Conclusions

Our results indicate that the human populations of these agricultural communities are at considerable risk of direct exposure to airborne contaminants, and that these contaminants are likely to enter the food chain through exposed crops and domesticated animals, potentially affecting urban populations. The brickmaking industry of Querétaro state, Mexico, under current conditions of operation, poses a serious threat of environmental contamination with persistent organic pollutants, as well a threat of direct and indirect impacts on rural populations. The inefficiency of combustion and unregulated mix of questionable fuel substances contribute to emissions of toxic air pollutants, and also provide the most efficient avenues of mitigating the threat. In these rural communities agricultural air quality is less impacted by traditional sources of agriculturally-sourced emissions than by co-located primary industries such as brickmaking.

Acknowledgements

DAG acknowledges the University of California Institute for Mexico and the United States--UC MEXUS and CEACA, Autonomous University of Queretaro, for partial support during these studies and the preparation of the manuscript.

References

ATSDR (2000). Toxicological profile for polychlorinated biphenyls (PCBs). Atlanta, GA: U.S,, Department of Health and Human Services, Public Health Service.

Bertazzi, P. A., Consonni, D., Bachetti, S., Rubagotti, M., Baccarelli, A., Zocchetti, C. and Pesatori, A. C. (2001). Health effects of dioxin exposure: a 20-year mortality study. Am J Epidemiol 153(11): 1031-44.

Blackman, A., Newbold, S., Shih, J., and Cook, J. (2002) The Benefits and Costs of informal Sector Pollution Control: Mexican Brick Kilns. Resources for the Future. Discussion paper 00-46.

Bytnerowicz, A., Miller, P.R. and Olszyk, D.M. (1987) Dry deposition of nitrate, ammonium and sulfate to a Ceanothus crassifolius canopy and surrogate surfaces. Atmoshsperic Environment 21: 1749-1757.

CIDA, Canadian International Development Agency (2002) Clean development mechanism project opportunities in Bangladesh: Pre-feasibility report on a brick manufacturing fuel substitution CDM project. The Pembina Institute.

Denison, M. S. and Heath-Pagliuso, S. (1998). The Ah receptor: a regulator of the biochemical and toxicological actions of structurally diverse chemicals. Bull Environ Contam Toxicol 61(5): 557-68.

Devkota, S.R. and C. Neupane, 1994. Industrial Pollution Inventory of Nepal, Industrial Pollution Control Management Project. UNDP/UNIDO/MOI/NEP/91-029. Kathmandu.

Dyke, P. H., Foan, C., Wenborn, M. and Coleman, P. J. (1997). A review of dioxin releases to land and water in the UK. Sci Total Environ. 207(2-3): 119-31.

Faqi, A. S., Dalsenter, P. R., Merker, H. J. and Chahoud, I. (1998). Effects on developmental landmarks and reproductive capability of 3,3',4,4'-tetrachlorobiphenyl and 3,3',4,4',5-pentachlorobiphenyl in offspring of rats exposed during pregnancy. Hum Exp Toxicol 17(7): 365-72.

Fiedler, H. (1996). Sources of PCDD/PCDF and impact on the environment. Chemosphere 32(1): 55-64. GCUNEP (2001). Stockholm Convention on Persistent Organic Pollitants. Governing Council of the United Nations Environment Programme.

Freer-Smith, P.H., Holloway, S. and Goodman, A. (1996) The uptake of particulates by an urban woodland: Site description and particulate composition. Environmental Pollution 95: 27-35.

Geyer, H. J., Schramm, K. W., Feicht, E. A., Behechti, A., Steinberg, C., Bruggemann, R., Poiger, H., Henkelmann, B. and Kettrup, A. (2002). Half-lives of tetra-, penta-, hexa-, hepta-, and octachlorodibenzo-p-dioxin in rats, monkeys, and humans--a critical review. Chemosphere 48(6): 631-44.

Guo, Y. L., Hsu, P. C., Hsu, C. C. and Lambert, G. H. (2000). Semen quality after prenatal exposure to polychlorinated biphenyls and dibenzofurans. Lancet 356(9237): 1240-1.

(INAFED) Instituto Nacional para el Federalismo y Desarrollo Municipal. Sistema Nacional de Información Municipal. http://www.inafed.gob.mx/wb2/

Kerkvliet, N. I. (2002). Recent advances in understanding the mechanisms of TCDD immunotoxicity. Int Immunopharmacol 2(2-3): 277-91.

Ma, T.H., Cabrera, G.L., Chen, R., Gill, B.S., Sandhu S.S., Valenberg, A.L.,Salamone, M.F. (1994) Tradescantia micronucleus bioassay. Mutation Research 310: 221-230.

Marquez, R. O. (2001) Appropriate Chemistry for the Economically Limited People of the Earth." Doctoral Dissertation, New Mexico State University.

Martin, F., Piearce, T., Hewer A., Phillips, D., Semple, K. (2005) A biomarker model of sublethal genotoxicity (DNA single-strand breaks and adducts) using the sentinel organism Aporrectodea longa in spiked soil. Environ Pollut. 38(2): 307-15.

Micieta, K. and Murin G. (1996) Microspore analysis for genotoxicity of polluted environment. Environmentaland Experimental Botany 36: 21-27.

Møller, P., Knudsen, E.K., Loft, S., and Wallin, H. (2000) The Comet Assay as a Rapid Test in Biomonitoring Occupational Exposure to DNA-damaging Agents and Effect of Confounding Factors. Cancer Epidemiology, Biomarkers & Prevention 9: 1005-1015.

Pitarque, M., Vaglenov, A., Nosko, M., Pavlova, S., Petkova, V., Hirvonen, A., Creus, A., Norppa, H., Marcos, R. (2002) Sister chromatid exchanges and micronuclei in peripheral lymphocytes of shoe factory workers exposed to solvents. Environ Health Perspect 110(4): 399-404.

Safe, S. (1993). Development of bioassays and approaches for the risk assessment of 2,3,7,8-tetrachlorodibenzo-p-dioxin and related compounds. Environ Health Perspect 101(Suppl 3): 317-25.

Safe, S. and Krishnan, V. (1995). Chlorinated hydrocarbons: estrogens and antiestrogens. Toxicol Lett 82-83: 731-6. Safe, S. H. (1995). Modulation of gene expression and endocrine response pathways by 2,3,7,8-tetrachlorodibenzo-p-dioxin and related compounds. Pharmacol Ther 67(2): 247-81.

Salomone, M., Geddle, J.A., Stuart, E. and Katz, M. (1890) Towards an improved micronucleus test: studies on 3 model agents, mitomycin C, cyclophosphamide and dimethylbenzanthracene. Mutatation Research 74(5):347-56.

Singh, N; McCoy, M. Tice, R., Schneider, E. (1988) A simple technique for quantitation of low levels of DNA damage in individual cells. Exp Cell Res 175(1):184-91.

(TCEQ) Texas Commission on Environmental Quality (2000), A Study of Brick-Making Processes along the Texas Portion of the U.S.-Mexico Border: Senate Bill 749. SFR-081-02

Comparison of Models Used for the Calculation of National Ammonia Emission Inventories from Agriculture in Europe

B. Reidy[1*], U. Dämmgen[2], H. Döhler[3], B. Eurich-Menden[3], F.K. van Evert[4], N.J. Hutchings[5], H.H. Luesink[6], H. Menzi[1], T.H. Misselbrook[7], G.-J. Monteny[8], J. Webb[9]
[1]Swiss College of Agriculture, Laenggasse 85, CH-3052 Zollikofen, Switzerland
[2]Federal Agric. Res. Cent., Inst. of Agroecology, Bundesallee 50, 38116 Braunschweig, Germany
[3]Association for Technology and Structures in Agriculture (KTBL), Bartningstrasse 49, 64289 Darmstadt, Germany;
[4]Plant Research International, P.O. Box 16, 6700 AA Wageningen, The Netherlands
[5]Danish Institute of Agricultural Sciences, Research Centre Foulum, Tjele, Denmark
[6]LEI, P.O. Box 29703, 2502 LS The Hague, The Netherlands
[7]Inst. of Grassland and Env. Research, North Wyke, Okehampton, Devon EX20 2SB, UK
[8]Wageningen UR, Agrotech. and Food Innovations B.V., 6700 AA Wageningen, The Netherlands
[9]ADAS Research, Woodthorne, Wergs Road, Wolverhampton WV6 8TQ, UK

Abstract

Six mass-conservative N-flow models, used to calculate national ammonia (NH_3) emissions from agriculture in different European countries, were compared using standard data sets. Two scenarios were run (for dairy cattle and for pigs) with three different levels of model standardisation: a) standardised inputs to all models (FF scenario); b) standard N excretion, but national values for emission factors (FN scenario); c) national values for N excretion and emission factors (NN scenario). Results of the FF scenario showed very good agreement (following iterative modifications to some of the models) between models, indicating that the underlying N flows of the different models are highly comparable. Differences between models for the FN and NN scenarios could be explained by differences in national management practices and climate. The congruency exercise led to an improved harmonisation of the national models.

Introduction

The Gothenburg Protocol of the UN Convention on Long-range Transboundary Air Pollution (UNECE 1999) requires the reporting of national annual emissions of ammonia (NH_3). Accurate inventories of agricultural NH_3 emissions are required to calculate the total national emissions, since they commonly account for more than 80 % of the total emissions (EMEP 2005). To allow a co-ordinated implementation of the Protocol, different national inventories should be comparable; at present they are not.

A core group of emission inventory experts has therefore inaugurated the EAGER network (European Agricultural Gaseous Emissions Inventory Researchers Network), with the aims of achieving a detailed overview of the present best available inventory techniques, compiling and harmonizing the available knowledge on emission factors (EF) and initiating a new generation of emission inventories that satisfies protocol requirements. As a first step in summarizing the available knowledge, the objective of the work reported in this paper was to determine the degree to which results obtained with different NH_3 emission models currently used for inventory calculations agree, and to evaluate any larger disagreements.

Methods

The models used for the comparisons have been used in the framework of the national NH_3 emission inventory calculations and manure policy analyses in different countries of Europe (Table 1). The models all use a mass-conservative N-flow approach starting with a specific amount of nitrogen (N) excreted by a defined livestock category and simulate the total ammoniacal nitrogen (TAN) flow over the different stages of emissions (grazing, housing, manure storage and application). Ammonia emissions are generally calculated with EF, where the EF is the percentage of the respective TAN pool emitted. Emissions were compared for a dairy cattle and a pig scenario, with different levels of model standardizations. At a first level of comparison, the congruency of the underlying N flow was tested (FF scenario). For this purpose, the national specific N excretions rates, TAN contents and EFs were replaced by a set of standardised values. At a second level of comparison, only the N excretion and TAN contents were standardised whereas the national EFs were used for the calculations (FN scenario). Finally, emissions were also calculated using the national N excretion rates, TAN contents and EFs (NN scenario).

Table 1. Models used in the comparisons.

Model	Country	Objectives of the model	Reference
DYNAMO	Switzerland	Estimation of the magnitude of NH_3 losses at the farm and national level	Reidy and Menzi, 2006
DanAm	Denmark	Estimation of the magnitude of NH_3 losses at the national level	Hutchings et al. 2001 1)
GAS-EM	Germany	Estimation of NH_3 and other N losses at the national and district levels	Dämmgen et al. 2002 2)
NARSES	United Kingdom	Estimation of the magnitude, spatial distribution and time course of agricultural NH_3 emissions at the national level	Webb and Misselbrook 2004
MAM	Netherlands	Manure policy analyses and estimation of NH_3 emissions at the farm and national level	Groenwold et al. 2002
FARMMIN	Netherlands	Ex-ante evaluation of the effect of management on profitability and nutrient losses	van Evert et al. 2003

1) An updated model fully based on the N-flow was used in this study.

2) An updated model fully was used in this study.

Results and Discussion

Running the models with the defined livestock and manure management parameters and the standardised N excretion and EF (scenario FF) resulted in very similar estimates of the NH_3 emission for the respective emission stages as well as total emissions. This indicates that the underlying N flows of the different models are highly comparable. The small differences observed could largely be explained with slight modifications of the N flow, either related to an altered partitioning of the N deposited during grazing and housing or to the extent other N transformations are taken into account (e.g. mineralization or denitrification processes) in the different models. It is worthwhile mentioning that initial simulations of this scenario had shown greater differences between the models but that when investigated, these were found to be due to errors in logic or in programming.

Differences between the models were more pronounced when the emissions were calculated with national emission factors and/or national N excretion rates (FN and NN scenario). The variation in the calculated emissions was primarily the result of the distinct national emission factors and N excretion rates. Both parameters reflect the specific livestock and manure management systems of the different countries and can be explained by:

- Differences in N excretion that result from differences in feeding practice (e.g. the protein concentration in the diet) or from different production intensities (e.g. milk yield per cow, growth rate per pig).
- Variations in the types of animal housing, storage and technology used for field applications.
- Variations in animal management (e.g. duration of the grazing period)
- Climatic factors

The variations in the calculated emissions of the FN and NN scenarios are therefore real, fully valid and are the reason why emission inventories are most appropriately constructed at a scale that reflects the heterogeneity in agriculture and climate.

Conclusions

The models compared are generally well congruent and existing differences in the results obtained can mostly be explained by existing differences of natural conditions and farm management in the countries from which the models originate. The congruency exercise has led to a greater harmonization of the structure and function of the models tested, because the scientific debate necessary to understand the variation in results from the different models generated awareness and consensus concerning the importance of some processes (e.g. mineralization).

References

Dämmgen, U., M. Lüttich, H. Döhler, B. Eurich-Menden, and B. Osterburg. 2002. GAS-EM – a procedure to calculate gaseous emissions from agriculture. *Landbauforschung Völkenrode* 52:19-42.

EMEP. 2005. Database of the national submissions to the UNECE LRTAP Convention maintained at EMEP. http://webdab.emep.int/ (February 14, 2006).

Hutchings, N. J., S.G. Sommer, J.M. Andersen, and W.A.H. Asman. 2001. A detailed ammonia emission inventory for Denmark. *Atmospheric Environment* 35:1959-1968.

Luesink, H.H., C.H.G. Daatselaar, G.J. Doornewaard, and H. Prins. 2004. Sociaal-economische effecten en nationaal mestoverschot bij varianten van gebruiksnormen. *LEI, Den Haag, Rapport 3.04.08*

Reidy, B., and H. Menzi. 2006. DYNAMO: An ammonia emission calculation model and its application for the Swiss ammonia emission inventory. *Proceedings of the Workshop on Agricultural Air Quality: State of the Science*. June 5-8, Bolger Center, Potomac, MD, USA.

UNECE, 1999a: Protocol to the 1979 Convention on Long-Range Transboundary Air Pollution to abate acidification, eutrophication and ground-level ozone. *United Nations Economic Commission for Europe (UNECE)*, Geneva

Van Evert, F., H. van der Meer, H. Berge, B. Rutgers, T. Schut, and J. Ketelaars. 2003. FARMMIN: Modeling crop-livestock nutrient flows. *Agronomy Abstracts 2003*, ASA/CSSA/SSSA, Madison, WI.

Webb, J., and T.H. Misselbrook. 2004. A mass-flow model of ammonia emissions from UK livestock production. *Atmospheric environment* 38, 2163-2176.

DYNAMO: An Ammonia Emission Calculation Model and Its Application for the Swiss Ammonia Emission Inventory

Beat Reidy and Harald Menzi
Swiss College of Agriculture, Zollikofen, Switzerland

Abstract

Within the framework of the Gothenburg protocol member countries will have to regularly report in detail on the of evolution ammonia emissions from agriculture. Present inventory approaches are generally not detailed enough to meet these requirements. A new emission inventory methodology has therefore been developed which is based on detailed survey data from farm and manure management and model calculations with an empirical mass flow model (DYNAMO). The calculations revealed that ammonia emissions from agriculture have decreased in Switzerland from 1990 to 2000 by approximately 20%. This change was primarily due to a substantial decrease in the livestock numbers and improved agricultural practices. The 2010 emission target of the Gothenburg protocol of a 13% reduction has therefore already been achieved.

Introduction

The Gothenburg Protocol will require member countries to regularly report on the evolution of ammonia emissions from agriculture. Existing emission inventory approaches are not sufficient to reproducibly detect relatively small changes of emissions (Stadelmann et al., 2006) because they are mainly based on expert assumptions and can only take into account farm management parameters to a very limited extent. For the calculation of a new Swiss ammonia emission inventory a new approach was therefore developed. It is based on emission calculations with am empirical model DYNAMO (**Dyn**amic **A**mmonia Emission **Mo**del) and on the results of a representative farm survey.

Methods

DYNAMO is an empirical mass-flow model following the N flow approach originally described by Menzi and Katz (1997) but with a widely extended set of farm management parameters. The model calculates emissions on the basis of the N-flow through the manure handling chain using emission factors in percent of the relevant amount of nitrogen present at each stage of emission. Considered emission stages include animal houses and hardstandings, manure storage and application, grazing, mineral fertilizer as well as crops and meadows. The model takes into account different animal categories and manure types and considers a series of different management practices relevant for ammonia emissions. Model input parameters on farm management were obtained from a questionnaire sent to 3880 farms. The stratified survey allowed the differentiation between nine regions (3 geographical regions x 3 altitude zones) and four farm types (Menzi and Reidy 2006). 1950 farms were eventually included in the emission calculations. The ammonia emissions for each of these farms were individually calculated with DYNAMO and the results up-scaled to the national level.

Results and Discussion

According to the ammonia emission inventory for the year 2000, agriculture is responsible for about 94% of the total anthropogenic emissions in Switzerland. Of the agricultural emissions, 88% originate from livestock production and manure management. Twelve percent of the emissions can be attributed to crop production, either to mineral fertilizer and sewage sludge application or to direct ammonia emissions from crops. With respect to livestock classes, the most important species are cattle and pigs, which contribute more than 90% of the emissions from animal production. In terms of emissions stages, 58% of the emissions result from manure application (Figure 1). Although the use of low emission application techniques (band application systems) has considerably increased in the last few years, still more than 85% of the slurry is spread with broadcasting systems as revealed by the survey. Primarily due to a strong increase in loose housing systems and the recent widespread introduction of hardstandings on cattle and pig farms the contribution of livestock housing systems to the total emissions from livestock has increased since 1990 from 24 (Stadelmann et al., 1998) to 28%. The greatest part of the slurry is still stored in covered slurry tanks. Manure storage therefore contributes only 12% of the total emissions from livestock production and manure management.

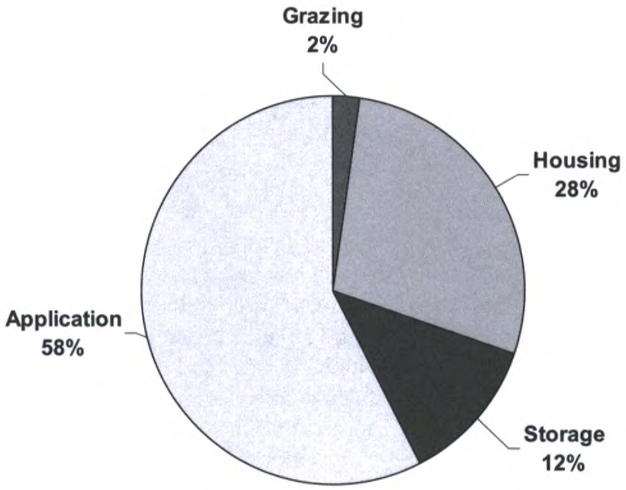


Figure 1. Contribution of each emission stage to the total ammonia emissions from livestock production and manure management in the year 2000.

Average annual NH_3-N emissions per hectare agricultural area in the different regions ranged up to 38 kg. Maximum annual emissions of 60 kg ha^{-1} NH_3-N can be observed in the central and eastern part of Switzerland. In these regions average livestock densities of more than two livestock units per hectare agricultural area can frequently be observed.

Nevertheless, primarily thanks to a substantial reduction of the overall livestock numbers and improved agricultural practices the total emissions from agriculture decreased by about 20% since 1990. The 2010 emission reduction target of 13% of the Gothenburg protocol has therefore already been exceeded.

References

Menzi, H., and B. Reidy. 2006. Assessment of the ammonia emission abatement potential for distinct geographical regions and altitudinal zones in Switzerland. *Proceedings of the Workshop on Agricultural Air Quality: State of the Science.* June 5-8, Bolger Center, Potomac, MD, USA.

Menzi, H., and P.E. Katz. 1997. A differentiated approach to calculate ammonia emissions from animal husbandry. In: J.A.M. Voermans and G.J. Monteny (Eds). "*Ammonia and odour emissions from animal production facilities*", Proc. International Symposium, Vinkeloord, NL, 6-10 October 1997, 35-42.

Stadelmann, F.X., B. Achermann, H.J. Lehmann, H. Menzi, S. Pfefferli, U. Sieber, and A. Zimmermann. 1998. Ammonia emissions in Switzerland: Present situation, development, technical and economic assessment of abatement measures, recommendations. *Institut für Umweltschutz und Landwirtschaft Liebefeld (IUL) und Forschungsanstalt für Agrarwirtschaft und Landtechnik (FAT).* 56 pp.

Effect of Air Pollution on Crop Productivity in China: Application of an Agricultural Ecosystem Model (AEM)

Wei Ren, Hanqin Tian, and Mingliang Liu

Institute of Geographical Sciences and Natural Resources Research, Chinese Academy of Sciences, Beijing, 100101, PR China.

Abstract

Evidence from field observation has shown that tropospheric ozone concentration in both urban and rural areas has increased significantly. Emissions of ozone precursors are expected to be double in the next 20 years in China and the levels of photo-oxidants may increase substantially. The individual mean values of related trace gases (NO, NO_X, SO_2 and CO) were much higher than that of background level. Strong seasonal variations of observed data in the sites were expounded with peaks in winter. To assess the effect of air pollution on crop productivity, we have developed the Agricultural Ecosystem Model (AEM), which fully couples crop growth processes and major biogeochemical cycles with hydrological cycles at a daily time step, especially adding air pollution (ozone, aerosol) effects on production processes. Our integrative assessment with the AEM suggests that elevated tropospheric ozone concentration has led to a significant decrease in the yield of four major grain crops (wheat, rice, corn and soybean). Of the four major grain crops, soybean yield is the most sensitive to elevated tropospheric ozone concentration, with a loss of more than 20%. Results drawn from our analysis imply that rising tropospheric ozone concentration could potentially influence China's capacity in feeding its growing population.

New Standards for Odour Emissions from Pig Facilities in Denmark

A. L. Riis
The National Committee for Pig Production, DANISH BACON & MEAT COUNCIL,
Copenhagen, Denmark

Abstract

The aim of this study was to determine the average odour emission from the most common pig facilities in Denmark during the summer in order to find new standards for odour emissions. The summer period was selected as odour emissions from finishing pigs was found to be highest during this period and is of great concern during public administration in the regulation of air pollutants.

To determine the odour emission from the nine most common categories of pig facilities in Denmark, 216 odour samples were collected. The nine categories of pig facilities were: dry sows kept in individual crates with partially slatted floors, dry sows kept loose, lactating sows kept in individual crates with partially or fully slatted floor, respectively, weaners kept in pens with partially or fully slatted floor, respectively, and finishers kept in pens with partially slatted, fully slatted or drained floor, respectively. Each category was represented by four intensive pig units, which all were mechanically ventilated. Odour samples were collected from each unit on three days with two samples per day. The samples were collected from the exhaust ventilation air in Tedlar® bags, and odour concentration was determined by an accredited odour laboratory. The ventilation rate of each house type was measured by a Fancom measuring fan.

The odour emission from dry sows did not differ significantly between the types of facility. This was also observed for weaners and therefore one number for odour emission is given for each of these stages of production. No difference was found between fully slatted floor and drained floor in finisher facilities. The odour emissions given in $OU_E\ s^{-1}\ animal^{-1}$ were 16 for dry sows, 72 for lactating sows with partially slatted floor, 100 for lactating sows with fully slatted floor, 7 for weaners, 19 for finishers with partially slatted floor and 29 for finishers with fully slatted or drained floor.

Furthermore, the ammonia concentration was measured in the exhaust ventilation air by Kitagawa Gas Detector Tubes at each odour sample. The ammonia emissions were 10.9, 13.4, 25.3, 0.1, 1.7, 4.6 and 6.2 g $d^{-1}\ animal^{-1}$ for dry sows, lactating sows with partially slatted floor, lactating sows with fully slatted floor, weaners with partially slatted floor, weaners with fully slatted floor, finishers with partially slatted floor and finishers with fully slatted or drained floor.

Introduction

In cases concerning air pollution with odour from livestock buildings in Denmark, emission standards for odour based on measurements from German livestock buildings in the 1980s (Oldenburg, 1989) have been used so far. Consequently, there was an acute need for emission standards of odour measured in Danish livestock buildings in 2005. Therefore, as part of the Agreement on the Action Plan for the Aquatic Environment III 2005-2015 in Denmark (Danish Ministry of the Environment, 2004), new standards for odour emissions from livestock facilities were selected as an area that required research. The new standards for odour emissions will form part of a new odour guide from the Danish Ministry of the Environment.

Odour emissions from different categories of pig facilities have been measured in other countries (Mol and Ogink, 2002; Hayes et al., 2005; Gay et al., 2003; Lim et al., 2001; Zhu et al., 2000; Verdoes and Ogink, 1997), but not in Denmark.

The main aim of the present study was to determine the average odour emission from the most common pig facilities in Denmark during the summer in order to find new standards for odour emissions. The summer period was selected as odour emissions from finishing pigs was found to be highest during this period and is of great concern during public administration in the regulation of air pollutants. Secondarily, the ammonia emissions from the pig facilities were determined at each odour sample. However, only the odour emissions found in this study are used for new regulation in Denmark.

Methods

The nine most common categories of pig facilities in Denmark were selected for this project. Each category was represented by four intensive pig units, which all were mechanically ventilated. The nine categories were:

- Dry sows kept in individual crates with partially slatted floor
- Dry sows kept loose
- Lactating sows kept in individual crates with partially slatted floor
- Lactating sows kept in individual crates with fully slatted floor
- Weaners kept in pens with partially slatted floor
- Weaners kept in pens with fully slatted floor
- Finishers kept in pens with partially slatted floor
- Finishers kept in pens with drained floor
- Finishers kept in pens with fully slatted floor.

Olfactometric odour samples were collected in each unit on three days with two samples per day. Samples were collected from the exhaust ventilation air in Tedlar® bags containing 30 L in compliance with European olfactometric standard EN:13725 (CEN, 2003). On each sampling day, the Tedlar® bags were filled with 0.7 L per minute between 12.00 and 1.00 pm for the first sample and after 1.30 pm for the second sample. A total of 216 olfactometric odour samples were collected in the summer period from June to October. Odour concentration was determined by an accredited olfactometric laboratory according to European olfactometric standard EN:13725 (CEN, 2003). For calculation of the odour emission, the ventilation rate in the pig facility for each odour sample was determined with a Fancom measuring fan (FarmTech a/s, Herlufmagle, Denmark).

To determine the average odour emission from a livestock facility in the summer, the outside temperature was preferred to be averagely 20 ºC at sampling time to ensure maximum ventilation in the facilities. Typically, piglets are weaned at four to five weeks of age and therefore emissions of odour were measured 14 days after farrowing. The odour emission from weaners was measured at 19 kg since they are normally housed in the weaner facility from 7 to 30 kg. Finishers weighed averagely 65 kg when odour measurements took place. For dry sows, the production is continuously and therefore odour measurements were not planned for specific stages in the production period. At each sample day the numbers of pigs were counted and the average weight were estimated from weight at insertion, date of insertion and average daily gain.

In connection with each odour sample, the ammonia concentration was measured in the exhaust ventilation air by Kitagawa Gas Detector Tubes 105SD (Mikrolab, Aarhus, Denmark).

Data Analysis

Odour emission per animal was calculated as:

$$\mathrm{OU_E/s\ per\ animal} = \left(\frac{\mathrm{L \times Q}}{\mathrm{W \times 3600}}\right),$$

where L is the odour concentration, $\mathrm{OU_E/m^3}$; Q is the ventilation rate, $\mathrm{m^3/h}$; and W is the total number of pigs in facility, no.

The ammonia emission was estimated by using the equation:

$$\mathrm{g\ NH_3\text{-}N/d\ per\ animal} = \left(\frac{\mathrm{M \times V \times Q \times 24}}{\mathrm{R \times T \times 1000 \times W}}\right),$$

where M is the mol weight of N, g/mol; V is the volume, ppm = $\mathrm{ml/m^3}$; Q is the ventilation rate, $\mathrm{m^3/h}$; R is the gas constant, 0,0821 Lxatm/molxK; T is the temperature in Kelvin, K; and W is the number of pigs in facility, no.

The ammonia emissions and the logarithm transformed odour emissions were processed with an analysis of variance in the MIXED Procedure in SAS (SAS Inst. Inc., Cary, NC).

Results and Discussion

The average odour emissions per animal from the different pig facilities are presented in Table 1. In facilities for dry sows, no statistical difference was found in odour emission depending on whether sows were housed in individual crates or kept loose. However, in facilities for lactating sows, a tendency to lower odour emission was found, if farrowing crates had partially slatted floor compared with farrowing crates with fully slatted floor (P=0.08). The odour emission per animal from lactating sows in farrowing crates with partially slatted floor was 28 per cent lower compared with lactating sows in farrowing crates with fully slatted floor. For weaners, no statistical difference was found in odour emission depending on type of facility. Correspondingly, facilities for finishers did not reveal any difference in odour emission regardless of whether the pens had drained floor or fully slatted floor. However, a tendency to lower odour emission was found for finishers kept in pens with partially slatted floor compared with pens with fully slatted floor or drained floor, respectively (P=0.09). The odour emission from finishers kept in pens with partially slatted floor was 34 per cent lower compared with facilities for finishers with drained floor or fully slatted floor in pens.

Table 1. Odour emissions from different categories of pig facilities. The odour emissions are stated per animal and 5 per cent and 95 per cent quantiles are shown in brackets.

Category of pig facility	Number of observations	Odour emission, $OU_E\ s^{-1}$ animal^{-1} (5 per cent & 95 per cent quantiles)
Dry sows – Kept in individual crates or kept loose	48	16 (7 – 39)
Lactating sows – Kept in crates with partially slatted floor	24	72 (40 – 125)
Lactating sows – Kept in crates with fully slatted floor	24	100 (56 – 280)
Weaners – Kept in pens with partially slatted floor or fully slatted floor	48	7 (4 – 14)
Finishers – Kept in pens with partially slatted floor	24	19 (8 – 48)
Finishers – Kept in pens with drained or fully slatted floor	48	29 (13 – 78)

The connection between the observed odour emissions in the summer from different categories of pig facilities and an entire integrated production unit is illustrated in Figure 1. A unit that has a production capacity of 100 livestock units (approximately 100 sows plus finishers) where the farrowing crates for lactating sows have fully slatted floor and the pens for finishers have drained or fully slatted floor, the total odour emission will be 22.900 OU_E/second. However, if both the farrowing crates for lactating sows and the pens for finishers have partially slatted floor, the total odour emission will be reduced by 28 per cent, which corresponds to 16.600 OU_E/second. The figure illustrates that facilities for finishers account for more than 2/3 of the entire odour emission of an integrated production unit.

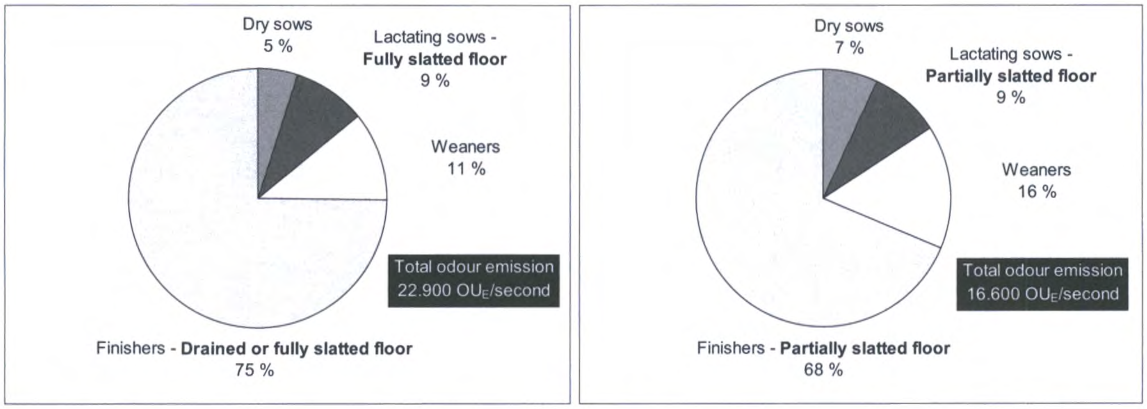


Figure 1. The figure to the left illustrates the distribution of odour emission in the summer from an integrated production unit with a capacity of 100 livestock units (approximately 100 sows plus finishers) where farrowing crates for lactating sows have fully slatted floor and pens for finishers consist of drained or fully slatted floor. The figure to the right illustrates the same distribution, but where both farrowing crates for lactating sows and pens for finishers have partially slatted floor.

The ammonia emissions from different categories of pig facilities are illustrated in Figure 2. No statistical difference was found in ammonia emission from dry sows regardless of whether they were housed in individual crates or kept loose. Correspondingly, no difference was found in ammonia emission from finishers regardless of whether the pens had drained floor or fully slatted floor. However, the ammonia emission was significantly lower from lactating sows in farrowing crates with partially slatted floor compared with crates with fully slatted floor ($P=0.01$). Also a significantly lower ammonia emission was found from weaners in pens with partially slatted floor compared with pens with fully slatted floor ($P=0.04$). In addition, the ammonia emission of finishers in pens with partially slatted floor was lower compared with pens with drained or fully slatted floor ($P=0.04$). Generally, the ammonia emission was higher per animal from dry and lactating sows compared with weaners and finishers.

Conclusions

In conclusion, the type of facility has a great impact on both odour and ammonia emissions from pig facilities in the summer. The odour emission per animal from lactating sows in farrowing crates with partially slatted floor was 28 per cent lower compared with the odour emission from lactating sows in farrowing crates with fully slatted floor. The odour emission per animal from finishers kept in pens with partially slatted floor was 34 per cent lower compared with facilities for finishers with drained or fully slatted floor in pens. Overall, the results illustrates that facilities for finishers account for more than 2/3 of the entire odour emission of an integrated production unit.

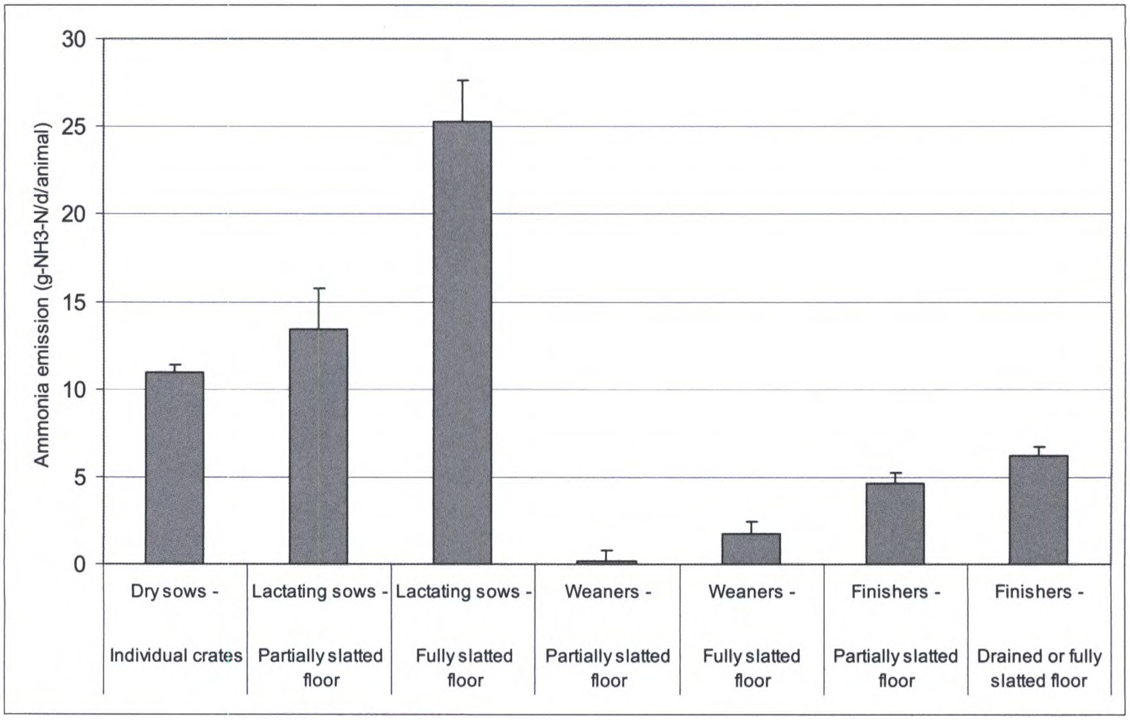


Figure 2. Ammonia emission from different categories of pig facilities measured at each odour sample in the summer period (June to October).

References

CEN. 2003. Air quality determination of odour concentration by dynamic olfactometry (EN13725). Brusssels, Belgium: European Committee for Standardization.

Danish Ministry of the Environment. 2003. Agreement on the Action Plan for the Aquatic Environment III 2005-2015 between the Danish Government, the Danish People's Party and the Christian Democrats. http://www.vmp3.dk/Files/Filer/English%20version/engelsk_oversaettelse.pdf (Feb. 9, 2006).

Gay, S.W., D.R. Schmidt, C.J. Clanton, K.A. Janni, L.D. Jacobsen, and S. Weisberg. 2003. Odor, total reduced sulfur, and ammonia emissions from animal housing facilities and manure storage units in Minnesota. *Applied Engineering in Agriculture* 19(3), pp. 347-360.

Hayes, E.T., T.P. Curran, and V.A. Dodd. 2005. Odour and ammonia emissions from intensive pig units in Ireland. *Bioresource Technology*. ARTICLE IN PRESS.

Lim, T.T. A.J. Heber, J.Q. Ni, A.L. Sutton, and D.T. Kelly. 2001. Characteristics and emission rates of odor from commercial swine nurseries. *Transactions of the ASAE* 44(5), pp. 1275-1282.

Mol, G., and N.W.M. Ogink. 2002. Geuremissies uit de veehouderij II, Overzichtsrapportage 2000 – 2002. *Imag Rapport 2002-09*, Wageningen, pp. 63.

Oldenburg, J. 1989. Geruchs- und ammoniak-emissionen aus der tierhaltung [Odour and ammonia emission of livestock husbandry]. *KTBL-Schrift 333, Kuratorium für Technik und Bauwesen in der Landwirtschaft e.V. KTBL*, Darmstadt.

Verdoes, E.D., and N.W.M. Ogink. 1997. Odour emission from pig houses with low ammonia emission. *In Proc. International Symposium on Ammonia and Odour Control from Animal Production Facilities, 2. eds. J.A.M. Voermans and G. Monteny*. Rosmalen, pp. 317-325. The Netherlands: Dutch Society of Agricultural Engineering (NVTL).

Zhu, J., L. Jacobsen, D. Schmidt, and R. Nicolai. 2000. Daily variations in odor and gas emissions from animal facilities. *Applied Engineering in Agriculture* 16(2), pp. 153-158.

A Simple Estimate of Nitrogen Deposition Across a Region of Variable Ammonia Emission Density

W.P. Robarge[1] and J.T. Walker[2]
[1]North Carolina State University, Department of Soil Science, Raleigh, NC, USA, 27695-7619
[2]U.S. EPA , National Risk Management Research Laboratory,
Air Pollution Prevention and Control Division, Research Triangle Park, NC 27711

Abstract

Animal production in eastern North Carolina, particularly for swine and poultry, is centered primarily in six counties. Ammonia emission densities arising from animal production in these six counties ranges from > 5000 to 2000-3000 NH_3-N km^{-2} yr^{-1}. The ammonia emission density for the remainder of eastern North Carolina declines steadily moving away from these six counties. The fate and transport of this emitted ammonia and its potential impacts on air quality and nutrient sensitive ecosystems within the state and along the coast is of major concern. We have followed a three-step approach in deriving a simple estimate of ammonia-N deposition across this region of variable ammonia emissions density. First, over 5 years of monitoring ambient ammonia chemistry in eastern North Carolina using annular denuder technology has yielded insight into the distribution of gaseous ammonia and fine particulate ammonium (PM2.5) throughout the region. Yearly mean gaseous ammonia varies by a factor of 5 going from high to low regions of ammonia emissions. Fine particulate ammonium concentrations vary by a factor of 2 to 3. Both gaseous ammonia and fine particulate ammonium demonstrate seasonal differences in concentration, as does the association of fine particulate ammonium with chloride, nitrate and sulfate. Second, our knowledge of the atmospheric ammonia chemistry across the region, combined with micrometeorological studies of ammonia-N deposition to various vegetative canopies, is used to estimate dry deposition of ammonia-N. These estimates are combined with N wet deposition measurements from several National Atmospheric Deposition Program collectors located in eastern North Carolina. Together, these data yield an estimate of ammonia-N deposition across this region of variable ammonia emission density. This number is compared to the estimate of total ammonia-N emissions from animal production. Third, the estimate of ammonia-N deposition is contrasted to projections of N deposition in this region using multi-media computer models. This comparison is used to assess the need for further monitoring of ambient atmospheric ammonia chemistry in the region, and to determine to presence of possible error in the databases, such as the NADP data sets. This work is part of our continuing efforts to characterize ambient ammonia chemistry across eastern North Carolina and to predict changes in wet and dry deposition of nitrogen across the region with changes in density of animal production facilities and the imposition of alternative technologies to reduce ammonia emissions.

Introduction

During the past decade, much attention has been paid to the swine industry in North Carolina (NC) and its potential negative impacts on the environment (Aneja et al., 2001). Many of the specific criticisms against the industry have been centered on swine operations as concentrated sources of nitrogen (N) emissions. In the gas phase, these emissions occur primarily as ammonia (NH_3) that is released directly into the atmosphere. Release of the water-soluble phase (ammonium ion; NH_4^+) can occur when transported off site in runoff events or after conversion to nitrate (NO_3^-) in the soil. The amount of ammonia-N emitted from swine operations in NC has been the subject of considerable debate. As a consequence, most of the research effort in the state has been directed toward quantifying NH_3 emissions from the relatively large swine production facilities located predominately in eastern North Carolina (Aneja et al., 2001). Relatively little effort has been directed to measuring or estimating the amount of ammonia emitted from swine operations that is actually being deposited in the state. Published reports have focused primarily on wet deposition (Walker et al., 2000; Whitall and Paerl, 2001). The fate and transport of this emitted ammonia and its potential impacts on air quality and nutrient sensitive ecosystems within the state and along the coast is of major concern. Discussed here is a portion of our efforts to derive a simple estimate of ammonia-N deposition derived from animal agriculture across this region of variable ammonia emissions density.

Ammonia Emission from Swine Operations

Figure 1 contains a schematic of a typical swine production facility in NC using a lagoon for waste treatment and land application for reuse of nutrients and disposal of excess water. Ammonia emissions can occur from each of the major components of this type of swine production facility. Ammonia emissions arise from the conversion of organic-N in urine and fecal matter into ammonium. This occurs on the floor of the housing units, within the pits underlying the houses, and within the lagoon. Conversion of organic-N is largely the result of microbial activity, which means that the rate of conversion to ammonium is temperature dependent. Temperature is considered a major controlling variable in regulating ammonia emissions from swine operations, thus emissions of ammonia (especially from lagoons and spray fields) are seasonal in nature (Visscher, et al., 2002).

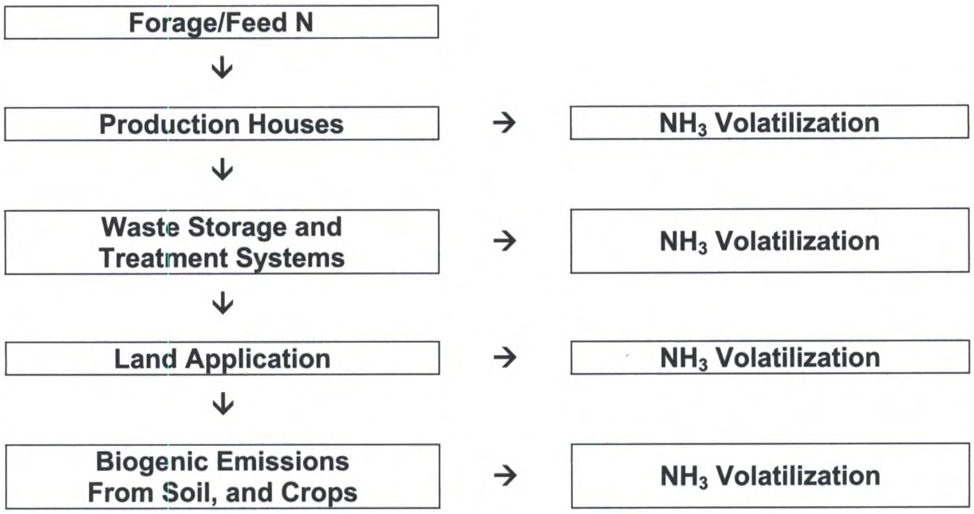


Figure 1. Major routes for ammonia emissions from a typical swine production facility in North Carolina. (Adapted from Aneja et al., 2001).

Recent estimates of emission factors published for swine operations in NC range from 7 +/- 2 kg NH3-N yr^{-1} hog^{-1} (Doorn et al., 2002) to 5.4 kg NH3-N yr^{-1} per finished hog (Aneja, et al., 2003). (Note: NH3-N means the units refer to the mass of N emitted as ammonia). Assuming a steady-state population of approximately 9.3 million swine in NC (the vast majority of which are finished hogs), this range in emission factors translates into a range of between 50,220,000 to 65,100,000 kg NH3-N emitted per year from swine operations. In English units this range translates to 55,240 to 71,610 tons of NH3-N per year. To put these numbers into proper perspective requires an estimate of the N excreted per year by the swine population in NC. Table 2 is an example calculation of the N excreted by the NC swine population in 1999: 102,000,000 kg of N. Contrasting the two sets of numbers suggests that between 50 to 64% of the N excreted by swine is emitted as ammonia. Doorn et al. (2002) reports an emission factor of 9 kg NH3-N/yr per hog which would mean 83,700,000 kg (92,100 tons) or > 80% of N excreted is emitted as ammonia. This number seems unreasonably high given the N applied to crops as fixed by required nutrient management plans, and the accumulation of N in the sludge in lagoons.

Table 2. Estimate of N excreted from swine production in 1999 for North Carolina (Sources: W. Cherry, NC Pork Council: Dr. T. Van Kempen, formerly An. Sci., NCSU, Raleigh, NC; personal communications).

Swine Production	N Excretion Factor	N Excreted (lbs N)	N Excreted (kg N)
14,358,900 (market hogs)	14.8 lbs N per pig	212,500,000	96,300,000
3,000,000 (20-50 wt class; sent out of state)	4 lbs N per pig	12,000,000	5,400,000
TOTAL=			101,700,00

Nitrogen Deposition

Wet Deposition

In the United States, the composition of rainfall has been monitored since 1978 by the National Atmospheric Deposition Program (NADP), which is now part of the National Trends Network. North Carolina is fortunate to have 9 active NADP active collectors, one of which is in the center of Sampson County at the Clinton Horticultural Crops Research Station (latitude 35.0258, longitude -78.2783, elevation 41 m; NADP designation NC35). Another is located in Bertie County, just outside of Lewiston, NC at the Peanut Belt Research Station (latitude 36.1325, longitude -77.1708; elevation 22 m; NADP designation NC03). The Lewiston location is of significance because it is in a region with a fairly low density of swine operations, and is also downwind (located northeast) of the six counties in NC with the highest density of swine operations.

Summaries for the wet deposition of NH4-N and NO3-N for 1980 to 2002 for the Sampson County site (NC35) and the Bertie County site (NC03) are provided in Tables 3 and 4. The data have been converted to mass of N as the respective chemical species to allow summation of terms to estimate total inorganic N inputs (NH4-N + NO3-N) per year as well as to calculate the percent contribution to total inorganic N from NH4-N. For the Sampson Co. site, there is an obvious increase in the wet deposition of NH4-N beginning in 1995 (Table 4), which corresponds to the period of rapid increase (1989 – 1997) in the swine population in eastern NC. There is no corresponding increase in NO3-N deposition, which appears to have remained constant for the past 22 years. There is no similar trend in the NH4-N data from Bertie Co. (Table 3). The similarity in NO3-N annual summaries between the two locations indicates that failure to not see an increase in NH4-N annual deposition at the Bertie Co. site is not due to an artifact in total volume of rainfall measured between the two locations.

Table 3. Summary for wet deposition of NH4-N and NO3-N recorded by the National Atmospheric Deposition Program Collector (NC35) in Sampson Co., NC (mean +/- std. dev.).

Period	NH4-N (kg N/ha)	NO3-N (kg N/ha)	Total N (kg/ha)	% NH4-N of Total N
1980-1989	1.52	2.26	3.77	40.0
	0.30	0.31	0.57	3.8
1998-2002	3.70	2.25	5.94	62.4
	0.40	0.41	0.77	2.5

The data in Tables 3 and 4 provide a number of insights as to the fate and transport of N from swine farms in eastern NC. First, as evident in the data summary for the years 1980-1989 for Sampson Co. (Table 3), and the data summary for the Bertie Co. site (Table 4), atmospheric deposition in general has been responsible for ~4 kg N ha^{-1} (as NH4-N and NO3-N) across eastern NC since 1980, before there was a significant increase in the swine population or change in style of management. Of this amount, ~1.5 kg N ha^{-1} yr^{-1} has been in the form of NH4-N. Second, in the region of the state with the highest density of swine operations, there has been an increase of NH4-N deposition of ~2.2 kg N ha^{-1}, or roughly a doubling of the

NH4-N deposition. This doubling is not evident at the Bertie Co. site nor is it present at other NADP sites in NC (data not shown). Third, since 1998, the amount of NH4-N deposition at the Sampson Co. site has remained fairly constant, demonstrating no more variation in annual yearly totals then is evident for the period 1980-1989, or for comparable periods at the Bertie Co. site. This suggests that the source of NH4-N influencing rainfall events in Sampson Co. has remained relatively constant since 1998. Whitall and Paerl (2001) monitored the composition of rainfall in the Neuse River Basin from 1996-1999 and concluded that counties with the highest density of swine operations, or counties immediately adjacent to these counties, did in fact experience an increase in the NH4-N content of the measured rainfall. Counties within the basin that did not meet these criteria did not experience an increase in NH4-N deposition.

Table 4. Summary for wet deposition of NH4-N and NO3-N recorded by the National Atmospheric Deposition Program Collector (NC03) in Bertie Co., NC (mean +/- std.dev.).

Period	NH4-N (kg N/ha)	NO3-N (kg N/ha)	Total N (kg/ha)	% NH4-N of Total N
1980-1989	1.45	2.41	3.86	37.2
	0.42	0.35	0.68	5.7
1998-2002	1.77	2.23	4.00	44.3
	0.28	0.33	0.60	1.8

Dry Deposition

Dry deposition of ammonia is partly a function of the ambient atmospheric ammonia concentration. In 1998, the authors began monitoring ambient atmospheric concentrations of ammonia and ammonium-based aerosols in Sampson Co., NC at the Clinton Horticultural Crops Research Station using annular denuder technology (Robarge et al., 2002). In succeeding years, this effort was expanded to include other areas in eastern North Carolina (Walker et. al., 2004), including the Bertie Co. location of NADP collector NC03 (Walker et al., 2006). Average NH_3 concentration from April, 2002 to October, 2003 at the Bertie Co. site was 1.05 $\mu g\ m^{-3}$ as opposed to an average concentration of 5.59 $\mu g\ m^{-3}$ measured in 2000 in Sampson Co. Mean gas concentrations were 0.30, 0.91, 0.43, and 3.17 $\mu g\ m^{-3}$ for HCl, HNO_3, HONO, and SO_2, respectively during the same period and differed little from similar measurements made in Sampson Co. in 2000. These observations suggest that aside from NH_3, regional gas chemistry across eastern NC may be relatively constant in atmospheric composition at ground level. Average NH_4^+ concentration at the Bertie Co. site was 0.74 $\mu g\ m^{-3}$, and the total concentration of inorganic ($NH_4^+ + NO_3^- + SO_4^= + Cl^-$) particulate matter with an aerosol diameter < 2.5 μm ($PM_{2.5}$) was 4.88 $\mu g\ m^{-3}$ from April 2002 to October 2003. These numbers are approximately half the values recorded in Sampson Co. in 2000 (1.76 $\mu g\ m^{-3}$ for NH_4^+ and 7.96 $\mu g\ m^{-3}$ for $NH_4^+ + NO_3^- + SO_4^= + Cl^-$) (Walker et. al., 2004). The relatively high concentrations of NH_3 observed at the Sampson Co. site are consistent with a relatively large source of ammonia-N emissions in the immediate region surrounding the site. The data are also consistent with the wet deposition data collected by NADP collector NC35 at the Sampson Co. site, and support the assumption of a gradient in dry deposition across eastern NC moving from areas with the highest density of ammonia emissions (> 5000 kg NH_3-N $km^{-2}\ yr^{-1}$; Duplin and Sampson Co.) to the relatively lowest (500 to 750 kg NH_3-N $km^{-2}\ yr^{-1}$, Bertie Co.).

A Simple Mass Balance for Emissions

The rainfall data from the NADP collector in Sampson Co., together with the extensive observations of ambient atmospheric ammonia and ammonium-based aerosol concentrations made at the same location using annular denuder technology, make a strong argument that a significant portion of the estimated 50,220,000 to 65,100,000 kg NH3-N emitted per year from swine farms in NC is being deposited locally in the counties with the highest density of animal operations in the state. However, the question still remains of whether this enhanced amount of local deposition can account for the majority NH3-N emitted. If not, then the rainfall data from other NADP collectors in the state, such as Bertie Co., NC, would suggest that the remaining NH3-N emissions not deposited locally are in fact not being deposited in the remainder of eastern NC. The following is a simple mass balance calculation in an attempt to address this issue.

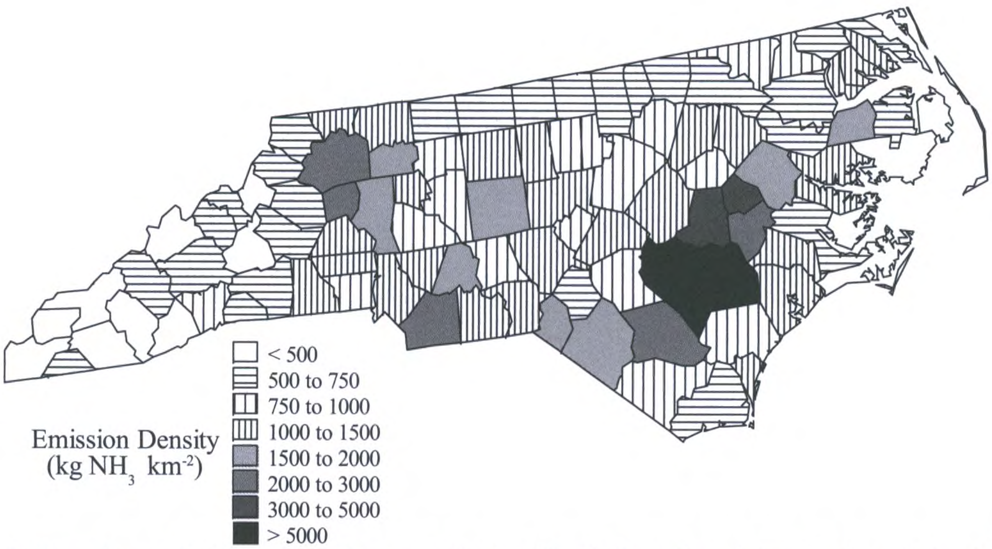


Figure 2. County-scale NH_3 emission density for North Carolina. Livestock activity data represent 2000 levels. All other activity data represent 1996 levels.

Figure 2 is a map of eastern NC highlighting the six counties with the highest density of swine operations in the state. For the simple mass balance calculation, the Coastal Plain region of the state was divided into two separate land areas – one area included the six counties with the highest density of animal operations (Bladen, Duplin, Green, Lenoir, Sampson and Wayne) and the other area included the remaining counties in the Coastal Plain. Note that the area with the six counties includes several of those observed by Whitall and Paerl (2001) to have relatively high amounts of NH4-N wet deposition. According to NC Department of Environment and Natural Resources records, the six county region contains approximately 1700 registered swine operations, and approximately 70% of the swine produced in NC. Selecting an average value of 57,400,000 kg for total NH3-N emissions from swine farms in the state yields an estimate of 40,200,000 kg of NH3-N emitted per year by the six designated counties.

The designated six counties occupy a region of approximately 3,660,000 acres or 1,481,100 hectares. From Table 3, the *additional* NH4-N observed in wet deposition at the Sampson Co. site for the period 1998-2002 averaged 2.2 kg NH4-N ha^{-1} yr^{-1} (1998-2002 average corrected for the 1980-1989 average). Whitall and Paerl (2001) conducted similar calculations for their more limited three-year data set for the Neuse River Basin and derived *additional* NH4-N estimates in wet deposition of 3-5 kg NH4-N ha^{-1} yr^{-1}. A value of 3.5 kg NH4-N ha^{-1} yr^{-1} was selected for the purposes of this calculation, which yields a total of 5,184,000 kg NH4-N (5700 tons of NH4-N) in wet deposition of *additional* N. This is roughly only 13% of the total assumed emissions from the six county area. The remaining counties in the Coastal Plain occupy an area of approximately 11,280,000 acres or 4,563,300 hectares including the sound and estuaries. Of the remaining 30%, 25% of the swine produced in NC were assumed to be in this area and emit approximately 14,400,000 kg of NH3-N per year. Reviewing the calculations in Table 4 indicates that the amount of additional NH4-N detected by the NADP collector at the Bertie Co. site (downwind of the six-county area) was 0.8 to 0 kg NH4-N ha^{-1} yr^{-1} depending your interpretation of the uncertainty associated with the average values. Assuming that NH4-N deposition would taper off moving away form the six county region, a value of 1 kg NH4-N ha^{-1} yr^{-1} was chosen for *additional* N in wet deposition, which yields a total of 4,563,300 kg NH4-N (5000 tons of NH4-N) in wet deposition of *additional* N. This is roughly 32% of the total assumed emissions from the remainder of the Coastal Plain region.

To complete the simple mass balance calculation for emissions requires an estimate of dry deposition within the two selected areas. It can be argued that dry deposition within the six county area would be enhanced due to the elevated levels of ambient ammonia and ammonium-based aerosol concentrations. As a first approximation, therefore, dry deposition was set equal to twice the wet deposition estimate of 3.5 kg NH4-N ha^{-1} yr^{-1} of *additional* N, or 10,368,000 kg NH4-N (11,400 tons of NH4-N). This brings the estimate of total *additional* N deposited in the six county area to 15,500,000 kg (17,100 tons), which only accounts for 38.5% of the estimated emissions of N from the swine farms in the six counties. For the remaining portion of the Coastal Plain, dry deposition was set equal to wet deposition, meaning that 64% of the emissions are accounted for in this area. Combining these estimates yields a total of 24,600,000 kg NH4-N for the calculated *additional* N from both wet and dry deposition across the Coastal Plain of North Carolina. Depending on selection of the emissions factor, this sum only accounts for 38 to 49% of the estimated emissions for NH_3 from large-scale swine operations in eastern NC.

Conclusions

It is acknowledged that these calculations are only approximate and cannot differentiate from other agricultural sources of ammonia in eastern NC. However, emission inventories in the state indicate that swine production is the dominant source of ammonia emissions in this area, and that changes in swine density of production are the most consistent with observed changes in wet deposition chemistry. These calculations also reflect actual field observations that have been made in eastern NC during the past five years, as well as general rules of thumb that have been derived from deposition studies on various ecosystems. As such, the attempted simple mass balance calculation can only account for ~ 40 to 50% of the estimates NH_3 emissions from swine. The remaining emissions, if real, must be transported outside of the Coastal Plain, either West into central NC, along the East Coast, and/or East out to the Atlantic Ocean.

References

Aneja, V.P., Nelson, D.R., Roelle, P.A., Walker, J.T., and Battye, W. 2003. Agrciultural ammonia emissions and ammonium concentrations associated with aerosols and precipitation in the southeast United States. *J. Geophys. Res.* 108(D4):4152.

Aneja, V.P., Roelle, P.A., Murray, G.C., Southerland, J., Erisman, J.W., Fowler, D., Asman, W.A.H., and Patni., N. 2001. Atmospheric nitrogen compounds II: emissions, transport, transformation, deposition and assessment. *Atmos. Environ.* 35:1903-1911.

Doorn, M.R.J., Natschke, D.F. and Meeuwissen, P.C. 2002. Review of Emission Factors and Methodologies to Estimate Ammonia Emissions from Animal Waste Handling. U.S. Environmental Protection Agency, RTP, NC. EPA-600/R-02-017.

Robarge, W. P., Walker, J. T., McCulloch, R. B., Murray, G., 2002. Atmospheric concentrations of ammonia and ammonium at an agricultural site in the southeast United States. *Atmos. Environ.* 36:1661-1674.

Visscher, A. De, Harper, L.A., Westerman, P.W., Liang, Z., Arogo,J., Sharpe, R.R., and Van Cleemput, O. 2002. Ammonia emissions from anaerobic swine lagoons: Model development. *J. Applied Meteorology.* 41:426-433.

Walker, J.T., Robarge, W.P., Shendrikar, A., and Kimball, H. 2006. Monitoring ambient ammonia chemistry in an agricultural region with a low density of animal production. Proceedings. Workshop on Agricultural Air Quality: State of the Science. Bolger Conference Center, Potomac, Maryland. June 5-8, 2006.

Walker, J.T., D. R. Whitall, W. Robarge, Paerl, H.W. 2004. Ambient ammonia and ammonium aerosol across a region of variable ammonia emission density. *Atmos. Environ.* 38:1235-1246.

Walker, J. T., Nelson, D., Aneja, V. P., 2000. Trends in ammonium concentration in precipitation and atmospheric ammonia emissions at a Coastal Plain site in North Carolina, USA. *Environ. Sci. Technol.* 34, 3527-3534.

Whitall, D. R. and Paerl, H. W. 2001. Spatiotemporal variability of wet atmospheric nitrogen deposition to the Neuse River estuary, North Carolina. *J. Environ. Qual.* 30(5):1508-151.

Monitoring Ambient Ammonia Chemistry in an Agricultural Region with a Low Density of Animal Production

Wayne Robarge[1], John Walker[2], Arun Shendrikar[3], and Hoke Kimball[3]
[1]North Carolina State University, Department of Soil Science, Raleigh, NC, USA, 27695-7619
[2]U.S. EPA , National Risk Management Research Laboratory,
Air Pollution Prevention and Control Division , Research Triangle Park, NC 27711
[3]North Carolina Department of Environment and Natural Resources,
Division of Air Quality, Raleigh, NC 27626

Abstract

We present several years of ambient ammonia (NH_3), ammonium (NH_4^+), hydrochloric acid (HCl), chloride (Cl^-), nitric acid (HNO_3), nitrate (NO_3^-), nitrous acid (HONO), sulfur dioxide (SO_2), and sulfate ($SO_4^=$) concentrations at a rural site in the Coastal Plain region of North Carolina. The site is located at the Peanut Belt Research Station (latitude 36.1325, longitude -77.1708; elevation 22 m) in Bertie Co., NC. The total NH_3 emission density of the county ranges from 500 to 750 kg NH_3-N km^{-2} yr^{-1}, and the county itself is approximately 80 km northeast of the region with the highest density of ammonia emissions (> 5000 kg NH_3-N km^{-2} yr^{-1}; Duplin and Sampson Co.) in eastern North Carolina. Average NH_3 concentration from April, 2002 to October, 2003 was 1.05 $\mu g\ m^{-3}$ as opposed to an average concentration of 5.59 $\mu g\ m^{-3}$ measured in 2000 in Sampson Co. Mean gas concentrations were 0.30, 0.91, 0.43, and 3.17 $\mu g m^{-3}$ for HCl, HNO3, HONO, and SO2, respectively during the same period and differed little from similar measurements made in Sampson Co. in 2000. These observations suggest that aside from ammonia, regional gas chemistry across eastern NC may be relatively constant in atmospheric composition at ground level. Average NH_4^+ concentration at the Bertie Co. site was 0.74 $\mu g\ m^{-3}$, and the total concentration of inorganic ($NH_4^+ + NO_3^- + SO_4^= + Cl^-$) particulate matter with an aerosol diameter < 2.5 μm ($PM_{2.5}$) was 4.88 $\mu g\ m^{-3}$ from April 2002 to October 2003. These numbers are approximately half the values recorded in Sampson Co. in 2000 (1.76 $\mu g\ m^{-3}$ for NH_4^+ and 7.96 $\mu g\ m^{-3}$ for $NH_4^+ + NO_3^- + SO_4^= + Cl^-$). The cumulative frequency distributions for aerosol sulfate for Bertie Co. from 2002-2003 versus those from Sampson Co. in 2000 suggest that the measured difference in ammonium-based aerosol mass < 2.5 μm between the two counties is due to differences in both sulfate and ammonium in the aerosol fraction. This work is part of our continuing efforts to characterize ambient ammonia chemistry across eastern North Carolina and to predict changes in wet and dry deposition of nitrogen across the region with changes in density of animal production facilities and the imposition of alternative technologies to reduce ammonia emissions.

Introduction

Loss of nitrogen through volatilization of ammonia (NH_3) from animal wastes represents a potential source of nitrogen loading via wet and dry deposition to surrounding terrestrial and aquatic ecosystems (Walker et al., 2000). This may be particularly true in eastern North Carolina (and especially in Sampson and Duplin counties) where there is a relatively high density of large-scale swine and poultry production facilities. Fate and transport of NH_3 emitted from effluent lagoons, housing units, or land application of animal wastes (effluent or litter) is dictated in part by the partitioning between ammonia and ammonium aerosols in the atmosphere. Ammonium aerosols form as a result of an acid-base reaction between NH_3 (a base) and acidic atmospheric species such as nitric acid (HNO_3), hydrochloric acid (HCl) or sulfur dioxide (SO_2). Ammonia has a relatively short residence time in the atmosphere, especially as compared to ammonium aerosols. Thus conversion of atmospheric NH_3 to ammonium aerosols favors transport of nitrogen lost by volatilization of NH_3 from animal wastes away from the immediate source area (such as portions of eastern North Carolina), enhancing the potential for its deposition in adjoining river basins or even across state boundaries.

Conversion of atmospheric NH_3 to ammonium aerosols also has broader implications for human health. In 1997, the U.S. EPA revised the national ambient air quality standard (NAAQS) for particulate matter (PM) to include specific rules for fine aerosol [$PM_{2.5}$ (d_p < 2.5 μm)] (US EPA, 1997). The revision set a standard of 15 $\mu g\ m^{-3}$ for the 3-year average of annual arithmetic mean concentrations, and a 24-hour standard of 65 μg m^{-3} based on the 3-year average of 98th percentile concentrations. In addition to adverse effects on human health, which form the basis for these standards, elevated atmospheric particulate concentrations degrade visibility and air quality in general.

The concentrations of NH_3 and ammonium aerosols in the atmosphere can be determined using annular denuder technology (Robarge et al., 2002). A denuder, in its most simple form, is a cylindrical tube coated with a reagent that selectively samples a specific gaseous component as air is drawn through the tube. Air is sucked through the denuder tubes, or series of tubes arranged in sequence and coated with different reagents, for a certain period of time and the sample volume is measured by a gas meter or estimated from the sample time and air flow, the later usually being controlled by a critical orifice. After sampling, the coating of the denuder is extracted and analyzed for the compound of interest. The coating used for a denuder tube is selected for the trace gases of interest, is stable under ambient temperature and changes in relative humidity, and is compatible with commonly used analytical techniques. Annular denuders represent an improvement over the basic cylindrical denuder tube design allowing for sampling at higher flow rates of air (Purdue, 1992). Due to their sampling rates, annular denuders can be adapted relatively easily for application in monitoring. They can also be readily adapted to other studies such as measuring dry deposition. Addition of a filter pack at the end of a series of annular denuder tubes allows the capture of fine particulates and aerosols that have been separated from the trace gases in the air flowing through the denuder tubes. Thus annular denuder technology, when properly configured, can provide a substantial amount of information about the concentrations of a number a trace gases in the atmosphere.

This project was designed to enhance the efforts already underway to monitor the atmospheric concentrations of ammonia gas and ammonium aerosols in eastern North Carolina. Specifically, annular denuder technology was selected to continue to provide daily (day/night 12 hour sampling periods) measurements of ambient ammonia/ammonium concentrations downwind (summer winds) of the relatively high density of large-scale animal production operations located in primarily Bladen, Duplin, Sampson and Wayne counties. These measurements compliment daily annular denuder measurements made in 1999 and 2000 at the Clinton Horticultural Crops Research Station (latitude 35.0258, longitude -78.2783, elevation 41 m) located in Sampson County. Presented in this paper are results for annular denuder data collected for the periods April 29, 2002 – August 4, 2002, and December 12, 2002 – October 12, 2003. Chemical analyses for the gaseous species has been completed and a summary of the data is included in this report together with previously published data from 1999 and 2000 (Robarge et al., 2002; Walker et al., 2004).

Methods

An annular denuder system (URG Corporation, Chapel Hill, NC) purchased was installed at the Peanut Belt Research Station (36.1325 degrees latitude, -77.1714 degrees longitude, elevation 22 meters) located North of Lewiston, North Carolina in Bertie County. The annular denuder system allows for 4 days (day and night cycles) of unattended operation. The computerized mass flow controller and associated mast to hold the annular denuder tubes were installed inside the fenced compound located near the headquarters building of the research station. This fenced in area also contains a National Atmospheric Deposition Program monitor (NC03) positioned at the research station. The research station is equipped with a fully functional weather station run by the State Climate Office of North Carolina.

The annular denuder system operated on 12-hour day- night cycles (0600 to 1800 hours day-cycle; 1800 to 0600 hours night-cycle), yielding 12-hour average concentrations. The system consisted of the following components: two 3-channel annular denuder tubes (30 mm x 242 mm length) coupled together to form one denuder tube combination, a total of 8 denuder tube combinations arranged in sequence along two 4-channel manifolds, two #30 aluminum Teflon coated cyclones (10 Lpm, < 2.5 micron cutoff) attached to each manifold, eight 2-stage Teflon filter packs with one attached to the bottom of each denuder tube combination, and a computerized sampling pump (8 channel) with accompanying mass flow controller. One of the annular denuder tubes in a denuder tube combination is coated with sodium carbonate and is used to trap acidic gases (HNO_2, HNO_3, HCl and SO_2). The second denuder tube in the denuder tube combination is coated with phosphorous acid and is used to trap NH_3. The Teflon filter pack contains a 2 micron Teflon filter (47 mm diameter) followed by a 0.8 micron Nylon filter. The Teflon filter essentially traps all of the particulates and aerosols < 2.5 microns in size that pass through the preceding aluminum Teflon-coated cyclone and two annular denuder tubes (intake height set at 2.65 m). The second Nylon filter is necessary to ensure quantitative recovery of ammonium because of potential secondary reactions that may occur on the Teflon filter. Use of the mass flow controllers allows precise control of air flow through the annular denuder tube assembly such that the total volume of air sampled is equal to the sampling period times the set flow-rate (typically 10 Lpm). The 8 channel sampling pump allowed unattended alternate day and night cycle sampling for a total of 4 days. During the initial sampling period (April 29, 2002 – August 4, 2002), the system was set to collect samples every other day. At the start of the second sampling period (December 12, 2002 – October 12, 2003), the sampling sequence was switched to four sequential days (Wednesday through Sunday) to facilitate maintenance of the site.

The annular denuder system was maintained by weekly trips to the research station from North Carolina State University at Raleigh, NC. These trips were typically made either Monday or Tuesday of the following week. Thus unexposed tubes were deployed for no longer than two days before start of sample collection (Wednesday 1800 hours), and recovered typically within one day following the end of a sampling session (Sunday 1800 hours). Flow rates were checked (BIOS DryCal DC-Lite Primary Flow Meter, Butler, NJ) bimonthly. The denuder tubes were color-coded using electrical tape to facilitate assembling the various denuder tube – filter pack combinations on site. The intake lines from the computerized mass flow controller were also color coded to ensure proper connection of denuder tube assemblies with proper position in the programmed sampling sequence. When assembled, the various color-coded tubes and connections provided a rapid visual check with a color chart carried by the field staff to ensure that all tubes were assembled and deployed properly.

Exposed annular denuder tubes were extracted within 24 hours of being returned to the research laboratory at North Carolina State University. The sample extracts and exposed filters were transferred to 20 mL plastic scintillation vials with screw cap (plastic liners) lids and stored at 4°C until analysis. All handling of the denuder tubes and filter packs (both in the field and in the laboratory) was done using gloves to minimize contamination, especially from chloride and sodium. Chemical analysis of extracted samples was performed by the Analytical Service Laboratory located in the Department of Soil Science, NCSU (Dr. W. Robarge, Supervisor). Standard colorimetric procedures were used to determine soluble NH4-N and NO3-N using a flow injection autoanalyzer (Lachat Model 8000 3-channel Autoanalyzer). Soluble Cl-, NO_2^- and $SO_4^=$ were determined using ion chromatography (Dionex Model 500 and Model 160 Ion Chromatographs with autosamplers).

Results and Discussion

Gaseous Species

The summary statistics for HCl, HNO_3, HONO, NH_3 and SO_2 as measured by annular denuder technology at the Lewiston, NC site are provided in Table 1. Also provided for comparison purposes, are data for the same gaseous species measured at the Clinton Horticultural Crops Research Station in 1999 and 2000. For HCl, HNO_3, HONO and SO_2, the measured concentrations at the Lewiston site are very similar to those measured in the center of Sampson County in 1999 and 2000 (Robarge et al., 2002; Walker et al., 2004). The agreement in mean values and standard deviations is exceptionally close, even though the absolute number of observations above the corresponding detection limits is substantially less than the data set for the Clinton site. The difference in total number of observations probably also explains why the recorded 12-hour averaged maximum values for the four gases are all higher at the Clinton site. Even so, the maximum values between the Clinton and Lewiston site differ by less than an order of magnitude, suggesting that there is little difference in the ambient concentrations of these gases measured at Lewiston in 2002/2003 compared to the Clinton site in 2000.

Similarity between the Lewiston and Clinton sites is also demonstrated by comparing the cumulative frequency distribution of SO_2 (Figure 1). Eighty percent of the observations at both sites are less than 6 ug SO_2 m^{-3}, although the overall distribution for the Lewiston data is shifted to lower concentrations. This shift amounts to a difference of roughly 1.0 ug SO_2 m^{-3} which may reflect some differences in local sources of SO_2, and/or differences in meteorology between years, since 2003 was a wet year compared to 2000.

Table 1. Summary statistics for atmospheric gases measured using annular denuder technology. Data for Lewiston site reflects two collection periods: April 29, 2002 to August 4, 2002, and December 12, 2002 to October 12, 2003. Data for Clinton site reflects collection period of January 1, 2000 to December 4, 2000.

Gaseous Species	Monitoring Site	N	Mean	S.D.	Min.	Max.
				----- ug/m3 -----		
HCl	Clinton	1168	0.43	0.45	0.10	5.96
	Lewiston	327	0.30	0.24	0.01	2.17
HNO3	Clinton	1168	0.94	1.02	0.09	7.34
	Lewiston	365	0.91	0.86	0.13	6.07
HONO	Clinton	1168	0.43	0.58	0.67	4.10
	Lewiston	178	0.43	0.33	0.09	1.91
NH3	Clinton	1168	5.59	5.13	0.06	36.4
	Lewiston	454	1.05	0.69	0.03	3.6
SO2	Clinton	1168	3.90	3.71	0.05	33.0
	Lewsiton	465	3.17	3.61	0.01	20.7

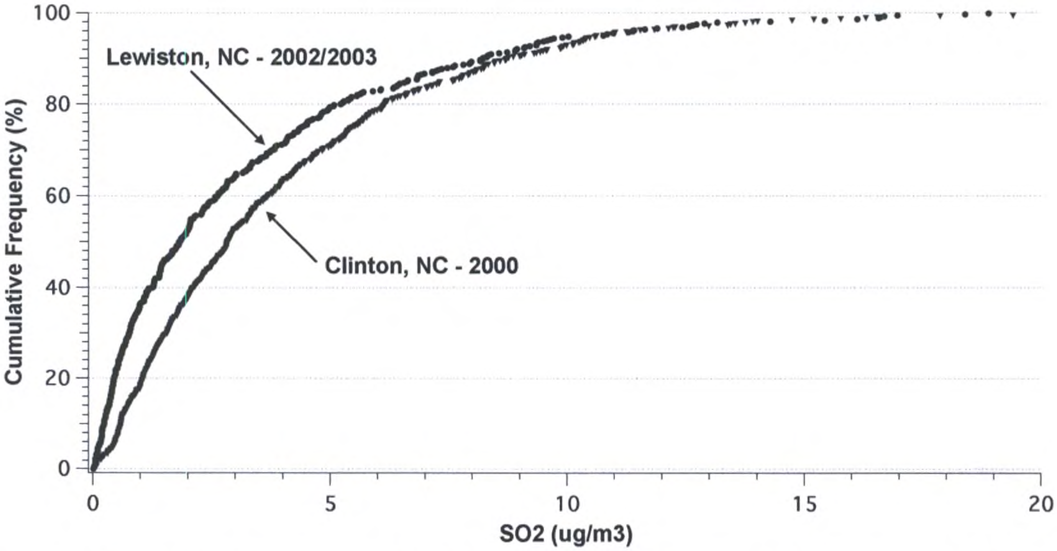


Figure 1. Cumulative frequency distribution for SO_2 measured using annular denuder technology at Lewiston, NC in 2002/2003 and Clinton, NC in 2000.

The only measured gaseous species using annular denuder technology that demonstrates a major difference in 12-hour average concentrations between the Lewiston and Clinton sites is ammonia. The mean concentrations vary by a factor of 5 (Table 1) and the observed maximum concentrations vary by a factor of 10. The difference in observed ammonia concentrations is even more striking when the cumulative frequencies for the observations from the two sites are compared (Figure 2). For the Lewiston site, 90% of the measured 12-hour average concentrations fall below 2 ug NH3-N m^{-3}, whereas almost 80% of the observations recorded at the Clinton site in 2000 are above 2 ug NH3-N m^{-3}. It is doubtful that this difference in gaseous ammonia concentrations can be attributed to differences in meteorology between 2000 and 2002/2003. Rather, the differences are consistent with differences in local source terms for ammonia emissions at these two locations. One logical explanation for differences in local source terms for ammonia would be the differences in animal production densities in Sampson and compared to Bertie County. The total NH_3 emission density for Sampson County is 4800 kg NH3-N km^{-2} yr^{-1}. For Bertie County the NH_3 emissions density is < 400 kg NH3-N km^{-2} yr^{-1} which corresponds to a difference by a factor of approximately 10 and is consistent with the observed difference in 12-hour average ammonia concentrations (Table 1).

Ammonium-Based PM2.5

The summary statistics for ammonium-based PM2.5 (NH_4Cl, NH_4NO_3, NH_4HSO_4 and $(NH_4)_2SO_4$) as measured by annular denuder technology at the Lewiston, NC site are provided in Table 2. Also provided for comparison purposes, are data for the same aerosol species measured at the Clinton Horticultural Crops Research Station in 1999 and 2000. The data in Table 2 reflect the analyses completed for the samples obtained from the Lewiston site for the periods April 29, 2002 to August 4, 2002, and December 12, 2002 to May 18, 2003.

Based on the limited analyses completed to date, the 12-hour average concentrations of ammonium, nitrate and sulfate appear lower at the Lewiston site in 2002/2003 than at the Clinton site in 2000. This is especially true for ammonium and nitrate, which differ by a factor of 3 or more in both the calculated mean values and maximum values observed to date as compared to the 2000 Clinton dataset. The differences in sulfate concentrations between the two monitoring is less obvious, and could be due to the difference in total number of observations between the datasets. However, cumulative frequency plots of the sulfate data (Figure 3) suggests that there is in fact a consistent difference between the two datasets that would require a substantial shift in measured aerosol sulfate concentrations during the later part of the year at the Lewiston site in order to obtain parity in the mean values between the two sample populations. This would seem unlikely to happen. Indeed contrasting temporal plots of ammonium-based $PM_{2.5}$ between the two samples sites (data not shown) suggests that the overall variation in aerosol concentration remains constant throughout the year and that the differences in ammonium-based $PM_{2.5}$ concentrations between the Clinton site in 2000 and the Lewiston site in 2002/2003 will remain upon completion of the analyses of the remaining aerosol samples.

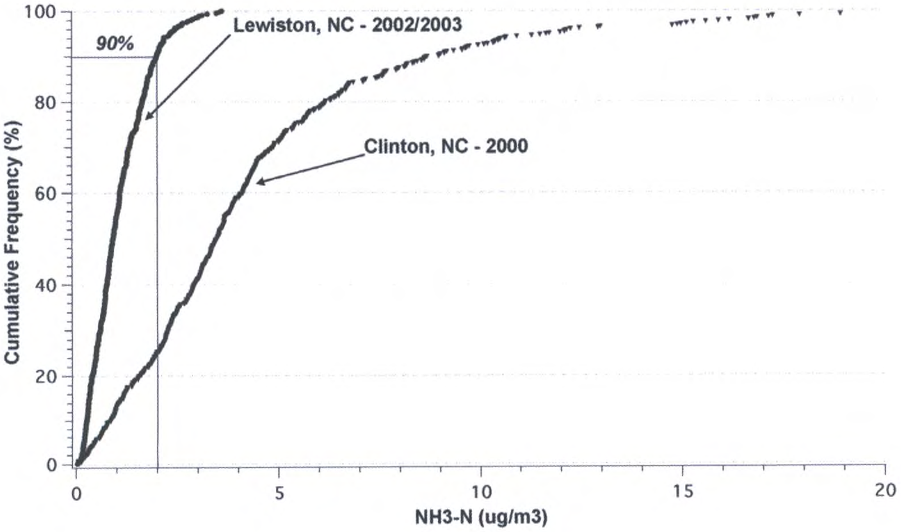


Figure 2. Cumulative frequency distribution for NH_3 measured using annular denuder technology at Lewiston, NC in 2002/2003 and Clinton, NC in 2000.

Table 2. Summary statistics for ammonium-based PM2.5 measured using annular denuder technology. Data for Lewiston site reflects two collection periods: April 29, 2002 to August 4, 2002, and December 12, 2002 to May 18, 2003. Data for Clinton site reflects collection period of January 1, 2000 to December 4, 2000.

Chemical Species	Monitoring Site	N	Mean	S.D.	Min.	Max.
				----- ug/m3 -----		
Cl	Clinton	1100	0.07	0.11	0.02	1.67
	Lewiston	127	0.34	0.37	0.05	2.58
NH4	Clinton	1100	1.76	1.38	0.06	10.1
	Lewiston	288	0.74	0.68	0.05	3.85
NO3	Clinton	1100	1.91	2.05	0.09	23.0
	Lewiston	211	0.20	0.21	0.03	1.38
SO4	Clinton	1100	4.22	3.65	0.05	20.7
	Lewiston	288	3.60	2.77	0.33	20.6

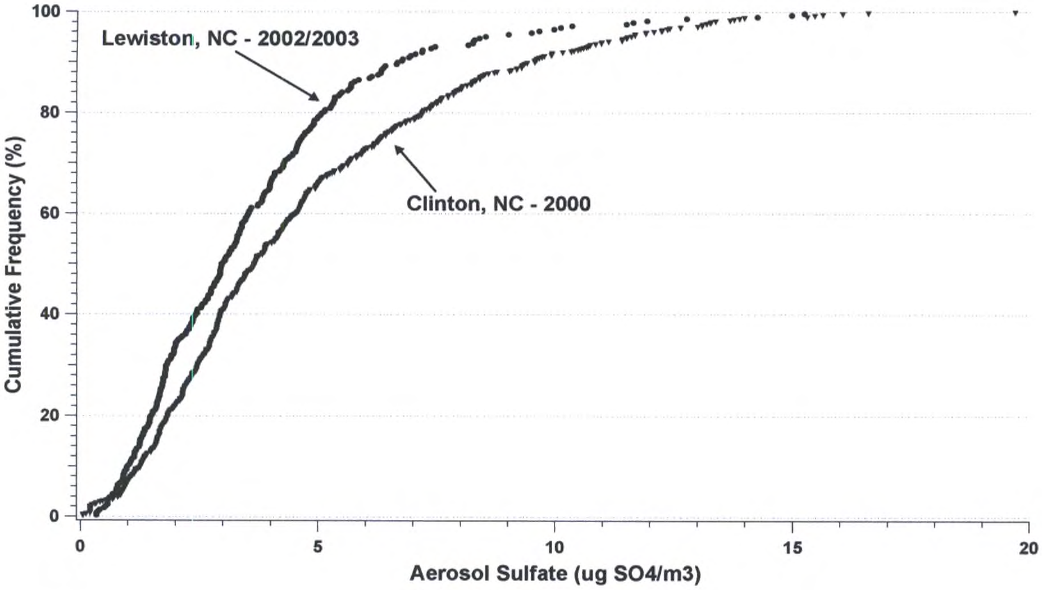


Figure 3. Cumulative frequency distribution for aerosol sulfate measured using annular denuder technology at Lewiston, NC in 2002/2003 and Clinton, NC in 2000.

Prior analysis of the annular denuder data collected at the Clinton site (Robarge et al., 2002; Walker et al., 2004) indicated that the average ammonium-based $PM_{2.5}$ mass concentration for the 2-year period 1999-2000 is 8.0 (±5.84) $\mu g\ m^{-3}$. On average, $SO_4^=$, NO_3^-, NH_4^+, and Cl^- account for 53, 24, 22, and 1% of total ammonium-based $PM_{2.5}$ mass. During both years, $SO_4^=$ becomes the primary constituent during warm months. Nitrate concentrations are higher during colder months. Chloride appears not contribute significantly to $PM_{2.5}$ at any time.

Given the results of studies which have investigated speciated $PM_{2.5}$ in the eastern U.S. (U.S. EPA, 1996; Tolocka et al., 2001), it may be reasonable to assume that, on average, ammonium-based aerosol accounts for less than 53% of total $PM_{2.5}$ mass. If this assumption is valid, the annual standard of 15 $\mu g\ m^{-3}$ may be routinely violated at the Clinton site, based on the observed 2-year average concentration (8.0 $\mu g\ m^{-3}$) and the distribution of daily values shown in Figure 8. During 2000, total $PM_{2.5}$ concentrations at 17 sites across the state of North Carolina ranged from 12.3 to 18.0 μg m^{-3} (NCDENR, 2002). Total $PM_{2.5}$ annual mean concentrations in Duplin, Lenoir, and Wayne counties, which are also within the six-county agricultural region described earlier, were 13.1, 12.7, and 15.9 μg m^{-3}, respectively, during 2000 (NCDENR, 2002). To date, the ammonium-based $PM_{2.5}$ concentration collected at the Lewiston site averages 4.6 (±3.4) μg m^{-3}. If the trends in measured ammonium-based $PM_{2.5}$ concentrations continue, it is unlikely that the $PM_{2.5}$ regulations will be violated at the Lewiston site.

Conclusions

The presence of excess NH_3 in the atmosphere influences the concentration and atmospheric chemistry of ammonium-based aerosol. At the Clinton site located in the center of Sampson County (estimated NH_3 emission density 4800 kg NH3-N km^{-2} yr^{-1}) prior investigations have shown that the relatively high concentration of atmospheric NH_3 establishes the condition of acid-gas limited ammonium aerosol formation during all seasons; i.e. no obvious seasonal trends in total ammonium-based $PM_{2.5}$ mass. At the Clinton site, ammonium-based aerosol concentrations are likely to be more sensitive to SO_2 and NO_x emissions than NH_3 emissions (Walker et al., 2006). The Lewiston site being monitored in this project provides an important test of this hypothesis. Ambient ammonium concentrations are significantly reduced suggesting more dependence of ammonium-based $PM_{2.5}$ on ammonia emissions or transport of ammonia into the site. Continued monitoring at the Lewiston site will provide much needed data on ambient N-containing inorganic gas and aerosol concentrations leading to a better understanding of the impact of emissions of ammonia on local and regional air quality.

Acknowledgements

We wish to acknowledge the support of the North Carolina Division of Air Quality, the North Carolina Pork Council, the National Pork Board, the North Carolina Water Resources Research Institute, and the US EPA for this project. We also sincerely appreciate the reliable field support of Lynette Mathis and Mark Barnes (North Carolina State University) throughout the course of the project.

References

NCDENR, 2002. North Carolina Department of Environment and Natural Resources, Division of Air Quality, Raleigh, NC

Purdue, L. J., 1992 Determination of the strong acidity of atmospheric fine-particles (< 2.5 μm) using annular denuder technology. Standard Method. Enhanced Method. EPA/600/R-93/037 (NTIS PB93-178234), U.S. EPA, Atmospheric Research and Exposure Assessment Laboratory, Research Triangle Park, NC, November.

Robarge, W.P., Walker, J.T., McCulloch, R. B., Murray, G., 2002. Atmospheric concentrations of ammonia and ammonium at an agricultural site in the southeast United States. *Atmos. Environ*. 36:1661-1674.

Tolocka, M. P., Solomon, P. A., Mitchell, W., Norris, G. A., Gemmill, D. B., Wiener, R. W., Vanderpool, R. W., Homolya, J. B., Rice, J., 2001. East versus West in th US: Chemical characteristics of $PM_{2.5}$ during the winter of 1999. *Aerosol Sci. Technol*. 34, 8-96.

U.S. EPA, 1997 *National Ambient Air Quality Standards for Particulate Matter: Final Rule.* Federal Register, Vol. 62, No. 138.

U.S. EPA, 1996. *Air Quality Criteria for Particulate Matter.* EPA/600/P-95/001aF. National Center for Environmental Assessment, Office of Research and Development, Research Triangle Park, NC.

Walker, J. T., Aneja, V. P., Dickey, D., 2000. Atmospheric Transport and Wet Deposition of Ammonium in North Carolina, USA. *Atmos. Environ.* 34, 3407-3418.

Walker, J.T., D. R. Whitall, W. Robarge, Paerl, H.W. 2004. Ambient ammonia and ammonium aerosol across a region of variable ammonia emission density. *Atmos. Environ.* 38:1235-1246.

Walker, J.T., Robarge, W.P., Shendrikar, A., and Kimball, H. 2006. Inorganic PM2.5 at an agricultural site. *Environ. Pollution.* 136: 258-271.

Carbon Dioxide Efflux from Poultry Litter Applied Soils under Conventional and Conservation Tillage Systems in North Alabama

Tiffany Roberson[1,2], E.Z. Nyakatawa[1], K.C. Reddy[1], and R.L. Raper[2]
[1]Alabama A & M University, Department of Plant and Soil Science, PO Box 1208, Normal, AL 35762.
[2]USDA-ARS, National Soil Dynamics Laboratory, 411 S. Donahue Drive, Auburn, AL 36832-5806.

Abstract

Elevated carbon dioxide (CO_2) levels from anthropogenic sources such as agriculture, use of fossil fuels, and deforestation pose problems such as global warming which can negatively impact natural ecosystems and food production. Intensive tillage can lead to C loss from agricultural soils. The objective of this study was to measure and document carbon dioxide loss and carbon storage in tilled and non-tilled cotton plots receiving poultry litter and ammonium nitrate as a nutrient source.

The atmosphere contained about 370 ppm CO_2 in 2002 as compared to about 280 ppm CO_2 before the industrial revolution (Brady and Weil, 2002) due to a phenomena known as global warming. It is speculated that future global CO_2 emissions to the atmosphere will increase from the current 7.4 billion tons of atmospheric carbon per year to about 26 billion tons per year by 2100 if major changes are not made in the way the world manages ecosystems and the way the world uses energy, particularly that from fossil fuels (Department of Energy, 2002). Agricultural ecosystems are important in the global context because of the large CO_2 flux to the atmosphere, and also because C storage in these systems can be sensitive to management practices such as tillage and cropping systems (West and Post, 2002). Soil conservation practices which can increase C levels include reduction in tillage such as use of mulch-tillage and no-tillage instead of conventional-tillage, planting cover crops such as winter rye [*Secale cereale* (L.)], and applying manures such as poultry litter (USDA Technical Note, 2001). Cover crops provide needed organic material which improves soil organic matter (Schertz and Kemper, 1994). Reddy et al. (2004) found that winter rye [*Secale cereale* (L.)] cover cropping increased surface residue cover by up to 35, 70, and 100% in CT, MT, and NT systems respectively. In addition to using conservation tillage, and growing cover crops, application of poultry litter can be used to increase soil carbon (Nyakatawa et al., 2001).

A field study to measure carbon dioxide efflux and C storage in tilled and non-tilled cotton [*Gossypium hirsutum* (L.)] plots receiving poultry litter as a nutrient source was conducted at the Tennessee Valley Research and Extension Center, Belle Mina, AL (34° 41' N, 86° 52'W) on a Decatur silt loam (clayey, kaolinitic thermic, Typic Paleudults) from 2003 to 2004. The treatments consisted of conventional-till, mulch-till and no-till tillage systems with winter rye [*Secale cereale* (L.)] cover cropping and ammonium nitrate and poultry litter sources of nitrogen. Conventional tillage was carried out using a moldboard plow in November and disking in April, followed by a field cultivator to prepare a smooth seedbed. In mulch tillage, a Lely rotary cultivator (Lely USA, Inc., Naples, FL) was used to destroy and partially incorporate crop residues to a depth of 5 to 7 cm in April before planting. No-tillage included planting directly into untilled soil using a Tye (Glascook Equipment and Sales, Veedersburg, IN) no-till planter. The winter rye cover crop variety 'Elbone' was planted in the fall and killed with glyphosphate herbicide about 7 d after flowering in the spring of 2003 and 2004. The time between killing of winter rye and cotton planting was about 6 wk each year to allow for total drying of residues. Ammonium nitrate was used at a single rate of 100 kg N ha^{-1}, the recommended N rate for cotton in the Tennessee Valley region. Poultry litter amounts necessary to supply 100 and 200 kg N ha^{-1} were calculated for application each year based on the N content of the poultry litter. Soil CO_2 efflux was measured using the LI-COR 6400 IRGA (LI-COR, Lincoln, NE) system attached to a LI-09 soil chamber (LI-COR, Lincoln, NE) and polyvinyl chloride (PVC) soil collars.

In 2003, CO_2 efflux in CT plots which averaged 4.4 $\mu mol\ m^{-2}\ s^{-1}$ was 16% and 63% significantly higher than in MT and NT respectively. In 2004, CO_2 efflux in mulch till which averaged 3.2 $\mu mol\ m^{-2}\ s^{-1}$ was 14% and 33% greater than CT and NT respectively. In 2003, total carbon was 4% and 13% higher in NT than in CT and MT plots. In 2004, total carbon was 10% and 7% higher in NT than in CT and MT plots

respectively. Plots receiving poultry litter released 18% more CO_2 than those receiving ammonium nitrate. Our study suggests that NT can reduce soil CO_2 emissions by 10.7 Mg CO_2 ha^{-1} during the cotton growing season of about 130 days. In summary, practicing conservation tillage methods such as no-tillage and mulch-tillage in addition to using recommended rates of poultry litter as a nutrient source and leaving plant residues on the soil surface, helps to sequester carbon in cotton production systems. Conservation tillage reduced CO_2 release to the atmosphere, and thus will in the long term play a significant role in reducing CO_2 to the atmosphere. Conservation tillage practices which promote carbon sequestration serve a vital role in the storage and release of total carbon from soil.

References

Brady, N.C, and R.R. Weil. The Nature and Properties of Soils. 13th ed. Prentice Hall. New Jersey, 2002.

Department of Energy. 2002. What is the potential for carbon sequestration? URL http://csite.esd.ornl.gov/rationale.html (visited 2006, February 1).

Nyakatawa, E.Z., C.K. Reddy, K.R. Sistani. 2001. Tillage, cover cropping, and poultry litter effects on selected soil chemical properties. *Soil and Tillage Research* 58: 69-79.

Reddy, K.C., E.Z. Nyakatawa, D.W. Reeves. 2004. Tillage and cropping systems: Tillage and poultry litter application effects on cotton growth and yield. *Agron. J.* 96:1641- 1650.

Schertz, D.L. and W. D. Kemper. 1994. Report on field review of NT cotton, Huntsville, AL September 22-23, 1994 by USDA/ARS/NRCS/Auburn University/Alabama A&M University.

U.S. Department of Agriculture (USDA) Technical Note. 2001. Long term agricultural management effects on soil carbon. Tech. Note No. 12. August, 2001.

West, T.O. and W.M. Post. 2002. Soil organic carbon sequestration rates by tillage and crop rotations: A global data analysis. *Soil Sci. Am. J.* 66:1930-1946.

GHG Fluxes from Agriculture and Land-Use Change in Russia

A.A. Romanovskaya
Institute of Global Climate and Ecology, Moscow, Russia

Abstract

Anthropogenic emissions of methane (CH_4) and nitrous oxide (N_2O) in agriculture and flux of carbon dioxide (CO_2) on abandoned agricultural land of Russian Federation are considered for the period from 1990 to 2003. Domestic livestock (major contributor) and rice cultivation are sources for CH_4. In 1990, its release to the atmosphere was 4993.3 Gg. In 2003 it dropped to 2247.5 Gg. N_2O emissions were estimated from manure management systems and agricultural soils taking into consideration input of nitrogen with synthetic fertilizers, animal wastes, decomposition of crop residues left on fields and cultivation of histosols. Crop residues are the main N_2O source in national agriculture. In 1990 total estimated anthropogenic N_2O emissions from agriculture were 713.1 Gg. In 2003 it decreased to 46% of 1990 level (328.0 Gg). The decrease in CH_4 and N_2O emissions is mainly caused by reduction in number of agricultural animals and poultry, decline in the area of managed land and decreased fertilizer use in the country. CO_2 fluxes from the abandoned agricultural lands that were under natural regrowth over the territory of Russia were estimated as annual changes in soil carbon stocks. Total area of abandoned agricultural land is 25.4 millions ha. The ROTHC model was employed in the investigation of carbon dynamics in soils. Average annual net-emissions over the territory of abandoned lands was 2,1 ± 1,8 Tg C/yr in 1990-1999. CO_2 removal from the atmosphere by soils was 5,7 ± 2,5 Tg C/yr on average in 2000-2003.

Introduction

The increase of greenhouse gas atmospheric emissions due to anthropogenic impact results in global warming that may cause negative implications on human activities and the general biosphere functioning. Carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) are the greenhouse gases whose human-induced emissions are subject to mandatory inventory under United Nations Framework Convention on Climate Change (UN FCCC, 1998). According to 1996 IPCC Revised Guidelines (IPCC, 1997) and Good Practice Guidance (IPCC, 2000), the agricultural sector is a source of anthropogenic CH_4 and N_2O emissions. Methane emissions come from enteric fermentation of domestic livestock, animal waste management systems (AWMS), and rice cultivation. Agricultural soils are considered the major anthropogenic source that accounts for more than 50% of global N_2O emission to the atmosphere (Khalil and Rasmussen, 1983; Bouwman et al., 2005). The intensity of N_2O formation and release to the atmosphere depends on content of soil nitrogen compounds. Input of organic (manure and human sewage) and mineral (synthetic) fertilizers, atmospheric deposition of nitrogen compounds of anthropogenic origin, decomposition of agricultural crop residues on fields, and cultivation of histosols rise nitrogen content in soil and consequently, increase N_2O emission from them (Bouwman et al., 2005; Mosier et al., 2005). Leaching and runoff of nitrogen compounds from fields induce subsequent formation of N_2O in groundwater, rivers and estuaries. These secondary sources are also associated to human activities in agricultural sector.

Soils play an important role in the global carbon cycle and can serve both as a sink and a source of carbon in the atmosphere. The organic carbon pool in the world soils is assessed at 1350 billion tons (Chapin and Matthews, 1994), which is equal to about 80% of the total pool of organic carbon in the terrestrial ecosystems, including vegetation (Land Use, 2000). The anthropogenic influences on soils affect the content of organic carbon and, thus, affect the global carbon balance in the biosphere. One of the main types of the anthropogenic influence on soils is related to the character of land use. For example, tree felling and plowing of virgin lands may result in a considerable loss of soil carbon and, hence, CO_2 release into the atmosphere, whereas the overgrowing of abandoned plowlands may lead to a gradual restoration of the soil humus pool and, thus, to carbon sequestration. In the 1990s, the global annual flux of carbon into the atmosphere under the impact of changes in the land use was equal to 2.2 ± 0.8 billion tons (Hougton, 2003). The main contribution to the carbon emission was due to forest cutting in tropical areas.

The aim of this work is to estimate anthropogenic emissions of CH_4 and N_2O from Russian agriculture and to study the changes in the total pool of organic carbon within the long-fallow lands of Russia for the period from 1990 to 2003.

Methods

The calculation of anthropogenic CH_4 and N_2O emissions was performed in accordance with Revised 1996 IPCC Guidelines (IPCC, 1997) and Good Practice Guidance (IPCC, 2000), and with the use of national statistical data (The Agricultural and Industry…, 1995; Agriculture in Russia, 1998, 2004). Methane emission from enteric fermentation and AWMS was estimated as a sum of emissions from particular animal categories and poultry, whereas the emission from rice cultivation is calculated for continuously flooded fields. Country-specific activity data on nitrogen content in manure and the split of various AWMS in Russia were derived to estimate N_2O emission from livestock categories. These are based on amount of nitrogen produced by domestic livestock and poultry as well as typical waste management practices applied within the country (Gytarsky et al, 2001). Default emission factors were applied to calculate N_2O and CH_4 emissions from enteric fermentation, AWMS and rice cultivation (IPCC, 1997; 2000). Country-specific emission factors were derived to estimate anthropogenic N_2O emission from the input of synthetic nitrogen fertilizers (Romanovskaya et al., 2001, 2002a, b). Nitrous oxide emissions from main types of cultivated soils (chernozems and soddy podzols) were estimated with the use of country specific emission factors (Romanovskaya *et al.*, 2001, 2002b). Calculations for other soils were made with the use of IPCC default emission factors. Nitrous oxide emissions from decomposition of agricultural crop residues (aboveground and underground) left on fields were also estimated with country-specific data on the amount of residues of various crops and their typical N content, including both N-fixing and non-N fixing crops (Romanovskaya et al., 2002a). The default IPCC methodology was applied to calculate emissions from cultivation of histosols in the country and indirect N_2O emissions in agriculture (IPCC, 1997; 2000).

To estimate changes of soil carbon pools within the abandoned lands the recently (after 1990) long-fallow lands of the Russian Federation are analyzed; in agreement with the Kyoto Protocol (Kyoto Protocol, 1998), the lands abandoned prior to 1990 are not considered, though some loss or accumulation of organic carbon may take place in them. Estimates of the areas of long-fallow lands were done on the basis of the official statistical data (The Agricultural and Industry…, 1995; Agriculture in Russia, 1998, 2004) via subtracting the area under croplands and annual fallow from the total area of arable land in separate regions. It was assumed that the soils that had not been plowed in 1990 remained in the fallow state afterwards, i.e., that the natural vegetation succession on them had a continuous character.

Changes in the reserves of organic carbon in the abandoned arable soils depend on the climatic parameters and the type of vegetation; these two factors dictate the annual input of organic remains into the soil and the rates of their decomposition, humification, and mineralization (Titlyanova, 1990; Chapin and Matthews, 1994; Orlov et al., 1996). The methods of mathematical modeling with due account for the entire set of parameters affecting the fate of soil organic carbon are usually applied to estimate the changes in the soil carbon pool. In our study, the ROTHC model was used (Coleman and Jenkinson, 1996; Jenkinson, 1990). This model was developed to simulate the soil carbon cycle in the cultivated soils. It is known that the properties and structural state of the formerly cultivated soils do not change much during the first two--three decades after the transfer of the former cropland into the category of long-fallow land. The initial input data of the model are the climatic data (the mean monthly temperature, precipitation, and evaporability), the soil properties (the initial content of organic carbon in the plow layer and the clay content in the soil), and the vegetation characteristics (the input of organic remains into the soil). This model makes it possible to calculate the soil organic carbon budget on a monthly basis; it can be used for long periods of forecasting. The models take into account four different pools of soil organic carbon: easily decomposable vegetation remains, difficulty decomposable vegetation remains, the microbial biomass, and the soil humus. The calculations are performed for each of these pools and for the pool of inert soil organic matter with high resistance to microbial decomposition.

In our calculations, the territory of Russia was subdivided into 40 regions with due account for the administrative division and for the similarity of soil and climatic conditions in the neighboring administrative regions (oblasts). Data on the monthly precipitation (mm) and temperature (°C) were obtained from the long-term records of weather stations (Reference Book on the Climate of the USSR, 1965; Hong-Kong Observatory, 2003) and averaged for the areas of particular regions. The climatic data for the period from 1960 to 1990 were mainly obtained from the website of the Hong-Cong observatory (Hong-Kong Observatory, 2003). The evaporability was calculated by the Thornthwaite method (Thornthwaite, 1948) on the basis of data on the mean monthly temperatures.

Information on the reserves of humus in different types of Russian soils was taken from the database compiled by Bolotina (1976), which has the most detailed information on the humus content in the plow layer (0--20 cm) for different types of soils differentiated with respect to the character of the land use. The carbon content in the soil humus was assumed to be equal to 58% (Kononova, 1984). Using this value, the reserves of humus in the plow layer were recalculated into the reserves of organic carbon. The data on the humus content in the plow layer of different soils were linked to the map of the administrative division of Russia. The distribution of plowed areas by different types of soils in each of the administrative regions was obtained from the statistical information of the Ministry of Agriculture (Distribution of Agricultural Lands…, 1980). On this basis, the average humus content in the humus of plowed soils was calculated for all the regions.

The databases developed by Bolotina (1976) and Sokolov and Rozov (1976) contain generalized information on the texture of the arable soils in the former Soviet Union; a distinction is made between loamy sandy, loamy, and clayey soils. The average content of the physical clay fraction in these groups of soils can be obtained from (Golubev, 1970). In our work, the physical clay content in the loamy sandy soils is assumed to be equal to 15%; for the loamy podzolic soils, 35%; for the loamy steppe (chernozemic) soils, 40%; for the clayey podzolic soils, 65%; and for the clayey chernozems, 72.5%. Taking into account the areas occupied by different soil types in the croplands of particular administrative regions (Distribution of Agricultural Lands…, 1980), the average content of the physical clay fraction in the cultivated soils of the regions can be determined.

The model distinguishes between different groups of soil organic carbon. The inert pool of soil humus is difficultly decomposable and remains virtually stable for decades. Thus, this pool of soil humus was excluded from further calculations (Jenkinson, 1990). The initial data for the other structural components of the soil organic matter were calculated according to the methodology suggested by the authors of the model (Coleman and Jenkinson, 1996; Jenkinson, 1990). It was supposed that the organic matter of the plowed soils was in a quasiequilibrium state with the environment before the cessation of the plowing. To adjust the model, the reserves of organic carbon in the plow layer (0--20 cm) were calculated for long time periods (up to 10 000 years) with due account for the soil and climatic conditions of a given region and for the different values of the annual input of postharvest residues into the soil. The organic carbon content in the postharvest residues of major types of cultivated crops was determined according to the method of Levin (Levin, 1977; Romanovskaya et al., 2002a). The calculation of the initial pool of organic carbon in the active components of the soil organic matter (labile humus, microbial biomass, and plant remains) was stopped when the total organic carbon content predicted by the model reached the values corresponding to the average organic carbon content in the soils of a given region calculated on the basis of the work of Bolotina (1976). The difference between the values calculated by the model and the values calculated by us on the basis of published data was less than 0.01 t C/ha.

The dynamics of the input of plant remains into the soil after the cessation of crop growing depends on the type of vegetation succession taking place on the former cropland (Gusev, 1932; Nersesyan and Shur-Bagdasaryan, 1989; Snakin, 1992; Snytko et al., 1986, 1988; Turnagaev and Pestereva, 1976). The annual productivity of plant communities in the succession can be expressed in percent of the maximum productivity of meadow communities in the given region. In turn, the latter can be calculated on the basis of data by Bazilevich (1993). Information on the productivity of meadow herbs in the zones of the middle and southern taiga, mixed and broad-leaved forests, forest-steppes, and steppes, as well as in the mountainous regions, has been averaged for the particular zones and, then, for the particular regions distinguished in our study.

It can also be supposed that the annual phytomass production of herbs in the plant successions on the abandoned croplands (long-fallow lands) is approximately equal to the amount of the plant remains getting into the soil. This assumption suggests that the development of vegetation successions on the abandoned croplands follows the natural pattern. Haymaking, pasturing, grass burning, and other kinds of anthropogenic impacts on the vegetation of the abandoned croplands are not taken into account. The minimal influence of the latter factors on the character of the vegetation successions on the recently abandoned croplands can be indirectly proved by the data on the bulk harvest of hay in Russia in the past decade. From 1990 to 2003, it decreased by 40% (The Agricultural and Industry…, 1995; Agriculture in Russia, 1998, 2004). Thus, we can conclude that no additional hayfields appeared in the country during this period.

Results and Discussion

Greenhouse gas emissions from agriculture in Russia are presented in the Table 1. In 1990 the total CO_2 eqivalent emissions from agricultural sources estimated as high as 325.9 Tg. During the period from 1990 to 2003 the total agricultural emissions dropped on 54.3% and in 2003 agriculture provided not more of 148.9 Tg CO_2 eq. The decrease in CH_4 and N_2O emissions is mainly caused by reduction in number of agricultural animals and poultry, decline in the area of managed land and decreased fertilizer use due to political and economical changes in the country.

The trends of the major greenhouse gases in agriculture during the period from 1990 to 2003 are given on the figure 1. Both gases show constant decline in their emissions. In 1990, CH_4 release to the atmosphere was 4993.3 Gg. In 2003 it dropped to 2247.5 Gg. Total estimated anthropogenic N_2O emissions from agriculture were 713.1 Gg in 1990 and in 2003 it decreased to 46% of 1990 level (328.0 Gg).

Contributions of various category sources to the total agricultural emissions in the country are rather stable all over the whole period of estimations (Figure 2 and 3). Enteric fermentation is the priority source of methane emission in national agricultural sector. Methane release from enteric fermentation depends on type, age, weight productivity and diet of animals, as well as the orientation of their growing. Ruminant animals have the most intensive fermentation and consequently, CH_4 emission. Almost 87% of CH_4 emissions from enteric fermentation come from dairy and non-dairy cattle. CH_4 emission from AWMS is mainly produced by cattle and swine farms. Rice cultivation contributes to 2.5% of total methane emission from national agriculture.

Table 1. Greenhouse gas emissions in Russian agriculture, Gg/yr.

