FT-IR Calibration and Concentration Measurements

Spectrum Quant+ (PERKIN ELMER software) was applied for concentration evaluation of the FT-IR spectra. The software operates with chemometric procedures (PLS: partial least-squares) based upon inputs from calibration spectra. This multi regression method was developed by Wold (1966). A cross-validation was used. For base line corrections, first derivatives were chosen. The spectra had to contain the range of gas concentrations of the gas (or gases) to be evaluated and, if possible, different concentration levels of disturbing gases like water vapor or others. Secondly, during the development of the calibration method, the most suitable wave number ranges for the calibration procedure had to be put in, or in other words, non-interesting and disturbing spectrum regions were to be "blanked". In the calibration methods, developed here, 15 to 30 standards (calibration spectra) were measured with a resolution of 0.2 cm^{-1} and a point interval of 0.05 cm^{-1}.

For multi gas analysis, the concentration distribution with the maximum of mutual linear independency was calculated. Then, specific instrument calibration mixtures (Linde HiQ® Specialty Gas) were applied or, in most of the cases, the gas mixtures were produced by using volume calibrated 10 dm^3 plastic bags (Linde Plastigas®; Linde, 2006) filled with nitrogen or filled with ambient air of known composition and adding definite quantities of pure gases by micro-liter syringes through the septum of the bags. It was found that the best calibration methods were achieved using the PLS1 algorithm of the software and blanking all spectra parts except regions with no or only weak overlapping spectral lines. These lines were determined by qualitative measurements with variable concentrations and by spectra simulation using USF-HITRAN-PC (HITRAN, 1992). It was found that the best accuracy could be achieved when the standard lines and sample lines did not differ in the magnitude of absorbance. Although it is possible to develop a calibration method, which works for e.g. CO_2 from 200 ppm till 20% by volume, usually strong deviations between predictions and real concentrations will be measured in the low or high edge of the concentration range. Therefore, for evaluation of high variable gas concentrations in the sample, e.g. from compost air, intensity adopted calibration software tools had to be applied.

Measuring System for Field Application – Flux Rates from Soil

The closed chamber technique is applied on an experimental field in order to obtain the seasonal and the spatial variability of N_2O and CH_4 soil fluxes. Gas flux measurements are performed several times weekly by means of an automated gas chromatograph (GC). The gas flux chambers are placed for the measurements at sealing rings (Y profile, sealing by water level) embedded in the soil. The gas flux chambers have a volume (V) to area (A) ratio of $V/A = 0.315$ m (volume 0.064 m^3, inner diameter 0.509 m). Two evacuated gas samplers (100 cm^3 bottles with Teflon sealing and vacuum taps) are connected to each box. The first is filled when the box is put on the water-sealed ring on the soil and the second one after about 60 minutes enclosure time. Then the samplers are connected with the automated GC-injection control system. The GC is fitted with an electron capture detector (ECD) and a flame ionization detector (FID). The operating temperatures for the ECD and the column temperature are 300 °C and 65 °C respectively. Both the pre-column (length 1 m) and the main column (length 3 m) are packed with Porapak Q (80/100 mesh). In one computer-controlled run up to 64 samples can be analyzed.

To apply a two point evaluation for gas flux measurements, the linearity of the concentration increase and the concentration gradient inside the gas flux chamber must be known. CO_2 and N_2O, both of them generated in the soil, have nearly equal diffusion constants. The easily measurable CO_2 served for the evaluation of linearity and mixing homogeneity of the measuring chamber (measurements at several heights in the closed chamber). Studies on the concentration increase of CO_2 in a flux chamber on bare soil and on grass sites demonstrated that the increase in concentration was linear (R^2 between 0.9973 and 0.9986) during measurement periods of 60 minutes (Fig. 8).

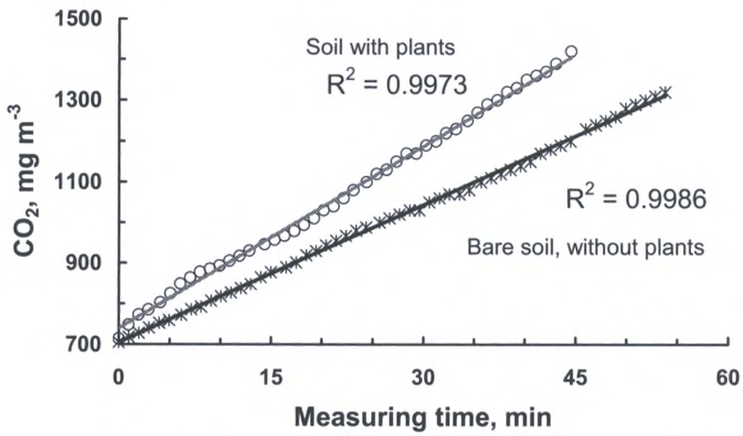

Figure 8. Increase of CO_2 in a gas flux chamber on bare soil and on soil with plants

Conclusions

Different emissions from agricultural operations have been investigated at the Leibniz-Institute for Agricultural Engineering Potsdam-Bornim (ATB) since several years. Gaseous emissions such as ammonia (NH_3), nitrous oxide (N_2O), methane (CH_4) and carbon dioxide (CO_2) were determined from livestock buildings on farms, manure storage facilities on laboratory scale, manure composting on farms and soils. Various methods were being applied and advanced for the special gases and emission sources.

The favored methodology for the determination of emissions from livestock buildings on farms was analyzing gas concentrations by a photoacustic IR analyzer and ascertaining the corresponding ventilation rates. The ascertainment of the ventilation rates depended on the ventilation of the building. For forced ventilated buildings, it was ascertained by calibrated ventilation fans and for natural ventilated buildings by tracer gases (SF_6 or Krypton 85). For the investigation of emissions from manure storage facilities on laboratory scale, the open / dynamic chamber method was preferred. Gas concentrations were determined by a photoacustic IR analyzer also used for building measurements. The closed chamber method was applied for measuring gas fluxes from farm manure composting and from soils. Gaseous emissions from manure composting (CH_4, CO_2, N_2O and NH_3) were determined by a high resolution FT-IR spectrometer. Emissions from soil (CH_4, CO_2 and N_2O) were measured with an automated gas chromatograph (GC) fitted with an electron capture detector (ECD) and a flame ionization detector (FID).

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A Methodology For Estimating Ammonia Emissions From Farm Manure Storage Using Passive Sampling and Atmospheric Dispersion Modeling

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Abstract

Ammonia emissions from stored farm manures form a small but significant proportion (~10%) of the total emitted annually from UK sources. These emissions are regulated by a number of factors including: the surface area of the store; the storage method; the physical form of the stored manure and the source of manure. A method was developed to estimate in-situ emissions from a range of store types in order to address the variability associated with these emissions. Passive diffusion samplers (Willems Badges) were used to determine vertical concentration profiles to a height of 4.5 m at a number of locations around each slurry store. An atmospheric dispersion model was applied to estimate the contribution from emissions from other on-farm ammonia sources to the measured data and also to determine the emission rate from the stored manure. A procedure was applied whereby the emission rates from each source were allowed to vary and a numerical equation solver was applied to backfit the modelled data set to the field measurements.

This methodology was applied at four farms that used two types of stores (weeping wall and earth banked lagoon) that are currently under-represented in UK emissions estimates. This paper reports the emissions for one of the farms that operated a weeping wall lagoon, a system which allows the partial separation of slurry solids and liquids, which can then be managed more conveniently as separate components. These measurements were collected on a monthly basis for a complete year, with each dataset being collected over 24 hours. Emissions were found to peak at 2 g NH_3-N m^{-2} day^{-1} in spring, reducing considerably over the summer and winter periods. The average emission from the farm was 0.7 g NH_3-N m^{-2} day^{-1} which agrees with other recent data on ammonia emissions from crusted slurry stores and, though it is within the range of data considered in the UK emissions inventory, suggests that emissions from this type of store may be currently overestimated in national predictions.

Introduction

Emissions of ammonia (NH_3) to the atmosphere in the UK are dominated by the contribution from the agricultural sector (Pain et al., 1998). These emissions originate from the surface application of agricultural wastes, livestock housings, slurry stores, fertiliser applications and grazing. Such inventories are relied upon to determine the total emission from UK agriculture and are increasingly being used to develop abatement strategies for targeting reductions in NH_3 emissions and to demonstrate compliance with international legislation on NH_3 emission ceilings (EC, 2001). The accuracy of the emission estimates from these inventories is dependant on the quality of the emission factors that are applied. For some of the agricultural practices included within the inventories, emissions estimates are based on a limited number of field trials that may not fully reflect the real-world variability. In particular, prior to this study there were no direct measurements of emissions from weeping wall or earth-banked lagoons with strainer boxes, which together account for approximately 30% of dairy slurry storage (Smith et al., 2001). These stores provide partial separation of slurry solids and liquids, with one or more sides of the store having slatted walls allowing liquid to drain to a separate storage area. The liquid and semi-solid manure fractions can then be managed separately. There are three distinct potential areas of NH_3 emission: a) the storage surface; b) the wetted slatted walls and drainage channels; c) the liquid storage area surface, and these may result in different emission characteristics than for a conventional slurry lagoon.

The overall project objectives included the partitioning of nutrients between the solid and liquid phases of the stored manure, but in this component of the study the aim was to quantify NH_3 emissions from the semi-solid and liquid storage fractions in weeping wall/strainer box lagoons. This paper describes the methodology developed to make such measurements and reports preliminary results for one of the study sites.

Estimation of Emission Fluxes

The dispersion of NH_3 emitted to the atmosphere can be determined at a farm-scale using mathematical models (Hill et al., 2001, Fowler et al., 1998). Such models are typically used to predict air concentrations and deposition fluxes arising from known, or estimated, rates of emission. However, these models can also be applied to back-calculate emission rates from known air concentrations.

Back-calculation techniques have been applied to determine emission rates from single isolated sources such as slurry applications (McInnes et al., 1985). For the sites that were investigated in this study NH_3 concentrations were likely to arise from a number of on-farm sources depending on the direction of the prevailing wind. Attempts were made to limit these influences during the site selection process, through the design of the field experiments and through conducting monitoring during periods when favourable wind directions were forecast.

