuming these proportions, our estimates of atmospheric N deposition in this region could be much higher than previously thought (~40 kg/ha.yr vs ~4 kg/ha.yr). Simulations using the CENTURY model reproduced the observed patterns and suggested that higher volatilization from urine and dung patches are accountable for higher N losses in grazed sites. Enriched soil $\delta^{15}\text{N}$ in grazed sites can result from high rates of N volatilization, due to the fact that the NH_3 products are heavily depleted in ^{15}N . However, in exclosures with deep soils, changes in soil $\delta^{15}\text{N}$ were small and not significant among grazed and ungrazed paired sites. In contrast, in exclosures with shallow soils not accumulating N, $\delta^{15}\text{N}$ values were significantly depleted in ungrazed (5.5‰) respect to grazed sites (6.6‰). Greater losses of N in the grazed matrix of these rangelands, which extend over more than 40 million hectares in the region, could be increasing the amount of reactive N in the atmosphere (mainly as NH_3), and increasing local N redeposition. Ongoing rain collections showed high NH_4 inputs in rain. As much as 4.02 mg/l were collected in one rain event in August-2005. Precarious extrapolation of these values to annual deposition gives an annual value of 48.2 Kg/ha.yr. Currently rain collectors are being installed to continue rain measurements. Our work suggests that domestic herbivores are opening the N cycle through increased N emissions and local redeposition of reactive nitrogen.



Agricultural Best Management Practices in Denmark

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Abstract

Indoor air quality and gaseous emissions from livestock facilities have been hot themes in Denmark during the last decade; much research is also in progress for the next decade. The driving forces are the EU directives, Danish rules, and the public concern about the human health and the environment, regarding dust, odour, and ammonia. With fulltime employment in pig and poultry houses, the dust burdens is of big concern for the workers, and comprehensive investigations are carried out in Denmark on methods to reduce the dust concentration in animal houses. The most efficient method today is spraying a small amount of rapeseed oil, controlled by animal activity. Nearby livestock facilities, the outdoor environment is especially disturbed by the odour emission from the facilities. It is well documented that the odour emission can be reduced very much by air scrubbers and bio-filters with good maintenance. The problems are the durability, maintenance requirements and energy consumption. Investigations are in progress in Denmark on how to reduce the odour emission by air scrubbers with low pressure losses. Ammonia emission is another hot theme because approval of new animal facilities and renewed facilities with increased production capacity is depending very much on the documented future ammonia emission, based on emission factors. There are big differences between the emission factors for different housing systems. For dairy cows in cubicle houses with slatted floor and underlying scrapers in the alleys, the emission is e.g. 0.06 kg NH₃-N per kg total N in manure, but 0.10 kg NH₃-N per kg total N for cubicles with solid floors. For layers in cages with manure belts, the emission is 0.10 kg NH₃-N per kg total N, which is very low compared to other types of layer houses. This illustrates that the housing system has a very big impact on the ammonia emission. For a certain system, the management has also a big impact. One of the efficient solutions in Denmark to reduce ammonia emission is to add 0.5% sulphuric acid to the slurry, which reduce the emission by 70%. In respect to reduce ammonia emission from slurry storages, the slurry tanks must in Denmark be covered with e.g. a tent or the slurry surface must be covered with a controlled compact floating layer.

Introduction

The animal production in Denmark has been undergoing drastic changes over the last five decades in respect to herd size, stocking density and housing systems (Pedersen, 2005). As a consequence, the indoor air quality and the emission from animal facilities are also developing fast over time. For instance five decades ago, the average number of growing-finishing pig per farm in Denmark was around 10, and today, 600. The space per growing-finishing pig went down from 1.6 m² in the 1950s to 0.7 m² in the 1990s and is today again slightly increased to 0.8 m², due to EU and Danish regulations. Until the 1950s, no slatted floors were used in Denmark. In the 1970s the use of slatted floors was growing very fast, and in the late 1990s, two thirds of all pens for growing-finishing pigs was with fully slatted floors without bedding. Due to EU and Danish regulations, fully slatted floor will be totally forbidden from 2013, and in the future slatted floor will only be allowed in a third of the pen surface for growing-finishing pigs. Together with another new EU rule, rooting material is obligatory for growing-finishing pigs, why it can be expected that there will be an increase in use of straw as bedding in many pig facilities. Also cooling by water spraying in pig houses in hot periods is today obligatory in Denmark. For cattle and poultry there have also been drastic changes over time. Altogether facilities for animal production are changing fast, and it may be expected that the present state-of-the-art will be out of date already in few years. As a consequence the indoor air quality and the emission will also change.