Gas: Category source	1990	1995	1996	1997	1998	1999	2000	2001	2002	2003
CH4: Enteric fermentation	4376.2	3233.6	2951.5	2605.7	2334.1	2108.0	2067.6	2009.2	1991.6	1954.4
CH4: Manure management	502.3	357.5	325.6	284.8	258.8	242.9	243.5	228.7	228.7	230.7
N2O : Manure management	183.75	131.4	117.9	103.0	93.4	86.4	85.3	82.0	82.2	82.1
CH4: Rice cultivation	114.8	68.4	68.8	60.4	58.4	69.2	70.0	61.6	59.6	62.4
N2O: Direct emissions from agricultural soils	338.5	221.9	211.9	206.2	179.5	175.5	182.7	181.7	182.1	175.0
N2O: Grazing of animals	29.1	20.8	18.7	16.3	14.6	13.2	12.9	12.6	12.6	12.5
N2O: Indirect emissions from agricultural soils	161.8	88.6	80.7	72.6	65.1	58.7	61.8	59.0	60.1	58.5
Total GHG emission, Tg CO2 eq.	325.93	220.30	203.28	185.38	164.97	154.32	156.26	152.23	152.35	148.87

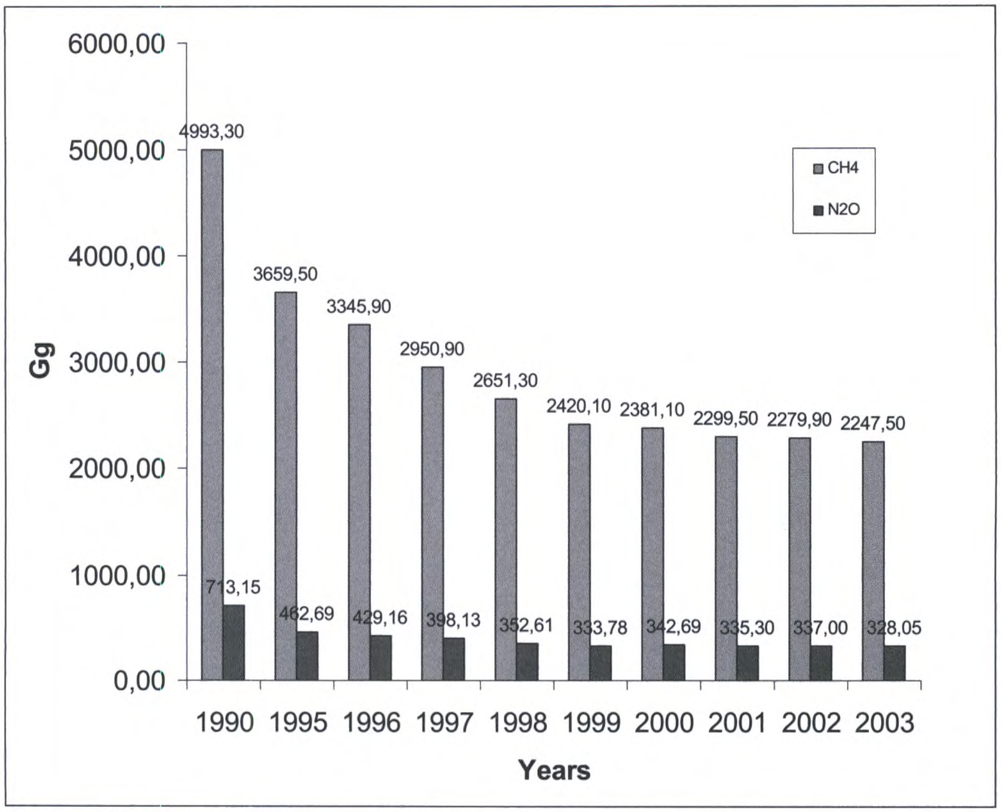


Figure 1. Trends of the methane and nitrous oxide emissions from agricultural sources.

N_2O emissions were estimated from manure management systems and agricultural soils taking into consideration input of nitrogen with synthetic fertilizers, animal wastes, decomposition of crop residues left on fields and cultivation of histosols. Crop residues are the main N_2O source in direct emissions from agricultural soils, contributing from 40% in 1990 up to 57% in 2003 to that category source. That corresponds to 13% and 21% contribution to the total agricultural emissions respectively (Figure 2 and 3). After 1990, the decrease in use of synthetic nitrogen fertilizers resulted in decrease of N_2O emission from this source and therefore its contribution. In 1990 the input of fertilizer nitrogen caused the release of 83 Gg N_2O. In 2003 emission dropped and became approximately 19.5% of the 1990 level (16 Gg N_2O).

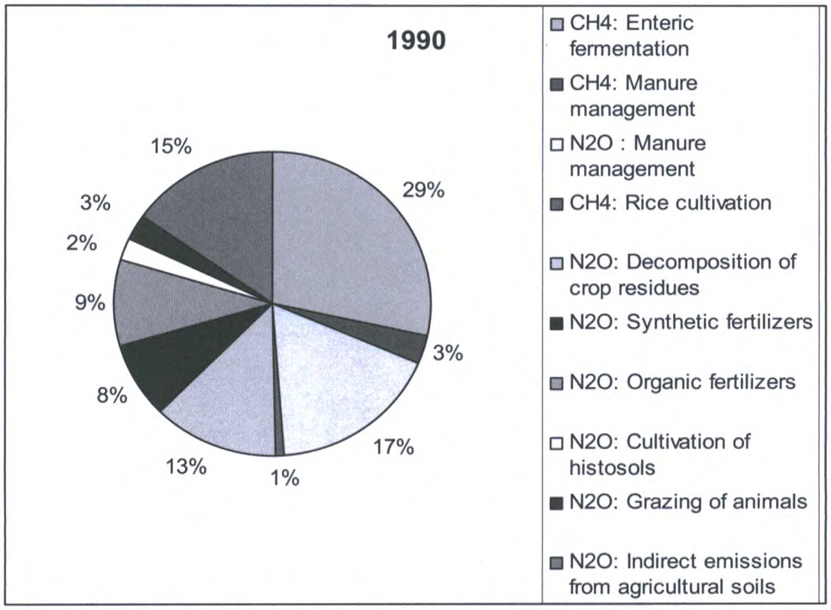


Figure 2. Contribution of different categories sources to the total GHG emissions from agriculture in 1990 (CO_2 eq.).

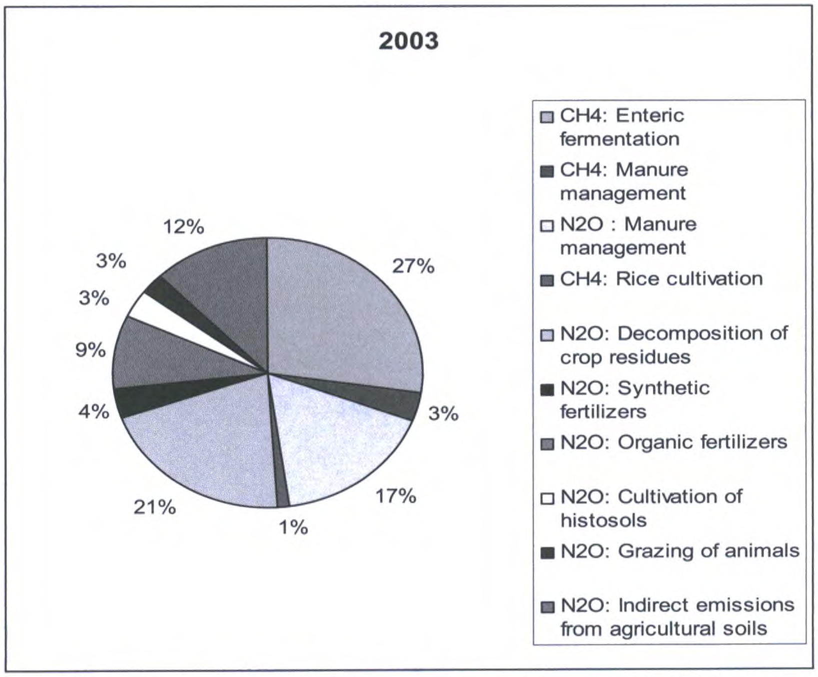


Figure 3. Contribution of different categories sources to the total GHG emissions from agriculture in 2003 (CO_2 eq.).

Soil Carbon in Abandoned Land

The results of our calculations indicate that the area of recently abandoned plowlands subjected to overgrowing with natural vegetation considerably increased in Russia from 1990 to 2003. In 1990, the total area of long-fallow lands comprised 0.5% of the total area of the cropland in Russia. In 2003, the area of long-fallow lands increased to 17.4% of the total area of the cropland, i.e., by almost 30 times. Overall, the area of abandoned cropland during these years reached 25.4 million ha.

The dynamics of the soil organic carbon on the abandoned croplands for some regions of Russia are shown in Fig. 4. In all of them, the general tendency is the same: during the first years after being abandoned, the former croplands lose organic carbon; then, as the vegetation succession leads to the development of more productive communities, the accumulation of organic carbon in the soils takes place. The rate of carbon accumulation may be higher than the rate of its loss during the first years. It should be noted that the rate of changes in the humus pool depends on the initial content of the organic carbon. In soils with an initially high humus content (e.g., in plowed soils of the central chernozemic zone and Western Siberia), these changes are relatively slow (Fig. 4). In contrast, in the low-humus soil, the dynamics of organic carbon upon the soil fallowing is more pronounced, especially in the zones with favorable climatic and vegetation conditions (e.g., in the plowed soils of Irkutsk oblast and Khabarovsk region). In Vologda oblast, the abandoned plowlands are overgrown with meadow vegetation of low productivity. Therefore, though the carbon accumulation in the soil begins on the third year of the succession, the rate of this process is relatively low. In general, the highest intensity of organic carbon accumulation in the overgrown soils of abandoned plowlands is the highest in the central region of Russia (up to 5.8 t C/ha per year) and in the Far East (up to 8 t C/ha per year), which is related to the high productivity of herbaceous biocenoses in the zones of mixed and broad-leaved forests. For the chernozemic regions of Russia, including Rostov oblast, Krasnodar region, and Stavropol region, the loss of organic carbon from the abandoned plowlands is especially significant. In 13 years, the total losses of soil organic carbon from the abandoned plowlands in these regions reached 4.8, 4.6, and 6.9 t C/ha. This is explained by the warm climate of these regions and, thus, the enhanced mineralization of the soil organic matter against the background of the low productivity of the secondary biocenoses in the dry steppe zone. The organic carbon accumulation in the abandoned and overgrown croplands in these regions begins in the ninth--tenth year after the beginning of the natural succession and proceeds at a low rate. Therefore, the net balance of carbon for the whole period under consideration (1990--2003) is negative in these regions.

Model estimations of the annual carbon flux in abandoned land during 1990-2003 shown on the Figure 5. Positive values on the figure 5 are emissions of CO_2 and negative values correspond to its sink. Before 1998 abandoned land in Russia loses soil carbon. Associated with ageing of soil after abandonment the annual plant production on unmanaged land is growing. That leads to the accumulation of the organic carbon in soils. Therefore after 1998 abandoned lands in Russia are responsible for net-sink of CO_2 with exception for 1999 then new large territories were abandoned as a result of economical crisis in 1998. Average annual net-emission over the territory of abandoned lands was 2.1 ± 1.8 Tg C/yr in 1990-1999. CO_2 removal from the atmosphere by soils was 5.7 ± 2.5 Tg C/yr on average in 2000-2003. As the largest territories of managed land were abandoned in southern regions of Russia, for those losses of soil carbon are typical, therefore, total balance of soil carbon on territory of abandoned lands in Russia during 1990-2003 shifted to emissions of 5.4 Tg C (19.8 Tg CO2).

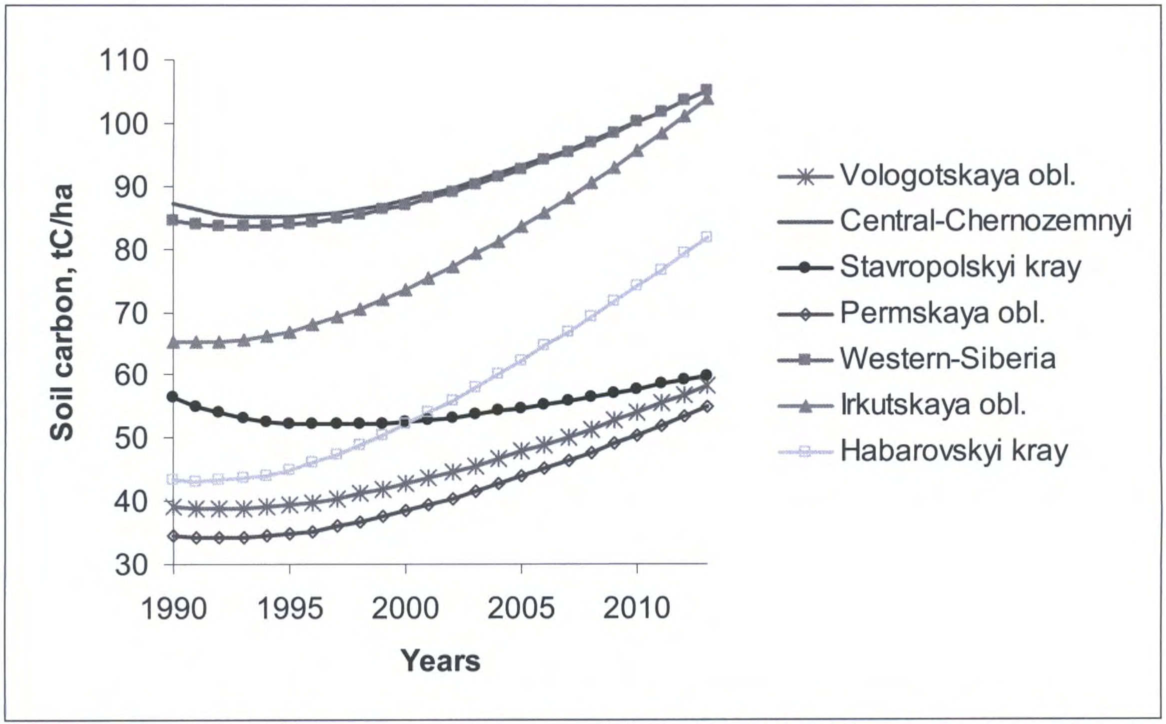


Figure 4. Dynamics of the soil organic carbon in abandoned land of some regions of Russia.

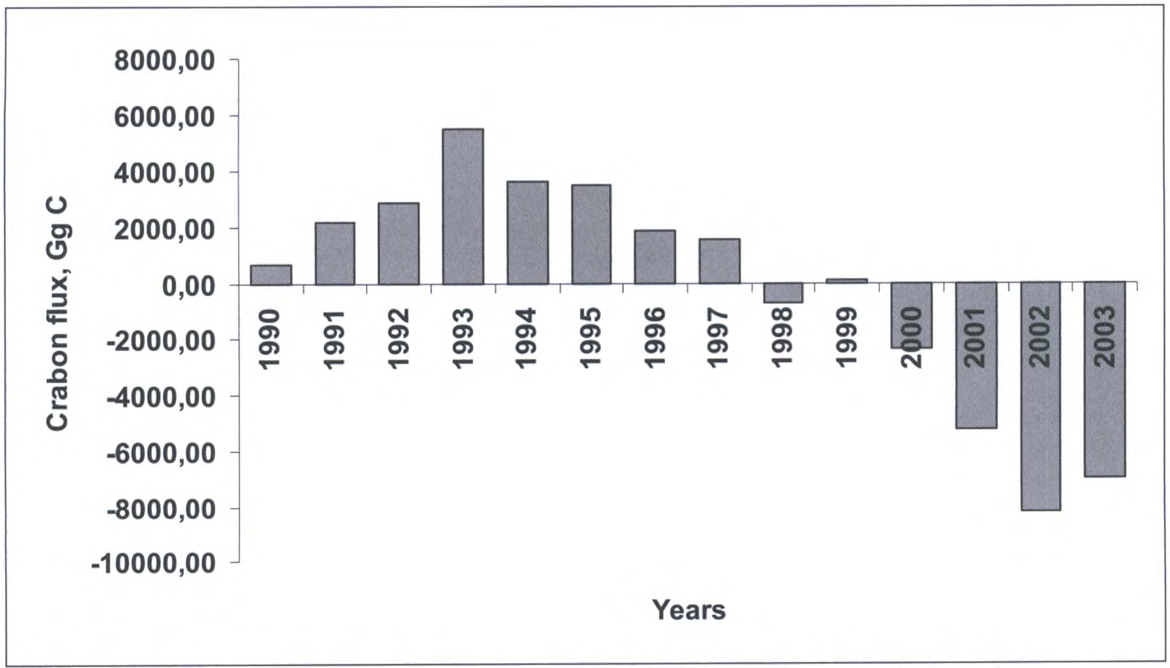


Figure 5.Annual flux of soil carbon on the territory of abandoned land in Russia.

At the same time, the current tendency in the dynamics of the soil organic carbon on the plowlands abandoned several years ago (Figs. 4) is directed towards its increased accumulation in the soils. It can be expected that this tendency will be preserved up to 2015--2020. Moreover, already in 2008--2012, when the obligations imposed by the Kyoto Protocol will come into force, the long-fallow soils in the Russian Federation will serve as a sink for carbon; it is expected that the pool of organic carbon in them will be higher than that in 1990. In other words, the long-fallow lands of Russia can be considered as a sink for carbon in 2008--2012. More accurate estimates of the capacity of this sink should be made in the future.

Conclusions

Methane and Nitrous Oxide Emissions from Agriculture

1) Enteric fermentation of ruminant animals provides the major contribution to anthropogenic CH_4 emission in Russian agriculture. The animal waste management systems comprise for no more than 13% of sectoral methane emission. In 1990, Russian agriculture was a source of 4993 Gg of CH_4 that is 3% of its global emission from agriculture. Later the emission dropped and in 2003 it was 45% of the 1990 level (2247 Gg).
2) Agricultural soils are the major source of anthropogenic N_2O emission in Russia. Within the period from 1990 to 2003, crop residues contributed from 30% to 57% of direct emissions from agricultural soils. In 1990 total N_2O emission was 713 Gg. In 2003 the emission dropped and became no more than 46% of the 1990 level (328 Gg).
3) The decrease in methane and nitrous oxide emissions in Russian agriculture between 1990-2003 is associated with a reduced input of nitrogen fertilizers, decline of cultivated area and population of domestic livestock and poultry in agriculture.

Soil Carbon in Abandoned Land

1) The area of long-fallow lands in Russia considerably increased from 1990 to 2003. In total, the area of abandoned croplands transferred into the category of long-fallow lands in Russia during these 14 years reached 25.4 million ha.
2) The general tendency in dynamics of soil carbon in abandoned land of different regions of Russia is the same: during the first years after being abandoned, the former croplands lose organic carbon; then, as the vegetation succession leads to the development of more productive communities, the accumulation of organic carbon in the soils takes place. The rate of carbon accumulation may be higher than the rate of its loss during the first years. However, the rate of changes in the humus pool depends on the initial content of the organic carbon.
3) Average annual net-emission over the territory of abandoned lands was 2.1 ± 1.8 Tg C/yr in 1990-1999. CO_2 removal from the atmosphere by soils was 5.7 ± 2.5 Tg C/yr on average in 2000-2003. As the largest territories of managed land were abandoned in southern regions of Russia, for those losses of soil carbon are typical, therefore, total balance of soil carbon on territory of abandoned lands in Russia during 1990-2003 shifted to emissions of 5.4 Tg C (19.8 Tg CO_2).

Acknowledgments

This study was supported by the Russian Foundation for Basic Research (project nos. 03-05-65085 and 05-05-65109) and Grant of the President on the Support of Young Scientists and Lead Science Schools on Problems of Soil Sciences of academician Y.A. Israel (no. NSH-1876.2003.5).

References

Agriculture in Russia. 1998. A Statistical Textbook. Moscow: Goskomstat [in Russian]

Agriculture in Russia. 2004. A Statistical Textbook. Moscow: Goskomstat [in Russian]

Bazilevich, N. I. 1993. Biological Productivity of Ecosystems in the Northern Eurasia. Moscow, Nauka. [in Russian].

Bolotina, N. I. 1976. Humus and Nitrogen Reserves in the Main Soil Types of the Soviet Union. In *Agrochemical Characterization of Soils of the Soviet Union* 15: 187--202.

Bouwman, A.F., G. Van Drecht, and K.W. van der Hoek. 2005. Surface N balances and reactive N loss to the environment from global intensive agricultural production systems for the period 1970-2030. *Science in China. Ser. C. Life Sciences* 48 (Special issue): 767-779.

Chapin, F. S., and E. Matthews. 1994. Boreal Carbon Pools: Approaches and Constrains in Global Extrapolations. In Carbon Cycling in Boreal Forests and Sub-Arctic Ecosystems. Ed. by T. S. Vinson, and T.P. Kolchugina. EPA, Corvallis, OR: 9-20.

Coleman, K., and D. S. Jenkinson. 1996. RothC-26.3, a Model for the Turnover of Carbon in Soil. In Evaluation of Soil Organic Matter Model. Ed. by D. S. Powlson, P. Smith, and J. U. Smith. NATO ASI Series, Springel, Berlin, 138: 237--246.

Distribution of Agricultural Lands in the Russian Federation by Soil Groups. 1980. Minsel'khoz RSFSR, Moscow. [in Russian].

Golubev, I. F. 1970. Soil Science with Basic Geobotanics. Moscow, Kolos. [in Russian].

Gusev, S. D. 1932. Vegetation and Dynamics of the Overgrowing of Long-Fallow Lands in the Verkhnii Ural Region. Nadezhdinsk, Ural'skii Kraeved. [in Russian].

Gytarsky, M.L., Zh.N.Lodzhyn, A.I. Nakhutin, V.A. Savin, R.T. Karaban’, R.M. Alexakhin, and I.M. Nazarov. 2001. Greenhouse Gas Emission From Agricultural Animals and Poultry in Agrarian Sector of Russia. *Agricultural Biology, Section Animal Biology* (6): 73-79.

Hong-Kong Observatory. 2003. 1961--1990 Global Climate Normals. Hong-Kong Observatory, WMO. http://www.hko.gov.hk/wxinfo/climat/world/eng/europe/russia/russia-e.htm.

Houghton, R. A. 2003. Revised Estimates of the Annual Net Flux of Carbon to the Atmosphere from Changes in Land Use and Land Management 1850—2000. *Tellus* 55B: 378-390.

IPCC, 1997. Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories. Paris, IPCC–OECD–IEA.

IPCC, 2000. IPCC Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories. Japan, IPCC, IGES.

Jenkinson, D. S. 1990. The Turnover of Organic Carbon and Nitrogen in Soil. *Phil. Trans. Roy. Soc*. 329: 361--368.

Khalil, M.A.K., and R.A. Rasmussen. 1983. Increase and seasonal cycles of nitrous oxides in the earth’s atmosphere. *Tellus* 35B: 161-169.

Kononova, M. M. 1984. Organic Matter and Soil Fertility. *Pochvovedenie* (8): 6—20.

Kyoto Protocol. 1998. Kyoto Protocol to the United Nations Framework Convention on Climate Change. UNFCCC--UNEP--IUC.

Land Use. 2000. Land Use, Land Use Change, and Forestry. A Special Report of the IPCC. Cambridge Univ. Press. Cambridge.

Levin, F. I. 1977. Content of Plant Remains in Field Crops and Its Determination from the Target Crop Yield. *Agrokhimiya* (8): 36--42.

Mosier, A. R., J.K. Syers, and J.R. Freney. 2005. Global assessment of nitrogen fertilizer: The SCOPE/IGBP Nitrogen Fertilizer Rapid Assessment Project. *Science in China. Ser. C. Life Sciences* 48 (Special issue): 759-766.

Nersesyan, T. Sh., and E. F. Shur-Bagdasaryan. 1989. Biological Productivity of Natural Phytocenoses. *Biol. Zh. Armenii* 42 (7): 684-687.

Orlov, D. S., O. N. Biryukova, and N. I. Sukhanova. 1996. Organic Matter in the Soils of Russian Federation. Moscow, Nauka. [in Russian].

Reference Book on the Climate of the USSR. 1965. Gidrometeoizdat, Leningrad. [in Russian].

Romanovskaya, A. A., M. L. Gytarsky, R. T. Karaban', and I. M. Nazarov. 2002a. Assessment of Nitrous Oxide Emission from Agricultural Plant Mortmass Unused in the Agrarian Sector of the Country. In Problems of Ecological Monitoring and Simulation of Ecosystems. Gidrometeoizdat, St. Petersburg, 18: 276-286 [in Russian].

Romanovskaya, A.A, M.L. Gytarsky, R.T. Karaban', D.E. Konyushkov, and I.M. Nazarov. 2001. The dynamics of nitrous oxide emission from the use of mineral fertilizers in Russia. In Optimizing Nitrogen Management in Food and Energy Production and Environmental Protection: Proceedings of 2nd International Nitrogen Conference on Science and Policy. *TheScientificWorld* 1 (S2): 336-342.

Romanovskaya, A.A., M.L. Gytarsky, R.T. Karaban', D.E. Konyushkov, and I.M. Nazarov. 2002b. Nitrous oxide emission from agricultural lands in Russia. *Mitigation and Adaptation Strategies for Global Change* 7(1): 31-43.

Snakin, V. V. 1992. Redox Potential of Soils and Productive Parameters of Grass Ecosystems. *Izv. Ross. Akad. Nauk, Ser. Biol.*(2), 295-300.

Snytko, V. A., L. G. Nefed'eva, and S. S. Dubynina. 1986. Grass Biogeocenoses of the Nazarovskaya Hollow and the Effect of Technogenesis on Their Productivity. In Productivity of Haylands and Pastures. Ed. by A. A. Titlyanova. Novosibirsk, Nauka. [in Russian].

Snytko, V. A., L. G. Nefed'eva, and S. S. Dubynina. 1988. Tendencies in the Restoration of Disturbed Lands. *Geogr. Prir. Resur.* (1): 56-61.

Sokolov, A. V., and N. N. Rozov. 1976. Soil-Agrochemical Zoning of the Soviet Union. In *Agrochemical Characterization of Soils of the Soviet Union* 15: 5--16.

The Agricultural and Industry Complex of Russia: Resources, Production, and Economy. 1995. A Statistical Textbook. Novosibirsk, Russian Academy of Agricultural Sciences (1). [in Russian].

Thornthwaite, C. W. 1948. An Approach toward a Rational Classification of Climate. *Geogr. Rev.* (38): 55--94.

Titlyanova, A. A. 1990. Primary Production and Humus Reserves in Ecosystems. In Problems of Soil Science in Siberia. Novosibirsk: 47--53 [in Russian].

Turganaev, V. V., and T. A. Pestereva. 1976. Dynamics of Vegetation on Abandoned Arable Lands in the Southern Regions of the Vyatka--Kama Basin (Udmurtiya). *Bot. Zh.*(6): 1265—1272.

UN FCCC, 1998. UN Framework Convention on Climate Change. UNFCCC--UNEP--IUC.

Nitrous Oxide Emissions from Denitrification in Swine Manure Treatment

Joe Rudek[1] and William L. Chameides[2]
[1]Environmental Defense, Raleigh, NC
[2] Environmental Defense, New York, New York

Abstract

Alternative manure treatment technologies such as anaerobic digesters are effective in preventing losses of undesirable gases to the atmosphere during the mineralization of organics. Consequently, these treatments result in increased quantities of ammonia nitrogen in the treated liquid, relative to that remaining in a lagoon. Denitrification is often considered as a means to convert the ammonia nitrogen into a benign end product, N_2. The production of nitrous oxide as a byproduct of (nitrification and) denitrification may be an environmental concern given its high global warming potential.

Discussion

Decades ago the anaerobic lagoon was designed, at least in part, to protect groundwater from nitrate contamination during animal waste treatment. This protection was accomplished by enhancing ammonia volatilization into the atmosphere. At the time, emitting ammonia into the atmosphere seemed sensible, given concerns about nitrate. Unfortunately the large fluxes of ammonia from the lagoons has led to significant health risks and environmental impacts. As efforts to find alternative animal waste treatment technologies proceed, it would be prudent to consider the potential for similar unintentional impacts that might not be apparent today. One such concern is nitrous oxide loss as a result of denitrification. While nitrous oxide loss during denitrification is generally only a small percentage (less than 2%) of the nitrogen being processed, its very high global warming potential (310 time CO_2) means even a *small* percentage loss can have a significant climatic impact. Reductions in methane emission, also a goal of many alternative waste treatment systems, could partially offset the net global warming potential of nitrous oxide lost during waste treatment. However, the global warming potential of methane is 21 times CO_2, nearly 15 times less potent than nitrous oxide. The impact of potential nitrous oxide loss in the advent of widespread adoption of denitrification as part of an alternative swine waste treatment system will be evaluated. The offsets and tradeoffs of simultaneous methane and ammonia loss reductions relative to increased nitrous oxide emissions in alternative treatment systems will also be considered.

Ammonia Emission Fluxes from a Dairy: Stalls, Lagoons, and Slurry Application

Brian Rumburg[1], George H. Mount[1], Brian K. Lamb[1], Hal Westberg[1], Jenny Filipy[1], David Yonge[2], Kris Johnson[3], Ron Kincaid[3],

[1]Laboratory for Atmospheric Research, Department of Civil & Environmental Engineering, Washington State University, Pullman, Washington, USA 99164-2910
[2]Center for Multiphase Environmental Research, Department of Civil & Environmental Engineering, Washington State University, Pullman, Washington, USA 99164-2910
[3]Department of Animal Sciences, Washington State University, Pullman, Washington, USA 99164-6310

Abstract

The effect of atmospheric ammonia (NH_3) emissions on the atmosphere, environment, and human health are not well understood due to a lack of information about emissions, transport, and fate. Ammonia in the atmosphere reacts to form fine aerosols which impact human health and its deposition has serious environmental impacts. Sulfate and nitrate in the atmosphere will react with NH_3 to form particulate matter less than 2.5 microns in aerodynamic diameter ($PM_{2.5}$) which is regulated by the Environmental Protection Agency (EPA). The largest anthropogenic source of atmospheric NH_3 is animal excreta. Dairy cows are the largest per animal emission source due to the large energy requirements of milk production. Short-path differential optical absorption spectroscopy (DOAS) was used to measure NH_3 concentrations, and an area source tracer ratio technique was used to calculate emission fluxes. Measurements were made from the housing area, waste lagoons, and sprinkler application. Emissions models were developed to understand the physical processes and local transport of NH_3. Housing emission fluxes for the summer averaged 8.1 ± 5.2 mg cow^{-1} s^{-1} at 18° C average temperature. The housing model had an error of ± 30% when compared to measured concentrations. Lagoon emission fluxes ranged from 30 μg m^{-2} s^{-1} at an air temperature of 11° C to 150 μg m^{-2} s^{-1} at an air temperature of 27° C. The lagoon model had an error of ± 21%. Slurry application emissions from the sprinkler were 18% of the slurry ammonia-N concentration. The initial measured flux from the field was 47 μg m^{-2} s^{-1} and this decreased to 17 μg m^{-2} s^{-1} during the experiment due to slurry infiltration into the soil. Annual emissions based upon theoretical models estimate emissions to be 130 kg NH_3 cow^{-1}.

Introduction

Ammonia emissions and the impact of atmospheric NH_3 on human health and the environment in the United States is not well understood due in part to a lack of measurement data from the major anthropogenic sources, waste from domesticated animals. The measurement of atmospheric NH_3 is difficult since it adheres to instrument inlet walls creating measurement artifacts. The United States Environmental Protection Agency (EPA) established a new standard for particulate matter with an aerodynamic diameter less than 2.5 microns ($PM_{2.5}$) in 1997. The passage of the $PM_{2.5}$ standard has led to interest in NH_3 emissions because NH_3 reacts with nitrate and sulfate species in the atmosphere to form $PM_{2.5}$. Current U.S. emission estimates are based upon European measurements where farming practices and regulations are different from the U.S., leading to possible errors in the current NH_3 emission estimates.

Most current emission estimates are annual estimates based upon a few measurements and do not reflect the influence of meteorology on emissions. Ammonia emissions vary seasonally due to the volatilization rate being temperature dependent. The production of NH_3 from the biological breakdown of organic N in animal waste is also temperature dependent. Cows produce the largest fraction of NH_3 globally, and of these, dairy cows have the largest per animal emissions due to their consumption of large amounts of protein for milk production. Emissions vary depending upon the housing, waste storage, and waste disposal methods.

The purpose of this work was to determine the total emissions of NH_3 from the milking dairy cows at the Washington State University (WSU) dairy and develop emissions algorithms for use in a regional air quality model. The emission sources measured include the housing area, waste lagoons, and waste application. This paper discusses the measurements of NH_3 emission fluxes and model results.

Methods

A series of concentration measurements and tracer experiments were conducted downwind of the milking cow housing, waste lagoons, and during sprinkler application of the waste at the WSU dairy from 2000 to 2004.

Instrumentation

Atmospheric NH_3 was measured using an open path differential optical absorption spectroscopy (DOAS) as described in Mount et al. (2002). The open-path method is advantageous because it does not suffer measurement problems due to NH_3 adhering to inlet walls. The instrument measures the photoabsorption of NH_3 in the mid-ultraviolet from 200 to 235 nm. The spectrograph is housed in a trailer along with the Xenon light source, collimating and collecting optics. Retro reflectors were used to direct the light beam to the collecting optics and then into the spectrograph. The spectrograph is a double crossed Czerny-Turner design with a silicon array diode detector. The instrument has a very high signal to noise ratio allowing high precision measurements as fast as every second. The detection limit is a few ppbv. For the experiment, spectra were co-added and recorded approximately every five minutes.

Tracer Ratio Flux Method

Fluxes from the dairy housing, waste lagoons, and waste application were measured using a tracer ratio technique. Sulfur hexafluoride (SF_6) was used as a tracer gas due to its low reactivity. The higher molecular weight of SF_6 compared to NH_3 has a negligible effect due to turbulent diffusion being the dominant transport mechanism (Kaharabata et al., 2000). The area source tracer ratio technique first discussed by Ludwig et al. (1983) was used to calculate the fluxes of NH_3. The method has been used to measure emissions of natural gas from urban areas (Lamb et al., 1995; Shorter et al., 1997), hydrocarbons from refinery waste impoundments (Howard et al., 1992), and greenhouse gases from cattle processing waste lagoons (Eklund, 1999). Tracer techniques have been shown to be in good agreement with other flux measurement techniques (Lamb et al., 1986). In this approach, a tracer is released as close as possible upwind of the area source, at a known rate and is measured downwind at the same location as the pollutant concentration from the area source. Knowing the tracer release rate and downwind concentrations it is possible to calculate the pollutant emission rate. The best conditions for tracer experiments are a steady wind so turbulence is the dominant dispersion mechanism and small σ_θ so the tracer plume traverses the area source.

The equation for the concentration downwind of an infinite line source is the following

$$C_l = 2Q_l / (\sigma_z u \ (2\pi)^{1/2}) \qquad (1)$$

where C_l is the line source concentration at ground level (g m^{-3}), Q_l is line source emission rate (g m^{-1} s^{-1}), σ_z is the vertical plume diffusion coefficient, and u is the wind speed (m s^{-1}). For this derivation the following assumption was made

$$\sigma_z = a\, x^b \qquad (2)$$

where a and b are empirical coefficients and x is the distance of the emission source from the receptor.

For an area source the above equation can be integrated to yield:

$$C_A = 2Q_A (X_u^{(1-b)} - X_d^{(1-b)}) / (u\, a(1-b)\, (2\pi)^{1/2}) \qquad \text{for } b \neq 1 \qquad (3)$$

where C_A is the area source concentration at ground level (g m^{-3}), Q_A is the area source emission rate (g m^{-2} s^{-1}), X_u is the distance to upwind edge of area source (m) and X_d is the distance to downwind edge of area source (m).

The area source and line source equations can be combined to solve for the flux from the area source. The only unmeasured variable is b which can be estimated from the tracer measurements or taken from the literature.

$$Q_A = (C_A\, Q_l\, (1 - b)) / (C_l\, (X_u^{1-b} - X_d^{1-b})\, x^b) \qquad (4)$$

Tracer experiments were conducted during periods of high and steady winds to ensure that SF_6 and the NH_3 plume was being measured. During low wind speeds and times of high σ_θ the SF_6 and NH_3 plumes separated and the results were inconsistent.

The SF_6 line source was located at the upper edge of the area sources, the tubing had restrictors placed every 4 m. Sequential 30 minute average portable syringe samplers were placed along the instrument path. The syringes were analyzed using a HP 5880A gas chromatograph (Hewlett Packard, Palo Alto, CA), with standards from Scott-Marrin (Riverside, CA). Line source emission rates were measured using a Alltech Digital Flow Check (Alltech, Deerfield, IL) flow meter.

Washington State University Dairy

The WSU Knott Dairy Center is a research and commercial dairy that also serves as a teaching and research laboratory in addition to supplying milk for the WSU creamery. The milk from the dairy is processed on campus and sold as cheese; students manage and take care of their own small herd and research is conducted on diet, milk production, waste production and cow health. The WSU dairy is located 8 km south of Pullman, Washington U.S. (N 46° 43.8', W 117° 10.1'). It has approximately 175 Holstein milking cows, 35 dry cows, 130 heifers and 75 calves.

The milking cows are housed in three-sided freestall barns with concrete floors. Waste, which includes milking cow manure and urine, bedding and milking parlor wash water from the stalls, is scraped into pits daily. The pits are flushed with recycled wastewater and the wastewater goes to a solids separator where solids greater than 0.3 cm in diameter are removed. From the solids separator the wastewater goes to a high solids lagoon (250 m^3) where additional solids are removed via gravity settling. The supernatant from the high solids lagoon goes to the low solids lagoon (9500 m^3) where additional settling occurs. The low solids lagoon is used as the primary storage lagoon for waste from the dairy. Two addition lagoons (19000 m^3) are used for long term storage of waste until the waste is applied to the surrounding grass fields in the late summer and early fall.

The milking cows are fed a diet of 26 kg of feed per day on a dry matter basis, the crude protein of the diet is 19%. The feed consists of alfalfa silage (25.6%), alfalfa hay (22.3%), concentrate (35.7%), wheat mill run (6.6%), and whole cottonseeds (9.9%). The cows consume 475 g day^{-1} of K and 790 g day^{-1} of N. The daily consumption of Na is is estimated to be 83 g day^{-1}, but it varies as the cows have unlimited access to salt. The daily milk production for the herd averages about 41 kg cow^{-1}.

Results and Discussion

Housing Emissions

The average measured flux from the housing area was 8.1 ± 5.2 mg cow^{-1} s^{-1} at an average temperature of 18° C. A mechanistic model of emissions was developed based upon a model of cow urea excretions and the reactions of urea to form liquid NH_3 and the volatilization rate from the liquid. The modeled concentrations had an error of ±30% compared to the measured concentrations. The model was most sensitive to changes in the pH. Emissions were highest during the summer with the maximum being about 80 kg NH_3 day^{-1} during warm windy conditions. Total annual emissions from the housing is 7000 kg NH_3 (40 kg NH_3 cow^{-1}). A N mass balance of the housing area showed that maximum emissions would be 7900 kg NH_3 yr^{-1} (Rumburg et al., 2006a).

Waste Storage Emissions

The NH_3 fluxes from the waste lagoons ranged from 30 to 150 μg m^{-2} s^{-1} depending upon temperature. A theoretical emissions model was developed based upon the temperature dependence of the ammonia concentration in the lagoons, volatilization rate and pH partitioning of NH_3 and NH_4^+ in solution. The theoretical emissions model predicts the downwind concentrations of NH_3 best when using the air temperature as opposed to the lagoon temperature. The normalized mean error of the model using the ambient air temperature as the lagoon temperature input was 21%. The model is very sensitive to changes in pH. The daily lagoon fluxes ranged from over 100 kg per day in the summer to 0 in the winter when the lagoons were frozen over. The annual emissions were calculated to be 9700 kg or (55 kg cow^{-1} yr^{-1}) based upon dairy lagoon operating conditions (Rumburg et al., 2006b).

Waste Application Emissions

The dominant emissions from the slurry application with the "Big Gun" sprinkler were measured to be from the sprinkler itself, with 18% of the NH_3 in solution volatilizing. Initial emission from the field were 47 μg m^2 s^{-1} which decreased to 17 μg m^2 s^{-1} in four and a half hours. An empirical exponential function was fit to the emissions data to model the concentration of slurry at the soil surface, and a theoretical emissions model was developed based upon the infiltration of slurry into the soil and the subsequent reactions of NH_3 and NH_4^+ in the soil complex. Volatilization from the soil is related to soil ammonia-N concentration, soil temperature and atmospheric convection at the soil surface. The total WSU dairy waste application emissions from the sprinkler is 4600 kg. The annual field emissions derived from the tracer experiment data are estimated to be 1300 kg. Total waste application emissions for the WSU dairy are approximately 5900 kg or 34 kg NH_3 per milking cow per year (Rumburg et al., 2005).

Conclusion

The annual per cow emissions from the housing, lagoons, and waste application is estimated to be 40, 55, and 34 kg NH_3 cow^{-1} yr^{-1}, respectively. The total annual emissions from the WSU dairy is estimated to be 130 kg NH_3 cow^{-1} yr^{-1} which is significantly higher than the EPA NEI estimate of 38 kg NH_3 cow^{-1} yr^{-1} based upon European data (EPA, 2004). Differences in farming practices increased N excretions based upon higher milk production levels are most likely the reason for the large differences in emissions.

References

Eklund, B., 1999. Comparison of line- and point-source releases of tracer gases. Atmospheric Environment, 33, 1065-1071.

Environmental Protection Agency (EPA), 2004. National emissions inventory - ammonia emissions from animal husbandry operations - draft report.

Howard, T., Lamb, B.K., Bamesberger, W.L., Zimmerman, P.R., 1992. Measurement of hydrocarbon emissions fluxes from refinery wastewater impoundments using atmospheric tracer techniques. Journal of the Air & Waste Management Association, 42, 1336-1344.

Kaharabata, S.K., Schuepp, P.H., Desjardins, R.L., 2000. Estimating methane emissions from dairy cattle housed in a barn and feedlot using an atmospheric tracer. Environmental Science & Technology, 34, 3296-3302.

Lamb, B.K., McManus, J.B., Shorter, J.H., Kolb, C.E., Mosher, B., Harriss, R.C., Allwine, E., Blaha, D., Howard, T., Guenther, A., Lott, R.A., Siverson, R., Westberg, H., Zimmerman, P., 1995. Development of atmospheric methods to measure methane emissions from natural gas facilities and urban areas. Environmental Science & Technology, 29, 1468-1479.

Ludwig, F.L., Liston, E.M., Salas, L.J., 1983. Tracer techniques fro estimating emissions from inaccessible ground level sources. Atmospheric Environment, 17 (11), 2167-2172.

Mount, G.H., Rumburg, B., Havig, J., Lamb, B., Westberg, H., Yonge, D., Johnson, K., Kinciad, R., 2002. Measurement of atmopsheric ammonia at a dairy using differential optical absorption spectroscopy in the mid-ultraviolet. Atmospheric Environment, 36, 1799-1810.

Rumburg, B., Mount, G.H., Lamb, B., Westberg, H., Filipy, J., Yonge, D., Kincaid, R., Johnson, K. Measurement and Modeling of Atmospheric Flux of Ammonia from Dairy Milking Cow Housing. submitted to Atmospheric Environment 2006a.

Rumburg, B., Mount, G.H., Yonge, D., Lamb, B., Westberg, H., Neger, M., Filipy, J., Kincaid, R., Johnson, K. Atmospheric Flux of Ammonia from an Anaerobic Dairy Waste Lagoon. submitted to Atmospheric Environment 2006b.

Rumburg, B., Mount, G.H., Yonge, D., Lamb, B., Westber, H., Filipy, J., Bays, J., Kincaid, R., Johnson, K. Atmospheric Flux of Ammonia from Sprinkler Application of Dairy Waste. submitted to Atmospheric Environment 2005.

Shorter, J.H., McManus, J.B., Kolb, C.E., Allwine, E.J., Siverson, R., Lamb, B.K., Mosher, B.W., Harriss,R.C., Howard, T., Lott, R.A., 1997. Collection of leakage statistics in the natural gas system by tracer methods. Environmental Science & Technology, 31, 2012-2019.

Characterizing Ammonia Emissions from Potential Environmentally Superior Technologies for Hog Farms in Eastern North Carolina

I. C. Rumsey[1], V. P. Aneja[1], S. P. Arya[1], D-S. Kim[1], W.P. Robarge[2],
D.Dickey[3], L.S. Stefanski[3], H. Semunegus[1], H.L. Arkinson[1], K.S. Bajwa[1], W.W. Stephens[1], S.B. Goetz[1], L. Todd[4], K. Mottus[4], C.M. Williams[5]
[1]Department of Marine, Earth and Atmospheric Sciences, North Carolina State University, Raleigh, NC 27695-8208, USA
[2]Department of Soil Sciences, North Carolina State University, Raleigh, NC 27695-8208, USA
[3]Department of Statistics, North Carolina State University, Raleigh, NC 27695, USA
[4]Department of Environmental Science and Engineering, University of North Carolina-Chapel Hill, Chapel Hill, NC 27599-7431, USA
[5]Animal and Poultry Waste Management Center, North Carolina State University, Raleigh, NC 27695, USA

Abstract

Ammonia fluxes from anaerobic lagoons and hog houses at two conventional hog farms in eastern North Carolina were measured and analyzed to elucidate their temporal variability (seasonal and diurnal), and to derive regression relationships between ammonia flux and physico-chemical properties. These emissions are compared and contrasted with emissions from eleven (11) Environmentally Superior Technologies (EST's): (1) AHA Hunt farm- Sequencing Batch Reactor (SBR); (2) Barham farm – In-ground ambient temperature anaerobic digester/energy recovery/greenhouse vegetable production system; (3) Brown's of Carolina (BOC) Farm #93 – Upflow biofilteration system (EKOKAN) ; (4) Carrolls farm- Aerobic Blanket System (ABS); (5) Corbett #1,– Solids separation/gasification for energy and ash recovery centralized system (BEST); (6) Corbett # 2 farm – solid separation/ reciprocating water technology (ReCip); (7) Corbett #3,4- Solids separation/gasification for energy and ash recovery centralized system (BEST); (8) Goshen Ridge Farm- Solids separation/nitrification-denitrification/soluble phosphorus removal/solids processing system (Super Soils); (9) Harrells farm- Permeable Bio-cover System (PBS); (10) Howard farm – solid separation/ constructed wetlands; (11) Vestal farm- Recycling of Energy, Nutrients and Water System (RENEW). Measurements of ammonia flux were limited to two, two-week long periods, representing warm and cool seasons. A flow-through dynamic chamber system and two sets of open-path FTIR spectrometers measured continuously NH_3 fluxes from water holding structures and emissions from barns at the EST and conventional LST sites. For comparing the emissions from water holding structures at ESTs with those from lagoons at conventional sites under differing environmental conditions, a conventional statistical-observational model for lagoon NH_3 flux was developed, using multiple linear regression analysis of continuous NH_3 flux measurement data against the relevant environmental parameters measured at the two conventional farms, during two different seasons of year. Percentage reductions in NH_3 emissions from different components of the EST, as well as the whole farm were evaluated. These results were all normalized by the appropriate nitrogen excretion rate at the EST farm, as well as the appropriate conventional farm. This study shows that emissions were reduced by some ESTs under some of the environmental conditions. However, based on our evaluation results and analysis, and available information in the scientific literature, most of the evaluated alternative technologies may require additional technical modifications to be qualified as Environmentally Superior as defined by the NC Attorney Generals Agreements.

Introduction

Ammonia is the most biologically active reduced form of nitrogen, and is a very important alkaline constituent in the atmosphere. NH_3 is emitted by a large number of sources such as biomass burning, fossil fuel combustion, emissions from human excreta, soils and agricultural crops, synthetic fertilizers and animal waste (Oliver et al., 1996; Bouwman et al., 1997). In North Carolina, Hog waste releases an estimated 68,540 tons of nitrogen per year (Aneja et al., 1998). This relates to an increase in commercial swine facilities since the 1960's, and particularly since 1980, where the hog population has increased from about 3 million to a present population of around 10 million. Ammonia reacts with a variety of acidic atmospheric species, namely nitric acid (HNO_3), hydrochloric acid (HCl) and sulfuric acid (H_2SO_4). These reactions result in the formation of ammonium aerosols, the majority of which are of the fine particle size (<2.5 μm). There are a variety of environmental consequences associated with the presence of atmospheric NH_3 and its deposition including soil acidification, aquatic eutrophication and fine particulate matter formation. In North Carolina, the main environmental consequence is the nutrient loading of coastal plain river systems (Aneja et al., 1998).

The conventional lagoon and spray technology (LST), is the current system used in North Carolina to manage hog waste. It consists of anaerobic lagoons to store hog waste (~98% liquid) and effluent from the lagoons is sprayed on surrounding crop fields as a nutrient source (Aneja et al. 2001). Four distinct components and associated processes at LST release NH_3 to the atmosphere: (1) production houses, (2) waste storage and treatment systems such as lagoons, (3) land application through injection or spraying, and (4) biogenic emissions from soils and crops (Aneja et al., 2001). The need for developing sustainable solutions for managing the hog waste problem is critical for shaping the future of hog farms in North Carolina. As a result of this, an agreement between the North Carolina Attorney General and several commercial hog farming companies was reached to develop environmentally superior technologies (ESTs) for swine facilities. Program OPEN (Odor, Pathogens, and Emissions of Nitrogen) is an integrated study of the emissions of ammonia, odor and odorants and pathogens from potential environmentally superior technologies for swine facilities. Program OPEN aims to evaluate the EST's at swine facilities to determine if they would be able to substantially reduce atmospheric emissions of NH_3, pathogens and odor from their observed or estimated emissions from the conventional lagoon and spray technology used at selected LST hog farms (Moore farm and Stokes farm) in different (warm and cold) seasons or observation periods.

As part of Program OPEN, this study focuses on the emissions of NH_3 from the different components/processes involved in swine waste treatment, including hog houses, water-holding structures, and spray fields at eleven (11) potential EST sites: (1) AHA Hunt farm- Sequencing Batch Reactor (SBR); (2) Barham farm – In-ground ambient temperature anaerobic digester/energy recovery/greenhouse vegetable production system; (3) Brown's of Carolina (BOC) Farm #93 – Upflow biofilteration system (EKOKAN) ; (4) Carrolls farm- Aerobic Blanket System (ABS); (5) Corbett #1,– Solids separation/gasification for energy and ash recovery centralized system (BEST); (6) Corbett # 2 farm – solid separation/ reciprocating water technology (ReCip); (7) Corbett #3,4- Solids separation/gasification for energy and ash recovery centralized system (BEST); (8) Goshen Ridge Farm- Solids separation/nitrification-denitrification/soluble phosphorus removal/solids processing system (Super Soils); (9) Harrells farm- Permeable Bio-cover System (PBS); (10) Howard farm – solid separation/ constructed wetlands; (11) Vestal farm- Recycling of Energy, Nutrients and Water System (RENEW).

Methodology and Experimental Set-up at EST Sites

Ammonia flux measurements were made during two different seasons at two conventional (i.e. LST) and 15 EST swine farms in eastern North Carolina. Descriptions of the potential ESTs can be found on the website produced by the Air & Poultry Waste Management Center, NCSU, Raleigh, NC (A&PWMC, 2005. http://www.cals.ncsu.edu/waste_mgt/smithfield_projects/smithfieldsite.htm). NH_3 fluxes from water-holding structures and other area sources at the potential ESTs and conventional farms were measured by a dynamic flow-through chamber system. The emissions ofNH_3 from hog houses at the potential EST and conventional farms were measured by using Open-Path Fourier Transform Infrared (OP-FTIR) spectroscopy. Emissions were normalized by nitrogen excretion rates, which were based on hog population and feed data. The percentage reduction of NH_3 emissions are calculated for the whole farm for each potential EST.

Approach to Evaluate Ammonia Emissions from Water-Holding Structures at EST Farms

At each environmentally superior technology and conventional site, the estimated ammonia emissions were limited to about two two-week periods, representing both a warm and a cold season. However, since measurements at different sites were made at different times of the year, environmental conditions are likely to be different at different sites, even during a representative warm or cold season. There is a need for accounting for these differences in our relative comparisons of the various alternative and conventional technologies.

It is suggested that the estimated emissions from an environmentally superior technology for each measurement period be compared with the estimated emissions from conventional sites, after the later are adjusted for the average environmental parameters (most likely lagoon temperature and/or air temperature) observed at the former (EST) site. A rational basis for this adjustment for somewhat different environmental conditions could be the multiple regression model developed between ammonia emissions and measured environmental parameters at the two conventional sites. Such a comparison would not require highly uncertain extrapolations of emissions at EST sites beyond the two measurement periods. It would also provide sound basis for ranking the various ESTs based on their comparisons with conventional sites for each of the warm and cold seasons. An algorithmic flow diagram for the evaluation of NH_3 emissions from water holding structures at the EST farms is shown in Figure 1. Estimated ammonia emission from animal houses at a potential EST were compared to the estimated ammonia emissions from similar houses at a conventional farm (either Moore farm-tunnel ventilated, or Stokes farm-naturally ventilated), depending on the type of the house ventilation used at the EST farm, for the same season.

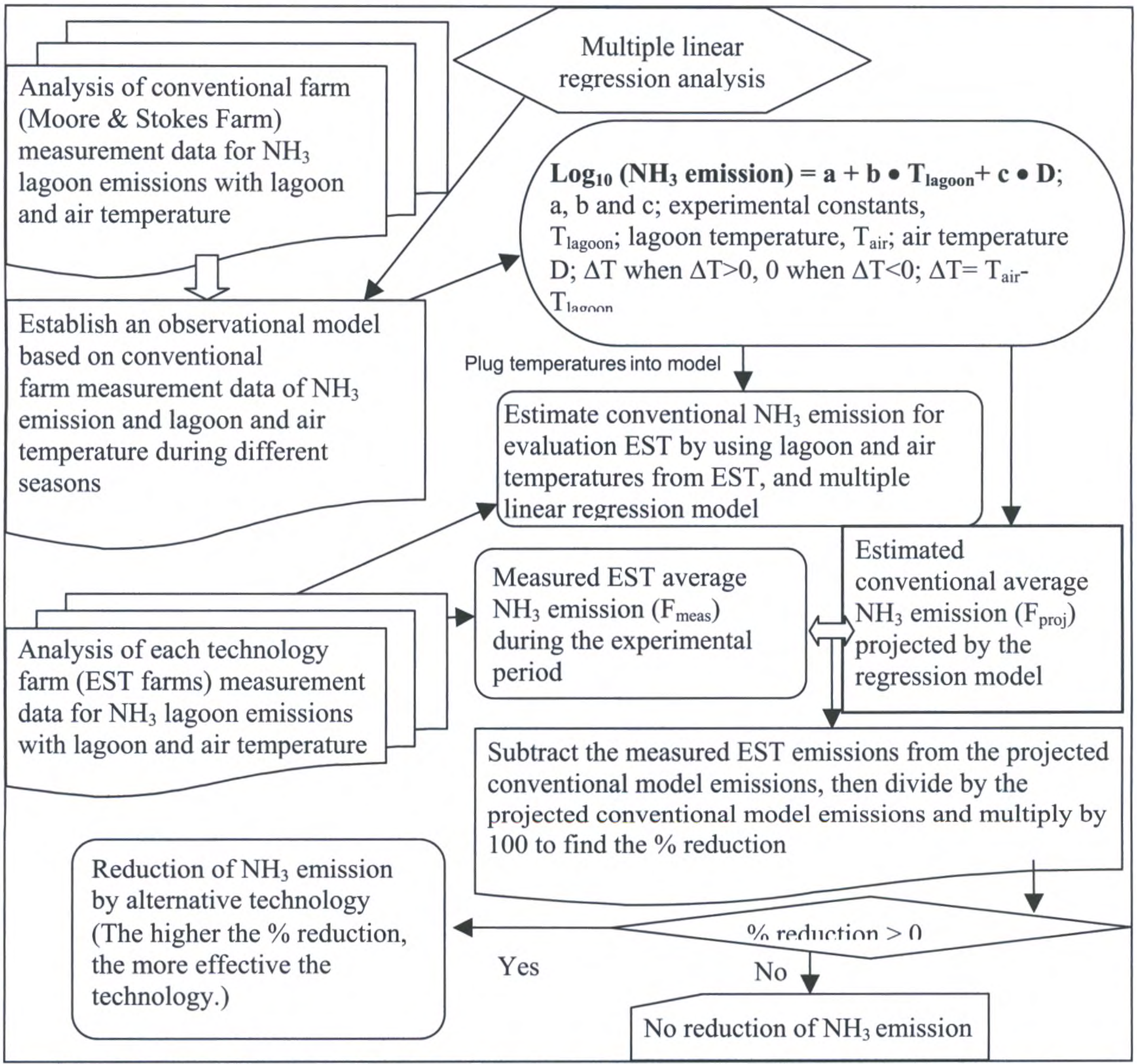


Figure 1. Algorithm flow chart for evaluation of Environmentally Superior Technology (EST) ammonia emissions from water-holding structures.

Results and Discussion

Nitrogen Excretion Rates Based on Animal Feeding

Animal weight, feed consumption and feed nutrient analysis data were used to estimate total nitrogen excretion rate produced at each experimental farm. Based on the nitrogen mass balance equation with given animal feed information (Table 1), nitrogen excretion rate (E) in unit of kg-N wk^{-1} (1000 kg-lw)$^{-1}$ was determined using the following equation:

$$E = \frac{F_c \times N_f \times (1-e_r)}{\overline{w}} \times 1000\,, \qquad (1)$$

where F_c is the feed consumed (kg pig^{-1} wk^{-1}), N_f is the fraction of nitrogen content in feed, e_r is the feed efficiency rate (ratio of average gain of nitrogen to nitrogen intake) [PigCHAMP, 1999], and $\overline{w}$ is the average live animal weight (kg/pig). Nitrogen excretion at each farm was calculated in term of the same units as NH_3-N emissions estimated from different components of the EST farm (e.g., lagoon, barns, etc) and shown in Table 1. All NH_3 emissions were normalized by nitrogen excretion rate (E) for the farm, and are called %E. Thus, %E represents the loss rate of ammonia from a source, as a percentage of N-excretion rate. A potential EST was evaluated by comparison of %E value from the EST ($\%E_{EST}$) farm to %E value from a baseline conventional farm ($\%E_{CONV}$), and percent reduction of NH_3-N can be estimated as

$$\%\,\mathrm{Re}\,duction = \frac{\left(\%E_{CONV} - \%E_{EST}\right)}{\%E_{CONV}} \times 100 \qquad (2)$$

Such percentage reductions can be estimated, separately for water holding structures, animal houses/barns, etc., as well as for the whole EST farm.

Multiple Regression Model for Lagoon NH_3 Flux Model Based on Conventional Farm Measurements

Two conventional farms were chosen for obtaining background measurements of NH_3, and evaluating EST's. Measurements at the conventional farms were conducted by using the same methods as used at EST's in two different seasons, including the fall (September – October, 2002; and winter January – February, 2003) measurement campaigns. Relationship between NH_3 flux and lagoon temperature, pH and TKN, as well as certain environmental parameters were examined over a relatively wide range of lagoon temperatures (~ 2°C to ~35°C) and lagoon air temperature differences that were observed during the field campaigns of fall and winter months at both the conventional farms.

The multiple regression equation based on flux measurement data from two conventional farms is given by

$$\mathrm{Log}_{10}F = 3.8655 + 0.04491(T_l) - 0.05946(D). \qquad (3)$$

Here, F denotes the average NH_3-N emission from the conventional lagoon in μg min^{-1}/1000 kg-lw, T_l is the lagoon temperature in °C, and D is a hot-air variable that is equal to zero if lagoon is warmer than air, but is equal to $\Delta T = T_a - T_l$ when $T_a > T_l$ and T_a is air temperature in °C at 10m height. This regression model was used to estimate the projected NH_3-N flux from lagoons at the LST baseline farms to compare with the measured NH_3-N flux from water-holding structures at an EST site, for the average values of T_l and D observed at the latter.

Table 1. The summary of animal weight, feed consumed, N-content and N-excretion at potential Environmentally Superior Technologies (ESTs) farms.

Farm Information	No. of pigs	Average pig weight (kg/pig)	Total pig weight (kg)	Feed consumed (kg/pig/wk)	N-content (%)	N-excretion (kgN/wk /1000kg-lw)
AHA Hunt						
Feb- Mar. 2004	10,909	59.2	645,813	13.97	2.38	4.08
Apr. 2004	12,106	66.1	800,206	14.46	2.34	3.29
Barham						
Apr. 2002	4,000	238.1	952,560	12.84	2.25^{Δ} /$3.09^{\dagger}$	1.65
Nov. 2002	4,000	238.1	952,560	15.92	2.38^{Δ} /$3.43^{\dagger}$	1.77
BOC # 93						
Apr. 2003	4,221	82.7	348,994	11.93	2.78	2.82
June. 2003	4,373	48.0	209,952	14.41	3.24	5.25
Carrolls						
Mar-Apr. 2004	6,332	59.2	374,854	12.89	2.56	3.90
June. 2004	6,095	59.7	363,872	13.21	2.67	4.13
Corbett # 1						
Oct. 2003	3,386	55.4	187,584	15.44	3.01	5.86
Dec. 2003	2,680	104.7	280,596	16.27	2.15	2.34
Corbett # 2						
Mar. 2003	1,249	98.5	123,054	16.27	2.76	3.19
June 2003	1,485	70.3	104,396	14.47	3.08	4.50
Corbett # 3,4						
Sep. 2003	3,620	106.8	386,616	12.74	2.03	1.69
Dec. 2003	4,122	76.3	314,509	8.98	2.49	2.05
Goshen Ridge						
Apr. 2003	3,519	93.4	328,675	17.03	2.67	3.41
Jan. 2004	3,814	61.9	235,972	14.67	2.42	4.02
Feb-Mar. 2004	3,138	99.8	313,172	16.18	2.49	2.83
Harrells						
Feb. 2004	2,071	110.3	228,431	18.62	2.26	2.67
Jun. 2004	2,417	86.5	209,071	16.25	2.06	2.71
Aug. 2004	2,615	50.9	133,103	12.61	2.98	5.17
Howard						
Jun. 2002	3,618	64.3	232,777	13.24	2.59	3.73
Dec. 2002	3,881	96.8	375,762	15.58	2.25	2.53
Vestal						
Mar. 2004	9,507	38.3	364,118	10.03	2.79	5.03
Aug. 2004	10,248	44.7	458,086	11.02	3.17	5.47

$^{\Delta}$N-content of the feed in gestation houses; †N-content of the feed in farrowing houses.

NH_3 Emissions from Water-Holding Structures

Water-holding structures emissions from 11 EST farms (AHA Hunt, Barham, BOC #93, Carrolls, Corbett #1, Corbett # 2, Corbett # 3,4, Goshen Ridge, Harrells, Howard, and Vestal) were calculated from measurements of NH_3 flux from EST farms, water-holding structure surface area, and farm production data (number of pigs, average pig weight, and feed consumed) during two approximately two-week long measurement periods, representing both warm and cool seasons. Emissions at the EST's are normalized live animal weight in the units of kg-N/wk/1000kg-lw. Farm production data for the EST farms are given in Table 1. Total estimated emissions are presented in Table 2.

Total emissions (kg-N/1000kg-lw/wk), normalized for nitrogen excretion were calculated for each experimental period for each EST farm (see Table 2). The emissions for the cool season for all EST farms ranged from 0.02-0.25, with a mean value of 0.13. The emissions for the warm season for all EST farms ranged from 0.02-2.14, with a mean value of 0.48. For nine out of eleven farms the total emissions were higher in the warm season than the cold season, the exceptions were Corbett # 3,4 and Goshen Ridge. This suggests that the seasonal variability of meteorological parameters such as lagoon and air temperature are significant factors in influencing NH_3 from hog facilities.

NH_3 Emissions from Housing Units

During each two-week sampling period at the EST farm sites, FTIR measurements were conducted to measure the NH_3 flux from the ventilation systems at the hog houses. Hog houses employ two different types of ventilation systems, mechanical or tunnel ventilation and natural ventilation. At Barham, BOC # 93, Carrolls and Howard farm mechanical ventilation was employed. For AHA Hunt, Corbett # 1,2,3,4, Goshen Ridge, Harrells, and Vestal farm the hog houses had natural ventilation. For the mechanically ventilated hog houses, the OP-FTIR spectrometers were placed adjacent to the fans of the houses and the open-path beam was placed at the centerline of the fan opening. The operation of the fans (monitoring when they are on or off) was performed during the entire sampling period. NH_3 emissions from hog houses were estimated based on FTIR path averaged NH_3 concentration measurement. The average NH_3 emission from the hog houses was calculated by multiplying the 15 minute path-averaged concentrations of NH_3 (measured as ammonia mg/m^3) across the midline of the fans exhausting the houses by the measured or factory calibrated fan rates (m^3/min).

To calculate the average NH_3 flux from the naturally ventilated houses, air-flow measurements were made by sampling at one location along each of the four sections of the building on the upwind side while the OP-FTIR was deployed. Each location was sampled for 30-60 seconds and the high and low readings recorded for all four locations over a 5-7 minute period of time. The high and low wind velocity readings were used to calculate the average wind velocity. The curtain opening for each section was measured and the volume of air per second (ventilation rate) flowing through the upwind side of the barn was calculated as the sum of curtain openings times the average wind velocities for the four sections of the building. The net ammonia concentrations associated with emissions from the building were obtained by subtracting the upwind readings from the downwind readings using the OP-FTIR and then converting the difference to concentrations of ammonia. A moving average was then applied to the concentration data to reduce the effect of wind variations (times when the wind deviated from the predominate direction). Flux from the building was obtained by multiplying net ammonia concentration times the corresponding ventilation rate. The flux calculations were then normalized by the total live weight of swine in the house (1000 kg-lw).

Table 3 shows the overall averaged NH_3 emissions estimated by OP-FTIR measurements from the hog barns at 11 EST farms during the sampling periods. Higher emissions from the barns were experienced during the warm period at ten out of eleven EST farms, the exception being Barham farm.

Table 2. Estimated NH_3 emissions from water-holding structures at EST farms during the experimental periods.

Farm name and sampling period	Total emission/ 1000kg-lw (kg-N/1000kg-lw/wk)	Farm name and sampling period	Total emission/ 1000kg-lw (kg-N/1000kg-lw/wk)	Farm name and sampling period	Total emission/ 1000kg-lw (kg-N/1000kg-lw/wk)
AHA Hunt Feb-Mar. 2004	0.17	**Corbett # 1 Oct. 2003**	0.33	**Goshen Ridge Feb-Mar. 2004**	0.04
AHA Hunt Apr. 2004	0.56	**Corbett # 1 Dec. 2003**	0.12	**Harrells Feb. 2004**	0.07
Barham Apr. 2002	0.31	**Corbett # 2 Mar. 2003**	0.18	**Harrells Jun. 2004**	2.14
Barham Nov. 2002	0.07	**Corbett # 2 June. 2003**	0.35	**Harrells Aug. 2004**	1.55
BOC # 93 Apr. 2003	0.02	**Corbett # 3,4 Sep. 2003**	0.10	**Howard Jun. 2002**	1.2
BOC # 93 Jun. 2003	0.03	**Corbett # 3,4 Dec. 2003**	0.23	**Howard Dec. 2002**	0.25
Carrolls Mar-Apr. 2004	0.22	**Goshen Ridge Apr. 2003**	0.02	**Vestal Mar. 2004**	0.02
Carrolls June-July. 2004	0.23	**Goshen Ridge Jan. 2004**	-♣	**Vestal Aug. 2004**	0.05

♣ Emissions from water-holding structures were not measured during this period.

Table 3. Estimated NH_3 emission from the hog houses at EST farms during the experimental periods (OP-FTIR measurements).

EST farms	Sampling periods	Barn emissions (kg-N/1000kg-lw/wk)
AHA Hunt	February-March, 2004	0.01
	April, 2004	0.71
Barham	April, 2002	0.34
	November, 2002	0.49
BOC # 93	April, 2003	0.57
	June, 2003	1.29
Carrolls	March-April, 2004	0.98
	June-July, 2004	1.15
Corbett # 1	October, 2003	0.16
	December, 2003	0.008
Corbett # 2	March, 2003	0.12
	June, 2003	0.49
Corbett # 3,4	September, 2003	0.48
	December, 2003	0.16
Goshen Ridge	April-May, 2003	0.72
	January, 2004	0.01
Harrells	January-February, 2004	0.05
	June, 2004	0.27
Howard	June, 2002	1.42
	December, 2002	0.85
Vestal	March, 2004	0.07
	August, 2004	0.75

Evaluation of Ammonia Emissions from EST Farms

In order to evaluate the potential percentage reduction of NH_3 emissions for the whole farm, both water-holding structures and barns were taken into consideration. To achieve this, potential EST emissions were compared to projected emissions at the conventional LST farms. The projected emissions from the LST farms were adjusted to the environmental conditions i.e. air and lagoon temperature, which have been determined to be statistically significant to ammonia emissions. The projected emissions from the barns are the measured emissions for the corresponding season from the two conventional farms, Moore and Stokes, respectively. Moore farm emissions are used to project emissions for farms with tunnel or forced ventilation. Stokes farm emissions are used to project emissions for farms with natural ventilation. To calculate the total % reduction, the sum of projected emissions and measured emissions are taken for the water-holding structures and barns are taken. These numbers are then used to calculate total % reduction using the same process that is applied individually for water-holding structures and barns.

Table 4 shows the $\%E_{EST}$, $\% E_{CONV}$ and the % reduction from the EST farms. Out of eleven farms, five show % reduction in NH_3 emissions for both experimental periods. The BEST technology employed at Corbett # 1 was the most successful of these, with a reduction of 71.8% in the warm season, and 66.0% in the cool season. The next largest % reduction was the ISSUES-RENEW system employed at Vestal farm with % reductions of 54.0 and 31.1 for cool and warm seasons, respectively. Other technologies that reduced ammonia emissions in both seasons included the EKOKAN technology at BOC # 93 farm, which had % reductions of 23.5 and 43.3 for the cool and warm seasons, respectively. Another technology with reductions in both seasons was the ISSUES-PBS, which was located at Carrolls farm. There was a small reduction of 8.1% in the cool season, and a larger reduction of 49.5% in the warm season. The ReCIP technology, which was employed at Corbett # 2 farm produced small reductions in both seasons, 20.9% in the cool season and 9.7% in the warm season.

Five technologies had mixed results, reducing emissions in one experimental period, but enhancing in the other. The largest variance in % reductions was the PCS technology located at Harrells farm. In the cool season there was a % reduction of 69.4, but in the warm season, there was a % reduction of -109.9. The Super Soils technology, which was located at Goshen Ridge farm also had a large variance in % reductions. A huge reduction of 98.5 % was calculated for the cool season, but for the warm season % reduction was -1.9. The Sequencing Batch Reactor employed at AHA Hunt farm was also moderately successful in the cool season with a % reduction of 67.2%, but emissions were slightly enhanced in the warm season (% reduction = -4.9). The BEST technology at Corbett # 3,4 showed the same seasonal pattern. % reduction values were 17.0 and -29.2 for the cool and warm seasons, respectively. The technology at Barham farm had a marginal reduction of 2.5 % in the cool season. In the warm season, emissions were enhanced with a % reduction of -11.9%.

The constructed wetlands at Howard farm was the only technology that showed enhancement of emissions in both seasons. The technology had % reduction values of -50.9 and -62.6, respectively.

Conclusions

Eleven potential ESTs were evaluated to determine if they would substantially reduce atmospheric emissions of ammonia at the swine facilities from their estimated emissions from the conventional lagoon and spray technology used at selected LST hog farms in two different experimental periods. Five out of eleven potential ESTs showed % reduction in NH_3 emissions for both periods. The BEST technology employed at Corbett # 1 had the highest % reductions of 71.8% and 66.0% for the warm and cool seasons, respectively.

Table 4. % E_{EST}, %E_{CONV} and % reduction values from the EST farms during the experimental periods

EST Farms	Sampling Periods	Emission Sources	%E_{EST}	% E_{EST} (WHS + house)	%E_{CONV}	% E_{CONV} (lagoon + house)	% reduction
AHA Hunt farm	Feb-Mar ,04	WHS	4.2	4.4	3.4	13.4	67.2
		house	0.2		10.0		
	Apr,04	WHS	17.0	36.5	24.8	34.8	-4.9
		house	19.5		10.0		
Barham farm	April -02	WHS	18.8	39.4	11.3	35.2	-11.9
		house	20.6		23.9		
	Nov-02	WHS	4.0	31.7	9.7	32.5	2.5
		house	27.7		22.8		
BOC # 93	Apr,03	WHS	8.2	28.4	14.3	37.1	23.5
		house	20.2		22.8		
	Jun,03	WHS	11.0	35.6	38.9	62.8	43.3
		house	24.6		23.9		
Carrolls farm	Mar-Apr,04	WHS	5.6	30.7	10.6	33.4	8.1
		house	25.1		22.8		
	Jun-Jul,04	WHS	5.6	33.4	42.2	66.1	49.5
		house	27.8		23.9		
Corbett # 1 farm	Oct,03	WHS	5.6	8.3	19.4	29.4	71.8
		house	2.7		10.0		
	Dec,03	WHS	5.1	5.4	5.9	15.9	66.0
		house	0.3		10.0		
Corbett # 2 farm	Mar-03	WHS	11.0	14.8	8.7	18.7	20.9
		house	3.8		10.0		
	Jun-03	WHS	18.0	28.9	22.0	32.0	9.7
		house	10.9		10.0		
Corbett # 3,4 farm	Sep,03	WHS	13.6	42.0	22.5	32.5	-29.2
		house	28.4		10.0		
	Dec,03	WHS	4.9	12.7	5.3	15.3	17.0
		house	7.8		10.0		
Goshen Ridge	Apr-May,03	WHS	0.6	21.7	11.3	21.3	-1.9
		house	21.1		10.0		
	Feb-Mar ,04 Jan, 04	WHS	0.1	0.3	9.7	19.7	98.5
		house	0.2		10.0		
Harrells	Jan-Feb ,04	WHS	2.6	4.5	4.7	14.7	69.4
		house	1.9		10.0		
	Jun,04	WHS	79.0	89.0	32.4	42.4	-109.9
		house	10.0		10.0		
	August,04	WHS	30.0	-	29.6		
Howard farm	Jun-02	WHS	31.9	70.0	22.5	46.4	- 50.9
		house	38.1		23.9		
	Dec-02	WHS	9.5	43.1	3.7	26.5	-62.6
		house	33.6		22.8		
Vestal	Mar, 04	WHS	7.8	9.2	10.0	20.0	54.0
		house	1.4		10.0		
	August,04	WHS	19.6	33.3	38.3	48.3	31.1
		house	13.7		10.0		

WHS = Water-holding structures

References

A&PWMC, 2005. Development of Environmentally Superior Technologies for Swine WasteManagement per Agreements Between the North Carolina Attorney General, Smithfield Foods, Premium Standard Farms, and Frontline Farmers. Animal & Poultry Waste Management Center, NCSU, Raleigh. http://www.cals.ncsu.edu/waste_mgt/smithfield_projects/ smithfieldsite.htm (February 1, 2006).

Aneja, V.P., B.P. Malik, Q. Tong, D. Kang, and J.H. Overton, 2001. Measurement and Modeling of Ammonia Emissions at Waste Treatment Lagoon-Atmospheric Interface. *Water, Air, and Soil Pollution: Focus* 1: 177-188.

Aneja, V.P., G. Murray, and J. Southerland, 1998. *Proceedings of the Workshop on Atmospheric Nitrogen Compounds: Emissions, Transport, Transformation, Deposition, and Assessment*. North Carolina State University, Raleigh, NC, pp299.

Bouwman, A.F., D.S. Lee, W.A.H. Asman, F.J. Dentener, K.W. Van der Hoek, and J.G.J. Oliver. 1997. A Global High-Resolution Emission Inventory for Ammonia. *Global Biogeochemical Cycles*, 11: 561-587.

Olivier, J.G.J., Bouwman, A.F., Van der Maas, C.W.M., Berdowski, J.J.M., Veldt, C., Bloos, J.P.J., Visschedijk, A.J.H., Zandveld, P.Y.J. and J.L. Haverlag, 1996. Description of EDGAR Version 2.0: A set of global emission inventories of greenhouse gases and ozone depleting substances for all anthropogenic and most natural sources on a per country basis and on 1°×1° grid. RIVM Techn. *Report 771060 002; TNO-MEP Report R96/119.* , National Institute of Public Health and the Environment/Netherlands Organization for Applied Scientific Research, Bilthoven.

PigCHAMP, 1999. *National Summary Reports*. PigCHAMP Inc., Ames, IA.

Soil Carbon Dioxide Emission as Influenced by Irrigation, Tillage, Cropping System, and Nitrogen Fertilization

U.M. Sainju*, J.D. Jabro, and W.B. Stevens
USDA-ARS-NPARL, 1500 North Central Avenue, Sidney, MT
* Corresponding author (usainju@sidney.ars.usda.gov)

Abstract

Soil and crop management practices can influence CO_2 emission from crop and grasslands and therefore on global warming. We examined the effects of two irrigation systems (irrigated vs. non-irrigated) and six management practices [no-till malt barley (*Hordeum vulgaris* L.) with 67 or 134 kg N ha^{-1} (NTBFN), no-till malt-barley with 0 kg N ha^{-1} (NTBON), conventional-till malt barley with 67 or 134 kg N ha^{-1} (CTBFN), conventional-till malt barley with 0 kg N ha^{-1} (CTBON), no-till pea (*Pisum sativum* L.) with 0 kg N ha^{-1} (NTPON), and undisturbed alfalfa (*Medicago sativa* L.) and grasses with 0 kg N ha^{-1} (UAGON)] on soil surface CO_2 flux and soil temperature and water content at the 0 to 15 cm depth. Weekly CO_2 flux, soil temperature, and soil water content were monitored during the crop growing season from May to November 2005 in Lihen sandy loam (sandy, mixed, frigid, Entic Haplustolls) in western North Dakota. Irrigation increased CO_2 flux by 27% compared with non-irrigation by increasing soil water content during dry periods. Similarly, tillage increased CO_2 flux by 58% compared with non-tillage by increasing soil temperature. The CO_2 flux was 1.5 to 2.5-fold greater in tilled than in non-tilled treatments following heavy rain or irrigation. Nitrogen fertilization increased CO_2 flux compared with no N fertilization in 2 out of 17 measurements while cropping system did not influence CO_2 flux. The CO_2 flux in undisturbed alfalfa and grasses was similar to that in no-tilled crops. The CO_2 flux was linearly related with soil temperature and daily average air temperature at the time of CO_2 measurement. Tillage followed by heavy rain or irrigation during the crop growing season drastically increased CO_2 flux in the coarse-textured soil previously managed under Conservation Reserve Program (CRP) planting for more than 20 yr.

Introduction

Global warming due to increased concentration of greenhouse gases, such as CO_2, in the atmosphere is a major concern. One of the significant sources of CO_2 emission is from agricultural practices, which contribute about 25% of total anthropogenic emissions (Duxbury, 1994; 1995). Soil can act as both source and sink of atmospheric CO_2. The CO_2 fixed in plant biomass through photosynthesis can be stored in the soil as organic C by converting plant residue into soil organic matter after the residue is returned to the soil. While management practices, such as tillage, can increase CO_2 emission from the soil by disrupting soil aggregates, increasing aeration, incorporating plant residue, and oxidizing soil organic C (Reicoscky and Lindstrom, 1993; Beare et al., 1994; Jastrow et al., 1996), no-tillage and increased cropping intensity can increase C storage in the soil (Lal et al., 1995; Paustian et al., 1995). Respiration by plant roots and soil microflora and fauna also contribute a major portion of CO_2 emission from the soil (Rochette and Flanagan, 1997; Curtin et al., 2000). The CO_2 emission from the soil to the atmosphere is the primary mechanism of C loss from the soil (Parkin and Kaspar, 2003) and provides an early indication of C sequestration in the soil when changes in soil organic C due to management practices are not detectable within a short period (Fortin et al., 1996; Grant, 1997).

Reduced tillage is regarded as one of the most effective agricultural practices to reduce CO_2 emission and sequester atmospheric C in the soil (Kern and Johnson, 1993; Lal and Kimble, 1997; Curtin et al., 2000; Al-Kaisi and Yin; 2005). Decreased tillage intensity reduces soil disturbance and microbial activities, which in turn, lowers CO_2 emission (Lal and Kimble, 1997; Curtin et al., 2000). Moreover, no-till with surface residue can further reduce CO_2 emission compared to no-till without residue (Al-Kaisi and Yin, 2005). In contrast, increased tillage intensity increased CO_2 emission by increasing aeration due to increased soil disturbance (Roberts and Chan, 1990) and to physical degassing of dissolved CO_2 from the soil solution (Reicoscky and Lindstrom, 1993; Jackson et al., 2003). The effect of tillage on CO_2 emission, however, was short-lived (<24 h) (Reicoscky and Lindstrom, 1993; Ellert and Janzen, 1999). Rewetting of dry soil due to irrigation or rainfall increased CO_2 flux by increasing microbial activities, C mineralization, and respiration (Sparling and Ross, 1988; Van Gestel et al., 1993; Calderon and Jackson, 2002).

Cropping system can influence CO_2 emission by affecting on the quality and quantity of residue returned to the soil (Curtin et al., 2000; Al-Kaisi and Yin; 2005; Amos et al., 2005). Increased above- and belowground biomass production can increase the amount of residue returned to the soil (Sainju et al., 2005), thereby increasing CO_2 flux (Franzluebbers et al., 1995; Curtin et al., 2000; Al-Kaisi and Yin; 2005). Increased belowground biomass production can also increase root and rhizosphere respiration, thereby increasing CO_2 flux (Amos et al., 2005). Similarly, residue quality, such as C/N ratio, can alter the decomposition rate of residue (Kuo et al., 1997; Sainju et al., 2002), thereby influencing CO_2 emission (Al-Kaisi and Yin, 2005). Nitrogen fertilization had little effect on CO_2 emission from the soil surface (Rochette and Gregorich, 1998; Wagai et al., 1998; Amos et al., 2005).

Management practices can affect soil temperature and water content (Curtin et al., 2000; Al-Kaisi and Yin, 2005) which directly influence CO_2 emission rates (Bajracharya et al., 2000b; Parkin and Kaspar, 2003; Amos et al., 2005). For example, tillage can dry the soil but no-till can increase soil water content and reduce soil temperature because of residue accumulation at the soil surface (Curtin et al., 2000; Calderon and Jackson, 2002; Al-Kaisi and Yin, 2005). Similarly, cropping system can reduce soil temperature by providing shade with increased biomass growth but can reduce soil water content due to increased evapotranspiration (Amos et al., 2005).

Little is known about the combined effects of irrigation, tillage, cropping system, and N fertilization on CO_2 flux in transitional land that has been converted from Conservation Reserve Program (CRP) planting to agricultural use. The CO_2 flux from the conversion of no-till to conventional till, from dryland to irrigated, or from less intensive to more intensive cropping system could be different from the flux obtained during the reverse trend of land conversion. We hypothesized that the individual and combined uses of irrigation, tillage, cropping system, and N fertilization have differential effects on soil surface CO_2 flux in the transitional land. Our objectives were to: (1) determine the effects of irrigation, tillage, cropping system, and N fertilization on soil surface CO_2 flux during the crop growing season in land converted from CRP to agricultural use, and (2) compare CO_2 fluxes in the agricultural system and undisturbed alfalfa and grasses.

Materials and Methods

The experiment was conducted in 2005 on a transitional land that was converted from CRP planting to agricultural use in the Nesson Valley in western North Dakota. The resident vegetation was dominated by alfalfa (*Medicago sativa* L.), crested wheatgrass [*Agropyron cristatum* (L.) Gaertn], and western wheatgrass [*Pascopyron* smithii (Rydb.) A. Love]. The soil was a Lihen sandy loam (sandy, mixed, frigid, Entic Haplustolls). The soil sampled in April 2005 before the initiation of the experiment had 720 g kg^{-1} sand, 200 g kg^{-1} silt, 80 g kg^{-1} clay, 12.5 g kg^{-1} organic C, and 7.7 pH at the 0 to 20 cm depth.

The treatments consisted of two irrigation systems (irrigated vs. nonirrigated) and management practices consisting of six combined treatments of tillage, cropping system, and N fertilization rates [no-till malt barley with N fertilization (NTBFN), no-till malt barley with no N fertilization (NTBON), conventional-till malt barley with N fertilization (CTBFN), conventional-till malt barley with no N fertilization (CTBON), no-till pea with no N fertilization (NTPON), and undisturbed alfalfa and grasses with no N fertilization (UAGON)]. The N fertilization rate for irrigated no-till and conventional-till malt barley was 134 kg N ha^{-1} and for non-irrigated no-till and conventional-till malt barley was 67 kg N ha^{-1}. For conventional-till malt barley, plots were tilled with a rototiller to a depth of 10 cm in April 2005 to prepare seed bed and to kill the resident vegetation. The UAGON treatment consisted of alfalfa and grasses that were continued from previous vegetation. Resident vegetation in no-till plots, except the undisturbed alfalfa and grasses treatment, was killed by applying glyphosate [N-(phosphonomethyl) glycine)] at 3.5 kg a.i. (active ingredient) ha^{-1} prior to crop planting. The experiment was arranged in randomized complete block with irrigation as the main plot and management practices as the split-plot treatment. Each treatment had three replications. The size of the experimental unit was 10.6 x 3.0 m.

Figure 1. Soil surface CO_2 flux measurement with Environmental Gas Monitor chamber and data logger

In May 2005, malt barley (cv. Certified Tradition, Busch Agricultural Resources, Fargo, ND) was planted at 3.8 cm depth at the rate of 90 kg ha^{-1} in the irrigated treatment and 67 kg ha^{-1} in the non-irrigated treatment with a no-till drill. Similarly, pea (cv. Majorete, Macintosh Seed, Havre, MT) was planted at 200 kg ha^{-1} in irrigated and non-irrigated treatments. In irrigated malt barley, half of N fertilizer as urea (or 67 kg N ha^{-1}) was banded at planting and other half was broadcast at 4 wk after planting. In non-irrigated malt barley, all N fertilizer was banded at planting. The P fertilizer as triple super phosphate and K fertilizer as muriate of potash were banded at 25 kg P ha^{-1} and 21 kg K ha^{-1}, respectively, each to malt barley and pea at planting. The amount of N, P, and K fertilizers to malt barley and pea were based on soil test and crop requirement. No fertilizers were applied to alfalfa and grasses in UAGON treatment which resembled undisturbed CRP planting. However, as with other management practices, half of UAGON treatment in split-plot arrangement received irrigation and other half did not. Appropriate types and amounts of herbicides and pesticides were used to control weeds and pests during growth and after harvest of malt barley and pea. In irrigated plots, water was applied at amounts ranged from 10 to 25 mm of rain from 17 June to 14 July 2005 (a total of 87 mm) based on soil moisture content. In July and August, malt barley and pea were hand harvested from an area of 0.5 x 4 m^2 and straw (leaves and stems) was returned to the soil.

Immediately after planting, soil surface CO_2 flux was measured weekly in all treatments during and after the crop growing season from May to November 2005 until the ground froze. All measurements were made between 9 A.M. and 12 A.M. of the day to reduce variability in CO_2 flux due to diurnal changes in temperature (Parkin and Kaspar, 2003). The CO_2 flux was measured with an Environmental Gas Monitor chamber attached to a data logger (model no. EGM-4, PP System, Haverhill, MA) (Figure 1). The chamber is 15 cm tall, 10 cm in diameter, and has capacity to measure CO_2 flux from 0 to 9.99 g CO_2-C m^{-2} h^{-1}. The chamber was placed at the soil surface for 2 min in each plot until CO_2 flux measurement was recorded in the data logger. A flag was placed as a marker in the plot where CO_2 flux was measured throughout the study period. At the time of CO_2 measurement, soil temperature near the chamber was measured from a depth of 0 to 15 cm from a probe attached to the data logger. Similarly, gravimetric soil water content was measured near the chamber by collecting soil sample from the 0 to 15 cm depth with a probe (2.5 cm diameter) every time CO_2 flux was measured. The moist soil was oven-dried at 110°C and water content was determined. Daily average air temperature and total rainfall during the study period were collected from a meteorological station located at 0.8 km from the study site.

Data for CO_2 flux, soil temperature, and soil water content were analyzed by using the Analysis of Repeated Measures procedure in the MIXED model of SAS (Littell et al., 1996). Irrigation was considered as the main plot, management practices as the split plot, and date of measurement as the repeated measure treatment in the analysis. Means were separated by using the least square means test when treatments and interactions were significant. Statistical significance was evaluated at $P \leq 0.05$ unless otherwise mentioned. Regression analysis was done between CO_2 flux and average daily air temperature, soil temperature, and soil water content to determine their relationships.

Results and Discussion

Climate

Rainfall and air temperature influence soil water content and soil temperature, which in turn, directly influence CO_2 flux from the soil surface (Bajracharya et al., 2000; Curtin et al., 2000; Parkin and Kaspar, 2003). Daily total rainfall was higher in May and June 2005 during the critical growing period of crops, and then declined from July to November as the crop matured (Figure 2A). Since air temperature is more correlated with CO_2 flux than soil temperature (Parkin and Kaspar, 2003), daily average air temperature was also measured to determine its influence on soil CO_2 flux, besides measuring soil temperature and water content. As expected, daily average air temperature increased from May to August and then declined (Figure 2B).

Effect of Irrigation

Soil temperature at the 0 to 15 cm depth followed a trend similar to that of air temperature (Figure 3B) but was less variable (Figure 2B). Irrigation resulted in a significant ($P \leq 0.001$) interaction with date of measurement on soil temperature, water content, and CO_2 flux (Table 1). Soil temperature was lower in irrigated than in non-irrigated treatment in Days 176, 195, 202, 208, and 217 of the year (Figure 2B). In contrast, soil water content at the 0 to 15 cm depth was higher in irrigated than in non-irrigated treatment in Days 176, 202, 208, and 217 of the year. It is not surprising to observe lower soil temperature and higher water content with irrigated treatment in these days when irrigation was done to meet the crop water requirement during periods of higher air temperature and lower rainfall (Figures 2A, 2B, 3C). Increased water content and evaporation from the soil surface probably reduces soil temperature, as wet soil is slower to change in temperature than dry soil (Bajracharya et al., 2000b; Parkin and Kaspar, 2003). Increased rainfall increased soil water content of both irrigated and non-irrigated treatments in May and June when no irrigation was done.

The CO_2 flux increased from 31 kg CO_2-C ha^{-1} d^{-1} in Day 122 to 427 kg CO_2-C ha^{-1} d^{-1} in Day 188 of the year, after which it declined (Figure 3A). The flux was higher in irrigated than in non-irrigated treatment in Days 158, 176, 179, and 188. Increased soil moisture content rather than a difference in soil temperature (Figures 3B and 3C) probably increased CO_2 flux with irrigation during these periods. Increased CO_2 flux after irrigation or after a heavy rain in dry soil resulting from increased C mineralization have been known (Howard and Howard, 1993; van Gestel et al., 1993; Curtin et al., 2000; Calderon and Jackson, 2002). Since a major portion of soil CO_2 flux is contributed by respiration from plant roots, rhizosphere, and microbial flora and fauna (Rochette and Flanagan, 1997; Curtin et al., 2000), it is likely that irrigation of dry soil increased microbial activities and CO_2 emission. Averaged across measurement dates, irrigation decreased soil temperature by 0.8°C and increased soil water content by 6% and CO_2 flux by 15% (Table 2).

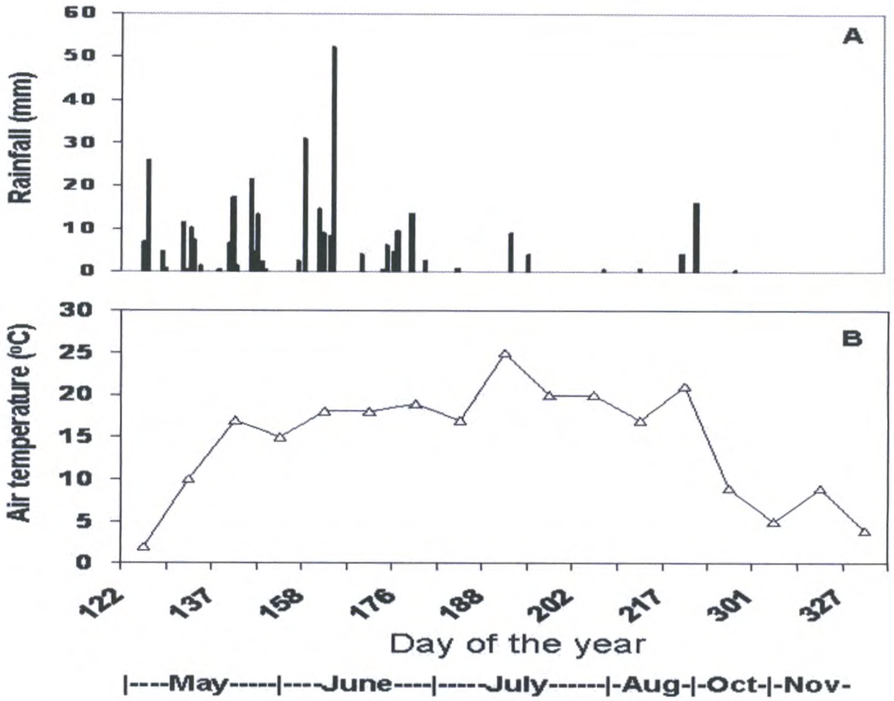


Figure 2. Daily (A) total rainfall and (B) average air temperature at the time of CO_2 flux measurement from May to November 2005 in western North Dakota.

Table 1. Analysis of variance for soil surface CO_2 flux and soil temperature and water content at the 0 to 15 cm depth.

Source	Soil CO_2 flux	Soil temperature	Soil water content
Irrigation (I)	**	*	*
Management practices (M)	***	***	***
I × M	NS†	**	NS
Date of measurement (D)	***	***	***
I × D	***	**	***
M × D	***	***	**
I × M × D	NS	*	*

* Significant at $P \leq 0.05$.
** Significant at $P \leq 0.01$.
*** Significant at $P \leq 0.001$.
† Not significant.

Table 2. Effects of irrigation and management practices on soil surface CO_2 flux and soil temperature and water content at the 0 to 15 cm depth averaged across dates of measurement.

Irrigation	Management practices†	Soil surface CO_2 flux ($kg\ CO_2\text{-}C\ ha^{-1}\ d^{-1}$)	Soil temperature (°C)	Soil water content ($g\ kg^{-1}$)
Irrigated		194a‡	14.8a	97.3a
Non-irrigated		169b	15.6b	91.9b
	NTBFN	164b	15.0c	97.1a
	NTPON	155b	15.2b	96.5a
	CTBFN	250a	15.0c	95.3a
	UAGON	150b	15.3b	86.9b
	NTBON	145b	15.2b	97.9a
	CTBON	242a	15.5a	94.7a

† Management practices are CTBFN, conventional-till malt barley with 67 or 134 kg N ha^{-1}; CTBON, conventional-till malt barley with 0 kg N ha^{-1}; NTBFN, no-till malt barley with 67 or 134 kg N ha^{-1}; NTBON, no-till malt barley with 0 kg N ha^{-1}; NTPON, no-till pea with 0 kg N ha^{-1}; and UAGON, undisturbed alfalfa and grasses with 0 kg N ha^{-1}.

‡ Numbers followed by different letters within a set of a column are significantly different at $P \leq 0.05$ by the least square means test.

Effect of Management Practices

Differences in soil temperature, water content, and CO_2 flux between management practices at some measurement dates resulted in a significant ($P \leq 0.01$) management practice × measurement date interaction (Table 1). Soil temperature was higher in UAGON than in all other practices, except in NTBON, in Day 176 (Figure 4B). Soil temperature was also higher in CTBON than in most of the other management practices from Day 195 to 217. While soil water content was consistently lower in UAGON from Day 137 to 158 and from Day 202 to 217, it was higher in NTPON from Day 144 to 166, higher in NTBON in Days 176, 179, 202, and higher in NTBFN in Days 208 and 217 than in most of the other management practices (Figure 4C). Increased soil temperature and decreased water content in UAGON was probably due to greater water requirement of alfalfa and grasses. Similarly, increased soil temperature and reduced water content in CTBON was likely a result of drying up of soil due to tillage and reduced plant growth. The plant biomass and yield were lower in CTBON (data not shown), probably due to low N availability, since N fertilizer was not applied in this treatment. Tillage can result in drying up of soil due to increase in water vapor flux (Kessavalou et al., 1998), and decreased shading with reduced plant growth can increase soil temperature (Amos et al., 2005). In contrast, increased soil water content and decreased soil temperature in NTBON, NTPON, and NTBFN was probably due to less soil disturbance, followed by residue accumulation at the soil surface. Increased soil water content and decreased soil temperature in no-till than in conventional till have been reported by several researchers (Curtin et al., 2000; Calderon and Jackson, 2002; Al-Kaisi and Yin, 2005).

Management practices, especially tillage, increased CO_2 flux more than the irrigation treatment by ≥159 kg CO_2-C ha^{-1} d^{-1} at certain measurement dates (Figure 4A). Drastic increases occurred in tilled treatments immediately following heavy rain or irrigation in Days 130, 158, 188, 195, 202, and 208 when CO_2 fluxes in CTBFN and CTBON were 1.5 to 2.5-fold higher than in other treatments. The flux was also higher in NTBFN and NTPON than in UAGON in Day 179 but was higher in UAGON than in NTPON in DAY 195. Similarly, the flux was higher in NTBFN than in NTBON and UAGON in Day 188 and was higher in CTBFN than in CTBON in Day 202.

The CO_2 flux of as much as 300 kg CO_2-C ha^{-1} d^{-1} following tillage and heavy rain in dry soil in northern Great Plains had been known (Curtin et al., 2000). Our results of CO_2 flux of as much as 600 kg CO_2-C ha^{-1} d^{-1} in CTBFN and CTBON following heavy rain or irrigation seems to be extreme but could be due to tillage in the soil that had been under CRP management for more than 20 years before the study was initiated. Tillage can result in an immediate short-term outburst of CO_2 due to decrease in partial pressure of CO_2 in soil air, followed by disturbance in soil aggregation and pores, and sudden release of CO_2 from the soil solution (Reicoscky et al., 1995; Ellert and Janzen, 1999; Rochette and Angers, 1999). The effect of tillage on the CO_2 flux was short-lived, as suggested by others (Reicosky and Lindstrom, 1993; Al-Kaisi and Yin, 2005), because fluxes were lower after tillage in the initial periods of study in Days 122, 137, and 144, regardless of management practices (Figure 4A). However, fluxes triggered in tilled treatments in Days 130, 158, 188, 195, 202, and 208 immediately after heavy rains and irrigation in dry soil. Under raindrop and irrigation impact, the soil surface may have sealed in the tilled treatment, temporarily trapping CO_2. As the water drained and soil surface dried, trapped CO_2 may have released in a brief, intense burst, as suggested by Curtin et al. (2000). Another possibility would be that tillage accompanied by rainfall and irrigation could have increased microbial activities, thereby increasing C mineralization and CO_2 flux. Better soil structure along with increased surface residue cover could have reduced surface sealing in the no-till treatment. Changes in soil organic C content due to short-term burst of CO_2 from tillage would be minor because only a small portion of soil organic matter mineralizes during the initial phase of tillage (Curtin et al., 2000; Al-Kaisi and Yin, 2005).Tillage also may have indirectly influenced CO_2 flux during these periods by increasing soil temperature (Figure 4B). Although CO_2 flux can be influenced by a range of factors including soil temperature, water content, wind speed, and CO_2 concentration gradient, greater fluxes in this study than those obtained by Curtin et al. (2000) could also be due to a difference in soil texture. Soil texture in our study was sandy loam and in their study was silt loam. The CO_2 flux increases with decrease in clay content (Parkin and Kaspar, 2003). Nevertheless, our study shows that tillage accompanied by heavy rain or irrigation can cause a sudden outburst of CO_2 during the crop growing season in the summer from areas previously under CRP management.

The CO_2 flux was similar under malt barley and pea in NTBON and NTPON, suggesting that cropping system did not influence CO_2 emission. Al-Kaisi and Yin (2005) reported no significant difference in CO_2 flux between corn (*Zea mays* L) and soybean (*Glycine max* L.). However, lower CO_2 flux in UAGON than in other treatments on some measurement dates was probably a result of decreased soil water content (Figures 4A, 4B, 4C). The greater CO_2 flux in NTBFN than in NTBON and in CTBFN than in CTBON in Days 188 and 202 suggests that N fertilization increased CO_2 flux when soil water content is adequate. Limited or no increase in CO_2 flux with increased N fertilization rate had been reported by several researchers (Rochette and Gregorich, 1998; Wagai et al., 1998; Amos et al., 2005).

Averaged across measurement dates, CTBFN and CTBON increased soil surface CO_2 flux by an average of 58% compared with other treatments (Table 2). These treatments also increased soil temperature and reduced soil water content, except in UAGON. Eliminating the short-term burst of CO_2 immediately after tillage (Al-Kaisi and Yin, 2005) which was not measured in this study, tillage can increase CO_2 flux by an average of 90 kg CO_2-C ha^{-1} d^{-1} (246 kg CO_2-C ha^{-1} d^{-1} in tilled vs. 156 kg CO_2-C ha^{-1} d^{-1} in non-tilled treatments) or 19.3 Mg CO_2-C ha^{-1} compared with no-tillage from May to November (214 d). If the loss was estimated for the entire year using the same rate of CO_2 emission, it would be 32.8 Mg CO_2-C ha^{-1} yr^{-1}. When CO_2 flux from tilled treatments during the peak emission periods in Days 130, 158, 179, 188, 195, and 202 were eliminated, the loss of CO_2 flux in tilled vs. non-tilled treatment averaged 4.7 Mg CO_2-C ha^{-1} yr^{-1}. The value could, however, be overestimated, because CO_2 flux rates are lower in winter than in summer (Follett, 1997; Bajracharya et al., 2000a). Furthermore, half of this CO_2 loss would be contributed by respiration from plant roots and soil microflora and fauna (Curtin et al., 2000). Soil C sequestration rate in northern Great Plains using no-till management compared with tilled treatment has been estimated at 5 to 6 Mg ha^{-1} yr^{-1} after 13 to 14 yr or about 400 kg C ha^{-1} yr^{-1} (Curtin et al., 2000). These data show that tillage can cause a drastic loss of soil C as CO_2 emission within a short period of time, but it takes a long time to sequester C using no-till practices.

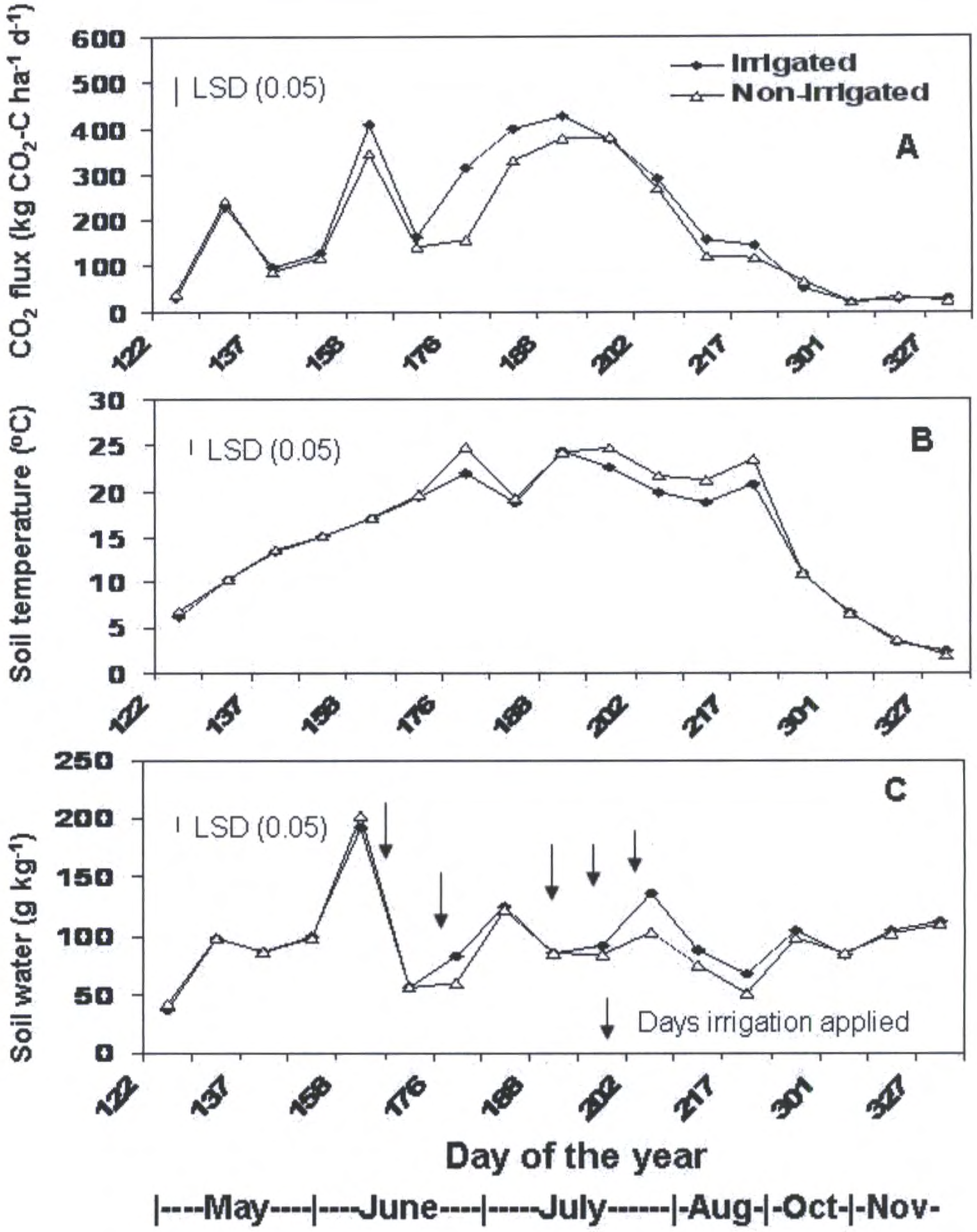


Figure 3. Effect of irrigation on (A) soil surface CO$_2$ flux and soil (B) temperature and (C) water content at the 0 to 15 cm depth averaged across management practices from May to November 2005 in western North Dakota. LSD (0.05) is the least significant difference between treatments at $P < 0.05$.

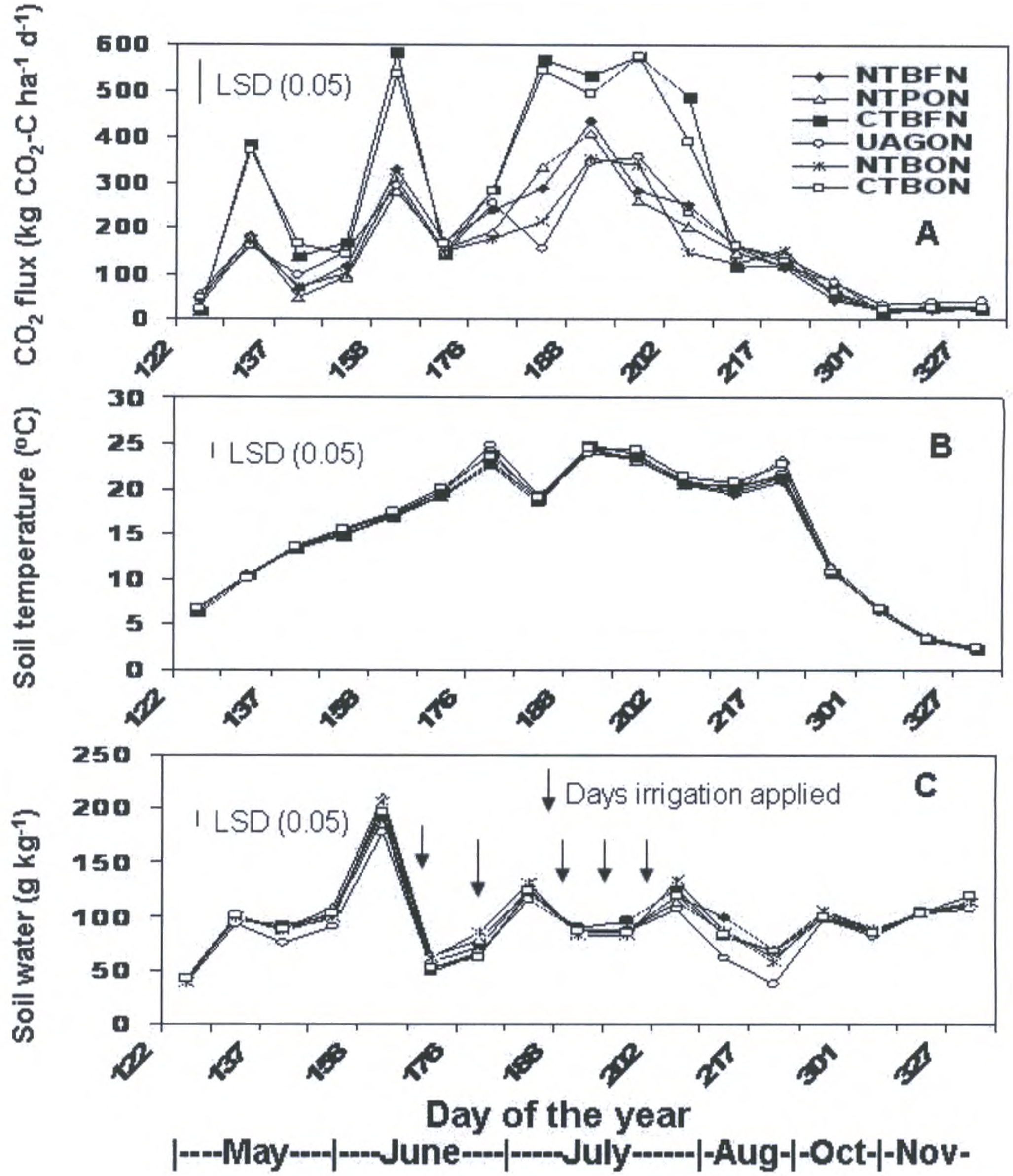


Figure 4. Effects of management practices on (A) soil surface CO_2 flux and soil (B) temperature and (C) water content at the 0 to 15 cm depth averaged across irrigation systems from May to November 2005 in western North Dakota. Management practices are CTBFN, conventional-till malt barley with 67 or 134 kg N ha^{-1}; CTBON, conventional-till malt barley with 0 kg N ha^{-1}; NTBFN, no-till malt barley with 67 or 134 kg N ha^{-1}; NTBON, no-till malt barley with 0 kg N ha^{-1}; NTPON, no-till pea with 0 kg N ha^{-1}; and UAGON, undisturbed alfalfa and grass with 0 kg N ha^{-1}. LSD (0.05) is the least significant difference between treatments at $P < 0.05$.