A modified back-calculation method was developed that applied an advanced atmospheric dispersion model along with a multi-variant statistical modelling tool to both determine emission rates from the slurry stores and to account for emissions from other on-farm sources.

Measurement of Air Concentrations

Air concentrations were measured at several heights surrounding each slurry lagoon. This enabled the statistical model to diagnose emission fluxes from the measured horizontal and vertical distributions of air concentrations. Vertical concentration distributions have been found to vary strongly with source type. Ammonia emitted from stored manure is advected by the wind immediately upon release, hence the vertical concentration profiles are similar to those from an equivalent sized surface-level area source, with concentrations reducing rapidly with height. However, NH_3 initially emitted into a naturally ventilated farm building is relatively well mixed within the confines of the building, being released through both the sides and roof of the structure. The dispersing plume from farm buildings therefore lacks the strong vertical signal of a plume from a ground level source, with measurements showing such plumes are almost uniform with height below the roofline of the building (Hill et al., 2001).

Passive sampling methods based on diffusion theory are useful when simultaneous measurements are required at numerous spatial locations and at varying heights, as they are light and have no requirement for electricity. The Willems badge sampler (Willems, 1990) was applied in these field experiments as this sampler can measure air concentrations over 24 hour periods with a limit of detection between 1 –5 µg NH_3-N m^{-3}, furthermore the sampler has a precision of the order of +/- 10 % (Hill, 2000).

Air concentrations (χ_a) are calculated, using Equation 1, from measurements of the mass of NH_3-N collected on the absorption filter (M_c) of the Willems badge minus a laboratory blank value (M_b), the combined boundary layer and filter resistance of the badge (R_b+R_f, 164 s m^{-1} at 15 °C), the area of the filter (A, 6.158 x 10^{-04} m^2) and the duration of the experiment, t (in seconds).

$$\chi_a = \frac{(M_c - M_b)(R_b + R_f)}{At} \qquad \text{Equation 1}$$

Willems badges were mounted on a number of masts at each of the experimental sites at heights of 0.5 m, 1.0 m, 2.0 m, 3.0 m and 4.5 m.

Atmospheric Dispersion Modeling

Dispersion factors (D in s m^{-3}) were calculated using the UK-ADMS atmospheric dispersion model. This model was chosen as it includes a realistic treatment of the physics of the atmospheric boundary layer, using Monin-Obukhov scaling, to describe the vertical variation in wind speed and turbulence in the atmosphere, and also as it incorporates a building effects module which allows for the streamline deflections and enhanced turbulence that occur downwind of animal housings.

The ADMS model was configured with the following input data:

a) Meteorological data for each measurement period, including wind speed and wind direction at a specified height (typically 2 m), air temperature and solar radiation.
b) Source location and dimensions for the major points of NH_3 emission at each of the farms, including the slurry lagoon, dirty water stores, the main buildings on the farm, open areas and feedlots.
c) A building configuration at each of the farms.
d) Estimates of the surface roughness length for the farm buildings and meteorological sites.
e) The position of each of the Willems badge monitoring points.

Multi-Variant Statistical Model for the Determination of Emission Rates

For each of the NH_3 emission sources (i), dispersion factors were calculated for the complete array of Willems badge monitoring points (j). Modelled air concentrations could then be determined at each Willems badge monitoring point ($\chi_{m(j)}$) from know emission rates for each source ($E_{(i)}$, in µg NH_3-N s^{-1}) using Equation 2.

$$\chi_{m(j)} = \sum_i D_{(ij)} E_{(i)}$$

Equation 2

The level of agreement between the modelled and measured air concentrations can be determined using the Normalised Mean Squared Error (NMSE), a statistic applied in the ASTM methodology for model evaluation (Irwin et al., 2003). Equation 3 shows an implementation of the NMSE using the dispersion modelling methodology applied herein and accounting for a constant background concentration (χ_b). Overbars denote ensemble means.

$$NMSE = \frac{\overline{\left[\chi_{a(j)} - (\chi_{m(j)} + \chi_b)\right]^2}}{\overline{\chi_a}\ \overline{(\chi_m + \chi_b)}}$$

Equation 3

A numerical equation solver was used to determine the emission rates for each source ($E_{(i)}$) and overall background concentration (χ_b) that resulted in the minimal value of NMSE. This methodology was applied to each of the experimental runs. The initial values for $E_{(i)}$ were obtained using the emission factors contained in Misselbrook et al. (2000). As values of $E_{(i)}$ calculated using this methodology incorporated the data from all the measured air concentrations, replication of the Willems badges at a single height was not required.

Experimental Setups

Field experiments were conducted at four farms, two sites in Devon and locations in Wiltshire and Nottinghamshire. This paper reports the store characteristics and preliminary results for one of the Devon sites, a dairy farm with approximately 185 cows.

An aerial photograph of the farm is shown in Figure 1. A large rectangular weeping wall type slurry store (25m wide and 34 m long) can be seen adjacent to the main cubicle house. Slurry from the cubicle house was scraped daily into the store via a ramp at the NW corner. The S and E walls of the store were slatted, allowing liquid to drain from the store via channels to two smaller 'dirty water' lagoons (at the bottom of Figure 1) from where it could be irrigated directly to the land. In addition, dairy parlour washings and yard run-off were channelled directly to the dirty water lagoons. Measurements were made on three sides of the slurry lagoon, close to the dirty water store and in the fields to the south of the lagoon (to provide an estimate of background concentrations). Those measurements collected close to the slurry store would be relatively unaffected by emissions from the farm buildings for wind directions from the south and south-east, though strong contributions would be expected for winds from the north-west and west.

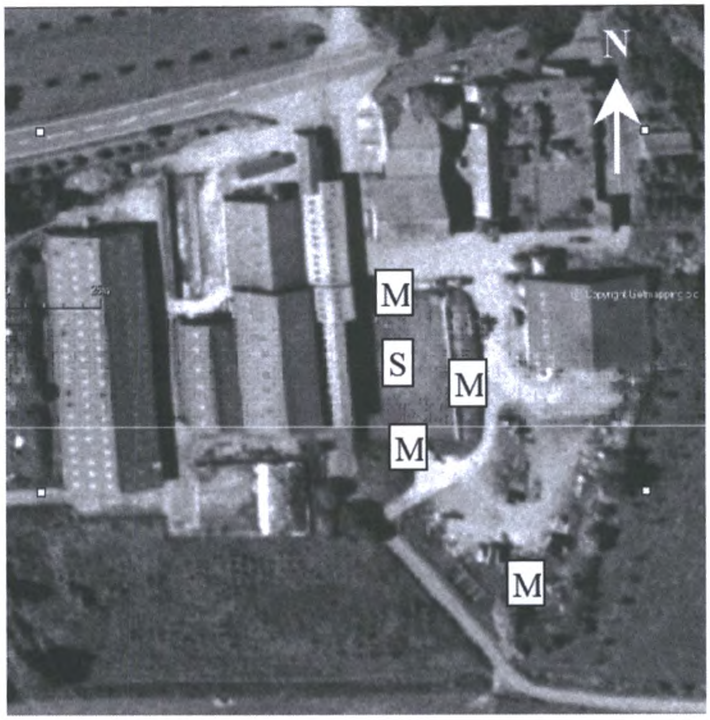

Figure 1. Aerial photograph of the study farm. The location of the slurry store is designated as (S), the locations of the on–farm measurement masts are designated as (M). A background mast was located in the fields to the south of the farm (not shown). Aerial photograph copyright Getmapping.com.

Results and Discussion

Meteorology

A summary of the meteorological data for the study farm is shown in Table 1. The meteorological data summary illustrates that a considerable variability in the wind directions often occurred over the duration of each of the measurement periods (shown by the standard deviation, sigma, of the wind direction). For many of the experimental runs, measured air concentrations could not be assumed to be only derived from emissions from the slurry store, confirming that the statistical methodology described previously was required in order to account for the influence of the additional on-farm NH_3 sources.

Table 1. Summary of meteorological measurements from the study farm. Data were collected at a 30 minute time resolution and a height of 2 m.

Start date	Duration (hours)	Wind speed ($m\ s^{-1}$)		Wind direction (degrees)		Temp. (^oC)	RH (%)	Solar radiation ($W\ m^{-2}$)		Rainfall (mm)
		Mean	Sigma	Mean	Sigma	Mean	Mean	Mean	Max	Total
12/09/02	24.0	1.9	1.6	86	70	15.2	88	125	524	1.0
24/10/02	24.0	2.5	1.6	214	74	11.8	90	64	293	4.0
03/12/02	25.5	1.6	1.4	347	92	7.8	92	27	151	4.8
15/01/03	23.5	1.9	1.6	260	51	4.6	93	27	214	0.0
26/02/03	23.5	3.2	0.9	122	7	7.9	91	17	134	0.2
15/04/03	21.5	1.2	1.1	152	77	12.1	74	162	730	0.0
16/06/03	26.0	1.5	1.1	217	68	15.9	80	285	826	0.0
24/07/03	26.0	2.0	1.1	206	91	15.6	92	135	816	24.8
12/11/03	27.0	1.4	1.2	250	97	8.6	92	45	313	0.0

Air Concentrations

The air concentrations measured during each of the experimental runs were used to diagnose the emission rates from the slurry store. This methodology was applied to the complete dataset of concentrations from each run and made use of the differences in the vertical concentration profiles from stored manure and from naturally ventilated farm buildings.

Vertical concentration profiles are shown in Figure 2 for a sampling mast where winds from the southwest (on 24-10-02) would include NH_3 concentrations from both the slurry store and the farm buildings, and from the southeast (on 26-02-03) where NH_3 concentrations would be predominantly derived from the slurry store. The profiles from 24-10-02 were interpreted using the atmospheric dispersion model showing that, at the lower sampling points in the profile, concentrations were mainly derived from the slurry store whilst at elevated points, to which emissions from the slurry store had not been vertically dispersed, NH_3 concentrations were derived from the farm buildings. The vertical concentration profile for 26-02-03 shows the situation where wind trajectories were unaffected by the farm buildings, hence the vertical concentration profile shows a classical exponential decrease with height, tending to background values at the upper measurement locations.