From a scientific point of view, very few scientists in the 1970s were concerned about the outdoor environment. It was mainly a question on how to improve the indoor climate in respect to improving efficiency of the animal production. The emission of e.g. ammonia emission via ventilation and evaporation of ammonia from storages was of little concern. In some way manure from animal production was

considered only as waste product, which could easily be replaced by fertilizers. In the 1980s the concern about the emission was growing and at the same time it was experienced that many pig farmers got problems with their lungs, due to dusty indoor climate. Today there is a very big awareness about the negative impact of ammonia on the environment. A review on ammonia emissions from animal production facilities is also shown by Pedersen et al. (2004), based on a CIGR ammonia conference held in Denmark in 2003. Another increasing problem of more cosmetic nature is the odour burdens for workers in animal facilities and especially for people living in the vicinity of animal facilities. Because odour is mix of many different chemical components in small concentrations, it is a real challenge to examine and get rid of the smell.

This paper deals with the three main air quality factors; dust, ammonia, and odour concentrations and emissions.

Dust

Dust Reduction Methods

In former time airborne dust was not recognized as a problem in Denmark, because the work in pig houses only covered a minor part of a working day. Today, the indoor climate is “improved” by insulation of floors, walls and roofs and better ventilation, but unfortunately the indoor air is much dustier, especially in pig and poultry houses, opposite the more moist cattle buildings. At the same time it is fulltime job to work in pig facilities. The dust problems in pig houses was experienced already in the 1960s, but the real awareness about the negative influence on lung function for stockmen was awakened later, when it was common with bigger herds and fulltime work in pig houses. Today all farmers and stockmen know that if they do not protect themselves, there is a big risk for reduced lung function after years of work in pig houses.

In the EU project PL 900703, it was shown (Takai et al. 1998) that the average dust concentration in cattle houses were 0.38 mg total dust and 0.07 mg respirable dust per m³ air. The concentration in pig houses averaged 2.19 mg total dust and 0.23 mg respirable dust per m³ of air, and in poultry houses 3.60 mg total dust and 0.45 mg respirable dust per m³ air. It shows that the dust concentrations in pig houses are around five times higher than in cattle houses, and that the dust concentration in poultry houses is even higher. The Danish hygienic limits for animal buildings are 3 mg total dust per m³. Because there is a big variation from one animal house to another, there are many pig and poultry houses where the dust concentration is above the threshold. These results should be considered in relation to the relative humidity, which in cattle buildings in Northern Europe is typical 80% but only about 65% in pig and poultry buildings with more dry feed (Seedorf et al. 1998). The dust burden is in fact a real problem for pig and poultry farmers, because long time exposure can reduce the lung function for humans. The lifetime for farm animals is only up to a few years, why long time exposure will normally not harm the animals, but because gases as e.g. ammonia are bound to the dust, it indirectly can harm the animals.

There are two main strategies to solve the dust problem, namely:

- To wear personal protection as e.g. dust mask or a helmet with fresh air supply
- To reduce the dust burdens in the animal house it self.

From the stockmen point of view the best solution is to reduce the dust burdens, because it is not comfortable to use mask or helmet. It must also be taken into account, that masks may only be used 3 hours per day, because it has a negative effect on breathing.