Relationship between Carbon Dioxide Emission and Temperature and Soil Water Content

A regression analysis between CO_2 flux and soil temperature at the 0 to 15 cm depth averaged across irrigation and management practices treatments and daily average air temperature at the time of CO_2 measurement revealed that CO_2 flux was linearly related ($R^2 = 0.55$ to 0.56, $P \leq 0.001$, n = 17) with both soil temperature and air temperature (Figures 5A and 5B). Both soil temperature and air temperature accounted for 55 to 56% of the variability in CO_2 flux with similar slopes. Soil water content at the 0 to 15 cm depth was weakly related ($R^2 = 0.18$, $P \leq 0.10$, n = 17) with CO_2 flux.

The significant relationship between CO_2 flux and soil temperature is well known (Follett, 1997; Bajracharya et al., 2000b; Parkin and Kaspar, 2003). High CO_2 flux usually occurs in the summer when soil temperature is higher and soil water content and substrate C availability is adequate, while low emission occurs in the winter when soil biological activity is minimal due to near-freezing soil temperature (Kirschbaum, 1995; Follett, 1997; Bajracharya et al., 2000a; 2000b). Parkin and Kaspar (2003) reported that soil surface CO_2 flux was better related with air temperature than with soil temperature because of plant residue at the soil surface. Our results showed that CO_2 flux was equally related with both soil temperature and daily average air temperature at the time of CO_2 flux measurement. Significant effects of irrigation and management practices on CO_2 flux and soil temperature and water content (Table 1) suggests that these treatments modified soil temperature and water content which influenced microbial activities and C mineralization, thereby resulting in different CO_2 fluxes between treatments at various measurement dates.

Conclusions

Results of this study showed that both irrigation and tillage influenced soil temperature and water content and increased soil surface CO_2 flux from areas previously managed under CRP planting for more than 20 years. Tillage followed by heavy rain or irrigation drastically increased the flux during the crop growing season from May to August 2005. The CO_2 flux was influenced by soil temperature and water content which were modified by treatments. Cropping system and N fertilization had limited or no effect on CO_2 flux in the first year after the establishment of treatments. The CO_2 flux in undisturbed alfalfa and grasses was similar to that in no-till malt barley and pea treatments. Long-range studies are needed to determine the effects of treatments on CO_2 flux under various climatic and environmental conditions to estimate C loss from the soil ecosystem.

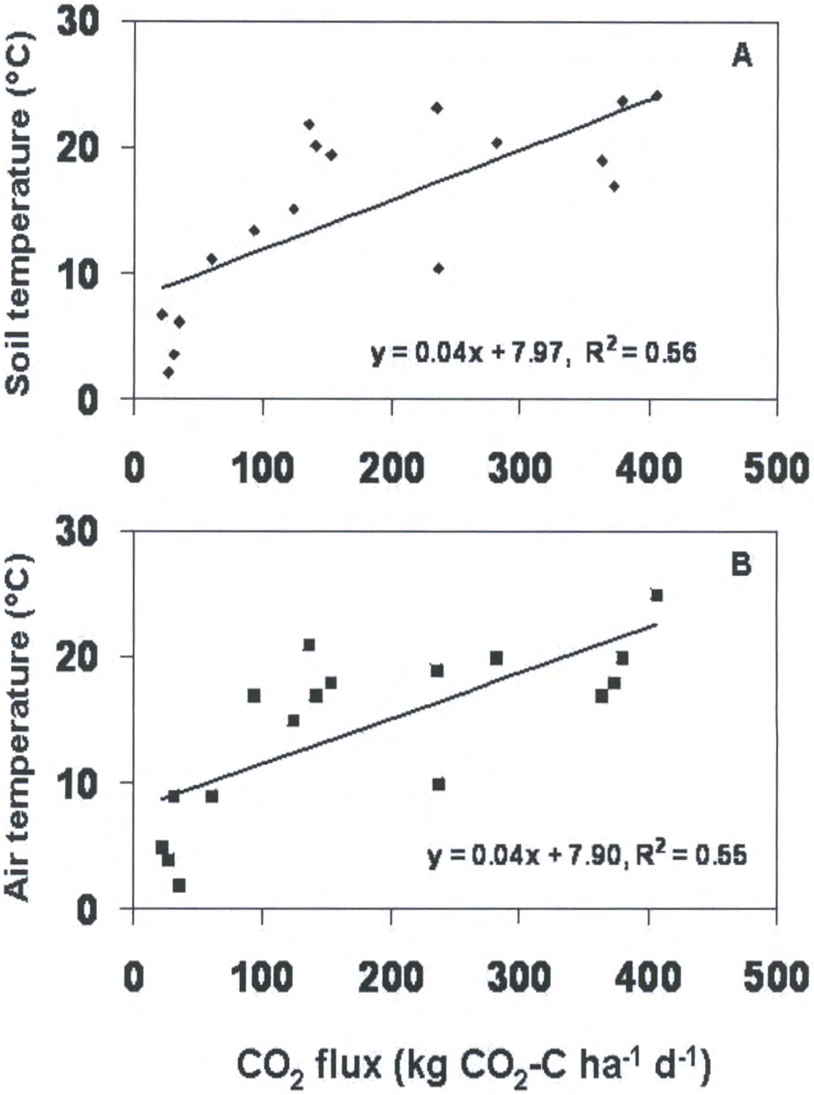


Figure 5. Relationship between soil surface CO_2 flux and (A) soil temperature at the 0 to 15 cm depth averaged across irrigation and management practices and (B) daily average air temperature at the time of CO_2 measurement.

Acknowledgement

We appreciate the help provided by John Rieger, Lane Hall, Bryan Gebhard, and Amy McGregor for weekly measurement of CO_2 flux in the field, Bill Iversen for irrigating experimental plots, and Robert Evans for providing support in conducting the research.

References

Al-Kaisi, M.M., and X. Yin. 2005. Tillage and crop residue effects on soil carbon and carbon dioxide emission in corn-soybean rotation. *J. Environ. Qual*. 34:437-445.

Amos, B., T.J. Arkebauer, and J. W. Doran. 2005. Soil surface fluxes of greenhouse gases in an irrigated maize-based agroecosystem. *Soil Sci. Soc. Am. J.* 69: 387-395.

Bajracharya, R.M., R. Lal, and J.M. Kimble. 2000a. Erosion effects on carbon dioxide concentration and carbon flux from an Ohio alfisol. *Soil Sci. Soc. Am. J.* 64:694-700.

Bajracharya, R.M., R. Lal, and J.M. Kimble 2000b. Diurnal and seasonal CO_2-C flux from soil as related to erosion phases in central Ohio. *Soil Sci. Soc. Am. J.* 64: 286-293.

Beare, M.H., M.L. Cabrera, P.F. Hendrix, and D.C. Coleman. 1994. Aggregate-protected and unprotected organic matter pools in conventional- and no-tillage soils. *Soil Sci. Soc. Am. J* 58:787-795.

Calderon, F.J., and L. Jackson. 2002. Rototillage, disking, and subsequent irrigation: Effects on soil nitrogen dynamics, microbial biomass, and carbon dioxide efflux. *J. Environ. Qual.* 31:752-758.

Curtin, D., H. Wang, F. Selles, B.G. McConkey, and C.A. Campbell. 2000. Tillage effects on carbon fluxes in continuous wheat and fallow-wheat rotations. *Soil Sci. Soc. Am. J.* 64: 2080-2086.

Duxbury, J.M. 1994. The significance of agricultural sources of greenhouse gases. *Fert. Res.* 38:151-163.

Duxbury, J.M. 1995. The significance of agricultural greenhouse gas emissions from soil of tropical agroecosystems. p. 279-291. *In* R. Lal (ed.) *Soil management and greenhouse effect*. Lewis Publ., Boca Raton, FL.

Ellert, B.H., and H.H. Janzen. 1999. Short-term influence of tillage on CO_2 fluxes from a semi-arid soil on the Canadian prairies. *Soil Tillage Res*. 50:21-32.

Follett, R.F. 1997. CRP and microbial biomass dynamics in temperate climates. p. 305-322. *In* R. Lal (ed.) *Management of carbon sequestration in soi*l. CRC Press, Boca Raton, FL.

Fortin, M.C., P. Rochette, and E. Pattey. 1996. Soil carbon dioxide fluxes from conventional and no-tillage small-grain cropping system. *Soil Sci. Soc. Am. J.* 60:1541-1547.

Franzluebbers, A.J., F.M. Hons, and D.A. Zuberer. 1995. Tillage-induced seasonal changes in soil physical properties affecting soil CO_2 evolution under intensive cropping. *Soil Tillage Res*. 34:41-60.

Grant, R.F. 1997. Changes in soil organic matter under different tillage and rotation: Mathematical modeling in *ecosys*. *Soil Sci. Soc. Am. J.* 61: 1159-1175.

Howard, D.M., and P.J.A. Howard. 1993. Relationships between CO_2 evolution, moisture content, and temperature for a range of soil types. *Soil Biol. Biochem*. 25:1537-1546.

Jackson, L.E., F.J. Calderon, K.L. Steenwerth, K.M. Scow, and D.E. Rolston. 2003. Responses of soil microbial processes and community structure to tillage events and implications for soil quality. *Geoderma* 114:305-317.

Jastrow, J.D., T.W. Boulton, and R.M. Miller. 1996. Carbon dynamics of aggregate-associated organic matter estimated by carbon-13 natural abundance. *Soil Sci. Soc. Am. J.* 60:801-807.

Kern, J.S., and M.G. Johnson. 1993. Conservation tillage impacts on national soil and atmospheric carbon levels. *Soil Sci. Soc. Am. J.* 57:200-210.

Kessavalou, A., A.R. Mosier, J.W. Doran, R.A. Drijber, D.J. Lyon, and O. Heinemeyer. 1998. Fluxes of carbon dioxide, nitrous oxide, and methane in grass sod and winter-fallow tillage management. *J. Environ. Qual*. 27:1094-1104.

Kirschbaum, M.U.F. 1995. The temperature dependence of soil organic matter decomposition and the effect of global warming on soil organic carbon storage *Soil Biol. Biochem.* 27:753-760.

Kuo, S., U.M. Sainju, and E.J. Jellum. 1997. Winter cover crop effects on soil organic carbon and carbohydrate. *Soil Sci. Soc. Am. J.* 61: 145-152.

Lal, R., J. Kimble, and B.A. Stewart. 1995. World soils as a source or sink for radiatively active gases. p. 1-8. *In* R. Lal et al. (ed.) *Soil management and greenhouse effect. Advances in Soil Science.* CRC Press, Boca Raton, FL.

Lal, R., and J.M. Kimble. 1997. Conservation tillage for carbon sequestration. *Nutr. Cycling Agroecosyst.* 49:243-253.

Littell, R.C., G.A. Milliken, W.W. Stroup, and R.D. Wolfinger. 1996. *SAS system for mixed models*. SAS Inst. Inc., Cary, NC.

Parkin, T.B., and T.C. Kaspar. 2003. Temperature controls on diurnal carbon dioxide flux: Implications for estimating soil carbon loss. *Soil Sci. Soc. Am. J.* 67: 1763-1772.

Paustian, K., G.P. Robertson, and E.T. Elliott. 1995. Management impacts on carbon storage and gas fluxes in mid-latitudes cropland. p. 69-83. *In* R. Lal et al. (ed) *Soils and global climate change. Advances in soil science.* CRC Press, Boca Raton, FL.

Reicosky, D.C., and M.J. Lindstrom. 1993. Fall tillage method: Effect on short-term carbon dioxide flux from soil. *Agron. J.* 85:1237-1243.

Reicosky, D.C., W.D. Kemper, G.W. Langdale, C.L. Douglas, Jr., and P.E. Rasmussen. 1995. Soil organic matter changes resulting from tillage and biomass production. *J. Soil Water Conserv*. 50:253-261.

Roberts, W.P., and K.Y. Chan. 1990. Tillage-induced increases in carbon dioxide loss from soil. *Soil Tillage Res*. 17:143-151.

Rochette, P. ,and D.A. Angers. 1999. Soil surface carbon dioxide fluxes induced by spring, summer, and fall moldboard plowing in a sandy loam. *Soil Sci. Soc. Am. J.* 63:621-628.

Rochette, P., and L.B. Flanagan. 1997. Quantifying rhizosphere respiration in a corn crop under field conditions. *Soil Sci. Soc. Am. J.*61:466-474.

Rochette, P. and E.G. Gregorich. 1998. Dynamics of soil microbial biomass C, soluble organic C, and CO_2 evolution after three years of manure application. *Can. J. Soil Sci.* 78:283-290.

Sainju U.M., B.P. Singh, and W.F. Whitehead. 2002. Long-term effects of tillage, cover crops, and nitrogen fertilization on organic carbon and nitrogen concentrations in sandy loam soils in Georgia, USA. *Soil Tillage Res*. 63:167-179.

Sainju, U.M., W.F. Whitehead, and B.P. Singh. 2005. Biculture legume-cereal cover crops for enhanced biomass yield and carbon and nitrogen. *Agron. J.* 97:1403-1412.

Sparling, G.P., and D.J. Ross. 1988. Microbial contributions to the increased nitrogen mineralization after air-drying of soils. *Plant Soil* 105:163-167.

Van Gestel, M., R. Merkx, and K. Vlassak. 1993. Microbial biomass responses to soil drying and wetting: The fast- and slow-growing microorganisms in soils from different climates. *Soil Biol. Biochem*. 25:109-123.

Wagai, R., K.R. Brye, S.T. Gower, J.M. Norman, and L.G. Bundy. 1998. Land use and environmental factors influencing soil surface CO_2 flux and microbial biomass in natural and managed ecosystems in southern Wisconsin. *Soil Biol. Biochem*. 30:1501-1509.

Influence of Biogenic Organic Compounds from Crops and Plantation Forests on Ozone and Secondary Organic Aerosols

Tanarit Sakulyanontvittaya[1], Jana Milford[1], Christine Wiedinmyer[2], Alex Guenther[2], Peter Harley[2], Detlev Helmig[3], Tiffany Duhl[3], John Ortega[3], and David Tanner[3].
[1]University of Colorado, Department of Mechanical Engineering, Boulder, CO 80309, USA
[2]National Center for Atmospheric Research, Boulder, CO 80307, USA; [3]Institute for Arctic and Alpine Research, University of Colorado, Boulder, CO 80309, USA

Abstract

Biogenic volatile organic compounds (BVOCs) such as isoprene, monoterpenes (MT) and sesquiterpenes (SQT) both react with and form ozone, and are also precursors of secondary organic aerosols. Crops and forests emit large numbers and quantities of BVOC. In this study, biogenic emissions of isoprene, MT, and SQT from crops, plantation forests, and natural vegetation are estimated with the Model of Emissions of Gases and Aerosols from Nature (MEGAN). While emissions estimates for isoprene and MT have been developed and refined over many years, SQT have received relatively little attention until recently. Emissions of SQT have been reported in the literature from several agricultural cultivars, including citrus trees, hybrid poplar, tobacco, corn, and loblolly pine. Preliminary SQT emission estimates for this study are based on data collected over the past several years through greenhouse and field measurements. Biogenic emission inventories developed using MEGAN are combined with inventories for non-biogenic emissions and input to the Community Multiscale Air Quality (CMAQ) model. CMAQ has been modified to include gas-phase chemistry and secondary aerosol formation from SQT and is run for July 2001 for a domain that covers the contiguous United States. The model results are evaluated by comparison with aerosol and ozone data from several air quality monitoring networks. Sensitivity analysis is performed to preliminarily compare the influence of biogenic emissions from agricultural activities on ozone and secondary organic aerosol with the influence of biogenic emissions from natural vegetation.

Monitoring the Precipitation Chemical Quality of Vaz Research Forest (North of Iran)

Ali Salahi[1], Shirin Geranfar[2], Shahram Banej-shafii[1], Ebrahim Azizkhani[1]
[1]Research Institute of Forest and Rangelands (RIFR), P.O.Box: 31585-343, Karaj, Iran, Email: ali.salahi@rifr-ac.ir , Fax:+98-261-6603482
[2]Amir –Kabir Industrial University, Tehran, Iran

Summary

Investigation on precipitation quality from lowland of Vaz Research Forest in Caspian Sea region to upland areas of 1000, 1600 and 2200 meters altitude had been carried out from 1999 to 2003. There are different amount of rainfall in varying elevations, and in natural cases showed that amount of Nitrate (NO_3^-), Sulfate (SO_4^{--}), Chloride (Cl^-), Ammonium (NH_4^+), Calcium(Ca^{++}) and Magnesium(Mg^{++}) have regular relation with altitude, and annual deposition of elements is decreased parallel with increased altitude. Nitrate wet deposition in 300, 1000, 1600 and 2200 meters were 250, 275, 193 and 104 kg/ha.yr respectively. Sulfate ion in this way including 21, 45, 28 and 2.5 kg/ha.yr wet deposition parallel with increasing mentioned altitudes. Chloride ion which originates mainly from spraying of sea-water, includes 1080, 620, 784 and 337 kg/ha.yr wet deposition regard to 300, 1000, 1600 and 2200 meters elevation respectively. Ammonium ion (NH_4^+) also including 19, 4, 8 and 4 kg/ha.yr related to 300, 1000, 1600 and 2200 meters elevation respectively. Annual wet deposition of Calcium (Ca^{++}) are 38, 22, 24 and 12 kg/ha.yr wet deposition regard to 300, 1000, 1600 and 2200 meters elevation respectively. Increased amount of NO_3^- and SO_4^{--}in 1000 meter altitude is reversed and different with decreased amounts parallel with increasing altitude, which originating from fossil fuel consumption by native community and resident people and anthropogenic effects. Acidity (pH) of precipitation were in natural and alkaline criteria, and total average of pH for precipitation regards to each elevation were more than 7 in pH value. Measurement of Nitrogen showed that there are annual wet deposition in amount of 56, 62, 44 and 23 kg/ha.yr wet deposition regard to 300, 1000, 1600 and 2200 meters elevation respectively.

Nitrogen wet deposition from 300 to 1600 meters elevation indicates receipt of N more than acceptance range of Europe, and wet deposition of sulfur is lower than acceptance and permitted level. Nitrogen Oxides (NOx) wet deposition is parallel with increase emission, while mentioned tendency for sulfur compounds are not strong (5).

A defined altitude consists of 300, 1000, 1600 and 2200 meters from sea level and in each elevation 3 rain sampler were installed to collect precipitation. These rain samplers were insulated regard to dust and air-born sedimentation and dry deposition. Rain samples were gathered inside clean and white polyethylene container during precipitating, and immediately were put in freezer to prevent any changes in rain and snow chemical quality also decomposition of nitrate ion. Chemical factors of precipitation samples were analyzed in analytical chemistry laboratory by following methods:

1) Sulfate ion (SO_4^{--}) with method of turbometeric and reading by spectrophotometer in length-wave of 340 nm(nano-meter)
2) Calcium and magnesium with method of complexometery, Moroxide indicator and Erio-Chrome Black Tea.
3) Chloride ion with method of Mohr
4) Using a flame photometer for Potassium and Sodium.
5) Ammonium with method of distillation and titration of existent Ammonium with Chloridric acid 0.01N (Normality).
6) Nitrate ion with method of spectrophotometer in length-wave of 220 and 275 nm (nano-meter) ultra-violet visibility.
7) Acidity with pH-meter.

Introduction

Plant species communities are influenced by some factors such as; precipitation, humidity and soil. Investigations on Vaz Research Forest were carried out in 1996 concern to meteorology, climatology, geology, geomorphology, typology and structures of forests and mountain rangelands (2).

Study on quality of precipitation which comes-down on Vaz Forest cover, lead to improving mentioned studies and more recognition of relation between atmospheric chemistry and precipitation chemical quality affecting on vegetation cover in different altitudes of Vaz Forest.

Previous study by Breedman (1986) showed that atmospheric deposition which can be tolerated by terrestrial ecosystems are between 32-45 kg/ha.yr concern to sulfur and 5.6 kg/ha.yr concern to nitrogen (1).

Results

Measurement of acidity in rain and snow samples in different elevation of Vaz Research Forest showed that the acidity based on pH value is in range of natural acidity or slightly alkaline. Measurement of pH in winter's snow samples and spring's snow samples indicated that winter's snow samples are 0.5 unit of pH value more acidic than spring's snow samples which resulted from more consumption of fossil fuels in winter and cold season, hence releasing and emission of acid forming gases into atmosphere to form attributed acids.

Analysis of Variance for acidity (pH) related to precipitation in Vaz Research Forest showed that there was significant difference ($p = 0.01$) for different treatments, and winter average pH value of snow samples showed that, they differ from other seasons.

Analysis of Variance for Nitrate ion (NO_3^-) related to precipitation in Vaz Research Forest showed that there was significant difference ($p = 0.01$) for different treatments, and seasonal precipitation sampling.

Analysis of Variance for Sulfate ion (SO_4^{--}) related to precipitation in Vaz Research Forest showed that there was significant difference ($p = 0.01$) for different treatments, different altitudes also between different sampling years.

Analysis of Variance for Chloride ion (Cl^-) related to precipitation in Vaz Research Forest showed that there was significant difference ($p = 0.01$) for different treatments, different altitudes also between different seasonal sampling.

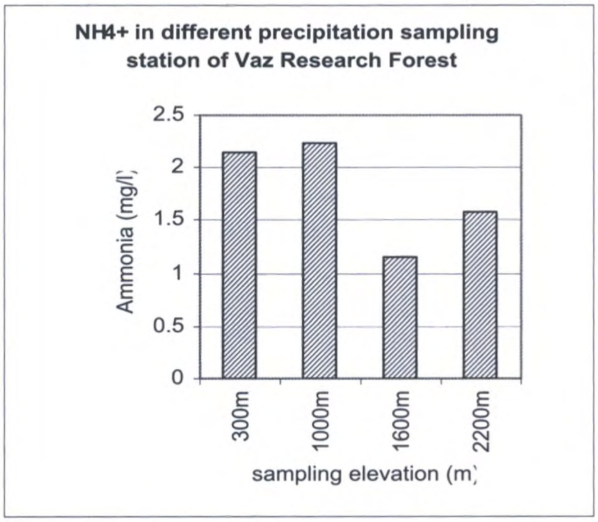
Analysis of Variance for Ammonium ion (NH_4^+) related to precipitation in Vaz Research Forest showed that there was significant difference ($p = 0.01$) for different treatments, different altitudes also between different seasons of sampling.

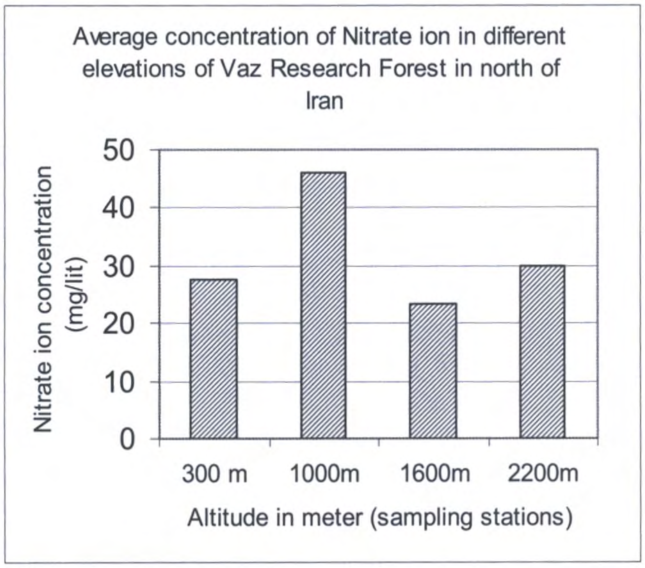
Analysis of Variance for Magnesium ion (Mg^{++}) related to precipitation in Vaz Research Forest showed that there was significant difference ($p = 0.01$) for different treatments, and between different seasons of sampling.

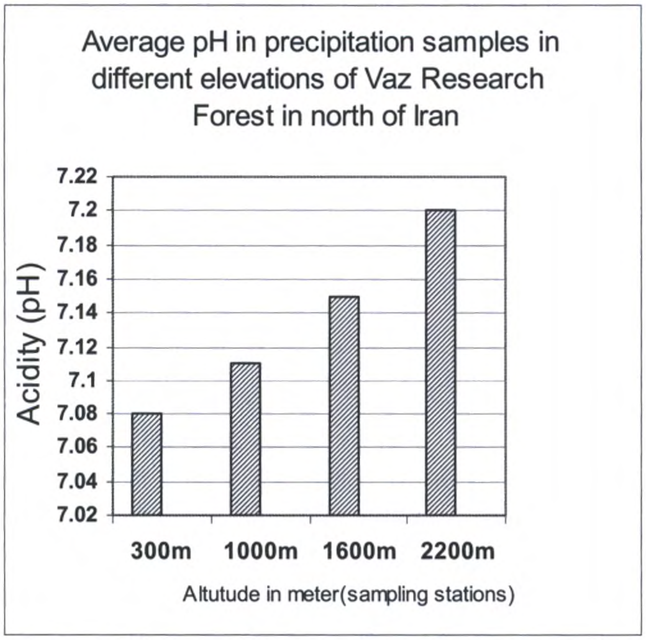
Analysis of Variance for Calcium ion (Ca^{++}) related to precipitation in Vaz Research Forest showed that there was significant difference ($p = 0.01$) for different treatments, different altitudes also between different seasons of sampling.

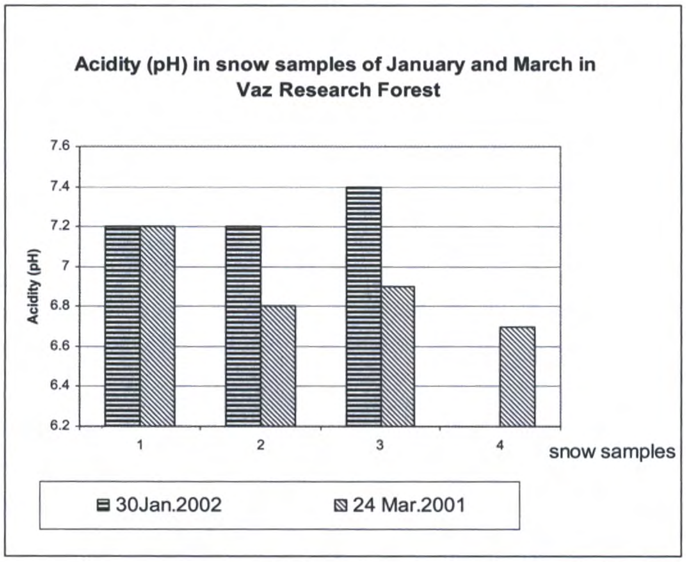
Analysis of Variance for Potassium ion (K^+) related to precipitation in Vaz Research Forest showed that there was significant difference ($p = 0.01$) for different altitudes.

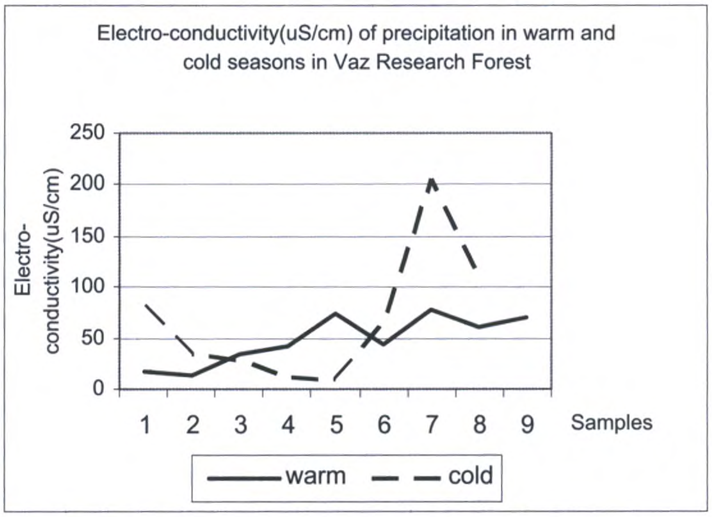
Analysis of Variance for Sodium ion (Na^+) related to precipitation in Vaz Research Forest showed that there was significant difference ($p = 0.01$) for sampling years, and different seasons of sampling.


NH4+ in different precipitation sampling station of Vaz Research Forest
Ammonia (mg/l)
2.5
2
1.5
1
0.5
0
300m
1000m
1600m
2200m
sampling elevation (m)


Average concentration of Nitrate ion in different elevations of Vaz Research Forest in north of Iran
Nitrate ion concentration (mg/lit)
50
40
30
20
10
0
300 m
1000m
1600m
2200m
Altitude in meter (sampling stations)


Average pH in precipitation samples in different elevations of Vaz Research Forest in north of Iran
Acidity (pH)
7.22
7.2
7.18
7.16
7.14
7.12
7.1
7.08
7.06
7.04
7.02
300m
1000m
1600m
2200m
Altutude in meter(sampling stations)


Acidity (pH) in snow samples of January and March in Vaz Research Forest
Acidity (pH)
7.6
7.4
7.2
7
6.8
6.6
6.4
6.2
1
2
3
4
snow samples
30Jan.2002
24 Mar.2001


Electro-conductivity(uS/cm) of precipitation in warm and cold seasons in Vaz Research Forest

Analysis of Variance for Electro-conductivity (EC) related to precipitation in Vaz Research Forest showed that there was significant difference ($p = 0.01$) for different treatments, different altitudes also between different seasons of sampling. Amount of EC in summer is at highest level and EC in winter's precipitation samples is in the lowest level.

Measurement of NO_3^- and SO_4^{--} in snow samples regards to different elevations of 1600, 2000 and 2200 meters showed that based on molarity, Nitrate molarity is 61 times of Sulfate molarity in snow samples. Weight-based comparison showed that weight equivalent of Nitrate ion is 35 times more than SO_4^{--}.

Discussion

Chemical analysis of precipitation in Vaz Research Forest showed that there is acidity in range of neutral, while pH in south-west of Norway and Sweden is between 4.1-4.2 which gradually increases to 4.5 in inner part of country (4). Acidity value in some urban area of Poland is between 5.1-5.3 and in rural area pH is between 4.3-4.95.

Primarily acidity is attributed to presence of Nitrate and Sulfates based on ion equivalent, and inside snow Nitrate existence was 60% more than Sulfate (8).

Measurement of NO_3^- and SO_4^{--} in snow samples regards to different elevations of 1600, 2000 and 2200 meters showed that based on molarity, Nitrate molarity is 61 times of Sulfate molarity in snow samples. Weight-based comparison showed that weight equivalent of Nitrate ion is 35 times more than SO_4^{--}.

Mineral Sulfur compounds

More than 60 compounds of mineral Sulfur which is released into atmosphere, can be detected. Some Sulfur content gases are produced by industrial processes and main sources of H_2S are found in urban areas and marshlands. The main source of SO_2 is anthropogenic and has a few natural sources, hence its concentration in urban area is higher than other places (3).

SO_4^{--} is one of the main components of atmospheric particles. The most amount of SO_4^{--} is found in urban atmosphere, but in some forests SO_4^{--} is higher than urban area due to transferring from urban area to forest atmosphere.

Gaseous phase of atmospheric H_2S is started as reaction with hydroxyl radical (OH.) and continues to form SO_2.

$$H_2S + OH. \rightarrow HS. + H_2O$$

$$HS. + 2O_2 \rightarrow HO_2. + SO_2$$

SO_2 in atmosphere reacts with hydroxyl radical and produces H_2SO_4.

$$SO_2 + OH^{\cdot} \rightarrow HSO3.$$

$$HSO_3^{\cdot} + O_2 \rightarrow SO_3 + HO_{2.}$$

$$SO_3 + H_2\,O \rightarrow + H_2\,SO_4$$

Produced H_2SO_4 forms the condensation nucleus for atmospheric water vapor and creates visible Sulfate aerosols existent throughout the troposphere (7).

Mineral Nitrogen compounds

N_2O and NH_3 emission source is high temperature combustion by Power-plants, vehicles, forest fire and solid waste burning. Gaseous Ammonia react slowly with hydroxyl radical and produces Nitrogen radical.

$$NH_3 + OH. \rightarrow NH_{2.} + H_2O$$

Probably this radical reacts with Ozon (O_3) and finally in a chain reaction lead to form Nitrogen Oxides. Solubility of NH3 in water is high hence most NH3 is decreased which is more than atmospheric gaseous phase disappearance. NO_2 can be oxidized both to Nitrate radical and to Nitric acid.

$$NO_2 + O_3 \rightarrow NO_3^{\cdot} + O_2$$

$$NO_2 + OH^{\cdot} \rightarrow HNO_3$$

O_3 and Nitrate radical are two active oxidant compounds inside the atmosphere which are responsible for decreasing half-life of some rare atmospheric gases neighborhood of urban area (9).

H_2S forms the main biological source of atmospheric Sulfur. With increasing amounts of SO_2 and NO_x, in case of more oxidation it falls down as acid with rain and snow, and in case of shortage of this gases also more existence of heavy metals like Cu, Zn, Cd, Pb, Hg, Ni and alkaline ions, led to neutralizing the atmospheric acidity and cause pH reach to higher pH (10).

Nitrate and Ammonium

Ammonia is released into atmosphere as gas form. More than 80% of gas emission originates from urea hydrolysis belong to animal urine. Ammonia after emission to atmosphere reminds and lasts as gas, also Ammonium ion in special form of Ammonium Sulfate (6).

Average amount of Nitrate based on mg/lit is increased parallel with increasing elevation from sea level. Average NO_3^- concentration at 300 meters altitude with amount of 27 mg/lit reach to 46 mg/lit at 1000 meters altitude due to anthropogenic emission.

Natural increase amount of NO_3^- from 300 meters elevation to 1000 and 2200 meters elevation is 20% increased in concentration. Annual wet deposition of NO_3^- in altitudes of 300, 1000, 1600 and 2200 meters from sea level is 250, 275, 93 and 104 kg/ha.yr respectively. The main reason for decreasing annual wet deposition of NO_3^- from 300m to 1000m and 2200m is referred to varying rainfall in different elevations. Rainfall in 300m is 900 mm, in 1000m and 1600m is 600mm and in 2200m is 300mm precipitation per year. Consequently with decreasing rainfall, wet deposition of NO_3^- in 1600 meters reach to one third. Wet deposition of NO_3^- in 1000 meters altitude increases rapidly because concentration of NO_3^- in 1000 meters increases, although precipitation per year which in 300 m is 900 mm, decreases to 600 mm precipitation in 1000 meters altitude, which has anthropogenic origin.

Acknowledgement

The authors would like to thank Mr. Karamali Zabihi from Natural resources Research Center of Mazandaran Province, also Dr. Hassan Askary head of "Research Division of Natural Resources Protection" and our colleague Mr. Moniri which helped us during our research.

References

Breedman, N. Van and J. Mulder, 1984. Atmospheric acid deposition: effects on the chemistry of forest soils, Acidification and its policy implications. Proceeding of an int. conference held in Amsterdam, 5-9 May 1986, Elsevier Scientific Publishers, 1986, Pp. 141-152.

Khaleghi, P. 1998. A Profile Caspian Forests, Vazrood Experimental Forest. ISBN: 964-473-038-0, 380 P. Research Institute of Forest and Rangelands (RIFR).

Kauppi, P. and J. Kamari. 1986. Acidification of forest soil -Ecological Modeling. Pp. 231-235, International Inst. Applied System Analysis, Laxenburg, Austria.

Min, E. and S. Lee, 1990, Effects of artificial acid precipitation on forest soil buffer capacities. Journal of Korean Forestry Society,1990. Pp. 376-387.

Salahi, A. and Geranfar, Sh., 2004. Air quality Management in Greater Tehran Metropolitan Area, 235-260. Elsom, D.M., Regional and Local Aspects of Air Quality Management, WIT PRESS, England, ISBN: 1- 85312- 952-6, 322 P.

Salahi, A., and Sh. Geranfar, 2001. Nitrogen deposition in the Greater Tehran Metropolitan Area. Proceedings of 2nd International Nitrogen Conference on Science and Policy. TheScientificWorld 1(S2) . Pp.261-265 , 2001.

Salahi, A. 1997. Environmental Aspects of Acid Precipitation Quality. ACID SNOW AND RAIN, proceeding of International Congress of Acid Snow and Rain (ICASR) 1997, Pp: 293-298.

SNV'S report, 1993.ACIDIFICATION still a major cause for CONCERN. enviro/NO. 16, DECEMBER 1993.

Susana B. 1993. Acid rain and toxics harming Nordic waters, enviro/ NO 15, July 1993. Publ. The Swedish Environmental Protection Agency (SNV).

William H. S. 1990. Air Pollution and Forests. Second Edition, ISBN 0-387-97084-3.

Particulate Matter ($PM_{2.5}$) Emissions from Burning of Wheat Stubble and Kentucky Blue Grass in Eastern Washington and Idaho

M. Schaaf[1], M. Schreuder[1], W. Johnston[2], C. Golob[2], R. Babbitt[3] and R. Susott[3]
[1]Air Sciences Inc. 421 SW 6th Avenue, Suite 1400, Portland, Oregon
[2]Department of Crop Sciences, Washington State University, Pullman, Washington
[3]Rocky Mountain Research Station- Fire Sciences Laboratory, 5775 US Highway 10, Missoula, Montanta

Abstract

This study compares particulate matter emissions from controlled experimental burns of wheat stubble fields and Kentucky Blue Grass fields in Eastern Washington and Idaho, United States. The experimental field burns tested for the influence of pre-burn residue loading, fire ignition pattern, and residue moisture on particulate matter emission factors. The results indicate that residue consumption was positively correlated with pre-burn residue loading. $PM_{2.5}$ emission factors for wheat ranged from 5-10 lbs $PM_{2.5}$ per ton residue consumed and appears to be predominantly influenced by pre-burn residue moisture content. Higher moisture content coincided with higher emission factors because of more incomplete burning of residue. In the Kentucky Blue Grass burns, emission factors, ranging from 25-150 lbs $PM_{2.5}$ per ton residue, were considerably higher than in the wheat burns. Moreover, the low residue treatments had higher emissions factors than the high residue treatment. This resulted in similar total $PM_{2.5}$ emissions on a per acre basis in high and low loading plots, as increased fuel consumption in high loading plots was offset by decreased emission factors, and vice versa. Differences in $PM_{2.5}$ emission factors are discussed in terms of their explanatory variables, specifically, residue loading, residue moisture, and the "architecture" of the fuel bed.

Introduction

Fire has long been used a management tool in both wheat and grass seed production in the inland Pacific Northwest. In the production of cereal grains fire is an effective tool to decrease the residue loading of the fields before the direct seeding of the next crop. The removal of high residue loadings can eliminate the need for specialized equipment needed to seed into heavy residues, and thereby reduce. Additional advantages of the use of fire to reduce residue loading include controlling disease(s) and enhancing mineral cycles in the soil. However, the fire emissions associated with the use of residue burning has also been recognized as a concern for public health in the Pacific Northwest. For example, since the late 1990s the Washington State Department of Ecology (WDOE) has been in negotiation with growers organizations to reduce the emissions from both wheat stubble burning as well as from blue grass burning. As part of this process the need for additional research to quantify smoke emissions from agricultural burning practices was recognized. The studies reported in this presentation address this need and evaluated field measurements of residue consumption and smoke emissions under several alternative residue and fire management practices.

Methods

In this study controlled experimental burns of wheat stubble fields and Kentucky Blue Grass fields were performed in Eastern Washington and Idaho, United States, respectively. Wheat field burns were performed both in the Spring and Fall season of 2000 using two fuel loading groups (high and low), and four different patterns of fire ignition. Fire ignition patterns consisted of mass ignition, the conventional method of ignition, as well as backing fire (wind direction opposite to direction of fire front), head fire (wind direction same as direction of fire front),and strip fire (similat to head fire, but ignited in strips). The Kentucky Blue grass experiments, performed in Summer 2001 tested for $PM_{2.5}$ emissions in high and low-loading (bailed) fields, on both irrigated and dry land farms. In all treatments pre-burn measurements included residue loading, and residue and soil moisture content. Post-burn measurements included residue loading. During the burns meteorological variables were recorded, and video recordings were taken to evaluate the development of the fire during the analytical phase. Smoke emissions were continuously samples at two FASS units (Fire Atmosphere Sampling Systems; Susott et al., 1991), and analyzed by the USFS Fire Sciences Laboratory, Missoula, Montana. Based on smoke emissions and fuel consumption, pollutant emission factors were calculated for CO_2, CO, CH_4, and particulate matter less than 2.5 micrometer in diameter, $PM_{2.5}$.

Figure 1. Example of a backing fire in a wheat field. Towers indicate the FASS units.

Results and Discussion

The results of the wheat experiments indicate that residue consumption was positively correlated with pre-burn residue loading. $PM_{2.5}$ emission factors, ranging from 5-10 lbs $PM_{2.5}$ per ton residue consumed, did not vary significantly by loading or ignition patterns, but were significantly lower in the spring than in the fall. The seasonal difference in $PM_{2.5}$ emission factors could be attributed mainly to higher fuel moisture in the fall compared to the spring.

Similar to the wheat stubble study, residue consumption in Kentucky Blue Grass increased with increasing pre-burn fuel loading. At two of the three field locations $PM_{2.5}$ emission factors, ranging from 25-150 lbs $PM_{2.5}$ per ton residue, were considerably higher in the low loading than in the high loading plots. This resulted in similar total $PM_{2.5}$ emissions on a per acre basis in high and low loading plots, as increased fuel consumption in high loading plots was offset by decreased emission factors, and vice versa. At the third location the high loading plots had both high $PM_{2.5}$ emission factors and high fuel consumption, leading to the highest total $PM_{2.5}$ emissions per acre. Differences in $PM_{2.5}$ emission factors in the blue grass study appeared most correlated with fuel bed characteristics, for example, stubble height, layering of the fuel bed, fuel bed depth and bulk density, and drying time between harvesting and residue burning. These factors had a pronounced effect on combustion efficiency, and therefore on the production of incomplete combustion products, including carbon monoxide, methane, and $PM_{2.5}$.

References

Susott, R., D. Ward, R. Babbitt, and D. Latham. 1991. Fire dynamics and chemistry of large fires. In: *Global Biomass Burning: Atmospheric Climate and Biosphere Implications.* Edited by J.S. Leveine, MIT Press, Cambrige, Massachusetts.

Key Odour Compounds Identified from Animal Production Units Using Gas Chromatography-Olfactometry-Mass Spectrometry

A. Schäfer[1], A. Feilberg[2], A.P. Adamsen[3], H. Scherrebech[1] and A.M. Nielsen[1]
[1]Danish Meat Research Institute, Maglegaardsvej 2, DK-4000 Roskilde, Denmark
[2]Danish Technological Institute, Kongsvang Allé 29, DK-8000-Aarhus, Denmark
[3]LugtTek A/S, Niels Pedersens Allé 2, DK-8830 Tjele, Denmark

Abstract

Controlling odour nuisance from pig production units implies knowledge of the key odour compounds in the air. Air samples collected before and after a biological filter were analyzed using Detection Frequency Analysis based on Gas Chromatography in combination with two Olfactometry devices and Mass Spectrometry (GC-O-MS). Some of the odours were only detected by the assessors, indicating a higher detection limit of the mass spectrometric detector compared to the assessors. The quantitative analysis showed a significant reduction of several of the measured compounds.

Introduction

Reducing odour nuisance from animal production units is a top priority for the agricultural sector in Denmark. To solve this problem, it is necessary to gain knowledge of the compounds that are essential for the odour experienced by the neighbours of the production units. Knowledge of the key odour compounds can be used in an applied effort to control the odour nuisance via management, feeding or, for example, chemical/biological precipitation of odour compounds from the ventilation air.

The aim of this study is to reveal the key odour compounds from production facilities (finishers) using Gas Chromatography in combination with two Olfactometry ports and Mass Spectrometry (GC-O-MS). Ranking of odour active compounds is performed by the Detection Frequency Analysis (Pollien et al., 1997).

Methods

Volatile organic compounds in air from housing units were collected using sample tubes containing a combination of Tenax TA and Carbograph. The analyses were performed on a Hewlett Packard GC/MS 6890N/5973. Thermal desorption was performed on a Perkin Elmer ATD400. The capillary column was an Innowax (30 m, 0.25 mm ID, 0.25 μm film thickness) from J&W. The GC oven program was 40°C for 5 min, 10°C/min to 110°C, 20°C/min to 260°C for 10 min. Compounds were tentatively identified using NIST/EPA/NIH Mass Spectral Database or by comparison to authentic standards. The GC was equipped with two ODP2 sniffing ports from GERSTEL.

The detection frequency (DF) method was performed, using 6 assessors for each sample. The assessors were trained in the sniffing technique on artificial standards and real samples. The assessors had to place the detected odours from the samples into one of nine odour categories. The categories comprised "chemical/synthetic" (CS), "earthy/mushroom" (EM), "fresh/green/nature" (FGN), "nut/roasted/fried" (NRF), "baked" (B), "sourish/sweet/fruit" (SSF), "sulphur/onion/cabbage" (SOC), "nauseating/rancid/sour" (NRS) and "animal/faecal" (AF).

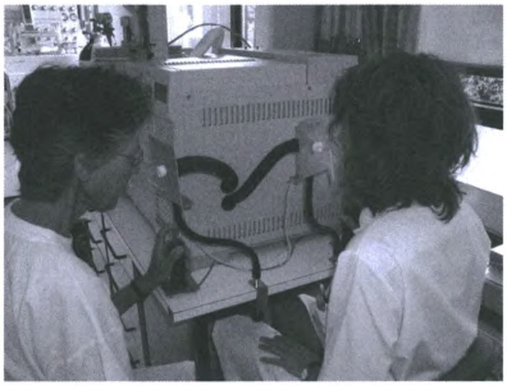


Figure 1. Two assessors sniffing samples from a housing unit.

Results and Discussion

Results from measurements in a housing unit with finisher pigs, where the ventilation air was passed through a biological filter, are shown. Sampling was performed simultaneously before and after the filter. The quantitative amount of detected compounds are shown in table 1. The concentration of the measured compounds for air before and after the biological filter shows a reduction (***) for 2,3-butanedione, 1-butanol, 3-methyl-1-butanol, acetic acid, benzaldehyde, propanoic acid, butanoic acid, 3-methylbutanoic acid and phenol.

Table 1. The amount is an average of three samplings (ng/l air).

Compound	BF 050301	BF 050308	AF 050301	AF 050308	LOQ
Dimethylsulfide	9	3	10	4	2
2,3-Butandione***	60	40	<	<	20
alpha-Pinene	4	<	4	<	2
Dimethyldisulfide	2	<	4	<	2
Hexanal**	40	30	20	20	6
3-Carene	2	<	2	<	2
1-Butanol***	4	3	0	<	2
Heptanal**	4	3	0	<	3
Acetic acid***	500	400	80	50	30
Benzaldehyde***	9	6	3	2	2
Propanoic acid***	300	200	30	20	20
Butanoic acid***	100	100	10	<	20
Acetophenone	0.2	0	0.6	0.2	0.2
3-Methyl-butanoic acid***	20	30	<	0	20
Pentanoic acid*	<	<	<	<	20
Phenol***	20	9	<	<	2
4-Methylphenol**	100	50	8	5	2
4-Ethylphenol**	5	3	<	<	2
Indole*	3	1	0.3	<	0.2
Skatole	3	<	0.4	<	0.2

< detected, but lower than the limit of quantification (LOQ). 0 not detected.
BF – before biological filter. AF – after biological filter.
* 5% significance level. ** 1% significance level. *** 0.1% significance level.

By means of the detection frequency analysis, the assessor detected 28 odours before the filter and 22 after the filter. Some of the detected compounds are shown in table 2 with the corresponding odour descriptor from flavornet, the category of odour chosen by the majority of the assessors and the number of assessors who detected an odour before and after the biological filter on two different days.

The MS analysis revealed the identity of most of the compounds. Among the compounds with the highest detection frequency, trimethylamine was only detected by the assessors before the filter and not after, while dimethylsulfide was unaffected by the filtering. 3-hydroxy-2-butanone was quantitatively reduced after the filter, but at the same retention time the detection frequency was significantly higher after the filter than before. This was not consistent with the reduced amount of 3-hydroxy-2-butanone. The chosen odour category did not correspond with the odour descriptors for 3-hydroxy-2-butanone from the literature either. The identity of the sniffed hidden compound still remains unknown. Two other compounds, 6-methyl-5-hepten-2-one and 1-hydroxy-2-propanone (tentatively identified), present in very low amounts, had a high detection frequency - these two compounds were detected only because olfactometry was used in combination with mass spectrometry.

Table 2. Result of the detection frequency analysis. BF – before filter. AF – after filter.

				Number of assessors			
LRI	Identification	Odour descriptor**	The favourite category of odour chosen by the assessors	BF 05.03.01	AF 05.03.01	BF 05.03.08	AF 05.03.08
655 716	Trimethylamine Dimethylsulfide	cabbage, sulphur fish	NRS, SOC	8	6	10	5
888	2-Butanone*	plastic, glue	SOC	4		2	3
968	2,3-Butanedione	butter, rancid	NRS	6	6	6	6
1077	Hexanal	grass, tallow, fat	FGN	4		4	5
1106	2-Methyl-1-propanol*	wine, solvent, bitter	CS, NRF	6		3	2
1230	3-Methyl-1-butanol	whisky, malt, burnt	NRF, EM	5	3	3	2
1266 1274	Octanal 3-Hydroxy-2-butanone	fat, soap, lemon butter, cream	B, SOC, EM	3	7	3	6
1290	1-Hydroxy-2-propanone*	-	SSF, EM	8		5	
1327	6-Methyl-5-hepten-2-one*	pepper, mushroom, rubber	NRF	6	4	4	5
1335	Sulphur dioxide*	pungent	SOC, CS	6	5	7	3
1382	Nonanal	fat, citrus, green	EM	2	2	2	3
1445	1-Octen-3-ol*	mushroom	EM, CS	5	4	2	3
1453	Acetic acid	sour	CS, EM, SSF	10	3	4	7
1513	Benzaldehyde	almond	CS	4	5	4	2
1549	Propanoic acid	pungent, rancid, soy	CS, NRF	2	3	5	3
1585	2-Methyl-propanoic acid	rancid, butter-like	FGN	6		4	4
1647	Butanoic acid	rancid, butter-like	NRS	10	2	9	7
1643	Acetophenone	must, flower, almond	CS, AF	2	3	2	
1724	3-Methylbutanoic acid	-	NRS, AF	4	3		4
2073	4-Methyl-phenol	medicine, smoke	AF	9	11	8	9
2167	4-Ethyl-phenol	must, phenol, spice	AF	6	7	5	
2460	Indole	faecal	AF	6	4		

* Tentative identification. ** Flavornet www.flavornet.com.

Conclusions

Quantitative measurements of selected compounds in air from housing units show reduction passing a biological filter. This information alone might not be adequate for reducing odour nuisance experienced by neighbours. Because of the low odour threshold of some of the compounds, the sensitive nose is a valuable tool for detecting these compounds. The combination of GC-O-MS enhances the amount of information obtained from the analysis of the different production facilities and helps combat odour nuisance from the animal production units. Further studies have to reveal the identity of odour compounds with high detection frequency.

References

Pollien, P., A. Ott, F. Montigon, M. Baumgartner, R. Muñoz-Box & A. Chaintreau. 1997. Hyphenated Headspace-Gas Chromatography-Sniffing Technique: Screening of Impact Odorants and Quantitative Aromagram Comparisons. J. Agric. Food Chem. 1997(45):2635-2637.

Assessment of Odour Annoyance by the Use of Dispersion Models and Odour Impact Criteria: A Sensitivity Study

G. Schauberger[1], M. Piringer[2] and E. Petz[2]

[1]Institute of Medical Physics and Biostatistics, University of Veterinary Medicine Vienna, Austria
[2]Central Institute for Meteorology and Geodynamics, Department of Environmental Meteorology, Vienna, Austria

Abstract

The Austrian odour dispersion model (AODM) is a Gaussian model suitable for the prediction of ambient odour concentrations. Based on cumulative frequency distributions of calculated odour concentrations at receptor points, separation distances are obtained defined by odour impact criteria chosen as a combination of odour threshold and probability of threshold exceedance.

Here, the AODM is used to calculate separation distances for various odour impact criteria which are defined by an odour threshold and the exceedance probability for a site in the Austrian North-alpine foreland. For these direction-dependent separation distances we analysed the sensitivity of the separation distance for the two parameters of the odour impact criteria.

We selected typical odour impact criteria used in various countries around the world. The definitions of these national odour impact criteria differ to quite an extent.

By using the separation distance as a final proof value, we analysed the variation of the separation distance due to the change of the odour impact criteria. The results show a sensitivity depending on the direction. The sensitivity is less pronounces for the prevailing wind directions. The separation distances show less variation with the wind direction for a higher protection level. The lower the protection level the higher the influence of the prevailing wind direction.

Materials and Methods

Only a brief summary is given here; details are found e. g. in Schauberger et al. (2002).

Short Description of the AODM

The calculation of odour release is based on a steady-state balance of the sensible heat flux, used to calculate the indoor temperature, and the ventilation rate of the livestock unit (Schauberger et al., 2000b). The corresponding odour flow in OU/m³ is assessed by a simple model for odour release described by Schauberger et al. (1999 & 2000b). The chosen system parameters typical for a livestock building in middle Europe (Schauberger et al., 1993) can be found in detail in Schauberger et al. (2001 and 2002). The results were calculated for a mechanically ventilated pig fattening unit with 1000 pigs. The following parameters were calculated every half-hour over the two year period: outlet air temperature, outlet air velocity, volume flow of the ventilation system, odour concentration of the outlet air. The odour flow in OU/s is calculated by the product of the volume flow of the building in m³/s and odour concentration of the outlet air in OU/m³.

The mean ambient odour concentrations are calculated using the Austrian Gaussian regulatory dispersion model (ÖNorm M 9440, 1992/96), a Gaussian plume model for single stack emissions. The model has been validated internationally (e. g. Pechinger and Petz, 1999). The mean odour concentrations of the dispersion model are transformed to instantaneous values depending on wind velocity and atmospheric stability. The meteorological background to calculate the instantaneous values using a peak-to-mean parameterisation is described in detail by Schauberger et al. (2000a).

Calculating Sensation and Separation Distance

The separation distance is calculated for eight wind direction classes (sectors of 45°) in two steps: First, sensation distances, defined as distances from the source where the momentary odour concentration is equal to the selected odour threshold, are calculated for each half–hourly period of the meteorological 2-

year time series. The second step is the calculation of the separation distance. Therefore, selected limits of the combination of odour concentration threshold T and probability of the threshold exceedance p are taken.

By the combination of these two values the protection level is defined. In Tab. 1 various national odour impact criteria were summarised. Based on these two values, we investigated the sensitivity of the separation distance in relation to the odour impact criteria. The separation distance was calculated for following combinations: the odour threshold 1, 3, and 5 OU/m³ and the exceedance probability 1, 3, 5, and 7%. The sensitivity of the separation distance S was analysed for a constant odour threshold T (Fig. 2 and Tab. 3) and for constant exceedance probability p (Fig. 3. and Tab. 4). For these two cases we used the slope of a regression line. The slope gives the change of the separation distance ΔS for the change of the odour threshold ΔT or the exceedance probability Δp (Tab. 3 and 4). The coefficient of variation of the separation distance was calculated for all 12 odour impact criteria to express the direction-depending variation (Tab. 5). The lower the coefficient of variation the higher the isotropy of the separation distances.

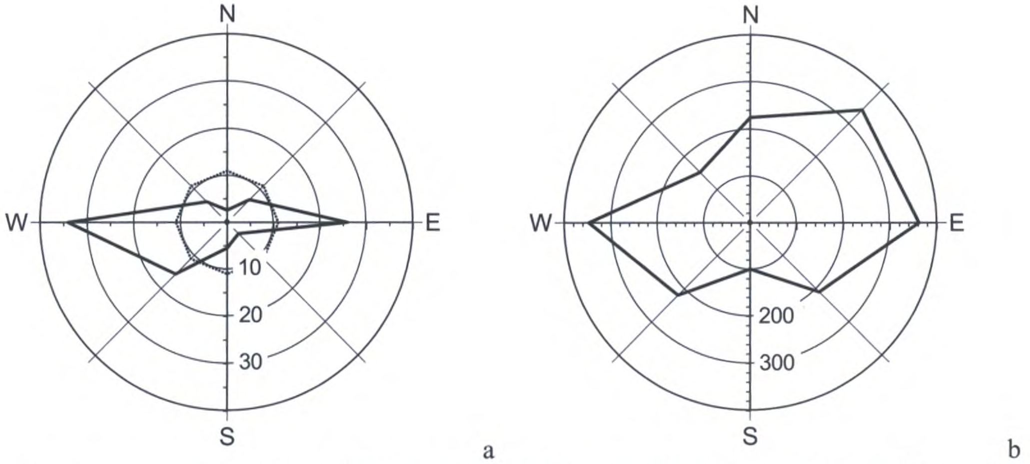


Figure 1. Polar diagram of (a) relative frequency distribution of the wind direction at Wels for eight 45° sectors. Calm conditions according to the ÖNorm M 9440 (1992/1996) with wind velocity less than 0.7 m/s evenly distributed over all directions (dotted line) and (b) separation distances (in m) calculated by the model for eight wind directions based on an odour concentration threshold of 1 OU/m³ and the probability of the threshold exceedance of 3 %.

As an example we selected a threshold of 1 OU/m³ and a probability of the threshold exceedance of 3% indicating that, during a typical year, there are 525 out of 17520 half hourly periods (3%) during which the ambient odour concentrations will be momentarily above 1 OU/m³ (Fig. 1). On the basis of the cumulative probability of the sensation distances for each of the eight wind direction sectors, the separation distances are calculated for the selected odour impact criterion. For a selected wind direction sector, the distance at which this definition is fulfilled, is called separation distance. E.g. for North wind, the corresponding separation distance points to the South of the odour source (Schauberger et al., 2002).

Table 1. Odour impact criteria: Limits of odour concentration and exceeding probability used in Austria, Germany, UK, Australia, The Netherlands, Denmark, New Zealand and Massachusetts (USA) (Schauberger et al.,2001)

Odour impact criteria[1]	Land use category[2]	Comment
Germany		
1 OU/m³ / 10%	pure residential areas and residential areas	
1 OU/m³ / 15%	residential and structured areas	
UK		
10 OU/m³ / 2%		Serious annoyance expected with near certainty
5 OU/m³ / 2%		Generally acceptable for existing installations. Emissions from stacks or large area sources may be acceptable at the relaxed end of the range
1 OU/m³ / 2%		No serious annoyance expected in the large majority of cases
1 OU/m³ / 0.5%		Safe target value for new sources
10 OU/m³ / 0.01%		Safe target value for new sources applicable to highly intermittent sources
Austria		
1 OU/m³ / 8% and 3 OU/m³ / 3%		threshold for reasonable odour sensation for medical purpose
Australia		
5 OU/m³ / 0.5%	rural and urban area	
2 OU/m³ / 0.5%	residential area	New South Wales
10 OU/m³ / 0.5%	residential areas	Victoria
The Netherlands		
1 OU/m³ / 2%	residential areas	existing units
1 OU/m³ / 0.5%	residential areas	new installations
1 OU/m³ / 5%	residential areas outside of villages and business areas	
Denmark		
5 - 10 OU/m³ / 0.1%		plants
0.6 – 20 OU/m³ / 1%		surrounding
New Zealand		
2 OU/m³ / 0.5%		property boundary
Massachusetts, USA		
5 OU/m³ / 0.5%		plant boundary

[1] Odour concentration threshold (OU/m^3) / exceeding probability p (%)
[2] The lend use category varies the accepted protection level

Meteorological Conditions

The meteorological data for January 30, 1992 to January 31, 1994 were collected at Wels, a site representative of the Austrian flatlands north of the Alps. The sample interval was 30 minutes. The city of Wels in Upper Austria is a regional shopping and business centre with a population of about 50,000. The surrounding area is rather flat and consists mainly of farmland. The mean wind velocity 10 m above the mean roof top level of 15 m is 2.2 m/s with a maximum velocity of about 13 m/s. The distribution of wind directions is shown in Fig. 1a.

Discrete stability classes have been determined based on sun elevation angle, cloud cover and low cloud base height, and wind speed (Reuter, 1970). The cloud data are measured at the Linz-Hörsching airport, about 13 km from Wels. Within the Reuter scheme, classes 2 to 7 can occur in Austria. Stability classes 2 and 3, which by definition occur only during daylight hours in a well-mixed boundary layer, class 3 allowing also for cases of high wind velocity and moderate cloud cover, occur in 26% of all cases. Stability class 4, representing cloudy and/or windy conditions including precipitation or fog, occurs day and night (43 %). Class 5 occurs with higher wind velocity during nights with low cloud cover, a situation which is not observed frequently at Wels (6 %). Classes 6 and 7 are relevant for clear nights, when a surface inversion, caused by radiative cooling, traps pollutants near the ground. Such situations occur in 25% of all cases.

Northerly and southerly winds show a behaviour which suggests an influence of the North-South oriented Alm river valley running into the Alpine foreland south of Wels. Northerly up-valley winds are more frequent during daytime, southerly down-valley winds more frequent during night. Therefore northerly winds are frequently associated with stability classes 2 to 4, southerly winds with classes 4 to 7. For both wind directions, the average wind velocity is rather small, with the 75 %-percentile at 1.1 m/s for North and at 1.9 m/s for South wind, respectively. In accordance with these findings, odour sensation at the separation distance for northerly winds (all half-hours) shows a maximum during daytime (between 7:00 and 20:00) and occurs frequently more often during the spring and summer months. For southerly winds, the odour sensation at the separation distance has its maximum in the evening (after 18:00), is large throughout the night, and shows a local maximum in the morning (before 6:00). South wind is more frequent from late summer through autumn to January.

Table 2: Relative frequency of the eight classes of wind directions

Wind direction	Relative frequency (%)
N	2.6
NE	6.8
E	25.9
SE	3.4
S	5.5
SW	15.6
W	34.1
NW	6.2

East and West winds are the dominant directions at Wels. Both directions show no strong variation over the day and some but no systematic variability across the year. Both directions are associated with much stronger wind velocities than North and South wind: the most frequent velocities for East wind are around 3 m/s, for West wind around 4 m/s. Maximum velocities are around 9 m/s for East wind and around 13 m/s for West wind. The distribution of stability classes with East and West winds is relatively similar to the overall distribution, due to the large frequency of these directions. Stability class 4 dominates, especially for West wind frequently in conjunction with high wind velocities, cloudiness, and rain. Classes 2 and 3 as well as 6 and 7 are more common with East wind associated with anticyclonic conditions. For East and West winds, odour sensation at the separation distance takes place more often in the second half of the day, with peaks around 22 CET, and from October to January. For both directions, the dependence of odour sensation on wind velocity shows several peaks, mostly at 1 and from 3 to 5 m/s. For East wind, odour sensation occurs only with stability classes 4 to 7; for West wind, it occurs with classes 4 to 6; classes 2 and 3 are free from odour sensation for the selected odour impact criterion, which is an effect of the large separation distances for these directions.

Results

The sensitivity of the separation distance S was calculated analysed for a constant odour threshold T (Fig. 2 and Tab. 3) and for constant exceedance probability p (Fig. 3. and Tab. 4).

For a certain odour threshold (e.g. 1 OU/m³ in Fig 2a) the sensitivity shows a minimum for the prevailing wind directions East, South-West, and East. The sensitivity of these wind directions lies between 7 and 12 m/%. The other directions this gradient is about 3 to 4-fold higher. For the highest protection level (1 OU/m³ / 1%) the coefficient of variation was lowest with about 36%. This means that the separation distance S doesn't change a lot with the direction. With growing exceedance probability the direction-dependent variability of the separation distances increases.

The influence of the exceedance probability is reduced for higher odour thresholds (Fig. 2b and 2c). For an odour threshold of 5 OU/m³ the sensitivity for the three prevailing wind directions is about 3 to 8m/% and about the 2-fold for the other wind directions which are more related to the valley wind system.

Table 3. Change of the separation distance ΔS (m) for increasing exceeding probability Δp (%).

	$\Delta S / \Delta p$ (m / %)		
Winddirection	Odour threshold T (OU/m³)		
	1	3	5
N	-17	-7	-7
NE	-39	-20	-17
E	-8	-5	-5
SE	-34	-16	-15
S	-37	-17	-15
SW	-12	-9	-8
W	-7	-4	-3
NW	-40	-21	-19

Table 4. Change of the separation distance ΔS (m) for an increasing odour threshold ΔT (OU/m³).

	$\Delta S / \Delta T$ (m / OU/m³)			
Winddirection	Exceeding probability p (%)			
	1	3	5	7
N	-28	-11	-11	-11
NE	-44	-30	-20	-11
E	-49	-44	-44	-44
SE	-42	-20	-13	-13
S	-44	-34	-19	-13
SW	-50	-47	-45	-43
W	-52	-47	-46	-46
NW	-43	-28	-17	-12

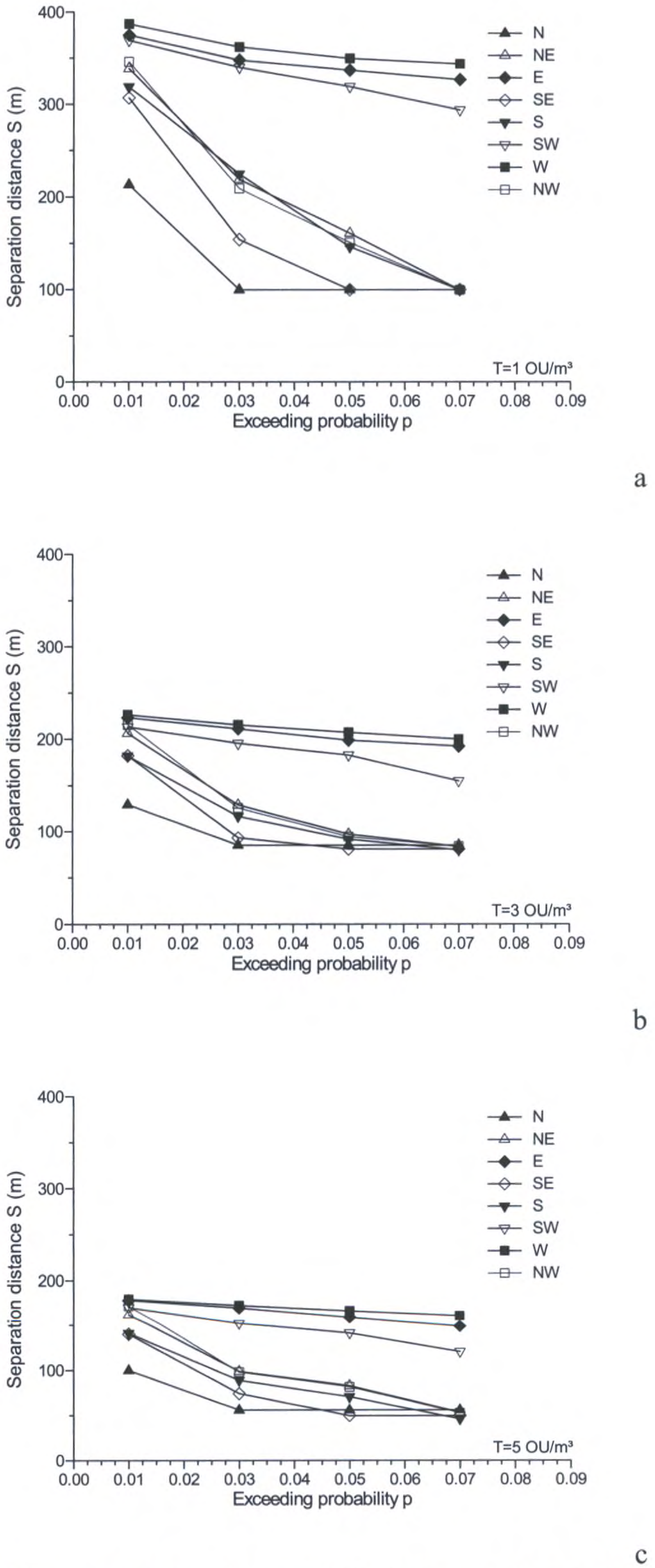


Figure 2. Sensitivity of the separation distance S (m) as a function of the exceeding probability p for various odour thresholds T of 1 OU/m² (a), 3 OU/m² (b), and 5 OU/m² (c).

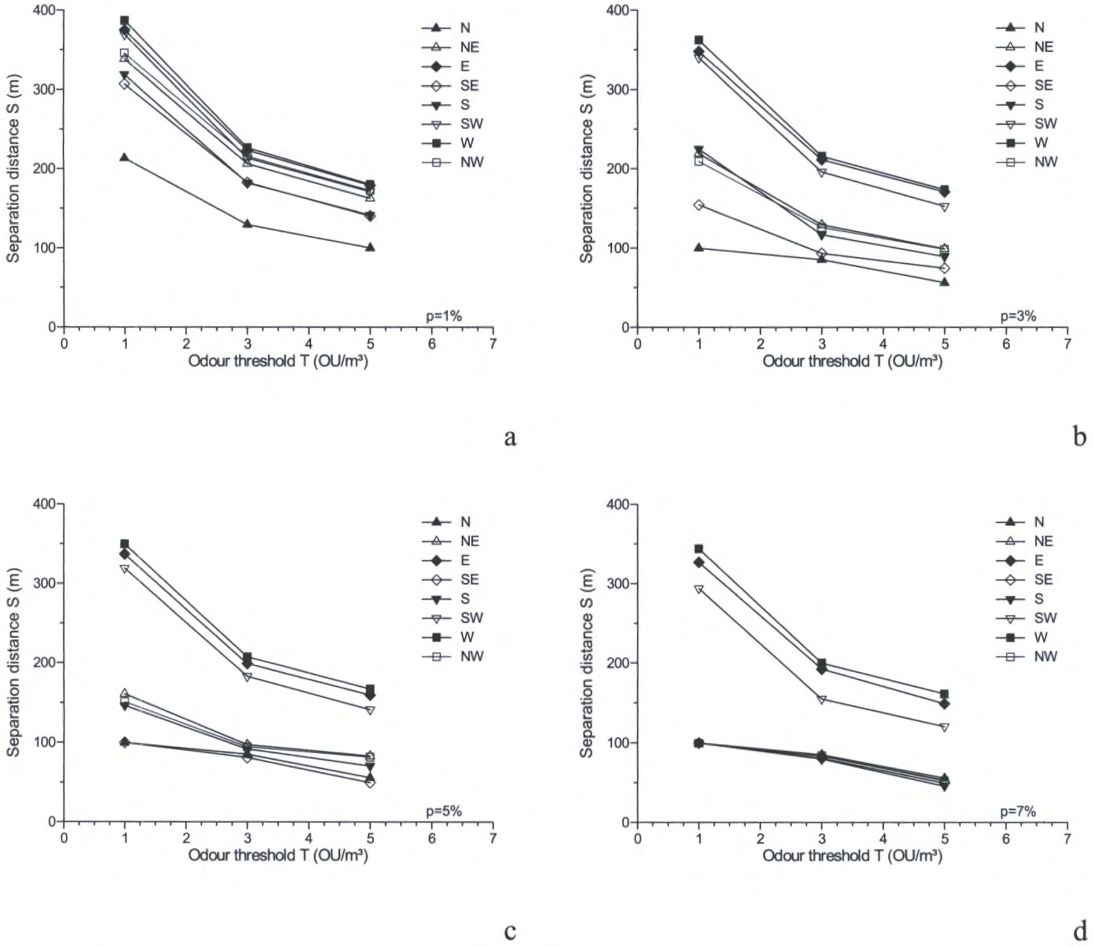


Figure 3. Sensitivity of the separation distance S (m) as a function of the odour threshold T for various exceeding probabilities p of 1 % (a), 3 % (b), 5 % (b), and 7 % (d).

For an exceedance probability of 1 % (Fig 3a) the sensitivity is nearly the same for all wind directions. For an increasing exceedance probability a separation for the prevailing wind directions (E, SW, W) and the other wind directions could be found. For an exceedance probability of 7% the sensitivity is about 45 m / OU/m³ compared to 10 to 13 m / OU/m³ for those wind directions which are dominated by the local wind system.

Table 5. Coefficient of variation of the separation distance S (OU/m³) for the selected odour impact criteria as a measure for the isotropy

	Coefficient of variation (%)			
Odour threshold T (OU/m³)	Exceeding probability p (%)			
	1	3	5	7
1	0.368	0.497	0.590	0.680
3	0.367	0.475	0.522	0.532
5	0.372	0.498	0.550	0.629

Discussion

From Table 1 it is apparent that odour thresholds in combination with their exceeding probabilities are explicitly related to land-use categories in Germany, the Netherlands, and Australia only. In all these countries, residential areas, in which, apart from existing installations, animal farming usually is not allowed, are best protected. However, the threshold systems are different. In Germany and the Netherlands, only the exceeding probability varies according to the land-use category. In Australia, the odour threshold varies, whereas the exceeding probability is fixed. In the UK, the odour thresholds are related to different levels of annoyance. Depending on the kind of odour threshold fulfilled for the investigated farm, the level of annoyance can be determined. Property domains are relevant for the validity of odour thresholds in Denmark, New Zealand, and Massachusetts, USA. Medical aspects led to the definition of the Austrian odour threshold.

The definitions of the various national odour impact criteria differ to quite an extent. Miedema and Ham (1988) and Miedema et al. (2000) found a strong relationship between the 98 percentile of the odour concentration and the percentage of the highly annoyed neighbours. They used an ambient odour concentration for an integration time of 1 hour, calculated by a dispersion model without consideration of the peak-to mean ratio. In Germany, the odour impact criterion is defined by an exceedance probability of 10% for a threshold of 1 OU/m^3. To apply this odour impact criterion, the calculated odour concentration (one hour mean value of the regulatory dispersion model) is multiplied by a constant factor of 10 by using the Gaussian dispersion model or the factor 4 for the recently publishes Lagrange models (AUSTAL 2000G).

Apart from the exceedance probability, the odour concentration threshold of the impact criterion is of importance. The odour is measured by the human nose as a sensor by comparing a diluted odour sample with odour free air. This means that the detection threshold of 1 OU/m^3 can only be perceived in an odour free environment (laboratory). Therefore the perceived odour concentration in the field must be higher than 1 OU/m^3 to be distinguished against the background concentration. Field experiments must be designed such that an odour source can be distinguished against the background odour. Nicell (1994) assumes an odour concentration of 3 OU/m^3 to allow for a discrimination, and one of 5 OU/m3 for unmistakable perception (also defined as a complaint level).

Further on, the perception of the odour intensity goes with the logarithm of the odour concentration according to the Weber-Fechner law (e. g., Misselbrook et al., 1993). Based upon laboratory-based experiments on perceived intensity, the Environment Agency, UK (2002), defines: 1 OU/m^3 is the point of detection, 5 OU/m^3 is a faint odour, and 10 OU/m^3 is a distinct odour. The discrepancy between the definition of 1 OU/m^3 in the laboratory by using odour free air and the situation in the field was solved by introducing the sniffing unit (van Langenhove and van Broeck, 2001; Defoer and van Langenhove, 2003).

All these aspects of the odour impact criteria show the difficulties to compare various methods used in different countries.

Conclusions

The odour impact criteria were selected predominantly by the administration and the legislation. The argumentation for the selection of the two values of the odour threshold and the exceedance probability seems not very sound.

By the fact that the odour impact criteria are the interface between the dispersion model and the annoyance potential a critical evaluation of the selected values should be done.

References

Defoer, N.; van Langenhove, H.; 2003. Determination of odour emissions from pig farms for regulatory purposes in Flanders. in International Symposium on Gaseous and Odour Emissions from Animal Production Facilities. 152-160, Horsens, Denmark.

Environment Agency, 2002. Integrated Pollution Prevention and Control (IPPC). Horizontal Guidance for Odour Part 1 – Regulation and Permitting (draft). Scientific and Technical Information Service, Bristol, UK.

Miedema, H. M. E.; Ham, J. M.; 1988. Odour annoyance in residential areas. *Atmospheric Environment*, 22 (11), 2501-2507

Miedema, H. M. E.; Walpot, J.I.; Vos, H.; Steunberg, C.F.; 2000. Exposure-annoyance relationship for odour from industrial sources. *Atmospheric Environment*, 43, 2927-2936.

Misselbrook, T.H.; Clarkson, Pain, C.R.; B.F.; 1993. Relationship between concentration and intensity of odour for pig slurry and broiler houses. *J of Agricultural Engineering Research*, 55, 163-169

Nicell, J.A.; 1994. Development of the odour impact model as a regulatory strategy. Int. J. *Environment and Pollution*, 4, 124-138.

ÖNorm M 9440, 1992/1996. Ausbreitung von luftverunreinigenden Stoffen in der Atmosphäre; Berechnung von Immissionskonzentrationen und Ermittlung von Schornsteinhöhen [Dispersion of pollutants in the atmosphere – calculation of the ambient air concentrations and determination of stack heights]. Österreichisches Normungsinstitut, Vienna.

Pechinger, U., Petz, E., 1999. Model evaluation of the Austrian Gaussian plume model ON M 9440: comparison with the Kincaid data set. *Int. J. Environ. Pollut.* 5, 338 – 349.

Reuter, H.; 1970. Die Ausbreitungsbedingungen von Luftverunreinigungen in Abhängigkeit von meteorologischen Parametern [Dispersion conditions of airborne pollutants in dependence on meteorological parameters]. *Archiv für Meteorologie und Geophysik, Bioklimatologie* A, 19, 173-186

Schauberger, G., Wudy, W., Cernohorsky, W., 1993. Statistisch-repräsentative Erhebung der Stallklimasituation in Rinder- und Schweineställen in Niederösterreich - Tierbestand, Gebäude und Lüftung. *Wiener Tierärztliche Monatsschrift*. German. 80, 364-374.

Schauberger, G.; Piringer, M.; Petz, E.; 1999. Diurnal and annual variation of odour emission of animal houses: a model calculation for fattening pigs. *Journal of Agricultural Engineering Research*, 74 (3) 251-259

Schauberger, G.; Piringer, M.; Petz, E.; 2000a. Diurnal and annual variation of the sensation distance of odour emitted by livestock buildings calculated by the Austrian odour dispersion model (AODM) *Atmospheric Environment*, 34 (28) 4839-4851

Schauberger, G.; Piringer, M.; Petz, E.; 2000b. Steady-state balance model to calculate the indoor climate of livestock buildings demonstrated for fattening pigs. *International Journal of Biometeorology*, 43 (4) 154-162

Schauberger, G.; Piringer, M.; Petz, E.; 2001. Separation distance to avoid odour nuisance due to livestock calculated by the Austrian odour dispersion model (AODM). *Agriculture, Ecosystems & Environment*, 87 (1) 13-28

Schauberger, G.; Piringer, M.; Petz, E.; 2002. Calculating direction-dependent separation distance by a dispersion model to avoid livestock odour annoyance. *Biosystems Engineering*, 82 (1) 1-24

van Langenhove, H.; van Broeck, G.; 2001. Applicability of sniffing team observations: experience of field measurements *Water Science & Technology*, 44, 65–70

Measuring Hydrogen Sulfide Emissions from Manure Samples in a Laboratory Setting

D.R. Schmidt, B.C. Martinez, and C.J. Clanton

Department of Biosystems and Agricultural Engineering, University of Minnesota, St. Paul, MN

Abstract

Measurement of gas emissions from area sources (feedlots, lagoons, manure storages, etc.) offers a considerable challenge. These emissions are dependent on manure characteristics and the microclimate near the area source. Several methods are available to measure these emissions including micrometeorological techniques, wind tunnels, and flux chambers. There are tradeoffs between accuracy and cost with each measurement method. This research effort was conducted in a controlled laboratory experiments to accurately measure gas emissions from manure for the ultimate use in mass transfer models.

The project objectives were to develop a laboratory experimental procedure that would be an accurate, low-cost method gas emission measurement technique in a laboratory setting. Initially, this protocol could be used to determine general trends in emission and relative rankings in gas emission mitigation strategies. Ultimately, this laboratory technique could then be applied to economically quantify actual emissions from individual area sources for regulatory purposes and could be used to compare and quantify emission reductions resulting from gas control technologies.

Hydrogen sulfide (H_2S) emissions from a sample of 100 ml fresh swine manure were evaluated using a flask method (figure 1). The total flask volume was 250 ml. Nitrogen gas was used as a carrier at six different airflow rates to demonstrate the relationship between H_2S emissions and airflow. H_2S concentration in the exhaust gas was initially measured every two minutes until concentration stabilized, then every 15 minutes for 480 minutes using a Jerome® meter.

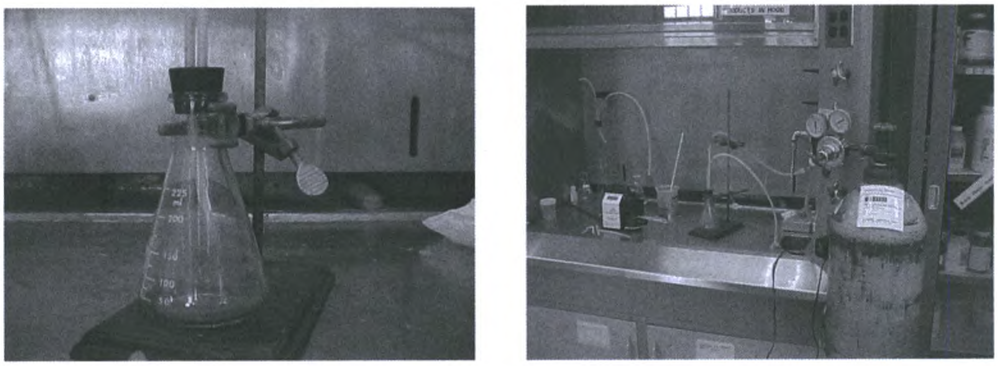


Figure 1. Experimental Setup Photo.

The observed mean H_2S air concentrations and emission rates for six airflow rates are:

	50-100 minutes		400-480 minutes	
Airflow rate L/min	Concentration ppm	Emission µg / min	Concentration ppm	Emission µg / min
0.34	1.92	0.84	1.10	0.48
1.5	0.76	1.46	0.47	0.92
3.0	0.57	2.22	0.33	1.27
4.0	0.39	2.03	0.22	1.13
6.0	0.31	2.40	0.13	0.97
8.0	0.26	2.66	0.12	1.27

After an equilibration period of 10 min, H_2S concentration increased in the airflow. For all airflow rates, except 0.34 L/min, the H_2S concentration stabilized and became fairly constant after 40 min. Then the H_2S concentration slowly decreased from 50 min to the end of data collection. For the 0.34 L/min rate, the H_2S concentration showed a distinct peak at 50 minutes with a maximum value of 2.10 ppm before decreasing at a much faster rate than the other five airflow rates. There is a decrease in H_2S concentration with the diluting affect of increased airflow rate. Note that the mean H_2S concentration collected between 50 and 100 min is approximately twice the mean concentration collected a time greater than 400 min.

Emission rates increased during the initial period (first 30-50 minutes), followed by a slow decay curve. This initial increase and subsequent decrease could not be explained with the amount of data taken but does show proof of concept for the experimental method. The emission information takes into consideration the airflow rate in the calculation of emission rates. Preliminary observation indicates the emission rate might level off or become independent of the airflow rate at about 3 L/min, which may indicate a shift from transfer rate limitation across the surface to H_2S generation within the manure for determining the overall mass transfer rate.

In a second laboratory study, part of a single swine manure sample was used at full strength (100%), labeled Strong and part of the manure sampled was diluted with distilled water (80% water:20% manure), labeled Weak for a Strong:Weak ratio of 5:1 ratio. Adjusting the Weak sample (20% dilution) data by multiplying the observed concentrations and emissions by a factor of five resulted in significant differences (p-value ≤ 0.05) compared to the Strong sample (100% dilution) data. This indicates a non-linear or an interaction of manure strength with airflow rate and/or the air collection process and final H_2S concentrations and emission rates.

Preliminary results are encouraging to continue developing a laboratory technique to quantify gas emissions from manure. Controlling the laboratory environment around the sample and data collection portion of the research (eliminating factors that interfere with data analysis) is critical to the success of the project and to the ultimate value of the data to the livestock and poultry industries.

Coupling Gaseous Ammonia, Fine Particulate Matter, and the Role of Precursor Transport in North Carolina

Curtis Scott and Viney P. Aneja
North Carolina State University, Raleigh, NC 27695, USA

Abstract

Approximately 10 million hogs reside in South Eastern North Carolina emitting 275 tons/day of ammonia. Ammonia can enhance fine particulate matter, or $PM_{2.5}$, through a series of chemical reactions. Several sites surrounding the hog farms (Forsyth, Guilford, New Hanover, Buncombe, Wake North Carolina) were selected to obtain $PM_{2.5}$ data from the IMPROVE monitoring network. The data collected from 2000 through 2004, summertime months (June, July and August) were examined because of the unique meteorological conditions conducive to PM2.5 formation. This was compared for seasonal variation by data collected during other seasons. The $PM_{2.5}$ data is divided into three groups. Group A (12AM – 11PM) entails the entire Julian day, group B (8AM – 8PM) is considered daylight hours, and Group C (10AM – 4PM) is considered the peak photochemical period. The three groups are used to gauge whether or not $PM_{2.5}$ formation is affected by photochemical reactions. The HYSPLIT (Hybrid Single-Particle Lagrangian Integrated Trajectory Model) computed 48 hour back trajectories on air masses for each of the associated peak high $PM_{2.5}$ episodes at both the 10 m (surface) and the 5000 m levels in order to determine if the air masses originated from the designated hog farm area. The correlations among meteorological variables (e.g., wind speed and direction, temperature, relative humidity) and the mass concentrations of $PM_{2.5}$ and its composition are being analyzed using several statistical methods (e.g., simple/multiple linear regression).

Using Modeled Atmospheric Dispersion of Bioaerosols Emitted from a Planned Multi-Site Consumer Egg Production Facility to Consider Public Health Concerns

Jens Seedorf

Institute for Animal Hygiene, Welfare and Behaviour of Farm Animals,
University of Veterinary Medicine, Hannover, 30559, Germany

Abstract

Gaseous and particulate pollutants in livestock buildings are potential health hazards for both animals and humans. Because ventilation systems transport loads of pollutants into the surrounding air, it is likely that this also represents a threat to the environment. Housed poultry flocks are among the animal production facilities which generally cause high emissions of particulate matter. Eggs are often produced commercially in large-scale laying hen confinement units, which may have a lasting negative effect on the environment and on nearby residents in particular. Licensing of new livestock buildings is therefore a difficult procedure under the current legislation, which attempts to balance the interests of all parties involved. The so-called bioaerosols (dust, dust-related compounds, microorganisms) are the emissions most suspected of having adverse health effects on residents in the vicinity of such livestock operations. Therefore, dispersion models are normally and officially used to assess the risks to the environment and public health. In this context we conducted a case study using a numerical dispersion model to predict the receptor concentrations of bioaerosols in the ambient air of a multi-site livestock building complex with nearly 1,000,000 laying hens in 31 livestock buildings equipped with more than 500 exhaust chimneys. However, the data necessary for input into the model is mostly based on the literature. In many cases such data are not available (e.g. the decrease in vitality of microorganisms due to their half-life times in the airborne state), are not really representative (e.g. emission rates), or are of unknown precision (e.g. particle size distribution and derived deposition rates). These factors are a source of uncertainty in the predictability of the model. In order to overcome this lack of data to some extent, we conducted a series of measurements in an existing laying hen building to verify the proposed input data for the assumed emission rates. In some cases (e.g. dust), the emission rates were found to differ only negligibly from those given in the literature). In others (e.g. total bacteria), the values measured exceeded those in the literature by a factor of 13. In addition to these uncertainty factors, we discuss the no less important interrelationship between the calculated receptor concentration field and public health issues.

On-line Measurement of Airborne Particulate Matter with an Optical Particle Sizer in Three Different Laying Hen Husbandry Systems

J. Seedorf[1] and T. Banhazi[2]
[1]Institute for Animal Hygiene, Welfare and Behaviour of Farm Animals, University of Veterinary Medicine Hannover, Foundation, Germany
[2]South Australian Research & Development Institute (SARDI), University of Adelaide, Roseworthy Campus, Australia

Abstract

The concentartions of airborne particles were measured using a direct-reading optical particle sizer in three different sections of an experimental barn (cages, floor, aviary) housing laying hens. In each section, six 24 hours surveys were conducted. The aviary compartment have shown the highest average total particle concentration of 624,813 particles per liter air. Only 21.5 % and 9.3 % of this concentration was detectable in the floor systems and in the caged compartment, respectively. The difference between the concentrations measured using the optical particle sizer and the gravimetrical method was large. Therefore, further validations will be necessary to ensure that the values directly displayed by the optical particle sizer will closely correspond with gravimetric measurements.

Apart from particle measurements, additional activity monitoring was made with infrared motion sensors to explore the association between these two previously mentioned parameters. A clear relationship between animal activity and airborne particle concentrations were demonstrated by this study, in agreement with previous results. A time-series graph of dust concentrations recorded in the aviary section of the experimental building is shown as an example. The variation in dust concentration could be partially related to the lighting regime used in the compartment and partially to human activities, such as control walks through the section.

Introduction

The traditional gravimetric measurement method is frequently used for the determination of the airborne particle concentrations in livestock buildings. This method is cumulative over time and cannot be used to show possible peak concentrations during the measurement period. Another disadvantage of cumulative dust sampling methods is the lack of information on particle size distribution. One exception is the operation of multi-stage cascade impactors with specific particle size-related cut-offs, which are able to collect information on size distribution. However, the number of impactor stages that can be used in these instruments is limited and therefore information collected on specific particle size ranges is often unsatisfactory. On the other hand, detailed information on particle size distribution is desirable, if particle properties in relation to inhalability and particle settling velocities are to be better understood. Particle size will ultimately impact on the respiratory health of farm workers and/or animals, and on passive dust reduction rates by sedimentation, especially under the low air velocities encountered in livestock buildings. Furthermore, the magnitude of the mass flow of emitted dust is dependent on the overall ratio among all airborne particle sizes. The emission of larger particles will result in relatively higher emission loads in the environment, when compared to small particles on a weight basis.

The utilization of optical particle counters and sizers (capable of continuously measuring dust concentrations) is one way to overcome the shortcomings of more traditional dust sampling methods. Differences in animal activity, ventilation rates and manure management systems used in different types of animal buildings may affect the quantity and quality of airborne dust (Banhazi et al., 2004). This is true in existing poultry keeping systems, where laying hens can be kept in cages, aviaries or on the floor.

The aims of this study were to (1) assess the utilization of an optical particle sizer in poultry buildings, (2) demonstrate its usefulness to monitor changes in airborne particle concentrations over time and (3) relate these changes to animal and human activity using infra-red motion sensors. Particle measurements were therefore made in an experimental laying hen house containing all three husbandry systems in separate compartments, thus allowing the comparison of dust concentrations between the different poultry housing systems.

Methods

Livestock Building

In an experimental livestock building, laying hens were kept in a barn with conventional cages, in an aviary and in a floor-based system with an extra scratching area. Due to the structure of the experimental livestock building, the length of each compartment was approximately 73 m but the width of each forced ventilated compartment varied between 3.5 and 5.2 m. In each of these three compartments approximately 1,500 hens were housed. The same commercially available feed was used during the trial in all compartments and the in the cage system dried manure was removed by conveyor belts.

Measurements

On-line measurements were carried out with an optical particle sizer, fitted with 15 different size channels from 0.30 µm to 20 µm (Aerosol Spectrometer 1.108, Grimm Aerosol Technik GmbH & Co. KG, Ainring, Germany). Recorded data were displayed as particle per liter (p/l). An integrated and removable backup filter allowed gravimetric measurement of deposited dust and thus allowing comparison between gravimetrically determined dust concentration and concentration calculated by the instrument itself.

A 24-hour measurement cycle starting from 06.00 h was chosen with a time resolution of 1 minute. After each monitoring period completed in one compartment the device was moved to the next compartment. Six measurement cycles in each husbandry system within one month were conducted in winter. The instrument was installed about 1.5 to 1.6 m above the floor.

Animal activity detection was carried out with infra-red motion sensors in conjunction with a signal-processing interface, as described previously (Pedersen and Pedersen, 1995). Two sensors along the length of the compartments were placed near the sidewalls to detect activity up to 6 meters from the sensors. The sensors were aligned to the longitudinal axis of the barns where most of the hens spent their time usually.

Data Processing

Recorded data sets were checked for inaccurate values and outliers. The maximum count range was 2 millions particles per liter air according to the manufacturer's specifications. Even if this maximum value was exceeded the value recorded was kept at 2 million particles per liter. Such extreme dust concentrations were only recorded in the aviary systems.

Due to the large variations within the dataset, the particle concentrations and activity values are presented as median values together with percentiles, minimum and maximum values. Mann-Whitney U-test was used to test the significance of different treatments as described by the housing systems. Spearman rank correlation (r_S) was used to show an interrelationship between particle concentrations and the magnitude of flock activity.

Results and Discussion

The monitoring of particle concentrations has revealed significant differences ($p<0.01$) between the three investigated laying hen compartments (treatments). Figure 1 clearly shows that particle concentrations were the lowest in the caged system while the aviary system produced the highest concentrations. The mean particle concentrations (expressed as arithmetic mean of the median values shown in Fig. 1) calculated were 58,078 p/l air in the cage barn, 134,486 p/l air in the floor system and 624,813 p/l air in the aviary system. The differences between device-readings and gravimetric measurements were considerable. The particle sizer recorded 3.4 times more dust compared with the gravimetrically determined dust quantities. As consequence, direct concentration readings have to be used with caution. Adjustment between the two readings might be possible if a correction factor was introduced. Such correction factor would need to take into consideration the specific physical density of livestock dust and other factors potentially influencing the accuracy of the optical readings. For example, the dust composition may vary over time in relation to activities such as feeding or re-bedding of buildings, when particles with different densities and size could appear in the air livestock buildings. Furthermore, the shape and surface of particles play an important role in terms of their light scattering properties and therefore their final size classification. However, recent investigations with a photo-optical instrument (similar to what was used in this study) have shown that a correction factor could help to re-adjust the predicted dust concentrations to correspond with gravimetrically obtained results (Taylor and Reynolds, 2001).

When comparing total particle counts with particle counts up to 4 µm in diameter (respirable dust fraction) the ratios between these fractions were different between the compartments. In the caged system 97.7 % of the particles were classified as respirable particles. In the aviary only 96.8 % of all signals recorded by the instrument were recognized as respirable dust fraction, which indicated the presence of greater number of larger particles in the aviary system than in the air of the caged system. The availability of a 3-dimensional space enriched with different structures obviously motivated the birds to fly and be more active generally. Turbulences associated with the increased bird activity would have stirred-up settled dust, thus shifting particle sizes distribution towards the larger particle sizes. Interestingly, the highest ratio was found in the floor system with 98.5 %. It was hypothesized that birds had reduced wing movements in the floor system, resulting in smaller amounts of larger particles released. On the other had, caged birds might flap their wing more often, due to the limited cage space and thus generating greater amounts of larger particles. However, further work is required to confirm these theories.

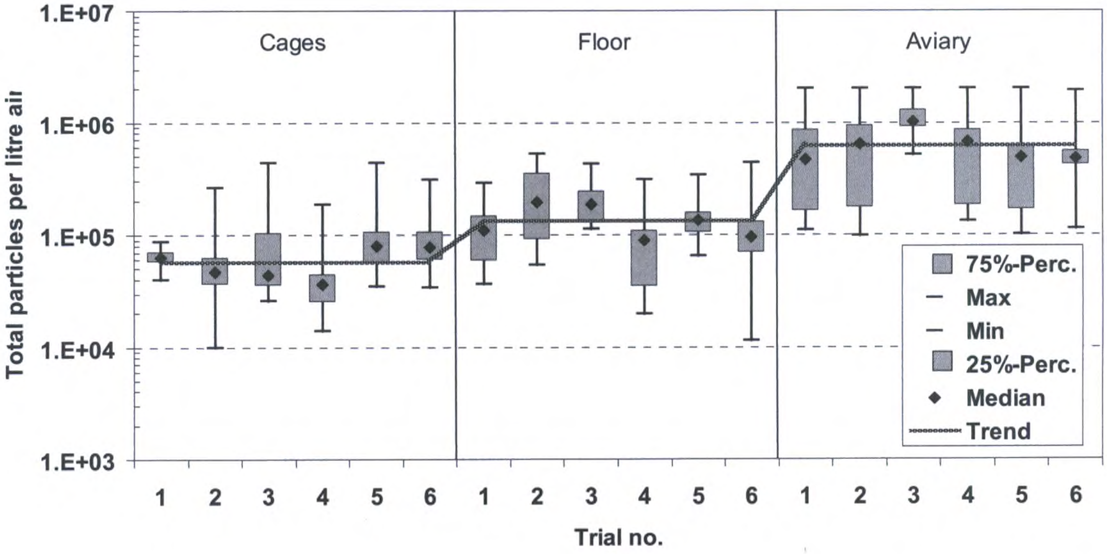


Figure 1. Median total particle concentrations in the three different compartments for laying hens. Incorporated trend line indicates the overall differences in concentrations.

Particle concentrations in livestock buildings are considerably influenced by the activity levels of the animals (Pedersen, 1993; Nannen and Büscher, 2006). Animals in general and poultry in particular are considerable emitters of hair, skin and feathers debris, respectively. Locomotion and typical behaviors such as flying or scratching in bedding material could result in the release of significant amounts of dust. Daily control walks of the animal keepers could trigger enhanced animal activity. Considering these factors could help to explain the large variations in dust concentrations, as presented in Fig. 1. The scale of variation becomes more obvious in Figure 2, which display a significant and positive correlation between particle concentration and animal activity ($r_S = 0.66$, $p<0.001$). The light regime had an obvious effect on particle concentrations via triggering increased animal activity. Single events such as control walks could have direct effects (a2), no significant effects (a3) or prolonged effects (a1). Despite the obvious advantages of the on-line dust measurements helping to explore the dynamic nature of particle concentrations in livestock buildings, a time-dependent and accurate recording system of the potential influencing factors is essential to help interpreting the findings comprehensively.

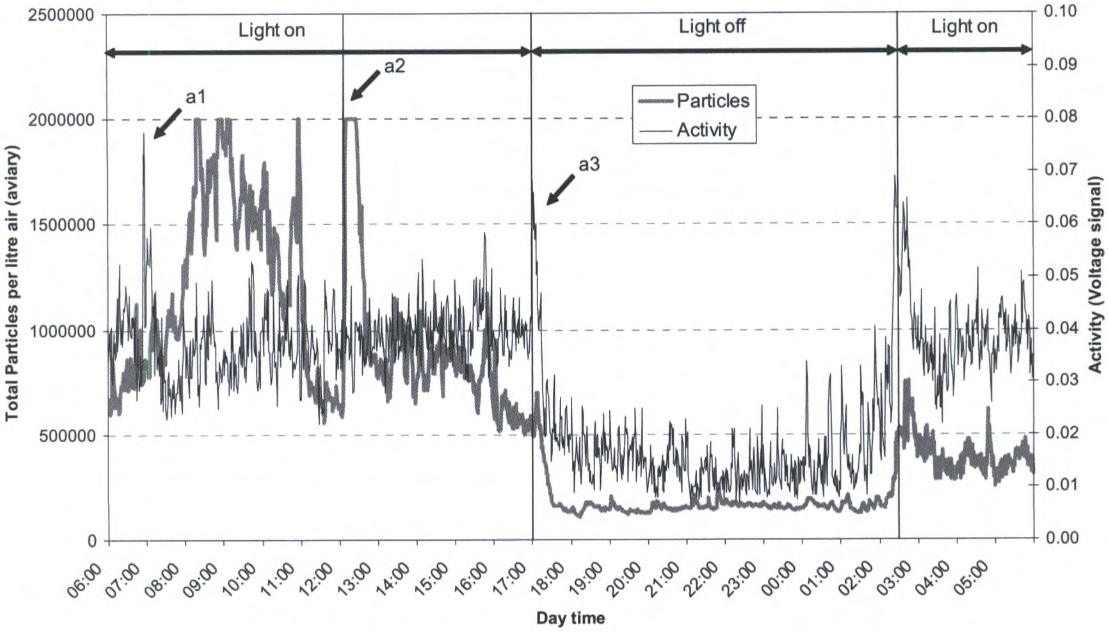


Figure 2. Example of a time-series of total particle concentration and monitored activities in the aviary system. Identified activities during the survey are marked with arrows (a1 to a3 = control walks).

Conclusions

Frequent monitoring of dust concentrations in livestock buildings is an important task, as high airborne particle concentrations could negatively affect the health of workers, animals and the environment. Traditionally, gravimetric dust sampling method is used to determine particle concentrations in the air of different livestock housing. However, this method does not provide continuous data over the sampling time. Direct-reading instruments such as optical particle sizer can overcome these limitations. These instruments, together with other on-line measuring techniques could facilitate the discovery of dynamic associations between environmental factors and aerial particle concentrations. To achieve this aim, particle measurements over 24 hours in conjunction with activity recordings were made in three different laying hen barns, where hens lived in cages, on floor and in an aviary. The following general conclusion can be drawn from this study:

1. Particle concentration is highest in the aviary system followed by the floor and the caged systems.
2. The different ratios between total and respirable particle counts (particles up to 4 µm in size) have shown that greater amounts of the larger particles were found in the aviary system in comparison to the other systems.
3. Animal activity and to some extent human interventions are considerably influencing the airborne particle concentrations in the aviary system.
4. A comprehensive recording system of factors potentially influencing particle concentrations is required to ensure reliable interpretation of the data.
5. The development and validation of a ‘correction factor’, which would re-adjust photo-optic measurements to correspond with gravimetric measurements would be desirable to provide researchers, health and environmental experts with reliable time-dependent data.

References

Banhazi, T., J. Seedorf, D.L. Rutley, and W.S. Pitchford. 2004. Factors affecting the concentrations of airborne particles in Australian piggery buildings. In *'Proceedings of ISAH Conference'*. St Malo, France. (Ed. F Madec) pp. 193-194. (ISPAIA)

Taylor, C.D., and S. J. Reynolds. 2001. Comparison of a direct-reading device to gravimetric methods for evaluating organic dust aerosols in an enclosed swine production environment. *Applied Occupational and Environmental Hygiene* 16(1): 78-83.

Nannen, C., and W. Büscher. 2006. Der Einfluss der Tieraktivität auf Staubfreisetzung und Partikelemission (The impact of animal activity on dust release and particle emissions). *Landtechnik*, 61(1): 36-37.

Pedersen, S. 1993. Time-based variation in airborne dust in respect to animal activity. *Proc. 4th Int. Symp. Livestock environment IV*, 6-9 July, Conventry, UK, 718-725.

Pedersen, S., and C.B. Pedersen. 1995. Animal activity measured by infrared detectors. *Journal of Agricultural Engineering Research* 61(4): 239-246.

Diurnal Variations of Ammonia and Hydrogen Sulfide Flux from a Dairy Manure Storage Pond in Idaho

Ron E. Sheffield[1] and Bruce Louks[2]

[1]University of Idaho, Twin Falls, ID 83301, USA; [2]Idaho Department of Environmental Quality,
Boise, ID 83706, USA

Abstract

Ammonia and hydrogen sulfide flux from a dairy manure storage pond were measured for 35-days on a 1,800-head dairy in south central Idaho. Continuous air samples were taken using a standardized wind tunnel floating on the pond and gas concentrations were averaged every 15-minutes using chemiluminescence. Diurnal variations of ammonia were found to be strongly correlated to solar radiation and temperature while variations of hydrogen sulfide were found to remain relatively constant. Average flux rates of 101.50 ug/m^2/s and 7.58 ug/m^2/s were measured for ammonia and hydrogen sulfide, respectively. Average emission rates for ammonia and hydrogen sulfide per milking cow were found to be 0.274 lbs/cow/d and 0.021 lbs/cow/d, respectively.

Ammonia-Based Air Permits for Idaho Dairies

R. E. Sheffield[1] and B. Louks[2]
[1]Waste Management Engineer, University of Idaho Extension, Twin Falls, Idaho
[2]Modeling and Monitoring Manager, Idaho Department of Environmental Quality, Boise, Idaho

Abstract

Over the past 5 years the dairy industry in Idaho has experienced tremendous growth in number of animals as larger operations are emerging on the rural landscape. This growth has heightened competition for air resources as population growth in Idaho has been moving the urban/rural interface further out into the rural areas and closer to dairy operations. Issues with dairy odor have been contentious over the past five years and public interest in regulating emissions from dairies has intensified. The Rules for the Control of Ammonia from Dairy Farms were developed at the request of the dairy producers and a local environmental advocacy group. The Rule allows for the implementation of a permit by Rule that would require dairies that would emit more then 100 tons of ammonia per year to implement various best management practices to reduce ammonia volatilization on the farm. During the rulemaking process it was agreed by all parties that the Rule would be re-visited annually to determine compliance issues and effectiveness in reducing ammonia emissions. As more scientific information emerges regarding ammonia emission factors, efforts to quantify emission reductions as a direct benefit of the Rule will be more credible and provide for better decision-making and environmental protection.

Introduction

The Independent Dairy Environmental Action League (IDEAL), on behalf of the Idaho Dairymen's Association (IDA) and the Idaho Conservation League (ICL) asked the Department of Environmental Quality (DEQ) to initiate a negotiated rulemaking to consider regulation of dairies, based on adoption of threshold numbers of dairy cows.

In this rulemaking, DEQ intends that dairies above the threshold numbers of cows will implement site-specific ammonia control best management practices (BMPs) and will be permitted by rule.

The text of the rule was developed by DEQ in conjunction with a negotiating committee made up of persons having an interest in the development of this rule. The committee represented individuals from the DEQ, Idaho Attorney Generals Office, Idaho Department of Agriculture (ISDA), IDA, Milk Producers of Idaho, Northwest Dairy Producers, ICL, Amalgamated Sugar Company, University of Idaho, and concerned citizens. The committee met regularly between April, 2005 and January, 2006 to develop this Rule.

The Environmental Quality Board approved a temporary Dairy Permitting rule on February 23, 2006 with an effective date of July 1, 2006. The temporary rule was published in the April Administrative Bulletin. Subsequently, it was published as a proposed rule in the May Bulletin. A public comment period was held from May 3 through June 5 culminating with a public hearing on June 5. Public comments will be reviewed and the proposed rule will be submitted to the Board for their consideration in mid-October. After the Board approves the proposed rule, a pending rule will be published in the Administrative Bulletin in January 2007 for consideration by the 2007 Legislature.

Who does the Rule effect?

The Rules for the Control of Ammonia from Dairy Farms are intended to provide a Permit by Rule to those dairies that may emit more than 100 tons of ammonia per year from their facility. The DEQ uses the annual 100 ton emission level as a threshold for requiring Permits to Construct for industrial, municipal and agricultural sources in Idaho. The threshold values (Table 1) for this rule were derived from manure excretion tables prepared by the American Society of Agricultural and Biological Engineers and the USDA – Natural Resources Conservation Service and were agreed upon by IDA and ICL. In addition to manure excretion, the IDA and ICL took into account their best professional judgment to account for differences in ammonia losses from various dairy housing types, manure storage systems, and land application practices.

The threshold values, presented in Table 1, make differentiations in terms of the type of dairy housing system as well as the type of liquid manure application system that is employed on the farm. Differences in threshold levels for drylot and free stall scrape dairies account for how each farm land-applies their manure and the relative amount of ammonia loss that typically occurs on the facility as well as during the application of manure and wastewater. Greater thresholds are allowed for dairies which export all of their manure and wastewater off the farm (Drylot = 7,089 head; Freestall Scrape = 3,893 head). Lower thresholds are allowed for dairies which apply their wastewater with center pivot drop nozzles or drag-hose (ground level) application systems. The lowest thresholds are used for openlot and freestall scrape dairies that apply wastewater with overhead pivot sprinklers or other conventional sprinkler systems. No differentiation is made between the type of farms and how solid manure is applied and/or incorporated. Lastly, freestall flush facilities, regardless of the farm's land application practices, greater than the threshold value of 2,293 head (1,638 AU) is subject to this rule.

Table 1. Animal Unit (AU) or mature cow thresholds to produce 100 tons NH_3/year.

Farm Type	Drylot	Free Stall/Scrape	Free Stall/Flush
	Animal Unit Threshold		
No land app	7,089	3,893	2,293
27% volatilization[1]	6,842	3,827	
80% volatilization[2]	6,397	3,700	
	Total Cow Threshold (Average weight of 1400 lbs)		
No land app	5,063	2,781	1,638
27% volatilization[1]	4,887	2,733	
80% volatilization[2]	4,569	2,643	

[1] *Assumes*: Expected level of N->NH_3 volatilization for: drop-hose or ground level liquid manure application

[2] *Assumes*: Expected level of N->NH_3 volatilization for: center pivot or other conventional sprinkler irrigation liquid manure application

The Rule will come into effect on July 1, 2006 with its adoption as a Temporary Rule by the Board of Environmental Quality on February 23, 2006. The temporary rule will be presented by the DEQ for public comment in 2006 and then will be voted upon by the 2007 Idaho Legislature for final adoption.

Rule Requirements

There are two requirements of the Rule: Registration and Inspection. Each dairy which exceeds the threshold must register within 15-days of the adoption of the Rule (July 16, 2006). Dairies, previously below the threshold levels, which expand beyond the threshold, must register within 15-days of their expansion. To register, dairies must provide the following information to the Idaho Department of Environmental Quality and the Idaho Department of Agriculture:

- Name, address, location of dairy farm, and telephone number,
- Type of dairy and the number of cows or Animal Units (AU), and
- Description of ammonia control BMPs used to meet the twenty-seven (27) point requirement.

Additionally, an initial qualifying inspection of each dairy will be completed within 30-days of receipt of the registration information. Inspections will be made to ensure a total of 27-points of BMP allocations are employed on the farm. Inspections will be made by personnel from the Idaho Department of Agriculture. Follow-up inspections will be conducted as part of ISDA's regular dairy inspection program.

The Rule recognizes that dairies and dairymen are unique and that no specific BMP is appropriate on all dairies. Therefore, a performance-based method, rather than requiring specific BMPs, was developed.

During the development of this Rule, various manure treatment systems and handling practices were evaluated for their effectiveness in reducing ammonia emissions (Table 2). Many scientific studies, extension bulletins, NRCS handbooks and EPA guidance documents were referred to in preparing the professional judgment toward to relative effectiveness in reducing ammonia and the allocation of points. An arbitrary point system, with a maximum of 20 points, was assigned to each practice; whereas, a practice receiving 20 points would equate to a system or practice that would result in a major reduction, approximately 70 percent, in ammonia emissions for that specific process. Each practice was rated on a year-round basis and as if all of the manure practically available for the practice was handled by the practice and variations due to normal seasonal use of each practice was taken into account in the points awarded to each BMP. Variations due to seasonal practices (such as corral harrowing or direct land application of liquid manure) and expected weather conditions have been factored into these ratings. Points awarded to land application practices assume that the practice is utilized on all manure that is applied. Points are allowed to be pro-rated to reflect actual waste treatment or handling that is occurring on each farm.