Emissions

Table 2 summarises the Normalised Mean Squared Error and Correlation squared (R^2) statistics from the multi-variant modelling approach used to derive emission rates from the measurements. These statistics are useful to illustrate the relative uncertainty that affected individual monitoring periods and show that for seven of the nine sampling runs, low values of NMSE (scatter between measured and modelled values) and R^2 values close to 0.9 were found illustrating accurate model estimates of emission rates.

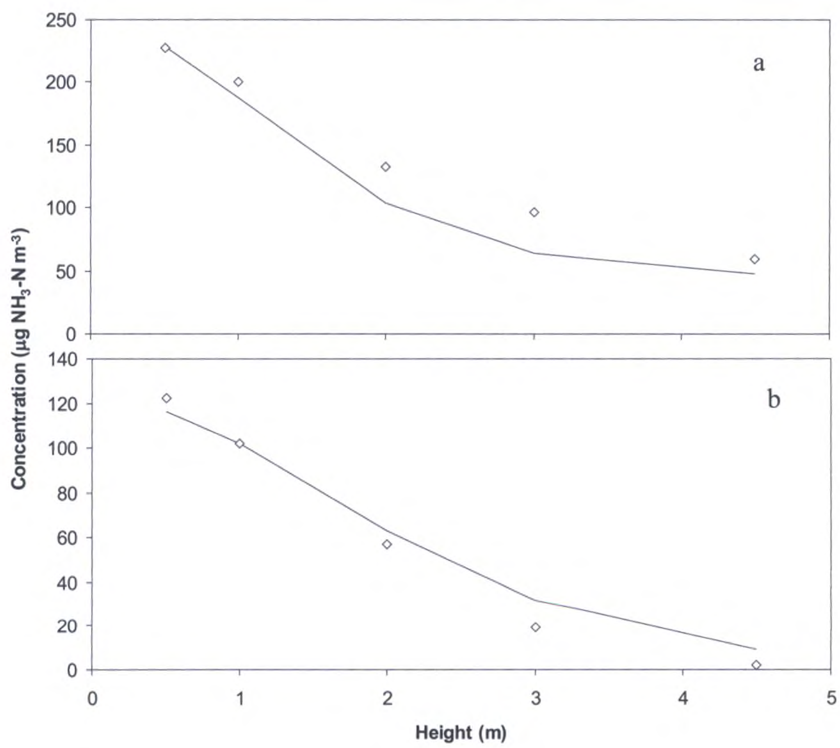

Figure 2. Air concentrations of NH_3-N measured (points) and diagnosed using the atmospheric dispersion model (line) for the northerly mast shown in Figure 1 for 2 periods: (a) when wind trajectories included influences from the buildings (24/10/02); and (b) when wind trajectories were consistently from the slurry store (26/02/03).

Table 3: Normalised Mean Squared Error and Correlation statistics determined following the multi-variant analysis used to diagnose emission rates from the slurry store.

Run	Date	NMSE	R^2
1	12/09/02	0.31	0.87
2	24/10/02	0.29	0.90
3	03/12/02	0.01	0.98
4	15/01/03	0.12	0.87
5	26/02/03	0.03	0.99
6	15/04/03	0.38	0.89
7	16/06/03	0.37	0.64
8	24/07/03	1.06	0.35
9	12/11/03	0.06	0.91

The timeseries of diagnosed emission rates (Figure 3) showed marked variations throughout the measurement period, with peak values of approximately 2 g NH_3-N m^{-2} day^{-1} occurring in spring. In contrast, a considerable reduction in emission rates was found over the winter and summer periods.

Using the data presented in Figure 3, an average emission rate from the slurry store was determined over the period of the measurements of 0.7 g NH_3-N m^{-2} day^{-1}. This compares with a value of 2.17 g m^{-2} day^{-1} that is applied for crusted slurry stores in the UK Ammonia Emissions Inventory (Misselbrook et al., 2000), indicating that the contribution from weeping wall stores may be currently overestimated. It should be noted however that the emissions reported herein are within the range of data used by Misselbrook et al. (2000) to derive their emission estimate.

Similar results were determined for the other sites studied (not presented here), where average emission rates were found to range between 0.4 – 0.7 g m^{-2} day^{-1}. Recent results from chamber studies have also reported similar emission rates from crusted slurry lagoons, with an average emission estimate of 0.5 g m^{-2} day^{-1} (Smith et al., 2004), illustrating the reproducibility of the data presented herein.

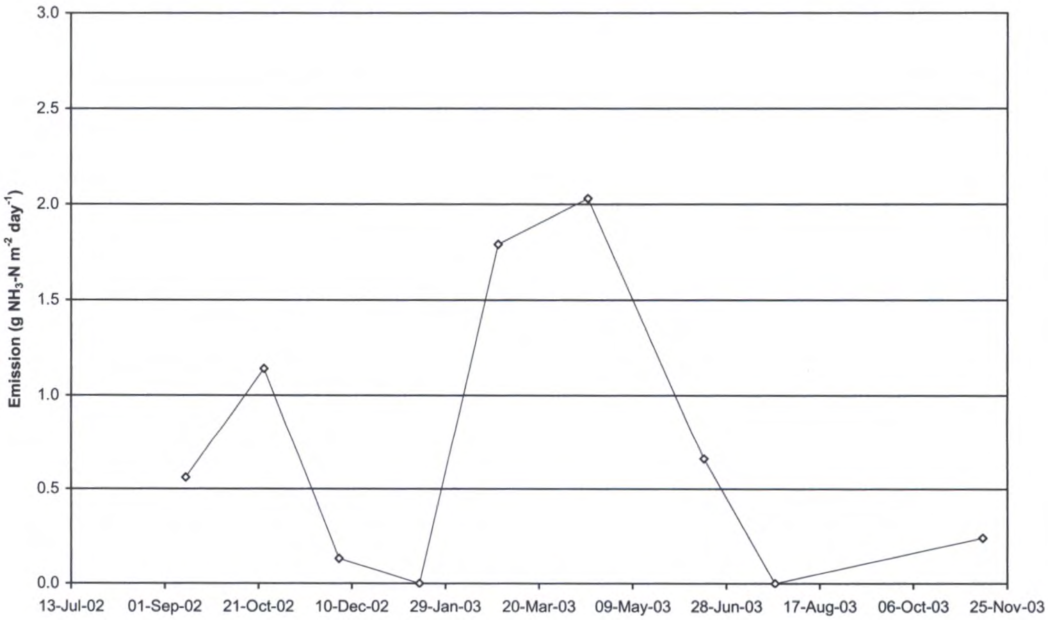

Figure 3. Timeseries of NH_3-N emissions from the slurry store.

Conclusions

A new in-situ method of deriving emission rates from farm waste stores, based on passive sampling of NH_3 concentrations in air and atmospheric dispersion modelling, was applied at four UK farms that used either weeping wall stores or earth banked lagoons with strainer boxes. The results from one of these farms that used a weeping wall store are reported herein. The measured air concentrations were found to be well described by a numerical modelling framework, with modelled vertical concentration profiles providing a good fit to the measured data both for the situation where air concentrations were derived from a single source (the slurry store) and where concentrations were combined from multiple on-farm sources (the slurry store and farm buildings). The diagnosed emission rates from the slurry store were found to vary throughout the year, with peak emissions occurring in spring of 2 g NH_3-N m^{-2} day^{-1}. Emissions averaged over the year were 0.7 g NH_3-N m^{-2} day^{-1} which, whilst within the range of data compiled for the UK Ammonia Emission Inventory, suggests that emissions from weeping wall stores may be currently overestimated.

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Agricultural Air Quality Modeling

Cumulative Dispersion and Deposition Modeling of Many CAFOs: RTI's Methodology for Ammonia Gas and PM Fine as Applied to the Smithfield Settlement Agreement

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Abstract

As part of the Smithfield Settlement Agreement, RTI developed and performed a modeling approach to predict the influence of ammonia emissions from multiple Concentrated Animal Feeding Operations (CAFOs) on watersheds and human health. RTI applied this modeling approach, using more than 2,000 swine operations in the eastern North Carolina covering five river basins. This paper presents the approach and results from that modeling effort and estimates gaseous ammonia emissions and fine particulate generation under baseline conditions and with simulated reductions. Emission factors were derived from the literature, and a variety of model units (e.g., lagoons) were used to accommodate differing capacities and types of swine operations. A U.S. EPA dispersion model was used to estimate ammonia concentrations and deposition rates from 12 model operations. A geographic information system (GIS) matched each of the 2,000 CAFOs to a model operation and mapped overlapping CAFO deposition zones of up to 50 km in radius each. The total nitrogen loads from swine facilities to the designated study area of five river basins were then calculated. The estimated concentration of ammonium salt fine particulates was estimated to assess the role of swine operations contributions to ambient fine particulate matter in the study area. Estimated impacts from swine operations of both gaseous ammonia and particulate ammonium are reported.

Introduction

The July 2000 agreement between the North Carolina Attorney General's Office and Smithfield Foods, Premium Standard Farms (subsequent), and their North Carolina subsidiaries called for the development of environmentally superior alternatives to lagoon and sprayfield swine waste management. The agreement called for an economic feasibility study of candidate technologies. The study included an evaluation of both the costs to the industry and consumers and the benefits to society of the environmental improvements associated with adopting candidate technologies. The environmental benefits assessment consisted of a multimedia environmental analysis of releases from 2,295 swine operations' housing, lagoon, and sprayfield waste management systems (RTI, 2003; see Figure 1). The release of gaseous ammonia was studied for its transport, deposition, and impact as a gas but also its conversion to ammonium salt fine particulates, transport, and human health impact from inhalation exposure.

To attain the goals of the multimedia environmental analysis, RTI designed an integrated modeling framework, combining North Carolina's swine concentrated animal feeding operation (CAFO) inventory, growth-stage-specific emission factors, and air and surface water quality modeling.

This paper describes the modeling framework developed by RTI to predict the dispersion and deposition of ammonia air emissions from more than 2,000 swine CAFOs in the Smithfield Settlement Agreement (SSA) study area. The methodology uses emission factors derived from peer-reviewed literature; employs a variety of model facilities to accommodate different capacities and types of swine CAFOs; uses an existing, proven dispersion-deposition model; and processes and interprets the outcome spatially using a geographic information system (GIS). The modeling framework is part of RTI's larger *Integrated Benefits Assessment Tool* (RTI, 2003). This tool evaluates the economic, environmental, and human health benefits of alternative CAFO waste management techniques.