At RCB, investigation on methods to reduce the dust concentration in pig houses has been carried out since the early 1980s, including vacuum cleaning, Ionization, straw versus no straw, wet versus dry feeding, recirculation of filtrated air, electrostatic filter, fogging with water, fogging with rape seed oil and fat in diet. Also The National Committee for Pig Production has since the 1990s carried out research on dust in pig buildings. Different technical solutions have been carried out at RCB as official test or in research programs. Research at RCB showed in the early 1990s that the dust burden in animal houses is strongly related to the animal activity, why an animal activity sensing system based on PID (Passive Infrared Detector) was developed (Pedersen and Pedersen, 1995). As shown in Figure 1 the dust concentration in animal houses is strongly related to the animal activity.

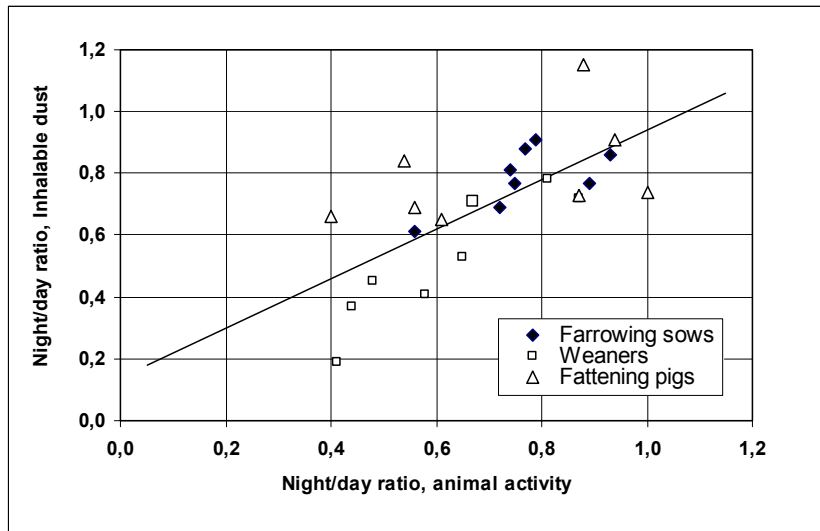


Figure 1. Correlation between animal activity and air borne dust in weaner house

Figure 2 shows another example (Pedersen and Takai, 1999) on how the diurnal variation in the dust concentration in a pig house is depending on the animal activity.

The measurements are carried out with a particle counter and shows good correlation between animal activity and the dust concentration.

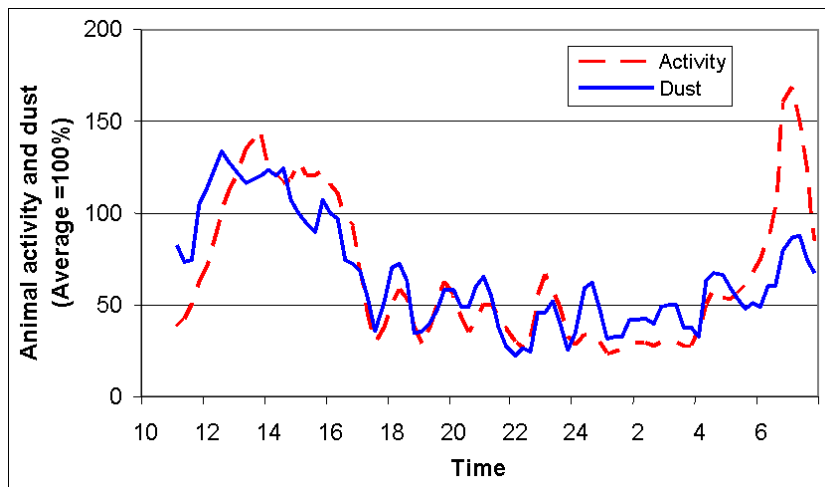


Figure 2. Dust in relation to animal activity in a weaner house

As mentioned above comprehensive work on how to reduce the dust burdens was initiated at RCB in the 1980s, and the possibility of abating the dust with spraying small amounts of oil in the animal house was tested. Many different techniques and types of oil were tested, and finally rape seed oil was selected, because it is an inexpensive vegetable oil. The results were promising already from the start. Figure 3 shows the high-pressure spraying of rape seed oil in growing-finishing house.