The emissions related to the management of exported manure are also addressed in the Rule. Dairy producers, under the Rule, can "take credit" for the management of ammonia conservation practices that occur on farms that receive their manure. For example, if a dairy exports the collected slurry (feces and urine) from a freestall scrape dairy to a neighboring farm which injects the manure a total of 25 points would be awarded: 15 for "Direct Utilization of Collected Slurry" and 10 points for "Soil Injection – Slurry", if used year-round or pro-rated to reflect the percentage of the year used. In order to take credit dairymen must keep records on the amount of exported material that left the farm. "3rd Party" receivers of the manure will not be inspected under this rule.

Three methods will be used under the Rule to monitor compliance with the Rule: Annual inspections, records and a deviation log. Each dairy will be inspected annually, as part of the existing wastewater management inspection, conducted by the Idaho Department of Agriculture. During annual inspections, the amount of ammonia control BMPs utilized will be compared to the registered control plan and the implementation of BMPs will be assessed. The inspector will also review applicable records such as manure export logs. Lastly, producers will be encouraged to keep a "Deviation Log" to report any equipment upsets and/or breakdowns that occur. Any repair that would be addressed in a reasonable timeframe would not constitute non-compliance with this rule.

Conclusions

Over the past 5 years the dairy industry in Idaho has experienced tremendous growth in number of animals as larger operations are emerging on the rural landscape. This growth has heightened competition for air resources as population growth in Idaho has been moving the urban/rural interface further out into the rural areas and closer to dairy operations. Issues with dairy odor have been contentious over the past five years and public interest in regulating emissions from dairies has intensified. Although Rules for the Control of Ammonia from Dairy Farms are not intended at controlling odor emissions, it will have beneficial aspects for odor. Ammonia emissions from any type of facility have a role in the formation of fine secondary aerosol particulates ($PM_{2.5}$) as it interacts with other compounds emitted by industrial and mobile sources in an airshed. The exact chemical process by which ammonia influences secondary aersol formation will vary across airsheds because the controlling parameters will vary. As required by the Clean Air Act, the Idaho DEQ does regulate the emissions of ammonia from large industrial sources. The dairy industry is recognized as the sector with the highest ammonia emissions in Idaho, and with the cooperation of all parties involved in negotiating this Rule, DEQ believes this a step in the right direction for controlling air pollution in Idaho. The Rule will be re-visited by the negotiated parties on an annual basis to determine compliance issues and effectiveness in reducing ammonia emissions. As more scientific information emerges regarding ammonia emission factors, efforts to quantify emission reductions as a direct benefit of the Rule will be more credible and provide for better decision-making.

Table 2. Ammonia Control Practices for Idaho Dairies.

System	Component	Ammonia Control Effectiveness [1]			Compliance Method[3]
		Open Lot	Freestall Scrape	Freestall Flush	
Waste Storage and Treatment Systems	Synthetic Lagoon Cover	15	20	20	1
	GeoteXtile Covers	10	13	13	1
	Solids Separation	3	3	3	3, 4
	Composting	4	4	4	1
	Separate Slurry and Liquid Manure Basins	6	10	-	1
	In-House Separation	0	12	0	1
	Direct Utilization of Collected Slurry	6	10	-	1, 3, 4
	Direct Utilization of Parlor Wastewater	10	10	10	1
	Direct Utilization of Flush Water	8	0	13	3, 4
	Anaerobic Digester	-	-	-	-
	Anaerobic Lagoon	-	-	-	-
	Aerated Lagoon	10	12	15	2
	Sequencing-Batch Reactor	15	20	20	2
	Lagoon Nitrification/Denitrification Systems	15	20	20	2
	Fixed-Media Aeration Systems	15	20	20	2
General Practices	Vegetative or Wooded Buffers (established)	7	7	7	1
	Vegetative or Wooded Buffers (establishing)	2	2	2	1
	Alternatives to Copper Sulfate	-	-	-	-
Freestall Barns	Scrape Built Up Manure	-	3	3	1
	Frequent Manure Removal	UD	UD	UD	-
	Tunnel Ventilation	-	-	-	-
	Tunnel Ventilation w/Biofilters	-	10	10	1
	Tunnel Ventilation w/Washing wall	-	10	10	3, 4
Open Lots and Corrals	Rapid Manure Removal	4	2	2	1, 2
	Corral Harrowing	4	2	2	1
	Surface Ammendments	10	5	5	2
	In-Corral Composting / Stockpiling	4	2	2	1
	Summertime Deep Bedding	10	5	5	1
Animal Nutrition	Manage Dietary Protein	2	2	2	2
Composting Practices	Alum Incorporation	12	8	6	2
	Carbon:Nitrogen Ratio (C:N) Ratio Manipulation	10	7.5	5	2
	Composting with Windrows	-	-	-	-
	Composting Static Pile	6	4.5	3	1
	Forced Aeration Composting	10	7.5	5	1
	Forced Aeration Composting with Biofilter	12	8	6	1
Land Application[2]	Soil Injection - Slurry	10	15	7.5	2
	Incorporation of manure within 24 hrs	10	10	10	2
	Incorporation of manure within 48 hrs	5	5	5	2
	Nitrification of lagoon effluent	10	10	15	3, 4
	Low Energy/Pressure Application Systems	7	7	10	1
	Freshwater Dilution	5	8	8	1, 2
	Pivot Drag Hoses	8	8	10	1
	Subsurface Drip Irrigation	10	10	12	1

[1] The ammonia emission reduction effectiveness of each practice is rated numerically based on practical year-round implementation. Variations due to seasonal practices and expected weather conditions have been factored into these ratings. Not implementing a BMP when it is not practicable to do so, does not reduce the point value assigned to the BMP, nor does it constitute failure to perform the BMP. UD indicates that the practice is still under development.

[2] Land application practices assume practice is conducted on all manure; points will be pro-rated to reflect actual waste treatment; points can be obtained on exported material with sufficient documentation.

[3] Method used by inspector to determine compliance:

1=Observation by Inspector

2=On-Site Recordkeeping Required

3, 4=Deviation Reporting Required. Equipment upsets and/or breakdowns shall be recorded in a deviation log and if repaired in a reasonable timeframe does not constitute non-compliance with this rule.

A Comparison of Ammonia Emission Rates Between a Tunnel and Sidewall Ventilated Broiler House

Ronald L. Siefert[1] and Joseph R. Scudlark[2]
[1]U.S. Naval Academy, Chemistry Department, Annapolis, MD, 21402, USA
[2]University of Delaware, Graduate College of Marine Studies, Lewes, DE 19958, USA

Abstract

Current estimates indicate that 25-30% of the total nitrogen loading to Chesapeake Bay is derived from atmospheric input[1]. In regions having intensive agricultural livestock activities, a large and increasing fraction of the nitrogen deposition is in the form of NH_x (NH_3 + NH_4^+); in some locations, it has been shown to be comparable to NO_3^- deposition[2, 3]. Despite this importance, NH_3 emission estimates are poorly constrained. A major reason is because NH_3 emission estimates rely on extrapolation of emission factors, which are based on many assumptions, and can be highly variable depending on the specific animal husbandry and manure disposal practices[4, 5]. In particular, many of the published emission factors are derived from studies conducted in Europe, where growing methods differ considerably from those practiced in the U.S.

Various approaches have been used to quantify NH_3 emissions from confined animal feeding operations, each with its own inherent limitations and level of uncertainty. We have developed a simple, novel, inverse plume dispersion modeling technique that employs gas-phase NH_3 concentrations obtained from a three-dimensional array of ~20 Ogawa passive samplers, for determining NH_3 emissions from poultry houses[6]. The passive samplers are deployed at 3 heights on a series of 10 m sampling towers, strategically situated in an array downwind from the house. Compared with traditional methods of NH_3 sampling (*e.g.,* denuders, gas scrubbers) passive samplers are ideally suited for this purpose since they are inexpensive, robust, do not require electricity or other supporting equipment (*e.g.,* pumps, mass flow controllers), and can be left unattended for extended periods. The samplers are typically deployed for 6-12 hr periods over the various stages of the grow-out cycle, and under differing meteorological regimes. A Lagrangian-based, inverse Gaussian dispersion plume model is fitted to the observed NH_3 concentration profile to determine the corresponding point source strength (*i.e.,* total NH_3 emissions from the house). Normalizing this value to the number of birds present in the house, allows one to calculate an emission factor corresponding to each sampling period (g NH_3-N/bird/day), which can be integrated to yield an emission factor for the entire flock. The primary advantage of this approach is that it allows for the determination of NH_3 emissions at various growth stages under "real world" growing conditions.

Our initial study involved an 11,500 bird, side-wall ventilated broiler house[6]. A total of 7 field deployments were performed at differing growth stages of two different summertime flocks, under both day and night meteorological conditions. Observed NH_3 emission factors ranged from 0.27 to 2.17 g NH_3-N/bird/day, with an arithmetic mean of 1.18 g NH_3-N/bird/day and flock-weighted mean of 0.74 g NH_3-N/bird/day.

In a comparative study reported here, NH_3 emissions were determined at a nearby, 18,600 bird tunnel ventilated poultry house under comparable summer conditions. A total of 4 separate plume characterization studies were conducted over the final 3 weeks of the 6 week grow-out cycle. Using a similar sampling grid of passive samplers and Gaussian dispersion modeling approach, the total NH_3 flux was determined for each field deployment. The mean emission factor observed at the tunnel ventilated house (0.11 g NH_3-N/ bird/day) was an order of magnitude lower than that previously observed at the same growth stage at the side-wall ventilated house. Although not all growing variables were measured, the large difference in emission factors between the two ventilation regimes suggest that modern, tunnel ventilated houses may result in a significant decreased in NH_3 emissions compared with traditional growing methods. We speculate that the lower emission rates we observe in the tunnel ventilated house may be due to differences in litter moisture resulting from greater air movement in the tunnel house, and in particular, the use of misters required for maintaining the optimal growing temperature in the side-wall house during summer.

References

Castro, M.S. and Driscoll, C., 2002. Atmospheric nitrogen deposition to estuaries in the mid-Atlantic and northeastern United States. *Environmental Science & Technology* 36, 3242-3249.

Scudlark, J.R., Jennings, J.J., Roadman, M.J., Savidge, K.B. and Ullman, W.J., 2005. Atmospheric nitrogen inputs to the Delaware Inland Bays: the role of ammonia. *Environmental Pollution* 135, 433-443.

Walker, J.T., Aneja, V.P., and Dickey, D.A., 2000. Atmospheric transport and wet deposition of ammonium in North Carolina. *Atmospheric Environment* 34, 3407-3418.

Battye, R., Battye, W., Overcash, C., and Fudge, S., 1994. Development and Selection of Ammonia Emission Factors. U.E. EPA Report EPA/600/R-94/190, Research Triangle Park, NC.

Chimka, C.T., Galloway, J.N., and Cosby, B.J., 1997. *Ammonia and the Chesapeake Bay Airshed.* Chesapeake Bay Program, Scientific and Technical Advisory Committee Publication 97-1, 49pp.

Siefert, R.L., Scudlark, J.R., Potter, A.G., Simonsen, K.A. and Savidge, K.B., 2004. Characterization of atmospheric ammonia emissions from a commercial chicken house on the Delmarva Peninsula. *Environmental Science & Technology* 38, 2769-2778.

Effect of Urease Inhibitor on Ammonia Emissions in Layer Houses

A. Singh[1]*, W.D. King[2], K.D. Casey[3], R.S. Gates[1] and T. Pescatore[2]
[1]Biosystems and Agricultural Engineering, University of Kentucky, Lexington, KY
[2]Animal and Food Sciences, University of Kentucky, Lexington, KY
[3]Agricultural and Research Extension Center, Texas A&M University, Amarillo, TX 79106

Extended Abstract

Air quality and potential gaseous emissions from livestock facilities are a concern as they are affecting human health and the environment. According to EPA about 86% of the national ammonia emissions are from miscellaneous sources that includes livestock and fertilizer. An inventory of aerial pollutant emissions emanating from USA poultry buildings estimated ammonia emissions to be 664 t/yr in 2002 and will rise to 720 t/yr in 2015.

Currently two major types of laying hen houses are being used in the USA, i.e. high-rise houses and manure-belt houses. For high-rise houses, solid manure is stored in the lower level of the building for about a year before removal. For manure-belt houses, manure drops onto a belt beneath cages and is removed from the house two to seven times a week. The major source of ammonia is the uric acid in bird feces which upon microbial decomposition results in ammonia. Since bird feces are the major source of ammonia in poultry facilities its treatment/management is vital to lower ammonia emissions.

Various methods have been used to control ammonia emissions from livestock facilities. Abatement of ammonia from livestock production by diet manipulation has been gaining popularity over the last few years. Studies have shown diet manipulation to be a useful tool to reduce air contaminants especially NH_3 emissions from livestock facilities. Various chemical and mineral amendments have been used to reduce ammonia emissions from animal manures. These chemicals include paraformaldehyde, zeolites like clinoptilolite, superphosphate (, phosphoric acid, ferrous sulphate, alum, acetic acid, propionic acid, and antibiotics. Though these chemical amendments seem to reduce ammonia emissions, they may add to pollution in some other form. For example, addition of phosphoric acid may enhance the P content of the litter whereas, iron in ferrous sulfate may be toxic to the birds though they both help abate ammonia emissions.

An alternative approach to decrease ammonia emissions would be to block the enzyme activity (urease) in the litter as ammonia is the byproduct of a five-step enzymatic degradation of uric acid. Urease inhibitors have been used as soil additive to control urease activity when urea-based fertilizers are applied. These inhibitors act by inhibiting the urease enzyme which converts urea to ammonia thus decreasing the loss of nitrogen as ammonia. Among the known inhibitors of urease, the most commonly used are phosphorodiamides and phosphorotriamides such as N-(n-butyl) thiophosphorictriamide (NBPT) and hydroxamic acid derivatives. More recent research on urease inhibitor has been focused on its ability to reduce ammonia emissions and conserve nitrogen in cattle feedlots. Laboratory studies have shown that phosphorodiamidate (urease inhibitor) when applied on weekly basis prevented up to 70% and 92% of the urea from being hydrolyzed after day 28. Both N-(n-butyl) thiophosphoric triamide and cyclohexyphosphoric triamide when applied to cattle feedlot pens successfully reduced ammonia emissions.

The objective of the research was to evaluate the effectiveness of commercially available urease inhibitor in reducing ammonia emissions from layer feces. Two experiments were conducted over a 21-day and 48 day period at the University of Kentucky Cold Stream Poultry Research Laboratory. The experiment was conducted in a caged layer room using 12 groups of cages. Each group of six cages housed twelve birds allowing 80 square inches per bird floor space. Ad libitum feed of a 16% CP Layer Mash and ad libitum water were provided. Ventilation was provided through ducts that delivered tempered external air to each of the individual cages by a 2000 CFM squirel cage blower. Exhaust was through a 36 inch sidewall fan at the end of the building with running speed based on room temperature inside the room. In the first study used, 5 kg broiler litter was placed in each of 12 containers measuring. Each container was placed below a group of six cages with 12 hens such that the manure from the birds dropped in the respective containers.

Six of the containers were treated with urease inhibitor, N-(n-butyl) thiophosphoric triamide (NBPT), acquired from Agrotain, at the rate of 0.036 ml/lb litter and the rest were retained as controls. The control containers were sprayed with de-ionized water at the same rate as urease inhibitor. The treatment and control containers were set up such that they alternate each other. Urease inhibitor was applied on day zero and reapplied after 7 days. In the second experiment the layer feces was allowed to accumulate in the containers for 10 day before applying the urease inhibitor. The urease inhibitor was applied at the start of the experiment and reapplied on day 31. Ammonia concentrations were measured every alternate day using an equilibrium chamber technique. The litter samples were collected once a week and analyzed for moisture, pH, total nitrogen, and total ammoniacal nitrogen.

The results indicated that there was a significant change in equilibrium ammonia concentration over time. In the first experiment with broiler litter as the base material, the effect of the urease inhibitor lasted for more than 14 days after urease reapplication. In the experiments with layer feces as the base material, there was a significantly lower ammonia concentration in the urease inhibitor treated containers and the effect lasted for one week. Conversely, after the second application the effect lasted for 14 days, thereafter the ammonia concentration in the urease treated feces started to build up. The nitrogen content of the litter also supports the ammonia concentration data. The study shows that urease inhibitor is effective in lowering ammonia emissions in layer houses.

Wet and Dry Deposition Studies at Remote, Agricultural and Urban Site of Semiarid Region of India

S.P. Singh[*1,2], P.Khare, K. Maharaj Kumari and S.S. Srivastava
[*1]Regional Research Institute, Nehru Garden, Kothrud, Pune, INDIA
[2]Dayalbagh Educational Institute (Deemed University), Dayalbagh Agra, INDIA
*Email: mtrpal@yahoo.com

Abstract

The wet and dry deposition studies at Remote, Agricultural and Urban site (Agra) in a semiarid region of India were carried out. Major anions viz. F^-, Cl^-, NO_3^-, SO_4^{2-} were analyzed using Ion Chromatograph (Dionex DX-500) and major cations viz Na^+, K^+, Ca^{2+}, Mg^{2+} were analyzed using Atomic Absorption Spectrophotometer (Perkin Elmer 2380) in both wet and dry deposition samples. NH_4^+ was analyzed colorimetrically using indophenol blue method. The rainwater pH at remote and urban sites was 6.6. However at agricultural site pH of rainwater was higher than the other two sites. Level of Cl^-, NO_3^-, NH_4^+, Ca_2^+ and Mg_2^+ were higher at agricultural site, while SO_4^{2-} was dominated at urban site. At remote site levels of ionic species were lower than the other sites. Analysis of sequential rainwater samples indicates that neutralization by NH_4^+ is an in cloud process, while that by Ca^{2+} and Mg^{2+} is below cloud process. NH_3 is known to exit as NH_4Cl, NH_4NO_3 and $(NH_4)_2SO_4$. These products may be formed in the atmospheric water droplets by scavenging of aerosols and reaction of gaseous species. Similarly, in dry deposition samples elevated levels of NO_3^-, SO_4^{2-}, NH_4^+, Ca_2^+ and Mg^{2+} were found at agricultural site. The high deposition rates at agricultural site may be attributed to local inputs while, lower deposition rates of major ions at urban site may be due to fine mode aerosols from anthropogenic sources. Wind rose study suggests minimal influence of long-range transport at remote site. Annual dry deposition rates are higher than the annual wet deposition rates. Wet and dry deposition data have been used to calculate the critical load of S and N for this region using a mathematical model. The present load of S and N are much lower than critical loads of S and N for this region. Our value of critical load agrees with the value of acidity assessed by RAINS Asia model for this area.

Key words: Wet and dry deposition, remote, agricultural and urban site.

Introduction

Deposition both, wet and dry are final steps in removal of trace constituents. The efficiency of wet deposition is controlled exclusively by atmospheric processes whereas the rate of dry deposition is mainly controlled by surface onto which the constituents are deposited. High concentration of total suspended particles (TSP) seems to play an important role in the chemistry of precipitation in India. This is due to thin forest cover and prevalence of turbulence and dry conditions for prolonged periods resulting in suspension of soil dust.

Industrialization has resulted in rapid increase in SO_2 and NO_x concentrations in the atmosphere in India. Considering the fact that coal contains 5% sulfur the total production of SO_2 from coal has increased from 0.36 mt in 1970 to 1.35 mt in 1994 (TERI, 1996). Acid rain usually described as European and North American problem is spreading to the developing countries of the world. Due to high economic growth emissions of SO_2 in Asia are set to rapidly increase from 34Mt in 1990 to about 110 Mt by 2020 if no control measures are taken. Recently considerable interest in acid deposition in India has been demonstrated by a number of short term research studies (Khemani et al., 1987, 1989; Mahadevan et al., 1984). Air pollution levels are steadily rising in metropolitan cities like Calcutta, Delhi and Bombay. Acid rain has already been reported since 1974 at a Chembur a major industrialized area in Bombay city with a large point source of SO_2 and at isolated pockets in Delhi and recently at Korba. The present communication describes wet and dry deposition study at a remote, agricultural and urban location of north-central India to assess the contribution of anthropogenic, agricultural and natural activities.

Experimental

The site description and analytical methodology are given elsewhere (Singh et al., 2001; Satsangi et al., 1998; Saxena et al., 1996).

Results and Discussion

Wet Deposition

In India tropical climatic conditions and predominantly alkaline rich soils have a neutralizing effect. The Thar desert in the north west and the sea and soil derived aerosols in the coastal areas are responsible for maintaining the pH in the alkaline range in the most parts of India. Higher temperature and sunlight increase the efficiency of atmospheric chemical reactions, particularly those transforming SO_2 and NO_x to acidic sulphates and nitrates. However, the background trends of pH in precipitation have been decreasing all over India (Mohan and Kumar, 1998).

The reference level commonly used to compare acidic precipitation with natural precipitation is pH 5.6, the pH that results from equilibrium of atmospheric CO_2 with precipitation. The average pH values at Rampur, Gopalpur and Agra are 6.6, 6.9 and 6.6,respectively.The pH values are by and large alkaline throughout India. The pH values are high (pH>7) at Srinagar, Allhabad, Ahmedabad, Jodhpur, Amritsar etc. The pH values are comparatively low, but still in alkaline range (pH 6-7) in industrial towns and cities ie at Nagpur and Visakhapatnam, Delhi, Agra, Bhopal etc. However, at places like Trivandram, Tuticorn and Nilgiri pH is still lower 5.7, 5.7 and 5.3 respectively. At Chembur, a highly industrialized location in Bombay city and Korba (MP) the pH is acidic; 4.5 and 4.8, respectively.

The wind blown dust particles representing soil characteristics reach the atmosphere and react with the falling raindrops whereby its soluble components dissolve, therein influence its pH value and the impact of nature of soil towards rain acidification is profound (Varma, 1989). It has been seen that in regions with pH >7, soil is alkaline in nature for e.g. Jodhpur etc. and acidic pH of soil in the regions with pH less than 6.

Chemical Composition of Rainwater in India

Chemical composition of rainwater is an integral measurement and helps to understand the relative importance of different sources. Figure 1 shows the chemical composition of rainwater for three different sites of north central India. Major water soluble anions and cations ranged between 11 to 56, 2.5 to 134 and 7.7 to 84 µeq l-1 at Rampur, Gopalpura and Agra. Higher levels of anions at Agra may be attributed to the anthropogenic activities while elevated levels of NO_3, Cl and cations at Gopalpura may be due to contribution of local agricultural activities. Comparatively lower levels of cations and approximately comparable levels of anions at Rampur may be due to local sources. Wind rose study suggest minimal influence of long-range transport at remote site.

The neutralization factors were calculated to evaluate the relative neutralization of rainwater by crustal components and ammonia. Neutralization factor were calculated as.

$$NF_{NH4} = \frac{[NH_4]}{2\,[SO_4]+[NO_3]}$$

$$NF_{Ca} = \frac{[Ca]}{[SO_4]+2\,[NO_3]}$$

$$NF_{Mg} = \frac{[Mg]}{[SO_4]+2[NO_3]}$$

The NF $_{NH4}$ was less than both NF$_{Ca}$ and NF $_{Mg}$ the factors being 0.56 for NH_4, 1.1 for Mg and 1.2 for Ca, revealing the crustal components neutralize a large fraction of available acid. Sequential rainwater analysis also suggest that neutralization by ammonia is a in-cloud process while neutralization by crustal component is a below cloud process.

Ionic Correlation and Factor Matrix

Correlation matrix is a common way of hypothesing potential precursors of ions in rainwater samples. Correlation between ions suggests the likely sources of pollutants and also indicates the gaseous reactions occurring in the atmosphere.

At all the three sites NO_3 and SO_4 showed good correlation in rainwater samples indicating their origin from similar sources. It is possible that a fraction of NO_3 and SO_4 may be derived from soil as a primary pollutant or may be associated with Ca, Mg and Na after the neutralization process. Ca, Mg and Na were found to be significantly correlated with SO_4 and NO_3. Ca, Mg which are essential components of soil showed a good correlation with each other implying a common origin. All these features indicate that wind carried dust and soil play a significant role in precipitation chemistry (Saxena et al., 1996; Khemani et al., 1985). Significant correlation of NH_4 with Ca, Mg, Na and K reflects the influence of agricultural activities.

Factor analysis was carried out by the principle component method. Initial factors were extracted from a matrix of correlations derived from standardized variables. Factors with eigen values greater than one were considered for varimax rotation to obtained the final factor matrix. The analysis identified four factors that contributed 85% of the variance to the data set. Factor analysis assumes that the Interco-relations among the original variables are generated by some smaller number of unobserved factors. In accordance with this assumption the grouping of elements/ions in each factor could be attributed to chemical, meteorological, physical reasons as well as to common sources.

Factor analysis study at Agra showed that Ca, SO_4, NO_3, HCO_3, Cl and Mg were grouped in one factor, which is considered as an indicative of soil factor. NH_4 and K grouped in a separate factor suggest the contribution of combustion activities. However, NH_3 emissions from decomposition of animal wastes and manure in the adjoining agricultural fields cannot be discounted. While last factor containing F was attributed to the process of lime pulverization in the vicinity of the sampling site.

Dry Deposition

Figure 2 shows dry deposition flux of water soluble ions at different locations. The dry deposition rates of soil derived cations Ca, Na, K, Mg ranged between 1.0 to 4.4, 1.0to 5.5 and 0.4 to 5.0 mg m^{-2} d^{-1}. The higher deposition rates of ionic species at Gopalpura, an agricultural site may be attributed to local input, while comparatively lower levels at Agra my be due to contribution of anthropogenic sources and existence of particle in fine mode. The minimum levels of anions and cations at Rampur indicate contribution from natural sources. Dry deposition of NO_3 and SO_4 are also of similar magnitude suggesting that they are also soil derived or associated with the soil elements. Annual wet and dry deposition rates were calculated considering annual rain depth, as 760mm. Annual dry deposition rates are higher than annual wet deposition rates, which implies that dry deposition is major phenomenon in this region.

Seasonal Variation in Dry Deposition Rates

The seasonally averaged dry deposition rates are shown in Figure 3. Dry deposition rates for all constituents are maximum during the winter and minimum during the monsoon season. The low monsoon fluxes are due to frequent rain showers, washout and also low emission fluxes from soil which settles during this period. In summer due to high temperature and strong winds, the atmosphere is unstable and turbulent and leads to a maximum deposition of pollutants. On the contrary in winter temperatures are much lower and calm, stable conditions prevail promoting stagnation of pollutants, which is further enhanced by frequent temperature inversions. Moreover, the persistence of humid and foggy conditions during winter would lead to maximum deposition rates and checking of re-suspension due to moistening of collection surface. These factors account for greater deposition during winter.

Critical Load (CL)

Critical load is defined as a quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge. If an ecosystem has a limited natural capacity to absorb or neutralize pollutants, then the critical load for that ecosystem would be low. Areas which are more able to withstand pollutants deposition have correspondingly higher critical loads (Hettelingh. 1992). An attempt is being made to present critical load of north-central India using SMB method. The calculation was performed using the method of Hettelingh et al.(1991). For details of calculation refer to our earlier paper (Satsangi et al., 1998). It was found that the present loads of S (2569.6 g ha^{-1} $year^{-1}$) and N (219.7 g ha^{-1} $year^{-1}$) are much lower than the critical loads of S and N (4947.2 and 16.91.4 g ha^{-1} $year^{-1}$, respectively) in the soil with respect to Anjan grass. Thus at present there are no chance of acidification of this region of India.

References

Khemani, L.T.; Momin, G.A., Naik, M.S., Rao, P.S.P.; Kumar, R., and Ramana, Murthy, Bh.V. (1985). Impact of alkaline particulates on the pH of rainwater in India. Water, Air and Soil Pollut. 24, 365-376.

Khemani, L.T.; Momin, G.A.; et al. (1987) Influence of alkaline particulates on pH of cloud and rain water in India. Atmospheric Environment 21, 1137-1145.

Khemani, L.T.; Momin, G.A., Rao, P.S.P.; Safai, P.D., Singh, G. and Kapoor, R.K.(1989) Spread of acid rain over in India. Atmospheric Environment 23 (4) 757-7620.

Hettelingh, J. P.; Downing R. J., de Smet PAM.(1991). Mapping critical loads for Europe. Coordination Centre for effects, Bilthoven, The Netherlands, Technical report No. 1, RIVM REPORT No. 25:910 1001.

Hettelingh, J. P.; Downing R. J., de Smet PAM. (1992). The critical loads concept for the cintrol of acidification. In: Schneider T, editor. Acidification research , evaluation and policy applications. Elsevier Science Publishers B. V.

Manju Mohan and Sanjay Kumar (1998). Review of acid rain potential in India: Future threats and remedial measures. Current Science, 75(6), 25th Sept. 1998.

Mahadevan, T.N., Meenakshy, V., and Sathe, A.P. (1984). Some observations on the chemical composition of precipitation in industrial area its use in air quality assessment. Mausam, 35, 84-90.

Saxena, A.; Kulshrestha, U.C.; Kumar, N.; Kumari, K. M.; and Srivastava, S. S. (1996) Charactrization of precipitation at Agra. Atmos. Environ. 30(20), 3405-3412.

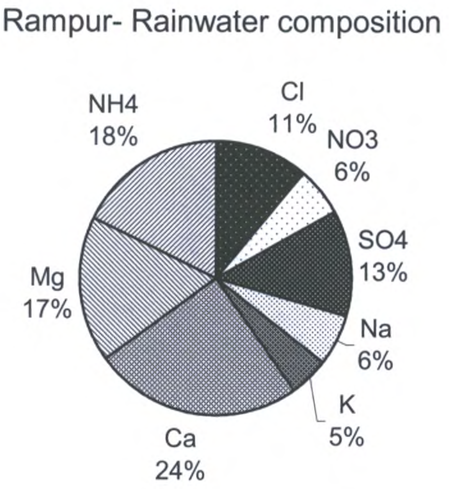
Satsangi, G. S.; Lakhani, A.; Khare, P.; Singh, S. P.; Kumari, K.M. and Srivastava, S.S. (1998). Composition of rainwater at a semi-arid rural site. Atmos. Environ., 32, 3783-3793.

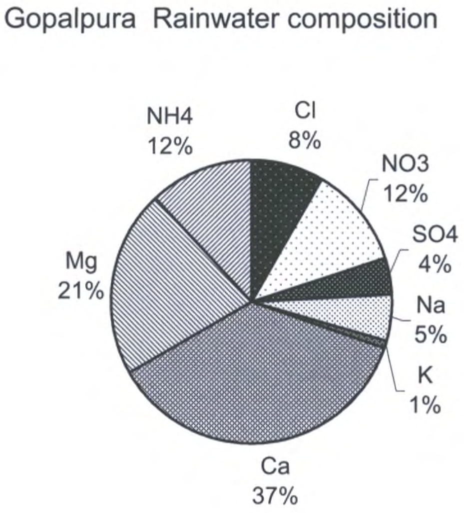
Satsangi, G. S.; Taneja, A.; Khare, P.; Singh, S. P.; Lakhani, A.; Kumari, K.M. and Srivastava, S.S. (1998). Deriving critical loads for the Agra region in India. The Science of the Total Environment, 222, 119-122.

Singh, S. P.; Khare, P., Satsangi, G. S., Lakhani, A.; Kumari, K.M. and Srivastava, S.S. (2001). Rainwater composition at a regional representative site in a semi-arid region of India. Water, Air and Soil Pollution. 127, 93-108.

TERI Energy Data Directory and yearbook 1996/97(TEDDY). New Delhi: Published by Tata Energy Research Institute, 1996.

Varma, G. S.(1989) Impact of soil derived aerosols on precipitation acidity in India. Atmospheric Environment 23, 2723-2728.





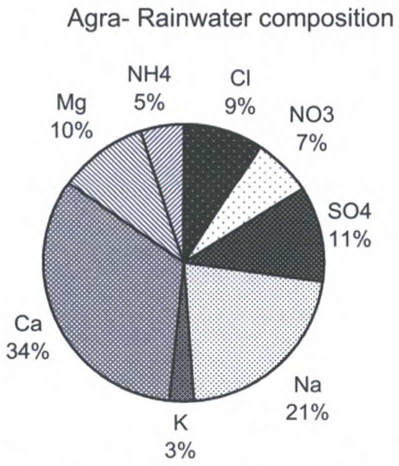


Figure 1. Chemical composition of rainwater at different sites.

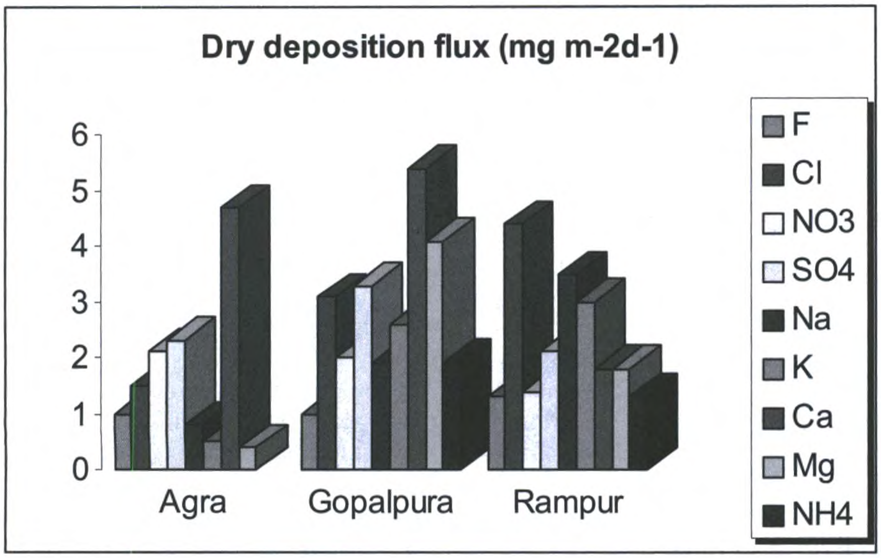


Figure 2. Dry deposition flux at different sites.

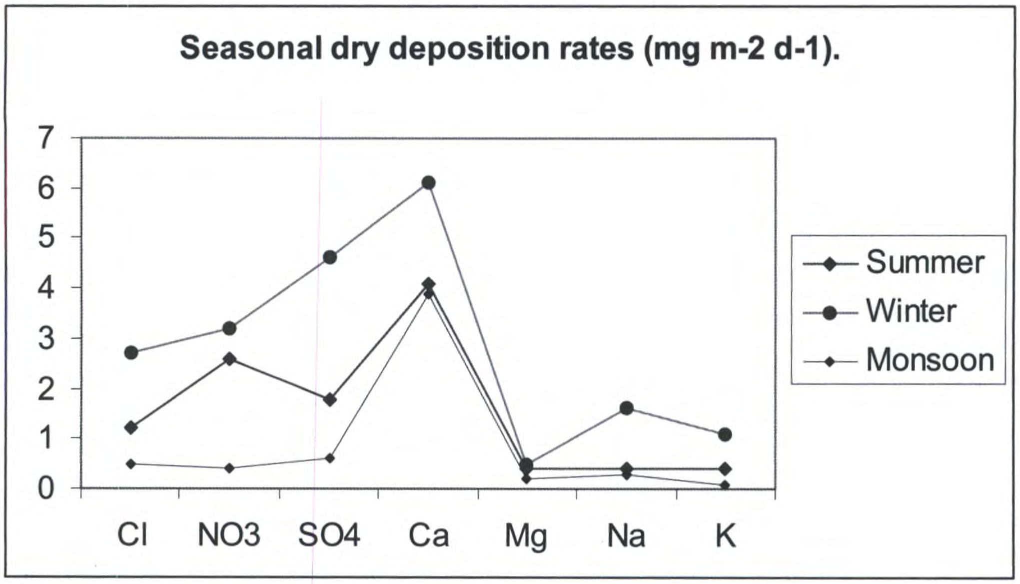


Figure 3. Seasonal Variation of dry deposition flux at different sites.

Current Issue State on Assessment of Chemical Composition of Precipitation and Its Impact on Regional Atmosphere Condition

Tatyana Smirnova and Galina Tolkacheva
Hydrometeorological Research Institute (NIGMI), Tashkent, Uzbekistan, 700052

Abstract

The scientific-technical revolution in XX century along with its tremendous positive effects on human beings has exerted a negative contribution in environment pollution, including atmosphere air pollution. As a result of this the atmosphere pollution with anthropogenic origin aerosols has increased. The Central Asian region is a region with unique combination of natural-climatic zones with sharp-continental climate, loess soil cover, plenty of solar days. These factors set conditions for intensive getting natural dust aerosol into the atmosphere. At present study of chemical composition of precipitation is a very actual issue for environmental pollution monitoring. The problems of revealing of pollution source, pollution degree change with time and chemical composition of pollutants ate worth a special attention. To the main anthropogenic air pollutants can be referred sulfur, nitrogen, carbon oxides and firm particles of diverse composition and origin. Heat and power, building, mining and chemical enterprises, transport and others are principal sources of air pollution on the territory of Tashkent region. As subjects of inquiry were chosen the points characterizing diverse nature-climatic zones including: soil-desert source of aerosol emission (on example of Muynak meteorological station); anthropogenic zone (on example of "Chatkal reserve" station of complex background monitoring). Chemical composition of precipitation has been investigated by methods of ion chromatography, atomic adsorptions, photocolorimetry and some others. It is necessary to stress that chemical composition of precipitation reflects impact of local sources of anthropogenic aerosol emission and can serve as an indicator of regional and global atmosphere pollution. Mineralization trends of precipitation have revealed common tendency of its increasing and as corollary, increasing of atmosphere pollution in the region with as natural as anthropogenic sources. The objective of the investigations is assessment and possible change of chemical composition of atmospheric precipitation from local sources on principal components for last 15 years.

Influence of Agricultural Ammonia Emissions on Particulate Matter in Switzerland

Christoph Spirig and Albrecht Neftel
Agroscope, Swiss Federal Research Institute for Agroecology and Agriculture, Zurich

Abstract

Exceeding levels of particulate matter is a major problem in Switzerland. Agriculture contributes to aerosols both with primary emissions and as an emitter of precursors for secondary aerosol formation. The latter contribution is highly uncertain as the relationship between precursor emissions and secondary aerosols is complex and non-linear.

We investigated the sensitivity of particulate matter to changes in agricultural ammonia emissions based on measurements of the Swiss national air pollution monitoring network. Process based and statistical models both predict that particulate matter in Switzerland can only moderately be influenced by ammonia emission controls. The reduction potential in livestock farming activities needs to be fully exploited to cause noticeable changes in particulate matter.

Introduction

Particulate matter (PM) concentrations commonly exceed the health standards in large parts of Europe. Significant fractions of PM are of secondary origin, such as sulfate, nitrate, ammonium and organic compounds. Cost-effective measures to reduce secondary aerosol depend on the knowledge of the complex relationships between PM and the emissions of gaseous precursors like ammonia and nitrogen oxides.

In Switzerland, more than 90% of the total ammonia emissions are caused by agriculture (Menzi et al., 1997). Several studies have shown that technical and operational measures in livestock farming have the potential to reduce these emissions significantly (Webb et al., 2005). The goal of this work is to investigate the relationship between the precursor ammonia and secondary aerosol and to predict the effect of realistic ammonia reductions on PM levels. As a complement to 3D-chemical transport models that are typically used for such tasks, we apply heavily observation based analyses here.

Methods

Two fundamentally different approaches were used to analyze the sensitivity of PM concentrations on changes of ammonia concentrations. The first category consists of process based models; the second category comprises statistical models in the broader sense, namely artificial neural networks. Both models were applied to describe the precursor-PM relationship based on measurements of the Swiss national air pollution network. At a station representing typical rural conditions, measurements of total ammonium (ammonia and particulate nitrate, TA), total nitrate (nitric acid and aerosol nitrate, TN) and particulate sulfate are available on a daily basis covering several years.

Process Based Modeling

The thermodynamic equilibrium model ISORROPIA (Nenes et al., 1998) was used to determine gas/particle partitioning of TA and TN. The applicability of the model was tested on several data sets from intensive observation periods, where measurements of both gaseous and particulate components of TA and TN were performed. The comparison of modeled and measured gas fractions of TA and TN showed that ISORROPIA successfully simulated gas-aerosol partitioning of these compounds and demonstrated that the assumption of equilibrium between gas and aerosol phases was reasonable. ISORROPIA was then used to calculate the gas-aerosol partitioning of inorganic compounds for the data set of the monitoring station Payerne, where measurements of total ammonium, nitrate and sulfate are available for several years. The sensitivity of the aerosol concentration on changes in ammonia was calculated by modifying the TA concentration and leaving all other inputs (essentially TN, sulfate, temperature, humidity) constant.

In order to assess how strongly the TA concentration can be influenced by changes in ammonia emissions, additional simulations with a one dimensional photochemical model (Spirig et al., 2002; Staffelbach et al., 1997) were performed. The model was run for summer and winter conditions with emissions representative of the Swiss Plateau as a base scenario. The model runs lasted 4 days, 75% to 25% of the base case emissions were chosen as reduction scenarios. The mean reduction in TA on the second day of the simulation was considered for the decrease of TA that can be achieved with reductions of ammonia emissions within Switzerland. Finally, the results of ISORROPIA and 1-D model were combined giving the sensitivity of aerosol concentrations on changes in ammonia emissions.

Artificial Neural Networks

As a contrast to the process based approach, neural networks were used to achieve the same task. The rationale behind this practice is the assumption that the aerosol concentration can be formulated as a function of independent variables:

$$PM10 = f(NO_x, SO_2, TA, TN, SO_4, T, H).$$

Neural networks of the type multilayer perceptron were proven to have the capability of approximating any smooth measurable function (Hornik et al., 1989). We therefore attempted to approximate f with such a multilayer perceptron network (one hidden layer with 15 neurons). The network was trained with a back propagation algorithm (including an early stopping method). For the network training, data from the monitoring network were fed to the network in random order, whereas roughly half of the measurements were used for determining the network weights and the rest was used for testing. In contrast to the process based model described above, the trained neural network can be considered as a purely empirical model. As with the process based model, sensitivities of the trained network output were determined by modification of the independent variable total ammonium.

Results

Thermodynamic Equilibrium Model

The decrease of inorganic aerosol mass as a consequence of a 10%-reduction in total ammonia (TA) for the Swiss monitoring network station Payerne is presented in Figure 1. Relative sensitivity is defined as percentage decrease of inorganic aerosol per percentage decrease of TA. The changes in aerosol mass are mostly under-proportional, but single cases with high sensitivity were found. These were mostly situations where water content was rather sensitive to TA changes. In this 10% reduction scenario, sensitivities during summer seasons are generally higher than those in winter. This behavior is compatible with the behavior of the ammonium nitrate equilibrium in an ammonia-saturated regime (Ansari and Pandis, 1998). This ammonia-saturated situation was confirmed by looking at the free ammonium (=moles of TA minus 2*moles of sulfate), which was positive throughout the whole data set.

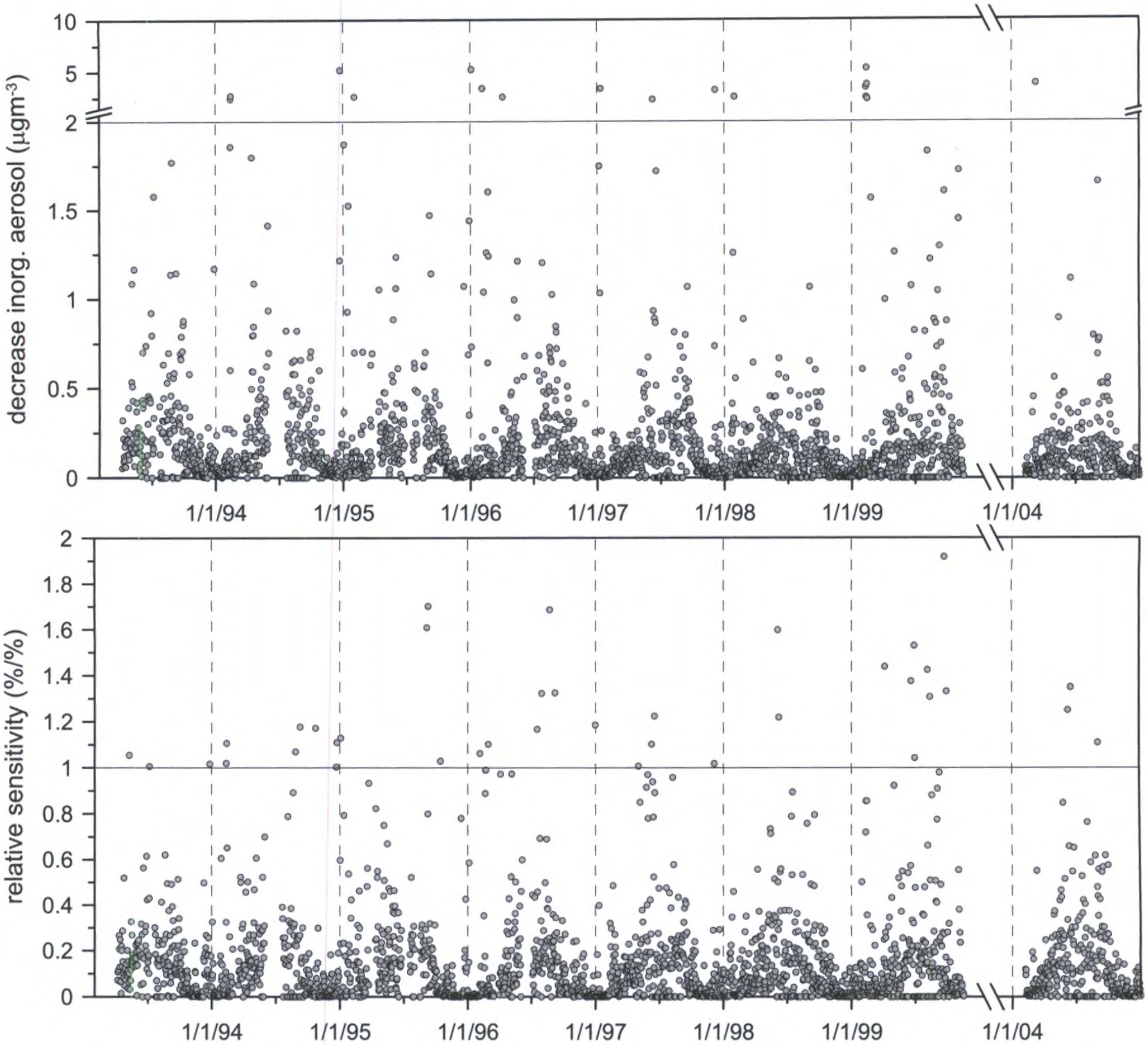


Figure 1. Decrease of inorganic aerosol mass after 10% reduction of total ammonium; absolute decrease, above, and relative sensitivity, below.

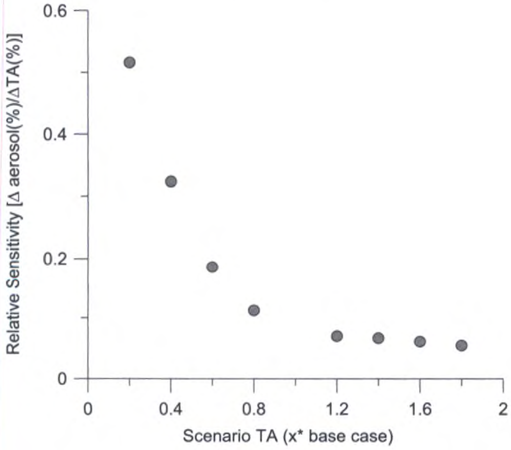


Figure 2. Relative sensitivities of inorganic aerosol mass towards reductions (0.2 ≤ x ≤0.8) and increases (1.2 ≤ x ≤1.8) of total ammonium.

The relative sensitivities increased when applying higher reductions. Figure 2 summarizes the mean relative sensitivities for various reduction/increase scenarios. The increase in sensitivity with stronger reductions is consistent with a regime becoming less ammonia-saturated.

Artificial Neural Network

Figure 3 shows the comparison between measured PM10 and neural network output. The trained network is obviously capable of simulating PM10 as a function of NO_x, SO_2, temperature, humidity, TA, TN, and sulfate.

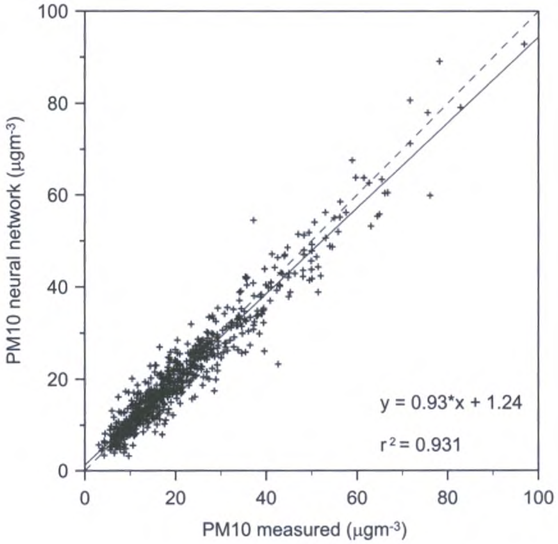


Figure 3. Comparison of measured PM10 with neural network output.

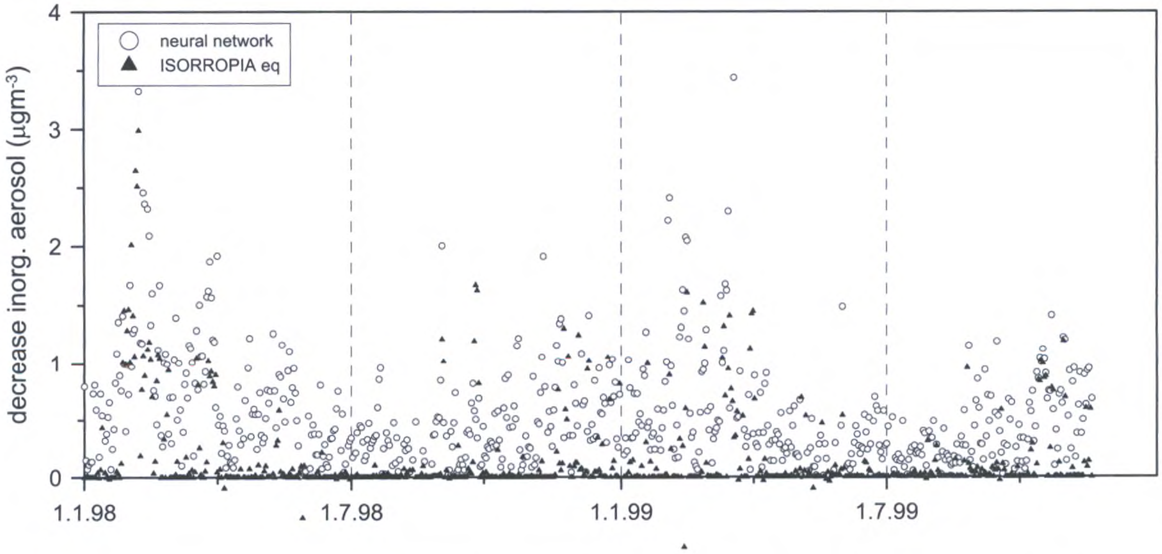


Figure 4. Decrease in inorganic aerosol mass after 10% reduction of TA as predicted by the neural network. ISORROPIA calculations shown here were performed under identical conditions as in PM10 measurements, i.e. equilibrated at 50% rel. humidity and 295 K.

The predicted decrease in PM10 by the network as a consequence of a 10% reduction in TA is presented in Figure 4. The neural network produced a different picture than the process based model (Figure 2), with higher sensitivities in winter. This discrepancy can mostly be explained by the reference conditions that are used for the PM10 measurement: Prior to quantification, the sample filters were equilibrated at 295K and 50% humidity. PM 10 measurements therefore represent an aerosol at these conditions while ISORROPIA calculated the composition at ambient conditions. If ISORROPIA calculations were performed at the PM10 standard conditions (i.e. 295 K and 50% humidity), the aerosol decreases behaved similarly to those of the neural network output (Figure 4). In addition, the decreases in aerosol mass became significantly smaller when applying these reference conditions. The neural network also predicted increasing sensitivities with increasing reductions.

Emission Reduction Scenarios

The change of total ammonia and total nitrate concentrations after reductions in ammonia and NOx emissions was calculated with the 1-D photochemical model. Table 1 summarizes the change in TA concentrations on the 2nd day of the model run for three reduction scenarios 25-75%. The sensitivity of the inorganic aerosol mass to NH_3 emission reductions was obtained by combining these 1D model results with the sensitivities of inorganic aerosol mass to changes in total ammonia. The qualitative behavior of the inorganic aerosol mass to changes in ammonia emissions is similar as the one found for the relationship between aerosol mass and total ammonia: the sensitivity is higher with increasing reductions. Small reductions in ammonia emissions will only cause negligible decreases of particulate matter.

Table 1. Results of 1D-model; reduction of total ammonium and inorganic aerosol after NH_3 emission changes.

Reduction NH_3 emissions	Reduction total ammonium	Reduction inorganic aerosol summer / winter
25 %	20 %	4% / 3%
50 %	39 %	10% / 8%
75 %	60 %	18% / 21%

Discussion and Conclusions

Both process based models and neuronal networks predicted that inorganic aerosol mass will react in a rather under-proportional manner to changes in total ammonia. It is a consequence of the largely ammonia-saturated regime in Switzerland. The qualitative agreement of the two fundamentally different model categories supports the plausibility of these findings.

Significant reductions of ammonia emissions are needed to achieve noticeable changes in particulate matter. Furthermore, the comparison of the aerosol compositions calculated at ambient and PM10-standard conditions (295K, 50% rel. humidity) revealed that changes in measured PM 10 will even be smaller.

Technical and operational measures in livestock farming have the potential to reduce ammonia emissions in Switzerland by about 30%. Given the sensitivities of the aerosol mass found here, the potential for reductions needs to be fully exploited to cause measurable decreases in PM 10.

On the other hand, ammonia reductions in this NH_3-saturated regime will cause nearly proportional decreases in gas phase ammonia. NH_3-emission controls thus offer a greater handle in reducing nitrogen deposition than particulate matter.

References

Ansari, A. S., and S. N. Pandis. 1998. Response of inorganic PM to precursor concentrations. *Environ. Sci. Technol.*, 32, 2706-2714.

Hornik, K., M. Stinchcombe, and H.White. 1989. Multilayer feedforward networks are universal approximators. *Neural Networks*, 2, 359-366.

Menzi, H., R. Frick, and R. Kaufmann. 1997. Ammoniak-Emissionen in der Schweiz: Ausmass und technische Beurteilung des Reduktionspotentials. *Schriftenreihe der FAL*, 26.

Nenes, A., S. N. Pandis, and C. Pilinis. 1998. ISORROPIA: A new thermodynamic equilibrium model for multiphase multicomponent inorganic aerosols. *Aquat. Geochem.*, 4, 123-152.

Spirig, C., A. Neftel, L. I. Kleinman, and J. Hjorth. 2002. NOx versus VOC limitation of O3 production in the Po valley: Local and integrated view based on observations. *J. Geophys. Res.*, 107, 8191, 10.1029/2001JD000561.

Staffelbach, T., A. Neftel, and L. W. Horowitz. 1997. Photochemical oxidant formation over southern Switzerland.2. Model results. *J. Geophys. Res.*, 102, 23363-23373.

Webb, J., H. Menzi, B.F. Pain, T. H. Misselbrook, U. Dammgen, H. Hendriks, and H. Dohler. 2005. Managing ammonia emissions from livestock production in Europe. *Environ. Pollut.*, 135, 399-406.

The Dispersion of the Ammonia Plume from a Beef Cattle Feedlot: Airborne and Ground-Level Measurements

Ralf M. Staebler[1], Shao-Meng Li[1], Cris Mihele[1], and Sean McGinn[2].
[1]Environment Canada, Toronto, ON, M3H 5T4, Canada; [2]Agriculture and Agri-Food Canada, Lethbridge, AB, T1J 4B1, Canada.

Abstract

Ammonia contributes significantly to the formation of particulate matter that has a detrimental effect on human respiratory and cardiovascular health. In Canada, approximately 45% of NH_3 emissions come from the cattle industry (based on data from 1995). As part of an effort to establish national emission inventories and to improve air quality forecasting capabilities, a study was recently conducted in Southern Alberta to investigate the emissions of a mid-size feedlot (20,000 head). A Cessna 207 equipped with fast NH_3 and NO_Y instrumentation as well as an aerosol mass spectrometer (AMS) was flown in a dense grid pattern (spacing between tracks ~ 200 m) covering an area of 5 by 10 km containing the 800 by 800 m feedlot, at several altitudes ranging from 30 to 300 m above ground. On the ground, this was supported by a more extensive suite of instruments on a mobile lab, as well as stationary measurements of long-path integrated NH_3 concentrations and stability parameters downwind of the feedlot. Three flights were conducted in late September 2005 under varying meteorological conditions, ranging from very calm and stagnant to strong westerly flow with near-neutral stratification. NH_3 mixing ratios up to 50 ppb were encountered, on the calm day up to 300 m above ground. The AMS showed some evidence of local nitrate formation. The data collected provide an unprecedented opportunity to validate and improve a Lagrangian dispersion model (WindTrax) used to predict emission rates from the concentration and turbulence measurements downwind of the source. The end goal of this project is two-fold: to provide better estimates of the local net emission rate, and to improve the characterization of the sub-grid-scale dispersion and transformation of NH_3 to account for these nonlinear processes in the national air quality models.

Preliminary Evaluation for Ecosystem Acidification due to Agricultural Activities Dynamical Modeling Approach

Iveta Steinberga and Lita Lizuma
Latvian Hydrometeorological Agency, Riga, LV1019, Latvia

Abstract

Soil acidification, caused by inputs of acidifying compounds, is characterized by a wide variety of long-term effects. The emissions of ammonia and nitrogen oxides have strongly increased in Europe in the second half of the 20th century. Ammonia is mainly volatized from intensive agricultural systems. Atmospheric nitrogen deposition rates from 20 - 100 kg ha^{-1} yr^{-1}. Depending on types of activities in country it is possible to separate and evaluate agricultural impact in ecosystem acidification processes. The Very Simple Dynamic model has been used for dynamic modeling, consisting of a set of mass balance equations describing the soil input and output data relationships and fluxes, and soil properties. In the short term, dynamic modeling of soil acidification can contribute to a better understanding of time delays of recovery in regions where critical loads are no longer exceeded and time delays of damage in regions where critical loads continue to be exceeded.

Reduction of Ammonia Emissions from Swine Lagoons Using Alternative Wastewater Treatment Technologies

A. A. Szögi and M.B. Vanotti

USDA-ARS Coastal Plains Soil, Water, and Plant Research Center, 2611 W. Lucas Street,
Florence, SC 29501

Abstract

There is a need for treatment technologies that can effectively address environmental concerns associated with ammonia (NH_3) emissions from anaerobic lagoons, typically used to manage manure. To meet this need, we conducted a study to determine the effects of water quality improvement in swine lagoons on NH_3 emission rates using alternative wastewater treatments. This determination was done in three contiguous swine production units that had similar animal production management and lagoons with similar surface area (about 0.9 ha each), but their waste management was substantially different. In the first production unit, a full-scale wastewater treatment plant produced a clean effluent that in turn converted the old lagoon into a water storage pond. In the second production unit, the traditional anaerobic lagoon treatment method was maintained as a control. In the third production unit, raw flushed manure was treated through a solid-liquid separation system before anaerobic lagoon storage. Passive flux samplers were used to measure simultaneously the NH_3 gas fluxes from the lagoons receiving treated water and the traditional anaerobic lagoon. Ammonia emissions from the traditional anaerobic lagoon (control) totaled 12,540 kg N/lagoon/year (13,633 kg N/ha/year). This result compares to lower NH_3 emissions of 3,355 kg N/lagoon/yr (or 3,647 kg N/ha/yr) from the anaerobic lagoon with solid-liquid separation and 1,210 kg N/lagoon/yr (or 1,311 kg N/ha/yr) from the converted lagoon. In the anaerobic lagoon with solid-liquid separation, total annual NH_3 emissions were reduced by 73% with respect to those of the traditional lagoon. In the converted lagoon, remarkable water quality improvements such as lower N concentrations substantially reduced annual NH_3 emissions by 90% with respect to those found in the traditional anaerobic lagoon. These results overall demonstrate that alternative new wastewater technologies can substantially reduce ammonia emissions from confined swine production.

Introduction

Anaerobic lagoons are widely used to treat and store liquid manure from confined swine production operations. During lagoon storage and treatment, urea and other organic nitrogen (N) compounds contained in liquid manure are converted into ammoniacal N that can be released as ammonia gas (NH_3) into the atmosphere. Increase of NH_3 emissions due to intensification of animal production on a local scale has been related to increasing atmospheric NH_3 deposition and air pollution in North Carolina's Coastal Plain region (Battye et al., 2003; Walker et al., 2000). Because of these environmental concerns, there is major interest in new swine manure treatment technologies that can reduce environmental problems associated with NH_3 emissions from anaerobic lagoons (Williams, 2001).

Ammonia emissions from traditional anaerobic swine lagoons depend on several factors, such as NH_3-N concentration, pH, temperature, wind speed, chemical and microbiological activities, and material transport processes (Arogo et al., 2003). In particular, NH_3 emissions from anaerobic swine lagoons have been shown to increase with NH_3-N concentrations and temperatures (Harper et al., 2004). Therefore, it appears obvious that lower N levels in lagoons will substantially reduce NH_3 emissions compared with a traditional anaerobic lagoon. The purpose of this research was to quantify the magnitude of NH_3 emissions reduction in two lagoons that received treated water from two new alternative wastewater treatment systems. One treatment system combined solid-liquid separation with removal of N and phosphorus (P) from the liquid phase (Vanotti et al., 2005) and the other wastewater treatment system consisted of solid-liquid separation treatment before anaerobic lagoon storage. Emissions from the two lagoons receiving treated water were compared with NH_3 emissions from a traditional treatment anaerobic lagoon; all three lagoons had similar construction design and animal production management.

Materials and Methods

The study was conducted on Goshen Ridge Farm near Mount Olive, Duplin County, North Carolina. The farm had three finishing units under identical animal production and waste treatment managements. Each unit had six barns with 4,360-head finishing pigs and a traditional anaerobic lagoon for treatment and storage of manure. Manure was collected in barns using slatted floors and a pit-recharge system typical of many farms in North Carolina. In each production unit, pits were drained weekly by gravity to the traditional anaerobic lagoons. Lagoon effluent was then used to recharge the pits of each production unit. Lagoon dimensions and monthly average live animal weight (LAW) computed from farm production records are presented in table 1. The relationship between N production by pigs and their weight was 0.3 kg N/1000 kg LAW/day (Szogi et al., 2006).

Table 1. Main characteristics of the three production units.

Production Unit	Lagoon Surface ha	Lagoon Volume m^3	Steady State Live Animal Weight kg
1	0.90	24,145	224,581
2	0.92	22,356	196,636
3	0.93	22,236	229,425

In unit 1, a full-scale wastewater treatment system was started in 2003 to treat all raw manure produced. The treatment system combined solid-liquid separation with removal of N and P from the liquid phase. The system treated raw manure flushed from the barns in three steps (Vanotti et al., 2006). The first step flocculated solids from raw flushed manure using polyacrylamide. This step produced separated solids that were transported off-site and converted to organic plant fertilizer, soil amendments, or energy at a centralized facility. In the second step, N management to reduce NH_3 emissions was accomplished by passing the liquid through a module where immobilized nitrifying bacteria transformed NH_3 into nitrate. Subsequent alkaline treatment of the wastewater in a P module precipitated calcium phosphate and killed pathogens. The treated water was recycled to refill the barn pit recharge system, and excess water was stored in the lagoon and later used for crop irrigation. As the treatment system recovered the manure solids and replaced the anaerobic lagoon liquid with cleaner water, it converted the anaerobic lagoon in unit 1 into a treated water pond (hereafter called lagoon 1 or converted lagoon). Unit 2 remained with the traditional anaerobic lagoon treatment and was used as a control (hereafter called lagoon 2 or traditional lagoon).

In unit 3, a full-scale solid-liquid separation system was started in 2004 to treat all raw manure produced. The system treated raw manure flushed from the barns using solid-liquid separation (one step). The manure was reacted with flocculant polymer (polyacrylamide) and separated with a self cleaning rotating screen (0.25-mm opening). Subsequently, a small filter press dewatered the manure solids. The separated liquid was stored in lagoon 3 (hereafter called lagoon 3 or anaerobic lagoon with solid separation) and later used to refill the barn pit recharge system. Separated solids generated in unit 3 were also transported off-site and processed into value-added products.

During 2004, NH_3 emissions were measured in all three lagoons. Ammonia emissions were determined with passive flux samplers using the method of Sommer et al. (1996). The passive samplers were placed at four fixed locations perpendicular to each other around each lagoon. This layout enclosed most of the lagoon surface within a circular sampling plot (figure 1). Nine data collection periods lasting 23 h each were scheduled from February to November 2004 for the three lagoons. Further details on NH_3 emission measurements using the passive flux method are described in Szogi et al. (2006).

Even though waste treatment in all three production units was substantially different, animal production management remained the same during the study. Total annual NH_3 emissions were obtained by fitting a Gaussian curve to daily NH_3-N emission data versus day of the year using Prism 4.0 software (GraphPad Software, Inc., San Diego, CA). All water analyses were performed according to Standard Methods for the Examination of Water and Wastewater (APHA, 1998).

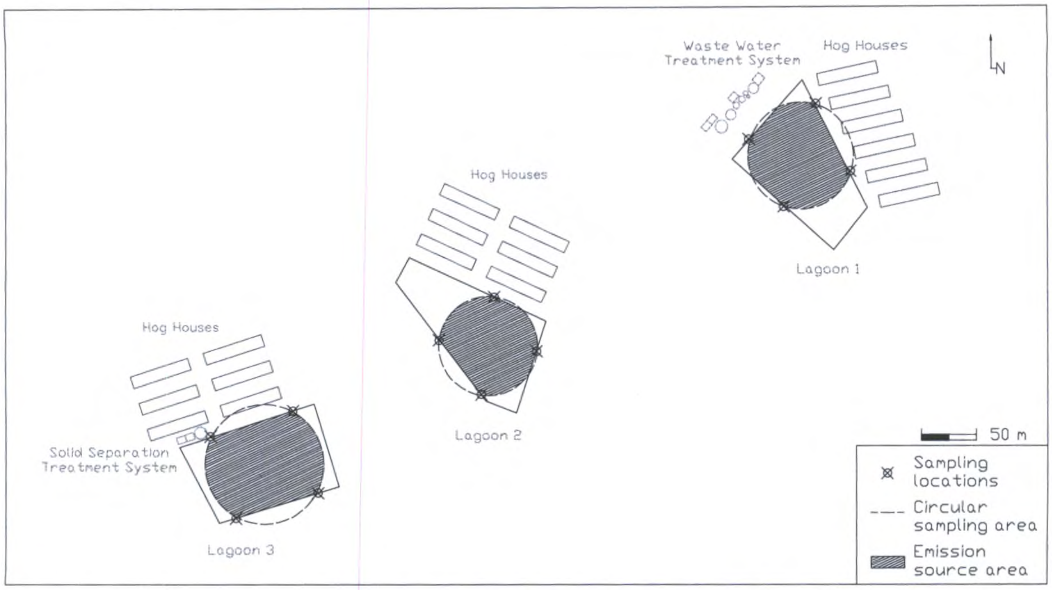


Figure 1. Schematic diagram of monitoring set-up for ammonia emission study for converted lagoon 1, traditional lagoon 2, and traditional lagoon 3 with solids separation, Duplin Co, NC.

Results and Discussion

Lagoon Water Quality

Monitoring of water quality was initiated in 2002, one year before the wastewater treatment plant started to operate in unit 1. The criterion to determine water quality improvement was the reduction in concentration of various water quality indicators (table 2). In 2002, the three lagoons were receiving flushed manure from the barns. Thus, the three anaerobic lagoons had similar annual mean pH, total ammoniacal N (TAN), total Kjeldahl N (TKN), and nitrate plus nitrate (NO_2+NO_3-N) concentrations (Jan.-Dec. 2002; table 2).

Beginning in February 2003, manure flush to lagoon 1 was halted and 100% of the liquid manure generated in the adjacent six barns was processed through the wastewater treatment plant (Figure 1). The quality of the liquid in lagoon 1 rapidly improved during 2003 as clean effluent from the treatment plant replaced anaerobic lagoon liquid, while water quality in lagoons 2 and 3 remained mostly unchanged. Annual (2003) average TAN and TKN levels in lagoon 1 declined 58% and 56%, respectively, with respect to lagoon 2 (control). In 2004, differences in TAN and TKN concentrations in lagoon 1 were even larger than in 2003; TAN declined 90% and TKN 81% with respect to lagoon 2 (table 2). In lagoon 1, the transition from anaerobic to aerobic water storage was noticeable in the first year of treatment. Dissolved oxygen (DO) concentrations in fall 2003 and winter 2004 (Oct. 2003 to March 2004, $n = 5$) at 0.15 m below the liquid surface averaged 3.5 mg/L in lagoon 1 and 0.5 mg/L in lagoon 2 (Szogi et al., 2006).

During 2004, the modular solid-liquid separation system in unit 3 separated > 90% of the manure solids but produced modest improvements on TAN and TKN levels in lagoon 3 (table 2). However, differences in average total solids (TS) content levels in lagoon 3 were 20% lower than lagoon 2 (control) and comparable to lagoon 1 as a result of solids removal (table 2).

Table 2. Change in water quality in three consecutive years for lagoon 1 before and after treatment plant was operational, lagoon 2 (control), and lagoon 3 before and after solids separation treatment, Goshen Farm, Duplin Co., NC[1].

Sampling Period	Lagoon	Treated	pH	TAN[2] (mg/L)	TKN (mg/L)	$NO_2 + NO_3$ (mg/L)	TS (g/L)
Jan.-Dec. 2002	1	N	8.0 (0.1)	464 (98)	506 (108)	0.08 (0.20)	4.2 (1.6)
	2	N	8.0 (0.2)	467 (118)	521 (122)	0.07 (0.21)	3.5 (0.9)
	3	N	8.0 (0.2)	469 (121)	517 (115)	0.07 (0.18)	3.6 (0.8)
Jan.-Dec. 2003	1	Y	8.1 (0.1)	186 (129)	230 (138)	4.1 (5.8)	2.7 (0.5)
	2	N	7.9 (0.1)	446 (102)	522 (127)	0.43 (1.4)	3.2 (0.7)
	3	N	7.9 (0.1)	375 (124)	439 (140)	0.41 (1.4)	2.8 (0.4)
Jan.-Dec. 2004	1	Y	8.1 (0.3)	37 (32)	76 (34)	20 (16)	2.3 (0.2)
	2	N	8.0 (0.2)	364 (88)	406 (79)	n.d.[3]	2.9 (0.2)
	3	Y	8.1 (0.2)	344 (141)	391 (140)	n.d.	2.3 (0.4)

[1] Data are annual means (standard deviation) of duplicate monthly composite samples.

[2] TAN = Total ammoniacal N; TKN = Total Kjeldahl N; $NO_2 + NO_3$ = Nitrite plus Nitrate; TSS = Total suspended solids.

[3] n.d. = not detected.

Reduction of Ammonia Emissions

Total annual NH_3 emissions in lagoons 1, 2 and 3 were calculated by fitting a Gaussian distribution to measured daily NH_3 emission values (figure 2). This curve was selected because it provided a good fit to the changes in daily NH_3 emissions throughout the year both in terms of R^2 and normality of residuals (residuals = emissions observed - emissions predicted) using the Shapiro-Wilk test statistic (Delong and Yuan, 1988). The total annual NH_3 emission for each lagoon is represented by the area under the curves in figure 2. On an annual basis, NH_3 emissions from the traditional lagoon (control) totaled 12,540 kg N/lagoon/yr (or 13,633 kg N/ha/yr) compared to 3,355 kg N/lagoon/yr (or 3,647 kg N/ha/yr) from the anaerobic lagoon 3 with solid separation treatment. Although water quality changes were modest in lagoon 3 with respect to the control (TAN, TKN and TS, Jan. – Dec. 2004, table 2), total annual NH_3 emissions reduction in lagoon 3 was 73% with respect to traditional lagoon 2. These results indicated that NH_3 emission reduction was more sensitive than the water quality to effect of the solid removal treatment. For the converted lagoon 1, reduction on annual NH_3 emissions was even larger with respect to traditional lagoon 2. Annually, NH_3 emissions from the converted lagoon totaled 1210 kg/lagoon/yr (or 1,311 kg N/ha/yr) compared to 12,540 kg N/lagoon/yr of the traditional lagoon. Lower N concentrations in the converted lagoon (lagoon 1, Jan.–Dec. 2004, table 2) substantially reduced annual NH_3 emissions by 90% with respect to those found in the traditional anaerobic lagoon. These results overall demonstrate that use of new wastewater technologies can substantially reduce ammonia emissions from confined swine production.

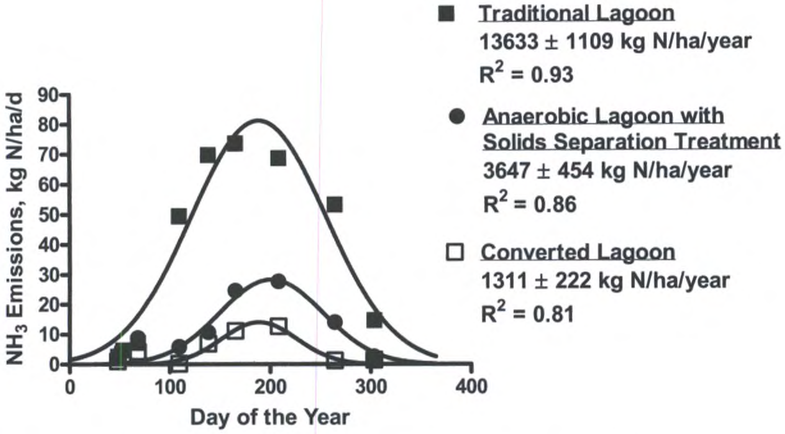


Figure 2. Change in daily rate of ammonia emissions throughout the year for the traditional lagoon, anaerobic lagoon with solids separation, and converted lagoon. The total annual ammonia (NH_3) emission for each lagoon system is represented by the area under each curve and indicated in the legend.

Conclusions

There is a need for treatment technologies that can effectively address environmental concerns associated with anaerobic lagoons. In particular, reduction of NH_3 emissions is a major environmental concern associated with confined swine production. In order to meet this need, two new wastewater treatment systems were demonstrated at full-scale in two of three 4,360-pig production units on a finishing farm in Duplin Co., NC. One treatment system combined solid-liquid separation with removal of N and P from the liquid phase, and the other wastewater treatment system consisted of solid-liquid separation before anaerobic lagoon storage. The third production unit was kept as a control using traditional anaerobic lagoon treatment.

In summary our findings indicate:

1) Ammonia emissions from the traditional anaerobic lagoon totaled 12,540 kg N/lagoon/year (13,633 kg N/ha/year). This result compares to lower NH_3 emissions of 3,355 kg N/lagoon/yr (or 3,647 kg N/ha/yr) from the anaerobic lagoon with solid-liquid separation, and 1,210 kg N/lagoon/yr (or 1,311 kg N/ha/yr) from the converted lagoon.
2) Although water quality improvements were modest in anaerobic lagoon with solid-liquid separation, total annual NH_3 emissions were reduced by 73% with respect to those of the traditional lagoon
3) Remarkable water quality improvements such as lower N concentrations in the converted lagoon substantially reduced annual NH_3 emissions by 90% with respect to those found in the traditional anaerobic lagoon.

Acknowledgements

This research was part of USDA-ARS National Program 206: Manure and By-product Utilization; CRIS Project 6657-13630-001-00D "Innovative Animal Manure Treatment Technologies for Enhanced Environmental Quality." Technology verification study was partially funded by North Carolina State University and its sponsor Smithfield Foods, Inc. through USDA Agreement 58-6657-2-202, NCSU Subcontract #2002-0478-02. The authors are grateful to Super Soil Systems USA, Clinton, NC for their contribution to the success of this project, and Premium Standard Farms for providing farm records on animal production. Mention of trade names or commercial products in this publication is solely for the purpose of providing specific information and does not imply recommendation or endorsement by the U.S. Department of Agriculture.

References

APHA. 1998. Standard Methods for the Examination of Water and Wastewater. 20th edition. American Public Health Association. Washington, DC.

Arogo, J., P.W. Westerman, and A.J. Heber. 2003. A review of ammonia emissions from confined swine feeding operations. *Trans. ASAE* 46(3): 805-817.

Battye, W., V.P. Aneja, and P.A. Roelle. 2003. Evaluation and improvement of ammonia emissions inventories. *Atmos. Environ.* 37(27):3873-3883.

Delong, D. M., and Y. C. Yuang. 1988. UNIVARIATE procedure. SAS Procedures Guide, Release 6.03. Cary, N.C.: SAS Institute, Inc.

Harper, L.A., R.R. Sharpe, T.B. Parkin, A. De Visscher, O. van Vleemput, and F.M. Byers. 2004. Nitrogen cycling through swine production systems: ammonia, dinitrogen, and nitrous oxide emissions. *J. Environ. Qual.* 33(4):1189-1201.

Sommer, S.G., E. Sibbesen, T. Nielsen, J.K. Schjorring, and J.E. Olesen. 1996. A passive flux sampler for measuring manure storage facilities. *J. Environ. Qual.* 25(2):241-247.

Szogi, A.A., M.B. Vanotti, and A.E. Stansbery. 2006. Reduction of ammonia emissions from treated anaerobic swine lagoons. *Trans. ASABE* 49(1): (In press).

Vanotti, M.B., A.A. Szogi, and P.G. Hunt. 2005. Wastewater treatment system. U.S. Patent 6,893,567. U.S. Patent Office.

Vanotti, M.B., A.A. Szogi, P.G. Hunt, P.D. Millner, and F.J. Humenik. 2006. Development of environmentally superior treatment systems to replace anaerobic swine lagoons in the USA. *Bioresource Technol.* (In press).

Walker, J., D. Nelson, and V.P. Aneja. 2000. Trends in ammonium concentration in precipitation and atmospheric ammonia emissions at a coastal plain site in North Carolina, U.S.A. *Environ. Sci. Technol.* 34(17):3527-3534.

Williams, C.M. 2001. Smithfield and Premium Standard Farms program implementation. p. 18-21. *In* G.B. Havenstein (ed.) International Symposium Addressing Animal Production and Environmental Issues, North Carolina State University, Raleigh, NC.

An Integrated Assessment of the Spatial Variability in Ozone-Induced Agricultural Crops Losses from Nitrogen Oxides Emissions

Daniel Q. Tong[1], Nicholas Z. Muller[2], Denise L. Mauzerall[1], and Robert O. Mendelsohn[2].
[1]Princeton University, Science, Technology and Environmental Policy Program, Woodrow Wilson School of Public and International Affairs, Princeton, NJ 08544, USA
[2]Yale University, School of Forestry and Environmental Studies, New Haven, CT 06511, USA

Abstract

Many recent nitrogen oxides (NO_X) control programs, including the EPA's NO_X SIP call and the latest CAIR programs, have adopted the "cap and trade" approach to mitigate regional NO_X emissions. An important premise of the cap-and-trade mechanism is that the marginal damages of NO_X emissions are equal across the trading domain. This paper examines the marginal ozone (O_3) damages on agricultural crops caused by NO_X emissions in different locations around the continental United States. We calculate O_3 damages using a new integrated assessment model that links pollution emissions to their chemical transformation and transport, to crops exposure, and to final impacts. We find that a fixed amount of increased NO_X emissions in a rural upwind state can cause crops losses more than 100 times as much as in a more urbanized Northeastern state. Although there are clear benefits to a "cap and trade" program which substantially reduces total emissions, the results of this paper, together with our previous studies on the marginal impacts of NO_X emissions on human health, raise questions about the fundamental assumption underlying the current "cap and trade" programs of NO_X emission control.

Air and Groundwater Pollution in Turkish Mediterranean Coast

Semra G. Tuncel* and Nur Banu Öztaş-Emek
Middle East Technical University, Department of Chemistry, 06531, Ankara, Türkiye
*Corresponding author, e-mail:semratun@metu.edu.tr; Fax: +90 312 2101280

Introduction

Kumluca, is a district of Antalya city at Turkish Mediterranean cost with very intensive agricultural activities employing greenhouses. Around 30% of green vegetable demand of European countries is met by Kumluca region. For many years, different classes of pesticides, including chlorinated and phosphorous pesticides, have been applied in the region. These fertilizers contaminated the air, soil and ground water. Present work investigate air and ground water pollution in the region.

The gaseous pollutants in the atmosphere are emitted from a variety of anthropogenic, natural sources and agricultural activities. These compounds are mixed, transported, and reacted and finally removed from the air back to the earth's surface.

Pesticide use in agriculture is increased in the last few decades, throughout the world for the sake of increasing the agricultural productivity to meet the increasing demand for food production. Even though there is a tendency to use less harmful and less persistent molecules, the contamination of air and aquatic environment by pesticides is a matter of great concern.

The aim of this thesis can be summarized as follows:

- Estimation of contribution of the agricultural activities to air pollution.
- Study of effects of agricultural activities for the surface and the ground water of the Turkish Mediterranean coast.

Results and Discussion

In Table 1, a summary of the statistics for the air pollutants are given: Parameters presented in the table include number of samples, arithmetic mean, geometric mean and the median values as well as standard deviation, variance, the maximum and the minimum concentrations and coefficient of variations. In order to understand the relative strengths of anthropogenic and sea salt contributions of sulfate in the Mediterranean atmosphere, the non-sea-salt sulfate (nss-SO_4^{2-}) concentrations were calculated using the following equation:

$$\text{nss-}SO_4^{2-} = \text{t-}SO_4^{2-} - 0.139\,[Cl^-]\,,$$

Where t-SO_4^{2-} is total sulfate and 0.139 is taken as the ratio of SO_4^{2-} to Cl^- in seawater.

The results of the analysis of chlorinated and phosphorus pesticides in well waters are presented in Table 5 and Table 6 respectively. The presented data are for spring sampling period, where the water level in the reservoirs was in their highest level. The measured concentrations were corrected with the percent recoveries. From the results, it is seen that, although the chlorinated pesticides are observed with a higher frequency than the phosphorus pesticides, their concentrations are lower than the limit values (0.1 µg/L). However, for some of the phosphorus pesticides, the values are above the limits for dichlorfon, phosphamidon, fenthion, azinphos-methyl and bromophos-ethyl for some samples.

Table 1 Summary of the statistics for the measured air pollutants (in μg m^{-3})

Parameter	NH_3(g)	NH_4^+	Cl^-	NO_3^-	SO_2 (g)	total SO_4^{2-}	nss-SO_4^{2}	HNO_3 (g)	O_3
# of Samples	**174**	**173**	**168**	**171**	**176**	**171**	**157**	**30**	**171**
Mean	**0.99**	**2.19**	**4.30**	**2.28**	**2.03**	**2.35**	**1.98**	**0.42**	**61.8**
Median	**0.64**	**1.99**	**2.01**	**1.94**	**1.97**	**1.87**	**1.46**	**0.35**	**61.9**
Standard Deviation	**1.08**	**1.17**	**6.94**	**1.77**	**1.06**	**2.01**	**1.87**	**0.28**	**14.54**
Minumum	**0.04**	**0.02**	**0.09**	**0.09**	**0.08**	**0.25**	**0.09**	**0.06**	**25.77**
Maximum	**6.44**	**6.08**	**58.3**	**8.94**	**5.89**	**12.5**	**11.4**	**1.33**	**98.56**
Range	**6.40**	**6.06**	**58.2**	**8.85**	**5.82**	**12.3**	**11.3**	**1.27**	**72.79**

Table 5. OC pesticide concentrations in ground waters

Name	% obs	av w/o ND (ng/L)	stdev	RSD	Min (ng/L)	Max (ng/L)
a hch	43	0.51	0.52	101.26	0.07	1.08
b hch	14	38.21			38.21	38.21
g hch	100	3.50	6.29	179.70	0.16	17.59
d hch	43	3.71	5.90	159.25	0.29	10.52
heptachlor	57	7.39	6.81	92.10	0.86	16.78
aldrin	71	28.67	24.39	85.09	1.37	60.77
hep endo	71	4.70	4.09	86.83	0.12	7.85
endosulfan	100	3.95	4.41	111.59	0.26	13.12
dieldrin	86	0.87	0.30	34.99	0.54	1.25
dde	57	1.68	1.08	64.01	0.57	3.16
endrin	100	3.27	0.94	28.60	2.48	5.18
b endos	100	3.79	0.91	23.89	2.82	5.20
endos sulf	100	6.29	2.26	35.87	3.83	10.13
ddt	100	4.72	0.96	20.34	3.57	5.94
methoxy	71	2.55	1.27	49.71	1.21	3.99

Table 6. OP pesticide concentrations in ground waters

Name	% obs	av w/o ND (ug/L)	st.dev	RSD	Min (ug/L)	Max (ug/L)
Methamidoph	0					
Dichlorfon	7	0.18			0.18	0.18
Diazinon	13	0.02	0.02	93.59	0.01	0.03
Phosphamid	27	0.09	0.03	30.08	0.06	0.12
Parathion	0					
Fenitrothion	0					
Pirimiphos	7	0.02			0.02	0.02
Malathion	0					
Fenthion	47	0.09	0.02	24.72	0.05	0.12
Chlorpyr.	0					
Bromo-methy	0					
Methidathion	7	0.09			0.09	0.09
Bromo-ethyl	20	0.28	0.18	63.08	0.08	0.40
Azinph-met	13	0.10	0.001	1.42	0.10	0.10

Methodological Study of Headspace Solid Phase Microextraction GC/MS Analysis of Odors Associated with Livestock Manure and Municipal Wastewater Sludge

Dikla Tweg[1,2], Yael Laor[1], Uzi Ravid[1], Yona Chen[2], Rima Baybikov[1], Shlomo Kapua[3], and Ilan Halachmi[1]

[1]Agricultural Research Organization, Newe Ya'ar Reseach Center, Ramat Yishay, 30095, Israel
(corresponding author: laor@volcani.agri.gov.il);
[2]Department of Soil and Water Sciences, Faculty of Agricultural, Food and Environmental Quality Sciences, The Hebrew University of Jerusalem, Rehovot, 76100, Israel;
[3]Israel Ministry of the Environment, Agro-Ecology division, Jerusalem, 34033, Israel.

Abstract

A large number of volatile organic compounds (VOCs) associated with livestock production are known to produce offensive odors which affect the air quality of nearby residents/workers. Land application of manure and municipal wastewater sludge at various degree of stability may also cause odor problems and conflicts between rural and urban residents. It is known that a comprehensive odor characterization and possible correlations between analytical measurements and human perception of offensiveness require a robust of advanced analytical techniques. Selecting proper sampling and an analytical approach for odor characterization is challenging. Headspace solid phase microextraction (HS-SPME) has been used in a large number of investigations for extracting odors from manure or from ambient air followed by gas chromatography mass spectrometry (GC/MS) analysis. Yet, reports on the examination of multiple methodological aspects related to the use of HS-SPME-GC/MS for manure and sludge odor characterization are rare in the literature. Here we present a methodological study that explored multiple aspects of the HS-SPME-GC/MS analysis, using calves' manure and municipal wastewater sludge as model matrices. Our methodological examination included elements of reproducibility, fiber type, extraction time, storage prior to SPME extraction, and sample preservation after extraction. Besides the specific tested element, analyses were conducted under identical conditions of fiber type, temperature and time of extraction. We show that: (i) HS-SPME extractions were more reproducible for some VOCs (phenolic compounds) than for others (some volatile fatty acids); (ii) one-week storage of manure/sludge at 4°C and -18°C, with and without 20% NaCl solution did not affect significantly odor profiling; (iii) DVB/CAR/PDMS fiber had higher affinity for phenolic compounds (e.g. *p*-cresol and indole) than CAR/PDMS fiber. Preferred affinity for volatile fatty acids and sulfuric compounds was less clear among the two fibers; and (iv) sample preservation after SPME extraction, was tested by inserting the SPME needle into a clean GC vial immediately after extraction and overnight storing at 4°C until GC-MS analysis. This method was generally successful but some decrease in specific VOCs was detected. Possible fiber contamination during storage was carefully minimized. Overall, if performed under controlled conditions, SPME can be considered as a reproducible semi-quantitative analytical approach for odor monitoring and characterization. It can be manipulated with fiber type and extraction conditions to focus on compounds of interest. A good method of sample preservation enables a safe use of this approach for ambient air measurements done in a distance from the analytical lab.

Mitigating Swine Odor with Strategically Designed Shelterbelt Systems: A Review

John Tyndall and Joe Colletti

Department of Natural Resource Ecology and Management, Iowa State University

Poster Abstract

Recent reports clearly indicate that odor emitted from concentrated livestock production facilities in the Midwest is a significant social problem that negatively impacts rural and state economies, human health, and the quality of rural life. Whereas livestock derived odors are ubiquitous with animal agriculture, four factors are thought to cause an increase in odor nuisance and a need for additional technological and management creativity. First, larger-scale livestock confinement production has led to increased concentrations of manure being stored and utilized in relatively small geographic locations. Second, urban/suburban expansion into the agricultural landscape has put many more people with limited agricultural experience into closer proximity to livestock production. Third, the current livestock odor problem is characterized by high concentrations of odorous emissions that travel across highly modified landscapes relatively devoid of any significant natural barriers that can impede, alter, absorb, or dissipate the odor plumes prior to contact with people. Because of this state of affairs a potential incremental approach to dealing with livestock odor is the use of shelterbelts (trees and shrubs) arranged in strategic designs near and within livestock facilities. This review outlines the various ways that shelterbelts can be an effective technology which bio-physically mitigates odor in a socio-economically responsible way thereby reducing social conflict from odor nuisance. The biophysical potential of shelterbelts to mitigate livestock odor arises from the tree/shrub impacts on the central characteristics and physical behavior of livestock odor. Specifically, as the majority of odors generated in animal facilities that are intense and detectable at appreciable distances travel as aerosols (i.e. particulates), there is compelling evidence that shelterbelts can ameliorate livestock odor by impeding the movement of these particulates. Because the odor source is near the ground and the tendency of the plume is to travel along the ground, shelterbelts of even modest heights (i.e. 20-30 ft) may be ideal for plume interception, disruption, and dilution. Shelterbelts can be adapted to fit almost any production situation and expected/experienced odor plume shape and timing. Depending on the shelterbelt design and tree/shrub species used, it can deal with the temporal changes to provide long term, year round plume interception, with increasing effectiveness over time. Additionally, more is also becoming known about how landscape aesthetics affect how people might perceive livestock odor, suggesting that landscape elements such as shelterbelts can lead to improvements and perhaps more positive interpretations of livestock odor and the farm systems that create them.

Introduction

Recent reports clearly indicate that odor emitted from concentrated livestock production facilities in the Midwest is a significant social problem that negatively impacts rural and state economies, human health, and the quality of rural life (Iowa CAFO Air Quality Study, 2002; Wing and Wolf, 2000; Thu and Durrenberger, 1998). Whereas livestock derived odors are ubiquitous with animal agriculture, four factors are thought to cause an increase in odor nuisance and a need for additional technological and management creativity. First, larger-scale livestock confinement production has led to increased concentrations of manure being stored and utilized in relatively small geographic locations. Second, urban/suburban expansion into the agricultural landscape has put many more people with limited agricultural experience into closer proximity to livestock production. Third, the current livestock odor problem is characterized by high concentrations of odorous emissions that travel across highly modified landscapes relatively devoid of any significant natural barriers that can impede, alter, absorb, or dissipate the odor plumes prior to contact with people. Because of this state of affairs a potential incremental approach to dealing with livestock odor is the use of shelterbelts (trees and shrubs) arranged in strategic designs near and within livestock facilities. This review outlines the various ways that shelterbelts can be an effective technology which bio-physically mitigates odor in a socio-economically responsible way thereby reducing social conflict from odor nuisance.

The biophysical potential of shelterbelts to mitigate livestock odor arises from the tree/shrub impacts on the central characteristics and physical behavior of livestock odor. Specifically, as the majority of odors generated in animal facilities that are intense and detectable at appreciable distances travel as aerosols (i.e. particulates), there is compelling evidence that shelterbelts can ameliorate livestock odor by impeding the movement of these particulates. Because the odor source is near the ground and the tendency of the plume is to travel along the ground, shelterbelts of even modest heights (i.e. 20-30 ft) may be ideal for plume interception, disruption, and dilution. Shelterbelts can be adapted to fit almost any production situation and expected/experienced odor plume shape and timing. Depending on the shelterbelt design and tree/shrub species used, it can deal with the temporal changes to provide long term, year round plume interception, with increasing effectiveness over time. Additionally, more is also becoming known about how landscape aesthetics affect how people might perceive livestock odor, suggesting that landscape elements such as shelterbelts can lead to more positive interpretations of livestock odor and the farm systems that create them.

The following are very brief sample excerpts from the review that show some of the underlying evidence that supports the use of shelterbelts for livestock odor mitigation. Based on evidence available in research literature, there are five primary, interacting, ways that shelterbelts can mitigate livestock odors:

1) Dilution of gas concentrations of odor into the lower atmosphere

- Shelterbelts create turbulence at the surface of the terrain that intercept and disrupt odor plumes traveling in laminar flow helping to push the plume into the lower atmosphere facilitating dilution (Bottcher et al., 2001; Bottcher et al., 2000; OCTF, 1998; SOTF, 1995; Takle, undated).
- Lowering wind speeds over storage lagoons can reduce convection of odorous compounds from the surface and allow for slower release of the odor plume which also facilitates dilution (Bottcher et al., 1999).

2) Encouraging dust and other aerosol deposition by reducing wind speeds

- Pesticide drift mitigation research suggests that due to reduced wind speeds drift pesticide will drop from the air stream. In broadleaf species, downwind drift reductions of 70% (no leaves) to 90% (in leaf) have been recorded (Porskamp et al., 1994).
- Numerical simulation of the effects of tall barriers around manure lagoons predicted reductions in downwind malodorous lagoon emissions of 26% to 92% (Liu et al., 1996).
- Wind tunnel modeling of a three-row shelterbelt system has quantified reductions of 35% to 56% in the downwind mass transport of odor –bearing particulates (dust and aerosols) (Laird, 1997; Thernelius, 1997).

3) Physical interception of dust and other aerosols

- Vegetation can and does filter airstreams of particulates. As air moves across vegetative surfaces, leaves and other aerial plant surfaces remove some of the dust, gas, and microbial constituents of airstreams. It is also generally accepted that trees and other woody vegetation (i.e. shrubs) are among the most efficient natural filtering structures in a landscape in part due to the very large total surface area of leafy plants, often exceeding the surface area of the soil containing those plants by over 400-fold (Schultze, 1982).
- Leaves with complex shapes and large circumference to area ratios collect particles most efficiently, thus, conifers may be more effective particle traps than deciduous species (Smith, 1994) and have a temporal advantage because of the leaves being on the trees year long.

4) By way of acting as a sink for the chemical constituents of the odorous pollution

- Volatile Organic Compounds (VOC's) –the offensive smell in livestock manure- have a distinct affinity to the lipophilic membrane (the cuticle) that covers plant leaves and needles. Studies are underway to examine the efficiency of various plants (Beattie et al., undated).
- Researchers have quantified measurable quantities of anthropocentric VOC's that have accumulated at the surface of plants (adsorption) and within the plants tissues (absorption) (Reischl et al., 1989; Reischl et al., 1987; Gaggi et al., 1985)

- Micro-organisms dominate the surface of plants (Preece and Dickenson,1971). These organisms also adsorb and absorb VOC's and provide additional surface area for pollution collection. These organisms also have the ability to metabolize and breakdown VOC's (Screiber and Schonherr, 1992; Muller, 1992; Beattie et al, undated).

5) Enhancing the aesthetics of pork production sites and rural landscapes.

- Professionals involved with livestock agriculture generally accept that a well-landscaped operation, which is visually pleasing or screened from view by landscaping is much more accepting to the public than one which is not (Lorimor, 1998; NPPC, 1995; Melvin, 1996). It is this notion of visual screening that has made landscaping and shelterbelts a common suggestion from agricultural engineers with regards to minimizing odor problems.
- If it is made known to neighbors and local communities that a shelterbelt is being used as a pollution (air or water) control tool, it may serve as very visible proof that a livestock producer is making an extra effort to control odor.

Shelterbelts are also a technology that can be considered "production technology neutral" and "size neutral" in that swine producers of all kinds – confinement, modified confinement, hoop house, pasture - and all sizes can plant designed shelterbelt systems. Shelterbelts, very uniquely, offer a technology that both producers and rural residents and communities can appropriately use, suggesting "user neutrality". Further, as opposed to other odor mitigating technologies that typically depreciate over time, shelterbelts may be the only odor control technology that theoretically increases in effectiveness over time. As with other "tree" based technologies used in agriculture, the effectiveness of shelterbelts in mitigating odor comes from providing complex ecological infrastructure within an otherwise ecologically simplified system (Schultze et al, 2000). As the trees grow larger, and more morphologically complex their ability to mitigate odors should become increasingly efficient. Though this improvement over time is contingent upon the health and maintenance of the shelterbelt systems and the continuance of hog production best management practices.

This research review makes it clear that the published information on the ability of shelterbelts to mitigate on-and off-farm livestock odor is limited and further bio-physical, economic, and social qualification and quantification of this technology is needed. Yet the existing evidence seems to indicate that shelterbelts can help incrementally to reduce odor pollution and sustain both the swine industry and the quality of rural life.

References Cited

Beattie, G., A. Disperito, and L. Halverson. Undated. Use of Plants and Plant-Associated Microbes to Reduce Odor Emission from Livestock Production Facilities. Research Report. Hog Odor and Waste Management Issues Production Research Program.
http://extension.agron.iastate.edu/immag/pr/ISUrep1.html (Last accessed February 15, 2006).

Bottcher, R.E., R.D. Munilla, and G.R. Baughman. 1999. Controlling dust and odor from buildings using windbreak walls. Proceedings: 1999 Animal Waste Management Symposium. Raleigh, North Carolina. January 27-28, 1999.

Bottcher, R.W., R.D. Munilla, G.R. Baughman, and K.M. Keener. 2000. Designs for windbreak walls for mitigating dust and odor emissions from tunnel ventilated swine buildings. pp.174-181 in: Swine Housing, Proc. of the 1st International Conference, Oct. 9-11, 2000, Des Moines, Iowa. American Society of Agricultural Engineers, St. Joseph, MI 49085-9659.

Bottcher, R.W., R.D. Munilla, K.Keener, and R.S. Gates. 2001. Dispersion of livestock building ventilation using windbreaks and ducts. Presentation at the 2001 ASAE Annual International Meeting, Sponsored by ASAE Sacramento Convention Center, Sacramento, California, USA July 30-August 1, 2001.

Gaggi, C., E. Bacci, D. Calamari, and R. Fanelli. 1985. Chlorinated hydrocarbons on plant foliage: An indication of the troposheric contamination level. Chemosphere, Vol.14, Nos.11/12:1673-1686.

Iowa CAFO Air Quality Study. 2002. Iowa (CAFO) Concentrated Animal Feeding Operations Air Quality Study. ISU/UI Study Group, University of Iowa College of Public Health, Iowa City, IA, 2002.

Laird, D.J. 1997. Wind tunnel testing of shelterbelt effects on dust emissions from swine production facilities. Thesis (M.S.)--Iowa State University.

Liu, Q., D.S.Bundy, and S.J. Hoff. 1996. The effectiveness of using tall barriers to reduce odor emission. Proceedings of the International Conference on Air Pollution from Agricultural Operations, Midwest Plan Service, Ames, Ia. pp. 403-407.

Lorimor, J. 1998. Iowa Odor Cotrol Demonstration Project: Landscaping. Cooperative Extension Service. Iowa State University Extension. Pm-1754h.

Melvin, S.W. 1996. Swine odor measurement and control issues reviewed. Feedstuffs. May 27, 1996. V. 68 (22) 12-14.

Muller, R. 1992. Bacterial degradation of xenobiotics. pp 35-58. In: (Eds.) Fry, J.C., G.M. Gadd, R.A. Herbert, C.W. Jones, and I.A. Watson-Craik. 1992. Microbial Control of Pollution. 48th Symposium of the Society for General Microbiology held at the University of Cardiff. NPPC, 1995;

Odor Control Task Force (OCTF). 1998. Board of Governors of the University of North Carolina. Control of odor emissions from animal Operations. http://www.cals.ncsu.edu/waste_mgt/control.htm (Link no longer accessible).

Porskamp, H.A.J., J.M.P.G. Michielsen, and Ir. J.F.M. Huijmans. 1994. The reduction of the drift of Pesticides in fruit growing by a wind-break. Dienst Landbouwkundig Onderzoek, Instituut voorMilieu-en Agritechniek, Rapport 94-29. Wageningen, 27pp.

Preece, T.F. and C.H. Dickinson. 1971. Ecology of Leaf Surface Micro-Organisms. Proceedings of an International Symposium held at the University of Newcastle Upon the Tyne. September 1970. Academic Press, London.

Reischl, A., M. Reissinger, and O. Hutzinger. 1987. Occurrence and distribution of atmospheric organic micropollutants in conifer needles. Chemosphere, Vol.16, Nos.10-12:2647-2652.

Reischl, A., M. Reissinger, H. Thoma, and O. Hutzinger. 1989. Accumulation of organic air constituents by plant surfaces. Chemosphere, Vol.18, Nos.1-6:561-568.

Schultze , R.S., J.P. Colletti, T.M. Isenhart, C.O. Marquez, W.W. Simpkins, and C.J. Ball. 2000. Riparian Forest Buffer Practices. Chapter 7, pp 189-282. In: Garrett, H.E., W.J. Rietveld, and R.F. Fisher (Eds.). North American Agroforestry: An Integrated Science and Practice. American Society of Agronomy, Inc. Madison, Wisconsin. 2000. p. 402.

Schulze, E.D. 1982. In: O.L. Lange, C.B. Osmund, H. Ziegler (Eds.), Encyclopedia of Plant Physiology. Vol. 12 B. Springer-Verlag, Berlin. pp. 615-676.

Schreiber, L., and J. Schonherr. 1992. Leaf surface microflora may significantly affect studies on foliar uptake of chemicals. Botanical Acta. 105: 345-347.

Smith, R. J. 1994. Dispersion of odours from ground level agricultural sources. J. agric. Engng. Res. 54:187-200.

Swine Odor Task Force (SOTF). 1995. Options for Managing Odor. North Carolina University, http://www.mtcnet.net/~jdhogg/ozone/odor/swineodr.html (Last accessed February 15, 2006).

Takle, E.S. Unknown Date. A numerical model feasibility study of managing swine odor by manipulation of atmospheric flow. A research proposal. Department of Agronomy, Iowa State University.

Thernelius, S.M. 1997. Wind tunnel testing of odor transportation from swine production facilities. Thesis (M.S.)--Iowa State University, 1997.

Thu, K.M. and E.P. Durrenberger, editors. 1998. Pigs, Profits, and Rural Communities. New York: State University of New York Press.

Wing, S. and S. Wolf. 2000. Intensive livestock operations, health, and quality of life among Eastern North Carolina Residents. Environmental Health Perspectives. Vol. 108, Number 3 March 2000.

Financial Feasibility of Using Shelterbelts for Swine Odor Mitigation

John Tyndall and Joe Colletti
Department of Natural Resource Ecology and Management, Iowa State University

Poster Abstract

This study is a farm-level financial feasibility examination of an odor mitigation technology that has been receiving increasing attention – the use of shelterbelts in and around swine production facilities. Shelterbelts are purposefully planted rows of trees and shrubs around the main sources of swine odor – the swine buildings, the manure storage systems, and crop fields that receive land applied manure. As current research suggests if shelterbelts are utilized appropriately (proper species, scale, location, etc.), they should play a significant incremental role within a "suite" of odor mitigation strategies, which collectively will reduce odor nuisance risk for all stakeholders in rural air quality. Yet despite the apparent promise of this technological approach, very little is known about the financial requirements for installation (i.e. site preparation and planting) and long term management of shelterbelt systems that have been designed for odor mitigation. Therefore, by using a series of hypothetical pork finishing farms and a number of shelterbelt design scenarios, the specific objectives of this discounted cash-flow oriented financial analysis are: 1) To assess the farm level financial nature of planting and maintaining odor mitigation shelterbelt systems over an extended planning horizon (20 years); 2) To compare various shelterbelt costs with known figures (WTP) on pork producer expenditures strictly for odor control; 3) To assess the effect of current federal programs (specifically the Environmental Quality Incentive Program and the Conservation Reserve Program) on reducing the financial burdens to swine producers for using shelterbelts. The results show that the expense in using shelterbelts in these varied cases, in most cases adds only a few pennies to overall production costs. The calculated costs for shelterbelt establishment and maintenance over a twenty-year period for all model pork finishing farms, depending upon the scenario, range from \$0.01 to \$0.82 per pig produced. The results of this study also emphasize the importance of cost share programs, particularly EQIP. Still, in most cases, both with and without cost share programming, the upfront and maintenance costs were well below known expenditures for odor management technology.

Introduction

While odors from swine production are ubiquitous with animal agriculture and have been historically tolerated by rural communities, structural changes in the US swine industry (i.e. increased farm size, increased geographic concentration of production facilities and therefore animal manure) have caused odor to become much more pervasive and offensive (Iowa CAFO Air Quality Study, 2002).

This study is a farm-level financial feasibility examination of an odor mitigation technology that has been receiving increasing attention – the use of shelterbelts in and around swine production facilities (MWPS, 2002). Shelterbelts are purposefully planted rows of trees and shrubs around the main sources of swine odor – the swine buildings, the manure storage systems, and crop fields that receive land applied manure.

As research suggests (Malone and Abbot-Donnelly, 2001; MWPS, 2002; Khan and Abbassi, 2001) if shelterbelts are utilized appropriately (proper species, scale, location, etc.), they should play a significant incremental role within a "suite" of odor mitigation strategies. Collectively, these strategies can reduce odor nuisance risk for all stakeholders in rural air quality. Yet despite the apparent promise of shelterbelts for odor mitigation, very little is known about the financial requirements for installation (i.e. site preparation and planting) and long term management of shelterbelt systems that have been designed for odor mitigation (Lorimor, 2002).

The purpose of this analysis is to show that the planting and long term maintenance costs of shelterbelts for odor mitigation are indeed within the range of (producer determined) acceptable expenses for odor management. Therefore, the specific objectives of this financial analysis of the use of shelterbelts for odor mitigation are:

- To assess the farm level financial nature of planting and maintaining odor mitigation shelterbelt systems over an extended planning horizon (20 years) for four different hypothetical model swine farms.
- To compare shelterbelt costs with known figures on pork producer expenditures strictly for odor control.
- To assess the effect of current federal programs (specifically the Environmental Quality Incentive Program and the Conservation Reserve Program) on reducing the financial burdens to swine producers for using shelterbelts.

A critical assumption to this financial analysis is that shelterbelts can be bio-physically effective in incrementally attenuating swine odor. Recent research suggests that shelterbelts can play significant roles in biophysically and socially mitigating odor in a socio-economically responsible way thereby reducing social conflict from odor and dust nuisance (Leuty, T. 2004; Malone et al., 2004 a; Malone et al., 2004 b; Malone and Abbot-Donnelly, 2001; Tyndall and Colletti, 2001).

Research has shown that because the source of livestock odor is near the ground and the tendency of the odor plume is to travel along the ground, shelterbelts of even modest heights (i.e. 20-30 ft) may be ideal for plume interception, disruption, and dilution (Bottcher, 2001, Heisler and DeWalle, 1988; Laird, 1997; Thernelius, 1997; Takle, 1983). Shelterbelts can be adapted to fit the production situation and expected/experienced odor plume shape and timing. Depending on the shelterbelt design and tree/shrub species used, this technology can provide long term, year round plume interception, with increasing effectiveness over time. More is also becoming known about how landscape aesthetics affect how people might perceive livestock odor, suggesting that landscape elements such as shelterbelts can lead to improvements and perhaps more positive interpretations of livestock odor and the farm systems that create them (Mikesell et al., 2001; Kreis, 1978).

Shelterbelt Designs for Four Hypothetical Pork Feeder Finisher Farms

Four hypothetical pork-finishing farms were developed by modeling them after real pork finishing farms in Iowa. The model farms serve as the frame for financial calculations of shelterbelts designed to mitigate odor emissions specific to those model farms (See Table 1 below for details regarding each model farm). Each model farm reflects different pork production technologies, building ventilation and manure storage. Each model farm has an accompanying model shelterbelt system designed specifically for the site specificity of each model farm and the represented production technologies. Additional assumptions underlying each model farm shelterbelt design and the farm level financial comparisons are as follows: There are two main odor management zones receiving odor mitigative attention with shelterbelts – 1) A shelterbelt component for the farm itself, which includes the immediate production site property, the swine buildings and the manure storage facilities; And 2) a shelterbelt component for a contiguous crop field that receives manure from the production facility for nutrient utilization purposes.

The swine farms are assumed to be located in central Iowa. Shelterbelt systems are designed with these wind patterns in mind.

Two planting stock cost scenarios were used: A high priced 6+ year old planting stock scenario and a low priced seedling stock scenario.

Table 1. Model Farm and shelterbelt system details.

	Farm A – Full confinement facility	Farm B– Full confinement facility	Farm C– Full confinement facility	Farm D – Hoop Barn Facility
Annual Pig Production	10,500	2,025	1,600	1,000
Building Ventilation Type	Tunnel	Mechanical	Natural	Natural
Manure Storage	Above ground concrete	Above ground concrete	Above ground concrete	Solid/bedding
Farm Acreage	5.91	1.38	0.90	0.83
Number of trees Planted	950	325	178	122
Number of Shrubs Planted	214	141	51	145
Application Field Acreage	660	160	100	25
Number of Application Field Trees	2,145	1,045	835	395

Financial Appraisal

This study used standard cash-flow analysis to examine the costs of establishing and managing a designed shelterbelt within and around the four model pork finishing facilities and shelterbelts that are planted around manure application fields. Costs examined include: site preparation, plant stock, shelterbelt establishment, shelterbelt repair and maintenance (long term management), and land rent (opportunity costs) for shelterbelts planted around crop fields. Measures of odor mitigation effectiveness from shelterbelts are not considered.