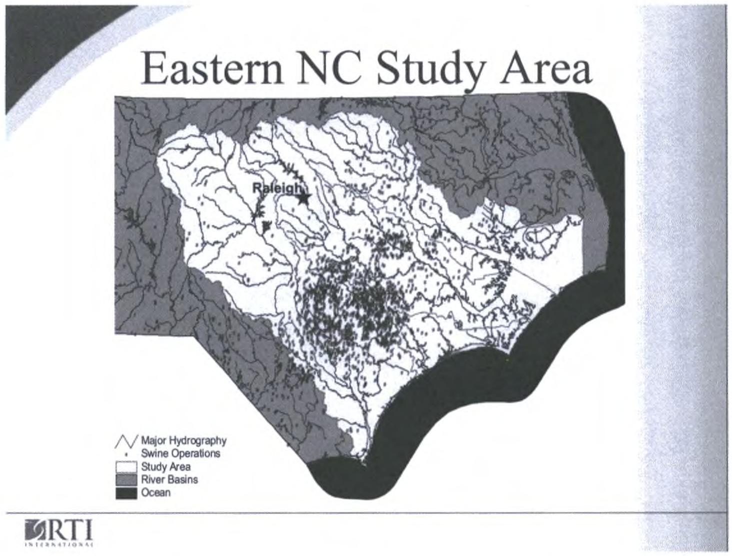

Figure 1. RTI studied the impacts of swine CAFO ammonia emissions on five major North Carolina river basins.

Method

RTI's approach required a number of data collection, modeling, and processing activities. Figure 2 depicts this multimedia approach as it was designed to estimate the environmental impact on watersheds of air emissions from multiple CAFOs. The following technical approach focuses on the air components.

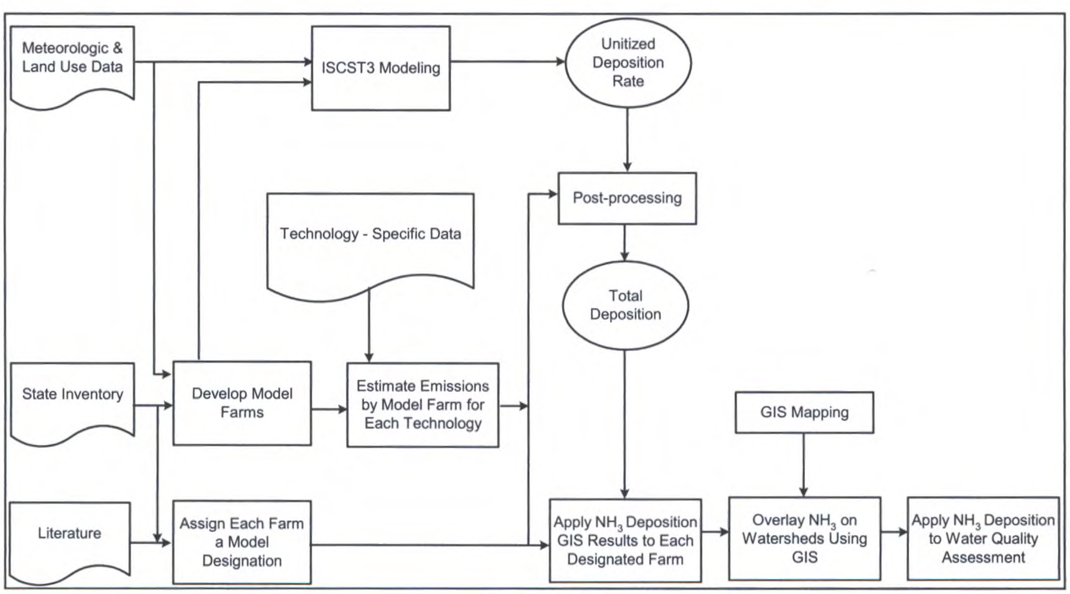

Figure 2. Ammonia atmospheric dispersion and deposition modeling approach

Data Collection and Model CAFO Development

RTI collected data from a variety of resources, including research studies, the literature, and other sources, to model ammonia emissions from swine CAFOs. We began with 2,295 swine CAFOs in five North Carolina river basins (Tar-Pamlico, Neuse, Cape Fear, White Oak, and New River). The State of North Carolina compiled a survey database that profiles these and other swine operations (N.C. DENR, 2002). This database served as the starting point for profiling the 2,295 CAFOs.

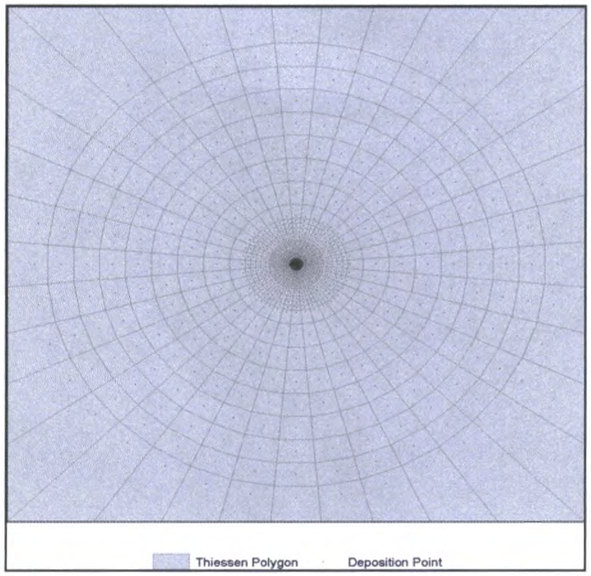

Figure 3. Deposition grid applied to each CAFO to model ammonia dispersion and deposition in a 50 km radius

RTI used each CAFO's geographic coordinates to assign these operations to one of three meteorological regions in the state. Each region possessed its own historical data on parameters such as wind speed, temperature, and land use. North Carolina's CAFO survey database also enabled RTI to assign each CAFO to one of four acreage categories (based on the swine population's steady-state live weight). This combination of three meteorological regions and four acreage categories yielded 12 model CAFOs. RTI next assigned each recorded CAFO in the study area to one of five growth stages (e.g., wean to feed) or span of growth stages (where multiple stages were raised on one CAFO). This assignment, in tandem with the 12 model CAFOs, resulted in each CAFO being characterized as one of 60 operating scenarios. RTI assumed that a model swine facility contains three ammonia-emitting sources: confinement houses with their waste collection systems, waste lagoon(s), and sprayfields. Given the variability in facility dimensions and layout, we chose to treat each model facility as a single-area emission source. The emission factor for the area source was a "composite" of the house, lagoon, and sprayfield emission factors. This composite approach was designed to allow emission factors of each source to be adjusted as source-specific control techniques are examined.

Emission factors were not available for each emission source in combination with each of the five growth stage categories. Therefore, RTI was required to use what emissions research was available from the literature (RTI, 2003) in order to derive emission factors based on steady-state live weight and the distribution of swine per CAFO by growth stage category. However, recent RTI advances to support EPA's development of WATER9 for AFOs could enable better, mass-balanced estimates of emissions. (Refer to "WATER9 – An Air Emission Model for Animal Feeding Operations – Software for Both Field Agents and Comprehensive Scientific Research" (Deerhake, Allen, and Nizich) in these proceedings.)

Unitized Deposition Modeling by Model CAFO

It is understood that gaseous ammonia deposits faster (nearer field) than other ammonia species such as fine particulate ammonium sulfate that may form following ammonia's release to the atmosphere. The dispersion/deposition of ammonia is characterized using a number of runs for model CAFOs where the emission rate is 1 mg per second per square meter (a unitized emission rate) and the chemical composition is 100% ammonia. As Figure 3 shows, this unitized ammonia emission was modeled to predict its dispersion and deposition in a 50 km radius.

RTI used the Industrial Source Complex Short-Term Model, Version 3 (ISCST3), version 02035, to model the dispersion as well as wet and dry deposition of ammonia (U.S. EPA, 1995). This model is an EPA-approved model that predicts atmospheric dispersion and deposition of specific chemical species up to about 50 km from the source. RTI performed "unitized" dispersion and deposition ISC modeling for the 12 model CAFOs (3 locations' meteorology x 4 CAFO sizes) which resulted in the dry and wet ammonia deposition estimates presented in Table 1. These loading rates were then applied to appropriate swine operations from the state inventory.

Table 1. ISCST3 Unitized Ammonia Deposition Results for Model CAFOs

12 Model CAFOs		Total Deposition (Mg/yr)	Dry Deposition (Mg/yr)	Wet Deposition (Mg/yr)
NWS Station (represents met. Region)	**CAFO Acreage**			
Norfolk, VA	500	33,522	33,177	345
Raleigh	500	39,956	39,566	390
Wilmington	500	36,846	36,360	486
Norfolk, VA	260	17,531	17,390	142
Raleigh	260	20,916	20,756	161
Wilmington	260	19,286	19,083	202
Norfolk, VA	100	6,823	6,784	39
Raleigh	100	8,155	8,110	44
Wilmington	100	7,517	7,460	57
Norfolk, VA	50	3,457	3,441	16
Raleigh	50	4,138	4,120	18
Wilmington	50	3,813	3,790	23

NWS = National Weather Service

Post-Processing of Unitized Dispersion and Deposition Modeling Results

Following unitized ammonia dispersion and deposition modeling, a post-processing system was run that portrays each CAFO in the study area as one of 12 model CAFOs with one of five operating scenarios. This approach predicted how much ammonia is emitted, dispersed, and deposited on and around each swine operation (up to 50 km). To develop a site-specific ammonia emission rate, annual average emission factors for each source (e.g., lagoon) were gleaned from the literature (RTI, 2003), accounting for animal type, growth stage, and technology when feasible.

RTI's modeling framework matched each of the CAFOs (2,295 CAFOs in the North Carolina study) to a model operation's predicted deposition loading field, in combination with the steady-state live weight reported by each CAFO operator in the North Carolina database. Once each CAFO was mapped, we then mapped overlapping CAFOs' ammonia deposition zones. Each CAFO's deposition loading field was predicted by creating Theissen polygons around each of 1,087 modeled deposition points in the 50 km radius.