Figure 3. Example of spraying of rape seed oil

To improve the system further, an experiment was carried out with rape seed spraying, controlled by animal activity.

Figure 4 summarizes the different methods tested at RCB, and it shows that reductions of up to 88% are obtained by spraying rape seed oil, controlled by animal activity, in such a way that spraying is only done when the animal activity is above the threshold. (Takai and Pedersen, 2000)

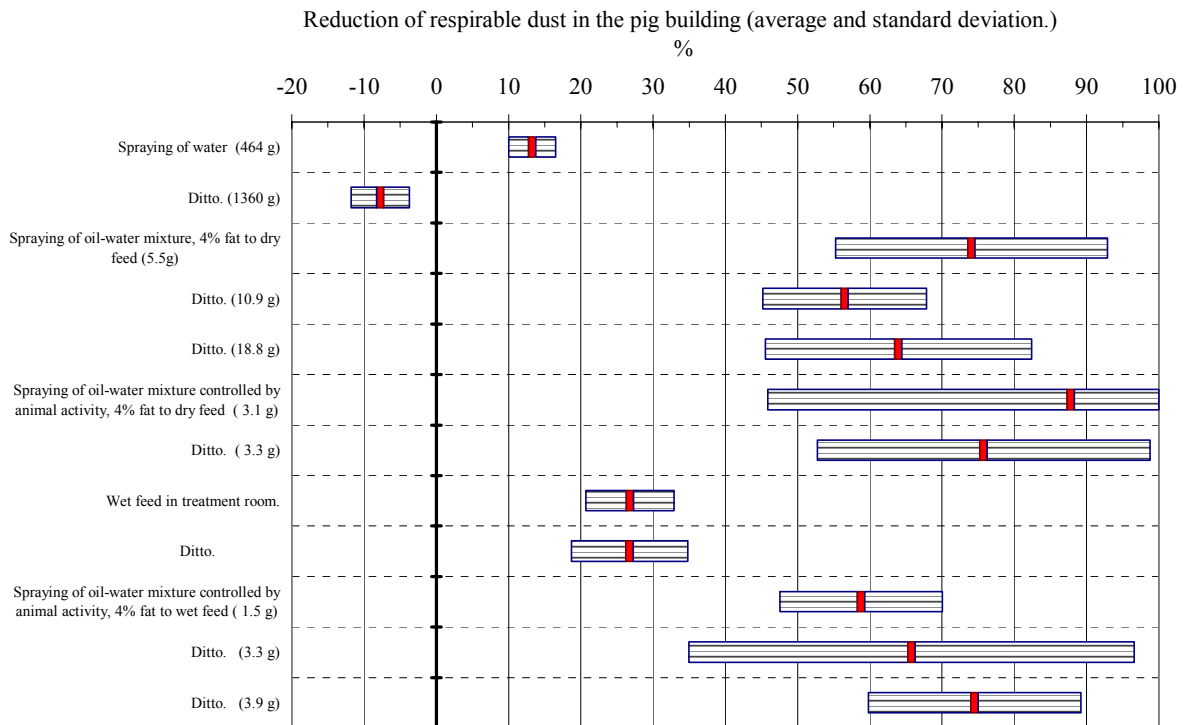


Figure 4. Reduction of respirable dust in pig buildings, from FCB tests (Rape-seed/water in g per day per pig)

Ammonia

Over the last four decades, ammonia measurements have often been a part of air quality investigations in animal houses in Denmark, because it has been one of the important parameters about indoor air quality in respect to animal health. In the 1970s with pig pens with partly slatted floor, bedding and around 1 m² pen area per pig, the average ammonia concentration was 5 ppm ammonia. In the 1990s Denmark was part of a comprehensive EU measuring program in Germany, the Netherlands, England, and Denmark, carried out in enclosed cattle, pigs, and poultry houses. In Table 1 shows the concentrations of ammonia measured in Danish houses in average of 10 houses in each group (Groot Kooerkamp et al. 1998)