All costs were discounted using standard discounting formulation (Klemperer, 1996). Final costs of the model shelterbelts are reported in two ways, first Present Value Cost (PVC) and PVC minus any governmental program payments (PVC^{GOV}) are established for each shelterbelt design. The second way is to apply a Capital recovery factor to total costs of each model shelterbelt and divide by the number of pigs produced annually; this presents total costs on a per unit of production (per pig) and spreads the costs out across all the pigs produced in a twenty-year period. This method is appropriate because the shelterbelt systems are designed to mitigate swine odor over an estimated ownership span (ISU, 1998) and therefore costs are spread over all the pigs produced at that facility over a 20 year period.

Across all the model farms the average costs associated only with the shelterbelt systems planted in and around the facility grounds range from $0.02 and $0.25 per pig produced (seedling price and high price respectively). When EQIP cost share programming is factored in, the average costs to producer's fall to $0.01 to $0.13 per pig. Looking at the combined average costs for establishment and management of shelterbelt systems around the facility grounds and the manure application fields of all four model facilities, the total costs without cost share come to $0.21 and $0.82 per pig produced (seedling/high price respectively). With cost share programming (in this case EQIP for the facility grounds and CRP for the application fields) the costs fall to $0.05 and $0.26 per pig produced. The largest facility had the highest upfront costs and the highest present value costs over 20 years, but ultimately slightly lower costs per pig produced due to the high production volume.

When these costs are compared to known (revealed) pork producer willingness to pay for on-farm odor management (USDA, 1996), the costs are in most cases well below what producers have spent in the past. The results of this study (both with and without cost share programming) show that the relatively low costs of establishing and maintaining shelterbelts for odor mitigation allow a "feasibility excess" of as much as $0.33 - $0.59 per pig produced that, ostensibly, could be spent on other complimentary odor management strategies. Financially there is room for shelterbelts to be part of a "suite" of odor management technology. It is also clear that EQIP benefits become more important as upfront costs of shelterbelt implementation increase (in this case the costs of the planting stock). This point is relevant for producers planning shelterbelt systems using older, larger planting stock.

However, a separate survey of NRCS EQIP agents revealed that while seven of the top ten hog producing states do recognize shelterbelts as an EQIP relevant odor mitigating Best Management Practice only a handful of pork producers around the country have received EQIP dollars specifically to use shelterbelts for odor mitigation.

General Conclusion

Based on the above cost information the use of shelterbelts for odor mitigation in these varied cases only add a few pennies to overall production costs. Expansion/promotion of cost share possibilities to producers likely will play a key role in future use of shelterbelts as an odor mitigating technology. Pork producers are in a legitimate quandary in that they face distinct risk of nuisance litigation at the same time that their profit margins are thin at best and very often negative on average. The costs of even pennies per pig can seem too much to bear under such economic conditions. As other studies have shown for some time now, when the government offers to help pay for agricultural externalities such as air and water pollution, farmers are much more likely to participate in promoted technological approaches (Purvis et al., 1989). This research demonstrates that shelterbelts can be a financially feasible part of overall plans to improve the sustainability of both rural communities and of the livestock industry.

References Cited

Bottcher, R.E. 2001. An environmental nuisance: Odor concentrated and transported by dust. *Chemical Senses* 26:327-331.

Heisler, G. M., and D.R. Dewalle. 1988. Effects of windbreak structure on windflow. Elsevier Science Publishers B.V., Amsterdam. *Agriculture, Ecosystems and Environment*,22/23 (1988) 41-69.

Iowa CAFO Air Quality Study. 2002. Iowa (CAFO) Concentrated Animal Feeding Operations Air Quality Study. ISU/UI Study Group, University of Iowa College of Public Health, Iowa City, IA, 2002.

Iowa State University. 1998. Livestock Issues: Q & A. Iowa State University, University Extension. Ames, Iowa. PM – 1741e. March 1998.

Khan, F.I. and S.A. Abbasi. 2001. Effective design of greenbelts using mathematical models. *Journal of Hazardous Materials*. B81: 33-65.

Klemperer, W.D. 1996. Forest Resource Economics and Finance. McGraw-Hill, Inc. New York. 515p.

Kreis, R.D. 1978. Limiting the environmental impact of animal production odors. *Environmental International*. Vol. 1, 247-275.

Laird, D.J. 1997. Wind tunnel testing of shelterbelt effects on dust emissions from swine production facilities. Thesis (M.S.)--Iowa State University.

Leuty, T. 2004. Using shelterbelts to reduce odors associated with livestock production barns. Ontario Ministry of Agriculture and Food. (cited 2/20/04).

Lorimor, J. 2002. Personal Communication. Associate Professor Emeritus, Specialty: Manure management and water quality. Iowa State University Department of Agriculture and Biosystems Engineering.

Malone, G.W. and D. Abbott-Donnelly. 2001. The benefits of planting trees around poultry farms. University of Delaware, College of Agriculture and Natural Resources, Extension Bulletin #159, December 2001.

Malone, G., G. Van Wicklen and S. Collier. 2004 a. Efficacy of Trees to Capture Emissions from Tunnel Ventilated Poultry Houses. Proceedings to International Poultry Scientific Forum. Jan. 27, 2004.

Malone, G., D. Hansen and G. Van Wicklen. 2004 b. Planting Trees around Poultry Farms: A Proactive Environmental Initiative. Proceedings to First World of Agroforestry. Orlando, FL. June 28, 2004.

Mikesell, R.E., K.B. Kephart, L.J. Ressler, and F.K. Willits. 2001. Comparison of neighbor and non-neighbor perceptions of large-scale animal agriculture. International Symposium on Animal Production and Environmental Issues. Oct 3-5, 2001. Research Triangle Park, NC.

Midwest Plan Service. 2002. Outdoor Air Quality. Manure Management Systems Series. MWPS – 18, Section 3.

Purvis, A. K., J. P. Hoehn & F.J. Pierce. 1989. Farmers response to a filter strip program: results from a contingent valuation survey. *Journal of Soil and Water Conservation*. Vol. 44, Num. 5. pp. 501-504.

Takle, E.S. 1983. Climatology of superadiabatic conditions for a rural area. *Journal of Climate and Applied Meteorology*. 22:1129-1132.

Thernelius, S.M. 1997. Wind tunnel testing of odor transportation from swine production facilities. Thesis (M.S.)--Iowa State University, 1997

Tyndall, J.C. and J.P. Colletti. 2001. Air Quality and Shelterbelts: Odor Mitigation and Livestock Production A Literature Review. Final Report. United States Department of Agriculture, National Agroforestry Center, Lincoln, Nebraska. Project Number: 4124-4521-48-3209.

United States Department of Agriculture (USDA). 1996. Economic Research Service, Natural Resources and Environment Division. Manure Management by U.S. Pork Producers in 1992. AREI UPDATES, no. 7, Aug. 1996.

Arkansas Swine Odor Survey

K.W. VanDevender
University of Arkansas, Division of Agriculture - Extension, Little Rock, Arkansas

Swine Odor Survey Participants

Arkansas Pork Producer's Association
National Pork Board
University of Arkansas, Division of Agriculure, Cooperative Extension Service
USDA, Natural Resource Conservation Service
Arkansas Department of Environmental Quality
Arkansas Natural Resources Commission
University of Arkansas, Division of Agriculure, Animal Science Department
Tyson Foods Inc. Swine Division
Cargill Pork
Arkansas Swine Producers
Individuals from the General Public

Background

This survey was conducted at the request of the Arkansas Pork Producer's Association. The objective was to make an unbiased assessment of the odors typically found on swine farms. This information can be used as a guide for future research, demonstration and educational efforts in addressing odor mitigation.

The survey was conducted on 36 randomly selected farms in 7 counties in northwest, westcentral and southwest Arkansas. From September 1996 to June 1997, 1,157 odor measurements were made at 253 locations.

This document is a summary of the final report for the survey. An electronic version of the report can be found at www.aragriculture.org/agengineering/anmanmortmgmt/swineodorsurvey.

Survey Setup

Odor Measurement Teams

The unbiased measurement of odors is complex. People evaluate odors based on their impressions of the strength and unpleasantness of the odor. As this is a subjective process, people perceive odors differently. For this reason, the survey used a team approach. This ensured that an average measurement could be determined at each location. To help ensure unbiased measurements, the team members included individuals from the Cooperative Extension Service, Natural Resource Conservation Service, Tyson, Cargill and the general public. There were seven teams, one for each county.

Odor Measurement Methods

Two different ranking methods were used to make odor measurements. Due to limited funds and time constraints, most of the team "sniffed" the odor and assigned a numeric rank. One team member used a scentometer to assign a numeric rank. A scentometer is an instrument with a series of different sized holes and activated charcoal filters. They allow purified air to be breathed prior to each measurement. The odor ranks are correlated to the ratio of odorous air to purified air. For both methods, the possible odor ranks ranged from "non-detectable" to "strongly offensive."

Table 1. Interpretation of Odor Rank Values

Odor Description	Scentometer Rank	Nasal Rank	Combined Rank*
Non-Detectable	1	1	1
Detectable But Non-Offensive	2	2	2
	3		
Mildly Offensive	4**	3	3
	5**		
Strongly Offensive	6***	4	4
	7***		

*Combined ranks used for combined measurements. The scentometer ranks were scaled prior to combining with nasal ranks.
**Scentometer odor ranks historically associated with the start of complaints.
***Scentometer odor ranks historically associated with serious nuisance odors.

Odor Measurement Process

Each team spent a day every other week collecting measurements. At each farm, the team moved upwind toward the potential odor source. Measurements were made at various distances. At each measurement site, team members recorded their odor measurement, the distance from the odor source, temperature, wind speed, relative humidity and any written comments.

Survey Results

Overall Odor Level Findings

The 250 scentometer and 907 nasal measurements were compared and found to be statistically similar, allowing the values to be combined into a single set of measurements. Of these 1,157 measurements, 85% had non-offensive odor ranks, 45% were made within 1/10 mile from the odor source and 99% percent were made within 1/2 mile. Less than 1% were made at a distance of one mile.

Table 2. Frequency of Odor Rank Measurements

Distance Ranges	Non-Detectable	Detectable But Non-Offensive	Mildly Offensive	Strongly Offensive
All Distances	57%	28%	13%	2%
0.0 to .1 mile	41%	35%	21%	3%
0.1 to .2 mile	52%	36%	11%	1%
0.2 to .3 mile	78%	19%	3%	0%
0.3 to .4 mile	100%	0%	0%	0%
0.4 to .5 mile	89%	5%	5%	1%
0.5 to 1 mile	14%	86%	0%	0%

Observer Trends

Analysis of the data revealed differences in how individuals evaluated odors. The individual with the highest average odor value (2.23) was a company field representative. The individual with the lowest average odor value (1.17) was a volunteer from the general public. Sorting and analyzing the measurements by class (agency, industry or the general public) found no differences in how the classes evaluated odors.

Natural Resources

Odor Source Trends

The two primary sources of odors were swine production facilities and land application of manure. The facilities included the houses, manure storage units and mortality disposal units. Odor measurements from both sources ranged from "non-detectable" to "strongly offensive." The average facilities odor value (1.51) was lower than the average application odor value (2.39).

Distance Trends

Odor intensity decreased as distance increased. There were also differences between the sources of odors. Within 1/10 mile of the source, the average odor value was 1.76 for the facilities and 2.58 for the manure applications. The maximum distance that an offensive odor was recorded was 3/10 mile for facilities and 1/2 mile for manure applications. Statistical tests also revealed that while odor intensity was related to distance, other factors also affected odor intensity.

Animal Population and Manure Storage Trends

Differences in odor measurements were found for the various types of swine production operations and manure storage units. However, there were no easily identifiable trends. The lack of trends was probably due to two factors. First, the facility-based measurements were for odors coming from both the swine houses and manure storage units. Also, the survey included a relatively small number of some types of production systems and manure storage systems. This resulted in a smaller number of measurements for some systems, making it more likely that other factors, such as weather and topography, obscured any trends in odor levels.

Appearance Trends

Farms with better appearance ranks were associated with lower odor measurements. The differences could be due to appearance influencing the assigned odor ranks. Another possibility is that differences in overall farm management result in true differences in odor levels.

Conclusions

- This survey provides information to help quantify existing odor levels typically associated with swine production.
- The small percentage of the odors found to be offensive implies that a significant portion of odor concerns is due to the relatively infrequent occurrence of offensive odors.
- An individual's association with swine production did not influence how odors were evaluated.
- The land application of manure tended to generate more odors than the facilities.
- Distance is a significant factor, but not the only factor, affecting odor intensity. Odor intensity was found to decrease as distances increased.
- Odor management practices will have to consider many factors to be effective. This is supported by the fact that other influencing factors masked odor trends with animal populations or manure storage units.
- Farms with a good overall appearance had lower odor levels than farms with a poor appearance.
- The survey results provide valuable information to guide future research, demonstration and educational efforts addressing odor mitigation.

References

VanDevender, K., 1998. Arkansas Swine Odor Survey. Project Report. University of Arkansas Division of Agriculture. www.aragriculture.org/agengineering/anmanmortmgmt/swineodorsurvey (Febuary 15, 2006)

Acknowledgement

Without the support of the individuals and organizations participating in this survey this project would not have been possible. The Arkansas Pork Producer's Association and the National Pork Board provided the funds to purchase the necessary equipment and supplies. The members of the odor teams contributed their time and effort. The landowners provided access to their farms for making odor measurements. The various agencies and industry groups provided support in the planning, farm selection, and implementation of the project.

Greenhouse Gas Emission Reductions and Carbon Credits from Implementation of Aerobic Manure Treatment Systems in Swine Farms

Matias B. Vanotti[1], Ariel A. Szögi[1] and Carlos A. Vives[2]
[1]US Department of Agriculture, ARS, Coastal Plains Soil, Water, and Plant Research Center
2611 W. Lucas Street, Florence, SC 29501, USA
[2]Agrosuper, Agricola Super Limitada, Rancagua c.c. 333, Chile

Abstract

Trading of carbon and NOx emission reductions is an attractive approach to help producers implement cleaner treatment technologies to replace current anaerobic lagoons. Our objectives were to determine greenhouse gas (GHG) emission reductions from implementation of aerobic technology (Supersoil project) in North Carolina swine farms. Emission reductions were determined using approved methodology in conjunction with monitoring information collected during full-scale demonstration of the new treatment system in a 4,360-head swine operation in North Carolina. Emission sources for the project and baseline manure management system were methane emissions from the decomposition of manure under anaerobic conditions and nitrous oxide emissions during storage and handling of manure in the manure management system. Emission reductions resulted from the difference between total project and baseline emissions. The project activity included an on-farm wastewater treatment system consisting of liquid-solid separation, treatment of the separated liquid using aerobic biological N removal, chemical disinfection and soluble P removal using lime. The project activity was completed with a centralized facility that used aerobic composting to process the separated solids. Replacement of the lagoon technology with the cleaner aerobic technology reduced GHG emissions 98.9%, from 4,712 Tonnes of carbon dioxide equivalents (CO2-eq) to 50 Tonnes CO2-eq/year. Total net emission reductions by the project activity in the 4,360-head finishing operation were 4,632.8 Tonnes CO2-eq per year. The dollar value from implementation of the Supersoil project in this swine farm was $9,960.54/year. This translates into a direct economic benefit to the producer of $0.91 per finished pig. Thus, GHG emission reductions and credits can help compensate for the higher installation cost of cleaner aerobic technologies and facilitate producer adoption of environmentally superior technologies to replace current anaerobic lagoons in North Carolina.

Introduction

Anaerobic lagoons are widely used to treat and store liquid manure from confined swine production facilities (Barker, 1996). Environmental and health concerns with the lagoon technology include emissions of ammonia (Aneja et al., 2000; Szogi et al., 2006), odors (Loughrin et al., 2006), pathogens (Sobsey et al., 2001), and water quality deterioration (Mallin, 2000). Widespread objection to the use of anaerobic lagoons for swine manure treatment in North Carolina prompted a state government-industry framework to search for alternative technologies that directly eliminate anaerobic lagoons as a method of treatment. In July 2000, the Attorney General of North Carolina reached an agreement with Smithfield Foods, Inc. and its subsidiaries (the largest hog producing companies in the USA) to develop and demonstrate environmentally superior waste management technologies for implementation onto farms located in North Carolina that are owned by these companies. In October 2000, the Attorney General reached a similar agreement with Premium Standard Farms, the second largest pork producer in the USA. The agreement defines an environmentally superior technology (EST) as any technology, or combination of technologies, that (1) is permittable by the appropriate governmental authority; (2) is determined to be technically, operationally, and economically feasible; and (3) meets the following five environmental performance standards (Williams, 2005):

1. Eliminate the discharge of animal waste to surface waters and groundwater through direct discharge, seepage, or runoff;
2. Substantially eliminate atmospheric emissions of ammonia;
3. Substantially eliminate the emission of odor that is detectable beyond the boundaries of the swine farm;
4. Substantially eliminate the release of disease-transmitting vectors and airborne pathogens;

5. Substantially eliminate nutrient and heavy metal contamination of soil and groundwater.

Selection of EST candidates to undergo performance verification involved a request of proposals and competitive review by the Agreement's Designee and a Panel representing government, environmental and community interests, the companies, and individuals with expertise in animal waste management, environmental science and public health, and economics and business management. This process yielded 18 technologies candidates from about 100 submitted projects. Subsequently, the selected technologies completed design, permitting, construction, startup, and performance verification under steady-state operational conditions. In July 2005, five of the 18 technologies tested were shown to be capable of meeting the environmental performance criteria necessary for the technologies to be considered environmentally superior (Williams, 2005). Only one of the technologies selected treated the entire waste stream from a swine farm (figure 1). The system was constructed and operated by Super Soil Systems USA of Clinton, NC, and the technology demonstration project was identified as "Supersoil Project." This on-farm technology used liquid-solid separation and aerobic processes to treat both the separated liquid and solids. It was developed to replace anaerobic lagoon technology commonly used in the USA to treat swine waste (Vanotti et al., 2005).

Figure 1. Full-scale wastewater treatment system (project activity) that replaced the anaerobic swine lagoon (background), Duplin County, North Carolina

The system had two components: 1) an on-farm wastewater treatment system consisting of liquid-solid separation using flocculants and screens, treatment of the separated liquid using aerobic biological N removal, and chemical disinfection and soluble P removal using lime, and 2) a centralized solids processing facility where separated manure solids were combined with cotton gin residue and aerobically composted to reduce the wastes into stable humus used to manufacture peat substitutes used in potting soil, soil amendments, and organic fertilizers. The on-farm system removed more than 97% of the suspended solids from wastewater. It removed 95% of total P in the liquid, 99% of its ammonia, more than 99% of its biochemical oxygen demand and odor-causing components, and produced a disinfected liquid effluent (Vanotti et al., 2006). In addition, the old wastewater lagoon was converted into clean water that substantially reduced odor and ammonia emissions (Loughrin et al., 2006; Szogi et al., 2006). The centralized facility produced quality composts that conserved 96.5% of the nitrogen into a stabilized product that met Class A biosolids standards due to high pathogen reduction (Vanotti, 2005).

Although this clean technology was determined to be technically and operationally feasible and it was able to meet the strict technical environmental performance standards of an environmentally superior technology (EST), a contingency project was subsequently planned to demonstrate a second-generation, lower-cost version of the treatment system (i.e., annual cost should be similar to baseline lagoon technology) to meet unconditional EST status. Second-generation technology development involved simplification of processes and operation based on lessons learned during testing of the first-generation system.

Capital investment is the most important barrier for widespread adoption of cleaner treatment technology due to higher costs involved compared to the baseline lagoon technology. On the other hand, proven environmental benefits from implementation of the new superior technologies are often difficult to translate in terms of direct economic benefits that can offset the investment barrier. Fortunately, new programs are being created on global reduction of anthropogenic emissions of greenhouse gases (GHG) that can help compensate for the higher installation cost of the cleaner technologies, and therefore favor technology adoption by producers. Such a program was recently implemented by Agricola Super Limitada (Agrosuper), the largest swine production company in Chile. The company initiated a voluntary adoption of advanced waste management systems (anaerobic and aerobic treatment of manure); implementation of the more expensive technology was greatly influenced by the adoption of the Kyoto Protocol and the Clean Development Mechanism. As a result, advanced technologies are being phased in gradually in all of Agrosuper's swine production units to replace the existing anaerobic lagoon technology. The company used revenues from the sale of Certified Emission Reductions (CERs) to partially finance the advanced waste management systems. This voluntary adoption case is significant to North Carolina because the company is phasing out lagoon technology that was implemented years ago using the North Carolina traditional anaerobic lagoon treatment model. To accomplish this purpose, Agrosuper developed a project activity at a 118,800 finishing swine facility in Chile that led to an approved UNFCCC/CCNUCC methodology (AM0006, 2004). The advantage of this methodology is that it considers aerobic components in addition to anaerobic digesters and flaring that are the focus of other approved methods for quantification of GHG emission reduction in animal manure systems (i.e., AM0016). Thus, the methodology is very suitable for quantification of GHG emission reductions in the Supersoil project which relies heavily on aerobic processes to treat the manure.

Our objectives were to determine greenhouse gas (GHG) emission reductions from implementation of the cleaner aerobic technology (Supersoil project) in North Carolina swine farms compared to the current anaerobic lagoon system (baseline scenario). GHG emission reductions were determined using approved methodology AM0006 in conjunction with monitoring information collected during full-scale demonstration of the treatment system.

Materials and Methods

The baseline activity was the traditional anaerobic lagoon–sprayfield technology for a farm with 4,360-head finishing pigs in North Carolina. The project activity consisted of the implemented advanced system (Supersoil project) in an identical farm. Determination of GHG emission reductions by the environmentally superior technology was made using approved methodology described in AM0006 (2004).

The AM0006 methodology includes the following emission sources for the project and baseline manure management system: 1) Methane (CH_4) emissions from the decomposition of manure under anaerobic conditions, and 2) Nitrous oxide (N_2O) emissions during storage and handling of manure in the manure management system. Baseline and project boundaries are shown in figures 2 and 3, respectively. Greenhouse gas emissions included in the boundary are calculated separately for the project and the baseline manure management system, using the same methodological approach. Emission reductions are the difference between total project and baseline emissions.

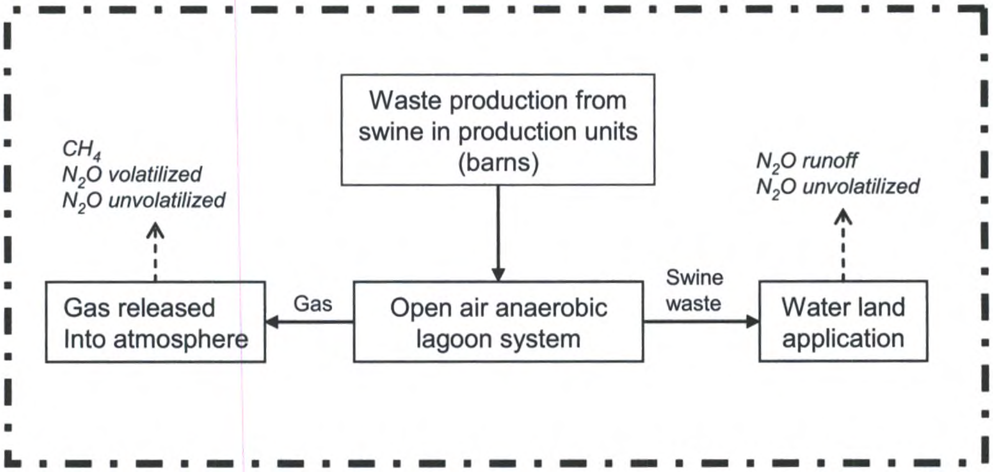


Figure 2. Baseline scenario boundary

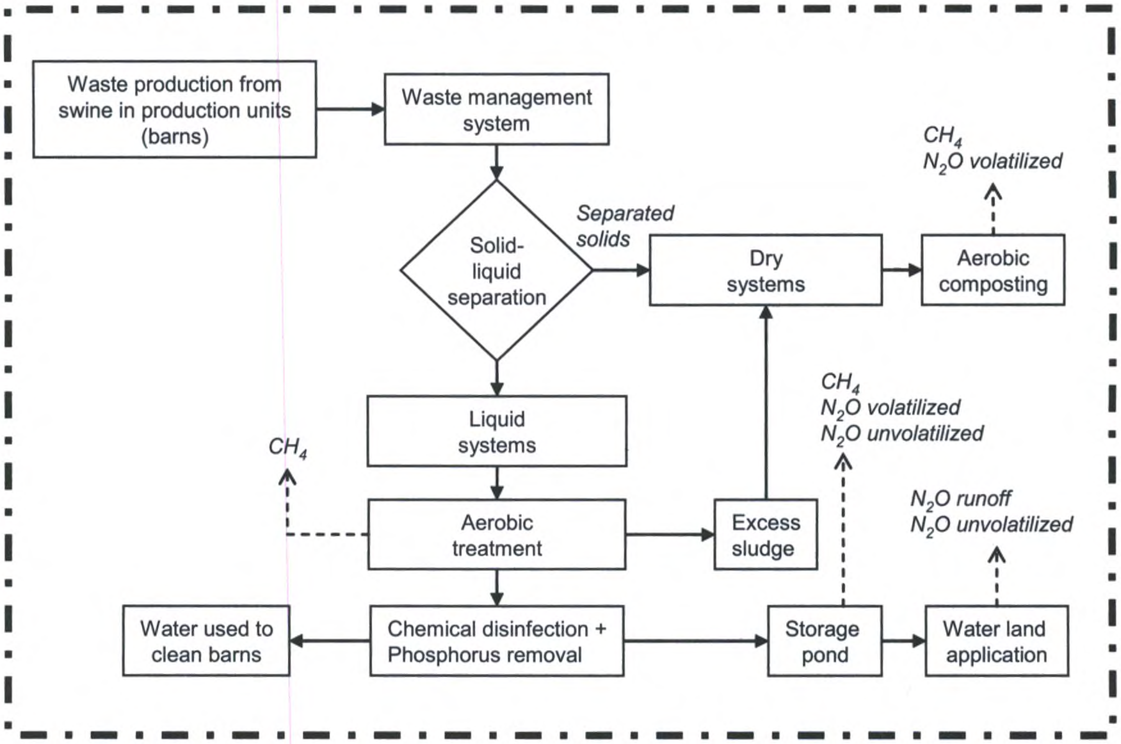


Figure 3. Project activity boundary

Emission factors used for each treatment stage were accepted values that are provided in the 1996 Revised IPCC Guidelines (IPCC, 1996) and in the IPPC Good Practice Guide (IPPC, 2000). Monitoring data and site specific information were obtained during full-scale project activity demonstration at Goshen Ridge Farm (on-farm treatment) near Mount Olive, Duplin County, North Carolina (Vanotti et al., 2006) and at Hickory Grove farm (composting facility) near Clinton, Sampson County, North Carolina (Vanotti, 2005). Total volatile solids (VS) and the nitrogen supplied to the manure management system were determined by the excretion rates of VS and N and the monitored livestock populations. Average pig weight (two years in six barns) was 73.08 kg per head. A partition variable (FracLIQUID) was created to divert the excreted

VS and N into the liquid system (figure 3). FracLIQUID was determined based on monitored BOD_5 and TN before and after solid-liquid separation. The difference (FracSOLID = 1 – FracLIQUID) determined the amount of VS or TN that was diverted into the dry system. For second and subsequent stages, methane emissions were calculated based on the measurement of the monitored BOD_5 and the quantity of manure flowing to that treatment stage (Option A in method AM0006). The BOD_5 was adjusted using monitored water temperature and the Van't-Hoff-Arrhenius relationship. Similarly, emissions of N_2O in second and subsequent stages were calculated based on measurements of the N content in the manure flowing to that treatment stage and monitored flow rates of the manure.

Emission reductions of CH_4 and N_2O were expressed in terms of CO_2 equivalents using approved Global Warming Potentials (21 for CH_4 and 310 for N_2O). Direct economic benefits from emission reductions were determined using current trading value (\$2.15 per Ton of CO_2) at the Chicago Climate Exchange (www.chicagoclimatex.com).

Results and Discussion

A total of 4,712 Tonnes of CO_2-eq were generated in a year by the baseline scenario (anaerobic lagoon–sprayfield technology) in the 4,230-head finishing operation in NC (table 1). Most (85.8%) of the GHG emissions were due to methane (CH_4) produced during anaerobic digestion in the open lagoons, and the remainder (14.2%) due to nitrous oxide (N_2O) emissions, mostly emitted during land application of the digested liquid. In contrast, implementation of the project activity (Supersoil project) on the same farm generated only 50 Tonnes CO_2-eq during the same one-year period that resulted in a 98.9% decrease in GHG emissions (table 2). Generation of methane in the project activity was < 0.5% of that produced by the baseline, and generation of N_2O in the project activity was 5% of that generated by the baseline.

Solid-liquid separation in the project activity diverted 65.6% of the VS and 39.8% of the TN contained in raw manure into the dry system (fig. 3). VS separation efficiencies of only 7% are typical for swine manure that goes through screening without flocculation treatment (Vanotti et al., 2002). However, high separation efficiencies in the project activity were obtained using PAM flocculation. Thus, the amount of VS diverted to dry and liquid systems is technology dependent and should be corrected for specific solid-liquid separation technology using actual monitoring information.

Table 1. Detailed baseline emissions for 4360-head finishing swine operation using anaerobic lagoon technology at Goshen Farm, Duplin Co., NC.

Emissions Source [1]	Emissions (Tonnes CO_2-eq per year) [2]
Lagoon CH_4	4,044.50
Lagoon N_2O (volatilized)	75.67
Lagoon N_2O (unvolatilized)	30.27
Land Application N_2O (unvolatilized)	278.35
Land Application N_2O	183.64
Total Baseline	4,712.43

[1] Baseline scenario boundary and emission sources shown in figure 2.

[2] Carbon dioxide (CO_2) equivalents. Global warming potential of methane (CH_4) and nitrous oxide (N_20) are 21 and 310, respectively.

Total annual emission reductions due to the project activity were calculated from the sum of CH_4 and N_2O annual emission reductions adjusted for leakage effects due to changes in electricity consumption (table 3). Electricity consumption was small (29.7 Tonnes CO_2-eq) compared to the emission reductions (4662.5 Tonnes CO_2-eq) by the project activity, and for this reason, the AM0006 methodology considers that electricity consumption by aerobic treatment should not be considered in the overall net reduction calculations. Nevertheless, we included this amount to add conservativeness in our GHG emission reduction determinations. Total net emission reductions by the project activity in the 4,360-head finishing operation were 4,632.81 Tonnes CO_2-eq per year (table 3). Implementation of aerobic systems is more advantageous than anaerobic systems in terms of carbon credits. For example, the project activity implemented by Agrosuper at their 118,800 swine operation in Chile reduced annual GHG emissions by 81,026 Tonnes CO_2-eq using anaerobic digester and flaring to replace anaerobic lagoon technology (baseline). In a second phase of the same project, they further reduced annual GHG emissions to a total of 116,993 Tonnes CO_2-eq with the installation of aerobic post-treatment of the liquid before land application.

Table 2. Detailed project emissions for 4360-head finishing swine operation using aerobic manure treatment system (Supersoil project) at Goshen Farm, Duplin Co., NC and aerobic composting of separated solids at centralized facility.

Emissions Source [1]	Emissions (Tonnes CO_2-eq per year) [2]
Aerobic Treatment of Separated Liquid	1.55
Storage Pond CH_4	0.30
Storage Pond N_2O (volatilized)	9.06
Storage Pond N_2O (unvolatilized)	0.90
Land Application N_2O (unvolatilized)	11.32
Land Application N_2O (runoff)	6.79
Aerobic Composting of Solids CH_4	14.71
Aerobic Composting of Solids N_2O	5.27
Total Project Activity	49.90

[1] Project activity boundary and emission sources shown in figure 3.
[2] Carbon dioxide (CO_2) equivalents. Global warming potential of methane (CH_4) and nitrous oxide (N_20) are 21 and 310, respectively.

Table 3. Overall results – Emission reductions per annum and dollar value for the implementation of the project activity using aerobic treatment system in the 4,360-head finishing swine operation in North Carolina.

CH_4 emission reductions (ER_{CH4}) due to project activity [1]	4,027.94 T CO_2-eq/year
N_2O emission reductions (ER_{N2O}) due to project activity [2]	634.59 T CO_2-eq/year
Leakage effect (L) from electricity consumption [3]	29.72 T CO_2-eq/year
Total net emission reductions (ER) due to project activity [4]	4,632.81 T CO_2-eq/year
Value of net emission reductions for 4,360-head farm [5]	$ 9,960.54 /year
Value of net emission reductions for each market pig produced [6]	$ 0.91 /finished pig

[1] Amount of CH_4 that would be emitted to the atmosphere during a crediting period of one year in the absence of the project activity (table 1) minus the amount of CH_4 emitted by the project activity in the same period (table 2), expressed in tons of CO_2 equivalents.
[2] Amount of N_2O that would be emitted to the atmosphere during a crediting period of one year in the absence of the project activity (table 1) minus the amount of N_2O emitted by the project activity in the same period (table 2), expressed in tons of CO_2 equivalents.
[3] Changes in electricity demand due to project activity (403.9 kwh/d x 365 d), expressed in terms of tons of CO_2 equivalents.
[4] Total annual emission reductions of the project are the sum of CH_4 and N_2O annual emission reductions adjusted for leakage effects ($ER = ER_{CH4} + ER_{N2O} - L$).
[5] Calculation uses current trading value of $2.15 per ton of CO_2 at the Chicago Climate Exchange (CCX) (March 2, 2006).
[6] Calculation uses actual turnover rate of 2.5 pigs/year monitored at Goshen Ridge farm. Thus, a 4,360-head farm produces 10,900 market pigs per year (0.91 = $9,960.54/10,900 finished pigs).

The dollar value from implementation of the Supersoil project in this farm was $9,960.54/year. This translates into an economic benefit of $0.91 per finished pig (table 3). We also projected these results to other farm sizes ranging from 4,000 to 12,000-head typically found in North Carolina and to a scenario of widespread adoption of the cleaner technology by most of the swine farms in North Carolina. Results of these calculations shown in table 4 indicate that implementation of aerobic systems can represent substantial direct economic benefits to swine producers in North Carolina. These benefits represent an income range from about $9,100/year to $27,500/year that can greatly help finance the installation cost of the environmentally superior technologies.

Table 4. Potential benefits from sale of GHG emision reduction credits due to installation of aerobic manure treatment systems (Supersoil project activity) on swine farms in North Carolina.

Farming scenario	Emission reductions (Tonnes CO_2-eq per year) [1]	Total value ($/year)
4,000-head farm	4,250	9,138
6,000-head farm	6,275	13,491
8,000-head farm	8,501	18,277
10,000-head farm	10,626	22,846
12,000-head farm	12,751	27,415
10,000,000 swine in North Carolina	10,625,711	22,845,279

[1] Projected amount of emission reductions based on results obtained by implemenation of project activity in the 4,360-head finishing facility at Goshen Ridge farm.

[2] Calculation of total dollar value uses projected annual emission reductions and current trading value of $2.15 per ton of CO_2 at the Chicago Climate Exchange (CCX) (March 2, 2006).

Conclusions

Our objectives were to determine greenhouse gas (GHG) emission reductions from implementation of environmentally superior technology in North Carolina swine farms. Emission reductions were determined using approved UNFCCC/CCNUCC methodology AM0006 that is appropriate for quantification of GHG emission reductions in manure management systems that utilize aerobic processes. We found that replacement of the lagoon technology with the cleaner aerobic technology in a 4,360-head swine operation reduced GHG emissions 98.9%, from 4,712 Tonnes of carbon dioxide (CO_2-eq) to 50 Tonnes CO_2-eq/year. The dollar value from implementation of the cleaner technology was $9,960.54/year. This translates into a direct economic benefit to the producer of $0.91 per finished pig. Therefore, GHG emission reductions can be an important component to facilitate producer adoption of environmentally superior technologies to replace current anaerobic lagoons in North Carolina.

Acknowledgements

This research was part of USDA-ARS National Program 206: Manure and By-product Utilization; CRIS Project 6657-13630-003-00D "Innovative Animal Manure Treatment Technologies for Enhanced Environmental Quality." Technology verification study of the project activity was partially funded by North Carolina State University and its sponsor Smithfield Foods, Inc. through USDA Agreement 58-6657-2-202, NCSU Subcontract #2002-0478-02. The authors are grateful to Super Soil Systems USA, Clinton, NC for help during project design and monitoring, and to Premium Standard Farms for providing farm records on animal weight and production. Mention of trade names or commercial products in this publication is solely for the purpose of providing specific information and does not imply recommendation or endorsement by the U.S. Department of Agriculture.

References

Aneja, V.P., J.P. Chauhan, and J.T. Walker. 2000. Characterization of atmospheric ammonia emissions from swine waste storage and treatment lagoons. *J. Geophys. Res.* 105(D9):11535-11545.

AM0006. 2004. "GHG emission reductions from manure management systems." Approved baseline methodology AM0006, United Nations Framework Convention on Climate Change (UNFCCC).

Barker, J.C. 1996. Lagoon design and management for livestock waste treatment and storage. North Carolina Cooperative Extension Service. Pub. No. EBAE 103-83.

IPCC. 1996. Revised 1996 IPCC guidelines for national greenhouse gas inventories. Intergovernmental Panel on Climate Change, National Greenhouse Gas Inventories Programme (IPCC-NGGIP). United Nations Framework Convention on Climate Change (UNFCCC).

IPCC. 2000. Good practice guidance and uncertainty management in national greenhouse gas inventories. Intergovernmental Panel on Climate Change, National Greenhouse Gas Inventories Programme (IPCC-NGGIP). United Nations Framework Convention on Climate Change (UNFCCC).

Loughrin, J.H., A.A. Szogi, and M.B. Vanotti. 2006. Reduction of malodorous compounds from a treated swine anaerobic lagoon. J. Environ. Qual.35(1):194-199

Mallin, M.A. 2000. Impacts of industrial animal production on rivers and estuaries. American Scientist 88(1):26-37.

Sobsey, M.D., L.A.Khatib, V.R. Hill, E. Alocilja, and S. Pillai. 2001. Pathogens in animal wastes and the impacts of waste management practices on their survival, transport and fate. In: White Papers on animal agriculture and the environment. MidWest Plan Service (MWPS), Iowa State University, Ames, Iowa, (Chapter 17).

Szogi, A.A., M.B.Vanotti, and A.E. Stansbery. 2006. Reduction of ammonia emissions from treated anaerobic swine lagoons. Transactions of the ASABE (in press)

Vanotti, M.B., D.M.C. Rashash, and P.G. Hunt. 2002. Solid-liquid separation of flushed swine manure with PAM: Effect of wastewater strength. Transactions of the ASAE 45(6):1959-1969.

Vanotti, M.B., A.A. Szogi, and P.G. Hunt. 2005. Wastewater treatment system. U.S. Patent 6,893,567. U.S. Patent Office.

Vanotti, M. B. 2005. Evaluation of Environmentally Superior Technology: Swine waste treatment system for elimination of lagoons, reduced environmental impact, and improved water quality (Centralized composting unit). Phase II: Final Report for Technology Determination per Agreements between NC Attorney General & Smithfield Foods, Premium Standard Farms, and Frontline Farmers. July 25, 2005. Available at: http://www.cals.ncsu.edu/waste_mgt/smithfield_projects/phase2report05/reports/A1.pdf

Vanotti, M.B., A.A. Szogi, P.G. Hunt, P.D. Millner, and F.J. Humenik. 2006. Development of environmentally superior treatment systems to replace anaerobic swine lagoons in the USA. *Bioresource Technol.* (in press).

Williams, C.M. 2005. Development of Environmentally Superior Technologies. Phase II Report for Technology Determination per Agreements Between the Attorney General of North Carolina and Smithfield Foods, Premium Standard Farms and Frontline Farmers. July 25, 2005. Available at: http://www.cals.ncsu.edu/waste_mgt/smithfield_projects/phase2report05/front.pdf

Impact of Air Pollution on Agricultural Crops in India

C.K. Varshney

Jawaharlal Nehru University, School of Environmental Sciences, New Delhi, 1110067, India

Abstract

Effect of ground level ozone on plants is attracting much attention as it has been shown to reduce the crop yield. Buildup of ground level ozone is viewed as a growing threat to food security. Studies carried out at New Delhi have shown that the ambient ozone concentration varies between 20 and 273 ug m^{-3} and the WHO one hour ozone standard is violated on many occasions. Values of ground level ozone reported from nine other widely separated stations in the country suggest that the build up of ground level ozone is fairly widespread in the country. Growing industrialization, urbanization and rapidly rising consumption of fossil fuels will further contribute to the build up of ground level ozone in the country. Investigations were undertaken to study the effect of ozone on four important India crop species viz., *Triticum*, *Phaseolus*, *Brassica* and *Spinacia* was under taken at Delhi. Plants were exposed to ambient levels of ozone and at different sites as well as plants treated with EDU for comparison. Measurements were made on growth, biomass, seed set, seed weight and yield loss was determined. The average yield loss in these four crop species was 8.57, 8.58, 14.38, and 14.37 per cent, respectively. The paper provides evidence to show that the ambient level of ground level ozone can potentially undermine the crop yield. In future, air pollution especially ground level ozone is likely to increase and may become a serious challenge for increasing agricultural yield in the country.

The ClearSky Field-Burning Decision Support System

J. K. Vaughan[1], K. Heitkamp[1], R. Jain[2], B. K. Lamb[1]
[1]Laboratory for Atmospheric Research, Washington State University, Pullman, Washington
[2]Golder Associates Ltd., Calgary, Alberta, Canada

Abstract

The ClearSky agricultural smoke dispersion modeling system was developed, beginning in 2002, as a decision support system for agricultural-burning smoke management in northern Idaho and eastern Washington. The ClearSky system features: 1) use of data from state burn-permitting programs to locate and characterize sources, 2) a web application giving authorized users the ability to submit potential burn scenarios for modeling, and 3) web-served graphics animations displaying burn scenario simulation air quality results. ClearSky is a NW-AIRQUEST (Northwest International Air Quality, Environmental Science and Technology Consortium) project and has developed in collaboration with the US Forest Service BlueSky/RAINS project for management of prescribed forest burning. The ClearSky modeling system uses a University of Washington MM5 forecast, at 4-km grid spacing, processed through CALMET to drive CALPUFF to simulate smoke dispersion from agricultural field burning. The emissions are simulated using a buoyant line source to represent an active flame front and a buoyant area source to represent the smoldering portion of a field. Parameters for the emissions and plume-rise modules were obtained from data collected during recent field studies and from sensitivity studies of the CALPUFF plume-rise modules. ClearSky was operated on a daily basis during the 2002 through 2005 burn seasons. An evaluation of the ClearSky system was conducted at the end of the 2002 season by re-running each burn day using actual burn data. Performance of the modeling system was analyzed by comparing 1) observed and MM5-predicted meteorology and 2) observed and ClearSky-predicted $PM_{2.5}$ concentrations at the monitoring stations revealing a notable vulnerability to meteorological forecast errors. Consequently, a new Ensemble ClearSky System is being developed which utilizes a suite of parallel, conterminous MM5 meteorological forecasts to generate a probabilistic forecast of plume impacts.

Introduction

Smoke from the burning of agricultural residue in the field or orchard is a major source of particulate matter in the Inland Northwest region comprising the Columbia Plateau areas of Washington, Idaho and Oregon. Field burning, Figure 1, is a common treatment both for the removal of cereal grain stubble and the residue associated with harvest of annual Kentucky bluegrass (KBG). In the latter crop, burning is generally thought to be an economical treatment to reduce disease and pests and stimulate new growth (Hardison, 1980), and hence is deemed a virtual necessity, although more nuanced and perhaps dissenting views can also be found (Chastain *et al.*, 1998). Rural population growth and the concomitant influx of people without a predisposition to suffer quietly the nuisance and/or health threats associated with agricultural burning have resulted in a politically and legally contentious situation wherein the economic interest of the farmer in being able to burn is being pitting against the health interests of residents. Consequently, the states of both Idaho and Washington as well as the Coeur d'Alene and Nez Perce tribal governments have environmental management programs charged with mitigating the potential nuisance and health threat associated with field burning; ClearSky is a decision support system that has been developed in response to requests from the state governments and tribal authorities to assist them in their environmental management duties; ClearSky should not be confused with the similarly named US EPA Clear Skies program. Basically, ClearSky is an automated Lagrangian puff dispersion modeling system that runs nightly during the burn season, resulting in predictions of $PM_{2.5}$ (atmospheric aerosol consisting of particulate matter of 2.5 μm aerodynamic diameter and smaller).

Figure 1. Wheat field stubble being burned in Eastern Washington

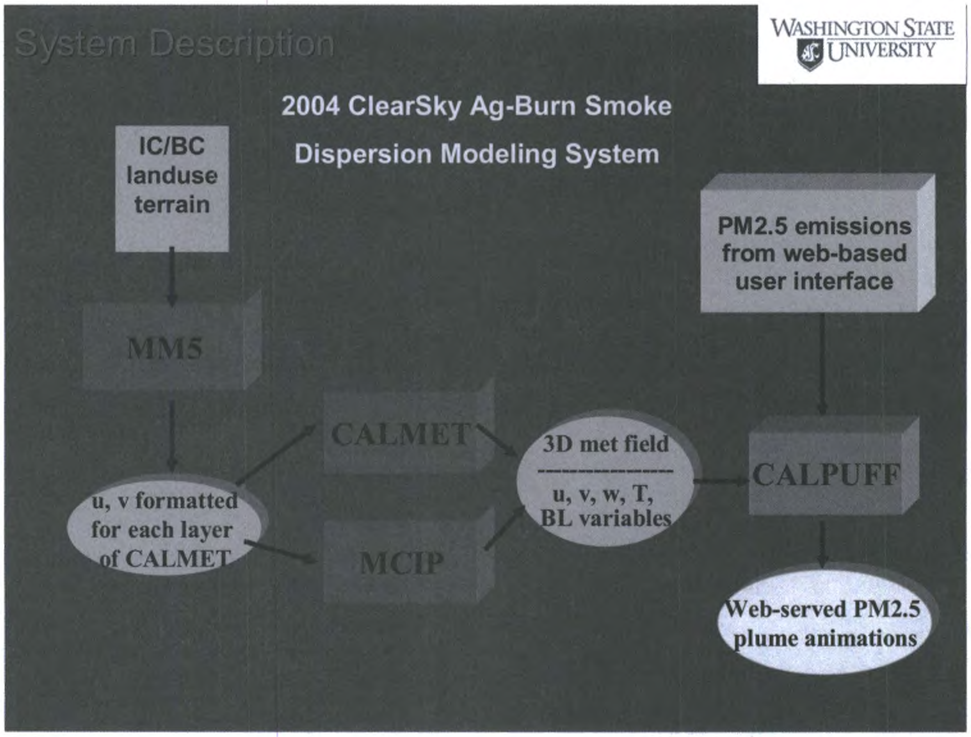
The ClearSky Decision Support System

As shown in Figure 2, the ClearSky system requires meteorological and emissions scenarios to run the CALPUFF Lagrangian puff dispersion model (Scire *et al.*, 1999). The required meteorology is provided by the University of Washington, Atmospheric Sciences Department, Mesoscale Meteorology group's nightly forecast operation (Mass *et al.*, 2003) generated using the Penn State Mesoscale Meteorological Model version 5 (MM5). The MM5 meteorology is reprocessed using CALMET (Scire *et al.*, 2000) into a format compatible with the CALPUFF model. Emissions for the CALPUFF dispersion model are generated based on field burning scenarios for each of the jurisdictions served by the system, and based on field studies of $PM_{2.5}$ emissions (Air Sciences Inc., 2003; Air Sciences Inc., 2004). These field burning scenarios are either provided on the previous day by field burning coordinators (state or tribal employees who in effect issue field burning permits) or are taken from predetermined default scenarios. Coordinators generate field burning scenarios by accessing a web-based tool that is informed by a (state or tribal) database of fields available and eligible for burning. Figure 3 shows a screen-shot of one of the scenario generator screens for selection of fields for simulated burning for the next day.

Results and Discussion

While ClearSky users value the system as part of a suite of tools they utilize in making decision as to whether to permit or recommend field burning, numerous opportunities exist to improve the ClearSky system. The ClearSky smoke ($PM_{2.5}$) animations provide an easily comprehended expression of the implications of the forecast meteorology, integrating the effects of vertical atmospheric structure and 3-d wind fields that evolve with time, and have become popular with regional smoke coordinators. However, when evaluated in detail, cases emerge where small errors in predicted winds result in CALPUFF predicting that plumes will miss receptors that they will in actuality hit. (Receptors are population centers with a $PM_{2.5}$ monitor.) Figure 4 shows an example of ClearSky CALPUFF results where a plume is predicted to miss the town of Athol, ID; but the monitor in Athol recorded a definite plume hit, due to relatively small errors in predicted winds. This demonstration of extreme dependency on winds has motivated experimentation with a new ensemble version of the ClearSky system.

The theory of ensemble meteorological modeling asserts that the variability in a parameter (e.g. wind speed) among several different simulations generated with similarly skillful models should represent the uncertainty in that variable. Ensemble average parameters have been demonstrated to be better predictors, statistically, than the parameter as predicted by any individual model (Delle Monache and Stull, 2003; Grimit, 2004). Based on these results, and using the University of Washington Mesoscale Ensemble forecast of 17 members, an ensemble ClearSky system is under development.



System Description

Washington State University

2004 ClearSky Ag-Burn Smoke Dispersion Modeling System

IC/BC landuse terrain

MM5

u, v formatted for each layer of CALMET

CALMET

MCIP

3D met field

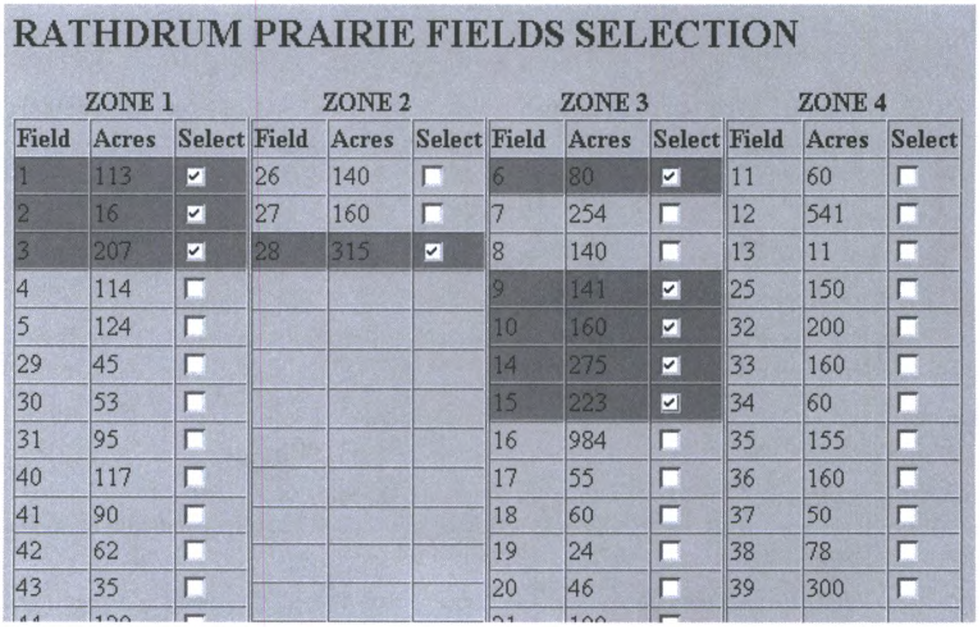
u, v, w, T, BL variables

PM2.5 emissions from web-based user interface

CALPUFF

Web-served PM2.5 plume animations

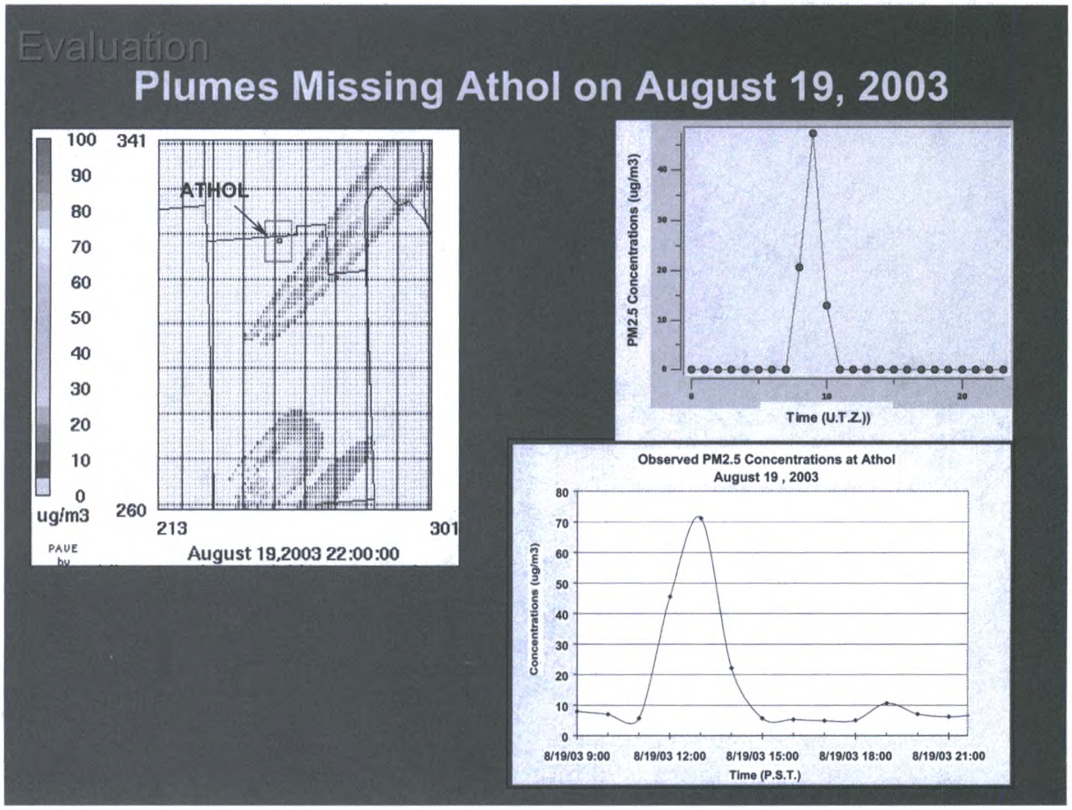
Figure 2. Schematic of ClearSky Decision Support System. Initial and boundary conditions for MM5 are taken from a GFS global weather model run. MCIP passes through MM5 results for boundary layer variables for recombination with CALMET 3-d winds for CALPUFF use. CALPUFF results are visualized and served via the project web-site for registered users.



RATHDRUM PRAIRIE FIELDS SELECTION

ZONE 1			ZONE 2			ZONE 3			ZONE 4		
Field	Acres	Select	Field	Acres	Select	Field	Acres	Select	Field	Acres	Select
1	113	☑	26	140	☐	6	80	☑	11	60	☐
2	16	☑	27	160	☐	7	254	☐	12	541	☐
3	207	☑	28	315	☑	8	140	☐	13	11	☐
4	114	☐				9	141	☑	25	150	☐
5	124	☐				10	160	☑	32	200	☐
29	45	☐				14	275	☑	33	160	☐
30	53	☐				15	223	☑	34	60	☐
31	95	☐				16	984	☐	35	155	☐
40	117	☐				17	55	☐	36	160	☐
41	90	☐				18	60	☐	37	50	☐
42	62	☐				19	24	☐	38	78	☐
43	35	☐				20	46	☐	39	300	☐

Figure 3. Screenshot of a field-burning scenario generator screen. The Rathdrum Prairie version is relatively simple; other scenario generators, for other areas, offer the burn coordinators control over ignition time.



Evaluation

Plumes Missing Athol on August 19, 2003

Figure 4. CALPUFF results show (upper left) a smoke plume missing Athol, ID, but the monitor (lower right) recording a plume hit. Modeled plume maximum and structure (upper right) are lower but similar to the Athol monitor results.

Conclusions

ClearSky has attained a high degree of functionality and a fair level of acceptance among the people responsible for managing field burning smoke hazards in the Inland Northwest, yet technical evaluation finds insufficient accuracy in $PM_{2.5}$ predictions. Since the errors are largely associated with errors in MM/CALMET forecast winds, a new ensemble ClearSky system is being developed for evaluation.

References

Air Sciences Inc., 2003. Cereal-grain residue open field burning study. Report for: Washington Dep. Of ecology, Washington Assoc. of Wheat Growers, and U.S. Environ. Protection Agency, Region 10. Project No. 152-02.

Air Sciences Inc., 2004: Quantifying post-emissions from bluegrass seed production field burning. http://www.ecy.wa.gov/programs/air/pdfs/bluegrass_final_report.pdf

Chastain, T.G., Young, W.C. III, Garbacik, C.J., Meints, P.D., Silberstein, T.B. 1998, Residue management and stand age does not affect seed quality in grass seed crops. http://cropandsoil.oregonstate.edu/seed-ext/Pub/1998/SR9908.html.

Grimit, E. P., 2004: Probabilistic Mesoscale Forecast Error Prediction Using Short-Range Ensembles. Ph.D. dissertation, University of Washington, Seattle, WA, 146 pp.

Delle Monache, L. D. and R. B. Stull, 2003: An ensemble air-quality forecast over western Europe during an ozone episode. *Atmospheric Environment*, **37**, 3469-3474.

Hardison, J.R., 1980. Role of fire for disease control in grass seed production. Plant Disease 64, 641-645.

Mass, C. F., Albright, M., Ovens, D., Steed, R., Grimit, E., Eckel, T., Lamb, B., Vaughan, J., Westrick, K., Storck, P., Colman, B., Hill, C., Maykut, N., Gilroy, M., Ferguson, S. A., Yetter, J., Sierchio, J. M., Bowman, C., Stender, R., Wilson, R. and Brown, W., 2003: Regional Environmental Prediction over the Pacific Northwest, Bull. Amer. Meteor. Soc., 84,1353-1366.

Scire, J.S., Robe, F.R., Fernau, M.E., Yamartino, R.J., 2000. A users guide for the CALMET meteorological model (Version 5). Earth Tech Inc.

Scire, J.S., Strimaitis, D.G., Yamartino, R.J., 1999. A users guide for the CALPUFF dispersion model (Version 5). Earth Tech Inc.

Nitrogen Losses from Organic Housing Systems for Fattening Pigs

Hans von Wachenfelt and Knut-Håkan Jeppsson
Department of Agricultural Biosystems and Technology, Swedish University of Agricultural Sciences, P.O. Box 86, S- 230 53 Alnarp, Sweden. Phone: +46 40 415485. Fax: +46 40 415475.
E-mail: Hans.von.Wachenfelt@jbt.slu.se

Abstract

In Sweden, organic pigs are generally produced according to the rules of the economic association KRAV. According to these rules, the pigs shall be held outdoors on pasture during summer period, whereas indoor housing with access to an outdoor pen is permitted only during winter period. The EU permitted systems for organic slaughter pig production with indoor housing and year round access to an outdoor pen with solid flooring and without additional access to plant covered soil are not common in Sweden. The aim of this project is to analyse and describe the consequences different production systems will have on manure management and nutrient balances for organic pig production. A research facility with four housing alternatives for organic fattening pig production has been built at the research farm Odarslöv situated in the south of Sweden. Inside, the housing alternatives are either deep litter or straw flow both combined with slatted floor area. Outside, two of the housing alternatives are based on access to only an outdoor pen with solid flooring, and the other two alternatives also have a yard with pasture during summer period. Nitrogen balances are calculated for the four housing alternatives. "Input" nitrogen in the form of feed, roughage and straw are determined via feed utilisation, feed composition and straw usage, respectively. The "output" nitrogen are determined by registration of pig growth, calculation of nitrogen contained in the pigs at slaughter, nitrogen in produced manure and urine, and the estimation of nitrogen accumulated on pasture. Each deep litter bed is weighed at cleaning and samples are analysed. Amount of manure from the slatted area is weighed each fortnight and analysed. Liquid manure from outdoor pens is weighed each fortnight and analysed. Soil tests to determine manure distribution and amounts in the yards are obtained before and after the growing period. For each housing alternative, the results will obtain a total picture of the nitrogen balance at pen level. The difference between "input" and "output" total nitrogen is mainly due to ammonia emission and nitrogen leaching. Preliminary results have shown that the pen balance method has a preliminary error of about ± 10%. The nitrogen loss from the pen alternatives is high, 2.5 - 3.0 kg per pig, probably depending on fodder nutrient constitution and large total pen area.

Introduction

In Sweden, organic pigs are generally produced according to the standards of the economic association KRAV (Krav, 2006). According to these standards, the pigs shall be held outdoors on pasture during summer period, whereas indoor housing with access to an outdoor pen is permitted only during winter period. The EU permitted systems for organic slaughter pig production (CEC, 1999), with indoor housing and year round access to an outdoor pen with solid flooring and without additional access to plant covered soil, are not common in Sweden. The aim of this paper is to analyse and describe the consequences different organic production systems will have on manure management and nitrogen losses from organic pig production.

In Sweden 84% of the total ammonia emission origins from agriculture, whereof 58% from cattle/milk production, 14% from pig production, 6% from horses and 5% from poultry production. The animal production sources of ammonia can be divided in 22% from animal houses, 34% from manure storage, 36% from manure spreading and 8% from pastures (SCB, 2003). In 1999 the Swedish Government adopted a new policy setting 15 national environmental goals. Out of these, "Only natural acidification" deals with ammonia emission from the agricultural sector. The aim is to reduce the ammonia emission from agriculture by 15% in 2010 from the level of 1995 (Swedish Environmental Protection Agency, 2005).

During the past 20 years our knowledge about ammonia emissions from agriculture has improved. Many researchers have investigated factors affecting these emissions and measures to reduce emissions from livestock buildings, manure storage and spreading. In the literature, the following factors is presented to influence the ammonia emissions in livestock buildings; nitrogen content of the manure, adsorption of ammoniacal nitrogen, urease activity, pH of the manure, manure temperature, C/N ratio of the manure, manure surface area, air movements in the building, air velocity above the manure surface, ventilation rate through the building, air temperature and the availability of oxygen in the manure (Jeppsson, 2000). In organic pig production, the large pen area per animal due to the aim for better animal welfare could be in conflict with the aim to minimize the nutrient losses. An outside yard and a pasture may increase the ammonia emission. Furthermore, the content in the organic fodder may cause additional nitrogen content of the manure compared with conventional pig production.

Material and Methods

A research facility with four housing alternatives for organic fattening pig production has been built at the research farm Odarslöv situated in the south of Sweden. Inside, the housing alternatives are either deep litter or straw flow both combined with a slatted floor area. Outside, the two inside housing alternatives have access to an outdoor pen with solid flooring. Furthermore, the outside housing alternatives have either access or no access to a yard with pasture during summer period.

The stable consists of 8 pens with 2 pens per pen design (figure 1). Each pen housing 16 finishing pigs per pen, growing from about 20 to 120 kg. The building is constructed as a non-insulated facility with natural ventilation. Pen dimensions are 3.6 x 5.6 m with an manure alley of 3.6 x 1.4 m and an outdoor yard of 3.6 x 5.0 m. Figure 3 show the layout and two sections of the building.

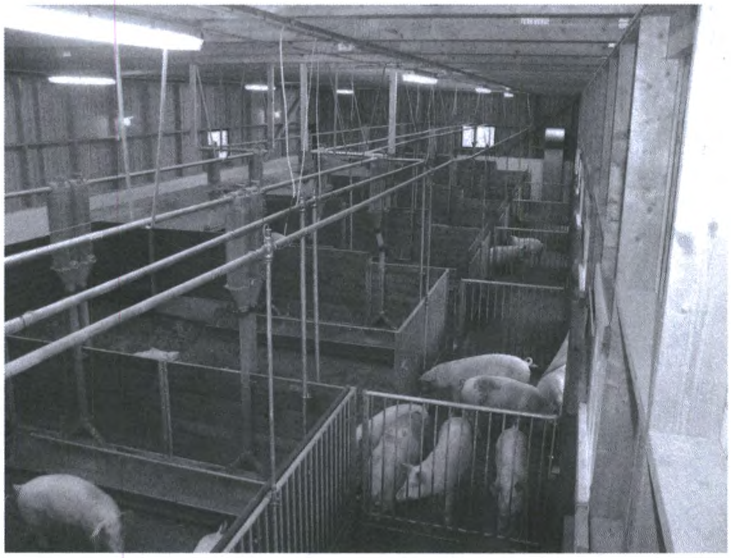


Figure 1. Research building for organic fattening pig production.

The deep litter pens (pen 1 – 4) are constructed with a 0.4 m deep and 2.5 m wide bedding area of deep litter in the front (lying area), followed by a 3.1 m wide concrete floor with two parallel fodder troughs along the long way of the pen (activity area). At the opposite end of the deep litter bed there is a manure alley with access to an outdoor yard. The manure alley consists of concrete slats with 1,0 m deep manure channel underneath (dung area).

Outdoor yard design was the same for all 8 pens (figure 2). It had an increased sloped flooring design with 2 percent slope in the first half of the area, with 7 percent slope in the next quarter and 10 percent slope in the last quarter of the outdoor yard. The 10 percent slope area was considered as prime dung area. At the lower end of this dung area a flushable channel collected the manure.

The straw flow pens (pen 5 – 8) had sloped flooring in the activity and dung area. The lying area had straw bedding of 100 mm, which was separated from the activity area by a 150 mm high threshold. The lying area was covered by a roof 1,5 m from the floor. On the activity area two parallel fodder troughs were mounted along the long way of the pen. At the end of the activity area a 10 percent sloped dung area conveyed the manure into the manure channel. From the sloped dung area the animals had access to a concrete slatted manure alley and an outdoor yard.

The pen nutrient balances have been analysed during two batches of finishing pigs. Two pens of each inside housing alternative were studied. In the winter season (fattening period A, Nov 3, 2004 – Mars 23, 2005) all four pens had access to an outdoor yard per pen. In the summer season (fattening period B, June 3, 2004 – Oct 7, 2004) one of each housing alternative had access to a pasture area of 96 m^2 per pig while the other had access to an outdoor yard. The analyses of the pen nutrient balances were made for pen no. 1, 3, 5, 7. (see figure 3).

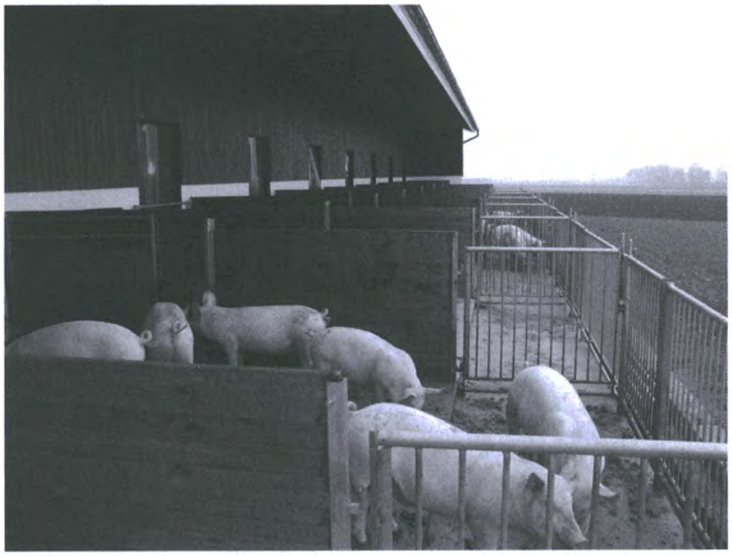


Figure 2. Outside yard at research building for organic fattening pigs

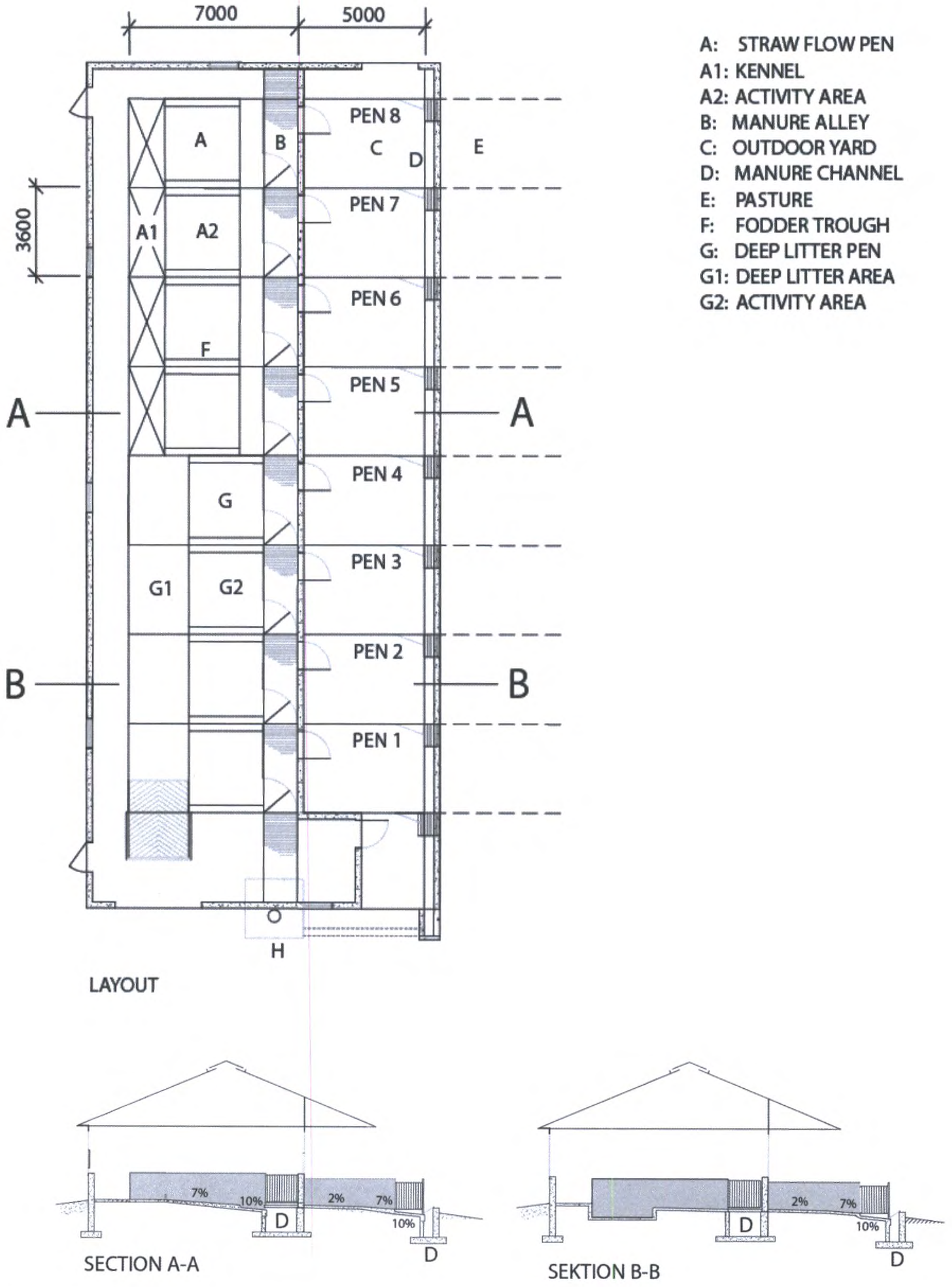


Figure 3. Layout and sections of the research building for organic fatting pigs

Input Values of Pen Nutrient Balance

Following parameters were regarded as significant input values in the nutrient balance for organic finishing pigs at pen level.

- Animal content of NPK (nitrogen, phosphorous, potassium)
- Fodder content of NPK during finishing period
- Grazing intake from pasture
- Hay and silage content of NPK
- Straw content of NPK

The input values of animal NPK per pen were based on entry date, weight, animal number, and the percentage NPK in body (Simonson, 1990). Fodder, hay, silage and straw values were all entered based on entry date, animal number, animal consumption, total solid content and NPK content from analysis.

Dry fodder pellets were given twice a day with time controlled water access. An all time access water nipple was situated in the manure alley in each pen. The fodder ration was volume metric measured and daily controlled by strain gauge sensors mounted at each leg of the fodder silo. Silage or hay was given at the top of the outdoor yard in a silage-hay bin. The amounts of straw and silage-hay were based on number of "loafs" of a bale per day, and regular measurements of loaf weights.

Output Values

The output values were:

- Animal content of NPK
- Liquid manure content of NPK, indoors and outdoors
- Straw bedding content of NPK

The output animal NPK per pen value consisted of animal number, weight at slaughter and the percentage NPK in body (Simonson, 1990). The amounts of liquid manure and samples for NPK concentration were measured at a 14 day interval and as soon as the animals had left the building the amounts of the straw bedding were weighed and sampled for NPK analysis.

Figure 4. Buckets with manure in the manure channel

The manure measurement was carried out by placing buckets underneath the slats in a cleaned indoor manure channel (figure 4). After a measuring period of 48 hours during which also the precipitation was measured, all buckets per pen in the indoor manure channel were weighed on an electronic scale (Flint AB). The scale was calibrated by loading and unloading weights up to 230 kg and the fit of the regression curve was by a $R^2 = 0, 9983$. The manure in the weighed buckets was emptied into a 100 litre barrel, thoroughly mixed and then sampled by intermittent sampling into one sample per pen when emptying the barrel. The same procedure was carried out for all four pens for the indoor manure channel.

In the outdoor yard both the manure channel and outdoor yard was cleaned before the buckets were put in place. After a measuring period of 48 hours, the outdoor yards were cleaned and the manure collected, measured and sampled together with the buckets from the outdoor manure channel following the same procedure as for the indoor manure channel.

Preliminary Results and Discussion

The pen nutrient balances for fattening period A show good agreements for total-P and total-K (Table 1). The difference between input and output is between ± 3 kg except for P in pen 3. The small difference in total-P and total-K shows that the nutrient balance method have an error of about ± 10%. In fattening period B, the differences in total-P and total-K for the pens with no access to pasture (pen 3 and 7) is slightly higher than during fattening period A, between 2.7 and 3.5 kg for total-P and 3.4 and 4.7 kg for total-K. This could indicate a systematic error and will be further examined.

The difference in total-N is between 40.6 and 54.9 kg per pen, during fattening period A. The difference is mainly due to ammonia emission from the pen. Thus, between 48 and 68% of the total-N in the manure is lost. No statistical difference between the two types of pens can be made but the results indicate an N loss of 50 kg from the straw flow pen (pen 5 and 7). The N loss from the pen with deep litter as lying area is 40 kg with "clean" bedding (pen 3) and 55 kg with "dirtier" bedding.

The cleanliness of the lying area with deep litter is crucial for the ammonia emission. A pen with "clean" deep litter seems to have a lower ammonia emission than the straw flow pens but with a "dirty" deep litter the N loss is about 15 kg higher and exceeds the straw flow pen. The N loss of about 50 kg corresponds with an ammonia emission of about 10 – 12 g day^{-1} m^{-2} during the fattening period. The ammonia emission corresponds with the results by Ivanova-Peneva & Aarnink (2005) investigating the ammonia emission from Dutch organic pig production. The ammonia emission was 1.9 – 2.7 g day^{-1} m^{-2} from clean pen areas and 11.4 - 13.3 g day^{-1} m^{-2} from fouled areas. The ammonia emission also corresponds with results from deep litter beds for conventional growing pigs, average 10 g day^{-1} m^{-2}, by Jeppsson (1998) and from outside yards for organic pig production, 9 – 12 g day^{-1} m^{-2} by von Wachenfelt (2002). The results show that the N loss is high, 3 kg per pig in average, probably depending on fodder nutrient constitution and large pen area per pig (2.6 m^2 per pig).

During fattening period B (Table 2), the pigs in pen 3 and 7 had no access to pasture. The difference in total-N is just below 40 kg corresponding to about 60% loss of total-N in the manure. From these pens the difference in nitrogen is mainly the ammonia emission from the pens. The N loss is 2.5 kg per pig.

The measured amount of NPK in the manure from the pens with access to pasture, pen 1 and 5, are lower than from the pens without access (pen 3 and 7). This indicates that some of the faeces and urine land in the pasture. Calculations preliminary show that 43 and 17% of the manure land on the pasture corresponding to 12 and 4 kg total-N, 8 and 4 kg total-P, and 11 and 4 kg total-K, respectively. The N-loss is 43 kg corresponding to 60% loss of total-N in the manure for pen 1 and 5, respectively.

Table 1. Preliminary pen nutrient balances for fattening period A (2004-11-03 – 2005-03-23). No access to pasture. 16 pigs per pen.

	Pen 1			Pen 3			Pen 5			Pen 7		
	N kg	**P** kg	**K** kg	**N** kg	**P** kg	**K** kg	**N** kg	**P** kg	**K** kg	**N** kg	**P** kg	**K** kg
Input values												
Animal	8.3	1.7	0.7	7.6	1.5	0.6	7.7	1.6	0.6	7.3	1.5	0.6
Fodder	118.8	26.8	28.3	116.9	26.3	27.8	116.4	26.3	27.7	116.6	26.3	27.8
Silage	0.0	0.0	0.0	1.2	0.0	1.2	0.0	0.0	0.0	1.2	0.0	1.2
Straw	3.9	0.6	5.5	3.4	0.5	4.8	1.7	0.2	2.4	1.5	0.2	2.1
Sum	**130.9**	**29.0**	**34.5**	**129.1**	**28.4**	**34.5**	**125.8**	**28.0**	**30.7**	**126.6**	**28.0**	**31.7**
Output values												
Animal	38.5	7.8	3.2	41.4	8.4	3.5	41.1	8.3	3.4	41.7	8.4	3.5
Manure	25.5	14.4	18.0	30.7	17.3	25.0	33.0	20.6	28.3	35.6	20.0	26.3
Bedding	12.0	4.1	14.1	16.3	4.6	16.6	2.7	1.0	2.3	1.2	0.4	1.4
Sum	**76.0**	**26.3**	**35.4**	**88.5**	**30.2**	**45.0**	**76.8**	**29.9**	**34.0**	**78.5**	**28.9**	**31.2**
Difference	**54.9**	**2.7**	**-0.9**	**40.6**	**-1.9**	**-10.5**	**49.0**	**-1.8**	**-3.3**	**48.2**	**-0.9**	**0.6**

Table 2. Preliminary pen nutrient balances for fattening period B (2004-06-03 – 2004-10-07). Pigs in pen 1 and 5 had access to pasture. 16 pigs per pen.

	Pen 1			Pen 3			Pen 5			Pen 7		
	N kg	**P** kg	**K** kg	**N** kg	**P** kg	**K** kg	**N** kg	**P** kg	**K** kg	**N** kg	**P** kg	**K** kg
Input values												
Animal	7.7	1.6	0.6	8.8	1.8	0.7	7.6	1.5	0.6	8.7	1.8	0.7
Fodder	95.6	24.7	25.5	96.5	25.0	25.7	95.0	24.6	25.3	96.4	24.9	25.7
Pasture[1]	11.1	1.0	8.0	-	-	-	5.2	0.5	3.8	-	-	-
Straw	1.4	0.2	2.0	1.4	0.2	2.0	0.4	0.1	0.6	0.4	0.1	0.6
Sum	**115.7**	**27.5**	**36.1**	**106.7**	**27.0**	**28.4**	**108.2**	**26.7**	**30.3**	**105.5**	**26.8**	**27.0**
Output values												
Animal	42.5	8.6	3.5	43.8	8.8	3.6	41.2	8.3	3.4	43.0	8.7	3.6
Manure[2]	25.9	14.9	24.4	24.2	14.6	19.0	24.2	14.5	20.5	23.3	14.5	18.5
Bedding	3.9	1.0	4.2	1.9	0.8	2.4	1.1	0.4	0.8	0.3	0.1	0.2
Sum	**72.3**	**24.6**	**32.1**	**69.9**	**24.3**	**25.0**	**66.5**	**23.2**	**24.7**	**66.6**	**23.3**	**22.3**
Difference	**43.4**	**3.0**	**3.9**	**36.8**	**2.7**	**3.4**	**41.7**	**3.5**	**5.6**	**38.9**	**3.5**	**4.7**
Manure												
in pen[3]	9.9	5.5	9.0	24.2	14.6	19.0	18.8	10.5	15.3	23.3	14.5	18.5
on pasture[4]	12.1	8.4	11.1				4.3	3.6	4.3			

[1] The amount of pasture calculated from the content of K in the manure; [2] Pen 1 and 5 calculated from average of pen 3 and 7 in relation to sum of input values; [3] Measured and analysed values; [4] Calculated values

Preliminary Conclusions

- The pen nutrient balance method has a preliminary error of about ± 10%.
- The ammonia emission from the pen areas is between 10 and 12 g day^{-1} m^{-2}.
- The nitrogen loss from the pen alternatives is high, 2.5 and 3.0 kg per pig, probably depending on fodder nutrient constitution and large total pen area.
- Between 20 and 50% of the NPK in the manure land on the pasture depending on how much the pigs use the pasture.

References

CEC. 1999. Council Regulation on Organic Livestock Production 1804/1999 of 19 July 1999 Supplementing Regulation (EEC) No. 2092/1991. Commission of the European Communities, Brussels

Ivanova-Peneva, S.G. & Aarnink, A.J.A. 2005. Ammonia emissions in organic pig production. Bau, Technik und Umwelt in der landwirtschaftlichen Nutztierhaltung, Braunschweig

Jeppsson, K-H. 1998. Ammonia emission from different deep-litter materials for growing-finishing pigs. Swedish J. agric. Res. 28: 197-206

Jeppsson, K-H. 2000. Aerial Environment in Uninsulated Livestock Buildings - Release of ammonia, carbon dioxide and water vapour from deep litter and effect of solar heat load on the interior thermal environment. Doctoral thesis, Swedish University of Agricultural Sciences, Agraria 245. Alnarp.

Krav. 2006. Krav´s regler [Krav´s standards]. Krav, Uppsala (in Swedish)

Simonson, A. 1990. Omsättning av kväve, fosfor och kalium i svinproduktionen [Transformation of nitrogen, phosphorous and potassium in pig production] Sveriges lantbruksuniversitet. Fakta Husdjur nr. 1, 1990. Uppsala (in swedish)

Swedish Environmental Protection Agency. 2005. Swedens environmental objectives- for the sake of our children - deFacto 2005. (ISBN 91-620-1240-1)

http://www.miljomal.nu (February 14, 2006)

von Wachenfelt, H. 2002. Betesdrift och utomhusytor för ekologiska svin [Organic pig production on pasture and outdoor areas], Specialmeddelande 236. Inst f jordbrukets biosystem och teknologi, SLU, Alnarp (in Swedish with summary in English)

Greenhouse Gas Emissions from Stored Animal Manure in Cold Climates

K.-H. Park[1], C. Wagner-Riddle[1], R. Gordon[2], V. Glass[2]
[1]Department of Land Resource Science, University of Guelph, Guelph, ON, N1G 2W1, Canada
[2]Department of Engineering, Nova Scotia Agricultural College, Truro, NS, B2N 5E3, Canada

Abstract

Current global warming has been linked to increases in greenhouse gas (GHG) concentrations. Animal manure is an important source of anthropogenic GHG, mostly of methane (CH_4) and nitrous oxide (N_2O), but environmental and animal factors affect and lend uncertainties to GHG emissions estimates from these sources. Country-specific emission estimates of these GHG can be obtained using IPCC 2000 guidelines, or suggested improvements, such as the USEPA approach for CH_4 emissions, which is based on monthly air temperature (Tair). These approaches have not been validated against measured CH_4 and N_2O fluxes for liquid swine and dairy manure storage in cold climates due to the scarcity of year-round studies. A four-tower micrometeorological mass balance method was used at three swine farms and two dairy farms in Ontario, Canada (annual Tair <10°C), from July 2000 to July 2003. Methane and N_2O concentrations were measured using two tunable diode laser trace gas analyzer. Mean N_2O fluxes were not significantly different from zero. Mean CH_4 fluxes obtained from half-hourly data varied between 4.6×10^{-3} to 1.05 mg m^{-2} s^{-1} for swine and 3.7×10^{-2} to 0.1 mg m^{-2} s^{-1} for dairy. The methane conversion factor for liquid swine manure stored in concrete tanks derived from measured fluxes was 0.23, comparable to the USEPA derived values of 0.22 – 0.25, but much lower than the IPCC recommended value for cold climates (0.39).

Introduction

Agricultural practices in Canada and globally have been identified as contributors to greenhouse gas emissions. In 1999, agriculture in Canada contributed 8.7% (or 61Mt) of total greenhouse gas emissions, an increase from 3.1% in 1990 (Government of Canada, 2001). The increased popularity of liquid manure handling systems at intensive livestock operations has raised environmental concerns regarding greenhouse gas emissions (Honeyman, 1996; Miner, 1999). Manure storage of this sort can occur for several months, leading to anaerobic decomposition and mostly the release of CH_4. IPCC (2000) recommends non-invasive and year-round measurements of GHG emissions in production systems, in order to reduce the large uncertainties in emission factors used for calculating national GHG emission inventories. The Micrometeorological Mass Balance (MMB) method, a non-invasive measurement technique, is suitable for heterogeneous source distributions and/or elevated sources such as waste storage sites (Denmead et al., 1998).

The research reported here was carried out to address some of the uncertainties in greenhouse gas emissions from stored liquid swine and dairy manure. The objectives of this research were as follows: 1) to quantify N_2O and CH_4 emissions from manure storage systems *in situ*; 2) to convert measured greenhouse gas emissions to emission factors and compare these with IPCC emission factor estimates.

Materials and Method

Three swine farms (Arkell, Jarvis, and Guelph) and one dairy farm (Bright) in Southern Ontario were chosen for this study. A mobile instrumentation trailer was moved between these sites to carry out measurements from June 2000 to July 2003. The micrometeorological mass balance method was used to measure CH_4 and N_2O fluxes at the four farms (Wagner-Riddle et al., 2006). This method requires gas concentration, wind speed and wind direction for calculations. The wind direction and wind speed were recorded using a mast mounted on the fence surrounding the manure tank. Each mast had four cup anemometers (F460, Climatronics Corp., Newton, PA) recording the wind speed at 25, 100, 200 and 350 cm above the concrete wall of the manure tanks. A wind vane (R.M. Young, Model 05102, Traverse, MI) was mounted on top of each mast (450 cm), to record the wind direction. Mean wind speed was recorded over 30 minutes, and mean wind direction was recorded over 5 minutes at Arkell and over 1 minute at the other sites. Four towers were placed around the tanks to ensure measurements of upwind and downwind air concentrations occurred during most periods. Each tower was equipped with four air sample intakes, placed at 25, 100, 200 and 350 cm above the manure tank wall. A vacuum pump (RA 0021, Busch, Virginia Beach, VA) was used to draw the samples from the intakes to the trailer. CH_4 and N_2O concentrations were quantified using two Tunable Diode Laser Trace Gas Analyzers (TDLTGA). Before reaching the TDLTGA the gas samples passed through a valve manifold unit (Campbell Scientific, Logan, Utah). This unit, capable of handling all 16 intakes, enabled the switching between the different sample intakes and the two analysers. As only one sample could be passed through each TDLTGA at any given time, the samples not going to the analysers were passed through a by-pass manifold and directed to the pump. CH_4 and N_2O fluxes were calculated using the mass balance method with the wind and concentration data as inputs. The mean fetch was calculated from 5- or 1-minute fetch values obtained from 5- or 1-minute mean wind direction data and averaged over 30 minutes.

Results

Nitrous oxide fluxes were small with mean value for several of the measurement period not significantly different from zero ($P<0.05$; data not shown). Methane fluxes at all of the sites were highly variable in time, a result of seasonal variation in temperature (Table 1). Methane fluxes from liquid swine manure were >0.5 mg m^{-2} s^{-1} during the warmest measurement periods, while fluxes from liquid dairy manure were much ~0.1 mg m^{-2} s^{-1}, as expected due to the lower CH_4 production potential of the latter.

As the measurements of CH_4 fluxes from liquid swine manure at the three studied sites comprised a period > 1 year, comparison with annual emission factors using the IPCC and USEPA approaches was possible. Measured CH_4 flux means were scaled using storage tank surface area and number of animals at each site. Consistent over-prediction of monthly CH_4 emissions for the January to April period when using manure temperature in the USEPA approach were observed due to formation of an ice layer which suppressed emission (Park et al., 2006). However, for GHG inventory purposes the USEPA approach yielded quite acceptable results, with annual EF of 6.5, and 7.5 kg CH_4 head^{-1} yr^{-1} with use of air temperature >7.5 °C, respectively, compared to the measured EF of 6.7 kg CH_4 head^{-1} yr^{-1}. Methane EF using IPCC Tier 2 for the three sites averaged 11.5 kg CH_4 head^{-1} yr^{-1}, an over-prediction consistent with the use of MCF = 0.39, when compared to the value derived from measurements (MCF = 0.23) and the USEPA derived values (MCF = 0.22 – 0.25).

Conclusions

The micrometeorological mass balance method, a non-intrusive flux measurement method, was used in the *quasi*-continuous measurement of GHG fluxes from stored liquid manure over ~ 2 years. Nitrous oxide fluxes were negligible for most of the measurement period. The annual CH_4 EF derived from our data was within ~10% of EF based on the USEPA approach. But, the measured EF was only ~60% of the EF derived using IPCC Tier 2. This discrepancy is due to an MCF factor (=0.39) recommended for climates with mean annual temperature <15 °C, which is clearly an over-estimate for cold climates (MCF<0.25).

Table 1: Mean methane flux measured for each location and measurement period. Arkell, Jarvis and Guelph are farms with liquid swine manure; Bright is a farm with liquid dairy manure. Mean air temperature is presented for each period. Standard error of mean flux shown in brackets.

Site	Measurement Period	Mean air temperature (ºC)	CH_4 (µg m^{-2} s^{-1})
Arkell	June 25 – July 18, 2000	16.7	706.3 (89)
	Oct. 24 – Nov. 5, 2000	7.2	250.7 (51)
	Jan. 9 – Jan 14, 2001	-5.6	53.6 (8)
	Mar. 22 – May 3, 2001	9.9	91.8 (11)
Jarvis	May 21 – July 16, 2001	18.5	583.4 (21)
	Nov. 8 – Nov. 28, 2001	7.4	174.1 (16)
Guelph	July 27 – Aug. 10, 2001	18.5	247.6 (50)
	Oct. 11 – Oct. 30, 2001	7.4	1054.8 (27)
	Jan. 14 – Mar. 28, 2002	-3.0	18.4 (2)
	Mar. 29 – April 18, 2002	7.3	38.2 (3)
Bright	Dec. 5, 2002 – April 16, 2003	-5.8	36.9 (5)
	April 17 – May 22, 2003	7.9	51.4 (4)
	May 23 – July 27, 2003	17.7	104.1 (6)

References

Denmead, O. T., L.A. Harper, J.R. Freney, D.W.T. Griffith, R. Leuning, and R.R. Sharpe. 1998. A mass balance method for non-intrusive measurements of surface-air trace gas exchange. *Atmospheric Environment*, 32(21): 3679-3688.

Government Canada. 2001. Canada's third national report on climate change : actions to meet commitments under the United Nations Framework Convention on Climate Change, ISBN 0-660-18694-2.

Honeyman, M.S. 1996. Sustainability issues of U.S. swine production. *Journal of Animal Science*, 74: 1410-1417.

IPCC. 2000. Good practice guidance and uncertainty management in national greenhouse gas inventories. J Penman, D Kruger, I Galbally, T Hiraishi, B Nyenzi, S Emmanul, L Buendia, R Hoppaus, T Martinsen, J Meijer, K Miwa and K Tanabe (Eds). The Institute for Global Environmental Strategies, Japan.

Miner, J.R. 1999. Alternatives to minimize the environmental impacts of large swine production units. *Journal of Animal Science*, 77: 440-444.

Park, K. -H., A. G. Thompson, M. Marinier, K. Clark, and C. Wagner-Riddle. 2006. Greenhouse gas emissions from stored liquid swine manure in a cold climate. *Atmospheric Environment* 40: 618-627.

Wagner-Riddle, C., K.H. Park, and G.W. Thurtell. 2006. A micrometeorological mass balance approach for greenhouse gas flux measurements from stored animal manure. *Agr. For. Meteorol.* (in press).

Aerobic Composting as a Strategy for Mitigation of Greenhouse Gas Emissions from Swine Manure

K.-H. Park[1], K. A. Kumudinie[1], C. Wagner-Riddle[1], R. Fleming[2], M. MacAlpine[2]
[1]Department of Land Resource Science, University of Guelph, Guelph, Ont., N1G 2W1, Canada
[2]Ridgetown College of Agricultural Technology, University of Guelph,
Ridgetown, Ont., N0P 2C0, Canada

Abstract

Agriculture has been recognized as a major contributor of greenhouse gases, releasing both nitrous oxide and methane. The collection and storage of livestock manure has been identified as a significant source of greenhouse gases. New strategies are being investigated as to their potential for reducing emissions of nitrous oxide and methane. Aerobic composting of the manure before storage is believed to have potential for mitigating the release of greenhouse gases, but research has not fully quantified emissions during the active and curing phases of this process. In addition, nitrous oxide emissions from soils after the field application of composted versus untreated liquid swine manure has not been considered in past studies. Here we report on two studies measuring: 1) methane and nitrous oxide emissions during the active in-vessel and curing phase of composting liquid swine manure with straw, and 2) nitrous oxide emissions after field application of untreated liquid swine manure in comparison to composted liquid swine manure, carried out in Ontario, Canada. A mega-chamber approach and eight large non-steady chambers were used to quantify CH_4 and N_2O fluxes during composting. A flux-gradient method was used to measure N_2O fluxes from field plots after application of composted and untreated manure in fall 2004 until corn planting in spring 2005. A tunable diode laser trace gas analyzer was used in both studies to measure gas concentrations. An analysis comparing GHG emissions during manure treatment (aerobic composting) through field application vs. non-treatment will be presented.

Introduction

Agriculture has been recognized as a major contributor of greenhouse gases (GHG), releasing both nitrous oxide and methane (Houghton et al., 2001; Mosier et al., 1998). The collection and storage of livestock manure has been identified as a significant source of methane and nitrous oxide (Rotmans et al., 1992; Khalil and Rasmussen, 1992). Treatment strategies, such as aerobic composting of the manure before storage, are being investigated as to their potential for stabilizing organic residuals and the fertilizer value for land application (Zeman et al., 2002). This treatment is also believed to have potential for mitigating the release of greenhouse gases (Thompson *et al*., 2004), but research has not fully quantified emissions during the active and curing phases of this process. In addition, nitrous oxide emissions from soils after the field application of composted versus untreated liquid swine manure have not been considered in past studies. The objectives of this study were: 1) to quantify methane (CH_4) and nitrous oxide (N_2O) fluxes during the active in-vessel and curing phase of composting liquid swine manure with straw, 2) to quantify N_2O fluxes after field application of untreated liquid swine manure in comparison to composted liquid swine manure, and 3) to compare overall GHG emissions from treated and untreated liquid swine manure.

Materials and Method

Composting

Measurements of CH_4 and N_2O fluxes were carried out at the composting facility of the Ridgetown College, Ontario. The facility utilized an in-vessel composting system, which consisted of three open-ended concrete channels (15.2 m long, 2.2 m wide and 1.6 m deep) housed in a covered building. The floors of the compost channels contained sub-channels connected to fans, which aerated the compost using a base level aeration regime coupled with a temperature feedback system (Fleming *et al*., 1999). Large round bales of wheat straw (a total of 1262 kg per channel) were placed in the channels, and manure was pumped from the holding tank to an hydraulic compost turner (Lagace System Corporation, Barrie, ON), which ran along steel tracks on the top of the channel walls, and dropped manure onto the straw while mixing both materials. Liquid swine manure was added three times over a 2 week period, each time at a rate of ~ 3.8 to 11.0 m^3. Fluxes were monitored over two composting batches July 30 to August 19, and September 14 to October 1, 2004. The default aeration regime was 3 min per hour and aeration regimes ranging from 2 min every 10 min to 4 min every 10 min were set up for each channel, according to compost temperature feedback provided by six thermocouples installed in each channel.

Eight fiberglass chambers, which consisted of 120 cm-long, 60 cm-diameter semi-circular domes with flat ends, were used to measure fluxes. Polyethylene tubing (12.7 mm i.d., 15.9 mm o.d.) connected to the inlet of these open chambers, delivered air from a blower at 189.2 L min^{-1}. At the same time, the compost building was used as a 'mega-chamber' by placing two fans with flow rate of 339.72 m^3 min^{-1} at each end of the building as described by Thompson et al. (2004). Methane and N_2O concentrations at the inlet and the outlet of fiberglass chambers and mega-chamber were quantified with two tunable diode laser Trace Gas Analyzer (TGA), one for CH_4 and one for N_2O (TGA100, Campbell Scientific Inc., Logan, UT). Mean gas concentrations over 10 min were recorded.

Curing

The in-vessel composting process described above, results in material that is placed out-of-doors as curing piles for 3-6 months before application to fields. In order to characterize fluxes from these piles over time, materials produced between May 13 to October 1, 2004, were placed in separate piles as each of eight composting batches concluded. Measurement of CH_4 and N_2O fluxes from these curing piles, ranging in age between 1 and 123 days at the start of measurements, was conducted from Oct. 1 to Oct. 13, 2004. One fiberglass chamber was placed at the top of the each curing pile, and CH_4 and N_2O fluxes were measured as explained above.

Application of Liquid Manure and Composted Manure to Crop Fields

The cured compost material was transported to the Arkell Research Farm, Ontario, for field application in November 2004. Three treatments were studied: application of composted swine manure, application of untreated liquid swine manure, and no manure application (control). Each treatment was applied to a 1-ha plot (100 m x 100 m). Untreated and treated manure were applied at a rate of 69 kg N ha^{-1} followed by chisel ploughing (80 % wheat stubble left on the field). Corn was planted in May 2005 and fertilized with urea and mono ammonium phosphate at a rate of 83kg N ha^{-1}.

Nitrous oxide flux from each plot was measured using a micrometeorological flux gradient method, using a TGA and eddy diffusivity coefficient derived from a wind profile (Wagner-Riddle et al., 1996). Measurements were carried out from end of January to June 2005.

Comparison of Greenhouse Gas Emissions from Treated and Untreated Manure

Methane and N_2O fluxes during composting and after compost application to fields were compared to fluxes generated by untreated liquid swine manure. Since composting occurred over May to October, this ~6 month period was chosen for the comparison of effect of treatment on GHG emissions during manure storage. Values for mean methane and nitrous oxide fluxes for untreated liquid swine manure stored in a concrete tank were taken from Park et al. (2006). Fluxes from treated and untreated manure were expressed on a volume of manure basis, and also expressed in terms of CO_2-equivalent by multiplying CH_4 fluxes by 23 and N_2O fluxes by 296, and adding these fluxes (IPCC, 2001). For the field application component, the time period considered was Nov. 2004 to Apr. 2005. Nitrous oxide fluxes measured for a 1-ha field were expressed in terms of volume of treated and untreated manure applied. Fluxes for November to January were estimated based on fluxes measured for February to April. Methane fluxes after field application of manure were considered negligible.

Results

The mean CH_4 and N_2O fluxes measured during composting and curing periods, and after land application are shown in Table 1. Methane and N_2O fluxes during composting quantified by the mega-chamber were 68.7 % and 157.5% of those quantified by the fiberglass chambers, respectively. This appears to be related to the spatial variability of compost temperature, and the placement of fiberglass chambers in the center of the composting channels, where highest temperatures tended to occur. Apparently high compost temperature led to higher CH_4 fluxes, but N_2O was suppressed under those conditions. The megachamber approach integrated fluxes from all regions of the compost channels, including those where temperature was not as high as the center.

Mean CH_4 and N_2O fluxes during curing were higher than during the in-vessel phase but the integrated emission considering emitting area and volume of manure treated were lower for curing. However, the duration of curing (~137 d) compared to the in-vessel phase, made overall GHG emissions higher for curing.

Accumulated CH_4 and N_2O fluxes for aerobic composting and liquid manure storage from May to October (154 d) were compared in terms of CO_2-equivalent emissions. Composting had total GHG emissions that were between 66.7 to 73.4% of those from liquid manure. The average N_2O fluxes from the plots receiving liquid manure and compost were 26.7 ng m^{-2} s^{-1} and 11.0 ng m^{-2} s^{-1}, respectively, from February to June 2005 (P=0.05). However, CO_2-equivalent emissions from these 1-ha plots were only between 5 and 21% of total GHG emissions from composting and untreated manure, respectively. The total overall GHG emission in CO_2 equivalent for aerobic composting was between 55.9 to 61.2% of the GHG emissions from untreated manure.

Conclusions

Methane and N2O emissions during storage and field application of treated and untreated liquid swine manure point to aerobic composting as a GHG mitigation practice. Comparison on a volume of manure basis for treated or untreated manure showed that most GHG emissions occur in the form of CH4, with N2O being negligible during liquid storage and <20% of GHG emissions during composting. Further comparison of manure treatments should consider other gaseous emissions such as ammonia.

Table 1. Methane and N2O fluxes associated with untreated and treated (aerobic composting) liquid swine manure during storage and field application over approximately 1 year.

Manure	Measuring Method	CH_4 flux		N_2O flux		CO_2 equivalent**
		µg m^{-2} s^{-1} (n*, SE)	µg L^{-1} s^{-1}	ng m^{-2} s^{-1} (n, SE)	ng L^{-1} s^{-1}	kg m^{-3} $period^{-1}$
Composting In-Vessel (17 d)	Mega Chamber	118.1 (30, 19.4)	0.449	1643.2 (29, 188.2)	6.3	17.9
	Fiberglass Chambers	171.9 (240, 14.7)	0.654	1039.9 (246, 81.3)	4.0	23.8
Curing (137 d)	Fiberglass chambers	289.6 (175, 45.8)	0.128	3239.8 (175, 269.9)	1.6	40.3
Land application (181 d)	Flux Gradient	n/a	n/a	11.0 (123, 1.7)	0.78	3.6
Total (335 d)						61.8 –67.7****
Untreated Storage (154 d)	Micromet mass balance	535.8 (684, 59.8)	0.281	0 - 337.6	~0.1	87.3
Land application (181 days)	Flux Gradient	n/a	n/a	26.7 (123, 2.7)	5.0	23.3
Total (335 d)						110.6

[*] number of observations (half-hourly for micrometeorological mass balance, daily for chamber and flux-gradient)

[**] calculated using a global warming potential (GWP) of CH_4 and N_2O (23 and 296 times higher than the GWP of CO_2, respectively); values expressed as total emission per period in days as indicated in first column

[***] Mean CH_4 and N_2O fluxes were calculated from monthly mean fluxes for May to October in Park et al. (2006)

[****] lower total obtained by using mega-chamber flux and higher total obtained with fiberglass chamber

References

Fleming, R., M. MacAlpine, and J. Morris. 1999. Final Report: Development of a Farm-Scale System to Compost Liquid Swine Manure, submitted to Ontario Pork. University of Guelph, Canada.

Houghton, J. T., Y. Ding, D. J. Griggs, M. Noguer, P. J. van der Linden, and D. Xiaosu. 2001. Climate Change 2001: The Scientific Basis Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change (IPCC). Cambridge University Press, UK.

Khalil, M. A. K., and R. A. Rasmussen. 1992. The global source of nitrous oxide. *Journal of Geophysical Research* 97: 14651-14660.

Mosier, A., C. Kroeze, C. Nevison, O. Oenema, S. Seitzinger, and O. van Cleemput. 1998. Closing the global N_2O budget: nitrous oxide emissions through the agricultural nitrogen cycle. *Nutrient Cycling in Agroecosystems* 52: 225-248.

Park, K. -H., A. G. Thompson, M. Marinier, K. Clark, and C. Wagner-Riddle. 2006. Greenhouse gas emissions from stored liquid swine manure in a cold climate. *Atmospheric Environment* 40: 618-627.

Rotmans, J., M. G. J. Den Elzen, M. S. Krol, R. J. Swart, and H. J. van der Woerd. 1992. Stabilizing atmospheric concentrations: towards international methane control. *Ambio* 21: 404-413.

Thompson, A.G., C. Wagner-Riddle, R. Fleming. 2004. Emissions of N_2O and CH_4 during the composting of liquid swine manure. *Environmental Monitoring and Assessment* 91: 87-104.

Wagner-Riddle , C., G. W. Thurtell, G. K. Kidd, and E. G. Beauchamp. 1996. Nitrous Oxide and Carbon Dioxide Fluxes from a Bare Soil Using a Micrometeorological Approach. *Journal of Environmental Quality* 25: 898-907.

Zeman, C., D. Depken, and M. Rich. 2002. Research on how the composting process impacts greenhouse gas emissions and global warming. *Compost Science & Utilization* 10: 72-86.

Modeling Fugitive Dust Emissions in Pinal County, Arizona

J.P. Engelbrecht[1] and K. Walch[2]
[1]Desert Research Institute, 2215 Raggio Parkway, Reno, NV 89512
[2]Pinal County Air Quality Control District, 31 N. Pinal Building F, Florence, AZ 85232

Abstract

This paper presents the results from a source attribution modeling study conducted on aerosol samples collected in Pinal County, Arizona.

In October and November of 2003, the Pinal County Air Quality Control District (PCAQCD) in collaboration with the Desert Research Institute (DRI) performed an air quality study in Pinal County, Arizona, in order to identify and quantify the contributions by emission sources to elevated PM_{10} and $PM_{2.5}$ particulate levels.

Low volume MiniVol samplers were operated on a US EPA one in three day 24-hour schedule, in parallel with existing Federal Reference Method (FRM) samplers. Five air quality monitoring sites located throughout the Pinal County agricultural basin were included in the study. The sites were located in the City of Casa Grande, the City of Coolidge, the Town of Stanfield, the Pinal County Housing Complex at Eleven Mile Corner, and a location locally known as Cowtown, which consists of several cattle feedlot companies. Additionally, three new fugitive dust samples were collected for re-suspension from a feedlot, dirt road and agricultural field.

Gravimetric analysis on the Teflon filters showed a substantial variation amongst the five sites with PM_{10} varying from 13 to 294 $\mu g/m^3$ (US EPA 24 hr National Ambient Air Quality Standard for PM_{10} = 150 $\mu g/m^3$) and for $PM_{2.5}$ varying from 3 to 183 $\mu g/m^3$ (US EPA 24 hr National Ambient Air Quality Standard for $PM_{2.5} = 65\ \mu g/m^3$)

A total of 130 ambient and 6 source filter pairs (Teflon and quartz fiber) were chemically analyzed by X-ray Fluorescence (XRF) spectrometry for 40 chemical elements, by Thermo Optical Reflectance (TOR) for 8 carbon species, by Ion Chromatography (IC) for 3 anions, by Atomic Absorption (AA) for 3 cations and automated colorimetry (AC) for ammonium.

The ambient samples together with the source profiles, including the new fugitive dust profiles and other profiles from the DRI's database were modeled by the Chemical Mass Balance (CMB) receptor model. Modeled source attributions include largely soil and feedlot dust, with minor contributions from vegetative burning, motor vehicle emissions, coal fired power plants, as well as secondary ammonium nitrate, and in a few instances secondary ammonium sulfate.

Project Description

This project (Walch and Gabrielson, 2005) was designed to identify the dominant sources of particulate matter in the Pinal County agricultural basin, which covers roughly the western third of the County. Low volume aerosol samplers capable of measuring particulate matter smaller than 2.5 microns ($PM_{2.5}$) and particulate matter smaller than 10 microns (PM_{10}) were used to collect ambient air samples at five existing monitoring sites. The Desert Research Institute (DRI) analyzed these filter samples for their mass and chemical species. These ambient results, together with measured source profiles, were modeled by the Chemical Mass Balance (CMB) receptor model to assess the source attributions within the Pinal County agricultural basin.

The ambient sampling was performed on a one in three day cycle, between October 1 and November 8, 2003. The ten individual days with the best data recovery were selected for chemical analysis. This sampling period was selected since elevated particulate matter concentrations have historically been recorded during the months of October and November. Several local sources contribute to elevated dust levels, the most notable one being cotton harvesting. Fall tillage is also taking place, and nine cotton gins throughout the agricultural basin are in full operation during this period. The cotton harvest also results in a substantial amount of traffic on agricultural field apron and other dirt roads, all contributing to elevated dust levels. The amount of construction activity in the area does not seem to vary seasonally, although it is steadily growing, providing a further source of fine particulate matter. The cooler ambient temperatures prevailing during this part of the season often create inversion conditions, trapping emissions from local sources.

Based on known elevated PM_{10} levels, the western portion of Pinal County was the area of focus for this study. Five existing monitoring sites were selected due to their proximity to potential local sources of fine particulate matter. They were the Casa Grande Downtown, Coolidge, Stanfield, Cowtown and Pinal County Housing sites (Fig. 1).

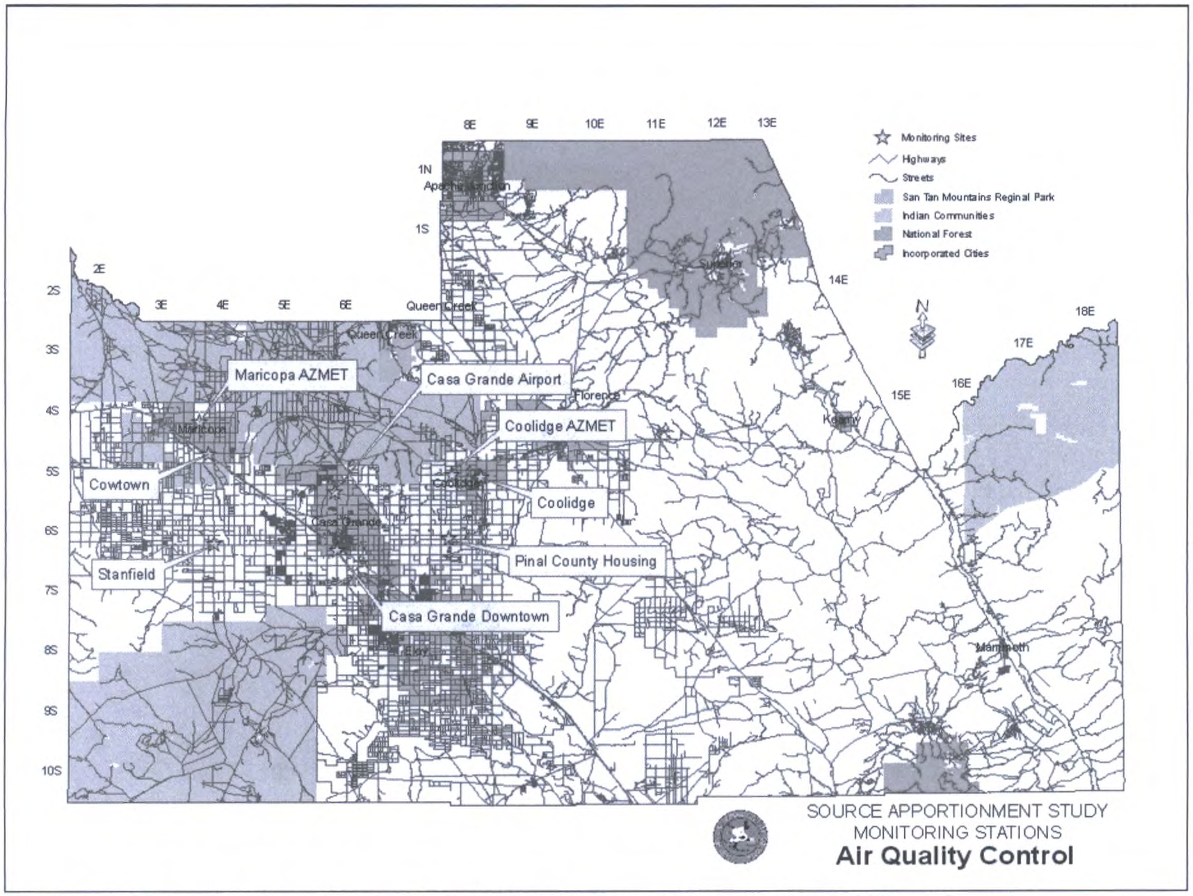


Figure 1. Map showing the air quality sites in the western sector of Pinal County

Methods

Air Monitors

Pinal County Air Quality Control District (PCAQCD) installed and operated twenty (20) Airmetrics MiniVol particulate matter samplers in the course of this study, four samplers at each of the five sites. At each site two samplers were equipped with PM_{10} and two with $PM_{2.5}$ size selective inlets. Two samples were collected on Teflon and two on quartz fiber filters for PM_{10} and $PM_{2.5}$. The samplers were run on a one in three day US EPA schedule. Each site also had other collocated sequential samplers, and continuous monitors.