Since each CAFO in the study area had geographic coordinates, RTI placed the appropriate deposition pattern on the ground (georeferenced) for each CAFO, creating a coverage of almost 2.5 million points. RTI then overlaid this large point coverage with the hydrologic unit (a.k.a. HUC) boundaries for those 14-digit HUCs in the study area. RTI calculated summary statistics for total, dry, and wet deposition on a HUC-by-HUC basis. As shown in Figure 4, deposited ammonia emissions can play an important role in nitrogen loading to surface waters from individual CAFOs and especially from multiple CAFOS. Note the contrast of modeled deposition receptor points from a single CAFO versus the cumulative effect of a cluster of CAFOs. Figure 4's first illustration represents the field of deposition for a single HUC from one CAFO. RTI modeling predicts a 3,000-head, southeastern U.S. CAFO can deposit approximately 58% of its estimated annual average ammonia emissions within 50 km. Warmer summer climates can increase emissions and, in turn, increase local deposition loading. Figure 4's second illustration represents the fields of deposition for multiple CAFOs located across at least eight contiguous HUCs yet whose 50 km radii overlap. Assuming that 58% of the total estimated ammonia emitted from one 3,000-head CAFO deposits within 50 km of its center, the deposition and subsequent stream loading impacts of multiple CAFOs can be significant both within their base HUC and in adjoining HUCs.

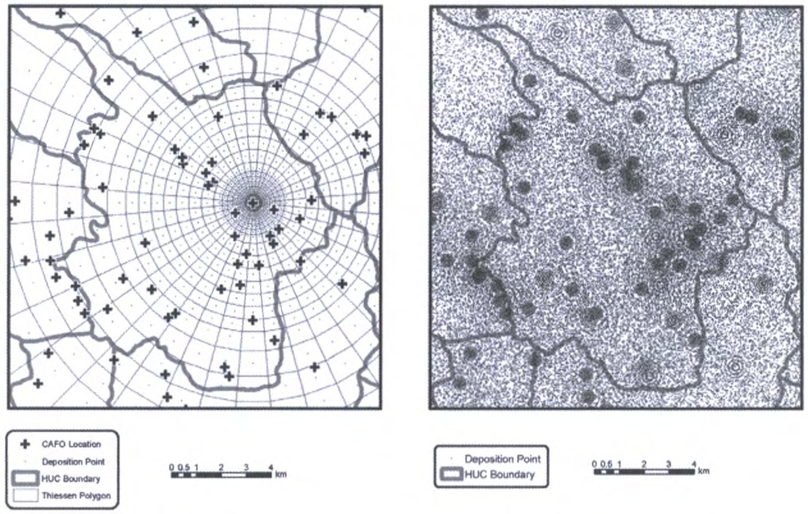

Figure 4. Comparative figures of a generic radial 50 km deposition grid centered on one swine CAFO (left) and the estimated cumulative deposition contribution from all swine CAFOs in the area (right)

GIS mapping of deposition offers a variety of options for study. In addition to hydrologic units, deposition can be mapped by airshed, county or state, climatology, census zone, terrain, industrial zone, natural and cultivated resources, or soil and water conservation districts. Such georefencing may aid in site selection, evaluation of new waste management technologies, crop planning, and development of site-specific nutrient management plans, for example.

Ammonia-to-Ammonium Conversion

Ammonia reacts with sulfuric, nitric, and hydrochloric acid gases ***n*** to form aerosols such as ammonium sulfate, ammonium bisulfate, ammonium nitrate, and ammonium chloride. Ammonium salts formed by these reactions can exist as solid particles or liquid droplets, depending on the amount of water vapor in the atmosphere. Ammonia preferentially reacts with sulfuric acid (H_2SO_4) to form ammonium bisulfate (NH_4HSO_4) and ammonium sulfate ($[NH_4]_2SO_4$) through equations 1 and 2:

$$NH_3(gas) + H_2SO_4(liquid) \rightarrow NH_4HSO_4(liquid) \tag{1}$$

$$NH_3(gas) + NH_4HSO_4(aq) \rightarrow (NH_4)_2SO_4(solid)or(liquid) \tag{2}$$

Ammonia can also undergo an equilibrium reaction with gas-phase nitric acid (HNO_3) in the atmosphere to form ammonium nitrate (NH_4NO_3) as shown in equation 3:

$$NH_3(gas) + HNO_3(gas) \leftrightarrow NH_4NO_3(solid)or(liquid) \tag{3}$$

Because sulfuric acid has a low vapor pressure, it seldom exists as a gas when water vapor is available. Nitric acid is much more volatile, so particulate nitrate is believed to be lower in concentration than sulfate (Seinfeld and Pandis, 1998; Pacyna and Benson, 1996). However, particulate nitrate can dominate in PM_{fine} when sulfate is limited.

In Search of Ammonia-to-Ammonium Reaction Rates

The dynamics of ammonium formation are far from understood with the kinetics being too complex to arrive at a conversion factor for the study. Factors such as local meteorology and the availability of sulfates and nitrates in the atmosphere complicate the prediction of ammonium conversion. As a result, RTI turned to the literature to find any monitoring studies of ammonium and ammonia. Robarge et al. (2002) performed a monitoring study in a 5 km radius of multiple swine operations. The measurements were taken at the Clinton Horticultural Crops Research Station located approximately 5 km north and east of Clinton, N.C. Three swine operations are located between 1.5 and 3.2 km east-northeast and east-southeast of the site. Three additional swine operations are located 3.2 to 5 km northwest of the site. Robarge's measurements represent ambient conditions. The fraction of ammonia in total ammonia (NH_x) from Robarge's ambient measurements was greater than 70%. Given the absence of ammonia-to-ammonium conversion factors, RTI decided to apply Robarge's ambient monitoring findings which represented ambient conditions and assume that ammonia gas (NH_3) was 70% of the total ammonia (NH_x) and that ammonium salt (NH_4^+) PM_{fine} was 30% of the total ammonia in this analysis. With this 30% value, we calculated a county annual average ammonium salt (PM_{fine}) concentration by multiplying the county average of ambient ammonia gas by 30%. This exercise estimated only ammonium salt PM_{fine} resulting from swine operation emissions. It did not estimate background ambient $PM_{fine,}$ resulting from other emission sources.

Results and Discussion

Gaseous Ammonia

Results of baseline modeling showed that when accounting for deposition only in the 50 km radius of each CAFO, about 34,000 Mg (over 37,000 short tons) of 2,295 CAFOs' ammonia emissions were deposited in the study area in one year (RTI, 2003). To reiterate, these values represent the sum of each of the 2,295 CAFO's deposition within a 50 km radius of each CAFO, excluding any ammonia that transports and deposits beyond the 50 km radius. As mentioned before, this modeling exercise presumes ammonia remains gaseous throughout its atmospheric dispersion within a 50 km radius. However, a fraction of ammonia may convert to an ammonium salt that is an aerosol (fine particulate). The 37,000 short tons of ammonia estimated by RTI compares well to the State of North Carolina's 1995 estimated statewide emissions (all sources) of 77,700 tons.

Figure 5 depicts the range of deposition by HUC. (There are more than 500 HUCs (or small watersheds) in the study area.) The greatest deposition occurs in counties with the greatest density of swine CAFOs. The 10 HUCs estimated to have the greatest ammonia deposition are all within the Cape Fear River basin. The 10 HUCs total about 5,700 Mg/yr of ammonia deposition, which is about 17% of the study area's total annual ammonia deposition.

The ranking of HUCs is a function of both the animal density and the density and proximity of CAFOs to one another. As Figure 5 demonstrates, when CAFOs are located near one another, the ammonia deposition for each CAFO's 50 km radius can overlap with another CAFO's, thus multiplying the ammonia deposition/loading to HUCs.

RTI's analysis of impacts to water quality from swine operations comprehensively addressed inputs of nitrogen and phosphorus from numerous sources and quantified swine sources independently from other sources. RTI developed methods to estimate watershed inputs for swine facilities both from runoff of nitrogen and phosphorus from land application of waste as well as deposition of ammonia. Additionally, we estimated municipal and industrial wastewater sources and runoff of both nitrogen and phosphorus and deposition of nitrogen from nonswine, nonpoint sources .

To address nitrogen inputs from swine facilities via runoff, we derived land application rates (kg N/yr) for each facility using facility inventory data received from the North Carolina Division of Water Quality and methods developed for the ammonia emissions and deposition assessment. To address atmospheric nitrogen inputs from swine facilities, we compiled output from the ammonia emissions and deposition modeling for each 14-digit watershed. Deposition rates were assumed to be uniformly distributed within each watershed. Direct deposition onto water surfaces for freshwater was calculated based on the watershed's deposition rate and the area in the watershed identified as water based on land cover data.

Indirect deposition was calculated based on an assumption of delivery (pass through) rates for different land cover categories.

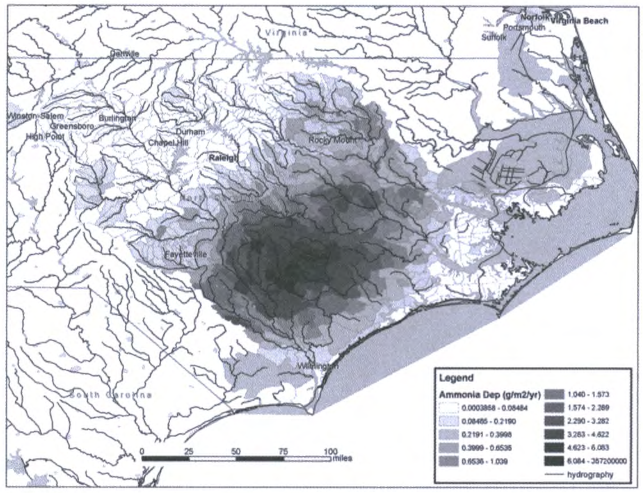

Figure 5. Modeled estimated ammonia deposition by HUC from 2,295 swine CAFOs in Eastern North Carolina

For nonswine atmospheric nitrogen inputs, we developed a method using available wet and dry deposition data from 1996 to 2000 from ambient monitoring sites, along with spatial interpolation. Data were compiled for reduced, oxidized, and organic nitrogen within or near the study area. We calculated total deposition as the sum of wet and dry deposition. We intentionally excluded data from several stations located relatively close to more concentrated swine activity.