Table 1. Ammonia concentrations and emissions measured in Denmark in the 1990s

	Ammonia concentration (ppm)		Ammonia emission (mg/h)	
	Average	Maximum	Per animal	Per HPU**)
Dairy cows, litter	2.7	18.6	560	485
Dairy cows cubicles	3.3	20.1	987	848
Beef cattle, slats	6.4	17.7	580	1166
Calves, litter	1.9	5.7	332	1024
Sows, slats	8.7	22.1	730	1234
Weaners, slats	5.3	17.9	46	750
Growing-finishing pigs, litter	9.1	21.7	394	2241
Growing-finishing pigs, slats	14.9	43.4	319	1635
Laying hens, deep litter	25.2	72.3	38.3	3643
Laying hens cages	6.1	14.5	7.7	711
Broilers, litter *)	8.0	40.3	8.9	343

*) Average of four measurements over a production circle

***) One HPU (Heat Producing Unit) is corresponding to 1000W in total heat production at 20°C

The investigation shows that in houses for finishers, the ammonia concentration in the 1990s is about 70% higher as found in the 1970s. It also shows that the ammonia concentration is much bigger in houses with slatted floor as in straw bedded pens. Since the 1990s new facilities with proper housing design and management often shows much lower ammonia concentrations.

For layers, the ammonia concentration and emission is very high for laying hens in deep litter.

Today almost all new building facilities for dairy cows are with cubicles and natural ventilation. It is open houses with very high air exchanges, and it is a real challenge to measure the ammonia emission from such facilities. Due to very big ventilation openings, which can be extended to an open sided building, it is necessary to measure the ventilation flow indirectly by tracer gases or by means of the increase in carbon dioxide concentration due to an animal CO₂ production of about 0.185 m³ per HPU per hour. Figure 5 shows the results from a Danish investigation based on carbon dioxide measurements (Zhang et al. 2005).

The ammonia emission, which is the product of the ammonia concentration and the ventilation flow, is strongly dependent on the indoor temperature and increases roughly proportional with the indoor temperature. In this example with naturally ventilated houses, the ventilation openings are normally fixed, why the ventilation rate can be considered as nearly independent of the indoor temperature. In houses with mechanical ventilation with temperature dependent ventilation rate, the relation between indoor temperature and ammonia emission will be different.

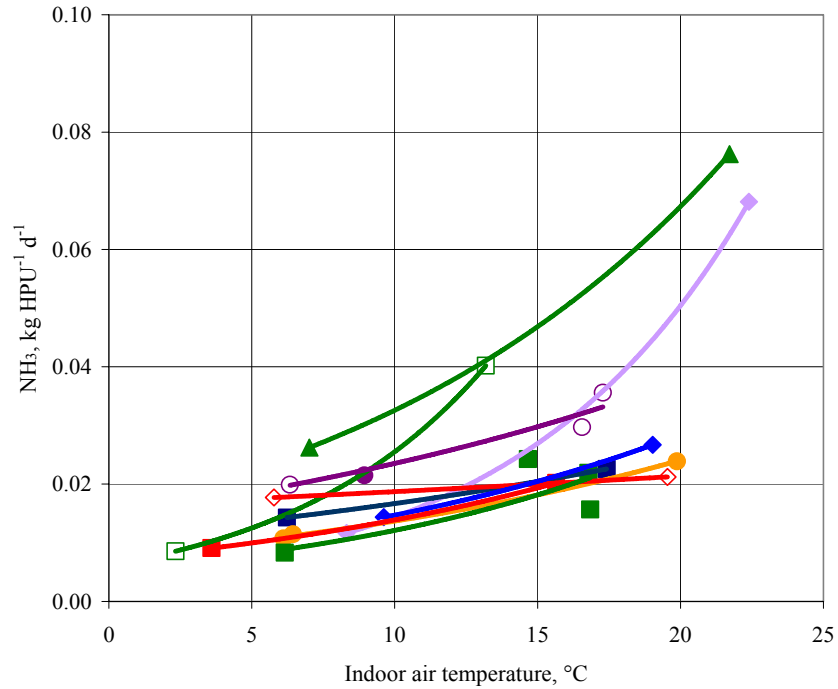


Figure 5. Average NH₃ emission rates for nine buildings with different slurry handling principles over the measurement period

Also the storages are contributing to the ammonia emission from animal facilities. In Denmark slurry tanks must be covered by a natural floating-layer of manure or be covered e.g. by a tent as shown in Figure 6. By use of a tent, the ammonia emission can be reduced to few percents.