Soil Source Sampling

Fugitive dust sampling was conducted close to the Cowtown site, in order to obtain more representative local soil source profiles for Receptor modeling purposes. This area was also selected for its previously recorded elevated PM_{10} concentrations and proximity to three major sources of particulate matter, feedlots, open field agriculture, and dirt roads. A significant additional source of particulate matter in the area is construction. The western portion of Pinal County where this study was focused is experiencing rapid development.

PCAQCD collected a surface soil grab sample from each of the potential major sources of fine particulate matter. The feedlot soil sample involved collecting one pound of fine surface material from each of five randomly selected feed pens, and combining these into one composite feedlot sample. Similarly the agricultural field soil samples involved collecting one pound of fine material from the surface of five randomly selected locations within an agricultural field immediately adjacent to the Cowtown ambient monitoring site, and combining these into one composite agricultural field sample. The dirt road soil samples involved collecting one pound of fine material from the road surface of five randomly selected locations along Hartman Road (a County dirt road bordering the site) and compositing these.

The DRI screened to minus 38 μm and re-suspended each composite sample, collecting filter sets of PM_{10} (minus 10 μm) and $PM_{2.5}$ (minus 2.5 μm) source samples from each of the three soil samples. These source filters were analyzed by the same methods as the ambient samples and provided chemical fingerprints (source profiles) for feedlot soil emissions, agricultural soil emissions and dirt road soil emissions (Figs. 2 and 3). The feedlot chemical fingerprints were significantly different to the other two sources, in that they contained substantial amounts of the carbon species. There was no significant difference between the agricultural field and the dirt road sample fingerprints (chemical profiles).

Filter Analyses and Modeling

The DRI weighed all Teflon filters and selected ten individual days for analysis by X-ray fluorescence spectrometry (XRF) for 40 elemental species. The number of Teflon filters chemically analyzed were as follows - ambient $PM_{10} = 50$, ambient $PM_{2.5} = 50$, source $PM_{10} = 3$, source $PM_{2.5} = 3$, field blanks = 10.

The DRI also analyzed the Pallflex quartz fiber filters for the cations Na^+ and K^+ by Atomic Absorption (AA) Spectroscopy, ammonium (NH_4^+) by Automated Colorimetry (AC), the anions SO_4^{2-}, NO_3^-, Cl^- by Ion Chromatography (IC), and the eight carbon species as well as carbonate ($CO_3^=$) by Thermal/Optical Reflectance (TOR) carbon analysis. The number of quartz filters analyzed were as follows; ambient $PM_{10} = 50$, ambient $PM_{2.5} = 50$, source $PM_{10} = 3$, source $PM_{2.5} = 3$, blanks = 10.

The DRI thereafter performed a source apportionment on the chemically analyzed filter sets by the Chemical Mass Balance (CMB) Version 8 receptor model (Friedlander, 1973; Watson *et al.*, 1997). This analysis involved using source profiles previously developed by DRI (Chow *et .al.,* 2004) together with the three new profiles from the Cowtown area.

Gravimetric Results

Summaries of the PM_{10} and $PM_{2.5}$ MiniVol gravimetric results from the five sites are presented in Table 1. The PM_{10} concentrations recorded by the MiniVol samples compared well to the continuous and sequential samplers co-located at each monitoring site. The average percent difference between the MiniVol samplers and co-located filter based Federal Reference Method samplers was 14%. A scatter plot comparing the two methods resulted in a slope of 0.6 and an R^2 value of 0.7. The average percent difference between the MiniVol samplers and co-located continuous Federal Equivalent Method samplers was elevated at 30%. Although a scatter plot comparing the two methods resulted in a slope of 0.9 and a R^2 value of 0.9

The Casa Grande sampling site, had a collocated $PM_{2.5}$ sequential FRM sampler. In this case the average Mini-Vol $PM_{2.5}/PM_{10}$ ratios listed in Table 1 were compared to the historical Casa Grande $PM_{2.5}/PM_{10}$ ratio of approximately 0.30. The average MiniVol $PM_{2.5}/PM_{10}$ ratios vary from 0.32 at the Stanfield site to 0.53 at the Coolidge and Cowtown sites. This comparison showed that the MiniVols likely overestimated the ambient $PM_{2.5}$ concentrations at times.

Table 1: PM_{10} and $PM_{2.5}$ gravimetric averages and ratios

Site	Avg. Mini-Vol PM_{10} ($\mu g/m^3$)	Avg. Mini-Vol $PM_{2.5}$ ($\mu g/m^3$)	Avg. Mini-Vol $PM_{2.5}/PM_{10}$ Ratio
Casa Grande	44	18	0.41
Coolidge	47	25	0.53
Cowtown	127	67	0.53
Pinal County Housing	60	25	0.42
Stanfield	55	18	0.32

Source Apportionment

DRI performed a source apportionment modeling (CMB version 8) on the ten of the thirteen sampling dates selected for this project. The ten runs with the best data recovery were selected for the chemical speciation and CMB modeling study. The remaining three runs were not analyzed due to budgetary constraints and thus are not included in this report. The sampling days included were October 3, 9, 12, 15, 18, 21, 27 and 30, as well as November 5 and 8, 2003. The data recovery for these ten runs was 95.5%.

Chemical Fingerprints (Source Profiles)

The chemical analyses of re-suspended filter samples provided chemical fingerprints for feedlot soil emissions, agricultural soil emissions and dirt road soil emissions (Figs. 2, 3). The PM_{10} and $PM_{2.5}$ Feedlot Dust samples differ from the PM_{10} and $PM_{2.5}$ Agricultural and Road Dust samples in that they contain a substantial amount of total carbon (TC).

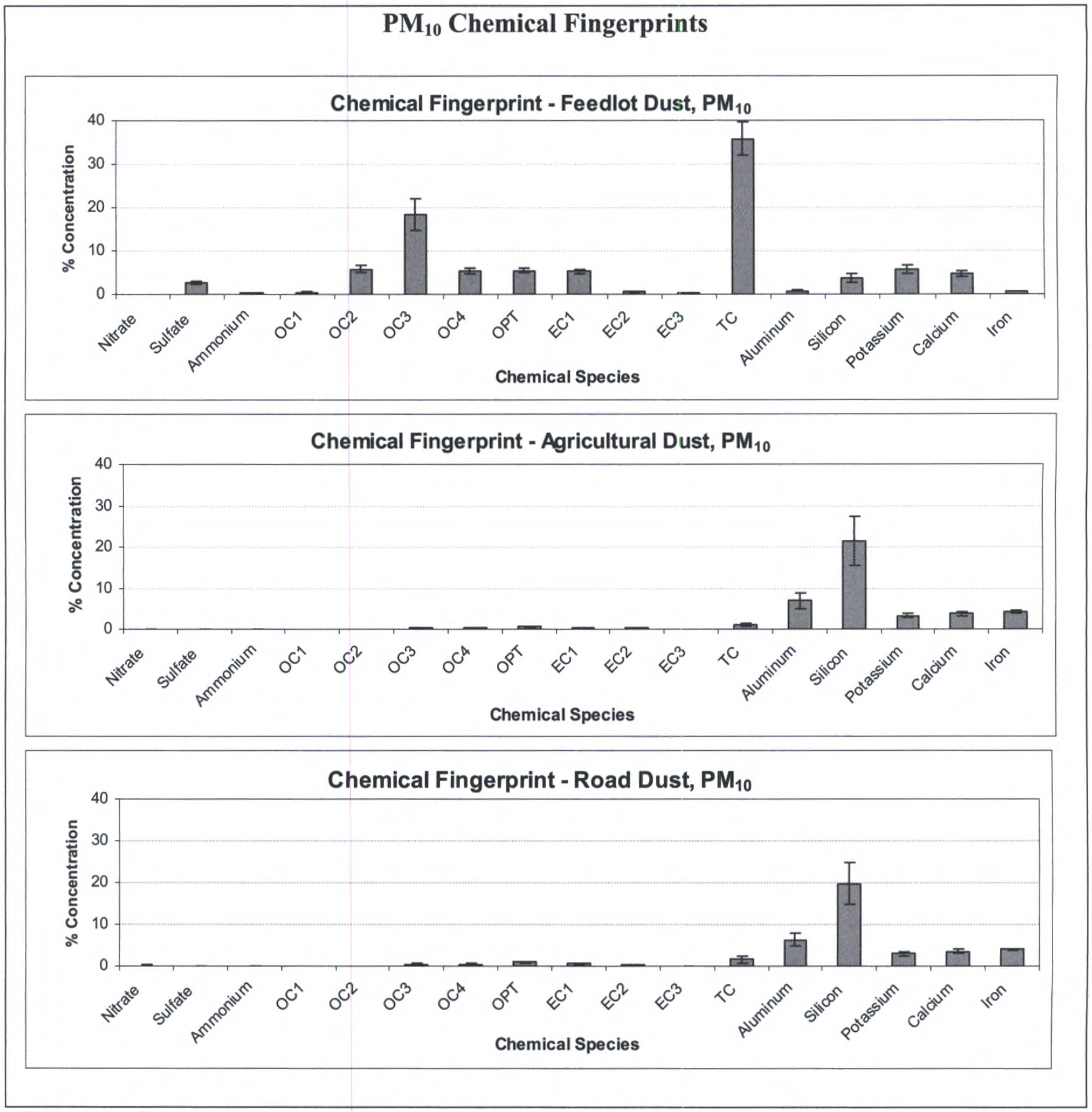


Figure 2. Chemical fingerprints (source profiles) of three PM$_{10}$ fugitive dusts from Pinal County. Concentrations of 17 chemical species and their analytical uncertainties (⊥), including water soluble nitrate, sulfate and ammonium, the eight carbon species (Organic Carbon (OC) 1 thru 4, Optical Pyrolysis by Thermal/Optical Reflectance (OPT) and Elemental Carbon (EC) 1 thru 3), as well as total carbon (TC), and five metals typical of most soils aluminum, silicon, potassium, calcium and iron.

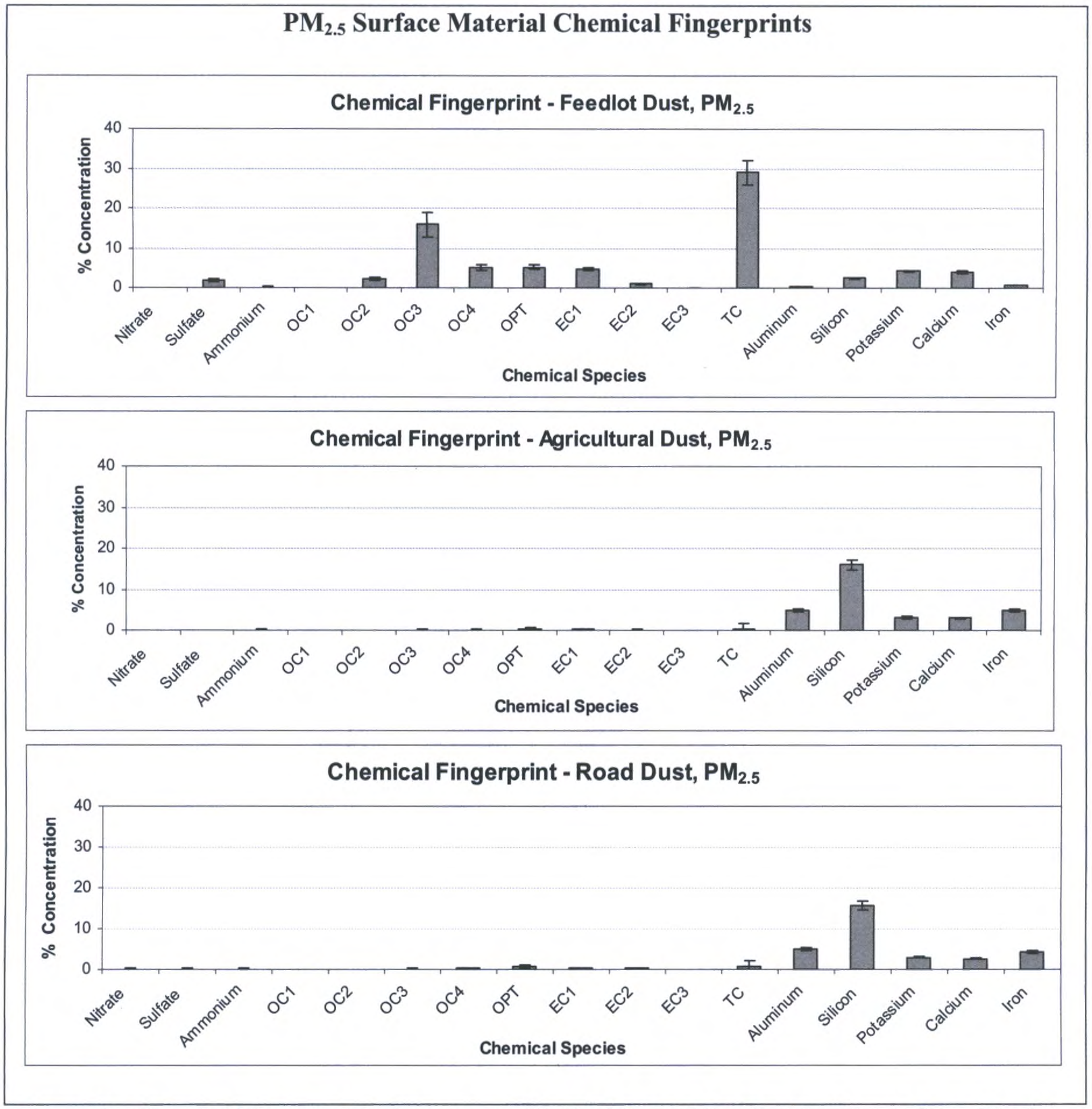


Figure 3. Chemical fingerprints (source profiles) of three $PM_{2.5}$ fugitive dusts from Pinal County. Concentrations of 17 chemical species and their analytical uncertainties (⊥), including water soluble nitrate, sulfate and ammonium, the eight carbon species (Organic Carbon (OC) 1 thru 4, Optical Pyrolyis by Thermal/Optical Reflectance (OPT) and Elemental Carbon (EC) 1 thru 3), as well as total carbon (TC), and five metals typical of most soils aluminum, silicon, potassium, calcium and iron.

Casa Grande

Casa Grande served as the control site due to its central location. Since the site is located in the downtown area of Casa Grande there are no agricultural fields, dirt roads, feedlots or large construction projects in the immediate area surrounding of the monitoring site. The average 24-hour PM_{10} concentration for this sampling site was 44 $\mu g/m^3$, the lowest of the five sites.

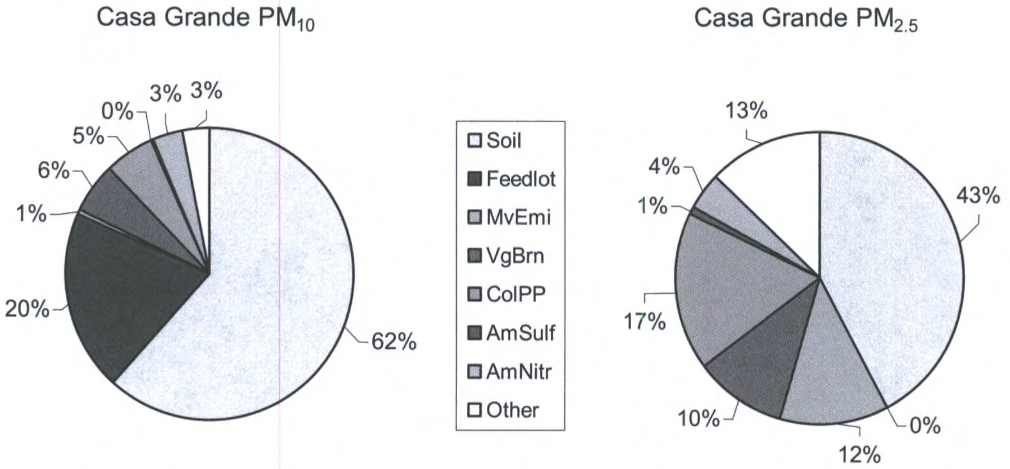


Figure 4. Average PM_{10} and $PM_{2.5}$ CMB modeled source attributions for Casa Grande.

For PM_{10} the major contribution is from agricultural, construction and dirt road soil (62%), followed by 20% from feedlot emissions (Fig. 4). The closest dairy is approximately two miles to the southeast of the site, and it is possible that the numerous dairies and feedlots located throughout the agricultural basin were contributing substantially to the recorded PM_{10} concentrations. It is also possible that agricultural fields were contributing to the modeled feedlot source since it is common practice to spread manure from dairies and feedlots on agricultural fields. This would explain feedlot emissions being identified in many samples.

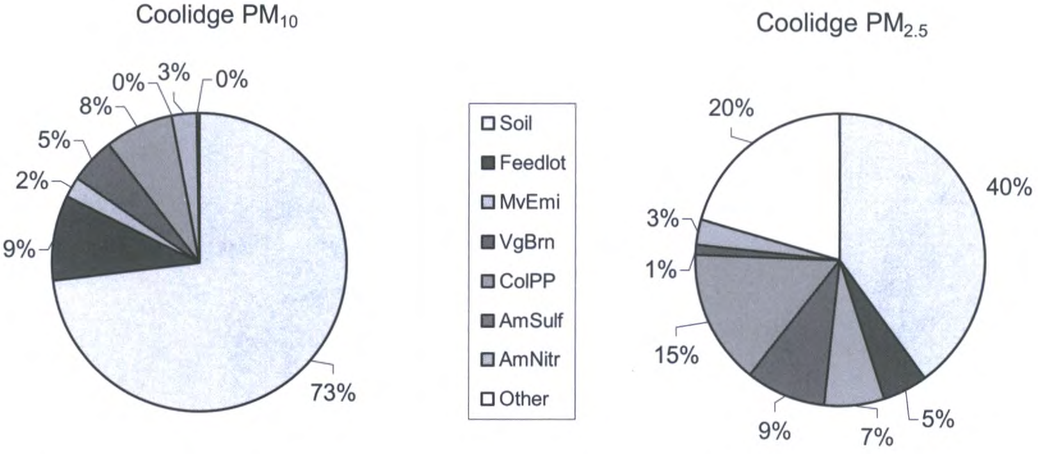
Vegetative burning and coal power plant emissions were other major sources of pollution. Open burning is commonly practiced throughout Pinal County. Countywide PCAQCD issued 253 burn permits during October 2003 that allowed for plant material to be disposed of by burning. Coal power plant emissions were identified in each individual sample and it appears that the emissions were originating from multiple point sources in the region. The closest coal power plant is approximately 70 miles to the southeast in Tucson. Ammonium Nitrate was the only remaining source contribution identified in all the samples, with an average of 3%. One sample, 10/9/03, identified ammonium sulfate with 3.8% and motor vehicle emissions with 5.3% as contributing sources.

The highest source contribution in all ten runs for $PM_{2.5}$ was the geological soil component, with an average contribution of 43% (Fig. 4). On average the second highest source contribution was coal power plants with 17%. Vegetative burning with an average source contribution of 10% and motor vehicle emissions with an average source contribution of 12% were the remaining dominant source types. The motor vehicle contribution was expected due to the proximity of several busy city streets. Ammonium nitrate and Ammonium sulfate were the remaining two identified sources with average contributions of 4% and 1% respectively.

The Casa Grande site was one of two sites in which feedlot emissions were not identified in the $PM_{2.5}$ fractions, the other being Stanfield.

Coolidge

The Coolidge site is on the eastern side of the agricultural basin, in a residential neighborhood and approximately ¼ of a mile west of presently cultivated agricultural fields. The average 24- hour PM_{10} mass concentration was one of the lowest measured in the course of this study, at 47 $\mu g/m^3$.


Coolidge PM_{10}

Coolidge $PM_{2.5}$

Figure 5. Average PM_{10} and $PM_{2.5}$ CMB modeled source attributions for Coolidge

The highest source contribution in all ten runs was the geological soil category with an average 73% (Fig. 5). On average the second highest contribution was the feedlot soil category with 9%, and feedlot emissions were only identified in four of the nine runs analyzed. The closest dairy is located approximately four miles northeast of the site and a second dairy is approximately five miles to the northwest. Coal fired power plant (8%) emissions and vegetative burning (5%) were the remaining two dominant sources.

Ammonium nitrate and motor vehicle emissions were the lowest sources with ammonium nitrate averaging 3% and motor vehicle emissions 2%. The small contribution from motor vehicle emissions was expected since the monitor is located along a low traffic residential street. Throughout the project secondary ammonium nitrate was found in small quantities in all but two PM_{10} samples. It is common practice to supplement agricultural soils with ammonium by bubbling gaseous ammonia into irrigation canals. Sewage treatment plants and animal manure are also local sources of ammonia.

The highest identified $PM_{2.5}$ source contribution in all ten runs was the geological soil category with an average contribution of 40% (Fig. 5).

Coal fired power plant emissions made up the third largest source category with an average of 15%. Vegetative burning with an average contribution of 9%, motor vehicle emissions with an average contribution of 7%, and feedlot soil emissions with an average contribution of 5% were the remaining significant sources. Vegetative burning was an expected source contribution at all the sites. Motor vehicle emissions were expected due to the site's proximity to a residential street.

Ammonium nitrate and ammonium sulfate were the remaining two sources with 3% and 1% respectively. Ammonium sulfate was only identified in three $PM_{2.5}$ samples at Coolidge, two in Stanfield, two in Casa Grande, and one PM_{10} sample in Casa Grande. The contribution was relatively small at all the sites, except Stanfield. The Coolidge and Casa Grande sites are both near heavily traveled railroad tracks. The diesel emissions, including sulfur dioxide, from the railroad locomotives may have contributed to the formation of ammonium sulfate at these sites.

Cowtown

The Cowtown site is close to several large feedlots. The site is approximately 300 yards from the closest feedlot and a grain handling operation, and completely surrounded by active and retired agricultural fields. The average 24-hour PM_{10} mass concentration at Cowtown was well above any other site, at 127 $\mu g/m^3$. Four samples had a recorded PM_{10} mass concentration above 200 $\mu g/m^3$, those from October 18, 21, and November 5, 8, 2003.

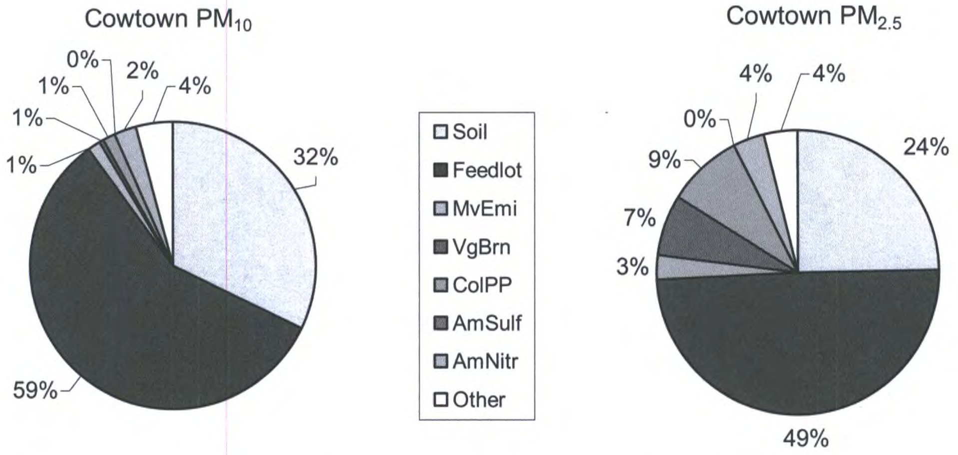


Figure 6. Average PM_{10} and $PM_{2.5}$ CMB modeled source attributions for Cowtown

The highest PM_{10} source contribution in all nine samples (one run was voided due to an equipment failure) was the feedlot soil category with an average contribution of 59% (Fig. 6). The second highest source contribution was geological soil with an average of 32%. This was the only site in the study where the geological soil was not the largest source.

On average the remaining source categories made up small contributions. Ammonium nitrate contributed on average 2%, which was expected due to the nearby feedlots serving as a large source of ammonia. Motor vehicle, open burning and coal power plant emissions each only contributed 1% on average.

For $PM_{2.5}$ the highest source contribution in nine of the ten runs was the feedlot soil category with an average of 49% (Fig. 6). The second highest source contribution for nine of the ten runs was the geological soil category with an average of 24%.

The average 24- hour $PM_{2.5}$ concentration at Cowtown was also well above any other site at 67$\mu g/m^3$. Three samples had recorded $PM_{2.5}$ mass concentrations above 100 $\mu g/m^3$, October 21, 27, and November 8, 2003. Coal power plant emissions were the third largest contributor with an average of 9%. Vegetative burning was also a significant contributor with an average of 7%.

Ammonium nitrate with an average contribution of 4% and motor vehicle emissions with an average contribution of 3% were the remaining identified source contributions. Both of these sources were expected to have a larger contribution due to the nearby feedlots serving as a source of ammonia and the proximity of a heavily traveled road.

Pinal County Housing

The Pinal County Housing site was selected due to it being in the heart of the agricultural basin. The site is surrounded by desert beyond which open field agriculture dominates the landscape for at least ten miles in all directions. The average 24-hour PM_{10} concentration for this site during the project was 60 $\mu g/m^3$.

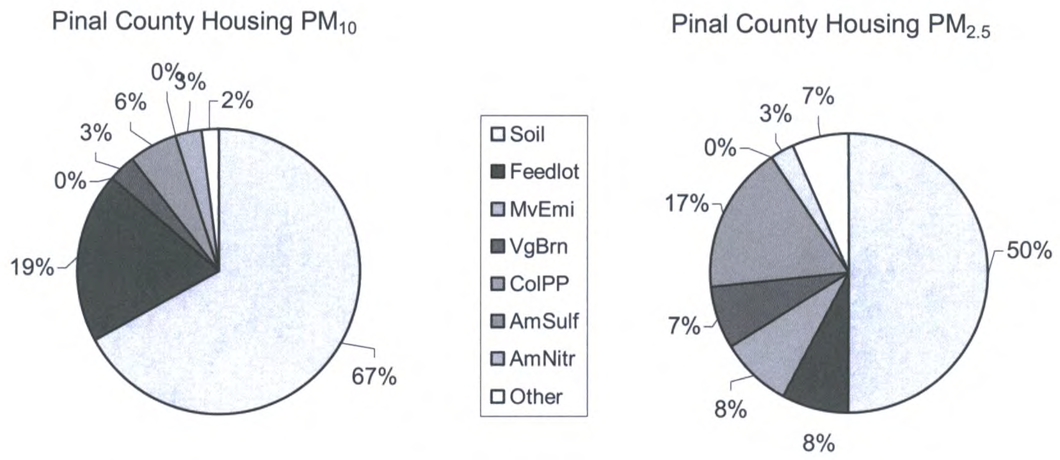


Figure 7. Average PM_{10} and $PM_{2.5}$ CMB modeled source attributions for Pinal County Housing

For PM_{10} the highest source contribution in all ten runs was the geological soil category with an average contribution of 67% (Fig. 7). The second highest source contribution for all the runs was the feedlot soil category with an average of 19%. The nearest dairy is approximately one mile south of the site and three others are approximately five miles west of the site. The feedlot soil emissions are attributed to the large number of dairies and feedlots throughout the agricultural basin, or manure spread over agricultural fields.

Coal power plant emissions made up the third largest category with an average of 6%. The only significant industrial processes nearby are a cotton gin ¾ of a mile to the south and a natural gas power plant approximately 2½ miles north of the site. This power plant operates solely on natural gas and only during peak power demands. Considering there is not a local source that could contribute to this source category, a long-range transport of the coal fired power plant emissions is suggested.

Vegetative burning and ammonium nitrate emissions were the remaining two identified source categories with both averaging 3% for the ten PM_{10} samples. Only two runs had an unidentified component, November 5, and November 8.

For $PM_{2.5}$ geological soil emissions were still the dominant source category with 50% when averaged for the nine samples (Fig. 7). One run was voided due to an equipment failure. Coal power plant emissions made up the second largest category with an average contribution of 17%. As mentioned above a long-range transport mechanism of power plant emissions may be responsible.

Motor Vehicle emissions averaged 8% and feedlot soil emissions 8% for the nine samples. October 30, 2003 had a large contribution identified as feedlot emissions with 39.9%, being the dominant modeled source for that day.

Vegetative burning and ammonium nitrate were the remaining two identified source contributions with an average of 7% and 3% respectively. As discussed previously vegetative burning was an expected source at all the sites. The Pinal County Housing site is located within a fenced area that houses the sewer lift station for the nearby County Housing subdivision. Even though this does create a localized source of ammonia it does not appear to have affected the source apportionment for the site. Ammonium sulfate was not identified in either the PM_{10} or $PM_{2.5}$ source apportionment and ammonium nitrate concentrations were comparable to all the other sites.

Stanfield

The Stanfield site is close to several feedlots and dairies. The community of Stanfield extends ¼ to ½ a mile in all directions from the site, beyond which open field agriculture dominates. Sizeable feedlot operations and dairies lie approximately three miles to the north, east, and west. The average 24-hour PM_{10} concentration at this site was 55 $\mu g/m^3$.

Sources identified from the quartz filter analysis, such as ammonium nitrate and ammonium sulfate, were not accounted for on October 15, 2003 due to a sampler failure. This explains why this is one of the two PM_{10} samples that did not show any ammonium nitrate.

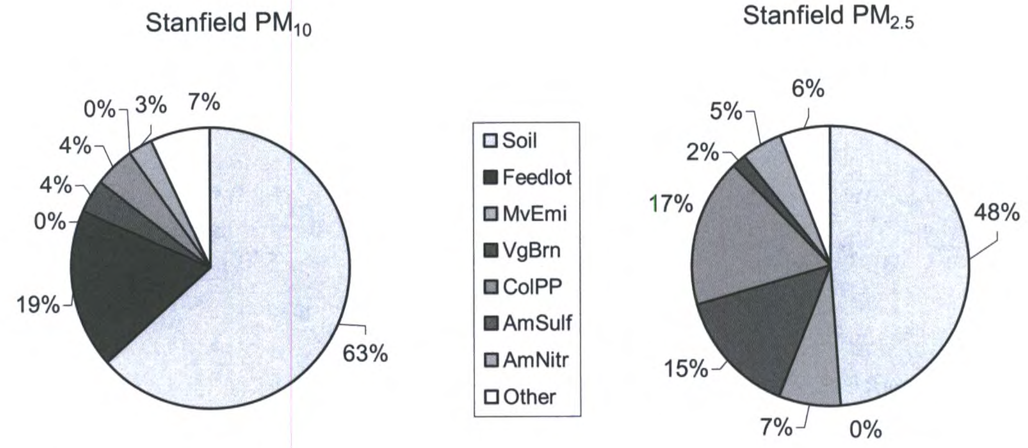


Figure 8. Average PM_{10} and $PM_{2.5}$ CMB modeled source attributions for Stanfield

For PM_{10} the highest identified source contribution in all ten runs was the geological soil category with an average contribution of 63% (Fig. 8). The second highest contribution for all the runs was the feedlot soil category with an average of 19%. Several dairies and feedlots are located approximately three miles to the north, east and west of the site.

Coal power plant and vegetative burning emissions made up the third and fourth largest categories, each with an average of 4%. There are no local industrial sources in the area that could be attributed to the coal power plant category. Ammonium nitrate was the remaining identified source category with an average of 3%. Both vegetative burning and ammonia nitrate emissions are expected, given the local conditions.

For $PM_{2.5}$, geological soil emissions were the dominant source category with 48% when averaged over ten samples (Fig. 8). Coal fired power plant emissions made up the second largest category with an average contribution of 17%. In nine of the ten runs coal power plant emissions were the second largest source and on October 12, 2003 coal power plant emissions were the dominant source with 22.8%.

Vegetative burning also made up a significant portion of the source attribution with 15% when averaged over the ten runs. Vegetative burning was the largest source contribution for October 30, 2003 and it is likely that on this day a local brush disposal fire contributed this amount of $PM_{2.5}$

Motor vehicle emissions made an average contribution of 7%, ammonium nitrate 5%, and ammonium sulfate 2%. This site is located adjacent to the Stanfield Fire Department and it is possible that the diesel emissions from an idling fire engine contributed to the motor vehicle emissions. The large number of feedlots and dairies in the area would serve as a source of ammonia for both the identified ammonium sulfate and ammonium nitrate components.

Conclusions

Mass measurements at five air quality monitoring sites in Pinal County over the period October 1 to November 8, 2003 showed elevated levels for both PM_{10} and $PM_{2.5}$. The Cowtown site (close to several feedlots) was the highest, being on average 127 $\mu g/m^3$ for PM_{10} and 60 $\mu g/m^3$ for $PM_{2.5}$ for this intensive sampling period.

Chemical fingerprints (source profiles) for three new sources from Pinal County are presented, namely Agricultural Dust, dirt Road Dust, and Feedlot Dust. Since most of the development in the area is being conducted on retired agricultural fields a construction source profile was not attempted. The Feedlot source profile differed significantly from the other two profiles in that it contained an elevated total carbon component. Since manure emissions could be distinctly identified a feedlot source profile was integrated into the modeling.

Source attribution modeling using the CMB receptor model showed that Soil (Agricultural + Dirt Road + Construction) was by far the greatest contributor to both PM_{10} and $PM_{2.5}$ at four of the five sites. Feedlot dust was the second largest contributor for PM_{10} at four sites, the only exception was Cowtown where Feedlot dust was shown to be the greatest at 59% for PM_{10} and 49% for $PM_{2.5}$. Minor modeled source types include Coal Fired Power Plant, Vegetative Burn, Motor Vehicle Emissions, Secondary Ammonium Nitrate and Secondary Ammonium Sulfate.

References

Chow, J.C., J.G. Watson, H. Kuhns, V. Etyemezian, D.H. Lowenthal, D. Crow, S.D. Kohl, J.P. Engelbrecht, and M.C. Green (2004). Source profiles for industrial, mobile, and area sources in the Big Bend Regional Aerosol Visibility and Observational study. *Chemosphere,* 54(9), 185-208.

Friedlander, S.K. (1973). Chemical element balances and identification of air pollution sources. *Environ.Sci.Technol.* 7:235-40.

Walch, K.G., Gabrielson, D.G., (2005). Pinal County Air Quality Control District Source Apportionment Study, Pinal County Air Quality Control District Publication.
http://www.co.pinal.az.us/AirQual/pdf/pinal_speciation_study.pdf (December 30, 2005)

Watson, J.G., Robinson, N.F., Lewis, C.W., Coulter, C.T., Chow, J.C., Fujita, E.M., Lowenthal, D.H., Corner, T.L., Henry, R.C., and Willis, R.D..(1997). *Chemical mass balance receptor model version 8 (CMB) users manual,* prepared for U.S. EPA, Research Triangle Park, NC, by Desert Research Institute, Reno, NV.

Dry Deposition of Ammonia in the Vicinity of a Swine Production Facility

John T. Walker[1] and Wayne Robarge[2].

[1]US Environmental Protection Agency, Research Triangle Park, NC 27711, USA; [2]North Carolina State University, Department of Soil Science, Raleigh NC, 27695, USA.

Abstract

While animal production facilities have been identified as important sources of atmospheric NH_3, there are no estimates of local NH_3 dry deposition for U.S. sites. Such estimates are necessary to estimate net farm-scale emissions and for assessing the risk of neighboring ecosystems to nitrogen deposition. This project investigates the dry deposition of NH_3 around a 5000 head swine production facility located in eastern North Carolina. Passive samplers are used to measure weekly-integrated NH_3 concentrations along horizontal gradients from the lagoon/barn complex out to a distance of 500 m. Dry deposition is estimated using a canopy compensation point model. Here we present data for the period April, 2003 to August, 2005 with emphasis on modeling results and uncertainty. Dry deposition rates range from 200 kg NH_3-N/ha/yr within 25 m of the lagoon/barn complex to 5 kg NH_3-N/ha/yr at a distance of 500 m. Assuming a steady-state emission factor of 6.0 kg NH_3/animal/yr, NH_3 dry deposition over the nearest 500 m from the barn/lagoon complex accounted for 11.6 (± 3.5)% of emissions between July, 2003 and July, 2004.

Study of Gas/Particle Partitioning Using Inorganic Thermodynamic Equilibrium Modules and Data from the California Regional $PM_{10}/PM_{2.5}$ Air Quality Study

Kai Wang[1], Yang Zhang[1], Mark Z. Jacobson[2], Jinyou Liang[3], and Karen Magliano[3]
[1]North Carolina State University, Raleigh, NC 27695
[2]Stanford University, Stanford, CA 94305
[3]California Air Resources Board, Sacramento, CA 95812

Abstract

Three thermodynamic equilibrium modules, ISORROPIA, AIM2, and EQUISOLV II are applied to simulate the gas/particle portioning for total amount of nitrate, ammonium, and chloride using observational data from the 1999-2001 California Regional PM_{10}/PM_{25} Air Quality Studies (CRPAQS). The gas/PM partitioning predicted by ISORROPIA and EQUISOLV II with a bulk equilibrium approach is quite similar and both modules can generally reproduce the observed partitioning. The observed size distributions are reproduced by EQUISOLV II with a size-resolved equilibrium approach for some cases but large deviations exist for other cases, due in part to the uncertainties in the size-resolved aerosol measurements. The likely causes for the deviations are being identified through diagnostic and sensitivity studies.

Introduction

Both the natural and anthropogenic particulate matter (PM) are known to play a significant role in many air pollution issues such as human heath problems, climate change, atmospheric visibility, stratospheric ozone depletion, acid deposition, and photochemical smog. Therefore, the knowledge of the physical and chemical properties, compositions, and processes of the aerosol particles is of great importance. The chemical compositions and formation mechanisms of PM are different for different size sections. Modeling the chemical composition and size distribution will improve our understanding of aerosol behavior and its impacts on air quality and climate change. In the past years, many inorganic and organic aerosol modules, most of which assume a thermodynamic equilibrium between the gas and aerosol phases for condensable species, have been developed to simulate the composition and phase state of PM. These include: EQUIL (Bassett and Seinfeld, 1983), MARS (Saxena et al., 1986), SEQUILIB (Pilinis and Seinfeld, 1987), AIM and AIM2 (Wexler and Seinfeld, 1991; Clegg et al., 1992, 1998), SCAPE and SCAPE2 (Kim et al., 1993a, b), EQUISOLV and EQUISOLV II (Jacobson et al., 1996; Jacobson, 1999), ISORROPIA (Nenes et al., 1998, 1999), and GFEMN (Ansari and Pandis, 1999a). These modules have been used in determining emission control strategies (e.g., Kumar et al., 1998), replicating ambient measurements (e.g., Fridlind and Jacobson, 2000; Moya et al., 2002; Campbell et al., 2002), and simulating PM in large-scale chemical transport models (e.g., Meng et al., 1998; Rodriguez and Dabdub, 2004; Zhang et al., 2004). A comprehensive review and performance evaluation of some of these modules have also been conducted in several studies (e.g., Ansari and Pandis, 1999b; Zhang et al., 2000).

In the present study, three aerosol equilibrium modules: ISORROPIA, AIM2, and EQUISOLV II are applied to study the gas/particle portioning of inorganic species using observational data from the 1999-2001 California Regional $PM_{10}/PM_{2.5}$ Air Quality Studies (CRPAQS). CRPAQS is a multi-year observational and modeling study that involves multiple organizations and is being conducted in northern California with a focus on the Central Valley, where the concentrations of PM frequently exceed the National Ambient Air Quality Standards, owing to the combination of mobile, industrial, agricultural, and residential sources (McDade, 2002). A comprehensive data analysis has been conducted using data from two CRPAQS sites (Angiola and Fresno) for the period of December 14, 2000 to January 18, 2001 during which PM concentrations (especially nitrate and ammonium) were the highest throughout the year. Two sets of observational data are used to set up inputs for model simulations for the two sites. The first set contains PM measurements (i.e., sulfate (SO_4^{2-}), nitrate (NO_3^{-}), ammonium (NH_4^{+}), chloride (Cl^{-}), sodium (Na^{+}), calcium (Ca^{+}), potassium (K^{+}), and magnesium (Mg^{2+})) that were obtained 5 times per day (3-8 hours average) using sequential filter sampler with Teflon/citric acid filter pack and corresponding gaseous measurements of nitric acid (HNO_3) and ammonia (NH_3) (referred to as episodic cases). The second set contains 24-hour average size-resolved PM measurements of the same species that were obtained using the Micro Orifice Uniform Deposit Impactor (MOUDI) samplers for nine stages over size range of 0.01-10 μm and corresponding gaseous HNO_3 and NH_3 (referred to as size-resolved cases). A total of 104 episodic cases (50 and 54 at Angiola and Fresno, respectively) and 8 size-resolved cases (4 at each site) are simulated.

In this paper, we compare the simulation results of ISORROPIA and EQUISOLV II against the observations. ISORROPIA simulates the partitioning using the bulk thermodynamic equilibrium approach only and it does not simulate equilibrium reactions involving Ca^+, K^+, and Mg^{2+}. EQUISOLV II simulates the partitioning using both the bulk and the size-resolved thermodynamic equilibrium approaches (referred to as bulk EQUISOLV II and size-resolved EQUISOLV II). The inputs for ISORROPIA and bulk EQUISOLV II include temperature (T), relative humidity (RH), total amount of species in the gas and aerosol phases such as total sulfate (TSO_4), total nitrate ($TNO_3 = HNO_3 + NO_3^-$), total ammonium ($TNH_4 = NH_3+NH_4^+$), and total chloride ($TCl = HCl+Cl^-$). Total amounts of sodium (Na^+), calcium (Ca^+), potassium (K^+), and magnesium (Mg^{2+}) are also included in the inputs for the bulk EQUISOLV II simulations. The inputs for size-resolved EQUISOLV II include T, RH, TNO_3, TNH_4, and size-resolved SO_4^{2-}, Cl^-, Na^+, Ca^+, K^+, and Mg^{2+}. A comparison is also performed between the simulation results of ISORROPIA and bulk EQUISOLV II and the results of bulk vs. size-resolved EQUISOLV II. The simulations using AIM2 are being set up and the results will be compared with those of ISORROPIA and EQUISOLV II and the CRPAQS observations. The performance of these thermodynamic modules in reproducing observed PM concentrations, size distributions, and gas-PM partitioning will be evaluated.

Summary of the Simulation Results

Three sets of simulations have been performed to simulate the partitioning of TNO_3, TNH_4, and TCl between the gas and aerosol phases: 1) bulk equilibrium simulations with ISORROPIA and bulk-EQUISOLV II for the 104 episodic cases; 2) bulk equilibrium simulations with ISORROPIA and bulk-EQUISOLV II for the 8 size-resolved cases with total amounts of species as inputs; and 3) size-resolved simulations with size-resolved EQUISOLV II for the 8 size-resolved cases with size-resolved PM composition as inputs except NO_3^- and NH_4^+ which are inputted together with their corresponding gases as TNO_3 and TNH_4.

For the bulk equilibrium simulations with ISORROPIA and bulk EQUISOLV II for the 104 episodic cases, ISORROPIA and bulk EQUISOLV II generally predict similar gas/PM partitioning of TNO_3, TNH_4, and TCl. Since all 104 cases are in sulfate-very poor (or ammonia-very rich) regime with the molar ratios of TNH_4 to TSO_4 (i.e., $[TNH_4]/[TSO_4]$) of 15-760 and more than 70% of those cases have high RHs (> 70%), this similarity is expected and is consistent with the findings of Ansari and Pandis (1999b) and Zhang et al. (2000). Some differences exist in the concentrations of H_2O, HCl, and H^+ predicted by both modules. At RH = 100%, both modules give extremely high H_2O amount ($> 10^5$ μg m^{-3} for ISORROPIA and $> 10^3$ μg m^{-3} for EQUISOLV II), indicating the limitation of the activity coefficients at RH = 100% used in both modules. ISORROPIA predicts higher (by a factor up to 4) concentrations of HCl than EQUISOLV II for almost all cases. In most cases, the aqueous particle predicted by ISORROPIA is less acidic than that predicted by bulk EQUISOLV II (with an average pH value of 5.07 from ISORROPIA vs. 4.09 from EQUISOLV II for the 104 episodic cases). Those differences are mainly caused by the different sets of equilibrium reactions simulated in both modules. The module capability to reproduce the observed gas/PM partitioning differs from site to site, depending on chemical and meteorological conditions at the site. The average temperature for the 50 episodic cases at Angiola is 13.7 °C, which is higher than that (9.4 °C) for the 54 episodic cases at Fresno. The average RH at Angiola is 84%, which is also higher than that at Fresno (72%). The observed average $[TNH_4]/[TSO_4]$ and $[TNO_3]/[TSO_4]$ are 112.6 and 11.8 at Angiola and 52.3 and 18.5 at Fresno, indicating that both sites are in sulfate-very poor (or ammonia-very rich) and nitrate-rich condition. For aerosol NO_3^-, 95% of predictions by ISORROPIA and bulk EQUISOLV II are within a factor of 1.5 of the observed values at both sites, with the correlation coefficient (*r*) of 0.8-0.81 at Angiola and 0.99 at Fresno. Both modules tend to slightly overpredict NO_3^- at Fresno and may either overpredict or underpredict NO_3^- at Angiola. For HNO_3, less than 20% of predictions by both modules at both sites are within a factor of 1.5 of the observed values, with very low *r* values of 0.14 at Angiola and 0.25-0.26 at Fresno. For aerosol NH_4^+, more than 95% of predictions by both modules at Fresno and only 70% of predictions at Angiola are within a factor of 1.5 of the observed values, with *r* values of 0.82 at Angiola and 0.96 at Fresno. Both modules reproduce well NH_4^+ at Fresno but tend to significantly underpredict NH_4^+ at Angiola. For NH_3, more than 90% of predictions by both modules at both sites are within a factor of 1.5 of the observed values, with *r* values of 0.92 at Angiola and 0.69 at Fresno. Both modules reproduce well NH_3 at Fresno but tend to overpredict NH_3 at Angiola. For aerosol Cl^-, both modules reproduce well with more than 95% of predictions within a factor of 1.5 of the observed values and an *r* value of nearly 1.0 at both sites.

For the bulk equilibrium simulations with ISORROPIA and bulk-EQUISOLV II for the 8 size-resolved cases, in most cases, the simulations and measurements are in excellent agreement for NH_3 and Cl^-, with a small overprediction by 0.9%, 2.7%, and 1.2% for NH_3 and a small underprediction by 3.3%, 1.3%, 2.5% for Cl^- for ISORROPIA, bulk EQUISOLV II, and size-resolved EQUISOLV II, respectively. ISORROPIA, bulk EQUISOLV II, and size-resolved EQUISOLV II also give good agreement for NO_3^- and NH_4^+, with a moderate overprediction by 13.8%, 14.1%, and 13.8% for NO_3^- and a moderate underprediction by 11.8%, 35%, and 16.2% for NH_4^+, respectively. Similar to the episodic cases, all modules fail to reproduce HNO_3, with a significant underprediction by 95.4%, 97.5%, and 95.4% for ISORROPIA, bulk EQUISOLV II, and size-resolved EQUISOLV II, respectively. Overall, the performance of ISORROPIA and EQUISOLV II is similar, with the results of size-resolved EQUISOLV II slightly better than bulk EQUISOLV II.

For the size-resolved EQUISOLV II simulations for the 8 size-resolved cases, the module reproduces the observed size distributions of NO_3^-, NH_4^+, and Cl^- for some cases but with large deviations for other cases. The observed NH_4^+ exists predominantly in the fine/accumulation modes and the observed NO_3^- exists in the accumulation/coarse modes, which are generally reproduced by the module. The performance of the module is generally better in the fine/accumulation modes than the coarse mode, as the thermodynamic equilibrium assumption holds primarily for fine PM but not for coarse PM. The growth rate to small particles, multiplied by the number concentration of small particles, is greater than that to large particles, multiplied by the number concentration of large particles. The rate of mass transfer between gases and small particles is thus generally faster than that between gases and large particles, rendering the equilibrium assumption a valid simplification in many cases. The simulated size distributions of NO_3^- are very similar to those of H_2O and the predicted NO_3^- concentrations are higher in the ammonia-rich size sections than those in the ammonia-poor size sections. One major factor leading to large deviations is likely associated with the large uncertainties in the size-resolved measurements. For example, some size bins (e.g., the first size bin (0.01 – 0.062 μm) and the second size bin (0.062 – 0.163 μm)) contain NH_4^+ and Mg^{2+} but no SO_4^{2-}, NO_3^-, and Cl^- in some cases, causing unrealistic charge-imbalance in the observations. All 8 samples contain excess cations that are not balanced. Other likely causes for the discrepancies between the simulated and observed size distributions are being identified through diagnostic and sensitivity studies.

Acknowledgements

This work is performed under the National Science Foundation Career Award No. ATM-0348819 and the Environmental Protection Agency Contract No. 4-92U-9609 at NCSU. The authors thank Athanasios Nenes of Georgia Institute of Technology, for providing the ISORROPIA source code; Simon Clegg of the University of East Anglia, for providing the AIM2 executable files; and Gregory O'Brien of California Air Resources Board, for providing the CRPAQS observational data.

References

Ansari, A.S., S.N. Pandis. 1999a. Prediction of multicomponent inorganic atmospheric aerosol behavior. *Atmospheric Environment* 33: 745-757.

Ansari, A.S., S.N. Pandis. 1999b. An analysis of four models predicting the partitioning of semi-volatile inorganic aerosol components. *Aerosol Science and Technology* 31: 129-153.

Bassett, M., J.H. Seinfeld. 1983. Atmospheric equilibrium model of sulfate and nitrate aerosol. *Atmospheric Environment* 17: 2237-2252.

Campbell, S.W., M.C. Evans, N.D. Poor. 2002. Predictions of size-resolved aerosol concentrations of ammonium, chloride and nitrate at a bayside site using EQUISOLV II. A*tmospheric Environment* 36: 4299-4307.

Clegg, S.L., P. Brimblecombe, A.S. Wexler. 1998. A thermodynamic model of the system H^+ -NH_4^+ -Na+ -SO_4^{2-} -NO_3^- -Cl^- -H_2O at 298.15 K. *Journal of Physical Chemistry* 102: 2155-2171.

Clegg, S.L., K.S. Pitzer, P. Brimblecombe. 1992. Thermodynamics of multicomponent, miscible, ionic solutions. II. Mixture including unsymmetrical electrolytes. *Journal of Physical Chemistry* 96: 9470-9479.

Fridlind, A.M. and M.Z. Jacobson. 2000. A study of gas-aerosol equilibrium and aerosol pH in the remote marine boundary layer during the First Aerosol Characterization Experiment (ACE 1). *Journal of Geophysical Research* 105: 17325-17340.

Jacobson, M.Z., A. Tabazadeh, R.P. Turco, 1996. Simulating equilibrium within aerosols and nonequilibrium between gases and aerosols. *Journal of Geophysical Research* 101: 9079-9091.

Jacobson, M.Z. 1999. Studying the effects of calcium and magnesium on size-distributed nitrate and ammonium with EQUISOLV II. *Atmospheric Environment* 33:3635-3649.

Kim, Y.P., J.H. Seinfeld, P. Saxena, 1993a. Atmospheric gas-aerosol equilibrium I. Thermodynamic model. *Aerosol Science and Technology* 19: 157-181.

Kim, Y.P., J.H. Seinfeld, P. Saxena, 1993b. Atmospheric gas-aerosol equilibrium II. Analysis of common approximations and activity coefficients calculation methods. *Aerosol Science and Technology* 19: 181-198.

Kumar, N., F.W. Lurmann, S.N. Pandis, A.S. Ansari. 1998. Final Report: analysis of atmospheric chemistry during 1995 integrated monitoring study. Final Report Prepared for the California Air Resources Board, Sacramento, CA, by Sonoma Technology Inc., Santa Rosa, CA.

McDade, C.E. 2002. California Regional $PM_{10}/PM_{2.5}$ Air Quality Study (CRPAQS) Introduction to Site Documentation Reports. California Regional PM_{10}/PM_{25} Air Quality Study Technical Committee c/o California Air Resources Board Sacramento, CA. by ENSR International Camarillo, CA.

Meng, Z., D., Dabdub, J.H. Seinfeld. 1998. Size-resolved and chemically resolved model of atmospheric aerosol dynamics. *Journal of Geophysical Research* 103: 3419-3435.

Moya, M., S.N. Pandis, and M.Z. Jacobson. 2002. Is the size distribution of urban aerosols determined by thermodynamic equilibrium?: An application to Southern California. *Atmospheric Environment* 36: 2349-2365.

Nenes, A., C. Pilinis, and S.N. Pandis. 1998. ISORROPIA: a new thermodynamic equilibrium model for multiphase multicomponent marine aerosols. *Aquatic Geochemistry* 4: 123-152.

Nenes, A., C. Pilinis, and S.N. Pandis. 1999. Continued development and testing of a new thermodynamic aerosol module for urban and regional air quality models. *Atmospheric Environment* 33: 1553-1560.

Pilinis, C., J.H. Seinfeld. 1987. Continued development of a general equilibrium model for inorganic multicomponent atmospheric aerosols. *Atmospheric Environment* 21: 2453-2466.

Rodriguez, M.A. and D. Dabdub. 2004. IMAGES-SCAPE2: A modeling study of size- and chemically resolved aerosol thermodynamics in a global chemical transport model. *Journal of Geophysical Research* 109: D02203.

Saxena, P., A.B. Hudischewskyj, C. Seigneur, J.H. Seinfeld. 1986. A comparative study of equilibrium approaches to the chemical characterization of secondary aerosols. *Atmospheric Environment* 20: 1471-1483

Wexler, A.S., J.H. Seinfeld. 1991. Second-generation inorganic aerosol model. *Atmospheric Environment* 25A: 2731-2748.

Zhang, Y., C. Seigneur, J.H. Seinfeld, M. Jacobson, S L. Clegg, and F.S. Binkowski. 2000. A comparative review of inorganic aerosol thermodynamic equilibrium modules: similarities, differences, and their likely causes. *Atmospheric Environment* 34: 117-137.

Zhang, Y., B. Pun, K. Vijayaraghavan, S.-Y. Wu, C. Seigneur, S. Pandis, M. Jacobson, A. Nenes, and J. H. Seinfeld, 2004, Development and application of the model of aerosol dynamics, reaction, ionization and dissolution (MADRID), *Journal of Geophysical Research* 109, D01202, doi:10.1029/2003JD003501.

Nitrogen Mass Balance for Spray Fields Fertilized with Liquid Swine Waste

S.C. Whalen[1] and J.T. DeBerardinis[2]
[1]University of North Carolina, Department of Environmental Sciences and Engineering,
Chapel Hill, NC 27599, USA
[2]North Carolina Department of Environment and Natural Resources, Division of Water Quality,
Raleigh, NC 27607, USA

Abstract

Increasingly, livestock is produced in the US in concentrated animal feeding operations (CAFOs), where large-scale confinement facilities house several thousand animals and collected waste is used as an organic fertilizer. In particular, eastern North Carolina has experienced the rapid proliferation of swine-producing CAFOs in the period 1980 to 2000 and stagnant growth thereafter. The current swine inventory in North Carolina stands at about 10 million animals concentrated in roughly 1500 production facilities. Waste at these facilities is collected in open-air lagoons and the liquid phase is land-applied by sprinkler irrigation. Accelerated eutrophication of coastal and inland waters and localized reports of groundwater contamination have centered attention on the fate of nitrogen in land-applied waste.

Although numerous investigations have focused on individual aspects of the fate and transport of nitrogenous liquid waste, none has attempted a detailed analysis of post-application transformations and losses of liquid waste-N. However, the post-application fate of N in land-applied liquid swine waste may involve several storage pools, transfers and transformations (Fig. 1). Nitrogenous material in anaerobic lagoonal swine waste consists of the inorganic reduced species NH_3-N and NH_4^+-N (=TAN; total ammoniacal nitrogen) as well as particulate and dissolved organic-N. The NH_3 fraction of TAN is immediately available for loss to the atmosphere via volatilization. Organic-N can be microbially mineralized to NH_4^+. Ammonium can engage in soil cation exchange reactions, volatilize to the atmosphere as NH_3 under appropriate conditions of soil pH, or oxidize to NO_3^--N through the action of nitrifying bacteria in aerobic soil zones. Nitrate can be reduced to N_2 (and to a much lesser extent N_2O) by denitrifying bacteria in anaerobic microzones, with the gaseous end products again lost to the atmosphere. Among other factors, rates of denitrification are dependent on microbial release of NH_4^+-N from mineralization of organic materials and subsequent oxidation of NH_4^+-N to NO_3^--N by nitrifying bacteria (nitrification) in aerobic soil zones. Inorganic-N (NO_3^- and NH_4^+) is available for consumption by microbes (immobilization) or plants and all organic and inorganic species are subject to export from the ecosystem via surface and ground waters, although NO_3^- shows the most mobility. Our study was aimed at providing an event-based (single application) analysis of physicochemical and biological interactions involved in soil N cycling dynamics and soil-atmosphere exchange of gaseous N that are mediated by each spray event. A comprehensive understanding of the post-application fate of fertilizers is essential for the development of Best Management Practices (BMPs) that aim to minimize offsite transport and maximize nutrient availability to the host crop.

On three occasions, we experimentally applied liquid lagoonal swine waste at typical doses of 1.25 and 2.5 cm (40 to 130 kg N ha^{-2}) to carefully defined plots in an active spray field on a on a corporate, farrow to half finish swine production facility located in eastern NC. A fifth, unfertilized plot served as a control. Waste collection and handling at the facility follow industry standards with the liquid phase applied by solid set sprinkler to a summer crop of coastal bermudagrass (*Cynodon dactylon L.*) that is overseeded with tall fescue (*Festuca* sp.) in the winter. Prior to experimentation, vegetation within each plot and the surrounding area was cut to a uniform height of 18 cm. Soil N pools and transformations were assessed for up to 19 d following waste addition. As necessary, soils were collected to a depth of 20 cm ("active soil zone") within plots with standard coring devices and cores were replaced with similar size cores from adjoining nonexperimental plots. These companion plots were fertilized at rates consistent with the experimental plots and were used expressly for the purpose of replacing removed soils from experimental plots.

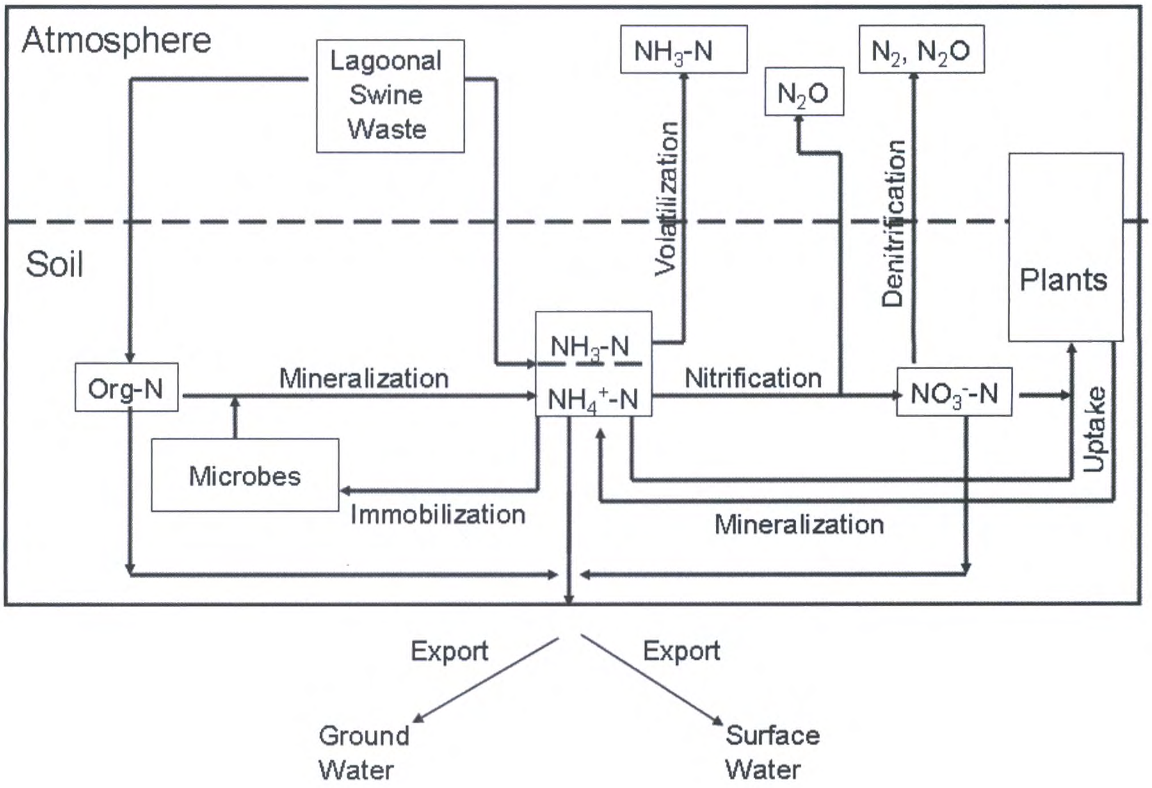


Figure 1. Fate of land-applied liquid swine effluent.

Soil cores returned to the laboratory were homogenized and analyzed for NO_3^--N, NH_4^+-N and basic physical properties following standard methodologies. Liquid swine waste and plant material were assayed for total-N (TN) by persulfate oxidation and dry combustion, respectively. Soil moisture and temperature were measured continuously in the active soil zone of all plots during each experimental period while rainfall was determined with a tipping bucket rain gauge. Ammonia volatilization was measured using a dynamic chamber technique. Rates of denitrification ($N_2 + N_2O$ production) were determined with the C_2H_2-block technique on soil cores periodically removed from each plot. Microbial biomass was determined on soil cores by the chloroform fumigation incubation method, while above- and belowground plant biomass-N was determined for soil cores on day 0 and at the termination each experiment. The time course for change in soil inorganic-N pools was assessed by periodic analysis of active zone cores for NO_3^--N and NH_4^+-N, while loss from the active zone was determined following rainfall and at the termination of the experiment by assessing inorganic-N concentrations in 10 cm core sections to a depth of 50 cm.

Swine waste in the three experimental fertilizations averaged 290 mg l^{-1} NH_4^+-N and 389 mg l^{-1} total-N, indicating that 75% of the waste was in the inorganic phase. Nitrate was absent, as waste storage lagoons are anaerobic. The predominance of NH_4^+ in the waste points to a high mineralization efficiency within the lagoon and further ensures that the waste is in a form readily available for plant assimilation, microbial immobilization and nitrification upon application to receiving fields. Moreover, liquid phase fertilizer readily penetrates the soil surface and makes immediate contact with plants and microbes, ensuring rapid post-application processing.

Ammonia volatilization occurred immediately upon field-application of liquid swine waste. Rates of volatilization were clearly dose-related, showing values around 100 and 200 mg NH_3-N $m^{-2}h^{-1}$ shortly after fertilizer application in the low and high dose plots, respectively. Ammonia volatilization returned to low, baseline levels within hours of waste application. Regionally, these highly sandy soils ensure rapid penetration of waste beneath the soil surface, limiting NH_3 volatilization to a brief post-application period.

Soil percent water filled pore space responded immediately to simulated spray events by increasing in value by 15 to 30% (depending on dose) over unamended, control plots. However, these sandy soils rapidly drained following fertilization with an accompanying decline in %WFPS.

Soil NH_4^+-N increased immediately upon waste addition in proportion to fertilizer dose, with values reaching ~25 μg N g_{dw}^{-1} soil (dw = dry weight). A rapid decline in NH_4^+-N was attributed in part to volatilization. However, the simultaneous increase in soil NO_3^--N indicated that liquid swine waste was rapidly nitrified upon application to receiving soils. Repeated fertilization of these spray fields and the immediate availability of a substrate source (NH_4^+-N) following fertilization apparently maintains an active residual population of nitrifying bacteria in spray field soils.

Elevated rates of denitrification were observed within 2 days of fertilization and declined rapidly thereafter (Fig. 4e). Swine waste provides a source of labile-C and promotes the development of anaerobic microzones by encouraging community respiration, both conditions that favor denitrification. However, the NO_3^--N supply to denitrifiers may have been inadequate at the time of fertilizer application, as day zero concentrations were low, at <0.5 μg g_{dw}^{-1} soil (dw = dry weight). Accordingly, denitrification immediately following fertilization was likely closely coupled to the oxidation of NH_4^+-N to NO_3^--N by nitrifiers. Rates of denitrification showed transient increases following each rainfall during the observational period. Highest rates of denitrification were around 5000 μg N m^{-2} h^{-1}.

Microbial biomass showed a consistent pattern of change in all experiments. Fertilization at the high dose resulted in a ~100% increase in microbial biomass at day 3, while a proportionately lesser increase of about 50% was observed in low dose plots. Microbial biomass declined in all plots by the termination of all three experiments, but, relative to the control, remained elevated by about and 22 to 30% and 35 to 50% in the low and high dose treatments.

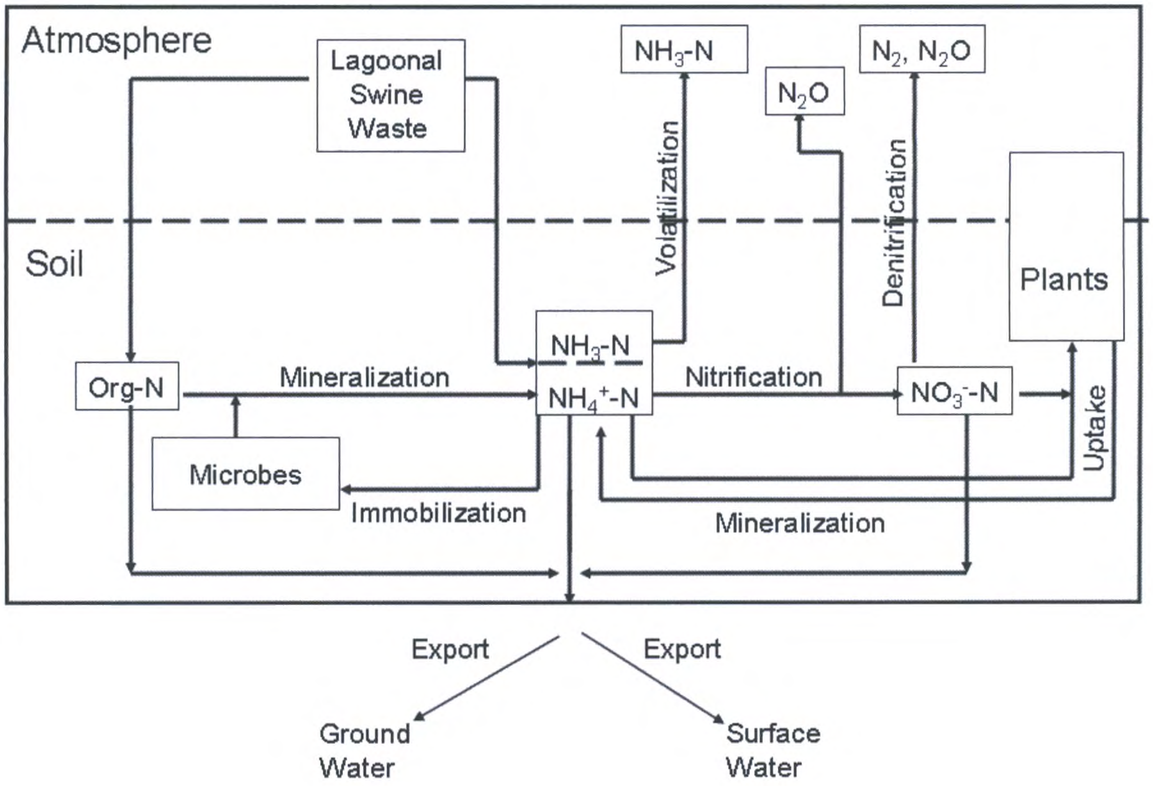
The mass of N immobilized by the end of the ~two week experiments varied from 0.6 to 0.9 g m^{-2} in the low dose treatments and from 0.7 to 1.2 g m^{-2} in the high dose treatments.

Relative to the control, plant biomass also increased in waste-amended plots, but the response was more variable than for microbial biomass. The mass of swine effluent-N assimilated into plant biomass varied from 2.0 g m^{-2} in the low dose treatment of experiment 1 to 9.7 g m^{-2} in the high dose plot of experiment 2. A dose response was evident in experiments 1 and 2, as the mass of effluent-N assimilated into plant biomass was about two-fold or more higher for the high versus low dose plots.

Fertilizer-N was transported below the microbially active surface 20 cm of soil at both loading rates for each experiment. Leaching losses varied over a factor of about 20, with values ranging from 0.2 g N m^{-2} for the low dose in Experiment 3 to 4.5 g N m^{-2} for the high dose in Experiment 2. The mass of N transported below the active zone appeared to be unrelated to rainfall, as largest values were associated with relatively little rainfall (6.4 cm). However, the largest mass losses did correspond with the highest loading rate in all experiments.

We constructed a nitrogen mass balance for the fertilizer-N by assessing rates and transformations in the soil N cycle in post-application observational periods of 14 to 18 d. We consistently recovered more N than applied, by an average of 130%. This was likely due to mineralization of endogenous organic-N, a reservoir that was not assessed. No dose-related response was evident with respect to the overall fate of the applied waste and the distribution among all reservoirs fell within the range of values reported for previous studies focusing on a single N transformation for land-applied liquid swine waste. Thus, liquid swine waste differs from other N fertilizers only in the rate of processing, due its immediate contact with and availability for plants and microbes. It is impossible to assess the error involved in individual terms of the mass balances. We therefore assumed that the error was proportional to the magnitude of each term in scaling down our overall N recovery to force a mass balance (e.g. total = 100%) in an effort to evaluate the fate of land-applied liquid swine waste-N. Roughly half (52%) of the waste-N was sequestered in plant material that can be removed from the site and recycled into livestock, while a smaller percentage (10%) remained in the soil as microbial biomass. A total of 22% was involved in unintended offsite transport. This was about equally divided between losses to the atmosphere via volatilization plus denitrification

(10%) and leaching loss (12%). The 16% of applied waste remaining in soil storage was assumed to be subject in the future to any of these fates.



Instrumentation Development for Monitoring Ammonia Emissions During Nutritional Trials in Dairy Freestall Housing and Under Laboratory Conditions

Eileen F. Wheeler[1], Patrick A. Topper[1], Gabriella A. Varga[2], Neil Brown[2], Victoria Blanes-Vidal[3],
A. Jud Heinrichs[2], Tom L. Richard[1], Robert E. Graves[1], Virginia Ishler[2], Geoffrey I. Zanton[2],
Meghan L. Moody[2]

[1]Agricultural & Biological Engineering, Pennsylvania State University, University Park, PA USA
[2]Dairy and Animal Science, Pennsylvania State University, University Park, PA USA
[3]Polytechnic University of Valencia, Department of Animal Science, Valencia, Spain

Overview

A one-year study assessed ammonia gas emissions from dairy cattle housing during dietary trials that evaluated nutritional modifications that limit nitrogen excretion. These trials were conducted in freestall as well as individual feeding facilities, housing either lactating dairy cows or heifers. Results provide comparisons among dietary strategies that were expected to reduce ammonia emissions. In addition, the results provide essential baseline gas emission data from freestall dairy housing that is lacking in the USA. A sample of collected information is presented here.

Several dietary strategies were developed and their effects on nitrogen balance and ammonia emissions were studied. The nutritional strategy reported in this paper is an evaluation of forage source fed to mature Holstein lactating cows. Forage consisted of corn silage with alfalfa or fescue-orchard grass silage. Each forage contained either coarse or fine ground corn. Two trials with cows in a replicated 4 x 4 Latin square design were conducted. One trial had ten cows in a controlled environment barn. A concurrent second trial was a larger scale version in a freestall barn of 120 cows. Diets were formulated to closely meet the cow's requirement for metabolizable protein such that rations contained 16.0 to 16.5% crude protein. The end result was more optimal nitrogen utilization efficiency, reduced ammonia emissions from housing, improved animal performance, and an indication of increased profits (reported elsewhere).

A mixture of feces and urine is necessary for ammonia formation and emission from dairy manure, with urease from the feces degrading the urea in the urine. In-barn measurements were completed to document the variability in emission rate from a barn environment that is partially dependent on incomplete mixing of feces and urine. To determine ammonia emissions under controlled conditions, experiments were also conducted within a laboratory with a set ratio of urine and feces from the 10 cows on dietary trial in the controlled environment barn.

Traditionally, ammonia emission from livestock housing has been estimated by measuring ventilation rate and ammonia concentration at the inlet and the outlet of the building. However, most dairy cattle are housed in naturally ventilated facilities where it is nearly impossible to accurately monitor emissions along large open sidewalls. Another approach to field measurement of gaseous emissions from animal houses lies in measuring the gas as it volatizes from the floor by means of enclosed flux chambers located over the manure.

Objectives

This paper presents a sample of data from the forage dietary trials of mature cows with an emphasis on instrumentation development to evaluate ammonia gas emissions. The objectives of the gas emissions evaluation work were: (1) To develop a barn-rugged, portable non-steady state (static) flux chamber system and measurement procedure to determine ammonia emissions from naturally-ventilated freestall dairy barns; and (2) To validate this measuring system by comparing flux chamber emission rates from dairy cow manure with whole building emission rates, calculated from ammonia balance in an experimental room; and (3) To determine gas flux mechanisms under controlled laboratory conditions using steady-state flux (dynamic) chamber system for evaluation of a standardized mixture of urine and feces from cows on dietary trials.

Materials and Methods

Methods for Objective 1: Procedures were developed to measure ammonia emissions from curtain-sidewall, naturally-ventilated dairy housing by measuring ammonia as it volatilized *in situ* from the manure-covered floor. Over 80 locations were measured within the freestall environment during a 10-hour experimental period; fifteen experimental days were distributed evenly throughout a 10-month timeframe to capture the variety of weather conditions experienced in a cold-climate like Pennsylvania. During freestall evaluations, air temperature, manure/floor temperature, and air speed near the manure surface were recorded. Instrumentation for in-barn measurements included a non-steady state flux chamber technique (Hutchinson and Mosier, 1981; Livingston G.P. and Hutchinson G.L., 1995) with a photoacoustic multi-gas analyzer (Innova, Denmark) for rapid, accurate gas detection over a 5 to 10 minute interval.

Gas emission from the manure is closely related to the air velocity on its surface, as convective transfer plays an important role in this process (Ni, 1999). Various flux chamber designs have been used for relative comparisons of gas evolution from surfaces. To complete objective 1 of this project, a relative comparison among the diets was an appropriate first step for *in situ* field evaluation of dietary influence on ammonia gas release. In addition, this project sought to quantify actual emissions data.

Methods for Objective 2: For validation of technique, the non-steady state flux chamber methodology was applied to a mechanically-ventilated building, called the experimental test room, for comparison with traditional whole building emissions measurements. Building emission rates can be obtained from the ammonia balance in the building by Equation 1:

$$E_{balance} = Q_{room} \cdot \left(C_{outlet} - C_{inlet}\right) \qquad (1)$$

where $E_{balance}$ is the ammonia emission rate from the building, calculated from the ammonia balance (in $g[NH_3]\cdot s^{-1}$); Q_{room} is the ventilation flow rate in the building (in $m^3 \cdot s^{-1}$); and C_{inlet} and C_{outlet} are the ammonia concentrations (in $g[NH_3]\cdot m^{-3}$) measured at the inlet and the outlet, respectively. Validation experiments were carried out in a positive-pressure ventilated test room with a manure-covered floor. The ammonia flux rate was measured with the non-steady state flux chamber at eight coordinates uniformly placed over the floor. Ammonia emissions from the building calculated from flux chamber measurements ($E_{chamber}$) were obtained as the product of: the averaged measured ammonia flux rate from the floor and the area of the manured surface. Air temperature, manure temperature, and air speed near the manure surface were recorded along with a laboratory analysis of manure pH and nutrients. Evaluations were made with both static and recirculation flow in the chamber. Recirculation flow was set to match test room conditions according to a relationship between flow and surface velocity in the chamber that was previously determined.

Ammonia flux rate from the manure covered by the chamber, f, in $g \cdot m^{-2} \cdot s^{-1}$, was calculated according to the model (Eqn 2) proposed by Hutchinson and Mosier (1981) for non-steady state chambers. This model calculates initial gas flux at the beginning of the sampling period.

$$f = \frac{V_c\left(C_1 - C_0\right)^2}{A_c t\left(2C_1 - C_2 - C_0\right)} Ln\frac{C_1 - C_0}{C_2 - C_1} \qquad (2)$$

where: C_0 is the background gas concentration (in $g[NH_3]\cdot m^{-3}$); C_1 is the gas concentration (in $g[NH_3]\cdot m^{-3}$) at a time t after placing the chamber on the surface; C_2 is the gas concentration (in $g[NH_3]\cdot m^{-3}$) at a time 2t after placing the chamber on the surface; V_c is the volume of the chamber (in m^3) and A_c is the area covered by the chamber (in m^2). In this work, the time (t, in seconds) between measuring C_0 and C_1, was set at 120 seconds. This model is non-linear, as it assumes that the rate of gas exchange is not uniform over the measurement period, but decreases as the gas is accumulated inside of the chamber. It is essential to confirm for each flux measurement that this non-linear assumption is fulfilled, by verifying another simple relation (Eqn 3).

$$\frac{C_1 - C_0}{C_2 - C_1} > 1 \qquad (3)$$

Methods for Objective 3: Additional evaluations for baseline emissions from dairy cow and heifer manure used bench-top, steady-state (dynamic) flux chamber analysis. Measurements were made under controlled conditions with collected urine-feces mixture from animals on the dietary trials. In brief, the dynamic chamber setup used the photoacoustic instrumentation to monitor gas concentration every 20-minutes from five 3.8L glass jars each containing 200 ml of a 50:50 urine:feces mixture. A sixth jar contained water as a control and check for cross-contamination of sampling lines and instrumentation. Mechanisms of gas evolution from manure can be determined from this method and are being investigated in on-going studies.

Results

Results from Objective 1: Results of freestall housing emissions presented in figure 1 were evaluated with the non-steady state flux chamber design. Ammonia emission results were on the lower end of other reported emissions, determined with a variety of techniques in several studies, with sometimes undocumented techniques. Cows on this trial were fed a diet close to the cows' requirement for protein so lower emissions were expected. One of the primary criticisms of chamber measurements is the perturbed micro-environment around the enclosed gas source. Air velocity is likely to be substantially different between the real barn environment and that enclosed in the "static" chamber.

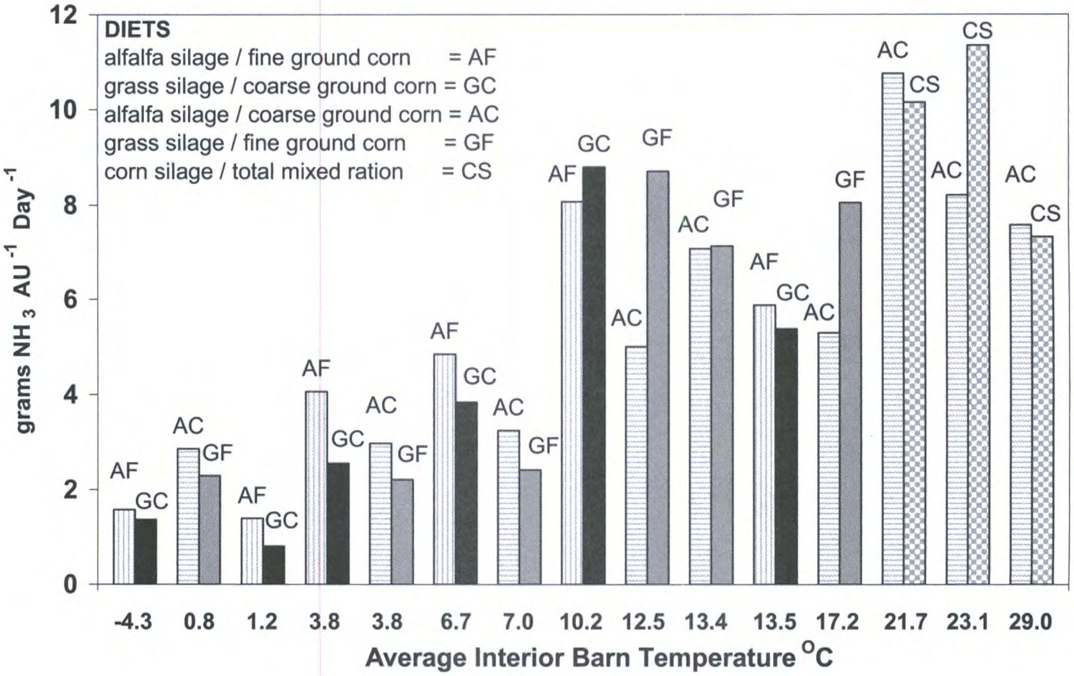


Figure 1. Freestall dairy housing emissions as measured with non-steady state (non-re-circulating) flux chamber during 15 trials over a 10 month period

Results from Objective 2: The addition of chamber air flow recirculation, to provide floor-level velocity similar to that experienced in the local environment, improved the accuracy of gas emission evaluations from the non steady-state chamber method. During initial evaluations of the non-steady state flux chamber design without recirculation airflow, results were not consistent with the measured test room emission rate. The non-recirculation flux chamber design underestimated by roughly 5 to 7 times the actual room emission. This disagreement was thought to be primarily due to a lack of air movement over the enclosed manure surface. Therefore, measuring and replicating air velocity over the manure was hypothesized to be important for achieving good agreement between flux chamber predictions and measured room emission.

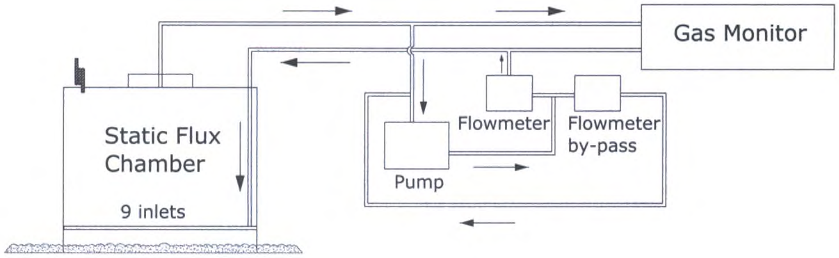


Figure 2. Schematic of non-steady-state (static) flux chamber instrumentation for monitoring ammonia emissions *in situ* in dairy housing with plumbing for periodic sampling (19 sec per minute) of chamber contents and chamber recirculation flow

Figure 3 shows gas concentrations measured in the flux chamber (C_0, C_1 and C_2, in $mg\left[NH_3\right]\cdot m^{-3}$), at eight sites on the experimental test room floor, for five validation experiments. The curvatures of the lines provide visual confirmation that all the flux measurements fulfilled the non-linear assumption of Equation 2. Experiments 1a, 1b and 1c used the same manure in three trials over three days. Manure used in Experiment 1c was monitored 48 hours after being placed on the floor and showed reduced emissions as it aged. Experiments 2a and 2b used the same manure in two trials on the same day.

Ammonia emissions from the experimental test room, calculated by means of the ammonia balance ($E_{balance}$), are presented in Table 1. Also presented are ammonia emissions calculated from the flux chamber measurements ($E_{chamber}$), in terms of average and standard deviation of the measurements at eight sites from the five field validation experiments. Ammonia emission calculated with the flux chamber method was between 9% and 37% lower than ammonia emission calculated from the test room mass balance. Finally, the relation between ammonia emissions calculated from both methods is shown in Figure 4. The coefficient of determination R^2 between them was 0.72.

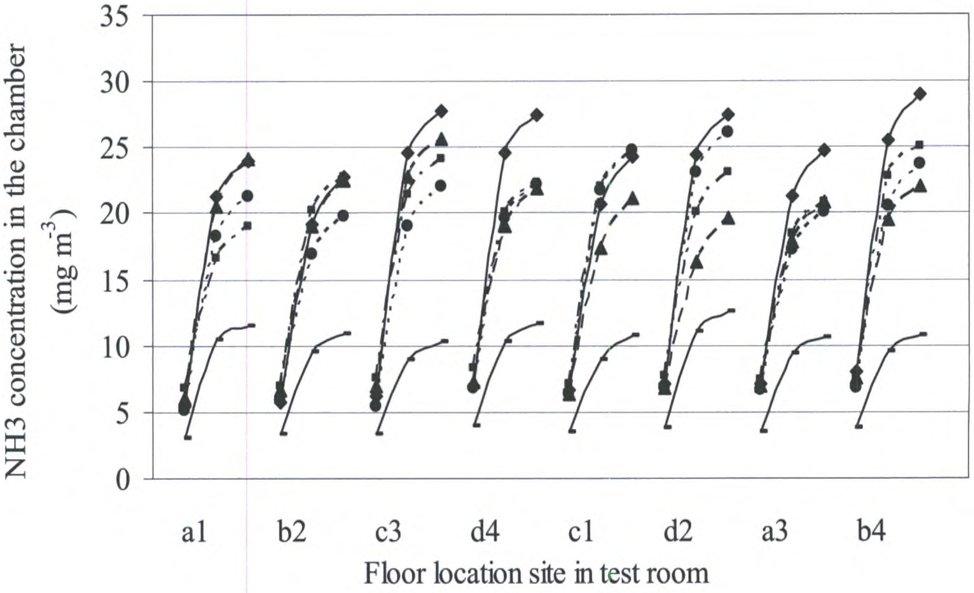


Figure 3. Increase in ammonia concentration over time in the flux chamber at eight floor sites in experimental test room. Locations "a" were closer to air inlet duct while "d" furthest away; locations 1 to 4 moving from one side or room to opposite side.
—◆— Experiment 1a _ _▲_ _ Experiment 1b
——— Experiment 1c ..●... Experiment 2a _ . ■ . _ Experiment 2b

Table 1. Comparison between ammonia emissions calculated from flux chamber measurement and from ammonia balance in the building.

Experiment number	Flux chamber recirculation Flowrate (l min^{-1})	Ammonia emissions, g h^{-1} m^{-2}			Bias %
		From ammonia balance	From flux chamber		
			Average	std.dev	
1a	18.4	0.270	0.246	0.046	9
1b	26.2	0.232	0.169	0.041	27
1c	11.0	0.141	0.092	0.017	35
2a	12.1	0.257	0.196	0.037	24
2b	14.4	0.313	0.197	0.041	37

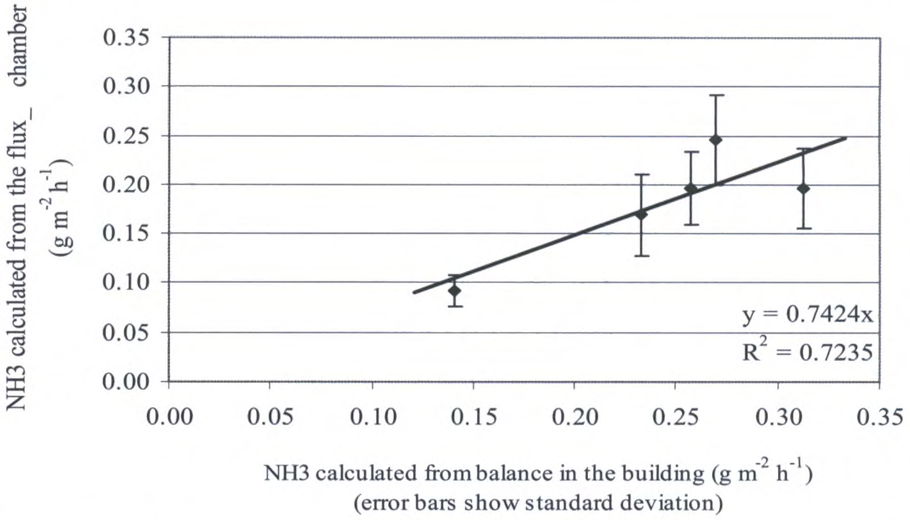


Figure 4. Ammonia emission calculated from the flux chamber and from the gas balance in the building

Several authors have reported validation results for the different methods for measuring ammonia emission rates. According to Richardson (1999), the uncertainties of current methods for determining emission rates from livestock farming sources are estimated to be on the order of 30%. The validation results obtained in this work are comparable to those obtained by previous authors for the other available methods (Demmers et al (2001); Scholtens (2004); Zhang et al. (2005)). However, the flux chamber method presents practical advantages that make it suitable for determining ammonia emission from certain naturally ventilated livestock buildings, such as free stall dairy barns with large open sidewalls, or when a comparison between ammonia flux rates from different surfaces of one single building has to be performed, such as for dairy housing with groups on different diets or manure storage structures. Ongoing work is continuing to improve the agreement between the flux chamber measurements and the experimental test room ammonia balance.

Results from Objective 3: There were 40 data sets evaluated to compare emissions over time from feces:urine samples collected from the cows on alfalfa and grass forage trial diets. A quadratic equation was fitted to each data set to compare the linear and quadratic coefficients among the diet treatments. Preliminary results from the steady-state flux chamber analysis are shown in table 2. There were significant differences among forage treatments with grass-based forage having reduced ammonia release relative to the alfalfa-based forage. (This finding was not clearly identified in the freestall evaluations under Objective 1) The diets with coarse or fine corn were not significantly different in terms of ammonia release.

Table 2. Comparison of curve-fitting parameters for steady-state emission rate versus diet statistical evaluations; values not of use in baseline emission data as presented here

Parameter	Diet Description				p-value		
Forage (F)	Alfalfa		Grass		F	C	F * C
Ground Corn (C)	Fine	Coarse	Fine	Coarse			
Linear Coefficient (ER=**a**T+bT^2)	4.31	4.50	3.14	3.84	0.037	0.28	0.54
Quadratic Coefficient (ER=aT+**b**T^2)	-0.040	-0.047	-0.027	-0.035	0.055	0.25	0.94
Total Ammonia (g/period)	0.215	0.210	0.166	0.199	0.066	0.39	0.23

ER= emission rate

Conclusions

A portable, non-steady state (static) flux chamber can be used to estimate ammonia emissions from natural ventilated buildings, being especially useful for measuring ammonia flux rates from different surfaces within one single building. The accuracy of the developed measurement method in the test room was comparable to those reported in literature for the other available gas emissions methods. Results of the dietary trials for mature cows and heifers will add to the ammonia emissions inventory from U.S. dairy farms while providing insight into potentially useful ammonia reduction strategies.

References

Demmers T. G. M., Phillips V. R., Short L. S., Burgess L. R., Hoxey R. P., Wathes C. M.. 2001. Validation of Ventilation Rate Measurement Methods and the Ammonia Emission from Naturally Ventilated Dairy and Beef Buildings in the United Kingdom. *J. of Agricultural Engineering Research*, Vol 79(1). 107-116.

Hutchinson G. L.; Mosier A.R. 1981. Improved Soil Cover Method for Field Measurement of Nitrous Oxide Fluxes. *Soil Sci. Soc. Am*. J. 45:311-316.

Livingston G.P. and Hutchinson G.L. 1995. Chapter 2: Enclosure-based measurement of trace gas exchange: applications sources of error. In *Biogenic Trace Gases: Measuring Emissions from Soil and Water*, P.A. Matson and R.C. Harriss. Blackwell Science Ltd. Oxford, England.

Ni, J.Q. 1999. Mechanistic models of ammonia release from liquid manure: a review. *Journal of Agricultural Engineering Research*, 72, no.1, pp. 1-17.

Richardson S. 1999. *Atmospheric emission inventory guidebook.* EMEP/CORINAIR, second edition. Prepared by the EMEP Task Force on Emission Inventories. European Environment Agency, Copenhagen.

Scholtens R., Dore C.J., Jones B. M. R., Lee D. S., Phillips V. R. 2004. Measuring ammonia emission rates from livestock buildings and manure stores—part 1: development and validation of external tracer ratio, internal tracer ratio and passive flux sampling methods. *Atmospheric Environment*, Vol 38(19) 3003-3015.

Zhang G., Strøm J.S., Li B., Rom H.B., Morsing S., Dahl P., Wang C. 2005. Emission of ammonia and other contaminant gases from naturally ventilated dairy cattle buildings Biosystems Engineering, Vol 92(3) 355-364.

Suits vs. Solutions: Incorporating Residents Perceptions into Optimal Air Quality Policy

Sara Whitmore, Jennie S. Popp, J. Martin Redfern
University of Arkansas, Fayetteville, Arkansas

Abstract

The Northwest Arkansas (NWAR) and Northeast Oklahoma (NEOK) area has a thriving economy that includes significant poultry production sector with numerous poultry grow-out houses in open-country, feed complexes, hatcheries, and processing plants. Major companies such as Wal-Mart, J.B. Hunt Transportation, and Tyson Foods have their corporate headquarters and expansive facilities in the region. From 1990 to 2000 the Northwest Arkansas area population grew 47%; since then it has grown another 22%. Many of the new residents are moving from urban areas, with little to no previous direct exposure to areas of production agriculture.

Currently, most environmental concerns in the region focus on water quality. However, as populations grow, concerns are turning towards air quality. Western Kentucky provides one example of this. A lawsuit against a poultry growing operation was filed by several residents near the facility, alleging that production management practices, designed to vent air in the poultry houses, led to degradated air for local residents. The court ruled in favor of the residents. Besides the general public looking at air quality, so are government agencies. Earlier this year, the EPA Animal Feeding Operations Consent Agreement recognized "…concerns raised regarding possible health impacts from Animal Feeding Operations (AFO) emissions", such as ammonia, hydrogen sulfide, and particulate matter, and as a result, AFOs may be subject to the requirements of the Clean Air Act and notification process of CERCLA.

This research examines perceptions of residents in the growing NWAR and NEOK region regarding current air quality, the need (if any) to improve air quality and their willingness to pay for this improvement. For the first time, a representative sample of roughly 2000 urban, suburban and rural residents in the region will be interviewed through a mail survey regarding their perceptions about: 1) air quality in general, 2) negative impacts of air quality that may impact them, 3) and how agriculture and other industries impact air quality. This questionnaire will use the contingent valuation methods to determine residents' willingness to pay for improved air quality. In addition to addressing the topics above it is expected that this survey will be able to highlight how differences in perceptions may be related to length of exposure to and knowledge of production agriculture. It is hoped that these results can be used as a catalyst to develop appropriate air quality policy in the region, before it is determined by the courts.

Introduction

The Northwest Arkansas and Northeast Oklahoma area has a thriving economy that includes a significant poultry production sector with numerous poultry grow-out houses in open-country, along with the necessary feed complexes, hatcheries, and processing plants. Other major companies such as Wal-Mart, J.B. Hunt Transportation, and Tyson Foods have their corporate headquarters and other facilities in the area. From 1990 to 2000 the Northwest Arkansas area has grown 47% in population, and in past 5 years has grown 22% (Hawkins, 2005).

With this growth people are moving into the area that are not familiar with production agriculture and agribusiness operations, which can cause many problems between the rural and urban residents. The issue of air quality has become of greater concern to the general public in Northwest Arkansas and Northeast Oklahoma.

Importance of the Study

Much of the concern about environmental issues in Northwest Arkansas and Northeast Oklahoma has been about water quality being impaired by phosphorous from point sources such as municipal waste water treatment plants, or runoff from non-point sources such as construction sites and farm fields. More recently air quality has become an issue of greater concern for governmental agencies as well as the general public.

In western Kentucky a lawsuit against a poultry growing operation was recently resolved; several residents near the facility alleged that when the houses vented air to reduce the level of ammonia gas inside, the air quality outside the houses was reduced, and this reduced air quality impacted them negatively. The court ruled in favor of the residents.

A January 21, 2005, EPA news release stated "The emissions of air pollutants and hazardous substances from certain (animal) feeding operations may be subject to requirements of the Clean Air Act and the notification process of the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA)." The January 31, 2005, Federal Register stated that the "EPA recognizes concerns raised regarding possible health impacts from AFO (Animal Feeding Operation) emissions", such as ammonia, hydrogen sulfide, particulate matter, and volatile organic compounds. In 2001, John Sweeten of Texas A&M University, stated "Until recently, air quality from CAFOs has received only secondary consideration, notwithstanding recently-increased public concerns and policy attention. Water and air quality protection are inseparable, …"

As with the water quality issue, any major change regarding emissions from poultry houses could impose financial costs on poultry farm operators and negatively impact the industry. At the same time, the recreational and retirement industries, which are benefited by natural amenities in the area, might be adversely affected by poorer air quality.

Methodology and Expected Results

The study area was chosen because it is believed that air quality will become a major issue in the next few years. The overall goal of the project is to measure area resident's perceptions of air quality and to measure their willingness to pay for improved air quality. For the first time, a representative sample of roughly 2000 urban and rural residents in the region will be sent a mail survey regarding their perceptions about: 1) air quality in general, 2) negative impacts of air quality that may impact them, 3) how agriculture and other industries impact air quality. In addition, questionnaire will use the contingent valuation methods to determine residents' willingness to pay for improved air quality.

It is also expected that this survey will be able to highlight how differences in perceptions may be related to length of exposure to and knowledge of production agriculture. This survey will provide valuable information that may be used not only to inform the general public, but may also be used as a catalyst to develop appropriate air quality policy in the region.

References

Bergman, Cynthia. 2005: "EPA Announces Air Quality Compliance Agreement for Animal Feeding Operations" *EPA Newsroom.* http://www.epa.gov/newsroom/newsreleases.htm *(*January 23, 2005)

Environmental Protection Agency. January 31, 2005.Animal Feeding Operations Consent Agreement and Final Order. *Federal Register*, Vol. 70, No. 19.

Hawkins, J. Northwest Arkansas Planning Commission, Personal Communication, October 2005.

Sweeten, John. 2001. Animal Production and Air Quality. *Agricultural Outlook Forum.* Texas A&M University Agricultural Research and Extension Center, Amarillo Texas.

United States District Court for the Western district of Kentucky. 2005. Civil Action No.: 4:02 CV-073-M4, Consent Decree.

AGLITE: A Multiwavelength Lidar for Aerosols

T.D. Wilkerson[1], G.E. Bingham[1], V.V. Zavyalov[1], J. Swasey[1], J.J. Hancock[1], B.G. Crowther[1], S.S. Cornelsen[2], C. Marchant[1], J.N. Cutts[1], D.C. Huish[3], C.L. Earl[1], J.M. Andersen[1], and M.L. Cox[1]
[1]Space Dynamics Laboratory, Research Division, North Logan, UT 84341, USA
[2]Campbell Scientific Inc., Logan, UT 84321, USA
[3]Utah State University, Mechanical Engineering Department, Logan, UT 84322, USA

Abstract

Agricultural operations produce a variety of particulates and gases that influence air quality. Agriculture, through wind erosion, tillage and harvest operations, burning, diesel-powered machinery and animal production operations, is a source of particulate matter that can enter human lungs and cause pulmonary problems. These emissions can negatively impact human health, property values, and the environment. The presence of buildings and other structures often make whole facility measurement capability a requirement for understanding the source strength and characteristics. The ability to use standoff methods to determine the movement and concentrations of emissions on a whole facility basis opens new capabilities for model development and verification. We report on the design, construction and operation of a new multiwavelength lidar developed with the Agricultural Research Service of the United States Department of Agriculture and its program on particle emissions from animal production facilities. The lidar incorporates a laser emitting simultaneous, pulsed NdYAG laser radiation at 355, 532 and 1064 nm at a pulse rate of 10 kHz. Lidar backscatter and extinction data are modeled to extract the aerosol information. All-reflective optics combined with dichroic and interferometric filters permit all the wavelength channels to be measured simultaneously, day or night, using photon counting by PMTs, an APD, and high speed data acquisition. The lidar is housed in a transportable trailer for all-weather operation at any accessible site. The laser beams are directed in both azimuth and elevation to targets of interest. We describe application of the lidar in a multidisciplinary atmospheric study at a swine production farm in Iowa. Aerosol plumes emitted from the facility were prominent phenomena, and their variations with temperature, turbulence, stability and feed cycle were studied, using arrays of particle samplers and turbulence detectors. Other lidar measurements focused on air motion as seen by long duration scans of the farm region. Successful operation of this lidar confirms the value of multiwavelength, eyesafe lidars for agricultural aerosol measurements.

The AGLITE Lidar System

The AGLITE lidar instrument used in this study is a three-wavelength lidar system (infrared, visible and ultraviolet emissions) designed and built at the Space Dynamics Laboratory (SDL) under contract from the Agricultural Research Service (ARS) of the United States Department of Agriculture. Figure 1 shows a schematic layout of the lidar system.

The commercial laser unit indicated on the left has four components: (1) a laser diode pump that drives (2) a Nd:YAG crystal laser at 1064 nm, (3) a frequency-doubling crystal to generate 532 nm, and (4) a mixing crystal for frequency-tripled output at 355 nm. These frequencies are employed simultaneously to probe the optical scattering by particles in the atmosphere. The physical properties of particles are then inferred from the observed variation of scattering with wavelength.

Figure 2 shows AGLITE mounted in the trailer used for field work at agricultural sites. Adjustments and final calibrations were most conveniently carried out at night to establish baseline optical signals without daytime interference. An exterior view of the lidar trailer is shown in Figure 3. After final optical adjustment in Utah, the lidar trailer was hauled to Iowa for field trials at a swine production facility, where the lidar was put into operation without needing any further adjustment. Lidar measurements at the Iowa site are also described elsewhere in the Workshop proceedings (Bingham et al., 2006; Hipps et al. 2006; Martin et al., 2006; Silva et al., 2006; Zavyalov et al., 2006).

The AGLITE laser system produces short pulses of light (~10 nsec duration) that are elastically backscattered by atmospheric particles into the steerable telescope, which, along with the laser output, is scanned in elevation and azimuth using the beam director. The Newtonian telescope has a diameter of 28 cm and a FOV of 0.46 mrad. Time resolution of each lidar "return" provides information on the density of particles as a function of distance ("range") from the lidar. The high laser repetition rate of 10 kHz allows the use of low pulse energy for eye safe operations at the close ranges required for agriculture applications. The time interval between successive 10 kHz pulses is long enough that unambiguous lidar range measurements can be made out to a distance of 15 km. The laser specifications on average output power provided by Photonics Industries are 4.1, 0.85 and 1.15 Watts, respectively, at 1064, 532 and 355 nm.

The closest lidar range for usable data is set by the effective entrance aperture of the telescope relative to the divergence of the laser transmission (Measures, 1984). We optically expand the Photonics Industries laser beam to a diameter of 10 mm and beam divergence approximately 0.2-0.3 mrad. This effectively determines our closest useful lidar range to be about 500 meters. Outgoing laser energy is monitored by photo-sensors, and this information is recorded in the data processing unit. The beam-separation unit (upper right, Figure 1) splits the backscattered light into the three wavelength channels appropriate to the laser transmission. Fluorescence observations are not employed in the present work.

For agricultural studies in general, the ability to operate in daytime is essential, because of the roles of sunlight, transpiration, convection and turbulence in the processes of interest. This requires that the lidar signals be observable against the intense daylight solar background. We have adopted photon-counting as the detection regime, in order to detect low intensity returns simultaneously on each channel, consistent with eye safety. Interference filters and an etalon are used in each detector channel to suppress optical cross talk between channels and background skylight, particularly at 532 nm. The detectors chosen for 355 and 532 nm are the photomultiplers (type 9954-A) made by Electron Tubes, Ltd. For the near IR light at 1064 nm we used the avalanche photodiode (SPCM Module) now made by Perkin Elmer, Inc. Approximate limits on the maximum count rate for the detectors are 50 million/sec for the PMTs and 10 million/sec for the APD. Details of the electronic system for AGLITE have been described by Cornelsen (2005).

Pulse counts from the photon counters are read out during each lidar return at a time resolution of 32 – 40 nsec by a digital processing unit. Counts are averaged across a predetermined set of laser pulses, displayed in a real time, and stored for further processing. Typical settings for the lidar operations are the following: time averaging of the return signal of 0.5-3 sec per one measurement (5,000-30,000 laser pulses), range resolution of 5-15 m up to maximum ranges of 0.5-15 km, and azimuth and elevation scans of 0.05-2° per sec. The AGLITE electronic control system automatically coordinates and synchronizes all the functions of the lidar, scanning turret, data acquisition system, digital camera, and weather station to provide a complete data package and makes it available to the operator for further analysis.

Technical details of the design and procedures of the Iowa lidar experiments are described by Zavyalov et al. (2006) in the proceedings of this Workshop. The AGLITE trailer was deployed approximately 650 meters east of the swine barn facility that was the target of the investigation. This location of the lidar system allowed full 3D volume measurements of particulate emission off of the three barn feeding operations from a single observation point.

Application of Lidar to Describe Air Masses and Air Motions

The particle density and motion of selected volumes of air can be visualized in various ways using lidar. The methods described here depend upon the existence of spatial variations of aerosol density and the fact that small aerosol particles are borne along with the wind and turbulent motions. This use of aerosols as tracers for air motion is an important addition to the sampling and analysis of aerosol size and chemistry. When plumes of fine particles are emitted from installations such as the swine barns studied in Iowa, their upward convection and subsequent downwind motion can be seen using sequential lidar scans in altitude and azimuth. Examples of azimuth and elevation scans are also discussed by Zavyalov et al. (2006). In addition, the persistence and motion of cloud and dust layers extending over large areas around a lidar station can be observed by means of azimuth scans at fixed angles of elevation.

Figure 4 shows an azimuth scan of cloud layers taken at 45° elevation, where the ranging capability of the lidar detects thick dust and cloud layers at various altitudes over the azimuth range of ± 20° about the west direction. (Because of the elevation angle, the altitude equals 0.71 times the lidar range.) Here the black/white color scale codes increasing lidar signal strength, and thus the relative aerosol density, as white. Successive azimuthal lidar scans, if they are taken sufficiently quickly, can provide details of the horizontal wind speed and its variation with altitude (Wilkerson et al., 2001). Though cloud and aerosol layers can also be detected using a lidar staring vertically above the lidar site, the simple zenith record is less informative about air motion patterns than the scans in azimuth and elevation.

Figure 5 is a set of elevation scans taken 5.79 minutes apart, showing large plumes of aerosol laden air in the atmospheric boundary layer out to distances of 2000 meters. An important aspect of these scans is the direction of the scan relative to the direction in which the plume is moving. For falling plumes of aerosols, a "down scan" follows the plume motion longer than for an "up scan". Therefore the image of a falling plume is lengthened in the "down" case and compressed in the "up" case and, vice versa, is distorted oppositely for a rising plume. The resulting effects in the images of the top of the aerosol layer (~ 500 meters) are clearly illustrated by the alternating scans in Figure 5.

For well defined clouds of aerosols, these elevation scans enable us to measure both the size and vertical velocity of clouds. In Iowa, making such visual observations during nighttime lidar scans near the swine barns, we were able to estimate upward velocities of aerosol plumes to be about 0.5 – 1.0 meter/sec at a height of 5-10 meters above ground. For well defined plumes rising or falling as they are borne along with the wind, the elevation traces taken like those in Figure 5 display prominent tilts toward or away from the lidar origin depending on both the motion and scan directions, and thus provide information on the vertical and horizontal transport in the boundary layer.

Conclusions

The three-wavelength AGLITE lidar has proved to be a fieldworthy, reliable and self contained system for monitoring and profiling the density and motion of aerosols in the air around agricultural installations. The use of three laser wavelengths has enabled us to observe significant variations in backscatter profiles depending on the particle origins. Time dependent records of vertical and horizontal motions have revealed both small scale and large scale motions of aerosol laden air masses originating from unpaved roads, large swine barn facilities, and ordinary atmospheric phenomena such as clouds and hazes.

In principle, atmospheric probing by lidar is able to obtain time dependent, three dimensional pictures of aerosol distributions in any region of interest such as the air space around animal feedlot and production facilities. Aerosols serve as valuable tracers of air motion. In practice, this faces limitations due to the time scale of atmospheric changes, such as turbulence versus the time required to make complete lidar scans of the region. In this paper and in companion papers in the Workshop, we illustrate that, by making the time dependent observations in two dimensions at a time, namely the lidar range versus a single geographic dimension (elevation or azimuth), one can obtain valuable information on the extent of clouds and on low altitude air motions both along and across the lidar beam direction. Then the synthesis of 3D interpretations of aerosol motion can follow from such 2D records as needed. We will continue to apply these lidar methods to the meteorological interpretation of aerosol motions seen in the emission of particulates from agricultural installations.

References

Bingham G. E., J. Hatfield, J. H. Prueger, T. D. Wilkerson, V. V. Zavyalov, R. L. Pfeiffer, L. Hipps, R. Martin, P. Silva, W. Eichinger. 2006. An Integrated Approach to Measuring Emissions from Confined Animal Feeding Operations at the Whole Facility Scale. Proceedings of the Workshop on Agricultural Air Quality: June 5-8.

Cornelsen, S.S. 2005. Electronics Design of the AGLITE Instrument, SDL Document No.SDL/05-093 and Master of Science Thesis in Electrical Engineering, Utah State University, Logan, UT, 113 pp.

Hipps, L., J.H. Prueger, J. Hatfield, G.E. Bingham, W. Eichinger, T.D. Wilkerson, V.V. Zavyalov, R. Martin, P. Silva. 2006. Integrating Lidar and Atmospheric Boundary Layer Measurements to Determine Fluxes and Dynamics of Particulate Emissions from an Agriculture Facility. Proceedings of the Workshop on Agricultural Air Quality: June 5-8.