Our results indicated that 62 percent of the swine delivery of nitrogen to free-flowing surface waters was estimated to occur via direct runoff, with the remainder through atmospheric deposition of ammonia upstream from estuarine waters. For the entire study area, swine facilities were predicted to contribute 28 percent of the atmospheric nitrogen inputs. Ammonia transported from swine facilities that deposits directly onto estuarine waters is estimated to deposit at rates of 0.01 to 0.04 kg/ha/yr for the different estuaries considered (Pamlico, Neuse, White Oak, and New), accounting for between 0.01 to 0.1 percent of the total estuarine loading. The rate of ammonia direct deposition to estuaries is estimated to be less than 1 percent of the estimated total nitrogen deposition rate accounting for nonswine sources, suggesting that local (indirect) ammonia gas transport and deposition is a more serious concern than ammonia transport directly to estuary waters. However, we cannot draw inferences about ammonium transport from swine facilities to estuaries because we did not attempt to model transport and deposition into the water system of swine waste as ammonium particles.

Ammonium Fine Particulates

RTI's assessment of benefits achieved from reducing CAFO ammonia emissions indicated that PM_{fine} plays a significant role. Table 3 lists the RTI atmospheric modeling exercise's estimated baseline ambient ammonium salt PM_{fine} concentrations attributable to swine operations for each of the counties in the eastern North Carolina study area. These data served as input to RTI's integrated benefits analysis of alternative waste management technologies.

The baseline average (i.e., presuming lagoon and sprayfield waste management) ammonium salt PM_{fine} concentration inside the five-river-basin study area was estimated as 0.592 $\mu g/m^3$. The county with the maximum estimated annual average ammonium salt PM_{fine} concentration was Duplin County at 3.576 $\mu g/m^3$. Duplin County's weighted annual arithmetic mean ambient concentration from all sources was 12.6 $\mu g/m^3$ (N.C. DENR, 2003). (For a point of reference, North Carolina's $PM_{2.5}$ (or PM_{fine}) standard is 15.0 $\mu g/m^3$ [NCAC 2D.410(a)]).

Table 3. Highlights of Modeled Estimated Average Annual Ambient Ammonium (NH_4^+) Concentrations ($\mu g/m^3$) for Selected Counties in the Study Area (descending order)

North Carolina Study Area County	Estimated Average Annual Ambient Ammonium (NH_4^+) Concentration ($\mu g/m^3$)	Number of Swine Operations
Duplin	3.576	514
Sampson	3.115	453
Greene	2.071	105
Wayne	1.915	146
Lenoir	1.551	78
Bladen	1.221	133
Jones	0.915	47
Johnston	0.652	61
Moore	0.081	9
New Hanover	0.051	0
Tyrrell	0.049	5
Wake	0.047	3
Montgomery	0.039	4
Durham	0.004	0
Dare	0.001	0

For ammonium salts PM_{fine}, a traditional EPA benefits estimation model was used to model the benefits of reducing PM_{fine} from swine CAFOs (U.S. EPA, 2003). EPA's model estimates reductions in annual incidence of specific health outcomes associated with reductions in ambient PM_{fine} concentrations.

RTI evaluated scenarios reducing total CAFO-modeled PM_{fine} by 10% and 50%. The largest estimated incidence reductions were for acute conditions, but the total value of these avoided cases was relatively small compared to the value of estimated reductions in premature deaths (on the order of tens to hundreds of millions of dollars). (NOTE: Although EPA's benefits model has been reviewed extensively, the selection and application of concentration reduction functions and valuation methods have uncertainties associated with them, e.g., the value assigned to mortality ($6-7 million per premature death avoided) (RTI, 2003)

Conclusions

RTI studied the relationship of CAFO ammonia emissions to increased nutrient loadings to surface waters and increased ambient air concentrations of PM_{fine} as part of the Smithfield Settlement Agreement research. The study necessitated a multimedia modeling framework to predict the fate and transport of ammonia. RTI created the framework to enable industry and government decision makers to evaluate the benefits of alternative waste management techniques. RTI applied its multimedia model to more than 2,000 swine CAFOs in eastern North Carolina. This model can be enhanced by applying RTI's recent improvements to the emission model WATER9 for AFOs.

Using dispersion and deposition models for ammonia gas in tandem with GIS and water quality models, RTI estimated the cumulative loadings to more than 500 small watersheds within five major North Carolina river basins. Results of baseline modeling showed about 34,000 Mg (37,000 short tons) of gaseous ammonia from 2,295 swine CAFOs deposited in the SSA study area in one year. This only accounts for deposition within a 50 km radius of each CAFO. These estimates were found to be consistent with state projections and research monitoring studies. Subsequent water quality modeling predicted that the rate of gaseous ammonia's direct deposition to estuaries is estimated to be less than 1 percent of the estimated total nitrogen deposition rate accounting for nonswine sources, suggesting that local (indirect) ammonia gas transport and deposition is a more serious concern than ammonia transport directly to estuary waters. However, we cannot draw inferences about ammonium transport from swine facilities to estuaries because we did not attempt to model transport and deposition into the water system of swine waste as ammonium particles.

RTI did estimate the potential for swine ammonia emissions to convert to ammonium fine particulates. These estimates indicate that a limited number of North Carolina counties with large populations of swine may contribute up to double-figure percentages of total monitored ambient PM_{fine} concentrations. An RTI benefits analysis indicated that substantial benefits ($3-4 million for each 1 percent of ammonia emissions reduced) can be achieved in terms of human health if ammonia emissions (as a precursor to PM_{fine}) can be reduced.

Efforts to reduce ammonia emissions will subsequently reduce ammonium and PM_{fine} formation, and dramatically increase the value of health benefits of the area. To enhance the understanding of the role of ammonium salts in ambient PM_{fine} concentrations and the role of CAFOs in the health effects from exposure to CAFO-generated PM_{fine}, additional research is needed on ammonium formation kinetics. Equally important is the need for more ambient monitoring studies that can:

- validate ammonium formation theory,
- build a spatial database of ambient observations to determine the role of climate and availability of sulfates, nitrates, and chlorides for salt formation, and
- determine the appropriateness of using ratios of spatially based ambient ammonia to ammonium and generate improved methods of estimation.

It was concluded that implementing waste management technologies to reduce both the CAFO-based ammonia deposition to waters and public exposure to ammonium salt PM_{fine} can result in significant benefits, especially the health benefits of PM_{fine} reduction.

Acknowledgements

RTI would like to thank North Carolina State University's Animal and Poultry Waste Management Center (Mike Williams, Ph.D., Director) for contracting RTI to perform the environmental and economic assessment (RTI, 2003). RTI also thanks its economists – Dr. George Van Houtven and Dr. Brian Murray – for inviting RTI's Environment, Health and Safety Division to collaborate on this study.

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Developing Manure-DNDC: Building a Process Based Biogeochemical Tool for Quantifying NH3, CH4 and N2O Emissions from California Dairies

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Abstract

Assessing the environmental impact of manure management is difficult due to high variability in the quality and quantity of animal waste, and in the numerous factors affecting the biogeochemical transformations of manure during storage, treatment and field application. There is an urgent need for scientifically sound, mass balance based, process models for quantifying air emissions from animal feeding operations. Measurement programs are essential, but must be supplemented by process-oriented modeling that incorporates mass balance constraints to extrapolate in both space and time (NRC, 2003). The time is right for moving beyond the inadequate emission factor approach by developing process based models for quantifying air emissions from animal feeding operations.

The dynamics of CH4, N2O and NH3 production/consumption is always controlled by several biochemical and geochemical reactions, namely decomposition, hydrolysis, nitrification, denitrification, ammonium adsorption, chemical equilibriums of ammonium/ammonia, and gas diffusion. These biogeochemical processes are currently simulated in our existing model called DeNitricifation-DeComposition, or DNDC. By tracking C and N dynamics under both aerobic and anaerobic conditions, these processes have been successful in simulating soil C sequestration and trace gas emissions and are well suited for estimating air emissions associated with manure production, storage, treatment and land application.

The current DNDC model has detailed processes for quantifying CH4, N2O and NH3 emissions from agroecosystems with fertilizer/manure application or animal grazing conditions but lacks algorithms for specifying fluxes under drylot, housing and storage conditions. We are now extending DNDC's applications by integrating the fundamental biogeochemical processes with housing and storage management practices. The new developments for our process-based, mass balance approach include (1) integration of detailed biogeochemical processes into the GHG emissions and NH3 volatilization under drylot, housing or storage conditions; (2) characterization of environmental factors under drylot, housing or storage conditions; and (3) characterization of quantity and quality of dairy waste. This paper provides an overview of our on-going project to develop GIS databases for California dairies, perform a field measurement program and perform model refinements to create a tool for quantifying air emissions from California dairies.

Background

Need for Process-based Biogeochemical Models: Accurate assessment of air emissions from dairies with emission factors is difficult due to: (1) high variability in the quality and quantity of animal waste, and (2) the numerous factors affecting the biogeochemical transformations of manure during collection, storage and field application. Measurement programs are essential but expensive and thus have not been extensively implemented. Therefore, process-based models that incorporate mass balance constraints are needed to extrapolate air emissions in both space and time (NRC, 2003). EPA has not yet developed such a model, relying instead on a simplified methodology for estimating air emissions from individual dairies, using "model" farms based on typical animal confinement, manure collection, solid separation, manure storage and stabilization, and techniques for land application of manure (EPA 2002).

Although it is well known that constant emission factors are not effective for quantifying GHG, ammonia, and ROG emissions from animal feeding operations (NRC 2003), managers and regulators generally lack access to tools that are both scientifically sound, capture the biogeochemical processes that impact emissions, and are relatively easy to use. There are a number of advantages to developing process-based models of element transformations and emissions from the *combined* components (animal feedlot, manure storage and handling, land application of manure) of animal feeding operations:

- Dynamic, process-based models, developed from laboratory and field studies, **do not rely on constant emission factors**. They assess the impact on emission factors of varying conditions (e.g., climate, storage facility, soils). These models will continue to improve as more field studies are conducted and published, and they do not obviate the need for a strong measurement program.
- By **enforcing a mass balance** in the model (i.e., conservation of mass), the sum of all emission factors are constrained to be ≤ 100% of inputs. This is both good bookkeeping and essential for evaluating trade-offs in mitigation strategies.
- Full system analysis with dynamic, process-based models can inexpensively and efficiently **evaluate mitigation scenarios** under various conditions, and can help target mitigation toward facility component(s) and/or operation(s) that cause the greatest emissions.
- Simultaneously provide estimates of all emission for **comprehensive assessments** of mitigation efforts. For example, *efforts to reduce methane (e.g. enhance aerobic manure management) may result in increased nitrous oxide emissions that could more than offset gains from methane reductions and result in a net increase in total greenhouse gas emissions*. Therefore, ***well validated models are critical for comprehensive analyses that capture all emissions to air and water***.