Figure 6. Slurry tank with tent for limiting the ammonia emission from storages

In Denmark the ammonia emission from animal facilities is strongly regulated by authorities, based on the knowledge of nitrogen release from the animals and evaporation factors. The nitrogen release is shown in Table 2, where N is total N including both ammonium N and organic N (Ministry of environment, 2003). The guidelines are under revision and attempted to be finished in 2006.

Table 2. Danish norm values for nitrogen release per animal

Specie	Type	Kg N in faeces + urine per animal per year
Pigs	Sows	26.72
	Weaners	0.65
	Growing-finishing pigs	3.14
	From weaning to slaughtering	3.79
Cattle (large races)	Dairy cows	127.3
	Calves, 0-6 months	5.8
	Heifers, 6 months to calving	30.8
	Bulls calves, 6 months to 382 days	24.3
	Cows with suckling calves	73.3
Mink	Female yearly with puppies	4.59
	Per produced fur	0.88
Poultry	Broilers, 35 days	0.0405
	Broilers, 40 days	0.0536
	Broilers, 45 days	0.0665
	Eco-broilers, 81 days	0,127
	Layers, cages	0,655
	Layers, floor keeping	0,857
	Layers with outdoor access	0,800
	Layers, ecological	0,879
	Brooding hens	0.907
	Turkeys, hens	0,481
	Turkeys, cocks	0,878
Ducks	0,173	
Geese	0,561	

For each type of animal and housing system, the Danish norm prescribes standard values for how much of the nitrogen, released from the housing system by faeces and urine, there is evaporated. Table 3 shows some of the evaporation factors. The table shows that there are big differences between the emission factors for different housing systems. For dairy cows in cubicle houses with slatted floor and underlying scrapers in the alleys, the emission is e.g. 0.06 kg NH₃-N per kg total N in manure, but 0.10 kg NH₃-N per kg total N for cubicles with solid floors. For layers in cages with manure belts, the emission is 0.10 kg NH₃-N per kg total ammonium, which is very low compared to other types of layer houses with floor keeping. This illustrates that the housing system has a very big impact on the ammonia emission. For a certain system, the management has also a big impact

Table 3. Emission factors from Danish animal housing facilities (per cent of excreted N in faeces and urine)

		Solid floor	Partly slatted floor	Fully slatted
Cattle	Dairy cows, tie stall, dung channel	5	3	
	Dairy cows, cubicles	10	6-8	
	Dairy cattle, deep litter	6-7		
Pigs	Pregnant sows	16	10	14
	Growing-finishing pigs	18	12	16
	Weaners,	25	10	16
	Farrowing sows,		10	20
Poultry	Broilers, traditional	20		
	Broilers, ecological	25		
	Layers, floor keeping, with outdoor access	30		
	Layers, cages, manure cellar	12		
	Layers, cages, manure conveyer	10		
	Layers, eggs for brooding	30		
	Turkeys, ducks, geese	20		

The ammonia emission can finally be calculated by multiplication of Kg N (faeces + urine per) animal per year in Table 2 with the evaporation factors in Table 3.

Many other factors impact on the ammonia emission as e.g. the feed composition in respect to the protein content.

In Danish investigations not published, it is indicated that the ammonia emission from growing-finishing pigs is about 25% lower for pigs using maize silage as rooting material than for pigs using straw. Another interesting result from the same experiments shows that the methane emission increases by about 80% using maize silage as rooting material compared to the use of straw.