Measures, R. M. 1984. *Laser remote sensing: Fundamentals and applications.* John Willey & Sons, Inc., New York.

Martin R.S., V. Doshi, and K. Moore. 2006. Determination of Particle (PM10 and PM2.5) and Gas-Phase Ammonia (NH3) Emissions from a Deep-Pit Swine Operation using Arrayed Field Measurements and Inverse Gaussian Plume Modeling. Proceedings of the Workshop on Agricultural Air Quality: June 5-8.

Silva, P., R.S. Martin, V. Doshi, K. Moore, and M. Erupe. 2006. Variations in Particle Composition and Size Distributions in and around a Deep Pit Swine Operation. Proceedings of the Workshop on Agricultural Air Quality: June 5-8.

Wilkerson, T.D., J.A. Sanders, I.Q. Andrus, G.K. Schwemmer, D.O. Miller, and D. Guerra. 2001. Horizontal Wind Measurements using the HARLIE Holographic Lidar. Proc.SPIE 4484, *Lidar Remote Sensing for Industry and Environment Monitoring II*, pp. 64 – 73.

Zavyalov, V.V., G.E. Bingham, T.D. Wilkerson, J. Swasey, C. Marchant, C. Rogers, R. Martin, P. Silva, and V. Doshi. 2006. Characterization of Particulate Emission from Animal Feeding Operations with Three-wavelength Lidar Using Simultaneous In-situ Point Measurements as Calibration Reference Sources. Proceedings of Workshop on Agricultural Air Quality: June 5-8.

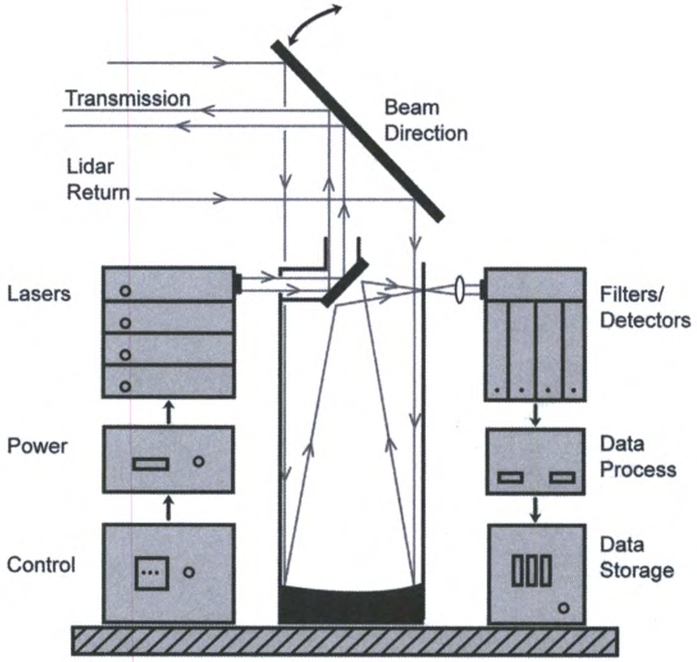


Figure 1. Schematic of AGLITE lidar design and construction

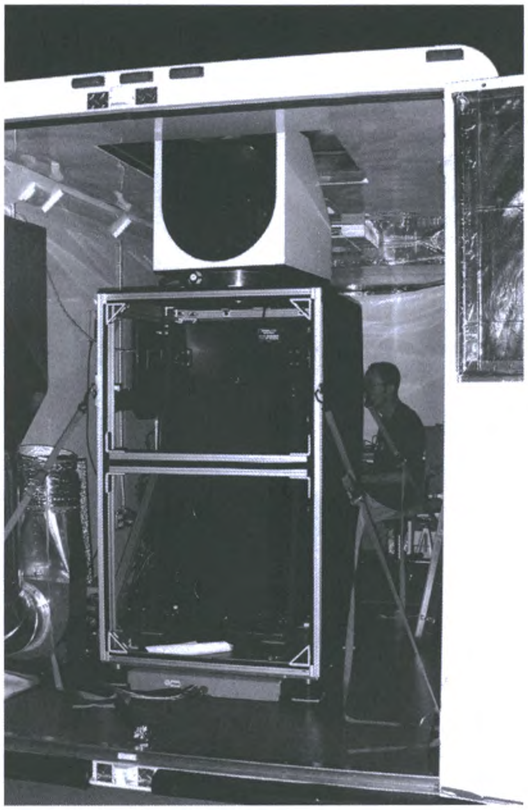


Figure 2. AGLITE lidar mounted vertically, viewed through open back door of trailer

Figure 3. Field deployment trailer for AGLITE lidar

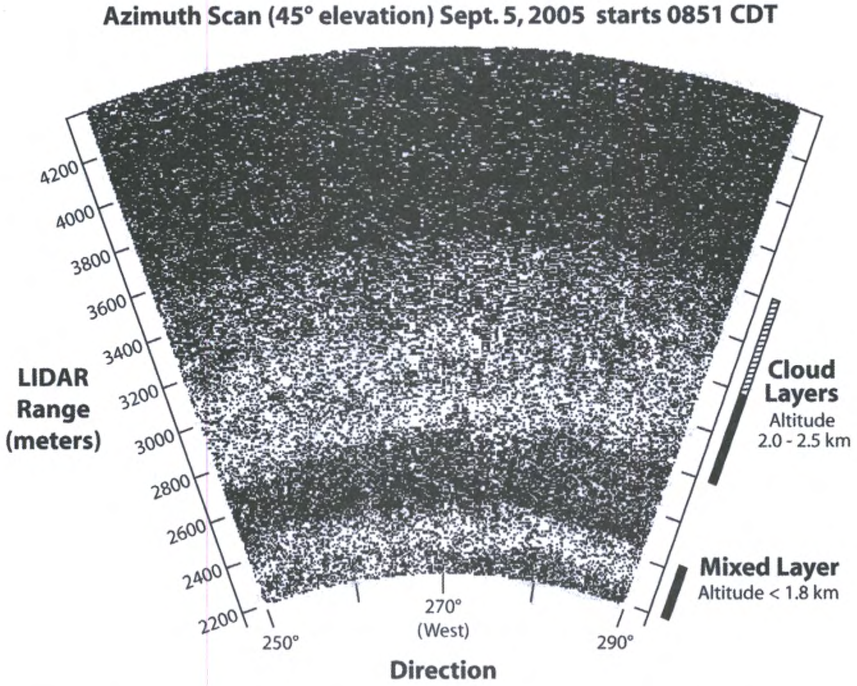


Figure 4. An azimuth lidar scan at 45°elevation showing cloud and aerosol layers to the west of the lidar site. Successive scans provide useful records of air motion.

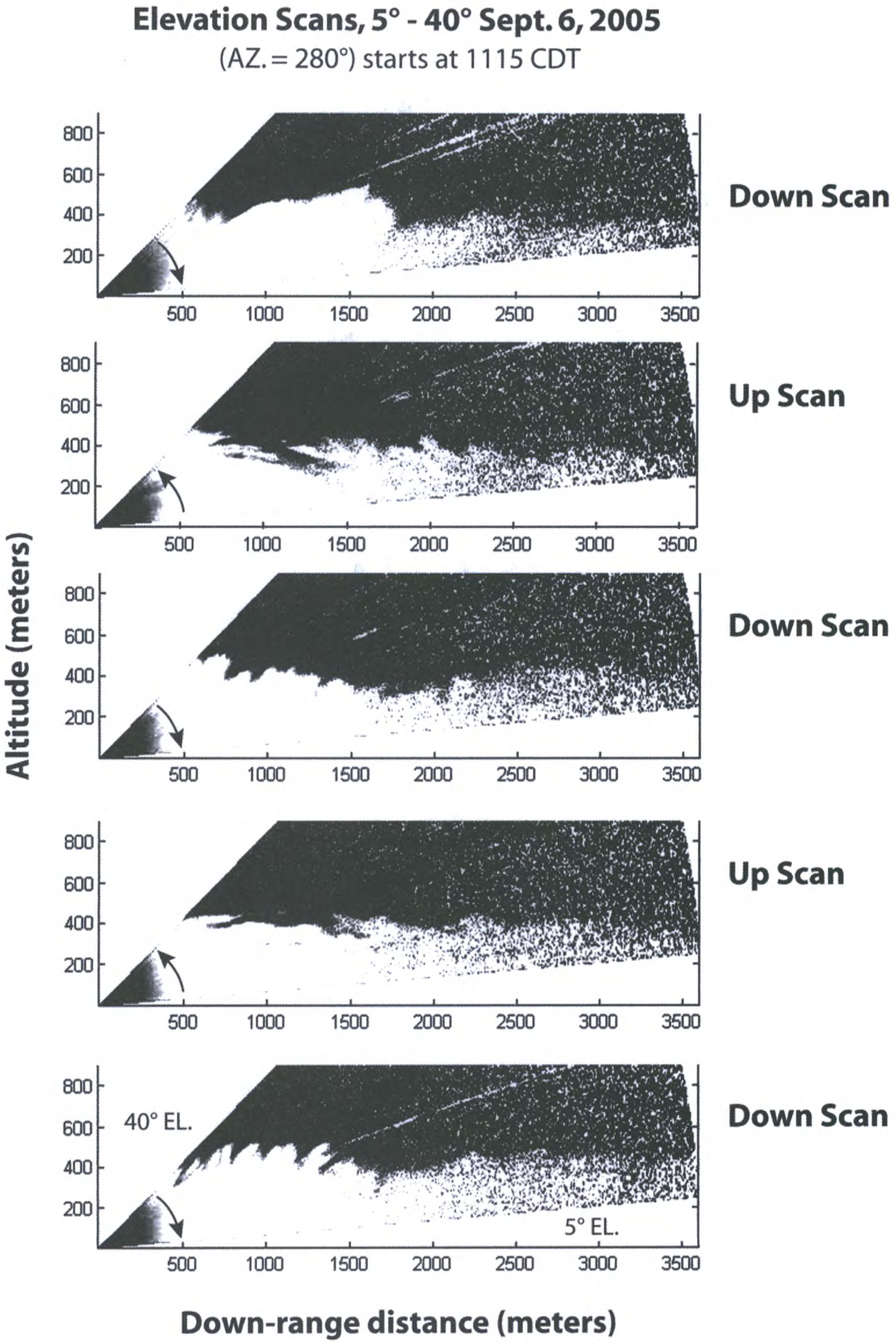


Figure 5. Successive lidar elevation scans at 5.79-minute intervals, looking upwind into boundary layer aerosols (lidar signal increases from black to white)

Modeling Transport and Chemistry of Ammonia in North Carolina: Seasonality and Process Analysis

Shiang-Yuh Wu[1], Jianlin Hu[2], and Yang Zhang[2]
[1]Department of Environmental Quality, Richmond, VA
[2]North Carolina State University, Raleigh, NC

Abstract

Modeling transport and chemistry of ammonia (NH_3) has been performed for August and December 2002 with a 4-km horizontal grid spacing in North Carolina (NC), USA. A probing tool, Process Analysis (PA), is employed to understand the major contributors to the formation and fate of major air pollutants. The most influential atmospheric processes for those pollutants are identified.

Introduction

Ammonia plays an important role in eco-environmental systems through various physical and chemical processes in air and water. Major sources of NH_3 include animal and human wastes, synthetic fertilizers, biomass burning, and fossil fuel combustion (Bouwman, 1997). The weather pattern and the emissions of nitrogen and sulfur oxides, and ammonia often strongly influence the formation of ammonium sulfate ($(NH_4)_2SO_4$) and ammonium nitrate (NH_4NO_3). In North Carolina (NC), there are three well developed physiographic divisions (Boyles, 2004). From east to west, they are the coastal plain, the pidemont, and the mountains with elevation ranging from sea level along the Atlantic coast to 6684 feet at the summit of Mount Mitchell, the highest peak in the eastern United States. The complex topography combined with synoptic weather patterns produce one of the most complex climates in the United States. Agriculture-livestock (A-L) is the largest source of NH_3 in NC and accounts for about 91% (482.9 tons day^{-1}) in August (Wu et al., 2005) and 81% (253.4 tons day^{-1}) in December of total NH_3 emissions. In this study, we conduct two 1-month model simulations (August amd December 2002) to examine the impact of A-L NH_3 emissions on the formation of $PM_{2.5}$ and to quantify the contributions of major processes to the formation of key air pollutants including O_3, $PM_{2.5}$, and its composition, and their precursors.

The US-EPA's modeling system, including the Pennsylvania State University (PSU)/National Center for Atmospheric Research (NCAR) Mesoscale Modeling System Generation 5 (MM5) version 3.7 (http://www.mmm.ucar.edu/mm5/mm5v3.html), the Carolina Environmental Program's (CEP) Sparse Matrix Operator Kernel Emissions (SMOKE) Modeling System version 2.1, and CMAQ (Binkowski and Roselle, 2003) version 4.4 with the PA technique is used in this study. The two 1-month simulations are conducted with a 4-km horizontal grid spacing over a domain covering nearly the entire North Carolina and a portion of adjacent states. The Integrated Process Rates (IPRs) and Integrated Reaction Rates (IRRs) in PA are calculated for both simulations. The results from PA-IPRs provide the relative contributions of individual physical and chemical processes to the formation of gas and PM species. The results from PA-IRRs provide individual gas-phase reaction rates that can be used to identify key chemical pathways for O_3, its precursors and precursors of secondary $PM_{2.5}$. The model input files are developed based on those for the simulations with 12-km horizontal grid spacing from the Visibility Improvement State and Tribal Association of the Southeast's (VISTAS) 2002 modeling program (http://www.vista-sesarm.org.asp). The model evaluation is conducted with surface measurements from five national networks: the Clean Air Status Trends Network (CASTNet), the Interagency Monitoring of Protected Visual Environments (IMPROVE), the EPA Speciation Trends Network (STN), the EPA Air Quality System (AQS), and the National Acid Deposition Program (NADP). Model predictions for August and December 2002 simulations evaluated include max 1-hr and 8-hr average O_3 mixing ratios, the 24-hr average mass concentrations of $PM_{2.5}$ and its composition, and wet deposition of sulafte, nitrate, and ammonium.

Summary of the Results

The preliminary evaluation for August and December 2002 shows a good overall performance for max 1-hr and 8-hr average O_3 mixing ratios with normalized mean biases (NMB) of -15% and -13% in August and NMBs of 4% and 7% in December. It also shows a relatively poor performance for the concentrations of

24-hr average $PM_{2.5}$ and its composition with NMBs of -72% to -18% for August and -59% to 107% for December for all networks. Overall the model underpredicts $PM_{2.5}$ and all its composition for August and overpredicts $PM_{2.5}$ and all its composition for December except SO_4^{2-} at the CASTnet and STN sites, organic matters (OM) at the STN sites, and BC at the IMPROVE and STN sites.

17 locations are selected from the STN, IMPROVE, and CASTNet sites for a detail analysis with PA-IPRs. The preliminary analysis shows that the emission is the most important contributor to the production of $PM_{2.5}$ and its major composition at the STN sites (primarily in urban areas) and the vertical transport is the largest contributor to their losses for both months. In general, the contribution of the PM processes to the mass change in $PM_{2.5}$ is higher in December than August for $PM_{2.5}$, NO_3^-, and NH_4^+. At the IMPROVE (remote locations) and CASTNet (suburban and rural) sites, the major processes are quite different from those at the STN sites. The vertical and horizontal transport dominate the production or loss of $PM_{2.5}$, indicating the importance of meteorology in determining the concentrations of $PM_{2.5}$ and its composition at those sites. Other important processes contributing to the production or loss of PM composition such as NO_3^-, NH_4^+, and secondary organic aerosols include PM processes and dry deposition.

The results from PA-IRRs are being analyzed. Major chemical pathways for O_3, its precursors, and precursors of secondary $PM_{2.5}$ in August and December will be identified.

Acknowledgements

This work is sponsored by the United States Department of Agriculture 2004-35112-14253, and the National Science Foundation Career Award No. ATM-0348819 at NCSU. Thanks are due to Mike Abraczinskas, George Bridgers, Wayne Cornelius, and Karen Harris of NCDENR for providing VISTAS's emissions and CMAQ modeling results with a 12-km grid spacing; Don Olerud, BAMS, for providing VISTAS's MM5 simulation results at a 12-km grid spacing.

References

Binkowski, F.S. and S.J. Roselle, 2003. Models-3 community multiscale air quality (CMAQ) model aerosol component, 1. Model description. *J. Geophys. Res.*, 108, 4183,

doi:10.1029/2001 JD001409.

Bouwman, A. F., D. S. Lee, W. A. H. Asman, F. J. Dentener, K. W. Van Der Hoek, and J. G. J Oliver, 1997. A global high resolution emission inventory for ammonia, *Global Biogeochem. Cycl.*, 11, 561-587, 1997.

Boyles, R. P., 2004. *North Carolina Climate*: A summary of Climate Normals and Averages at 19 Agricultural Research Stations. The State Climate Office of North Carolina, Technical Bulletin 322, September 2004.

Wu, S.-Y., S. Krishnan, J.-L. Hu, C. Misenis, Y. Zhang, V. P. Aneja, and R. Mathur, 2005. Simulating atmospheric fate of ammonia in Southeast U.S. using CMAQ with a 4-km resolution, the 4th Annual CMAS Models-3 User's Conference, Sep 26-28, 2005. Research Triangle Park, NC.

Effect of Diet on Air Emissions from Laying Hens of Different Ages

W. Wu[1], W. Powers[1], R. Angel[2], C.E. Hale, III[3], and T. Applegate[4]
[1]Iowa State University, 109 Kildee Hall, Ames, IA 50011, USA
[2]University of Maryland, 4131 Animal Sciences Center, College Park, MD 20742, USA
[3]Rose Acre Farms, Seymour, IN 47274 and [4]Purdue University, Lilly Hall of Life Sciences,
West Lafayette, IN 47907, USA

Abstract

Manure from poultry feeding operations is associated with diminished air quality. The objectives of the current study were to evaluate the effectiveness of feeding a reduced emission diet (R) containing 6.9% of a gypsum-zeolite mixture and a slightly reduced crude protein (CP) to 21-, 38-, and 59- wk old Hy-line W36 hens (trials 1, 2, and 3, respectively) on egg production and emissions of NH_3, H_2S, NO, NO_2, CO_2, CH_4 and non-methane total hydrocarbon as compared to feeding a commercial diet (C). At each age, 640 hens (BW = 1.36, 1.47, and 1.52 kg in trials 1, 2, and 3, respectively) were allocated randomly to eight environmental chambers for a 3-wk period. On an analyzed basis, the C diet contained 18.0, 17.0, and 16.2% CP and 0.25, 0.20, and 0.20% S in trials 1, 2, and 3, respectively. The R diet contained 17.0, 15.5, and 15.6% CP and 0.99, 1.20, and 1.10% S in trials 1, 2, and 3, respectively. Diets were formulated to contain similar calcium and phosphorus content. Gaseous concentrations were monitored from each chamber in a sequential manner resulting in 10 to 11 daily observations per chamber. Average daily egg weight (ADEW; 57.4 g), average daily egg production (ADEP; 82.5%), average daily feed intake (ADFI; 92.6 g) and BW change (BWC; 24.3 g), across ages, were unaffected by diet ($P>0.05$). Age affected ADEW (52.1, 58.9, and 61.2 g), ADEP (86.7, 87.1, and 73.7%), ADFI (86.8, 96.2, and 94.6g) and BWC (65.2, 17.3, and -9.7 g) in trials 1, 2, and 3, respectively ($P<0.01$). Diet ($P<0.01$) and age ($P<0.05$) affected NH_3 emissions. In trials 1, 2, and 3, daily NH_3 emissions from hens fed the R diets (185.5, 312.2, and 333.5 mg $bird^{-1}$) were less than those of hens fed the C diet (255.0, 560.5, and 616.3 mg $bird^{-1}$). Daily emissions of H_2S across trials from hens fed the R diet were 4.08 mg $bird^{-1}$ compared to 1.32 mg bird-1 from hens fed the C diet ($P<0.01$). Diet ($P<0.05$) and age ($P<0.05$) affected emissions of CO_2 and CH_4. A diet effect ($P<0.01$) on NO emissions and an age effect on SO_2 emissions ($P<0.01$) was observed. No diet or age effects ($P>0.05$) were observed for NO_2 and non-methane total hydrocarbons. Results demonstrate that diet and laying age influence air emissions.

Introduction

It is well documented that gaseous emissions from laying hen feeding operations can have potential negative impacts on the environment and on human and bird health. These emissions have been shown to be reduced through diet formulation. Feeding diets formulated to reduce excess crude protein inputs (Elwinger, et al, 1996) help to reduce nitrogen (N) excreted resulting in lower ammonia (NH_3) emissions. Acidogenic materials reduce manure pH resulting in the protonation of NH_3 to ammonium. Gypsum (calcium sulfate) is one of the acidogenic compounds that has been tested and can serve as a partial replacement for limestone as a calcium (Ca) source without reducing hen performance in laying hen diets (Keshavarz, 1991). Zeolite has been shown to be a beneficial feed additive that exhibits a strong preference for binding nitrogenous cations like ammonium resulting in lower NH_3 concentration (Nakaue and Koelliker, 1981; Cabuk, et al (2004)). A study conducted by Hale (2005) showed that using a reduced crude protein (CP) diet in combination with acidogenic materials such as gypsum and nitrogenous binding compounds like zeolite decreased NH_3 emission (as measured in vitro) from laying hen excreta. However, the effectiveness of feeding such a diet on all gaseous emissions has not been reported. The objective of the current study was to evaluate the effectiveness of feeding a reduced emissions diet (R) containing 6.9 % of a gypsum-zeolite mixture which replaced 35% of the limestone and slightly reduced CP to laying hens of different ages on egg production and emission of NH_3, H_2S, NO, NO_2, CO_2, CH_4 and non-methane total hydrocarbon as compared to feeding a commercial diet (C).

Materials and Methods

The study consisted of three trials utilizing birds which were initially 21, 38, 59 wks old. During each trial, 640 Hy-line W36 hens (BW = 1.36, 1.47, and 1.52 kg in trials 1, 2, and 3, respectively) were randomly assigned to one of eight air emissions chambers for a 3-wk period. Between each trial, chambers were completely cleaned. In each chamber, eight cages of 10 birds per cage were used (355 cm^2 per bird).

Feed, in mash form, and water were available for *ad libitum* consumption. On an analyzed basis, the C diet contained 18.0, 17.0, and 16.2% CP and 0.25, 0.20, and 0.20% S in trials 1, 2, and 3, respectively while the R diet contained 17.0, 15.5, and 15.6% CP and 0.99, 1.20, and 1.10% S in trials 1, 2, and 3, respectively. Diets were formulated to contain similar calcium and phosphorus concentrations. The R diet contained a 35% replacement of Ca with gypsum and clinoptilolite. All diets were formulated to meet National Research Council (1994) nutrient recommendations.

Gaseous concentration and air flow were monitored from each chamber in a sequential manner resulting in 10-11 daily observations per chamber. During each 15 minutes observation period, the concentration of NH_3, H_2S, NO, NO_2, CO_2, CH_4 and non-methane total hydrocarbon were recorded through computer.

Data were analyzed using a mixed model with the day as random variable. (SAS v 8.0) Emission data were adjusted for number of birds. Significance was accepted at or below a $P<0.05$.

Results and Discussion

Average daily egg weight (ADEW; 57.4 g), average daily egg production (ADEP; 82.5%), average daily feed intake (ADFI; 92.6 g) and BW change (BWC; 24.3 g), across ages, were unaffected by diet. Age affected ADEW (52.1, 58.9, and 61.2 g), ADEP (86.7, 87.1, and 73.7%), ADFI (86.8, 96.2, and 94.6g) and BWC (65.2, 17.3, and -9.7 g) in trials 1, 2, and 3, respectively.

In trials 1, 2, and 3, daily NH_3 emissions from hens fed the R diets (185.5, 312.2, and 333.5 mg $bird^{-1}$) were lower than those of hens fed the commercial diet (255.0, 560.5, and 616.3 mg $bird^{-1}$). Age effects on daily NH_3 emission were greater from older hens than from younger hens. Daily H_2S emissions from hens fed the R diets (1.6, 7.1 and 3.7 mg $bird^{-1}$) were lower than those of hens fed the commercial diet (0.5, 1.9 and 0.8 mg $bird^{-1}$). Daily CO_2 emissions from hens fed R diet (64626, 79799 and 80335 mg $bird^{-1}$) were less than those of hens fed the commercial diet (65627, 86243 and 82694 mg $bird^{-1}$). Additionally, age affected CO_2 emissions. Daily methane emissions from hens fed R diet (138.5, 28.2 and 9.4 mg bird-1) were lower than those of hens fed the commercial diet (153.1, 43.8 and 11 mg $bird^{-1}$). Age affected methane emission with younger birds producing more methane than older birds. Daily emissions of NO from hens fed R diets (0.71, 0.11 and 0.21 mg $bird^{-1}$) were less than those from control diet (0.94, 0.2 and 0.28 mg $bird^{-1}$). No diet or age effects were observed for NO_2 non- methane total hydrocarbons.

Diet acidification, CP reduction and zeolite supplementation reduced the emissions of NH_3 (by 39%), CO_2 (by 5%), CH_4 (by 17%) and NO (by 48%). Diet acidification increased the gaseous emissions of H_2S mainly because of the high concentration of sulfate in the acidifying agent used. More research is needed to address the increased H_2S emissions and to explore the impact of graded concentrations of the additive on emissions.

Implication

Emergency Planning and Community Right-to-Know Act (EPCRA) and Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) reporting requirement allow daily emissions 100lb of both NH_3 and H_2S from poultry farms. The current study illustrated that feeding a reduced emission diet could increase hen population by 40% without exceeding the requirements. Even though gypsum addition increases H_2S emissions considerably, these concentrations did not surpass reporting limits even for the larger laying hen complexes. Reporting requirements for laying hens operations for CERCLA/EPCRA, therefore, will be driven by NH_3.

References

Cabuk, M., A. Alcicek, M. Bozkurt, and S. Akkan, 2004. Effect of yucca schidigera and natural zeolite on broiler performance. Int. J. of Poultry Sci. 10:651-654.

Hale III, E.C. 2005. Reduction of ammonia emission, phosphorus excretion, and potassium excretion in laying hen manure through feed manupulation. 2005 Animal Waste Management Symposium, Research Triangle Park, North Carolina

Keshavarz, K. 1991. The effect of calcium sulfate (gypsum) in combination with different sources and forms of calcium carbonate on acid-base balance and eggshell quality. Poultry Sci. 70:1723-1731.

McCrory, D. F. and P. J. Hobbs. 2001. Additives to Reduce Ammonia and Odor Emissions from Livestock Wastes: A Review. J. Environ. Qual. 30:345–355.

National Research Council. 1994. Nutrient Requirements of Poultry, 9th rev. ed. National Academy Press, Washington, DC.

H.S. Nakaue, J.K. Koelliker. 1981. Studies with clinoptilolite in poultry. I. Effect of feeding varying levels of clinoptilolite (zeolite) to dwarf single comb white leghorn pullets and ammonia production. Poultry Sci. 60:944-949

Role of Leaf Surface Water in the Bi-Directional Ammonia Exchange Between the Atmosphere and Terrestrial Biosphere

Yihua Wu[1], John Walker[2], Christa Peters-Lidard[1], Donna Schwede[3],
Robin Dennis[3], Wayne Robarge[4]
[1]NASA/GSFC/HSB
[2]U.S.EPA/NRMRL
[3]NOAA/ARL/ASMD, in partnership with the USEPA
[4]NCSU/DSS

Abstract

A field experiment was conducted to study the ammonia exchange between plants and the atmosphere in a soybean field in Duplin County, North Carolina during the summer of 2002. Measurements indicate that the net canopy-scale ammonia exchange is bi-directional and has a significant diurnal cycle. In general, ammonia concentrations peak a few hours after sunrise. Deposition occurs in the evening and early morning hours while emission occurs in the late morning and early afternoon. To investigate the mechanisms that control the exchange process, a new model is developed based on the Multi-Layer BioChemical deposition (MLBC) model (Wu et al., 2003) with additional parameterizations for the leaf ammonia compensation point (Wu et al., 2006) and leaf surface water effects. The MLBC model considers biochemical processes such as photosynthesis, respiration, and membrane passive transport through the cuticle. The leaf surface water thickness is based on a simple water budget equation. Model results suggest that while accurate prediction of the stomatal compensation point is critical, leaf surface water also plays an important role in the net canopy-scale ammonia flux.

1. Introduction

It is well documented that leaf surface water influences the development of various foliar bacterial and fungal pathogens (Aston, 1979). Some studies suggest that leaf surface water plays a role in the deposition of pollutants on crops. Our field measurements of ammonia fluxes indicate that leaf surface water appears to enhance ammonia deposition to plants. The potential for the formation of free water on leaves is always present. Leaf surface water can originate from three sources: the air (dewfall and precipitation), the soil (dewrise) and the plant itself (guttation). Depending on environmental conditions, water on the leaf surface can be a thin film, discrete droplets, or a continuous liquid layer. Mikhail and Robens (1983) reported that a thin water film in the nanometer range was always present on surfaces at normal humidities in their investigation of a range of different hydrophilic materials. Leaf surface water can be a barrier for non-soluble gases (such as CO_2), or an intermediate medium (i.e., temporary sink and source) for soluble gases (such as NH_3 and SO_2). Leaf surface water also plays an active role in chemical processes. Fowler and Unsworth (1979), and Schuepp (1989) found that the deposition velocity for SO_2 was several times higher when the plant was covered by dew than for dry conditions. Weseley (1989) reported that dew increased the deposition of SO_2 and decreased the deposition of O_3. Observations by Finkelstein et al. (2000) also show that dew increases the deposition of SO_2. Plant leaf surfaces exposed to the atmosphere are continuously exposed to water vapor, so the potential for the formation of molecular films is always present. According to Flechard et al. (1999), the sink strength of the leaf surface is largely dependent on surface humidity and temperature. Dry deposition to the leaf surface for most gas species is reversible. For example, thin water films on plant surfaces may behave as perfect sinks for NH_3 for a limited period of time during wet periods after rain or during dewfall. However, as a dew layer or intercepted rainwater evaporates from the leaf surface, NH_3 could be released from the evaporating water drops since the NH_3 molecule is not immediately 'destroyed'. Therefore, it is very important to consider the role of water when investigating the exchange of ammonia between the atmosphere and biosphere. In this paper, we develop and evaluate a mechanistic model of ammonia uptake and emission from leaf surface water.

2. Materials and Methods

2.1 Leaf water budget

The leaf surface water balance is given by the following equation:

$$\frac{\Delta S}{\Delta t} = R_i + D - E \qquad for \qquad 0 \le S \le S_c \tag{1}$$

where S is water storage (mm), t is time (s), , R_i is the amount of rainfall intercepted by the leaf (mm s^{-1}), D is the rate of dew condensation (mm s^{-1}), E is the rate of leaf surface water evaporation (mm s^{-1}), and S_c is the leaf water storage capacity (mm).

There are a large number of rainfall interception models in the literature, which can be categorized into two types (Massman, 1983): static models (Merrian, 1960; Zinke, 1967; Norman and Campbell, 1983) and dynamic models (Gash, 1979; Massman, 1983; Zeng et al., 2000). The static models are simple and easy to use, but do not always give satisfactory quantitative results. On the other hand, the dynamic models are mechanistically more descriptive, but are complex. The dynamic models also rely on empirical drainage parameters that may change with canopy size and type. In this study, we use Norman and Campbell's static model in which the interception of rainfall is expressed as:

$$R_i = R_n(1 - e^{-0.5LAI}) \tag{2}$$

where R_n is rainfall rate (mm s^{-1}) and LAI is leaf area index (m^2 m^{-2}).

The leaf water storage capacity, S_c, is a function of canopy structure which depends on vegetation age and species (Hall et al., 1997; Kang et al., 2005, and Pypker et al., 2005). However, the commonly accepted value of 0.2 mm given by Noilhan and Planton (1989) is used in this study.

If we assume that dew formation and leaf surface water evaporation can not occur at the same time, then the potential rates for both dew formation and leaf surface water evaporation can be computed from the following equation:

$$H_2OFlux = \frac{Q_s - Q_a}{R_a + R_b} \tag{3}$$

$D_{\max} = H_2OFlux$ when H_2Oflux is negative

$E_{\max} = H_2OFlux$ when H_2Oflux is positive

where Q_s is the saturated specific humidity at the corresponding leaf surface temperature (g kg^{-1}), Q_a is the specific humidity at the corresponding air temperature (g kg^{-1}), R_a is the aerodynamic resistance (s m^{-1}), and R_b is the boundary layer resistance (s m^{-1}), D_{max} is the potential rate of dew formation, E_{max} is the potential rate of evaporation. To get the actual rates of dew and evaporation, D_{max} and E_{max} are multiplied by the wetness index and the fraction of wet leaf surface area. While the observed wetness index is used, the fraction of wet leaf surface area is computed as the following (Deardorff , 1978):

$$F_w = \left(\frac{S}{S_c}\right)^{0.67} \tag{4}$$

Ammonia deposition flux (F_d, μg m^{-2} s^{-1}) and emission flux (F_e, μg m^{-2} s^{-1}) to and from the leaf surface water are computed according to:

$$F_d = -\left(\frac{C_a}{R_a + R_b}\right) \text{ when } H_2OFlux < 0 \tag{5a}$$

$$\text{and } F_e = \left(\frac{C_s}{R_a + R_b}\right) * F \text{ when } H_2OFlux > 0 \tag{5b}$$

where C_a is ammonia ambient concentration (μg m^{-3}), and C_s is ammonia concentration at the leaf surface (μg m^{-3}), F is an empirical scale factor, given by $F = \left(\frac{E}{S_c}\right)^{0.2}$. These fluxes are added to the stomatal, cuticular and ground fluxes to yield the net canopy-scale flux.

2.2. Cuticular Resistance

Plant cuticles are a lipophilic polymer membrane that consists of an insoluble bipolymer cutin and waxlike lipids (Kerstiens, 1996). The cuticular resistance (R_{cut}) includes three parts in parallel: the cuticular resistance for dry leaf surfaces (R_{cd}), the cuticular resistance for wet leaf surfaces (R_{cw}), and the surface resistance for wet leaf surfaces (R_{sfc}). They are in s m^{-1}. Wu et al. (2003) used the following equation for R_{cut}:

$$\frac{1}{R_{cut}} = \frac{1-W_F}{R_{cd}} + \frac{W_F}{R_{cw}} + \frac{W_F}{R_{sfc}} \tag{6}$$

where W_F is the maximum weighting factor for wet leaf surface area, and is given by the following empirical equation:

$$W_F = \begin{cases} 0 & (H < H_1) \\ \dfrac{H-H_1}{H_2-H_1} & (H_1 \le H \le H_2) \\ 1 & (H > H_2) \end{cases} \tag{7}$$

where H_1 and H_2 are critical humidities (35, and 98 respectively). The parameterizations of R_{cd} and R_{cw} are based on membrane transport theory, and can be expressed as:

$$R_{cd} = \frac{X_{cm}}{D_c K_{ca}} \tag{8}$$

where X_{cm} is the thickness of the cuticle membrane (m), D_c is the diffusivity of a specific gas of interest in the cuticle (m^2 s^{-1}), and K_{ca} is the cuticle/air partitioning coefficient for a specific gas; and

$$R_{cw} = \frac{X_{wm}}{D_w K_{wa}} + \frac{X_{cm}}{D_c K_{cw}} \tag{9}$$

where X_{wm} is the thickness of the leaf water film (m), D_w is the diffusivity of a specific gas of interest in water (m^2 s^{-1}), K_{wa} and K_{cw} are water/air and cuticle/water partitioning coefficients (dimensionless), respectively. The thickness of the leaf water film, $X_{wm,}$ is estimated by the leaf surface water budget equation. The wet leaf surface resistance (R_{scf}) is computed by the following empirical equation:

$$R_{sfc} = \left(\frac{1}{D_a K_{wa} X_{wm}}\right)\left(\frac{1}{(0.5 + |7.0 - pH|)}\right) \tag{10}$$

where pH is the pH value of leaf surface water, D_a is the diffusivity of a specific gas of interest in the air (m^2 s^{-1}). The reader is referred to Wu et al. (2003) for a more detailed description of the cuticular resistance as implemented in the MLBC model.

2.3 Field Measurements

The MLBC model was evaluated against ammonia fluxes measured over soybean (Walker et al., 2006). The experiment was conducted in Duplin County, NC between June 18 and August 22, 2002. The site is located in the state's Upper Coastal Plain region which is characterized by relatively high ammonia emissions from animal production facilities and fertilizer application (Walker et al., 2000). Soil at this site is Norfolk series fine sandy loam. Ammonia concentrations showed considerable diurnal variability, reaching a maximum in the early morning and a minimum in the late evening. Deposition was observed at night and during the first few hours after sunrise, while emission typically occurred in the late morning and early afternoon (Figure 1). This diurnal cycle of ammonia flux is driven by at least three forces: diurnal variation of ambient ammonia concentration (Figure 1), diurnal variation of the plant ammonia compensation point, and diurnal variation of leaf surface water (Figure 1). Leaf surface water can affect ammonia exchange by: 1) acting as a temporary sink and source, and 2) modifying the plant cuticular resistance. As shown in Figure 1, there is a strong relationship between leaf surface wetness and ammonia flux: ammonia deposition increases with leaf surface wetness; ammonia emission occurs when leaf surface wetness drop to a threshold of 0.06. There appear to be two ammonia emission peaks: one at 11:00 AM (LT), and the other at about 14:00 PM (LT). The first peak is consistent with the evaporation of leaf surface water as indicated by the drop in leaf wetness between 7:00 and 11:00 AM. This suggests that the dissolved ammonia in leaf surface water is released back to the air. The second peak represents the diurnal maximum of the stomatal compensation point, which is a strong function of leaf temperature. Therefore, it is necessary to compute leaf surface water budget

3. Results and Discussion

The leaf surface water budget model described above was incorporated into the MLBC model (Wu et al., 2003) with the addition of ammonia compensation point (Wu et al, 2006), which was then evaluated using data from the field experiment. Three model runs were conducted: RUN1 with a constant cuticular resistance (8000 s m^{-1}) and without ammonia deposition and emission to and from leaf surface water; RUN2 with the parameterization for cuticular resistance and without ammonia deposition and emission to and from leaf surface water, and RUN3 with the parameterization for cuticular resistance and with ammonia deposition and emission to and from leaf surface water. Component fluxes (stomata, non-stomata [i.e., cuticle, leaf surface water and ground fluxes]) were also computed for RUN3. The average daily cycle of observed and modeled ammonia flux is shown in Figure 2. RUN1 and RUN2 have only one peak. The peak is around 15:00 PM (LT). RUN1 underestimated both deposition and emission most of the day while RUN2 overestimated deposition but underestimated emission. RUN3 has two peaks, one at 12:00 PM (LT), the other one at 15:00 PM (LT). The modeled peak values are close to the observed ones. RUN3 overestimated deposition. However, all runs are well within one standard deviation of the observations. The daily cycle of modeled component fluxes from RUN2 is shown in Figure 3. The figure clearly shows that the first peak is contributed by the release of the dissolved ammonia in leaf surface water when it evaporates, the second peak is contributed by stomatal emission. To further demonstrate this, the daily cycle of modeled stomatal and nonstomatal component fluxes is shown in Figure 4.

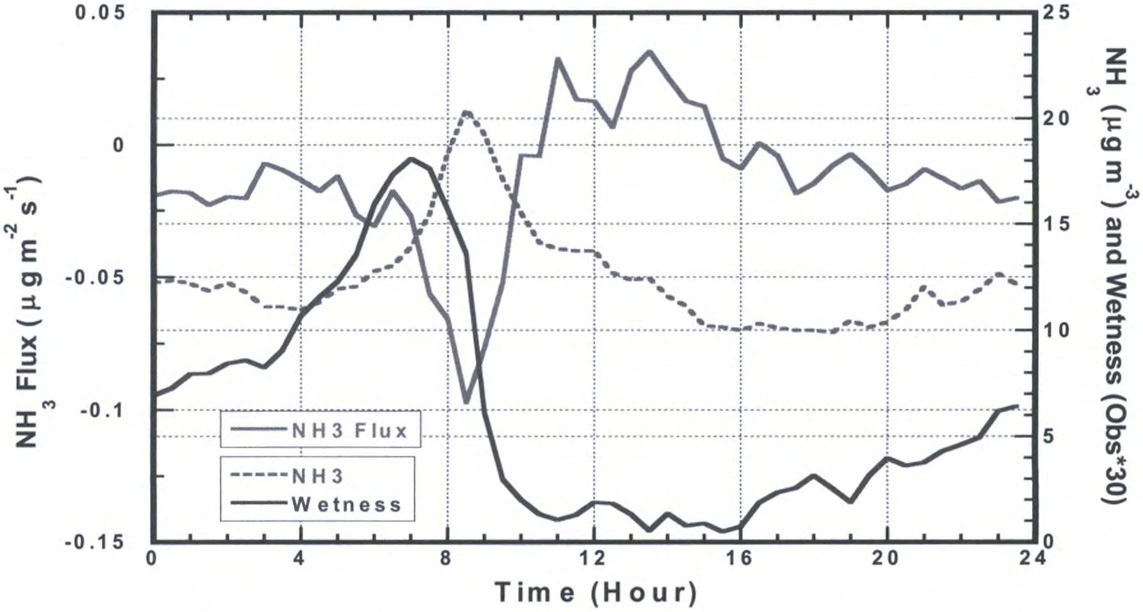


Figure 1. Average diurnal variations of observed NH_3 fluxes, leaf surface wetness and NH_3 concentration for the entire experiment period. Leaf wetness represents the fraction of the 30-minute measurement period during which the leaf wetness sensor was wet, and is multiplied by 30 in order to show on the same figure.

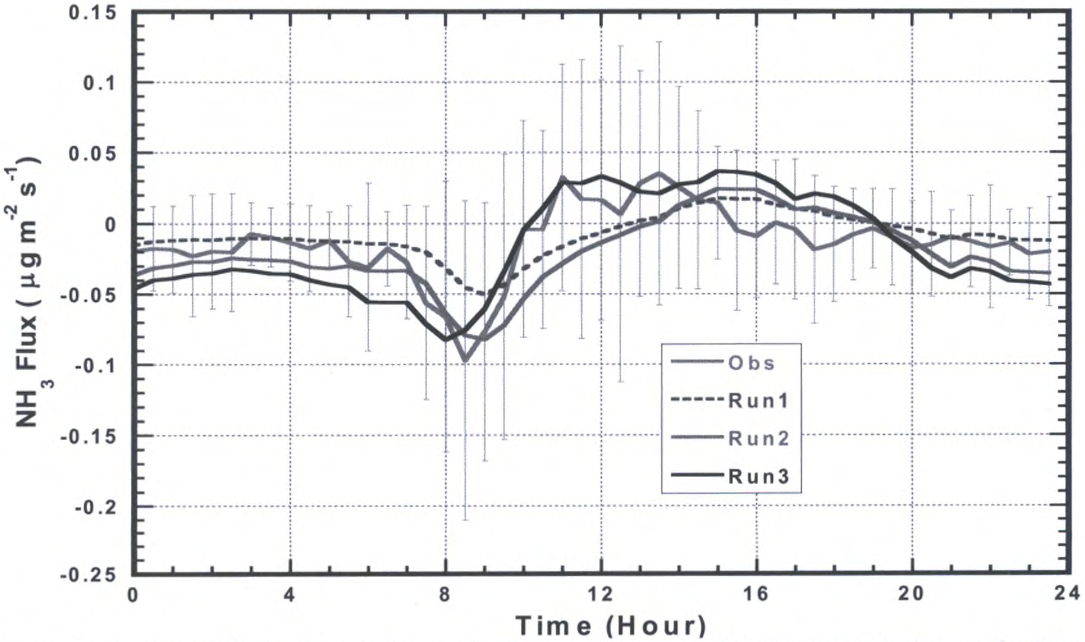


Figure 2. Average diurnal variations of observed and modeled ammonia fluxes for the entire experiment period.

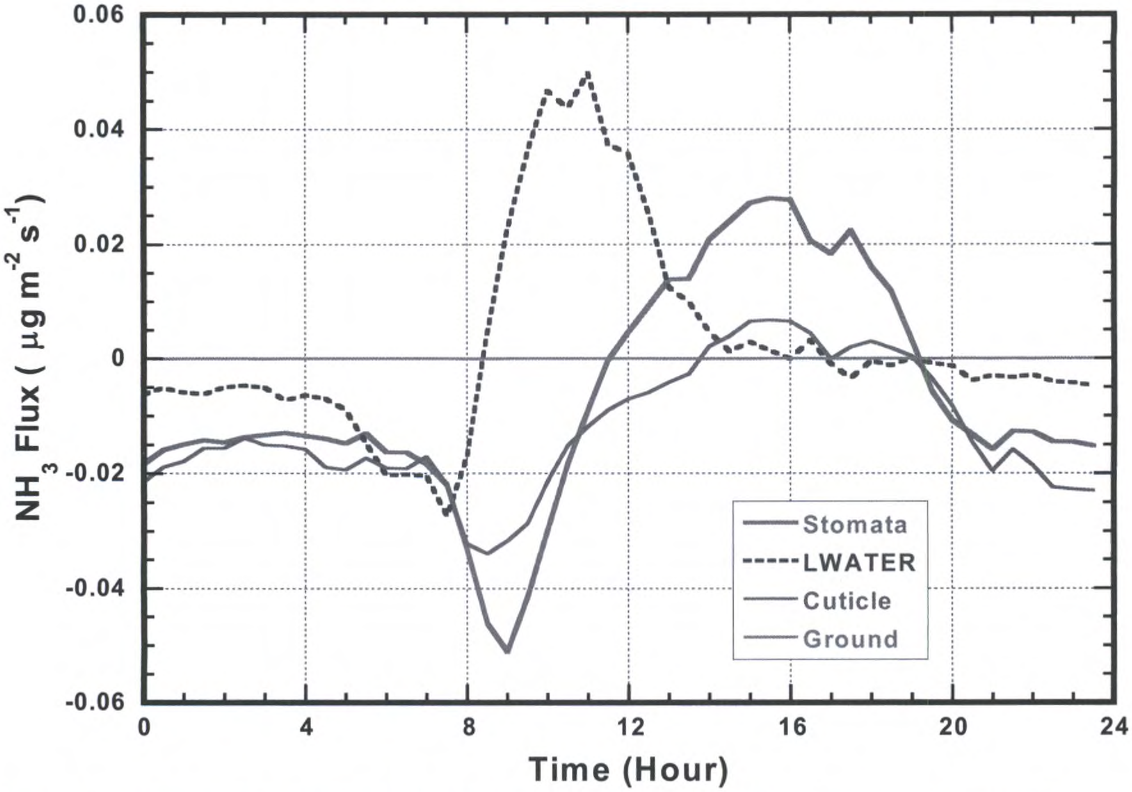


Figure 3. Average daily cycles of modeled stomatal, leaf surface water, cuticle and ground fluxes from RUN3 for the entire experiment period.

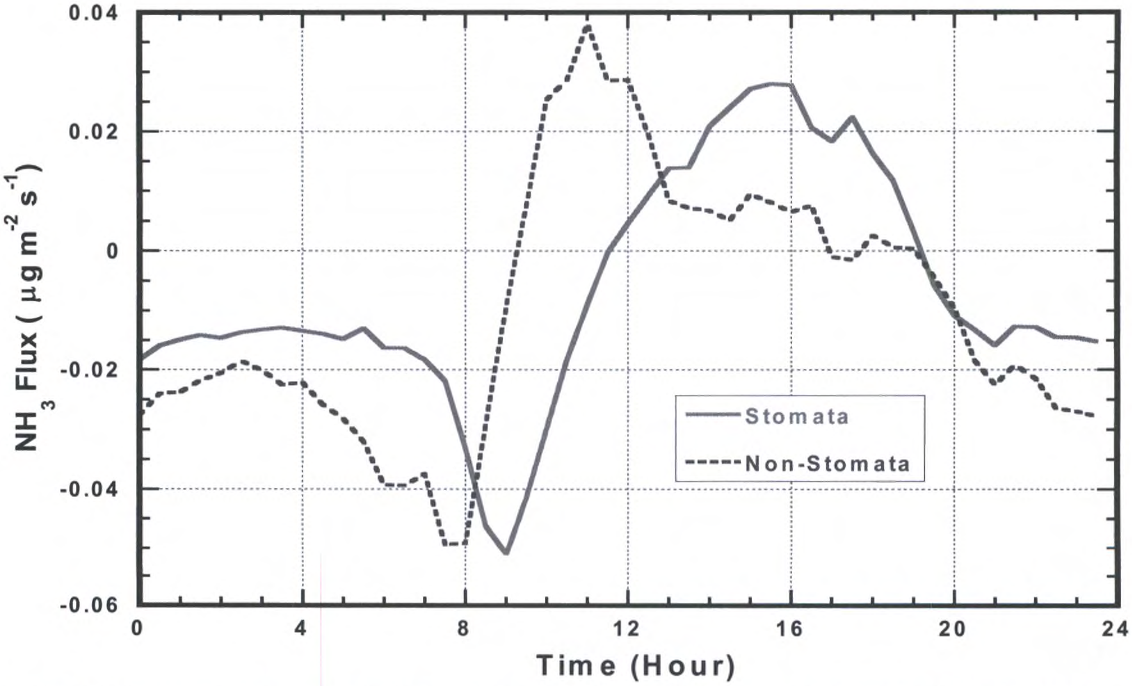


Figure 4. Average diurnal variations of modeled stomatal and nonstomatal fluxes from RUN3 for the entire experiment period.

4. Summary

The role of leaf surface water in the bi-directional ammonia exchange between the atmosphere and terrestrial biosphere was investigated using both measurements and modeling approaches. Fluxes measured over soybean during the summer of 2002 in Duplin County, North Carolina show that deposition occurred at night and during the first few hours after sunrise while emission was observed during the late morning and early afternoon. One emission peak occurred around 11:00 AM while a second peak occurred near 15:00 PM. Results from the new MLBC model clearly demonstrate that the two peaks are the result of leaf surface water evaporation and stomatal emission, respectively.

5. Acknowledgement

This project is sponsored by EPA/NOAA and NASA. The research presented here was performed under the Memorandum of Understanding between the U.S. Environmental Protection Agency (EPA) and the U.S. Department of Commerce's National Oceanic and Atmospheric Administration (NOAA) and under agreement number DW13921548. This work constitutes a contribution to the NOAA Air Quality Program. Although it has been reviewed by EPA and NOAA and approved for publication, it does not necessarily reflect their policies or views.

6. References

Aston, A.R., 1979: Rainfall interception by eight small trees, Journal of Hydrology, Vol. 42, pp 383-396.

Bouten, W., M. G. Schaap, J. Aerts and Aart W. M. Vermetten , 1996: Monitoring and modelling canopy water storage amounts in support of atmospheric deposition studies, Journal of Hydrology, Vol. 181, pp 305-321.

Deardorff, J.W. 1978. Efficient production of ground surface and moisture with inclusion of a layer of vegetation. J. Geophys. Res. 20, 1889-1903.

Finkelstein, P.L., T.G. Ellestad, J.F. Clarke, T.P. Meyers, D.B. Schwede, E.O. Hebert, and J.A. Neal, Ozone and sulfur dioxide dry deposition to forests: observations and model evaluation. J. Geophys. Res., Vol. 105 (D12), 15365-15377, 2000.

Flechard, C. R., D. Fowler, M. A. Sutton and J. N. Cape, A dynamic chemical model of bi-directional ammonia exchange between semi-natural vegetation and the atmosphere, Q. J. R. Meteorol. Soc., Vol. 125, 2611-2641, 1999.

Fowler, D. and Unsworth, M. H., Turbulent transfer of sulphur dioxide to a wheat crop. Q. J. R. Meteorol. Soc., Vol. 105, 767-783, 1979.

Gash, J.H.C., 1979: An analytical model of rainfall interception by forests. Q. J. R. Meteorol. Soc., Vol. 105, pp 43-55.

Hall, F.R., Downer, R.A., Cooper, J.A., Ebert, T.A., and Ferree, D.C. 1997. Changes in spray retention by apple leaves during the growing season. HortScience, 32, 858-860.

Kang, Y., Q. Wang and H. Liu, 2005: Winter wheat canopy interception and its influence factos under sprinkler irrigation, Agricultural Water Management, Vol. 74, pp 189-199.

Kerstiens, G., Plant Cuticles, an integrated functional approach. □ios Scientific Publishers, Oxford, UK., 1996.

Massman, W., 1983: The derivation and validation of a new model for the interception of rainfall by forests, Agricultural Meteorology, Vol. 28, pp 261-286.

Merrian, R.A., 1960: A note on the interception loss equation, J. Geophys. Res., Vol. 65, pp 3850-3851.

Mikhail, R. S. and Robens, E., Microstructure and thermal analysis of solid surface. Wiley, Chichester, pp. 56-66, 1983.

Noilhan, J., and Planton, S. 1989. A simple parameterization of land surface processes for meteorological models. Mon. Weath. Rev., Vol. 117, 536-549.

Norman J.M., and Campbell, G. 1983. Application of a plant environment model to problems in the environment. Adv. Irrig. Vol. 2, pp 155-188.

Seinfeld, J. H. and S. N. Pandis, Atmospheric chemistry and physics: From air pollution to climate change, John Wiley & Sons, INC, New York, 1998.

Pypker, T.G., B.J. Bond, T.E. Link, D. Marks and M.H. Unsworth, 2005: The importance of canopy structure in controlling the interception loss of rainfall: Examples from a young and an old-growth Douglas-fir forest, Agricul. and Forest Meteorology, Vol. 130, pp 113-129.

Walker, J. T., Aneja, V. P., Dickey, D., 2000. Atmospheric Transport and Wet Deposition of Ammonium in North Carolina, USA. Atmospheric Environment, Vol. 34, 3407-3418.

Walker, J.T.,Wayne P. Robarge, Yihua Wu, Tilden Meyers, 2006. Measurement of bi-directional ammonia exchange over soybean using the modified Bowen-ratio technique. In press, Agricultural and Forest Meteorology.

Wesely, M.L, Parameterization of surface resistance to gaseous dry deposition in regional-scale numerical models. Atmospheric Environment, Vol. 23, 1293-1304, 1989.

Wu, Y., J. Walker, C. Peters-Lidard, Donna Schwede, Robin Dennis and Wayne Robarge and, 2006: Modeling Bi-directional ammonia gas exchanges between the atmosphere and biosphere, I. Parameterization of ammonia compensation point (To be submitted to Agricultural and Forest Meteorology).

Wu, Y., B. Brashers, P. L. Finkelstein and J. E. Pleim, 2003: A multiplayer biochemical dry deposition model, I. Model Formulation, JGR, Vol. 108 (2003), No. D1,

Zeng, M., J.W. Shuttleworth, J.H. Gash, 2000: Influence of temporal variability of rainfall on interception loss. Part I. Point analysis. Journal of Hydrology, Vol. 228, pp 228-241.

Zinke, P.J., 1967: Forest interception studies in the United States. In: W.E. Sopper and H.W. Hull (Editors), International Symposium on Forest Hydrology, Pregamon Press, Oxford, pp 823.

Spatial Distribution of Ammonia Emissions on the Territory of the Czech Republic

M. Zapletal and P. Chroust

Centre for Environment and Land Assessment - Ekotoxa Opava, Horní nám. 2, 746 01 Opava,
Czech Republic

Abstract

Modeling of the spatial distribution of ammonia (NH_3) emissions on the territory of the Czech Republic within a 5x5 km grid is essential condition for the modelling of atmospheric transport and deposition flux of reduced nitrogen compounds and exceedances of critical loads of nutrient nitrogen. Spatial distribution of ammonia emissions is a suitable tool for strategy identification of ammonia emission reduction. Key input data to the model of distribution of ammonia emission are detailed data on number and positioning of cattle, horses, sheep, goats, pigs and poultry, data on mineral fertilisers usage, data on the industrial sources and processes, data on the other anthropogenic sources and land use data. For on assessment of total ammonia emission originating from livestock breeding the emission factors have been applied. Total ammonia emissions in 2002 were calculated as a sum of ammonia emission from livestock breeding, emission from mineral fertilisers usage, emission from industrial sources and processes and emission produced by human population on the territory of the Czech Republic on a 5x5 km grid. Total ammonia emission in the Czech Republic in 2002 was estimated to be 72.3 kt NH_3 year^{-1}. Ammonia emission from agricultural sources was estimated to be 68.4 kt NH_3 year^{-1}. There from cattle 29 kt NH_3 year^{-1}, pigs 29.9 kt NH_3 year^{-1}, poultry 7.5 kt NH_3 year^{-1}, horses, sheep and goats 0.21 kt NH_3 year^{-1}, mineral fertilisers usage 1.7 kt NH_3 year^{-1}. Ammonia emission from industrial sources and processes was estimated to be 0.8 kt NH_3 year^{-1}. Ammonia emission from human population was estimated to be 3.1 kt NH_3 year^{-1}.

Introduction

Atmospheric pollution by ammonia emissions involves an important environmental problem. Because of wet and dry ammonia depositions contribution to acidification and eutrofisation of ecosystems, ammonia is regarded to be a significant agent participating on present gradual extinction and damage of forests. The source of ammonia emissions in the Czech Republic are namely agricultural activities (housing of cattle, manure storage and application, fertilisers application) (Zapletal, 1997, 1998, 2001, 2002). Industrial production (heat treatment of coal, chemical industry) and other sources (transport, emissions from human population) represent a less contribution to ammonia emissions. The main source of ammonia in the nature is its generation due to linkage of atmospheric nitrogen through the bacterial processes cycling in a soil and consequent release to the atmosphere. The model on spatial distribution of emissions and deposition flux concentrations of ammonia on the territory of the Czech Republic on a 10x10 km grid in 1994 is presented in the work Zapletal (1997).

This study includes description, application and the results of modelling the spatial distribution of ammonia emissions on the territory of the Czech Republic on a 5x5 km grid level using emission model and detail emission inventory of all significant ammonia emission sources. Exact ammonia emissions spatial distribution is the essential input into transport and deposition models for a purpose of modelling atmospheric transport and deposition flux of reduced nitrogen compounds and for evaluation exceedances of critical loads of nitrogen and acidity on the territory of the Czech Republic. Spatially distinguished ammonia emissions are a useful aid for the determination strategy and tools of abatement of ammonia emissions.

Methods

Source Data

Ammonia emissions from livestock breeding (emissions from housing, manure and sewage storage, fertilisers and sewage application on field, emissions from grazing period), emissions from application mineral fertilisers used in agriculture, technologic emissions from industrial sources and emissions from people population were included in calculation of the total ammonia emission on the territory of the Czech Republic.

The main source of data for determination of spatial distribution of ammonia emissions on the territory of the Czech Republic was the Register of stables in 2001 and 2002 (Zapletal et al., 2004), including detail data on breeding and location of livestock stables (firstly cattle), and the Soil Register incorporating information on land use (Zapletal et al., 2004). Data on pigs and poultry numbers were derived from the database of the State Veterinary Administration in Liberec (Zapletal et al., 2004). The details for assignment of spatial distribution of ammonia emissions originating from industrial sources were provided by the Czech Hydrometeorological Institute (CHMI, 2002). Information about population of the Czech Republic were derived from available data of the Czech Statistical Office (CSO, 2003).

Methodology of Modeling the Spatial Distribution of Ammonia Emission

The ammonia emissions from livestock breeding were calculated using livestock numbers and average annual emission factor. The emissions from breeding cattle, horses, sheep and goats were calculated separately for all individual stables. Emissions from pig-breeding and poultry-farming were calculated for all sources recorded in the respective cadastral region. Spatially differentiated data on numbers of water poultry were not available. Emissions calculated in different spatial detail were summarised in GIS to a 5x5 km grid covering entire territory of the Czech Republic.

Emission factors have been used to assess the total ammonia emission from livestock breeding including four types of production NH_3 emissions (emissions from housing, emissions from farm-manure storage, emissions from farm-manure application and emissions from grazing period). These factors were suggested for individual livestock classes according to actual legislation (MECR, 2002) and conclusions of the Ammonia Expert Panel of the UN ECE (Hoek, 1998). Average emission factors used for cattle, pigs, poultry, horses, sheep and goats are presented in Table 1.

Emission factors have been used for the following animal categories: dairy-cows, calves, bulls, heifers, sucking-pigs, sows, fattening pigs, layers, broilers, geese and dugs. Emission factor for sows was determined as a weighted average of factors of sows and in-pig sows. Emission factors for turkey hens, turkey cocks, goats, sheep and horses were applicated according to results of the Ammonia Expert Panel of the UN ECE (Hoek, 1998).

Emissions from application of mineral fertilisers have been calculated according to area of agricultural land in individual land parcels (Zapletal et al., 2004) in a 5x5 km grid including arable land, gardens, vineyards and hop-gardens, excluding pastures and meadows that are not supposed to be fertilised with mineral fertilisers, according to average of converted nitrogenous fertilisers consumption on 1 ha of agricultural land (on basis of the Czech Statistic Office data) (CSO, 2003) and according to average factor of nitrogen loss from applied nitrogenous mineral fertilisers (2.94 %) (Dragosits et al., 1998). For an assessment of ammonia emissions from industrial sources, the Register of Emissions and Air Pollution Sources (CHMI, 2002) was used. Ammonia emissions from human population have been derived from database of the Czech Statistical Office (CSO, 2003) and from average amount of ammonia emitted by an individual, i.e. 0.3 kg NH_3 man^{-1} $year^{-1}$ (Klaassen, 1990).

The spatial scale of ammonia emissions varies with source data. Summarisation of source data of different spatial detail (cadastral area, parishes, point sources, land parcels) was carried out by means of the tools for spatial analyse in GIS.

Results and Discussion

The distribution of ammonia emissions is extremely variable in regional scale (district, county) and in local scale (surrounding of emission sources and stables). Ammonia emissions are concentrated in intensively covered agrarian areas at close vicinity of stables and storage facilities and industrial sources. Dominant share on the total ammonia emission presents emission from agricultural production (ranging as far as 95% of total emission), first of all emission from livestock breeding (housing, storage and application of manure, waste and grazing). In 2002, emissions from technologic sources participated on total ammonia emissions in the Czech Republic by 1 % and emissions from human population by 4 %.

Agricultural Sources

Emission factors, animal numbers and ammonia emissions for individual categories of animals in the Czech Republic in 2002 are presented in Table 1.

Table 1. Emission factors (kg NH_3 animal^{-1} year^{-1}), numbers of animals and ammonia emissions (kt NH_3 year^{-1}) for individual categories of animals in the Czech Republic in 2002

Animal category	Emission factor (kg NH_3 animal^{-1} year^{-1})	Animal numbers	Emission (kt NH_3 year^{-1})
Dairy cows	27.9	502 500	14.0
Cows	16.2	131 300	2.1
Heifers	16.2	341 500	5.5
Calves	16.2	226 400	3.7
Bulls	16.2	221 300	3.6
Other cattle	16.2	4 500	0.1
Total cattle		1 427 500	29.0
Total sows	17.44	289 000	5.0
Sucking-pigs	6.5	735 200	4.8
Pigs	8.3	2 423 200	20.1
Total pigs		3 447 400	29.9
Broilers	0.21	21 950 100	4.6
Layers	0.27	6 704 700	1.8
Turkey cocks and hens	0.92	893 600	0.8
Other poultry	0.21	288 100	0.1
Total scraping poultry		29 836 500	7,2
Geese and dugs	0.73	306 500	0.2
Total poultry		30 143 000	7.5
Horses	8	14 000	0.11
Sheep	1.34	65 000	0.09
Goats	1.34	3 800	0.01

Ammonia emissions from predominant sources, i.e. emissions from livestock breeding, tend to occur on specific land use types. Emissions from housing of livestock and from storage of farm manures are defined spatially by a zone of agricultural source. Emissions from applicated farm manures and mineral fertilisers occur in a point of application, i.e. on fields (arable land). Ammonia emissions emitted during grazing period of cattle, horses, sheep and goats are linked spatially with pastures.

Predominant part on the total ammonia emissions from livestock breeding pertains to pig breeding (45 %) and cattle breeding (44 %), i.e. together nearly 89 %. A part of poultry farming emissions is about 11 %. A part of emissions from breeding other livestock categories (horses, sheep, goats) is rather marginal (less than 1 %). Small equipments of capacity to 179 cattle peaces share on ammonia emissions from cattle breeding mostly (more than 58 % of total emissions from cattle breeding in the Czech Republic).

Mineral Fertilisers

Ammonia emissions from application mineral fertilisers on the territory of the Czech Republic in 2002 have been estimated to be in total 1.7 kt NH_3 year^{-1}. The total consumption of nitrogenous mineral fertilisers in the time period 1994/1995 to 1998/1999 decreased approximately by 10 % and raised during the period 1998/1999 to 2000/2001 again by about 13 %.

Industrial Sources and Human Population

From the Register of Emissions and Air Pollution Sources have been selected 252 industrial (technologic) ammonia emissions sources. These sources were localised on the territory of the Czech Republic by means of geographic coordinates and according to cadastral parcels classification. The total ammonia emission from industrial sources recorded in the Czech Republic in 2001 amounts 0.8 kt NH_3 year^{-1}.

Ammonia emission from human population on the territory of the Czech Republic calculated on basis of the data provided by the Census 2001 (CSO, 2003), was estimated to be 3.1 kt NH_3 year^{-1}.

Total Ammonia Emission

Total ammonia emission for the Czech Republic in 2002 calculated as a sum of ammonia emissions from cattle breeding, from mineral fertilisers application, from emissions produced by industry and human population was estimated to be 72.3 kt NH_3 year^{-1}. The spatial distribution of the total ammonia emissions (kg NH_3 – N ha^{-1} year^{-1}) from agricultural and other sources in the Czech Republic on a 5x5 km grid resolution in 2002 is represented in Figure 1.

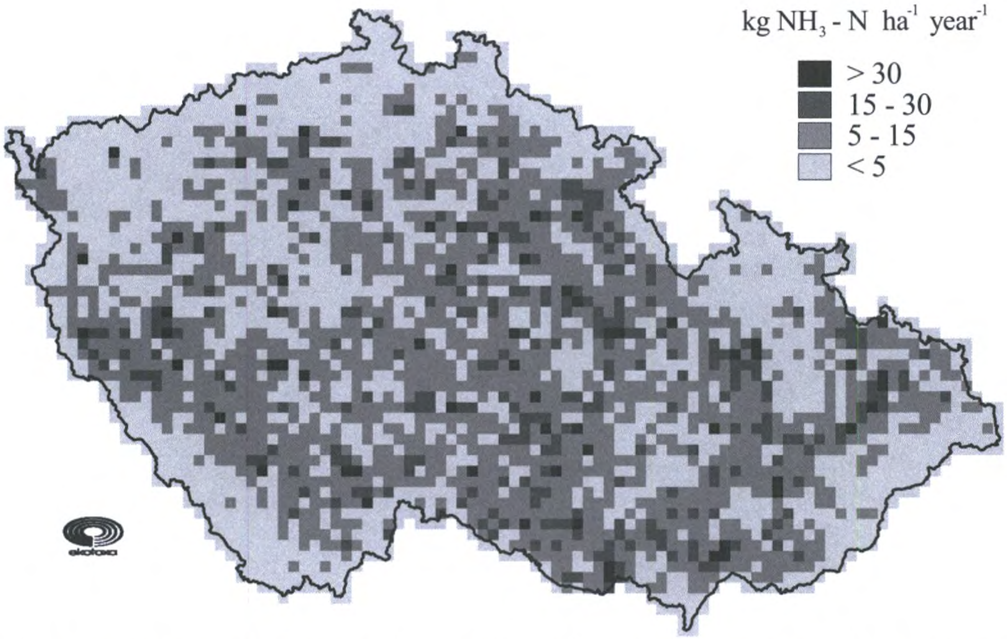


Figure 1. Ammonia emissions (kg NH_3 – N ha^{-1} year^{-1}) from agricultural and other sources in the Czech Republic on a 5x5 km grid in 2002

Conclusions

The actualised data on ammonia emissions including their location have been used for modelling the spatial distribution of ammonia emissions on the territory of the Czech Republic in 2002. The total values of ammonia emissions have been computed as a sum of ammonia emissions from livestock breeding, emissions from mineral fertilisers application, emissions from industrial sources and emissions produced by human population. These were distributed by means of GIS tools into a regular 5x5 km grid covering all of the territory of the Czech Republic.

From the results of spatial emission balance it follows that national emission ceiling for the Czech Republic by 2010 (80 kt NH_3 year^{-1}) (MECR, 2003) was not exceeded by the actual ammonia emission level in 2002 (72.3 kt NH_3 year^{-1}). A great scope at achievement emission ceiling by 2010 can be found at abidance good agricultural practice and spatial usage of verified technologies decreasing ammonia emissions from agricultural production.

Acknowledgement

This study was funded by the Ministry of Environment of the Czech Republic (project VaV/740/1/02).

References

CHMI. 2002. Air pollution in the Czech Republic in 2001. The Czech Hydrometeorological Institute, Prague, 161 pp.

CSO. 2003. Statistical Yearbook of the Czech Republic 2002. The Czech Statistical Office, Prague, 360 pp.

Dragosits, U., M.A. Sutton, C.J. Place, and A.A. Bayley.1998. Modelling the spatial distribution of agricultural ammonia emissions in the UK. In: van der Hoek, K.W, J. W. Erisman, S. Smeulders, J.R. Wisniewski, and J. Wisniewski (eds.), *Nitrogen, the Confer-N-s: Proceedings of First International Nitrogen Conference,* March 23-27, 1998. Noordwijkerhout. The Netherlands, Elsevier Science, Amsterdam, pp. 195 – 203.

Hoek van Der, K.W. 1998. Estimating Ammonia Emission Factors in Europe: Summary of the Work of the UNECE Ammonia Expert Panel. *Atmospheric Environment* 32: 315-316.

Klaassen, G.1990. Emissions of ammonia in Europe. IIASA WP-90-68. Laxenburg, Austria, 39 pp.

MECR. 2002. Decree of the Government of the Czech Republic No. 353/2002. Ministry of the Environment of the Czech Republic, Prague, 55 pp. (in Czech)

MECR. 2003. Decree of the Government of the Czech Republic No. 417/2003. Ministry of the Environment of the Czech Republic, Prague, 2 pp. (in Czech)

Zapletal, M. 1997. Atmospheric deposition of acidifying components on the territory of the Czech Republic. (Ph.D. Thesis., The Technical University of Mining and Metallurgy at Ostrava). Silesian University at Opava, Opava, 161 pp. (in Czech, summary in English)

Zapletal, M. 1998. Atmospheric deposition of nitrogen compounds in the Czech Republic. *Environmental Pollution* 29: 305-311.

Zapletal, M. 2001. Atmospheric deposition of nitrogen and sulphur compounds in the Czech Republic. *The ScientificWorld* 1 (S2): 294-303.

Zapletal, M. 2002. Atmospheric deposition of nitrogen and sulphur compounds in the Czech Republic. In: Galloway, J., E. Cowling, J.W.Erisman, J. Wisniewski, and C. Jordan (eds.): Optimizing Nitrogen Management in Food and Energy Production and Environmental Protection. Contributed Papers from the 2nd International Nitrogen Conference, 14 –18 October 2001. Potomac, Maryland, USA, Lisie, Abingdon, Exton, Tokyo, pp. 294-303.

Zapletal, M., P. Chroust, D. Kuňák, M. Sáňka, M. Fara, I. Skořepová, D. Fottová, T. Pačes, H. Kazmarová, P. Čupr, E. Budská, P. Fabiánek, and J. Seják. 2004. Efficiency investigation of measures for reduction of air pollution based on abatement of negative effects of pollutants on environmental compartments and human health. Project VaV 740/1/02. Ekotoxa Opava, Opava, 600 pp. (in Czech, summary in English)

Characterization of Particulate Emission from Animal Feeding Operations with Three-wavelength Lidar Using Simultaneous In-situ Point Measurements as Calibration Reference Sources

Vladimir V. Zavyalov[1], Gail E. Bingham[1], Thomas D. Wilkerson[1], Jason Swasey[1], Christian Marchant[1], Christopher Rogers[1], Randy Martin[2], Phil Silva[2], and Vishal Doshi[2]

[1]Space Dynamics Laboratory, 1695 North Research Parkway, North Logan, UT 84341
Tel: 435-797-4116 Fax: 435-797-4599 vladimir.zavyalov@sdl.usu.edu
[2]Utah State University, Logan, UT 84322

Abstract

Lidar (LIght Detection And Ranging) provides the means to quantitatively evaluate the spatial and temporal variability of particulate emissions from agricultural activities, including animal feeding operations. A three-wavelength portable scanning Lidar system built at the Space Dynamic Laboratory (SDL) is used to extract optical properties of the particulate matter from the return Lidar signal and to convert these optical properties to physical parameters including the spatial distribution of particulate concentration around the agricultural facility and its temporal variations. The inversion algorithm developed to retrieve physical parameters of the particulate matter takes advantage of measurements taken simultaneously at three different wavelengths (355, 532, and 1064 nm) and allows us to estimate the particle size distribution in the emitted plume as well; however, quantitative evaluation of particulate optical and physical properties from the Lidar signal is complicated by the complexity of particles composition, particle size distribution, and environmental conditions such as the ambient humidity. Additional independent measurements of particulate physical and chemical properties are needed to unambiguously calibrate and validate the particulate physical properties retrieved from the Lidar measurements. In this paper we present results of the particulate emission characterization obtained by simultaneous remote measurements with Lidar and point measurements at the feeding operation site with standard equipment including optical particle counters, portable PM_{10} and $PM_{2.5}$ ambient air samplers, multistage impactors, an aerosol mass spectrometer, and ion chromatography.

1. Introduction

Agricultural operations produce a variety of particulates and gases that influence ambient air quality and are important to the well-being of humans, animals, and plants. Concentrated Animal Feeding Operations (CAFO) are being investigated by government regulators as one of the major sources of air and water pollutants. The United States Department of Agriculture (USDA) and Agriculture Research Service (ARS) have developed a program to measure the level of pollutants crossing the CAFO boundary and the source and transport phenomenon associated with the pollutant release. The federal government regulations of CAFO are based on point-source pollution measurements located around CAFO facilities. These localized point measurements cannot adequately determine the spatial and temporal distribution of pollutant emission over extended atmospheric regions. Additionally, variations in environmental conditions and pollutant transport activities make it practically and economically infeasible to monitor actual pollutant source strength using point sensors. Lidar (LIght Detection And Ranging) technology (Measures, 1984) provides a means to derive quantitative information of particulate spatial distribution and optical properties over remote distances using one instrument located at a single convenient point. The Space Dynamics Laboratory (SDL) at Utah State University has teamed with ARS researchers to build a Lidar system for remote sensing of pollution from agricultural activities including CAFO. A combination of scanning geometry with a high repetition rate laser allow us to measure representative three dimensional (3D) cross sections of particulate clouds in a reasonable time of a few minutes. For agricultural assessment, the most representative Lidar scan pattern employs repeated vertical scans of the atmosphere on the upwind and downwind sides of a pollutant emission source. When combined with wind speed information, this pattern allows the estimate of source flux rates using the input-minus-output flux difference method (Hipps, 1995). High spatial resolution of 5m and 3D representation of the measured distribution makes the Lidar system a unique instrument for particulate flux measurements and monitoring of flux temporal and spatial variations.

The Lidar technique measures the return signal of laser light scattered by the atmosphere and is equivalent to the integral of the backscatter cross section of the particles (aerosols) present in the atmosphere with the particle size distribution as the weighting function. The aerosol backscatter cross section is uniquely determined by the physical and chemical properties of the aerosols (size, shape, and complex refractive index) and the laser wavelength. The wavelength dependence on the backscatter coefficient is mainly determined by the aerosol size distribution and refractive index. Different aerosol types have different refractive indices and size distributions, which implies that it is possible to discriminate aerosol types according to the wavelength dependence measured by a multiple wavelength Lidar. Using backscatter coefficients measured at several laser wavelengths, the physical properties of aerosols, including size distribution and particle concentration, can be retrieved by determining a solution to the Mie integral equations describing scattering properties of the aerosol (Bockmann, 2001). Thus a multiwavelength Lidar system can provide not only information on the 3D distribution of particulate matter but also information on the particulate size distribution in a 3D space with the ability to convert this information to the standard EPA (Environmental Protection Agency) mass concentration units like PM_{10}, $PM_{2.5}$, and $PM_{1.0}$.

Aerosol sounding techniques for the retrieval of physical aerosol parameters from multi-wavelength Lidar measurements have been developed since the 1980s and have made major progress in the past five years (Heintzenberg et al., 1981; Rajeev et al., 1998; Muller et al., 1999; Bockmann, 2001; Veselovskii et al., 2004). Unambiguous and stable retrieval of aerosol physical parameters requires measurements of backscatter coefficients at least at three laser wavelengths and aerosol extinction coefficients at least at two different wavelengths using additional Raman channels (Althausen et al., 2000; Bockmann, 2001). However, from the instrumental point of view, multi-wavelength Lidar systems with additional Raman channels are still very expensive and complicated to operate. Moreover, Raman signals are comparably weak and require significant integration time to achieve a reasonable signal to noise ratio to be useful for the retrievals (around 30-60 minutes for one measurement (Althausen et al., 2000)). Because of this, most Lidar systems with Raman channels typically operate at night to reduce background radiation from the atmosphere. For agricultural applications we need an inexpensive, robust and easily operated system that is able to provide particulate emission measurements in a matter of a few seconds under any meteorological and diurnal conditions and still be able to distinguish between different types of particulate emissions. A three-wavelength Lidar system appears to be a reasonable tradeoff between accuracy and stability of retrievals while providing the ability to operate under different environmental conditions with minimal measurement time. To date, a significant database of atmospheric aerosol characteristics has been obtained using a combination of satellite and ground based observations (Hess, 1998; Dubovik et al., 2002). Using this database, several researchers have shown that the physical properties of assumed aerosols can be successfully retrieved based on measurements of backscatter coefficients at only three wavelengths (Sasano et al., 1989; Rajeev et al., 1998; Del Guasta et al., 1994).

In this paper we present the initial results of the particulate emission characterization obtained by simultaneous remote measurements with a 3-wavelength Lidar system and in-situ point particulate measurements performed with standard EPA approved equipment. The combination of in-situ and remote measurements with the Lidar system pursues a twofold goal. First, particulate chemical and physical parameters measured in situ are used to make assumptions on the complex refractive index and type/shape of particle size distribution of particulate emissions present on the experiment site. In-situ measurements are also used to constrain the inverse solution to minimize overall errors and uncertainties in the Lidar measurements and the data analysis process. Second, in-situ measurements are used to calibrate and verify the results of Lidar retrievals. The experiment was conducted at the deep-pit swine production facility situated near Ames, in central Iowa, for approximately three weeks during August and September of 2005. An integrated system to measure whole facility emission was designed to characterize the complex structure and temporal/spatial variations in the particulate emission rates often associated with production operations (Bingham et al., 2006).

2. Experiment Setup and Overview of Employed Instrumentation

A schematic diagram of the deep-pit swine production facility and instrumentation employed on this site are shown in Figure 1.

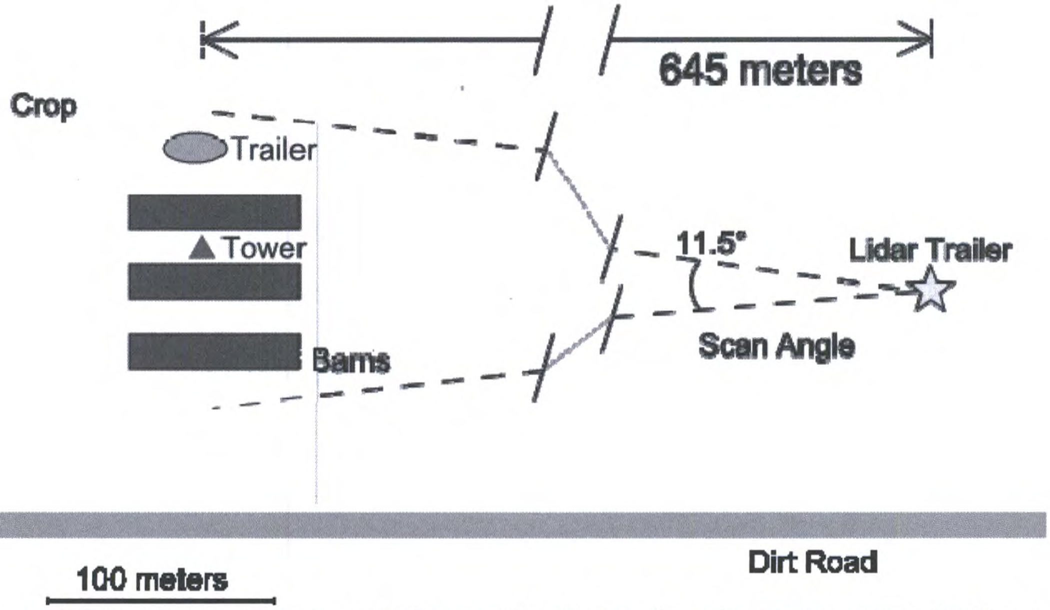


Figure 1. Experimental site layout showing locations of in-situ sensors and the Lidar system

The facility consisted of three separate, parallel barns, each of which housed around 1,250 pigs. The area around the facility was topographically flat and surrounded by fields of soybean and corn. A number of MetOne Optical Particle Counters (OPCs) 9722 were employed around this hog facility with units mounted at various heights on a tower erected at the center of the facility and a point source sensor trailer to provide information on particle size distribution continuously in real time. OPCs have the ability to count airborne particles in eight size ranges from 0.3 to 10 μm in diameter, with sampling time of 20 sec. A pair of Tisch Cascade impactors at the central tower and sensor trailer provided filter-based particle size fractionation and concentration measurements in a range of 0.37-9 μm. To measure chemical composition, real time particle ionic composition, and fine particle size distribution, the Aerodyne Aerosol Mass spectrometer (AMS) was deployed at the trailer. Portable $PM_{10}/PM_{2.5}$ (AirMetrics MiniVol) samplers and passive NH_3 (Ogawa Model 3300) samplers were arrayed vertically and horizontally around the three-barn production facility, and data were collected on a daily-averaged basis. The AirMetrics samplers were operated with $PM_{2.5}$ impactor separation heads for approximately the first half of the field study and were than switched to the PM_{10} heads for the remaining portion of the study. A monitoring Davis weather station was established approximately 40 m to the north of the nearest barn to record the typical suite of meteorological parameters (wind speed, direction, temperature, etc.) for determination of near-source atmospheric advection and dispersion. A detailed description of the instrumentation and results of the in-situ measurements of particulate and gas emission are reported by Martin et al. in the proceedings of this Workshop.

The AGLITE Lidar instrument used in this study is a three-wavelength lidar system being designed and constructed at SDL under a contract with the ARS. A single diode-pumped Nd:YAG laser operating simultaneously at fundamental (IR-near infrared 1.064 μm), doubled (V-visible 0.532 μm), and tripled (UV-near ultraviolet 0.355 μm) frequencies is used as a transmitter of short impulses of radiation to probe scattering particles in the atmosphere. The high repetition rate of 10 kHz allows the use of low pulse energy for eye safe operations at the close ranges required for agriculture applications. The laser beam diameter is 10 mm, and beam divergence is approximately 0.2-0.3 mrad after beam-expanding optics. Outgoing laser energy is monitored by photo-sensors, and this information is transmitted to the data processing unit. The laser light backscattered from particles in the atmosphere is directed by a scanning mirror to a Newtonian telescope with a main-mirror diameter of 28 cm and a field of view (FOV) of 0.46 mrad. The beam-separation unit is used to split up the return backscattered light at three different channels according to wavelength. A photon counting detection system is chosen to detect low return signal simultaneously on each channel. Interference filters are placed in front of each detector to suppress background daylight radiation from the atmosphere and optical cross talk between channels. The data from the photon counting unit are read out by a digital processing unit, averaged across a predetermined set of laser pulses, displayed in a real time, and stored and/or transmitted for further processing. The whole Lidar system was optimized for eye safe operation at all three wavelengths to allow full daylight operations at ranges from 0.5 to 15 km with a minimal range resolution of 5 m. Technical details of the Lidar design and construction are described by Wilkerson et al. in the proceedings of this Workshop.

The AGLITE Lidar system is trailer-mounted, and the scanning mirror is elevated above the trailer roof to provide a 270^0 azimuth by 45^0 elevation field of measurement. A digital camera co-aligned with the field of view of the Lidar and capturing imagery of the down-range scene is also mounted on the beam director. The AGLITE electronic control system automatically coordinates and synchronizes all the functions of the Lidar, scanning turret, data acquisition system, digital camera, and weather station to provide a complete data package and makes it available to the operator for further analysis. The Lidar trailer was placed at approximately 650 m east of the central tower (see Figure 1) and accompanied by a second weather station to monitor atmospheric conditions near the Lidar. This location of the Lidar system allowed full 3D volume measurements of particulate emissions from the three-barn feeding operations from a single observation point. Typical settings for the Lidar during operation are the following: accumulation time for return signal of 0.5-3 sec per measurement (5,000-30,000 laser pulses), range resolution of 5-15 m at ranges of 0.5-15 km, and azimuth and elevation scan speed of 0.05-2.0°/sec.

3. Inversion of the Lidar Signal to Retrieve Optical and Physical Properties of Aerosols

The Lidar return power from range R for two distinct classes of scatters may be written in the following form (Measures, 1984; Klett, 1985):

$$P(R,\lambda) = C \cdot \frac{P_0(\lambda)}{R^2} \cdot T^2(R,\lambda) \cdot \left[\beta_b(\lambda,R) + \beta_a(\lambda,R)\right] \qquad (1)$$

Where C= $\xi(R,\lambda) \cdot A_0 \cdot \Delta R = \xi(R,\lambda) \cdot A_0 \cdot \tau_d \cdot c/2$ is the Lidar calibration constant that includes losses in the transmitting and receiving optics $\xi(R,\lambda)$ (Lidar overlap geometrical form factor), the effective telescope area A_0, and range increment ΔR acquired by the Lidar that is defined by the detector integration pulse width τ_d and speed of light c. $P_0(\lambda)$ is a total laser power transmitted at wavelength λ. $\beta_b(\lambda,R)$ and $\beta_a(\lambda,R)$ stand for the backscatter coefficients of air molecules and aerosol particles, respectively. Round trip laser beam transmittance along the beam pass $T^2(R,\lambda)$ is defined by the equation:

$$T^2(R,\lambda) = \exp\ \{-2 \int_0^R [\alpha_b(\lambda,R') + \alpha_a(\lambda,R')]\partial R'\} \qquad (2)$$

Where $\alpha_b(\lambda,R)$ and $\alpha_a(\lambda,R)$ are extinction coefficients of air molecules and aerosols, respectively.

Retrieval of aerosol physical parameters from a raw Lidar signal involves four major steps:

1) Account for geometrical form factor of the telescope receiving optics and scattered sunlight background radiation. The geometrical form factor in equation (1) takes into account the overlap between the transmitted laser beam and the FOV of the telescope receiving optics with the central obstruction and can be estimated theoretically (Measures, 1984). In real life, this factor can change after setup of the portable Lidar system in the field, and the real geometrical form factor $\xi(R,\lambda)$ shall be determined experimentally. To calculate the geometrical form factor at field conditions, the polynomial regression method for an inhomogeneous atmosphere proposed by Dho et al., 1997 is used. This method does not require special atmospheric conditions and in many cases can be used during operational field measurements. During daylight observations the background radiation of sunlight scattered by the atmosphere dominates the Lidar return signal at long distances. For each Lidar measurement this background radiation is approximated by least

squares fitting to a constant value at distances of 13-15 km and then subtracted from the total Lidar return signal.

2) Calculate the optical parameters (backscatter and extinction coefficients) of the background aerosols and particulate emission from the feeding facility at three wavelengths, utilizing Klett's analytical solution for two scattering components (Klett, 1985). Typically, the molecular part of equation (1) can be calculated using standard atmosphere conditions at different altitudes and the well known and parameterized refractive index, backscatter and extinction properties of the air molecules (Bockmann, 2004). Aerosol backscatter and extinction coefficients remain two unknowns in a single Lidar equation that describes one particular Lidar measurement. We are using the standard solution of this equation proposed by Klett that involves a priori assumption of the relationship between aerosol extinction and backscatter coefficients that is usually called the Lidar ratio L:

$$L_a=\alpha_a(\lambda,R)/\beta_a(\lambda,R) \qquad (3)$$

Another assumption deals with selecting a boundary value to determine the constraint factor required by Klett's solution, usually a calibration or reference value of the extinction coefficient $\alpha_{aD}=\alpha_{aD}(\lambda,R_D)$ independently measured at a certain distance R_D.

The original solution of equation (1) has been deducted for typical atmospheric applications when the Lidar system is looking straight up so that molecular contribution is significant for altitudes above the aerosol boundary layer (ABL). For agricultural applications all measurements are conducted close to the ground, and the main contribution to atmospheric scattering is determined by aerosols while the molecular contribution is negligibly small. In this case we are still dealing with two distinct types of scatterers such as background aerosols and airborne particulate matter emitted from the agricultural activities. For this application, the original solution of equation (1) is still valid, and we attribute the subscript "b" to the atmospheric background aerosols while the subscript "a" will refer to the particulate emission. Typically the agricultural particulate emission is spatially localized around the emission source while the rest of the Lidar signal is dominated by the surrounding background aerosols. In close proximity to the emission source the background aerosol loading is typically homogeneous, and a standard slope method (see for instance Klett, 1985) can be applied to retrieve the extinction coefficient of the background aerosols. For each wavelength, the slope of a line that has been fit in a least-squares sense to the curve S(R) is used as an estimate of $\alpha_b(\lambda,R)$ over the interval where $S(\lambda,R)=\ln[P(R,\lambda)\cdot R^2]$ presents nearly a straight line.

3) Estimate parameters of particle size distribution for background aerosols and particulate emission using an iterative technique to minimize the difference between simulated (Mie theory) and measured extinction coefficients at three Lidar wavelengths.

Based on the OPC data measured in situ, we approximate the particle size distribution by a bimodal distribution, using the standard Power low function for the accumulation mode:

$$n(r)=N_1r^{\nu} \qquad (4)$$

and lognormal size distribution for the coarse particle mode:

$$n(r)=\frac{1}{r}\frac{N_2}{(2\pi)^{1/2}\cdot\ln\sigma}\cdot\exp[\frac{(\ln r-\lg r_m)^2}{2\cdot\ln^2\sigma}] \qquad (5)$$

Where N_1 is a constant related to the total number of particles with radius r in accumulation mode and ν is the size index that generally varies in a range of 3-5. N_2 represent the total density of particles in the coarse mode with mode radius of r_m and width of distribution σ.

To estimate the mode radius and particle number densities N_1 and N_2, we are using a modified version of the minimization technique described by Del Guasta (1994). The minimization function in our case is constructed on the differences between the extinction coefficients α_i, retrieved from the Lidar signal, and α_i^{calc}, calculated using Mie theory (Bockman, 2001) at three laser wavelengths. Arbitrary values of the Lidar ratio for both background aerosols L_b and particulate due to convective turbulence. The horizontal scan in Figure 3 was taken under west wind conditions, and the data show that both particulate emission and fugitive dust were swept by the wind toward the Lidar and extended horizontally as compared to Figure 2. The time series in Figure 4 was measured under south wind conditions when fugitive dust was blown from the south road, which was parallel to the line of Lidar range measurements. A weaker signal at a range of ~650 m represents particulate emission between swine barns measured at a height of ~6 m in a close proximity to the OPC sensor mounted on the tower.

Due to the spatial and temporal separation of different particulate emissions in the return Lidar signal, these events can be easily processed separately so that optical and physical properties of particulate emissions from different sources can be extracted using single Lidar scan. For all cases the return signals from fugitive dust clouds were about an order of magnitude stronger than the signal from barn particulate emissions, which in some cases only slightly exceeded Lidar returns from the background aerosols in the air surrounding the facility. The retrieval procedure briefly described in a previous section was tested on both cases, for which there are remarkably different physical origins of particulate emissions involved.

Proper conversion of Lidar data involves the construction of a model for the particulate composition and size distribution based on in-situ measurements. Following the OPAC database (Hess, 1998) we assumed that fugitive dust is composed of a mixture of quartz and clay materials. Particle size distribution was approximated by a bimodal distribution based on in-situ measurements with OPC sensors. The results of the retrievals are shown in Figure 5. Extinction coefficients retrieved at three laser wavelengths in step 2 are shown in Figure5A as a function of the distance from Lidar. The dust cloud chosen for conversion is extended for almost 1km from the west road toward Lidar and shows different ratio of extinction coefficients measured at different wavelengths. As a result, the parameters of particle size distributions estimated at different distances are different, which leads to a different ratio of small (accumulation mode) and large (coarse mode) particles within the same dust cloud.

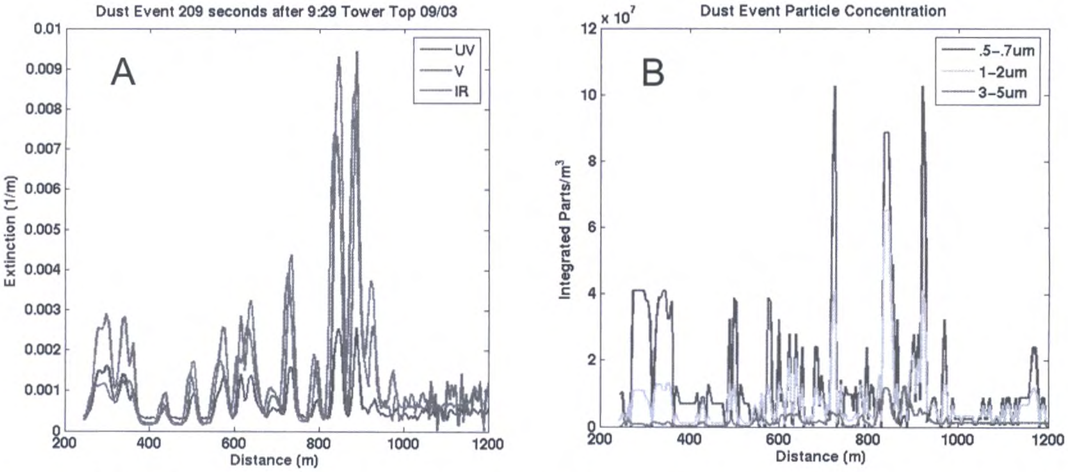


Figure 5. Optical and physical properties of fugitive dust cloud retrieved from the Lidar signal at different distances from the Lidar. A) Extinction coefficient at three Lidar wavelengths. B) Volume concentrations for dust particles with radius range of 0.5-0.7 µm, 1-2µm, and 3-5 µm estimated from extinction coefficients.

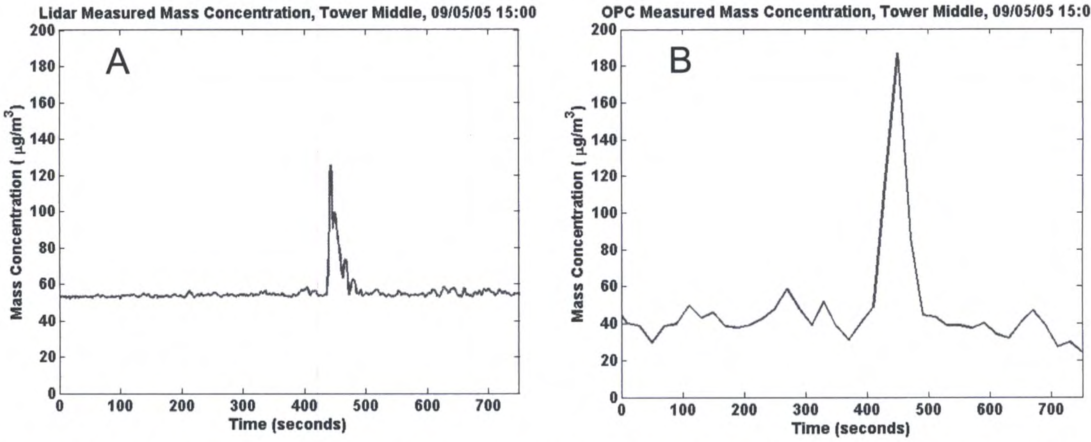


Figure 6. Time series of dust particles mass concentration PM_{10} as measured by the Lidar (graph A) and OPC sensor mounted on the top of the tower (graph B). Measurements are collocated in time and space.

Volume concentrations of particles with radiuses in a range of 0.5-0.7 μm, 1-2 μm, and 3-5 μm were calculated from size distributions estimated in the retrieval step 3 and are shown in Figure 5B. It is seen that the large particle concentration decreases while the concentration of small particles increases with the distance from road along the wind, showing settlement of large particles from the dust cloud.

Mass concentration PM_{10} estimated from the Lidar signal is compared with in-situ measurements by OPC in Figure 6. Both measurements represent a time series of particulate emission measured simultaneously at the top of the tower, where the peak concentration represents a fugitive dust event and the base signal is mostly due to the background aerosols. Concentration of background aerosols and fugitive dust from the road measured by the Lidar are both in good quantitative agreement with coincident OPC measurements.

As mentioned previously, particulate emissions from the swine facility only slightly exceed the background aerosol loading. Lidar returns are still sensitive to these small variations, which can be easily spotted due to their spatial localization in the return Lidar signal. The OPC sensors provide continuous point measurements so that background aerosols and particulate emission can be distinguished only by the numbers of counts (intensity of the signal). Comparison of the OPC data measured between barns and far away from barns where only background aerosol is present shows that particulate emission counts are on the level of natural variability of the background aerosol loading. In this case it is difficult to extract exact information on the particulate size distribution from the OPC data. Taking this into account, we also approximated the particulate size distribution by a bimodal distribution, as in a case of fugitive dust and background aerosols. Preliminary chemical analysis of the particulate emission measured in situ (Martin, 2006) shows that its composition may be considered as a standard water soluble aerosol mixture composed from various kind of sulfates, nitrates, organic carbon, etc. (Hess, 1998). Assuming these approximations, Lidar returns from a particulate plume were processed and parameters of particulate size distribution were estimated as described in section 3. Once parameters of particle size distribution are estimated, the mass concentration of $PM_{2.5}$ is calculated assuming an average particle density of 1.8 g/cm^3. The results of these calculations are shown in Figure 7 for two time series of Lidar measurements pointed at the middle of the tower and at its top.

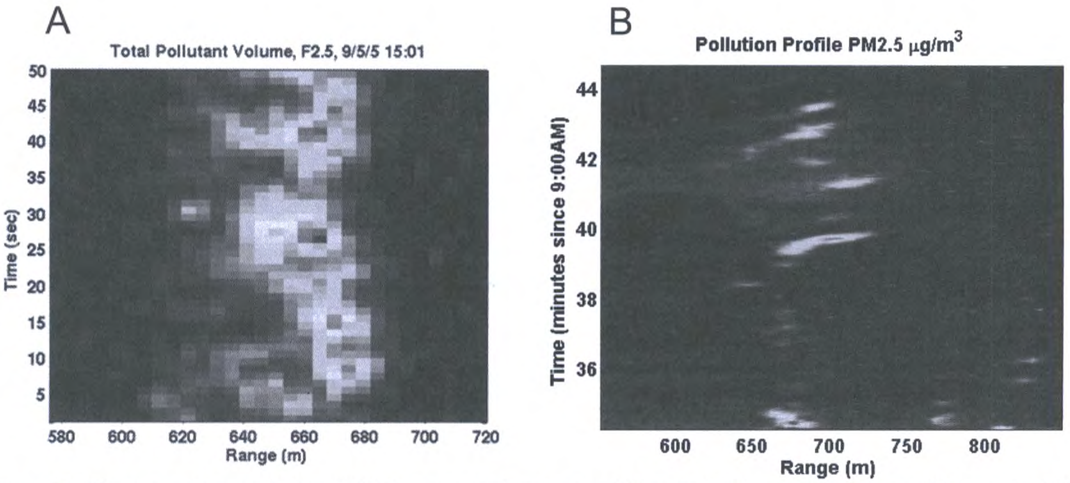


Figure 7. Comparison of the PM10 concentration at the middle of the tower (A) and at the tower top (B) as estimated from the Lidar returns.

It interesting to note that particulate was emitting from barns as periodical events with periodicity of 3-10 seconds as can be seen in both images of Figure 7. During these observations the Lidar accumulation time was set to 1 second, and that was enough to resolve the periodic nature of particulate emission. The accumulation time of OPC monitoring sensors was 20 seconds so that OPC counts represent a time average of periodic events that contributed to the inability of OPC sensors to resolve clearly particulate emissions and background aerosols.

Preliminary comparisons of the Lidar retrievals with in-situ $PM_{2.5}$ measurements shows that total $PM_{2.5}$ mass concentration agrees within an order of magnitude.

5. Conclusion

A three-wavelength portable scanning Lidar system has been developed at SDL to derive information of particulate spatial distribution and optical/physical properties of aerosols over remote distances using an instrument located at a single convenient point. Preliminary results discussed in this paper show the great potential of remote Lidar measurements to quantitatively characterize particulate emission from different sources. Lidar technology represents a unique technique to characterize spatial and temporal variations of particulate emission from any source met in field conditions. Additionally, the high measurement rate of the Lidar allows us to capture temporal variations in particulate emissions on the order of seconds that could not be resolved by most in-situ point instrumentation. The use of extinction/backscatter ratios derived from Lidar measurements at three laser wavelengths was found to be a promising method for remote measurements of the size distribution of particulate emissions present in the field. These emissions include background aerosols, emissions from the feeding facility, and fugitive dust from the road. The strength of Lidar returns from these sources varies by an order of magnitude, and the inversion algorithm developed to process three wavelengths Lidar data gives meaningful results for all sources of particulate emissions. Retrieval results for fugitive road dust are in a good agreement with coincident in-situ measurements by OPC sensors. The retrievals of mass concentration of particulate emission from the feeding operation agree on the order of magnitude with in-situ measurements performed with $PM_{2.5}$ ambient samplers. The main uncertainties involved in such a method are due to the incomplete knowledge of the particulate refractive index and parameters of particle size distribution to further constrain the iterative minimization technique employed in this method to estimate parameters of assumed size distribution. Further work is needed for both remote Lidar and in-situ point measurements to verify, calibrate, and correlate all types of measurements performed in the field.

References

Althausen D., D. Muller, A. Ansmann, U. Wandinger, H. Hube, E. Clauder, and S. Zorner. 2000. Scanning six-wavelength eleven channel aerosol lidar. *J. Atmos. Ocean. Technol.* 17: 1469–1482.

Bingham G. E., J. Hatfield, J. H. Prueger, T. D. Wilkerson, V. V. Zavyalov, R. L. Pfeiffer, L. Hipps, R. Martin, P. Silva, W. Eichinger. 2006. An Integrated Approach to Measuring Emissions from Confined Animal Feeding Operations at the Whole Facility Scale. Proceedings of the Workshop on Agricultural Air Quality: June 5-8.

Bockmann C. 2001. Hybrid regularization method for ill-posed inversion of multiwavelength lidar data in the retrieval of aerosol size distributions. *Appl. Opt.* 40: 1329–1342.

Bockmann C. , U. Wandinger, A. Ansmann, J. Bosenberg, V. Amiridis, A. Boselli, A. Delaval, F. De Tomasi, M. Frioud, I.V. Grigorov, A. Hagard, M. Horvat, M. Iarlori, L. Komguem, S. Kreipl, G. Larcheveque, V. Matthias, A. Papayannis, G. Pappalardo, F. Rocadenbosch, J. A. Rodrigues, J. Schneider, V. Shcherbakov, and M. Wiegner. 2004. Aerosol lidar intercomparison in the framework of the EARLINET project. 2. Aerosol backscatter algorithms. *Appl. Opt.* 43: 977-989.

Del Guasta M., M. Morandi, L. Stefanutti, B. Stein, and J. P. Wolf. 1994. Derivation of Mount Pinatubo stratospheric aerosol mean size distribution by means of a multiwavelength lidar. *Appl. Opt.* 33: 5690–5697.

Dho S. W., Y. J. Park, and H. J. Kong, 1997. Experimental determination of a geometrical form factor in a lidar equation for an inhomogeneous atmosphere. *Appl. Opt.* 36: 6009-6010.

Dubovik O., B. Holben, T. F. Eck, A. Smirnov, Y. J. Kaufman, M. D. King, D. Kino, D. Tanre, and I. Slutsker. 2002. Variability of absorption and optical properties of key aerosol types observed in worldwide locations. *J. Atm. Scie.* 59: 590-608.

Heintzenberg J., H. Muller, H. Quenzel, and E. Thomalla. 1981. Information content of optical data with respect to aerosol properties: numerical studies with a randomized minimization-search-technique inversion algorithm. *Appl. Opt.* 20: 1308–1315.

Hess M., P. Koepke, and I. Schult. 1998. Optical Properties of Aerosols and Clouds: The software package OPAC. *Bull. Am. Meteorol. Soc.* 79: 831-844.

Hinds W. C. 1998. *Aerosol technology. Properties, behavior, and measurement of airborne particles.* John Willey & Sons, Inc., New York.

Hipps L.E. and D.F. Zehr. 1995. Determination of evaporation from integrated profiles of humidity and temperature over an inhomogeneous surface. *Boundary-Layer Meteorol.* 75: 287- 299.

Klett J. D. 1985. Lidar inversion with variable backscatter/extinction ratio. *Appl. Opt.* 24: 1638-1683.

Martin R.S., V. Doshi, and K. Moore. 2006. Determination of Particle (PM10 and PM2.5) and Gas-Phase Ammonia (NH3) Emissions from a Deep-Pit Swine Operation using Arrayed Field Measurements and Inverse Gaussian Plume Modeling. Proceedings of the Workshop on Agricultural Air Quality: June 5-8.

Measures R. M. 1984. *Laser remote sensing: Fundamentals and applications.* John Willey & Sons, Inc., New York.

Muller D., U. Wandinger, and A. Ansmann. 1999. Microphysical particle parameters from extinction and backscatter lidar data by inversion with regularization: theory. *Appl. Opt.* 38: 2346–2357.

Rajeev K. and K. Parameswaran. 1998. Iterative method for the inversion of multiwavelength lidar signals to determine aerosol size distribution. *Appl. Opt.* 37: 4690–4700.

Sasano Y. and E. V. Browell. 1989. Light scattering characteristics of various aerosol types derived from multiple wavelength lidar observations. *Appl. Opt.* 28**:** 1670-1679.

Veselovskii I., A. Kolgotin, V. Griaznov, D. Muller, K. Franke, and D. N. Whiteman 2004. Inversion of multiwavelength Raman lidar data for retrieval of bimodal aerosol size distribution. *Appl. Opt.* 43: 1180-1195.

Wilkerson T.D., G. E. Bingham, V. V. Zavyalov, J. A. Swasey, J. J. Hancock, B. G. Crowther, S. S. Cornelsen, C. Marchant, J. N. Cutts, D. C. Huish, C. L. Earl, J. M. Andersen, and M. L. Cox. 2006. AGLITE: A Multiwavelength Lidar for Aerosol Size Distributions, Flux and Concentrations of Whole Facility Emissions. Proceedings of the Workshop on Agricultural Air Quality: June 5-8.

Particulate Matter Emissions from an Ohio Belt-Battery Layer Barn

L.Y. Zhao[1], T.T. Lim[2], A.J. Heber[3], H. Sun[4], C.A. Diehl[5], J.-Q. Ni[6], P.C. Tao[7], and S.M. Hanni[7]
[1]Assistant Professor, [4]Research Associate, The Ohio State University, Columbus, OH 43210
[2]Research Scientist, [3]Professor, [5]Research Engineer, [6]Technical Director, and [7]Reserch Assistant, Purdue University, West Lafayette, IN 49707

Abstract

Particulate matter (PM) emissions from concentrated poultry laying houses cause significant environmental and health concerns. Quantifying and characterizing PM emissions and effectiveness of mitigation technology for layer facilities will allow the poultry industry to control the particulate emissions more effectively. This study assessed PM emissions from a new belt-battery laying facility through six months of continuous measurement. PM10 concentration was continuously measured by using the Tapered Element Oscillating Microbalance (TEOM) system. Gravimetric samplers were used to sample total suspended particulate matter twice per week. Barn ventilation was estimated via field tests and continuous monitoring of fan control signals, vibration, and differential static pressure of all fans. The average daily mean (ADM) concentration of PM10 and TSP were 265±108 μg/dsm3 and 3070±675 μg/dsm3 at the barn exhaust locations, respectively. The ADM emission rates of PM10 and TSP from the belt battery layer barn were 20±18.9 mg/d-hen and 168±110 mg/d-hen, respectively. Based on these emission rates, 13.7 million hens would emit 100 tons of PM10 and 4.1 million hens would emit 250 tons of TSP per year, respectively. The PM10 concentration in the manure belt battery layer barn was only 45-53% of that in high-rise barns. PM10 emission rate of the belt battery barn was only about 62% that of the high-rise barns. Similar levels of TSP concentrations and emission rates in the manure belt battery barn were measured in comparison with the high-rise deep-pit layer barns.

Introduction

Air emissions from animal production facilities have caused public concern about human health, animal health, and the global environment (NRC, 2003). Dust particles carry odor, gases, and bacteria and therefore are of the greatest health concerns. In poultry buildings, the combination of dust and other air contaminants such as ammonia may cause respiratory disease, increased mortality rates, and reduced bird growth (Maghirang et al. 1991).

PM emissions are directly regulated under the Clean Air Act (CAA) in terms of ambient PM concentrations and emission thresholds for major emission sources. Among animal production facilities, poultry facilities create the most concern with regard to emitting the amounts of PM that could potentially violate CAA (Heber, 2004). Dust and ammonia emissions have created a major challenge for the viability and growth of the egg-laying industry. Therefore, understanding and controlling PM emissions to sustain the viability and growth of the poultry industry is of utmost importance. However, scientific data on PM emissions from poultry facilities are limited. Such insufficient scientific data has hindered the development of PM control technologies and effective management practices in the poultry industry. Collecting reliable emission data and accurately assessing PM control technologies and practices have become necessities.

Particulate emissions from poultry buildings were studied by European researchers at 81 poultry facilities in four countries over a 24-hour period in each of two seasons (Takai et al., 1998). This short term study showed that the mean inhalable and respirable dust emission rate from caged layer facilities with manure belts and deep pit were in the range of 398-872 and 24-161 mg/h-AU (AU=500 kg live weight), respectively. Statistically significant ($p<0.001$) effects of housing facilities on dust emissions were revealed. Lim et al. (2003) continuously studied PM concentrations and emissions of a representative US deep-pit laying hen house by using the microweighing (TEOM) and real-time ventilation monitoring technologies to determine variations of PM emissions during a 12-month period. Heber et al. (2005) has intensively quantified continuous baseline emissions of PM10 and TSP from a high-rise deep-pit poultry layer barn over a fifteen month period.