The following is a brief description of our efforts in building GIS databases for characterizing soil, climate conditions, and locations of California dairies, a brief outline of our field measurement plans for obtaining calibration and validation data for our biogeochemical process model, and our Manure-DNDC modeling framework.

GIS Database Development

To run the emission models and create an easy to use emission modeling system, spatially explicit data on dairy locations, soils, climate and general agricultural land use are needed. Access to these GIS data layers will enable users to perform statewide, air district, county and even individual dairy emission simulations. We are building a tool to automatically access, retrieve and process CIMIS (California Irrigation Management Information System) climate station databases (daily min T, max T, precipitation, and solar radiation) into Manure-DNDC model format. Manure-DNDC requires, at a minimum, temperature, precipitation, relative humidity, and wind speed. N deposition data derived from the National Atmospheric Deposition Program (NADP) maps will be built into the system to estimate mean deposition rates at the sub-county scale. Soil characteristics will be obtained from the NRCS STATSGO (1:250,000) and SUSRGO (1:12,000 to 1:63,000) databases. Area weighted ranges on soil pH, bulk density, texture and SOC will be compiled on a dairy and individual field basis using land use boundary derived from aerial photography (see insert in figure 1). These crop and dairy location maps have been obtained from California Department of Water Resources (DWR) land use products. In addition, agricultural census data on livestock counts will be incorporated into the GIS.

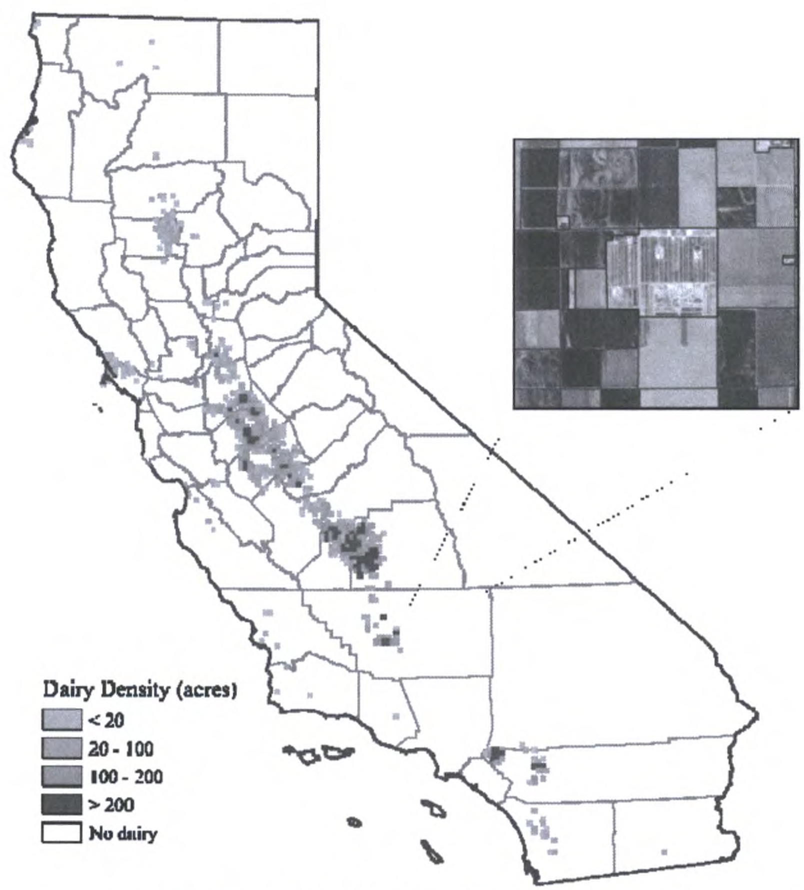
For the majority of the counties in California the spatial extents of dairies will be derived from the California DWR GIS database. Dairy cow density was calculated for each county my dividing total dairy cows in each county derived from the NASS database by the total area in dairies as defined by the DWR database. Dairy cows were disrupted across the DWR dairy location polygons by multiplying the county dairy cow density by the DWR dairy location polygon areas. A map of dairy cows distributed across the DWR database is given in Figure 1.

A few counties are still missing or only partially covered in the DWR database including Sonoma, San Bernardino and Riverside Counties. All three counties have moderate to large dairy operations. In the case of these counties additional datasets will be used to locate dairies including a dairy facility street address database provided by the South Coast Air Quality Management District for San Bernardino and Riverside Counties. USGS Digital Line Graph (DLG) files at a scale of 1:100,000 will be employed to address match the dairy locations to available street addresses.

Field Measurement Program

Model calibration and validation is critical and requires high quality field data. Field data, funded by this and other projects will be collected at several operating dairies and the UC Davis Emission Testing Facility:

- CH_4 and NH_3 measurements will be collected using active denuder/filter packs and two tunable diode laser systems.
- An open path FTIR system and canister samples will be used to collect N_2O emissions following land application of manure.
- The UC Davis emission testing facilities will be used to measure CH_4, NH_3, N_2O emissions under a wide range of animal housing and manure treatment/storage facilities using a INNOVA 1412 Photoacoustic Field Gas-Monitor.



Data Source: California Department of Water Resources land use survey database (http://www.landwateruse.water.ca.gov)

Figure 1. Dairy cow distribution in California

DeNitrification-DeComposition Model (DNDC) Refinement

Over the past decade, multi-agency support from EPA, NASA, USDA and NSF has guided the development, testing, and application of a research biogeochemical model of nitrogen (N) and carbon (C) cycling in soils. The process-oriented computer simulation model, Denitrification-Decomposition (DNDC), was developed based on the biogeochemical concepts for predicting soil biogeochemistry (Li et al. 1992, 1994, 1996; Li 2000). DNDC consists of two components. The first component (see figure 2), consisting of the soil climate, crop growth and decomposition sub-models, predicts soil temperature, moisture, pH, redox potential (Eh) and substrate concentration profiles (e.g. ammonium, nitrate, dissolved organic carbon) based on ecological drivers (e.g., climate, soil, vegetation and anthropogenic activity). The second component, consisting of the nitrification, denitrification and fermentation sub-models, predicts nitric oxide (NO), nitrous oxide (N2O), methane (CH4) and ammonia (NH3) fluxes based on the environmental variables in the soil. Classical laws of physics, chemistry and biology, and empirical equations generated from laboratory observations, were used in the model to parameterize each specific reaction. The entire model forms a bridge between basic ecological drivers including management of agro-ecological systems, and water, carbon, and nitrogen cycles. DNDC utilizes GIS databases with spatially and temporally differentiated information on climate, soil, vegetation and farming practices for local, regional and national scale analyses.

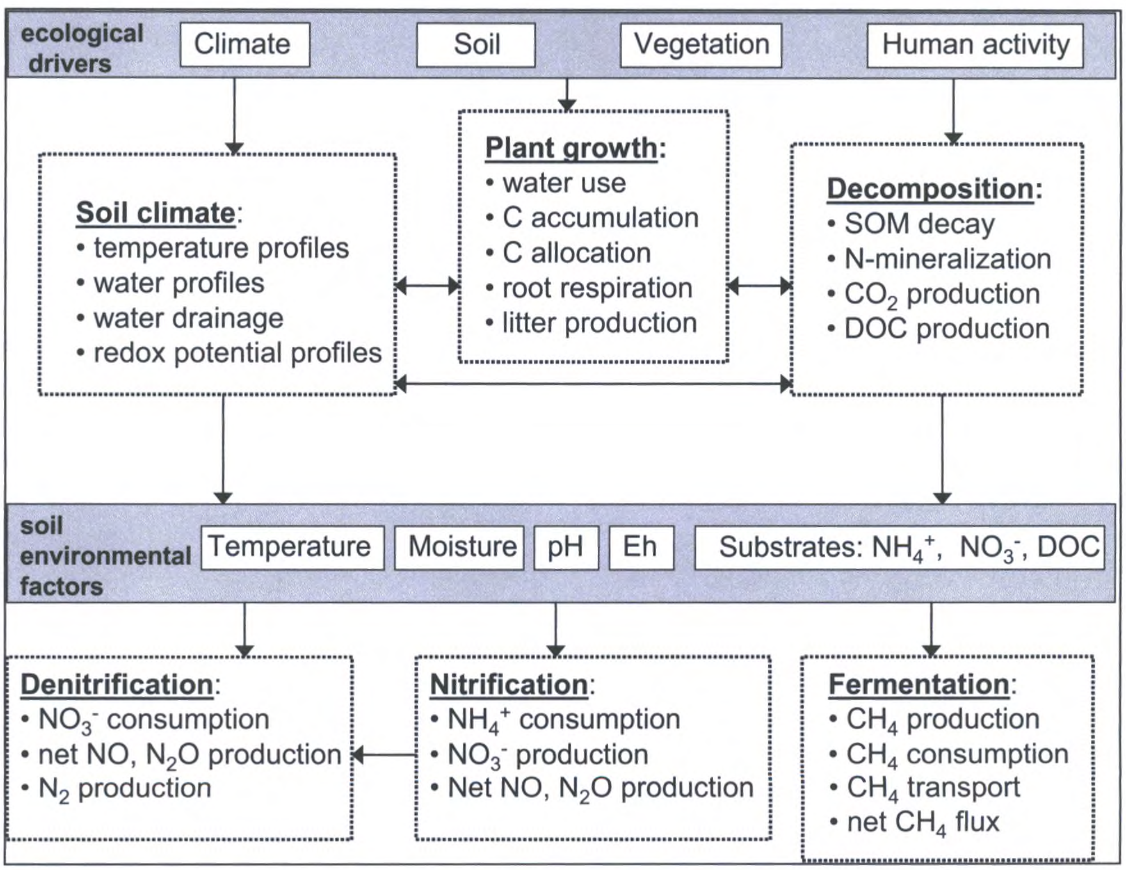


Figure 2. DNDC Model Framework

The core of DNDC is a soil biogeochemical model, which can be linked to vegetation models to predict carbon sequestration and nitrogen cycling for different ecosystems. DNDC has been linked to a crop model (Zhang et al. 2002, Li et al. 2004) to simulate crop growth, soil organic carbon (SOC) dynamics and emissions of dinitrogen (N2) and several trace gases including N2O, NO, NH3 and CH4 from both upland and wetland agricultural ecosystems. DNDC is a unique process-based biogeochemical model because it

(1) simulates both aerobic and anaerobic conditions, (2) tracks redox potential (Eh), (3) can provide a relatively complete suite of nutrient releases to air and water, including emissions of ammonia, greenhouse gases and nitrate leaching, and (4) contains tools for examining sensitivity and uncertainties in emission estimates. These capabilities are critical for quantifying whole farm emissions from California dairies. This model has been independently tested and validated by many researchers and under a wide range of conditions worldwide and now is utilized for national trace gas inventory studies in the U.S., Canada, the U.K., Germany, Italy, New Zealand, China, India, Japan, Thailand and the Philippines. The extensive validation and applications worldwide indicate that the fundamental processes embedded in DNDC have provided a sound basis for modeling C and N dynamics across a broad range of climatic zones, soil types and management regimes.