Mink Facilities

Housing facilities for mink are based on elevated cages, where the feces and urine drop from the cage bottom to the underlying slurry gutter, often in two-row open mink houses. Together with protein rich feed, it gives a high potential for ammonia emission. To show the influence of management and temperature on the emission, experiments were carried out in a two row open mink house as shown in Figure 8 (Pedersen and Sandbøl, 2002). The results show that the ammonia emission is very sensitive to both the surrounding temperature and the frequency of cleaning the slurry gutter.

BAT- Best Available Technique

In Europe, many countries have developed an evaluation system for evaporation of ammonia and greenhouse gases from different production systems under the umbrella BAT. The idea is to guide farmers to select environment friendly production systems. The BAT notes are also used by regional governments in giving permissions to changes and enlargements of existing animal production systems and to establishment of new farms. In Table 4 is shown some examples of Danish BAT notes:

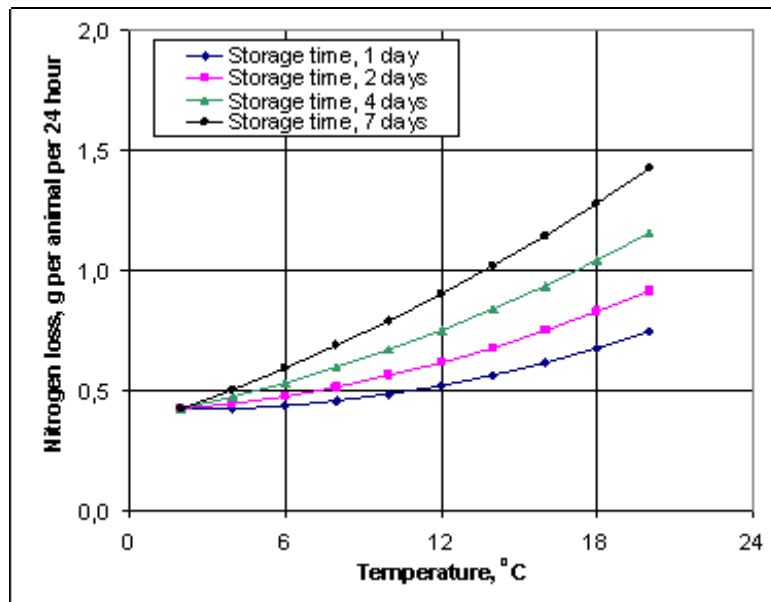


Figure 8. Nitrogen losses from two-row open house for mink of 1800 g

Table 4. Danish BAT notes on ammonia emission

Type and group number	Principles	Reference house	Reduction, %		
			Dust	Ammonia	Odour
Layers 105.02-51	Pit drying of manure, floor keeping deep litter and scraping area	Floor keeping, 2/3 deep litter, 1/3 manure cellar	-	50-60	-
Layers 105.02-52	Manure drying in layer houses, cages	Manure cellar	-	50-70	-
Pregnant sows 106.01-51	Cooling of bottom in manure cellar	Partly slatted floor	-	30	-
Farrowing sows 106.02-51	Farrowing pens with partly slatted floor	Fully slatted floor	-	50	-
Weaners 106.03-52	Two-climate pens, partly slatted floor	Fully slatted floor	-	30	-
Growing-finish. pigs 106.04-52	Partly (1/3) slatted floor	Fully slatted floor	-	15	-
Growing-finish. pigs 106.04-53	Cooling of bottom of slurry channel, partly slatted floor.	Fully slatted floor	-	30	-
Growing-finish. pigs 106.04-58	Fully slatted floor	Air washing with sulphuric acid cleaning of up to 60% of exhaust air	-	50	Small reduction
Dairy cattle 107.04-51	Pre-manufactured floors with urine grooves and drains	Cubicles and slatted floor above ring shaped manure channel with daily circulation	-	50	-

In addition to Table 4, there is a list of BAT candidates, not yet confirmed. Some of them regard reduction of the pH in slurry channels, by means of adding 0.5% sulphuric acid to the slurry and circulating it by a special technique. The reduction of ammonia emission is about 70%, which is very good, but the economy and other aspects have to be considered further.