In the U.S., high-rise deep-pit and manure belt laying facilities account for 73% and 23% of current laying hen production, respectively. The manure belt laying facilities are considered a newer facility design and account for 60% new facilities (Xin, 2005). Ammonia emission rates from the manure belt laying facilities range from 0.054-0.094 g / d-hen, significantly lower than that from traditional high-rise deep-pit laying facilities which ranges from 0.83-0.90 g/d-hen (Liang et al., 2005). Sun et al. (2005) confirmed significant ammonia mitigation effects of a manure belt battery laying facility through a continuous six-month measurement.

Nevertheless, the effects of the new manure belt laying facility on PM emissions have not been scientifically documented. Quantifying PM emissions from the new manure belt layer facilities will supply the poultry industry with vital information in the adoption of new facilities and the control of ammonia and PM emissions. Thus, improvement of air quality and compliance with federal air quality laws will be facilitated. The objective of this study was to quantify PM emissions from a belt-battery laying facility.

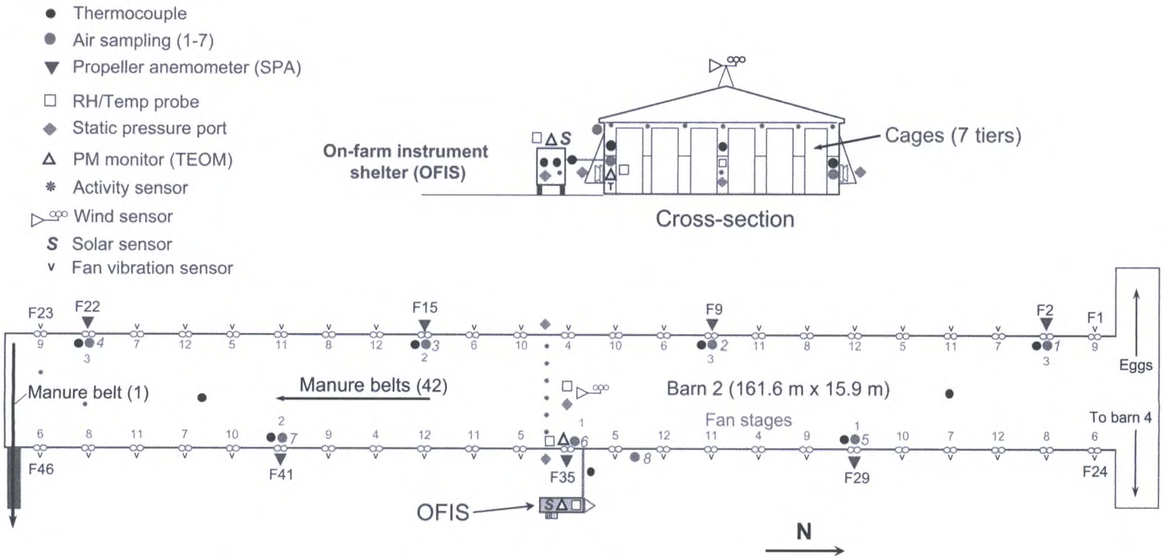
Materials and Methods

One commercial manure belt-battery layer house (MB barn) located in central Ohio was monitored for this study. On August 10, 2004, continuous PM emission measurements began to be conducted and continued for six months. Emission rates of particulate matter with an aerodynamic diameter equal or smaller than 10 micro ($PM_{10)}$ and total suspended particulate (TSP) were measured. A mobile air quality lab was used to host equipment and data acquisition systems. Tapered Element Oscillating Microbalances (TEOMs) were used to continuously monitor PM_{10} concentrations of the building's exhaust air and ambient air. Total suspended particulate concentration was measured periodically using a TSP gravimetric sampler. Barn ventilation and indoor environment was continuously monitored. Emission rates of TSP and PM_{10} were calculated by multiplying concentrations by total barn ventilation airflow rates.

Layer Barns

The MB barn had been converted from an old high-rise deep-pit barn in 2004. On the same site, another 15 barns were also in differing stages of the process of being converted from high rise barns to manure belt battery barns. The barns were oriented from north to south and were spaced 15.9-m apart. The barn used in the study was 161.6 m long x 15.9 m wide and housed approximately 168,000 hens in six rows of 7-tier crates. Figure 1 shows the schematic of the barn and sampling locations. Manure was collected on plastic belts under the cages and removed from the barn within 1 to 7 days. Air was blown on the belt to reduce manure moisture content. Continuous slot inlets over the top of each row of layer cages introduced fresh air from the attic into the barn and fourty-six122-cm belted axial fans (Model GP48G600MNA, S.N. REVA 4-04, ValAir, 2599 Old Philadelphia Pike, Bird 'N Hen, PA 17505) installed uniformly on the two side walls along the barn length exhausted the dirty air out of the barn. The fans in the barn were 6.4 m apart. The barn had 12 temperature sensors and was ventilated in 12 stages. Each stage consisted of four fans, except for stages 1 and 2, which have 3 fans each. Eggs are removed on conveyors into the egg processing plant. The lights are shut off from 8:00pm to 4:00am each night. Egg production along with water and feed consumption were recorded automatically, and daily mortalities were recorded manually.

Figure 8. Schematic of the belt battery layer barn and sampling locations



Measurement of Particulate Matter Concentration

The PM_{10} concentrations were continuously measured using the commercially available equipment, TEOM 1400a Ambient Particulate (PM10) Monitor (Rupprecht & Patashnick, Albany, NY). The TEOM PM_{10} measurement method was designated by the USEPA as an equivalent method (EPA Designation No. EQPM-1090-079). The TEOM equipment with $PM_{2.5}$ inlet is also used extensively in state and national $PM_{2.5}$ monitoring networks. The TEOM monitor is designed for ambient air monitoring, indoor air quality assessment, and exposure studies. The key to the device is a tapered element oscillating microbalance, which is an inertial mass measurement technique for making a direct measurement of the particle mass collected on a filter in real time. The device operates at an industry-standard, volume-controlled flow rate of 16.7 L/min. It can be outfitted with a variety of commercially available pre-separator inlets suitable for measuring TSP, PM_{10}, and $PM_{2.5}$ (Heber et al., 2002). The TEOM sampling head was placed near an exhaust fan and in a relatively low speed air stream. The TEOM pumps and controllers were stationed in the instrument shelter and provided vacuum to the filters via long vacuum tubes. The sample stream temperature was maintained at 50°C. The PM concentrations measured by the TEOMs were adjusted to report data at one atmosphere pressure (1 atm) and 20°C.

A three-point TSP gravimetric sampler (Wang et al, 1999) was used to collect TSP dust samples. The TSP dust sampler consisted of an isokinetic sampling head, a 37 mm diameter glass fiber filter (Millipore, 0.7-μm pore size), 37-mm diameter filter holder (Millipore Aerosol Analysis Monitor), 1.57 mm dia. critical venturi orifice, plastic tubing, and an electric air pump (Gast, model 1023-V131Q-G608X). The dust sampler was run 24 to 72 h to account for the changing PM levels over the course of one to three working days. The sampling heads were located at three different heights in front of an exhaust fan inlet (less than 0.5 m from the fan impellers). The locations of TSP sampling heads were carefully selected by using a portable vane thermoanemometer (Model 451126, Extech, Bohemia, NY),which matched the 2 m/s airflow speed required for isokinetic sampling. The filters, both new and dusty, were desiccated for 24 h before and after weighing. Acetone was used to rinse dust remnants off the sampling heads. PM concentration was calculated by dividing total PM mass collected by air volume sampled.

Ventilation and Barn Environment Measurement

Differential pressures were monitored continuously in the barns near the exhaust fans by using differential pressure transmitters (Model 2671-100-LB11-9KFN, Setra, Boxborough, MA) with a range of ±100 Pa and an accuracy of ±0.5 Pa. Atmospheric pressures were monitored with barometric pressure transducers in the TEOMs.

The operating status (on/off) of each fan stage was monitored via auxiliary contacts of fan motor control relays. Fan airflow capacities were measured in the field with a fan airflow numeration system (FANS) and a portable fan tester (Gates et al. 2004). Also, at the University of Illinois, fan airflow capacities were calibrated at an accuracy within 2% by using a fan test chamber. In addition, the operating status of each fan was individually monitored via vibration sensors (Ni et al., 2005).

The temperature of exhaust air was monitored using copper-constantan thermocouples (Type T). An electronic relative humidity (RH) and temperature transmitter (Model HMW61, Vaisala, Woburn, MA) was used to monitor temperature and relative humidity at a representative exhaust location in the barn.

Calculation of PM Emission Rates

PM emission rates were simply calculated as the product of PM concentration multiplied by barn ventilation airflow rate. Ventilation rate was adjusted to the standards if 1 atmospheric pressure and 20°C Equation 1 was used for TSP emission calculation. Equation 2 was used for TEOM PM_{10} emission calculation.

$$E = QC \qquad (1)$$

$$E = Q\frac{293}{(273+T)}\frac{P}{P_0}C_0 \qquad (2)$$

Where:

E	PM gross emission rate, µg/s
Q	Exhaust airflow rate at T, m^3/s
P	Pressure of exhaust air, atm
P_0	Standard pressure, 1 atm
C_0	PM concentration or concentration difference measured by TEOM(s), $\mu g/m^3$
T	Temperature of exhaust air, °C

The gross emission rates were calculated directly from exhaust PM10 concentrations multiplied by airflow rates. The net emission rates were calculated using the difference of exhaust PM10 concentrations and monthly average of ambient PM10 concentrations multiplied by barn airflow rates.

Data Analysis

General statistical data analysis was conducted initially to calculate average daily means (ADM) with 95% confidence interval and standard deviations of each measurement parameters. A two-way analysis of variance (ANOVA) with a complete randomized block model was used to analyze statistical differences of PM emission rates during light hours and dark hours and in winter months from the BB barn and tow high-rise deep-pit barns (HR). Paired t-test analysis was used to compare PM emission rates from the BB barn and two HR barns. The statistical significance level of 0.05 was used to judge whether the statistical hypothesis (there is no statistical difference) should be rejected ($P<0.05$) or not ($P>0.05$).

Results and Discussion

The reported results are average daily means. Basic statistics of the data were analyzed and the data were validated through cross-checking other condition parameters. With over 70% valid data, it is reasonable to conclude that these results have avoided the bias that can be caused by insufficient data.

Indoor Environment of the Barns

Daily mean barn temperature, RH, and ambient temperatures are presented in Figure 2. During the test period from Aug. 2004 to Jan. of 2005, daily average ambient temperature dropped from 26.2°C to -13.7°C with an overall average outdoor air temperature of 10.6 °C. The testing period covered hot, mild, and cold seasons. The ADM barn temperature was 20.8 °C at the middle tier cages and 22.9 °C at the barn exhausts. Since exhaust fans are located uniformly on two side walls and liner slot air inlets are located on the tops of each row of cages, the cold fresh air picked up heat, moisture and air emissions from hens as well as manure exhausted through the exhaust fans. Therefore, ADM temperature at the center middle tier cages was lower than that at the exhaust fans. The indoor temperature was slightly higher on hot days and slightly lower on cold days. Overall however, the barn temperatures were maintained relatively stable across seasons.

Daily mean RH of exhaust air ranged from 45 to 80% with an average RH of 60±6.4%. The ambient RH ranged from 48% to 98% with an average RH of 72±11%. Since the barn was not equipped with any cooling system, when outdoor RH and temperature were high in August and early September of 2004, the average indoor RH averaged 67±6.7%, which was much higher than the average RH of October to January, which was 58±5%. Therefore, the indoor RH fluctuated much more widely than did the indoor temperature.

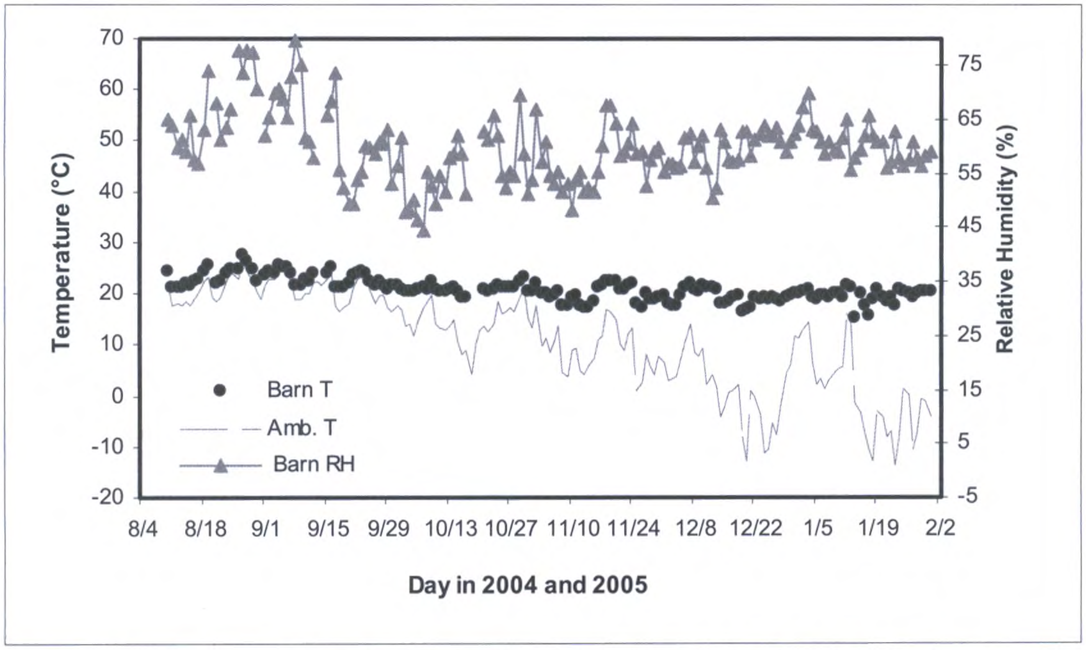


Figure 2. Daily mean temperature and relative humidity of the belt-battery barn

Figure 3 shows the building differential pressures and ventilation rates. The daily mean pressures were maintained at about -20 Pa with small fluctuations most of the time. In January, due to ice built up on the baffled air inlet and a small number of running fans, large static pressure fluctuations of about ±10 Pa was observed. Affected by weather conditions, barn ventilation rates were higher in hot August and lower in cold January. Daily mean ventilation rates ranged from 26 to 337 dsm^3/s with an average of 129±98 dsm^3/s. Ventilation rate fluctuated significantly on hot days.

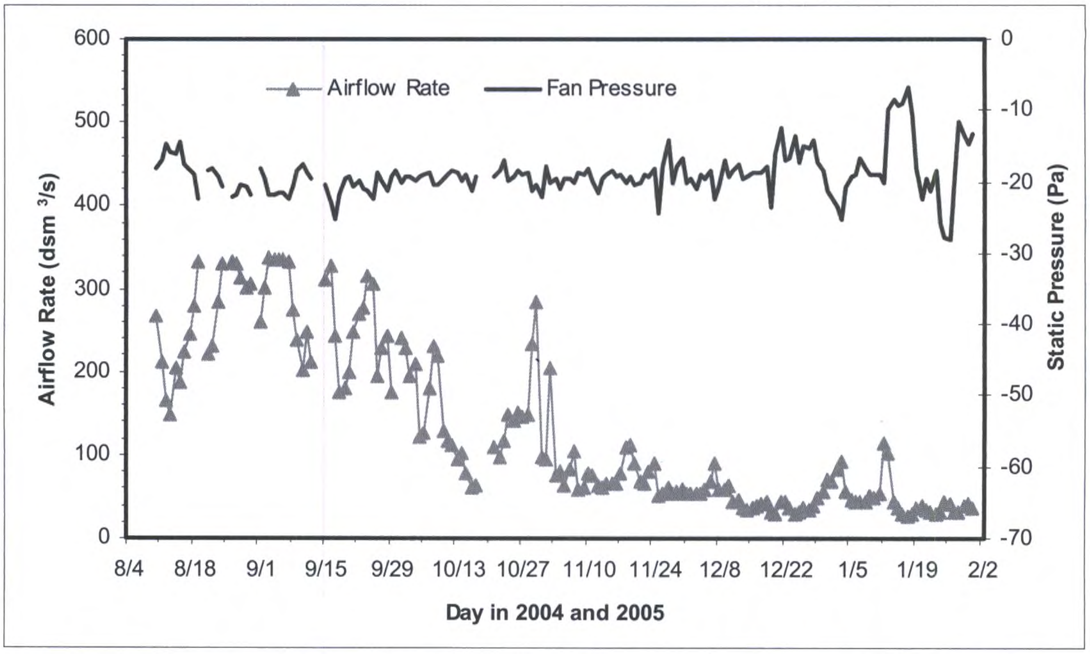


Figure 3. Daily mean ventilation rate and barn differential pressure of the Belt-Battery barn

The initial inventory was 167,741 hens, with the average hen weighing 1.44 kg. The hens grew to 1.7 kg in early December 2004 and lost weight to 1.63 kg by the end of the study on Jan 31, 2005. The ending inventory was 166,053 hens. The average inventory was 166,985 hens and the average hen weight was 1.60±0.1 kg. The total live mass ranged from 485 to 567 AU with an average of 535±25 AU.

PM10 Concentration and Emission

Figure 4 shows daily means of PM_{10} concentrations and emissions from the MB barn. In August and early September, the PM_{10} concentration decreased dramatically from a 10-d average of 597±85 $\mu g/dsm^3$ to a relatively stable state with an average concentration of 239±63 $\mu g/dsm^3$. The overall average PM_{10} concentration was 265±108 $\mu g/dsm^3$. The initial high concentration was mainly due to new flock in the barn. New flocks are generally very active when adapting to a new environment. Such highly active flocks contribute to high PM emission and concentration. Typically it takes about six weeks for a new flock to adapt to new cage environment.

PM_{10} concentration of the MB barn was about 45%-53% that of one HR barns reported in Lim et al. (2005), which affiliated with the same poultry company, housed similar number of hens, and had standard diet. The overall six-month average PM_{10} concentrations of the HR barns were 565±195 and 500±153 $\mu g/dsm^3$ for the HR barn1 and 2, respectively, which agree well with the annual average PM_{10} concentrations,518±74 $\mu g/dsm^3$, of a HR barn in Indiana (Lim et al, 2003),

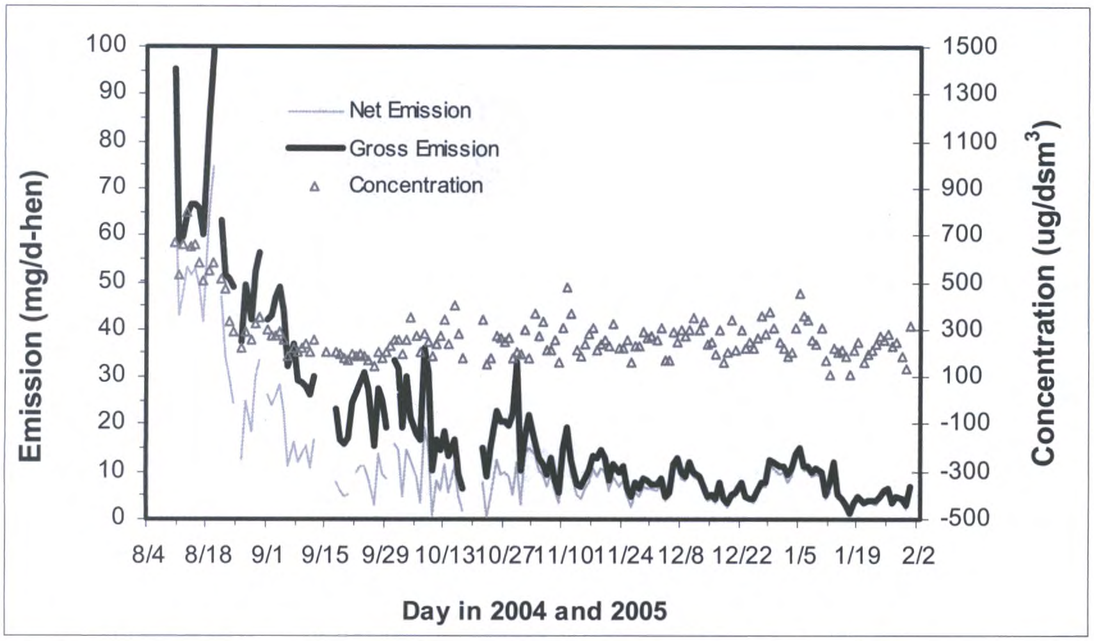


Figure 4. Daily mean particulate matter (PM_{10}) concentrations and emission rates of the Belt-Battery barn

The ambient PM_{10} concentration was used to calculate net PM_{10} emission. Ambient PM_{10} concentration ranged from 0 to 99% of PM_{10} concentration at the barn exhaust fan with an average of 31±0.2%. In the first three months (hot days), the ambient PM_{10} accounted for 47% of the exhaust concentration. In the second three months (cold days), it accounted for 18% of the exhaust concentration. Re-entrainment of exhausted particles into the poultry barn contributed to an approximately 30% increase of the PM_{10} emission on average in the six-month period. In hot days, because of high ventilation rate and relatively small separation distance between layer barns, the re-entrainment was about 47% of PM_{10} emission. Therefore, the gross PM_{10} emission rates could include a significant fraction of PM_{10} that was re-entrained into the barn.

The overall six-month average PM_{10} emission rate was 20±18.9 mg/d-hen. However, the PM_{10} emission rate was as high as 95 mg/d-hen in the beginning of the test and decreased dramatically in the first few weeks to an average of 19.8±8 mg/d-hen in October. As the weather got cold and barn ventilation rate decreased, the PM_{10} emission rate fluctuated and then slowly decreased. The lowest PM_{10} emission rate in January averaged 7 ±3.6mg/d-hen, which was 7% of the initial PM10 emission rate and 27% of PM10 emission rate in late August.

The MB barns and the two HR barns (Lim et al., 2005) had very similar decreasing trends in PM10 emissions as the weather got cold and ventilation rates decreased. The MB barn initially had a much higher PM_{10} emission due to the new flock at the beginning of the study. However, the HR barns had a much higher overall PM_{10} emission. The overall average PM10 emissions were 30±13.4 and 35±33 mg/d-hen for the HR barn 1 and 2 (Lim et al. 2005) respectively. An annual average PM10 emissions,16±3.4 g/d-AU, which is about 51 mg/d-hen if an average of 3.2 lbs of hen weight were assumed, from an Indiana HR barn was reported by Lim et al. (2003). From this limited study, it is found that the MB barn had significantly lower PM10 concentration and emission than the HR barns.

It is interesting to observe that PM_{10} concentration in the MB barn was not affected by the decreased building ventilation rate, in contrast with PM_{10} concentration in the HR barns which increased slowly as ventilation rates decreased and the weather cooled down. However, because emission rate is a product of ventilation rate and concentration, PM_{10} emission rates of layer barns were strongly affected by ventilation rate. Thus, PM10 emission rates of the layer barns decreased as the ambient temperature decreased.

TSP Concentration and Emission

Figure 5 shows mean TSP concentrations and emission rates from the MB barn 1. At the exhaust of the MB barn, TSP concentration increased as ventilation rate decreased due to lower outdoor temperature. The concentration fluctuated day by day, but overall increased from 1767 to 4247 μg/dsm^3 with a six-month average of 3070±675 μg/dsm^3. The HR barn and MB barn had TSP concentrations very similar to the HR barn 1, which had an average TSP concentration of 2,795±745 μg/dsm^3 (Lim et al., 2005). Annual average TSP concentration of an Indiana layer barn was 1887±563 μg/dsm^3 (Lim et al. 2003), which is lower that of the Ohio layer barns. Indoor environment, management practices, building ventilation, diet, and hen activity will likely affect PM emission and concentrations.

The TSP emission decreased dramatically in September and October and then reached a relatively stable state in the colder months. The average TSP emission rates were 272±96 mg/d-hen in September and October and 93±24 mg/d-hen in the colder months. Both the HR barn and the MB barn barns had decreasing trends in TSP emission as outdoor temperature decreased. However, TSP emission in the MB barn had a high initial value of 500 mg/d-hen and a relative steep decrease while the HR barn had an initial high value of 359 μg/dsm^3 and a relatively slow decrease. It is very likely that the new flock contributed to the high initial TSP concentration and emission. Ventilation rates were also strongly associated with TSP emission in the layer barns. In cold months (Nov., Jan. and Feb.), TSP emissions were stable for both types of barns with an average TSP emission of 93±24 and 103±31 mg/d-hen from the MB barn and the HR barn, respectively. The overall mean TSP emission rate was 168±110 and 146±96.6 mg/d-hen from the MB barn and the HR barn, respectively. Statistical analysis of variance showed that TSP emissions from the two types of barns in cold months were not statistically different (P=0.4485). A paired t-test showed that TSP emissions in the warm months also showed no statistical difference (P=0.395). The annual average TSP emission from the Indiana layer barn was 63±15 g/d-AU (Lim et al. 2003), which is about 201mg/d-hen and higher than that of the Ohio layer barns.

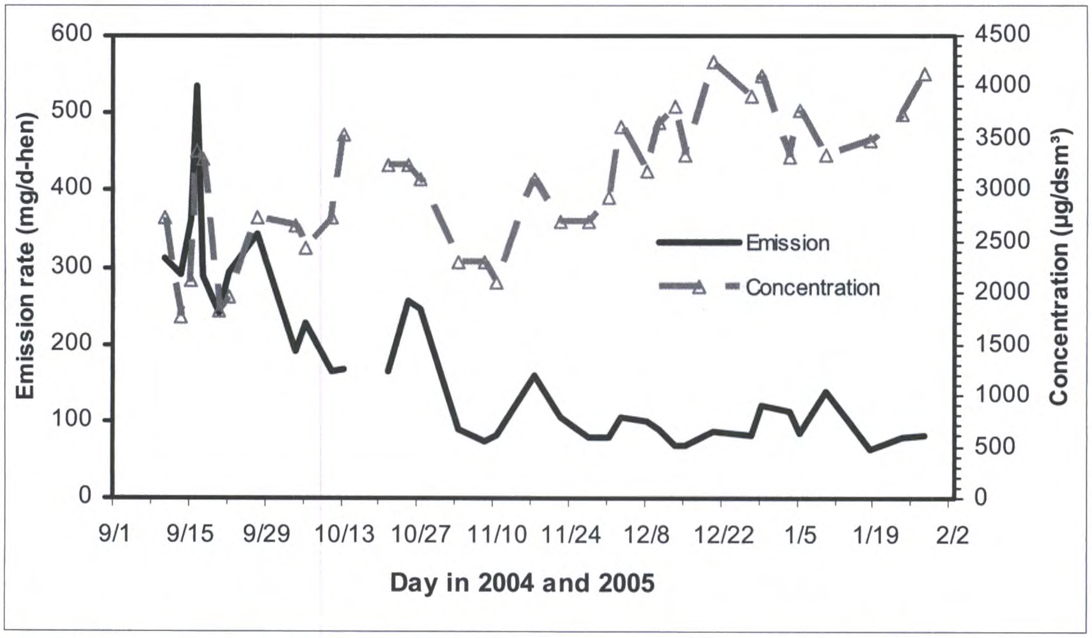


Figure 5. Mean total suspended particles (TSP) concentrations and emissions

The ratios of PM_{10} to TSP were 5-16% with an average of 9±3% in the MB barn and 14-41% with an average of 25±5% in the HR barn.

According to the emission rates, it would take about 4.1 million hens in the MB barn or 4.7 million hens in the HR barn to emit 250 tons of TSP per year, which is the emission threshold regulated by the CAA Prevention of Significant Deterioration (PSD) permit program. It would take about 13.7 million hens in the MB barn or 8.5 million hens in the HR barns to emit 100 tons of PM_{10} per year, which is the emission threshold regulated by the CAA Title V permit.

Diurnal Variations in PM_{10} Concentrations and Emission Rates

Figure 6 shows daily variations in PM10 mass concentrations at an exhaust of the MB layer barn. The continuous measurement results clearly indicated effects of light control on indoor PM10 concentrations. Each day at 4:00 a.m. when the lights were turned on, PM10 concentrations increased significantly. Each day when the lights were turned off at 8:00 p.m., the indoor PM10 concentrations significantly dropped. At the end of the day, before the lights were turned off, dust concentration reached its daily peak. These results agree with previous studies (Carpenter, 1986; Maghirang et al., 1991). Statistical analysis shows that dust concentrations during light hours were statistically different from those during dark hours ($P<0.05$).

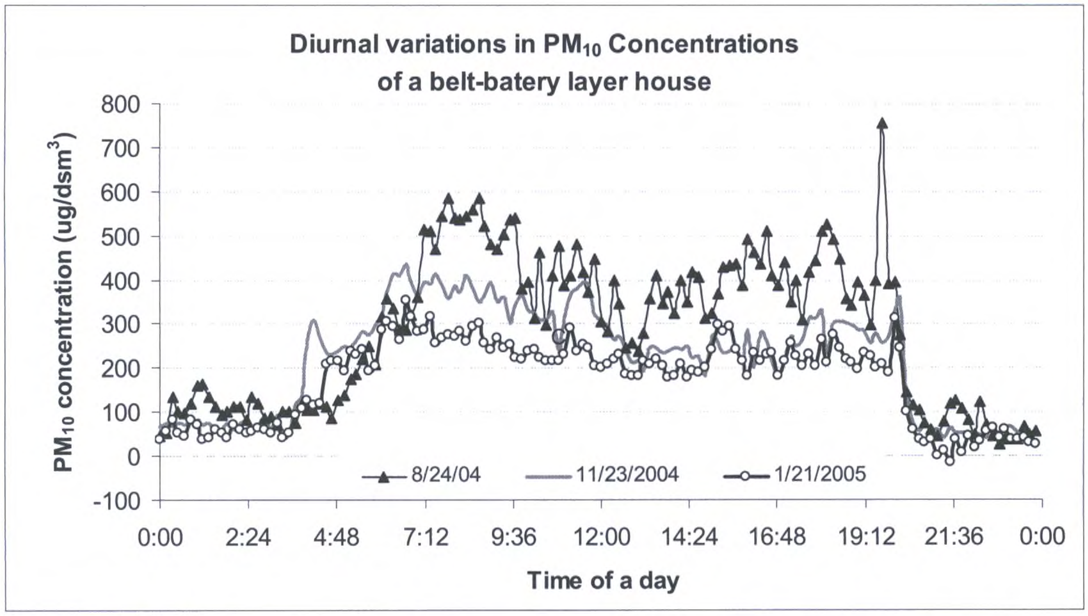


Figure 6. Diurnal variations in PM10 concentrations at an exhaust fan of the manure belt-battery layer house

Figure 7 shows daily variations in PM10 emission rates of the MB layer barn. The 10-minute average emission rates within 24-hour periods show strong effects of light control on PM10 emission rates in August and weaker effects in November. There is no clear diurnal pattern in January due to minimum ventilation. PM10 emission rates increased significantly when the lights were turned on at 4:00 am and dropped significantly as the lights were turned off at 8:00 pm in August. In January, when minimum ventilation was in operation, there was no clear diurnal trend in PM10 emission rates. The diurnal emission rate patterns agree with the concentration diurnal patters very well on the day in August, but not on the January day. Statistical analysis shows that dust PM10 emission rates during light hours were statistically different from rates during dark hours on the days in August and November ($P<0.05$), but not on the day in January.

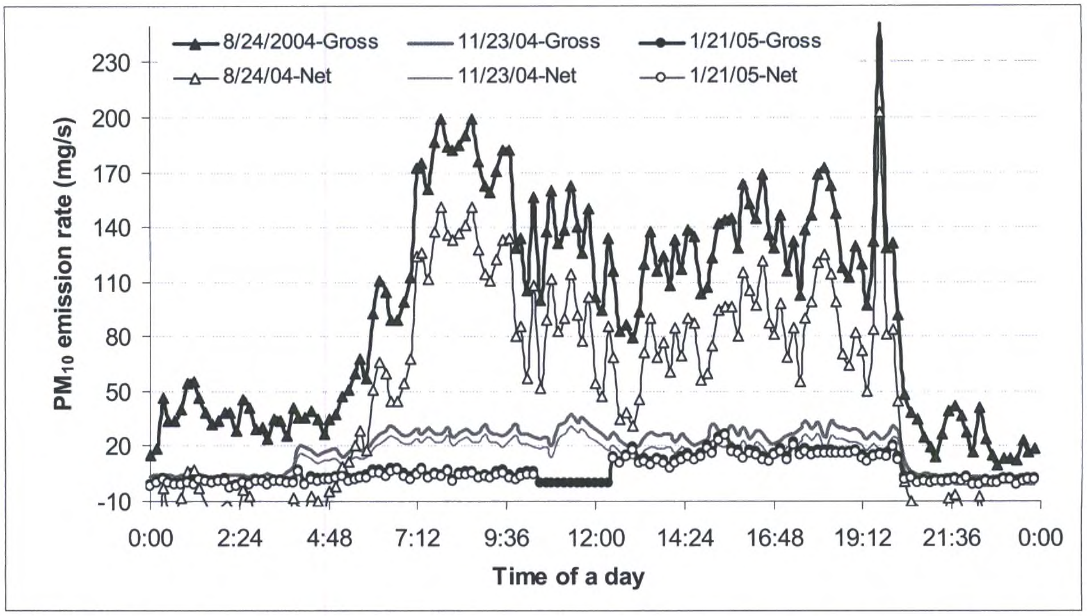


Figure 7. Diurnal variations in PM10 emission rates of a manure belt-battery layer house

Figure 8 shows an example of daily variations in barn airflow, PM_{10} concentrations, and PM_{10} emission rates. It confirms the dominant light control effects on PM10 indoor concentration and emission rates.

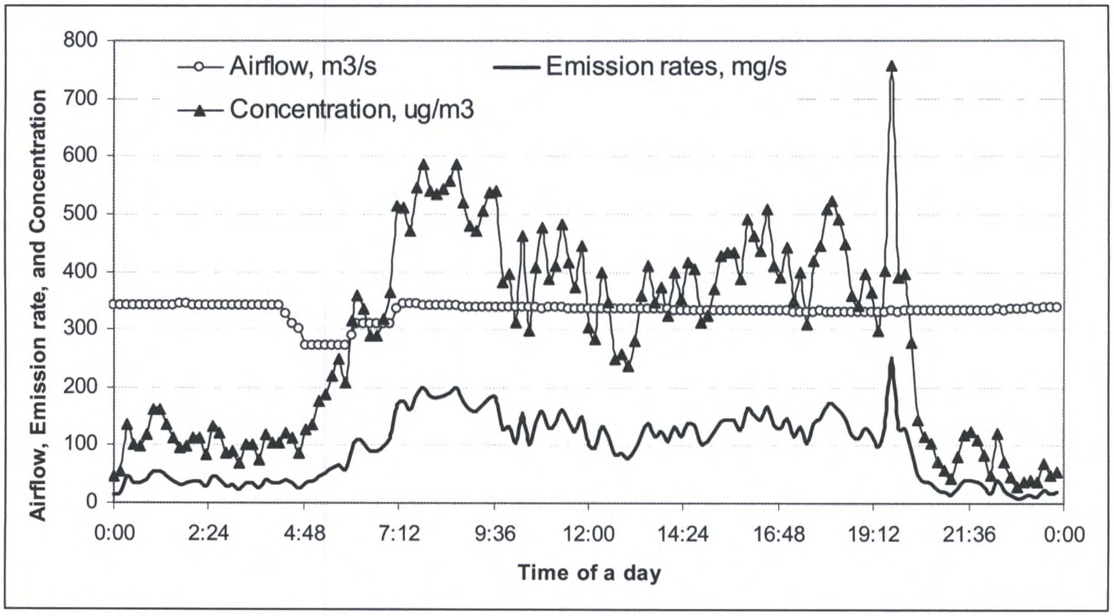


Figure 8. Example daily variations in barn airflow, PM10 concentrations, and PM10 emission rates

Conclusions

The overall average PM_{10} concentration of the MB barn was 265±108 μg/dsm^3. The overall average PM_{10} emissions were 20±18.9 mg/d-hen.

The TSP concentration of the MB barn fluctuated day by day, but overall increased from 1643 to 4025 μg/dsm^3 with a six-month average of 3070±675 μg/dsm^3.

TSP emissions of both MB and HR barns decreased as outdoor temperature decreased. However, in cold months, TSP emissions were stable for both types of barns, with average TSP emission rates of 93±24 and 103±31 mg/d-hen from the MB barn and the HR barn, respectively. The overall mean TSP emission rates were 168±110 and 146±97 mg/d-hen from the MB barn and HR barn 1, respectively. Statistical analysis of variance showed that TSP emissions from the two barns in cold months were not statistically different ($P=0.4485>0.05$). A paired t-test showed that TSP emissions in the warm months were not statistically different either ($P=0.395$).

According to the emission rates, it would take about 4.1 million hens in the MB barn or 4.7 million hens in the HR barn to emit 250 tons of TSP per year. It would take about 13.7 million hens in the MB barn or 8.5 million hens in the HR barns to emit 100 tons of PM_{10} per year.

New hen flock contributed to higher TSP and PM10 concentrations and emissions from the MB barn. High ventilation rates were also strongly associated with high PM10 and TSP emissions in the layer barns.

During high ventilation rate periods, there are strong diurnal patterns of PM10 concentrations and emission rates associated with the light control schedule. PM10 concentrations and emission rates during light hours of high ventilation rate periods are significantly higher ($P<0.05$) than during dark hours of high ventilation rate periods. There is no clear diurnal pattern in PM10 concentrations and emission rates during minimum ventilation periods.

References

Carpenter, G.A. 1998. Dust in livestock buildings – review of some aspects. *J. Agric. Eng. Res.* 33:227-241.

EPA. 2005. The U.S. EPA air compliance consent agreement with animal feeding operations. http://www.epa.gov/compliance/resources/agreements/caa/cafo-agr-050121.pdf. Last accessed July 28, 2005.

Heber, A.J., T.-T. Lim; J.Z. Gallien; J.-Q. Ni; P.C. Tao, A.M. Millmier, L.D. Jacobson, J.A. Koziel, S.J. Hoff, Y. Zhang, and G.B. Baughman. 2002. Quality assured measurements of animal building emissions: Part 2. Particulate matter concentrations. *Symposium on Air Quality Measurement Methods and Technology*, San Francisco, CA. Nov. 13-25, Air and Waste Management Assoc.: Pittsburgh, PA.

Heber, A.J. 2004. Measurement and control of ammonia, dust and odor emissions from layer houses. *Multi-State Poultry Meeting*, Indianapolis, IN, May 25-27

Heber, A.J., T.T. Lim, J.-Q. Ni, P.-C. Tao, A.M. Schmidt, J.A. Koziel, S.A. Hoff, L.D. Jacobson, Y. Zhang, and G.B. Baughman. 2005. Quality assured measurements of animal building emissions: Part 2. Particulate matter concentrations. *Journal of Air and Waste Management,* accepted pending revision.

Jerez, S., Y. Zhang, J. McClure, L.D. Jacobson, A.J. Heber, J. Koziel, S.J. Hoff, and D.B. Beasley. (*in press*). Comparison of total particulate emission using tapered element oscillating microbalance and a TSP sampler. *AWMA Journal.*

Liang, Y., H. Xin, E.G. Wheeler, R.S. Gates, H. Li, J.S. Zajaczkowski, P. Topper, K.D. Casey and F.J. Zajaczkowski. 2004, Ammonia emission for US poultry houses: layer hens. ASAE Paper No. 044104. St. Joseph, Mich.: ASAE.

Lim, T.-T., A.J. Heber, J.-Q. Ni, J.Z. Gallien, and H. Xin. 2003. Air quality measurements at a laying hen house: particulate matter concentrations and emissions. *Air Pollution from Agricultural Operations*, Durham, NC, USA, Octber 12-15., p.249-256.

Lim, T.T., H. Sun, J.–Q. Ni, L. Y. Zhao, C. Diehl, A.J. Heber, and P.C. Tao. 2005. Effect of flow-through particulate impaction curtain on PM emissions from a high-rise layer barn. ASAE paper 054013. St. Joseph, Mich.: ASAE.

Maghirang, R.G., H.B. Manbeck, W.B. Roush, and F.V. Muir. 1991. Air contaminant distributions in a commercial laying house. *Trans. ASAE* 34: 2171-2180.

Ni, J.–Q, A.J. Heber, H. Sun, S.M. Hanni, C. Diehl, T.T. Lim, L.Y. Zhao, and M. Darr. 2005. Application of vibration sensors for fan monitoring to improve measurement quality at a large layer barn. ASAE paper 054013. St. Joseph, Mich.: ASAE.

Ji-Qin Ni1, Technical Director; Albert J. Heber1, Professor; Hua-Wei Sun2, Research Associate; Sam M. Hanni1, Research Assistant; Claude A. Diehl1, Research Engineer; Teng T. Lim1, Research Scientist; Lingying Zhao2, Assistant Professor; Matt Darr2, Research Associate

NRC. 2003. *Air emissions from animal feeding operations: current knowledge, future needs.* Washington, DC: The National Research Council of the National Academies, The National Academies Press.

Sun, H, T.T. Lim, J.–Q. Ni, A.J. Heber, L.Y. Zhao, C.A. Diehl, P.C. Tao, and S. Hanni. 2005. Ammonia emissions from a belt-battery layer house. ASAE paper 054166. St. Joseph, Mich.: ASAE.

Takai, H., S. Pedersen, J.O. Johnsen, J.H.M. Metz, P.W.G.G. Koerkamp, G.H. Uenk, V.R. Phillips, M.R. Holden, R.W. Sneath, J.L. Short, R.P. White, J. Hartung, J. Seedorf, M. Schroder, K.H. Linkert, and C.M. Wathes. 1998. Concentrations and emissions of airborne dust in livestock buildings in Northern Europe. *J. Agri. Engng Res.* 70(1): 59-77.

Xin, H. 2005. Ammonia emissions & potential mitigation options for U.S. poultry facilities. Presentation at the North Atlantic Poultry Health and Management Conference, May 10. http://www.abe.iastate.edu/Xin/Atlantic-talk-5-10-2005-Xin.pdf.

Wang, X., Y. Zhang, L. Y. Zhao, and G. L. Riskowski. 1999. Development of a multi-point aerosol sampler using critical flow control devices. *Transactions of ASHRAE* 105(2): SE-99-17-03.

Optimization of Air Sampling Strategies for Monitoring Ammonia Emissions from Poultry Layer Facilities

Lingying Zhao[1], Albert J. Heber[2], Teng T. Lim[2], J.-Q. Ni[2], and P.C. Tao[2]
[1]Ohio State University, Columbus, OH 43210, USA
[2]Purdue University, West Lafayette, IN 49707, USA

Abstract

Ammonia from concentrated poultry layer houses result in significant environmental and health concerns. The emission data are highly needed by government agency, producers, and researchers to regulate and mitigate the emissions from poultry facilities. Since the complex spatial and temporal variations of the ammonia emission problem due to many affecting factors, accurately estimating and monitoring ammonia emissions from poultry laying houses have been challenging. The Purdue mobile lab method for air emission measurement, which continuously monitoring air emissions from poultry facilities, can provide relatively complete and accurate ammonia emission information, but it involves high cost and long monitoring time. Sufficient air sampling rates for monitoring ammonia emissions are needed information for wide application of this methodology and other air monitoring methodologies. This study analyzed ammonia emission data of several 6-month continuous, intensive measurements of large egg laying houses using the Purdue method with two sets of ammonia gas analyzers, that one switched between nine exhaust locations and the other kept at one location. Analysis of variance was used to evaluate spatial and temporal variations of ammonia emissions. Factors associated with spatial and temporal variation of air emissions were evaluated with sensitivity analysis. The study results showed that spatial variation of ammonia emission in a conventional high-rise layer facility is relatively small. Air sampling periods should be at least two weeks for each major seasonal weather condition. The optimum length of time for determining semi-annual emission factors is six weeks, such as two weeks in January, March, and May, respectively.

Predicting NH_3 Emissions from Manure N for Livestock Facilities and Storages: A Modified Mass Balance Approach.

H. M. Keener[1] and L. Zhao

Abstract

NH_3 levels and resulting emissions during the handling of manure at animal-production facilities have significant health, safety, odor-generation, and environmental impacts. In addition, NH_3 levels are impacting the profitability and growth of the livestock industry and forcing adoption of new technology. Determining NH_3 emissions is costly using the current approach of measuring ammonia concentrations and airflow from a facility. This paper looks at using N-balances to determine the upper limit on NH_3-N emissions for both forced and naturally ventilated livestock facilities. The analysis uses a controlled volume approach for inputs and outputs from the system and N/ash ratios. Considered in the analysis are body growth, milk and egg production, mortality and leachate as materials leaving the system. However, this method does not distinguished for losses of N as N_2 or NO_x's. Although the method can not predict daily cycles in emissions or maximum concentrations, it can provide an accurate estimate of the maximum theoretical levels for NH_3-N emissions over long production cycles (several days to months) if sampling gives good precision. As such, results are to be interpreted as daily, weekly or yearly averages. It does not require measuring total masses of materials into and from the system, but is dependent on obtaining adequate, representative samples of feed and manure at entry/exit points of the housing and storage systems. Generalized equations for predicting emissions from all classes of livestock operations are presented. Research using this N-balance method was simple, low cost, and accurate (based on results and reported literature values) in predicting upper limits on NH_3 emissions in the air leaving a 1.6 million caged layer poultry facility using two types of manure management: belt/composting and deep pit. Specific results based on the study showed clear advantages of belt/composting over conventional deep-pit systems, with total emissions less than half that of the conventional caged layer systems.

Keywords: ammonia, emissions ,livestock environment, poultry manure, nitrogen balance, odor.

Background

During the past decade, emissions from concentrated feeding livestock facilities have become a significant environmental issue as it relates to odors and greenhouse gases, in particular NH_3. Already, in Western Europe, legislation is in place to require the reduction of NH_3 emissions by up to 50% (Groot Koerkamp, 1994). This issue is a major constraint to the profitability and growth of livestock industries and mandate new approaches to livestock housing and manure management. Evaluating the effects of these new methods on emissions from livestock buildings for full scale operations can be quite expensive and labor intensive using current methodologies of continuous monitoring of airflows and emission levels (Heber et al., 2001; NRC 2003).

The goal of this study was to develop the theory for a modified mass balance approach to predicting NH_3-N emissions for livestock operations and apply this method to a 1.6 million bird poultry facility in Ohio using two types of manure management: belt/composting and deep pit.

Theory

N balances for animal production system enable predicting upper limits on ammonia emissions . Figure 1 is a schematic of an animal production system showing inputs and outputs, generalized for the case of body growth, milk and egg production. It also includes mortality and leachate as materials leaving the system. Analysis of this production system for NH_3-N assumes no other gaseous losses of N.

---

[1] The authors are Harold M. Keener, professor, and Lingying Zhao, assistant professor, Department of Food, Agricultural, and Biological Engineering, Ohio Agricultural Research and Development Center, Ohio State University. Corresponding author: Harold M. Keener. e-mail: keener.3@osu.edu.

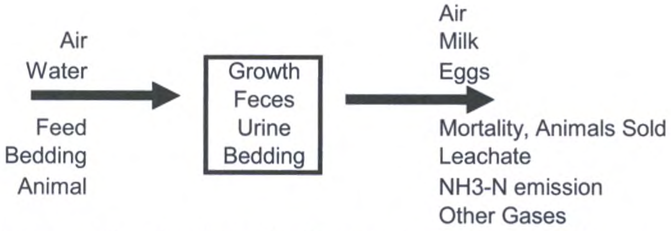


Figure 1. Flow diagram of animal production system showing inputs, storage and output variables.

To simplified the presentation of equations used in the analysis the notations presented in Table 1 were adopted. The nitrogen balance for the system based on ($N_{storage} = N_{in} - N_{out}$) is given by

$$\sum_{i=6}^{9} x_{N_i}\frac{dm_i}{d\theta} \cong \sum_{i=1}^{5} x_{N_i}m_i' - \sum_{i=10}^{14} x_{N_i}m_i' - N_{15} \qquad (1)$$

For an ash balance on the system the equation is

$$\sum_{i=6}^{9} x_{A_i}\frac{dm_i}{d\theta} = \sum_{i=1}^{5} x_{A_i}m_i' - \sum_{i=10}^{14} x_{A_i}m_i' \qquad (2)$$

Table 1. Variables and nomenclature used in mass balance equations for prediction NH_3-N emissions from livestock facilities.

Variables

θ = time, day

m_i = mass of i, kg

mi' = mass flow rate, kg/day

dmi/dθ = rate of change, kg/day

x_{Ni} = nitrogen content, dec

x_{Ai} = ash content, dec.

Ri - x_{Ni}/x_{Ai}, nitrogen to ash ratio, dec

Ni = total nitrogen in i, kg/day

Subscript

i = ...

1, air in

2, water in

3, feed in

4, bedding in

5, animals in

Subscript

i =

6, growth of animals in system

7, feces in system

8, urine in system

9, bedding in system

10, air out of system

11, milk out of system

12, eggs out of system

13, mortality, animals sold out of system

....14, leachate

....15, NH3-N emission

A, ash

N, nitrogen

M, manure

Solving equation 1 for N_{15} (the nitrogen emitted as NH_3-N gives

$$N_{15} \le \sum_{i=1}^{5} x_{N_i}m_i' - \sum_{i=6}^{9} x_{N_i}\frac{dm_i}{d\theta} - \sum_{i=10}^{14} x_{N_i}m_i' \qquad (3)$$

$$= \sum_{i=3}^{5} x_{N_i}m_i' - x_{N6}\frac{dm_6}{d\theta} - \sum_{i=7}^{9} x_{N_i}\frac{dm_i}{d\theta} - \sum_{i=11}^{14} x_{N_i}m_i'$$

The final form of (3) assumes N in the air entering the system passes through the system and no N is in the water entering the system. The term $\sum_{i=7}^{9} x_{N_i}\frac{dm_i}{d\theta}$ is the total nitrogen in the manure.

Using equation 2, the total ash in the manure is given by

$$\sum_{i=7}^{9} x_{A_i}\frac{dm_i}{d\theta} = \sum_{i=3}^{5} x_{A_i}m_i' - x_{A6}\frac{dm_6}{d\theta} - \sum_{i=11}^{14} x_{A_i}m_i' \tag{4}$$

Now from sampling the manure and laboratory analysis the manure N/A ratio, R_M, can be evaluate. But it is also true that

$$R_M = \sum_{i=7}^{9} x_{N_i}\frac{dm_i}{d\theta} \Big/ \sum_{i=7}^{9} x_{A_i}\frac{dm_i}{d\theta} \tag{5}$$

Substituting (5) and (4) into (3) gives

$$N_{15} \le \sum_{i=3}^{5} x_{N_i}m_i' - x_{N6}\frac{dm_6}{d\theta} - \sum_{i=11}^{14} x_{N_i}m_i' - R_m\left[\sum_{i=3}^{5} x_{A_i}m_i' - x_{A6}\frac{dm_6}{d\theta} - \sum_{i=11}^{14} x_{A_i}m_i'\right] \tag{6}$$

The advantage of (6) over (1) for doing N mass balance and estimating NH_3-N losses is no weighting of the manure is required. A second advantage of (6) compared to the conventional method of continuous sampling of airflow rates and NH_3-N concentration for exhausting air is avoidance of the high equipment cost, maintenance and labor to operate continuous monitoring equipment. However, this method does not distinguished for losses of N as N_2 or NO_x's. Also, it would not predict daily cycles in emissions or maximum concentrations. It can however, provide an accurate estimate of the maximum theoretical levels for NH_3-N emissions over long production cycles (several days to months) if sampling gives good precision. As such, results are to be interpreted as daily, weekly or yearly averages.

Eq 6 can be used to predict the upper limit on NH_3-N emissions for open and/or naturally ventilated building. Such structures do not lend themselves to the more common method of measuring airflows and NH_3 concentrations leaving a building. For example, for a concrete free stall dairy barn with lactating cows, terms containing 5, 6, 12, 13, and 14 would probably be zero. For a deep pit swine finishing house, the terms containing 4, 5, 9, 11, 12, and 14 would probably be zero. Data for the non-zero terms would be based on time periods conductive to accurate sampling.

Keener et al., (2002) used eq. 6 for analysis of two types of caged layer facilities. Information from that paper are presented here to illustrate how the method can be applied. In that study, results using the modified mass balance approach were compared with measured emissions based on airflows and exhaust NH_3-N concentrations and with published literature. For analysis purposes, it was assumed the terms $\frac{dm_6}{d\theta}$, m_4', m_5', m_{11}', m_{13}' and m_{14}' were $\cong$ zero. Because the birds were mature, the assumption on $\frac{dm_6}{d\theta}$ (i.e. changes in their body composition, storage or release of ash and nitrogen) could be justified. Also, no bedding was used, no animals were placed, no milk was produced, mortality was very low, and no leachate was observed during the analysis periods. The equation to evaluate N_{15} becomes

$$N_{15} \le x_{N_3}m_3' - x_{N_{12}}m_{12}' - R_m\left[x_{A_3}m_3' - x_{A_{12}}m_{12}'\right] \tag{6b}$$

For yearly emissions per bird, eq. 6b becomes

$$EM_{NH3\text{-}N} = 365\ N_{15}/n_b \quad (\text{kg } NH_3\text{-N bird}^{-1}\ \text{yr}^{-1}), \tag{7}$$

where n_b was the number of birds in the building.

Experimental Facilities

The Mad River facility of the Daylay Egg Farm, located near West Mansfield, Ohio, is a modern, 1.6 million bird, caged layer facility for the production of chicken eggs (fig. 2). The birds are housed in a total of eight buildings. Four buildings are of the deep-pit design and house 150,000 birds each. Manure is removed yearly from these buildings and directly land applied. The other four buildings were built in a 1997 expansion and house 250,000 birds each. They incorporate a belt conveyor system for manure removal. Manure on the belt is delivered to two separate buildings (12 lanes and 6 lanes, respectively) where it is composted, using Salmet composters, into a dry product suitable for fertilizer applications.

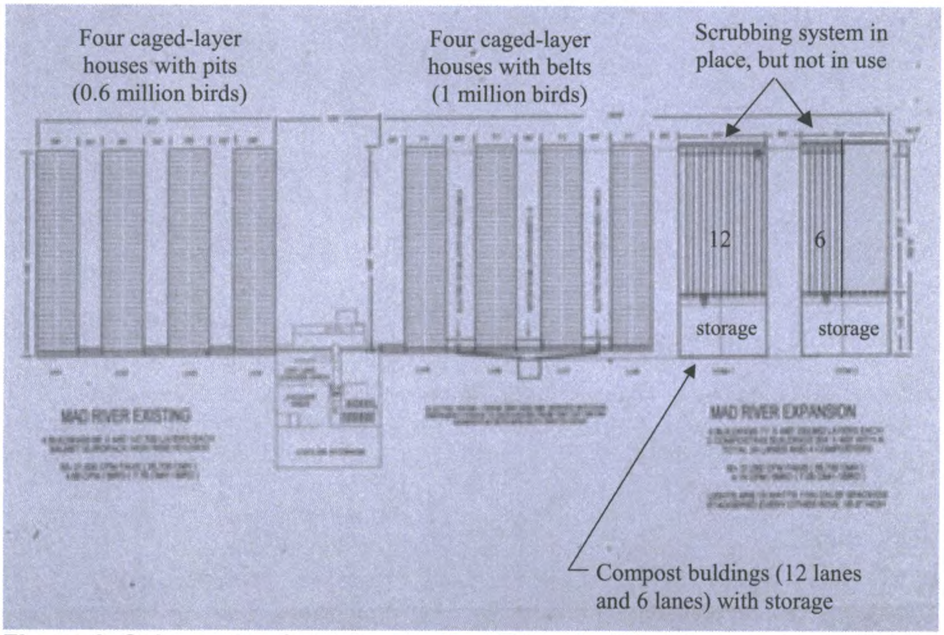


Figure 2. Schematic of the Daylay Mad River facility (Keener, et al., 2002).

All of the buildings have significant ventilation. The deep-pit caged layer houses each have 36 side-mounted hooded fans (18 per side) located in the pit area, which draw air down across the birds (maximum rate 0.13 m^3 min^{-1} $bird^{-1}$, 4.7 cfm $bird^{-1}$) and out. The new layer buildings are vented through the roof using a bank of nine large computer-controlled fans on each house (0.119 m^3 min^{-1} $bird^{-1}$, 4.2 cfm $bird^{-1}$). Inlets are located along the entire sidewall length. The composting buildings are vented at one end using eight fans on one building and four fans on the other, each with a capacity of 566 m^3 min^{-1} (20,000 cfm).

Experimental Procedure

Sampling data on ammonia measurements were made on 22-23 March and 17-18 July 2000 in cooperation with the Wisconsin Department of Natural Resources (Keener, et al., 2002). Instantaneous ammonia concentration measurements were made from deep-pit caged layer houses 1 and 4 at the exit of the hooded exhaust fans using colorimetric stain tubes with ranges of 0.25 to 3 ppm, 2 to 30 ppm, or 5 to 70 ppm, ±15% error (National Dräger Inc., Pittsburgh, Pa.). These measurements were also made for belt caged layer house 8 at the entrance to the exhaust fans and for the two compost buildings at the exit from the shuttered fans. Airflows from fans were based on rated capacity and no verification of accuracy was attempted, as random operation of fans was a much greater source of error. Results from calculations using airflows were used only as a gauge of the range for NH_3 emissions and not a prediction of absolute amounts lost.

On day 2 of each test, manure and compost samples were collected. Eight manure samples were collected from the four deep-pit caged layer houses, two from each house. Each sample was a composite of grab samples from the top or the middle of the manure windrows at four or five different locations. For the four caged layer houses using manure belts, five manure samples were collected randomly over a 30-minute interval from the cross conveyor belt transferring manure to the compost building. For the compost buildings, five samples of finished compost were collected, one each from lanes 2, 6, and 10 in building 1 and from lanes 2 and 4 in building 2. All samples were sent to the OARDC/OSU analytical laboratory for analysis of pH, solids, ash, total carbon, inorganic carbon, NH_3-N, NO_3-N, and total N. The feed was analyzed for moisture, ash, and nitrogen. Values used for egg composition were estimated from general literature. Solution of the mass balance made use of the fact that feeding rate was known on a per bird basis and that ash per bird could be defined at all points in the system.

Results

Table 2 gives daily feed rate, daily egg production rate along with chemical composition of feed, eggs, deep-pit manure (buildings 1-4), belt manure (buildings 5-8) and compost (compost buildings). The deep-pit manure had an N/ash content of 0.06072 and 0.09143 for March and July, respectively. Belt manure was 1 to 3 days old and had an N/ash content of 0.18705 in March and 0.19096 in July. The compost had a N/ash content of 0.16729 for March and 0.16373 for July.

Table 2. Mean (μ) and standard deviation (sd) of feed rate, producion level and chemical analysis of feed, egg, manures, and compost used in nitrogen balances for Daylay facilities on 22 March and 18 July 2000 (Keener et al., 2002)

Description		per Bird kg/day	DM (%)	Ash (%)	TC (%)	IC (%)	N (%)	NH_3-N (μg g^{-1})	NO_3-N (μg g^{-1})	Rm N/ash
Feed input	μ	0.09091	89.60	13.73			2.82			
	sd	0.00455	0.15	0.65			0.31			
Eggs	--	0.05227	34.10	10.00			2.05			
22 March 2000										
Deep-pit manure	μ		40.20	48.42	26.91	3.39	2.94	19038	81.1	0.06072
(buildings 1-4)	sd		10.05	9.09	3.83	0.67	0.81	11295	23.9	
Belt manure	μ		46.99	30.26	33.37	1.94	5.66	6810.9	232.4	0.18705
(buildings 5-8)	sd		6.65	1.96	1.06	0.31	0.99	2631.4	57.3	
Compost buildings	μ		82.02	37.18	31.11	2.14	6.22	8150.2	256.1	0.16729
	sd		0.97	1.80	1.07	0.23	0.24	763.3	38.7	
18 July 2000										
Deep-pit manure	μ		70.81	43.97	28.96	2.78	4.02	4782.9	93.8	0.09143
(buildings 1-4)	sd		12.19	5.08	1.76	0.44	0.77	1551.6	14.1	
Belt manure	μ		51.71	30.74	33.85	1.68	5.87	6566.3	211.9	0.19096
(buildings 5-8)	sd		8.07	3.41	2.25	0.38	0.48	2578.1	24.7	
Compost buildings	μ		90.20	34.08	32.28	1.92	5.58	4846.6	213.1	0.016373
	sd		0.91	1.32	1.11	0.14	0.57	1245.4	37.4	

Table 3 summarizes the nitrogen balances for two caged layer manure management systems based on N fed to the animal, N in the eggs produced, and N emitted by using equation 7. Results showed that the average N retained for the March and July studies was 0.556 kg $bird^{-1}$ yr^{-1} and 0.209 kg $bird^{-1}$ yr^{-1} for the belt/compost system compost and conventional deep-pit manure, respectively. This represented 81% and 30% of N excreted by the birds, respectively.

Table 3 also lists the nitrogen emissions results for March and July. Total nitrogen emission during March for a conventional caged layer deep-pit system (DP) was 0.477 kg (1.05 lb) N $bird^{-1}$ yr^{-1}. This would be 0.631 kg NH_3 $bird^{-1}$ yr^{-1} and is very similar to the value that Battye et al. (1994) reported of 0.598 kg NH_3 $bird^{-1}$ yr^{-1} for laying hens >6 months of age. Total nitrogen emission during March, based on compost N level, for the belt/compost caged layer system (CLB-CB) was calculated to be 0.129 kg (0.284 lb) N $bird^{-1}$ yr^{-1}. This would be 0.166 kg NH_3 $bird^{-1}$ yr^{-1} and is 25% of the value for the deep-pit system. For the DP and CLB-CB systems, the calculated NH_3-N loss using estimated airflow rates and measured ammonia concentrations in the exhaust air were 0.550 and 0.437 kg (1.213 and 0.963 lb) N $bird^{-1}$ yr^{-1}, respectively. Losses of NH_3 only from the layer building for the CLB-CB system, using a mass balance approach, were estimated at 0.057 kg $bird^{-1}$ yr^{-1}, compared to 0.034 kg $bird^{-1}$ yr^{-1} cited by Groot Koerkamp (1994). Results in March using airflows and NH_3 concentrations were very approximate because airflows were varying during the data collection period. For the conventional deep-pit caged layer system, the exhaust streams measured 5 to 60 ppm NH_3 v/v when all exhaust fans (32 per building) were running, while the belt caged layer system had 5 to 10 ppm NH_3 v/v (Keener et al., 2002).

Table 3. Nitrogen balance (kg $bird^{-1}$ yr^{-1}) for Daylay operation, March and July 2000.

System[a]	Measured Variable	Feed N	Egg N	N manure DP or CB	Total NH_3-N loss	NH_3-N loss CLB	NH_3-N loss CB
March							
DP	Manure Ash & N	0.823	0.133	0.213[b]	0.477		
DP	NH_3 loss	0.823	0.133		0.550		
CLB-CB	Manure & Compost Ash & N	0.823	0.133	0.561[b]	0.129	0.057	0.072[d]
CLB-CB	NH_3 loss	0.823	0.133		0.437[c]	0.179[c]	0.258[c,d]
July							
DP	Manure Ash & N	0.823	0.133	0.204[b]	0.486		
DP	NH_3 loss	0.823	0.133		0.513		
CLB-CB	Manure & Compost Ash & N	0.823	0.133	0.550[b]	0.140	0.041	0.099[d]
CLB-CB	NH_3 loss	0.823	0.133		0.225	0.039	0.186[d]

[a] DP = caged layer deep-pit manure storage, CLB = caged layer manure belt, and CB = compost building.

[b] Remainder term using (feed N - egg N - total emission N)

[c] Values believed to be high. Caged layer fans were being modulated. Compost building fans were not in full operation.

[d] Base on Total NH_3-N loss - NH_3-N loss CLB.

Total losses of NH_3-N, using manure or compost N levels, for the July test showed (table 3) similar values to the March study, with 0.486 kg (1.07 lb) N $bird^{-1}$ yr^{-1} for DP and 0.140 kg (0.308 lb) N $bird^{-1}$ yr^{-1} for the CLB-CB. Thus, NH_3 losses for the CLB-CB system were 29% of the deep-pit system in the July study. The NH_3 losses from the laying house in July for the CLB-CB system were estimated at 0.040 kg $bird^{-1}$ yr^{-1}, compared to 0.034 kg $bird^{-1}$ yr^{-1} cited by Groot Koerkamp (1994). These results show that the belt/compost system has a major advantage over the deep-pit system in terms of ammonia emissions. For the DP and CLB-CB systems, the calculated NH_3-N losses in July using airflow rates and ammonia concentrations in the exhaust air were 0.513 and 0.225 kg (1.129 and 0.495 lb) N $bird^{-1}$ yr^{-1}, respectively. Again, results using airflows and NH_3 concentrations were approximate because airflows were varying during the data collection period. For the conventional deep-pit caged layer system, the exhaust streams measured 7 to 25 ppm NH_3 v/v when all exhaust fans (32 per building) were running, while the belt caged layer system had 1 to 1.5 ppm NH_3 v/v (Keener et al, 2002).

Discussion

Results using nitrogen and ash levels for determining upper limits for NH_3 emissions for the caged layer facilities studied gave NH_3 emission values similar to those reported in the literature. The N/ash method was straightforward to implement and appears to be reasonably accurate. Values for July were similar to those for March, namely 0.129 and 0.140 kg NH_3-N $bird^{-1}$ yr^{-1} for CLB-CB and 0.477 and 0.486 kg NH_3-N $bird^{-1}$ yr^{-1} for the DP systems, respectively. Emission values calculated using airflow rates and gas concentrations were 20% to 250% higher and were subject to much greater uncertainty. The problem was that airflows were non-steady state in the March tests. With variable airflow rates, NH_3 levels in the pit increased with low airflow and decreased with high airflow and were dependent on previous airflow rates. This is because NH_3 emission from manure is affected by pit air and manure NH_3 levels, which in turn are affected by prior events. In July, airflow rates were essentially constant with all fans running at maximum output; thus, the resulting emission rates using airflow were much closer to the values based on manure nitrogen. The results clearly showed that the CLB-CB system would have ≅28% less emissions than the DP system. In addition, from the standpoint of N conservation, the CLB-CB compost retained over twice as much nitrogen as the DP manure while being a dry product at 82% to 90% dry matter and 5.6% to 6.2% N.

Conclusions

The method of using ash as a reference value in calculating NH_3 emissions for caged layer poultry was an accurate and straightforward way to determine the upper limits of emissions on a per bird basis. This approach is lower cost, easily implemented and can provide accurate estimates of the upper limits on NH_3 emissions than the current methods of quantifying airflow rates and NH_3 concentrations in exhaust streams.

Acknowledgements

This paper was adapted from the paper by Keener and Michel (2005) given at the Symposium on the State of Science of Animal Manure and Waste Management, San Antonio, TX. Salaries and research support are provided by the State and Federal funding appropriated to the Ohio Agricultural Research and Development Center, The Ohio State University. Use of a commercial trade name or company in this paper does not imply endorsement of the products but is included only to assist the reader.

References

Battye, R., W. Battye, C. Overcash, and S. Fudge. 1994. Development and selection of ammonia emission factors. Final report prepared for USEPA Contract Number 68-D3-0034. Washington, D.C.: U.S. EPA.

Groot Koerkamp, P. W. G. 1994. Review on emissions of ammonia from housing systems for laying hens in relation to sources, processes, building design, and manure handling. *J. Agric. Eng. Res*. 59(2): 73-87.

Heber, A.J., J.Q.Ni, B.L. Haymore, R.K. Duggirala, and K.M. Keener. 2001. Air quality and emission measurement methodology at swine finishing buildings. *Transactions of ASAE* 44(6):1765-1778.

Keener, H.M., D.L. Elwell, and D. Grande. 2002. NH3 emissions and N-balances for 1.6 million caged layer facility: manure belt/composting system vs deep pit operation. *Transactions of ASAE.* 45(6):1977-1984.

Keener, H.M. and F.C. Michel Jr. 2005. Predicting NH_3 emissions from manure N for caged layer facilities. A modified mass balance approach. Symposium on the State of Science of Animal Manure and Waste Management. 1/5-7. Marriott River Center, San Antonio, TX. Published on CD (search at: http://www.cals.ncsu.edu/waste_mgt/natlcenter/sanantonio/proceedings.htm)

NRC. 2003. Air Emissions from Animal Feeding Operations: Current Knowledge, Future Needs. The National Research Council of the National Academies. The National Academies Press, Washington, D.C.

Yang, P., J. C. Lorimor, and H. Xin. 2000. Nitrogen losses from laying hen manure in commercial high-rise layer facilities. *Trans. ASAE* 43(6): 1771-1780.

Ozone Damage to Crops in Southern Africa: An Initial Modeling Study

M. Zunckel[1], L.D. Emberson[2] and M. Sowden[1]
[1]CSIR, P O Box 17001, Congella 4013, South Africa
[2]Stockholm Environmental Institute, University of York, Heslington, York YO10 5DD

Abstract

The Cross Border Impact Assessment Project (CAPIA) was designed to develop an understanding of regional surface ozone concentrations and their potential risk to agriculture in southern Africa. Surface ozone concentrations were estimated using the Comprehensive Air Quality Model with extensions (CAMx). The initial assessment of ozone risk to maize was characterised using the Accumulated exposure Over a Threshold of 40 ppb (AOT40). Modelled ozone concentrations exceed 40 ppb over much of suothern Africa, suggesting that the potential for ozone damage to maize exists across the region. The AOT40 approach has limitations; the most notable being its inability to account for modifying factors that limit the amount of pollutnat taken up by the plant. The aim of this research is to investigate the feasibility of including the stomatal flux algorithms in the CAMx model, and so improve the estimates of ozone uptake in plants and the subsequent risk of ozone damage posed to crops. The initial model results indicate that the areas with elevated ozone concentrations are not the same as those with the highest ozone fluxes, suggesting that application of the more biologically relevant flux-based risk assessment methods would identify different regions within the modelling domain where damage to maize is more likely to occur. In addition, the algorithms in CAMx tend to underestimate both the deposition velocity and ozone flux in comparison to the flux method. Lastly, the maximum modelled total ozone fluxes are above the critical stomatal flux values of 6 nmol m^{-2} s^{-1} currently defined and applied within Europe to assess risk and economic impacts of ozone to agricultural crops.

Introduction

In many parts of the world surface ozone is considered to be the most prevalent and damaging air pollutant to which plants are exposed (Emberson *et al*, 2001a). Precursor emissions from multiple source types in the southern African region include those from biomass burning (Scholes *et al*, 1996), large and small industry and mining, transport ((Fleming and van der Merwe, 2002) and the combustion of wood and fossil fuels in domestic areas. In addition, natural emissions from biogenic sources have been shown to be significant (Greenberg *et al*, 2003; Harley *et al*, 2003). With high insolation and a dominant anticyclonic circulation that imposes long atmospheric residence times for the mixture of pollutants, an ideal environment exists in which ozone can form. Indeed, monitoring at Maun, a remote rural site in Botswana (Zunckel *et al*, 2004) has indicated that ozone concentrations in this remote area often exceed those typically experienced in urban environments. Despite this, surface ozone is monitored at only a few sites. With the exception of the Global Atmosphere Watch station at Cape Point where surface ozone concentrations have been logged since 1983 (Brunke and Scheel, 1998), the measurement records cover relatively short time periods. This dearth of ozone data implies that the understanding of surface ozone concentrations over the region is limited as is the understanding of potential impacts on vegetation.

Following suggestions by van Tienhoven *et al* (2005) that southern African vegetation may be at risk to damage by ozone, the Cross Border Air Pollution Impact Assessment (CAPIA) project was designed to develop an understanding of regional scale surface ozone concentrations, and to assess the potential risk of air pollution on agriculture. In the absence of comprehensive monitored ozone data, a modelling approach was required to meet the objectives of CAPIA. Using available data on anthropogenic and biogenic emissions and regional scale meteorology, ambient surface ozone concentrations were estimated using the Comprehensive Air Quality Model with Extensions (CAMx) (ENVIRON, 2003). The European approach known as the Accumulated exposure Over a Threshold of 40 ppb (AOT40) (Fuhrer *et al*. 1997) was used to assess the risk to maize in the CAPIA project. van Tienhoven *et al* (2006 identified that the AOT40 was exceeded over large areas of southern Africa and suggested that maize and other agricultural crops were indeed at risk to ozone damage.

The AOT40 is an approach that identifies the potential risk of ozone damage to vegetation based on the ambient concentration to which the plant is exposed (Fuhrer et al. 1997). This approach has some limitations; the most notable being its inability to account for modifying factors that limit the amount of pollutant actually taken up by the plant. These generally occur through modifications to stomatal conductance caused by local environmental conditions such as low humdities and high soil water stresses (Musselman and Massman, 1999). These factors are not considered by the AOT40 approach since damage is only related to the external pollutant concentration rather than the absorbed pollutant dose. The limitations of using only the AOT40 approach for the CAPIA risk assessments were recognised and resulted in the decision to use the recently developed "flux based" risk assessment method

This paper provides an overview of the theory and standard approach used in CAMx to calculate dry deposition of ozone. It also provides a theoretical discussion of the "flux modelling", emphasising the differences between the two methodologies. An initial comparison between ambient concentrations calculated using the standard CAMx dry deposition for ozone and the flux algorithms are presented.

Methods

Dry Deposition Modelling in CAMx

Analogous to an electrical circuit, the movement of aerosols or gases through a plant canopy and onto plant surfaces and the ground surface is typically modelled as a combination of resistances in series and parallel. Each branch of the circuit represents a different path by which material may be deposited. For example, pollutants may transfer to the sites of biological action within the leaves of the plant canopy through the stomatal openings to the mesophyll tissue. They may also deposit on the external surfaces of the plant canopy or move through the canopy and deposit directly on the ground surface. As ozone is a gas, this discussion considers deposition of gases only.

The factor that links the rate of dry deposition of a gas to the ambient concentration is the deposition velocity, where

$$F = -V_d * C \qquad \text{(Eq. 1)}$$

C is the ambient concentration of the gas, V_d is the deposition velocity and F is the deposition rate or flux. The negative notation indicates a downward flux. The ambient concentration is typically measured or modelled, the latter being the case within the CAPIA project. Wesley and Hicks (1977) and later Wesley (1989) developed a resistance model that incorporates the major resistances to deposition which may be described by the following equation:

$$V_d = \frac{1}{R_a + R_b + R_s} \qquad \text{(Eq. 2)}$$

R_a is the aerodynamic resistance and represents bulk transport between some reference height and the plant canopy. In the case of the CAMx modelling performed within the CAPIA project this height is 10 m above the ground surface. The pollutant transport within this part of the atmosphere results from turbulent diffusion. The magnitude of the R_a term depends on the intensity of turbulent motion, which in turn, depends on insolation, wind speed, surface roughness and the near-surface lapse temperature rate. As a result, R_a is a minimum on warm sunny days with strong mixing induced by surface heating and mechanical turbulence and a maximum on cool nights with calm winds and suppressed mixing.

In CAMx (ENVIRON, 2003) R_a is calculated from:

$$R_a = \frac{1}{ku_*}\left[\ln\left(\frac{z}{z_0}\right) - \Phi_h\right] \qquad \text{(Eq. 3)}$$

where

u_*	frictional velocity (m/s), which is a function of the landuse type which is an input requirement in CAMx.
k	von Karman constant.
z	reference height (10m)

z_0 roughness length, which is also a function of landuse.
Φ_h stability correction term.

R_b is the quasi-laminar sub-layer resistance that represents molecular diffusion through the thin layer of air that is directly in contact with the surface to which deposition takes place. It is mostly dependant on the molecular diffusivity of each pollutant species, which in turn is dependant on the friction velocity (u_*), von Karman's constant (k) and the Schmidt number (S_c) which is the ratio of air viscosity to molecular diffusivity of the chemical species.

$$R_b = \frac{2S_c^{2/3}}{ku_*} \qquad \text{(Eq. 4)}$$

The surface resistance, R_s, is expressed as a combination of serial and parallel resistances that depend on the physical and chemical characteristics of the surface in question. Over vegetated land surfaces R_s is given by the following equation:

$$R_s = \frac{1}{\dfrac{1}{r_{st} + r_m} + \dfrac{1}{r_{uc}} + \dfrac{1}{r_{dc} + r_{cl}} + \dfrac{1}{r_{ac} + r_{gs}}} \qquad \text{(Eq. 5)}$$

The first serial resistance represents the pathway into the stomatal and mesophyllic portions of the active plant. The second resistance represents the pathway into the upper canopy and the third is the pathway into the lower canopy. The fourth resistance is the pathway to the ground surface. Some of the resistances are dependant on season and landuse type which are included in the dry deposition model (Wesley, 1989), and so included in CAMx. Other resistances are adjusted in CAMx to account for variation in insolation, moisture stress and surface wetness.

The CAMx modelling domain is typically divided into user defined spatial grids. The underlying surface is gridded accordingly and variations in land use type across the modelling domain are captured in the model by the allocation of the dominant land use category to each grid block. The plant specific resistance algorithms described above are then scaled-up by applying them in each grid block.

The European Flux Model

In the European flux model, total ozone deposition velocity is also calculated using the 3-resistance formulation (Eq.2), similar to Wesley's (1986) method which is used in CAMx. The resistances include the aerodynamic resistance (R_a), the boundary layer resistance (R_b) and the surface resistance (R_{sur}). R_a and R_b are calculated using the same principles described in equations 3 and 4 respectively. R_{sur} comprises a plant canopy resistance and a resistance to the underlying soil similar to the Wesley (1989) approach. It is given by:

$$R_{sur} = \frac{1}{\dfrac{LAI}{R_{sto}} + \dfrac{LAI}{R_{ext}} + \dfrac{1}{R_{inc} + R_{soil}}} \qquad \text{(Eq. 6)}$$

where

R_{inc}	canopy aerodynamic resistance
R_{soil}	soil resistance
R_{ext}	external resistance
R_{sto}	land cover-specific stomatal resistance which is the resistance to ozone uptake through the stomata
LAI	leaf area index,

R_{ext} and R_{soil} are constants. R_{inc} is calculated with:

$$R_{inc} = \frac{(b * LAI * h)}{u_*} \qquad \text{(Eq. 7)}$$

where

b	an empirical constant taken as 14 m^{-1}
h	the vegetation height
u_*	friction velocity

A schematic of the various resistances and their configuration is presented in Figure 1.

One of the key differences between the CAMx and European deposition models is the approach to calculating stomatal resistance. To incorporate certain aspects of the European deposition model into the CAMx model two modules were developed. Module 1 is a canopy stomatal resistance module and Module 2 is a leaf stomatal flux module. The application of the former can be used to calculate total dry deposition to maize using the stomatal formulations that are unique to the European deposition model. This will allow comparisons of dry deposition to maize that are solely dependant upon the calculation of the stomatal component, considered a key component of total deposition during the growing season. The application of Module 2 would give an indication of the ozone uptake to upper canopy leaves of maize. Comparisons of the spatial pattern of cumulative ozone flux with AOT40 could then be used to assess the consequences of using concentration rather than flux based approaches to identify the risk posed to vegetation from surface ozone.

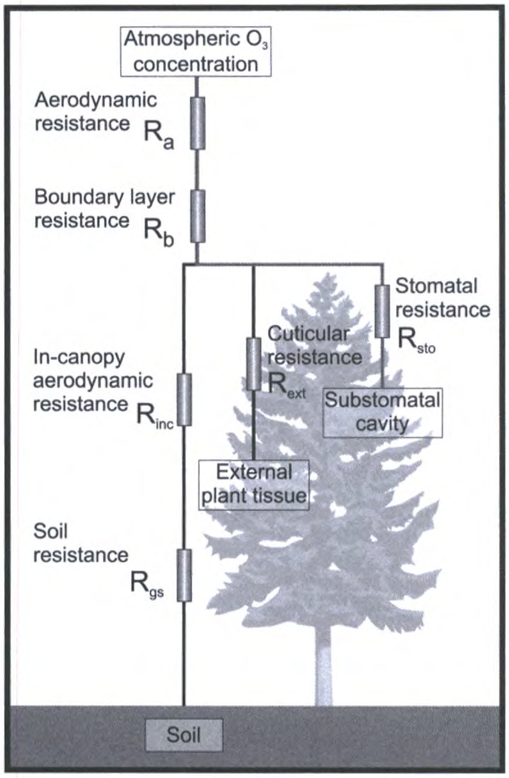


Figure 1: A schematic of the resistances to dry deposition of a pollutant gas (Emberson *et al.* 2001b)

Module 1 was developed to take the place of the existing stomatal resistance (defined by r_{st} in CAMx as in Eq. 8). The existing r_{st} is used to estimate surface resistance and is incorporated in the overall resistance scheme according to the parallel resistance approach of Wesley and Hicks (1977) and Wesley (1989). The stomatal resistance (r_{st}) is calculated with:

$$r_{st} = \text{diffract} * r_j * (1 + (200 / (\text{solflux} + 0.1))^2) * (400 / (t_s * (40 - t_s))) \qquad \text{(Eq. 8)}$$

where

diffract	ratio of molecular diffusivity of water to species;
r_j	baseline minimum stomatal resistance (s/m);
solflux	solar radiation flux (W m^{-2});
t_s	surface temperature (^oC).

In the European flux model Module 1 replaces the calculation of r_{st} (described above) with R_{sto} (described below). Module 2 calculates g_{sto} (stomatal conductance in mmol O_3 m^{-2} s^{-1} on a projected leaf area basis). From the latter, stomatal ozone flux to a representative leaf at the top of the canopy could be calculated.

Parameters that are applied in the Module 1 and 2 for maize are listed in Table 1.

Module 1: Canopy Stomatal resistance (R_{sto})

Module 1 calculates canopy stomatal resistance (R_{sto}) in units of s m^{-1} according to :-

$$R_{sto} = \ [(g_{max} \ * \ f_{phen} \ * f_{light} * \max \{f_{min}, (f_{temp} * f_{VPD} * f_{SWP})\}) / 41000\ ]^{-1} \qquad \text{(Eq. 9)}$$

41000 is the factor to convert from mmol m^{-2} s^{-1} to m s^{-1} (Jones, 1992).

g_{max}	is the maximum stomatal conductance in units of mmol O_3 m^{-2} s^{-1} expressed on a projected leaf area basis
f_{phen}	is the relative f determined by leaf age
f_{min}	is the minimum daytime stomatal conductance observed under field conditions
f_{light}	is the relative mean canopy f determined by irradiance
f_{temp}	is the relative f determined by temperature
f_{VPD}	is the relative f determined by the leaf-to-air vapour pressure deficit (VPD)
f_{SWP}	is the relative f determined by the soil water potential (SWP), (related to soil moisture deficit, SMD)

The relative g factors are expressed on a scale of 0-1 and used to modify $g_{max.}$ Capitals denote the whole canopy value; small case denotes a single leaf.

The variables required to calculate R_{sto} in Eq. 10 are either assigned constant values (Table 1) or calculated using the following set of equations:

Table 1: Parameters applied in the calculation of R_{sto} and g_{sto}.

Parameter	Description	Value	Unit
R_{ext}	Resistance of the exterior plant parts to uptake or destruction of ozone	2500	s/m
R_{soil}	Resistance to destruction or absorption at the soil surface	200	s/m
LAI_{max}	Maximum LAI during the growing season	3.5	M^2/m^2
h	Maximum plant height	2	M
g_{max}	maximum stomatal conductance in units of mmol O_3 m^{-2} s^{-1} expressed on a projected leaf area basis	150	mmol O_3 m^{-2} s^{-1} (P)
f_{phen}	Relative g determined by leaf age	1	
f_{min}	Minimum daytime stomatal conductance observed under field conditions	0.2	
f_{light}	relative mean canopy g determined by irradiance	see function	
α	for f_{light} function	-0.005	
f_{temp}	relative g determined by temperature	see function	
T_{min}	for f_{temp} function	0	
T_{opt}	for f_{temp} function	25	
T_{max}	for f_{temp} function	51	
f_{VPD}	relative g determined by the leaf-to-air vapour pressure deficit (VPD)	see function	
VPD_{max}	for f_{VPD} function	1	
VPD_{min}	for f_{VPD} function	2.5	
f_{SWP}	relative g determined by the soil water potential (SWP), (related to soil moisture deficit, SMD)	see functions	

Irradiance (f_{light})

For Module 1, the application of a canopy radiative transfer model is necessary to estimate the influence of irradiance (which changes with canopy depth) on the stomatal conductance (g_{sto}) of sunlit and shaded leaf portions of the canopy.

The f_{light} function requires radiation measured as phothsynthetically active radiation (PAR) in μmol m^{-2} s^{-1}.

$$\text{PAR } (\mu\text{mol m}^{-2}\text{ s}^{-1}) = \text{Solar radiation } (\text{Wm}^{-2}) /2 * 4.57 \qquad \text{(Eq. 10)}$$

Application of the canopy radiative transfer model requires the evaluation of the solar elevation (sinβ) to relate the "height" of the sun in the sky with the penetration of irradiance into the canopy. sinß is calculated according to the following equation, with angles in degrees.

$$\text{solar declination } (\delta) = -23.4 * \text{COS}(360*(\text{day of year}+10)/365) \qquad \text{(Eq. 11)}$$

$$\sin\beta = \sin(\text{latitude})*\sin(\delta)+\cos(\text{latitude})*\cos(\delta)+\text{time of day function} \qquad \text{(Eq. 12)}$$

The canopy radiative transfer model is used to estimate the PAR for both sunlit and shaded canopy portions. This requires that the Idir and Idiff fractions of the total PAR are calculated. This is achieved using a simplified version of the method developed by Weiss & Norman (1985) to estimate the fraction of PAR that is direct irradiance (If_{dir}) using equation 14. The remaining fraction being the diffuse irradiance component (If_{diff}):-

$$If_{dir} = -0.000000084 * PAR^2 + 0.00041 * PAR + 0.4075 \qquad \text{(Eq. 13)}$$

$$If_{diff} = 1 - If_{dir} \qquad \text{(Eq. 14)}$$

The respective fractions of direct and diffuse irradiance can then be used to estimate the absolute irradiance at the top of the canopy.

$$I_{dir} = If_{dir} * PAR \qquad \text{(Eq. 15)}$$

$$I_{diff} = If_{diff} * PAR \qquad \text{(Eq. 16)}$$

The necessary input parameters are then available to apply the canopy radiative transfer model so that the irradiance falling on the sunlit and shaded portions of the canopy as a whole can be estimated:-

$$PAR_{shade} = I_{diff} * \exp(-0.5 * LAI^{0.7}) + 0.07 * I_{dir} * (1.1 - 0.1 * LAI) * \exp[-\sin\beta] \qquad \text{(Eq. 17)}$$

I_{diff} is the flux density of diffuse PAR above the canopy. LAI is the leaf area index

$$PAR_{sun} = I_{dir} * \cos(\theta) / \sin\beta + PAR_{shade} \qquad \text{(Eq. 18)}$$

I_{dir} is the flux density of direct PAR above the canopy.

θ is the angle between a leaf and the sun and is assumed to be constant to 60 degrees.

PAR_{sun} and PAR_{shade} are the flux densities of PAR on sun and shaded leaves. Scaling from the leaf to canopy level is achieved by calculating f_{light} [equation 20 to 24] for sunlit and shaded fractions proportionally (in terms of LAI) for the whole canopy. Where:

$$f_{light}sun = [1 - \exp(\alpha * PAR_{sun})] \qquad \text{(Eq. 19)}$$

$$f_{light}shade = [1 - \exp(\alpha * PAR_{shade})] \qquad \text{(Eq. 20)}$$

$$LAI_{sun} = [1 - \exp(-0.5 * LAI / \sin\beta)] * 2 \sin\beta \qquad \text{(Eq. 21)}$$

$$LAI_{shade} = LAI - LAI_{sun} \qquad \text{(Eq. 22)}$$

$$f_{light} = f_{light}sun * LAI_{sun} / LAI + f_{light}shade * LAI_{shade} / LAI \qquad \text{(Eq. 23)}$$

$f_{light}sun$	is the f_{light} value for sunlit leaves
LAI_{sun}	is the sunlit leaf area of the canopy.
LAI_{shade}	is the shaded area of the canopy
LAI	is the leaf area index
$f_{light}shade$	is the f_{light} value for shaded leaves

Temperature function (f_{temp})

The f_{temp} function estimates the influence of temperature on g_{sto}. Here it is assumed that temperature is constant with depth throughout the canopy.

$$f_{temp} = \max\{f_{min}, [(T-T_{min}) / (T_{opt}-T_{min})] * [(T_{max}-T) / (T_{max}-T_{opt})]^{bt}\} \qquad \text{(Eq. 24)}$$

T is the air temperature in °C,

T_{min} and T_{max} are the minimum and maximum temperatures at which stomatal closure occurs to f_{min},

T_{opt} is the optimum temperature and

bt is defined as: $bt = (T_{max}-T_{opt}) / (T_{opt}-T_{min})$

Vapour Pressure Deficit function (f_{VPD})

The f_{VPD} function estimates the influence of Vapour Pressure Deficit (VPD) on g_{sto}. Input data available from CAMx are water vapour mixing ration in ppm (w_p) and atmospheric pressure in hPa (p).

Vapour pressure is calculated using the following equations:-

Calculate water vapour in g/kg (w_s)

$$w_s = (w_p * 1000) / 10^6 \qquad \text{(Eq. 25)}$$

Calculate vapour pressure in hPa (e)

$$e = \frac{w_s * p}{1000 * 0.622} \qquad \text{(Eq. 26)}$$

Saturated vapour pressure (e_s) in hPa which is a function of temperature is required to convert from vapour pressure to vapour pressure deficit.

$$e_s = 611.21 * \exp\left(\frac{17.502 * T}{240.97 + T}\right) / 100 \qquad \text{(Eq. 27)}$$

e_s, the saturation vapour pressure is in Pa. T is the air temperature in °C.

The calculation of vapour pressure deficit, VPD, in kPa is then

$$\text{VPD} = \min\{0, (e_s - e)/100\} \qquad \text{(Eq. 28)}$$

The resulting VPD value (divided by 1000 to give VPD in kPa) can then be used in the following f_{VPD} function:-

$$f_{VPD} = \min\{1, \max\{f_{min}, ((1-f_{min}) * (VPD_{min} - VPD) / (VPD_{min} - VPD_{max})) + f_{min}\}\} \qquad \text{(Eq. 29)}$$

Soil water status (f_{SWP})

The CAMx model identifies three levels of soil water potential; these are assigned to the following f_{SWP} values, where f_{SWP} represents the function deterring the influence of soil water status on g_{sto}. Although this is a simplification of the model, only allowing step-changes in g_{sto} response to SWP it will provide a spatial indication of the relative importance of the SWP component.

No water stress / irrigated crops	$f_{SWP} = 1$	
Medium water stress	$f_{SWP} = 0.5$	(Eq. 30)
Extreme water stress	$f_{SWP} = f_{min}$	

Module 2: Leaf stomatal conductance (g_{sto})

Module 2 calculates g_{sto} (stomatal conductance in mmol O_3 m^{-2} s^{-1} on a projected leaf area basis). From this stomatal ozone flux to a representative leaf at the top of the canopy can be evaluated.

$$g_{sto} = (g_{max} * f_{phen} * f_{light} * \max\{f_{min}, (f_{temp} * f_{VPD} * f_{SWP})\} \qquad \text{(Eq. 31)}$$

With the exception of g_{sto}, the only other variation between Module 1 and Module 2 is in the calculation of irradiance, f_{light}. For Module 1 the application of a canopy radiative transfer model is necessary to estimate the influence of irradiance (which changes with canopy depth) on the stomatal conductance of sunlit and shaded leaf portions of the canopy. For module 2, stomatal conductance is estimated only for a leaf in the upper canopy and hence can be calculated simply as a function of the irradiance reaching the top of the canopy as follows:

$$f_{light} = 1 - \exp(\alpha * PAR) \qquad \text{(Eq. 32)}$$

PAR is the phothsynthetically active radiation and α is constant for f_{light}.

The calculation of top leaf stomatal ozone flux ($fluxO_3$) can then be made using Eq. 33.

$$fluxO_3 = g_{sto} * O_3 \qquad \text{(Eq. 33)}$$

Results and Discussion

As an initial method of comparing the CAMx and flux methodologies, a single 50 x 50 km model grid cell over Zimbabwe was selected to investigate model outputs from a 5-day period covering the 10 to 14 January 2001. The results in Fig 3 (top) show clearly the diurnal variation in ozone concentration over Mashonaland, with concentrations reaching more than 90 ppb in the middle of the day. As may be expected, the deposition velocities (Vd) calculated with both the CAMx and the European methodologies (Fig 3, middle) also show strong diurnal variations.However, the V_d values estimated using the European method are consistently higher than those calculated using the CAMx routine during the daytime periods.

Due to the relationship between the ambient concentration and V_d, the resultant ozone flux is low at night and reaches maximum values during the daytime period. The European flux model values are consistently greater than those estimated using CAMx. For CAMx, ozone fluxes range from 0 to about 6 nmol O_3 m^{-2} s^{-1}, compared with the European method values, which range from near zero values at night to average daytime maxima of more than 10 nmol O_3 m^{-2} s^{-1} , reaching more than 15 nmol O_3 m^{-2} s^{-1} on 13 January 2001. The higher ozone fluxes estimated using the European model most likely reflect the higher maximum stomatal conductance value assigned to maize in the European model.

It is also interesting to note that the two days with the highest ozone concentrations (10 (0 to 24 hours) and 11 January (24 -48 hours)) are not always those with the highest calculated ozone fluxes. This apparent incongruity would possibly be even more striking if stomatal fluxes were compared with ozone concentrations and highlights the need to use flux rather than concentration based approaches for ozone risk assessments.

The comparison is expanded to a portion of the maize producing area of Mashonaland West to Mashonaland East and Mashonaland Central in Zimbabwe. Here a 550 km by 550 km 'window' is extracted from the southern Africa modelling domain. In Fig 4 each panel represents an area of approximately 300 000 km^2, running from grid cell 33 to 44 in an east to west direction and from 34 to 44 in a south to north direction. A single modelled value represents each cell. Unlike the diurnal depiction in Fig. 3, the results shown in Fig. 4 are a snap-shot over the defined domain showing results for a single hour, namely 16:00 on 10 January 2001.

For this hour the modelled concentrations of ozone range between 20 and 100 ppb over the model window (Fig. 3, left). Concentrations exceed the threshold value of 40 ppb suggesting that damage to vegetation may be expected over an extensive area covering almost the entire southern half of the window.

One might expect high ozone fluxes to coincide with areas of high ozone concentrations, given the relationship in Eq. 1. Generally speaking this assumption is valid for the CAMx deposition (Fig. 3, centre) with the highest fluxes occurring over the southern parts of the window, ranging from low values and reaching 6 nmol O_3 m^{-2} s^{-1}. It is not a perfect match however, and the area of maximum fluxes occur to the south of the maximum ambient concentrations. The amount of damage predicted by ozone flux models depends on the toxicity of the absorbed dose. The toxicity levels of absorbed ozone to maize growing in southern Africa is not yet known, but European work for wheat and potato has shown that damage could occur at fluxes above 6 nmol m^{-2} s^{-1}.

As seen in Fig. 3 for the modelling period in cell 38:38, the ozone flux estimated with the European method is consistently higher than that estimated with the CAMx methodology (Fig. 4 centre and right) though there is agreement in the general area where the ozone flux is highest. However, the European model also identifies areas of high ozone fluxes in the northeastern parts of the window (Fig. 4, right).

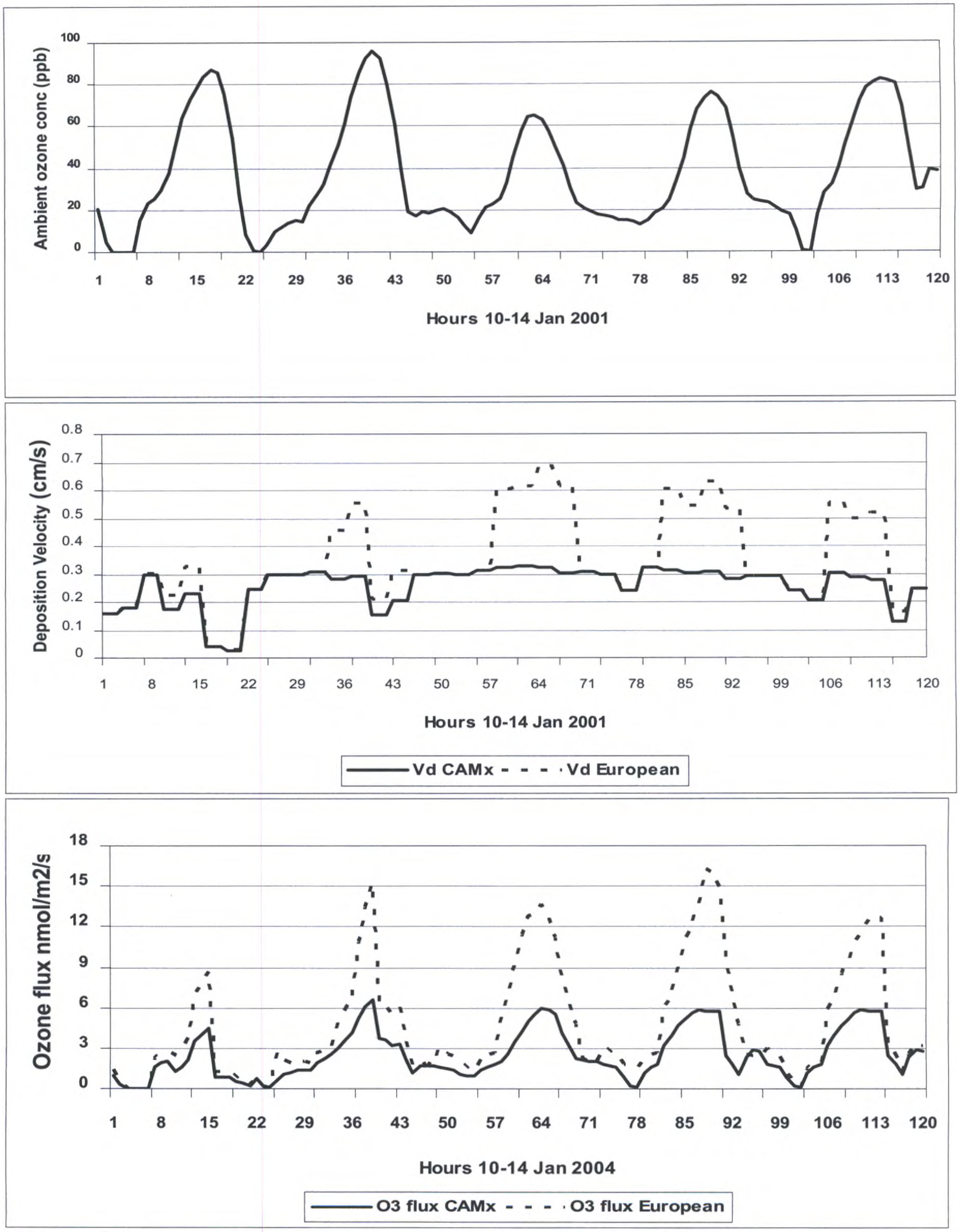


Figure 3: Model results for cell (38:38) in Zimbabwe for 120 hour modelling period 10 to 14 January 2001 with ozone concentration in ppb (top), and the corresponding deposition velocities (centre), and total ozone fluxes (bottom) the latter two components calculated with both the CAMx and European methodologies.

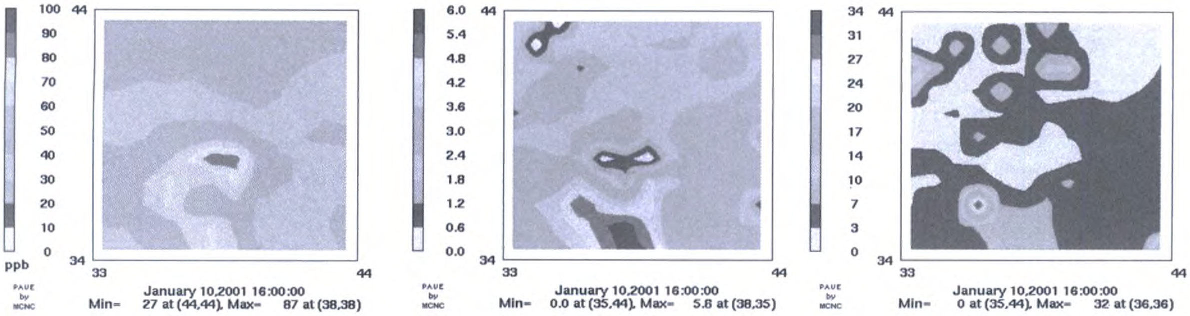


Figure 4: Modelled ambient ozone concentrations in ppb (left). Modelled CAMx total ozone flux (centre) and modelled flux using the European stomatal flux algorithms over Mashonaland, Zimbabwe at 16:00 on 10 January 2001. Values given in nmol O_3 m^{-2} s^{-1}.

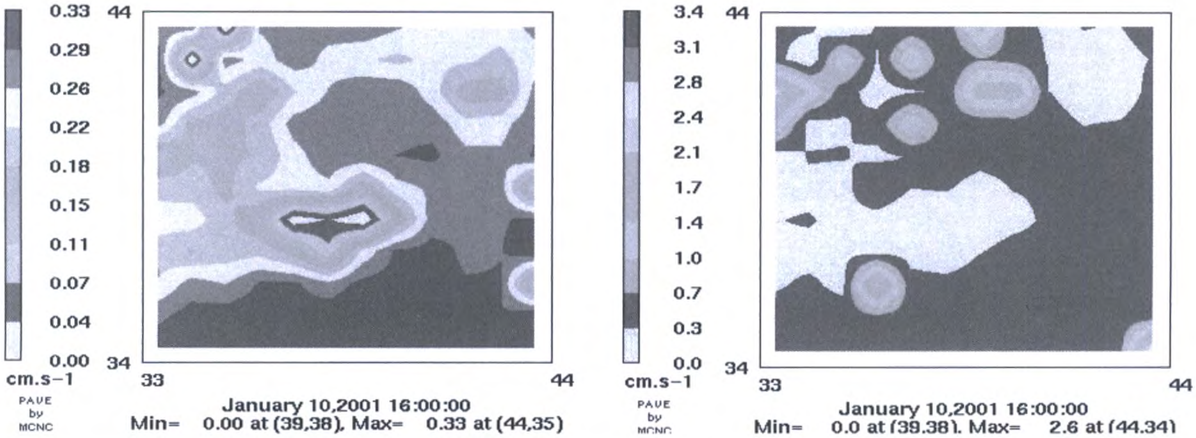


Figure 5: Modelled ozone deposition velocity in cm s^{-1} over Mashonaland, Zimbabwe at 16:00on 10 January 2001, estimated with the CAMx (left) and the European flux algorithm (right).

An interesting comparison is made when evaluating the modelled CAMx ozone fluxes (Fig. 4 centre) and the European ozone fluxes (Fig. 4 right). Again the maximum flux occurs in the south of the window where rates range between 3 and 6 nmol O_3 m^{-2} s^{-1} for the CAMx routine. These are generally higher for the European method with ozone fluxes in the south ranging from 3 to more than 30 nmol O_3 m^{-2} s^{-1} in one particular area. This is most likely due to the European model being parameterised specifically for maize rather than a more generalised "agricultural" land-cover type. Also interesting is the area of relatively higher ozone fluxes in the northwestern parts of the window predicted using the European method that coincides with relatively low ambient ozone concentrations. This suggests that the incorporation of additional environmental parameters using the European method is able to identify instances when lower ozone concentrations are capable of producing higher ozone fluxes due to the low resistance to ozone uptake and deposition. The European flux estimates reach 14 nmol O_3 m^{-2} s^{-1}.

Examination of the deposition velocities (V_d) (Fig. 5) adds some additional insights. Again, V_d calculated with the CAMx algorithms is consistently lower than the European method at 16:00 on 10 January, also seen in Fig. 3. Both approaches return V_d values in the region of 0.3 cm s^{-1} over the largest part of the window, but the maximums returned with the European method are much higher than that with CAMx. The highest CAMx values occur in the southwestern part of the window, in the region of 0.35 cm s^{-1}, decreasing almost consistently towards the northeast. The European flux model shows a contrasting picture with patches of higher V_d ranging between 0.3 and 1.7 cm s^{-1}, with high deposition velocities in the south and in the north east, coinciding with the observed maximums of ozone flux (Fig. 4, right). It is useful to make preliminary comparisons of the model output values with deposition velocities for maize measured under field conditions to give some indication of how well the different models are performing. Deposition velocities for maize growing in The Netherlands were measured by van Pul & Jacobs (1994). These measurement data found V_d values in the range of 0.2 to 1.6 cm s^{-1} which would indicate that the predictions of deposition velocity made using the European stomatal flux algorithms are closer to realistic values for this species. However, additional work will be necessary to fully understand the model outputs in relation to observed measurements under location specific climatic conditions.

Conclusions

It is commonly recognised that using concentration based approaches, such as the AOT40, to assess potential damage to agricultural crops has some limitations. The most notable of these are due to the modifying factors that limit the amount of pollutant actually taken up by the plant and hence uncouple the relationship between concentration and damage. These generally occur through modifications to stomatal conductance caused by local environmental conditions such as low humidities and high soil water stresses. These factors are not considered by the AOT40 approach since damage is only related to the external pollutant concentration rather than the absorbed pollutant dose.

As such, a "flux based" risk assessment method (currently being developed as part of an ozone deposition model for use in Europe) has been applied in this work to assess the possibility of using an additional, more biologically relevant risk assessment method within the CAPIA project. The aim of the work was to initiate the application of certain components of the European deposition module in the standard CAMx model, in particular to investigate the feasibility of including the stomatal flux algorithms, and so improve the estimates of ozone uptake in plants and the risk posed to crops.

The modification of the CAMx model has been successfully completed and initial results, using a single day's model output over Zimbabwe, illustrate the perceived risk across the region using both concentration and flux based approaches. Analyses of these data highlight the spatial differences between ambient concentrations as calculated for CAPIA, and ozone fluxes calculated by both the CAMx and European deposition models. The main conclusions of this research are:

- Modelled ambient ozone concentrations exceed the 40 ppb threshold over much of southern Africa. This would suggest that application of the concentration based (AOT40) risk assessment method would indicate the potential for ozone damage to maize in the region.
- The areas where ozone concentrations are elevated are not the same as those with the highest ozone fluxes. This suggests that application of the more biologically relevant flux-based risk assessment methods would identify different regions within the modelling domain as being those where most damage to maize is likely to occur.
- The CAMx model tends to underestimate both the deposition velocity and actual ozone flux in comparison to the flux method. This is most likely due to the flux method modelling for a specific species (i.e. maize) rather than a more generic land-cover type (i.e. agricultural crops).
- The maximum total ozone fluxes are above the critical stomatal flux values of 6 nmol m^{-2} s^{-1} currently defined and applied within Europe to assess risk and economic impacts of ozone to agricultural crops.

This preliminary work has shown that both the concentration and flux based approaches indicate that ground level ozone concentrations could result in damage to maize across southern Africa. However, the areas of maize growing identified as being at most risk from ozone varies spatially dependant upon which of the two approaches are used. This is due to the flux-based method reducing the uncertainty of the risk assessment by incorporating modifying factors and relating potential damage to absorbed dose rather than concentration. The initial work has also shown there to be differences between the current approaches to estimating deposition in CAMx compared to the species-specific modelling conducted using the European approach. It is important that the initial work is expanded in order to develop improved understandings of ozone deposition and uptake by agricultural crops.

References

Emberson L. D., Ashmore M. R., Murray F., Kuylenstierna J. C. I., Percy K. E., Izuta T., Zheng H., Sheu B. H., Liu C. P., Agrawal M., Wahid A., Abdel-Latif N. M, van Tienhoven M, de Bauer L. I. and Domingos, 2001a. Impacts of air pollutants on vegetation in developing countries, *Water, Air and Soil Pollution*, 130, 107-118.

Emberson L.D., Ashmore M.R., Simpson D., Tuovinen J.-P., and Cambridge H.M., 2001b Modelling and mapping ozone deposition in Europe. *Water, Air and Soil Pollution,* 130, 577-582.

ENVIRON, 2003. User's guide to the Comprehensive Air Quality Model with Extensions (CAMx), ENVIRON International Corporation, 101 Rowland Way, Novato, CA 94945.

Fleming G. and van der Merwe M., 2002. Spatial disaggregation of greenhouse gas emissions inventory data for Africa south of the equator, http://gis.esri.com/library

Fuhrer, J., Skärby, L., and Ashmore, M.R., 1997, Critical levels for ozone effects on vegetation in Europe. *Environmental Pollution*, 97 (1-2), 91-106.

Greenberg J.P, Guenther A., Harley P., Otter L., Veenendaal E. M., Hewitt C. N., James A. E. and Owen S. M., 2003. Eddy flux and leaf-level measurement of biogenic VOC emissions from mopane woodland of Botswana, *Journal of Geophysical Research*, 108(D13), 8 466, doi:10.1029/2002JD002317,2003.

Harley P., Otter L., Guenther A. and Greenberg J., 2003. Micrometeorological and leaf-level measurements of isoprene emissions from a southern African savanna, *Journal of Geophysical Research*, 108(D13), 8 468, doi:10.1029/2002JD002592,2003.

Scholes R. J., Kendall J. and Justice C. O., 1996. Emissions of trace gases and aerosol particles due to vegetation burning in southern Africa, *Journal of Geophysical Research*, 101, D19, 23 677-23 682.

van Pul W.A.J. and Jacobs A.F.G. (1994) The conductance of a maize crop and the underlying soil to ozone under various environmental conditions, Boundary-Layer Meteorology, 69, 83-99.

Wesley M. L. and Hicks B.B., 1977, Some Factors that affect the deposition rates of sulphur dioxide and similar gases on vegetation, Journal of the Air Pollution Control Association, 27(11), 1110-1116.

Wesley M. L., 1989. Paramererization of surface resistance to gaseous dry deposition in regional-scale numerical models, *Atmospheric Environment*, 23, 1293-1304.

van Tienhoven A. M., Otter L., Lenkopane M., Venjonoka, K and Zunckel M., 2005. Assessment of ozone impacts on vegetation in southern Africa and directions for future research, *South African Journal of Science,* 101, 143-148.

van Tienhoven A. M., Zunckel M., Emberson L., Koosailee, A. and Otter L., 2005. Preliminary assessment of ozone impacts to maize (*Zea Mays*) in southern Africa, *Environmental Pollution*, 140(2006), 220-230.

Zunckel M., Venjonoka K., Pienaar J. J., Brunke E-G., Pretorius O., Koosaliee A., Raghunandan A. and. van Tienhoven M. A., 2004. Surface ozone over southern Africa: Synthesis of monitoring results during the Cross Border Impact Assessment project, *Atmospheric Environment*, 38(2004), 6139-6147.

Author Index