Odour Emission

Much research is carried out worldwide over the last decades on climatization of animal houses, and many examples are shown on how to improve indoor climate and to reduce the emission from animal houses. Why are those strategies and technical solutions not implemented in praxis? The answer is simple: because it is primarily a question about economy. In the 1970s some few Dutch farmers with their farms in villages installed air scrubbers in combination to existing exhaust fans in pig houses. Visits on sites convinced the author that the odour was reduced very much, and the remaining odour was turned to a mouldy smell. Also measurements carried out by SBI (Danish Building Research Institute) and by DLU (The Agricultural Experimental Center, Ørritslevgaard) in the 1970s, showed by olfactorimetric that it was possible to reduce the odour very much. In the last three decades many experiments are carried out worldwide on air scrubbers and bio-filters, and today it is well proved that it is possible to reduce the odour. Due to increasing production cost for electricity, maintenance and manpower, very few odour reduction plants are taken into use. The coming years of research on bio-scrubbers and bio-filters will probably give a better understanding of the mechanisms which control the efficiency of such systems.

Many other examples of odour reduction are tried. For instance, feeding the animal with small amounts of plant products, mixing the slurry with clay minerals, etc., but sustainable solutions are not yet seen.

Measurements on odour emission in ongoing at more research units in Denmark. Some recent results carried out at Danish Agricultural Advisory Service, are shown in Table 5

Table 5. Odour measurements in cattle and poultry houses

		Odour production	
		Per animal (OU/s)	Per 1000 kg (OU/s)
Cattle	Dairy cows, tie- and loose housing	145	242
Poultry	Hens in cages	0.75	398
	Hens in floor keeping	1.68	897
	Broilers	1.77	1217

The odour emission from broilers seems very high compared to cattle, expressed per 1000 kg body mass, but the manure production of 1000 kg broiler is much higher than from 1000 kg dairy cows. Expressed in another way, the total heat production, related to the metabolic mass of animals, broiler produces about eight times as much heat as 1000 kg dairy cow, which must be taken in consideration when evaluating the data.

A big task for the future will be to learn more about the components include in the odour from animal houses and to get a better understanding of the behavior of the different chemical components in respect to half-time reduction, etc. In Table 6 is shown the results of a Danish experiment, where the results of an odour panel are compared to results from a chromatography method. The ranking is based on the number of panel members who detected the individual odorants. (Kai & Schäfer, 2004). The percentage is referring to how many out of seven panelists were able to identify some smell. The table just illustrates a step in the right direction of a long path leading to sufficient understanding of odour emission from animal production facilities.

Table 6. GC- Olfactometry analysis versus odor panel description

Pct.	GC-Olfactometry analysis Odor panel description	GC-MS analysis Odor compound
100	Sweat, sour, pig house, cheese	Butanoic acid
100	Pig odour, forest floor	Not identified
100	Sour, pig house, feet	Not identified
100	Flower, fresh, fermented, sour	Not identified
86	Pig house, faecal	4-methyl-phenol
86	Forest floor	Not identified
86	Faecal, pig house, unpleasant	Not identified
71	Sour	2-methyl-propionic acid
71		Not identified
71	Mushroom	Not identified
71	Boiled rice	Not identified
71		Not identified
71	Sour, unpleasant, cheese	Not identified
71	Tar	Not identified
71	Faecal	Not identified
71	Pig house	Indole
57	Pig house	Octanoic acid
57		Not identified
57		Not identified
43		Propanoic acid
43		Phenol
43	Feet	Not identified
43	Burnt, tar	Not identified
43	Tar	Not identified
43		Not identified

Conclusion

The challenge for the next decades will be:

- To improve the indoor climate in respect to humans and animals and minimizing of emission of dust, gases and odour from animal facilities;
- To develop efficient inexpensive air cleaners for exhaust air, with low investment, running costs and need maintenance;
- To meet the continuous fast changes in animal housing systems with appropriate solutions

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