Example Modeling Framework for Estimating Ammonia Emissions from Dairies

Ammonia (NH3) emissions are a common phenomenon in terrestrial ecosystems although the flux magnitudes vary greatly. High NH3 fluxes have been observed in many agro-ecosystems due to the N cycling enhanced by either livestock husbandry or manure/fertilizer application in farmlands. Animal excretion under confinement or grazing conditions, manure storage, and manure/fertilizer application is a significant source of N into the agroecosystems. For example, the N existing in form of urea in animal urine can be hydrolyzed by urease in the soils at feeding lot, grazed pasture, or storage stand to form ammonium (NH4+). NH4+ can convert to NH3 through several chemical reactions driven by the equilibriums between NH4+, NH3 in liquid phase (NH3(l)) and NH3 in gas phase (NH3(g)). This conversion usually takes place in several hours or several days. The N existing in solid organic matter in animal dung will undergo decomposition (or mineralization) first, which is the major process converting the organic N into inorganic NH4+, before the N can join the soil NH4+ pool to continue its transformation to NH3. This conversion usually takes several weeks or months. Both of the fast and slow paths are important to NH3 emissions from agricultural sources. The fast path usually constitutes high, episodic NH3 fluxes following animal excretion, manure amendment or fertilizer application; and the slow path supports the consistent "background" NH3 emissions at broad scale resulting from decomposition of organic matter from manure or crop residue. However, no matter through which path, the dynamics of NH3 production is always controlled by several biochemical or geochemical reactions, namely decomposition, hydrolysis, nitrification, denitrification, ammonium adsorption, chemical equilibriums of ammonium, and ammonia gas diffusion. These fundamental reactions exist in DNDC to quantify both the spatial and temporal variability and non-linear nature of NH3 volatilization from dairies.

The entire process of elemental transformations from fresh animal wastes to inorganic nutrients dispersing in the air, soil or water is controlled by a suite of above-listed reactions. As soon as fresh waste is excreted from animals, it will begin to undergo a series of biochemical and geochemical processes including decomposition, ammonification, nitrification, denitrification, ammonia volatilization, and leaching. Regardless of location (e.g. freestalls, dylots, storage sites, or crop fields), rates of these processes are controlled by environmental factors, including: gravity, radiation, temperature, moisture, pH, redox potential (Eh), and substrate concentration gradients. Quantifying how these environmental factors affect the biogeochemical processes, as well as how the primary drivers (e.g., climate, soil, vegetation and management) affect environmental factors, is our core task for modeling the life cycle of manure at dairies in California. Our modeling approach and framework for estimating NH3 emissions, as an example, from dairies is presented in Figure 3. This approach and the DNDC general modeling framework shown in Figure 2 and as outlined in Table 1 is currently being implemented to create the Manure-DNDC dairy tool for California.

	Manure Production	Manure Storage/Processing	Manure Application
Environmental factors	Air temperature Precipitation Wind speed/direction	Air temperature Precipitation Wind speed/direction	Meteorological data Soil properties Vegetation type
Management factors	Manure quantity and quality Freestall, exercise pens, drylots Temperature Ventilation Duration before removal Removal technique (flushing, scrape)	Temperature Moisture Manure texture Additions Duration before land application	Crop type/rotation Tillage depth Manure application rate Manure C&N content Other fertilization Irrigation Weeding Grazing
Model Simulations	Manure temp., moisture, pH, C&N content Decomposition Denitrification NH3 volatilization N2O, NO, N2, CH4 emissions N & C leaching	Manure temp., moisture, pH, C&N content Decomposition Denitrification NH3 volatilization H2S, N2O, NO, N2, CH4 emissions N & C leaching	Manure temp., moisture, pH, C&N content Decomposition Denitrification N uptake by plants NH3 volatilization N2O, NO, N2, CH4 emissions N & C leaching

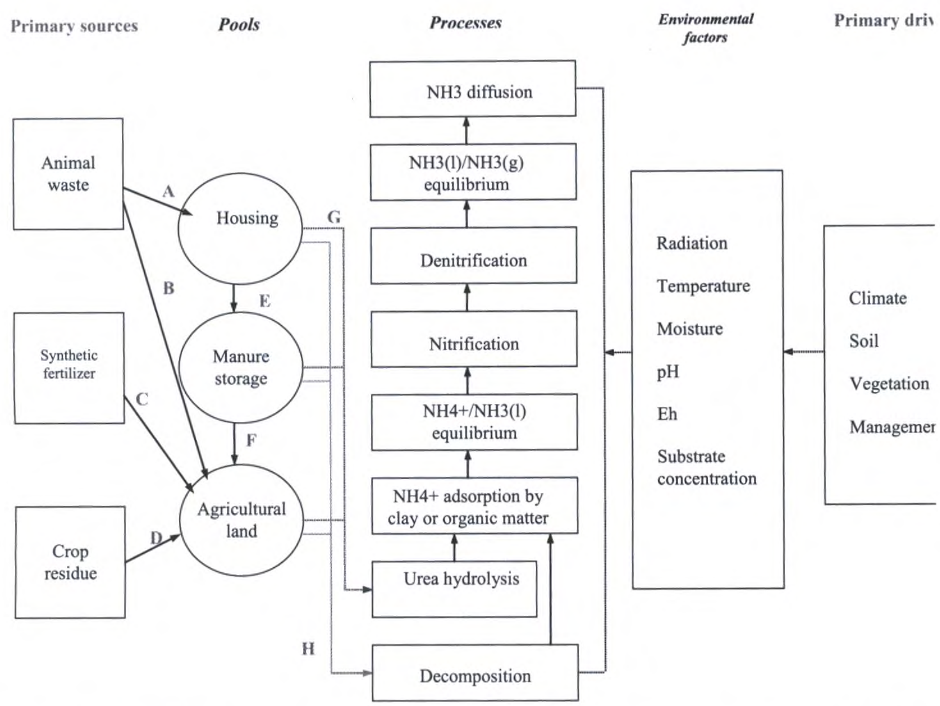


Figure 3. Framework of a process-based ammonia model for agricultural sources. A- Animal housing; B- Grazing; C- Fertilization; D- Crop residue incorporation; E- Manure storage; F- Manure application; G- Dissolved N (e.g., urea etc.) in liquid phase; H- Solid manure or soil organic matter.

Expected Results

From this project we anticipate the following results:

- Field measurements of CH_4, N_2O and NH_3 emissions from 5 operating dairies,
- CH_4, N_2O and NH_3 emissions from controlled facility (UC Davis) with complete mass balance
- Database on manure management practices for California dairies.
- Process-based model for spatially explicit estimates of CH_4, N_2O and NH_3 emissions from manure production, storage, processing and land application phase of manure management. Model will operate at scales ranging from individual dairies to air districts.
- Statewide emission inventory estimate of CH_4, N_2O and NH_3, VOC and H_2S (VOC and H_2S estimates based on best available emission factors).

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Considerations for Detailed Land Surface/Vegetation Representation in Air Quality Models

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Abstract

We discuss the significance of vegetation (particularly the treatment of canopy resistance and seasonality of vegetation density) and land surface representation on the different aspects of coupled meteorological and environmental modeling with specific focus on air quality models.

An overview of the land – atmosphere coupling and the evolution in representing the land surface/ vegetation will be discussed. Using specific example, we will illustrate the implication of the surface representations on meteorological forecasts at various scales. We will then focus on the impact of different methods to represent vegetation in the coupled weather/land-surface model on the dry deposition modeling and the need for considering detailed photosynthesis-based canopy schemes for reducing the uncertainty with deposition estimation.

Recent developments with a dry deposition modeling approach that includes vegetation-atmosphere interactions through photosynthesis/carbon assimilation in a coupled soil-vegetation-atmosphere transfer (SVAT) model, together with various methods to specify the seasonal variation of vegetation density, will be presented. Results from the coupled model studies to estimate observed deposition velocity estimates for ozone over agricultural fields, and the enhancements needed to model the bi-directional exchange for ammonia deposition near an animal agricultural facility will be presented. The presentation will conclude with specific recommendations regarding various land surface/vegetation parameterizations for agricultural air quality models.

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Measuring Gas and Odor Emissions from Swine and Dairy Manure Using a Microtunnel

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Extended Abstract

Preliminary work using a 250-ml Erlenmeyer flask, which air was blown into the flask containing 100 ml of manure to measure gas concentrations gave encouraging results that emission rates can be determined in a laboratory setting. The next steps were to determine the gas capture efficiency of the setup through a mass balance approach and develop an air collection device resembling a field wind tunnel used to collect air samples.

Ammonia Mass Balance

A mass balance approach was used to determine the amount of ammonia captured using a boric acid trap and ammonia distillation. To evaluate the capture efficiency, a known amount of ammonium chloride solution was used instead of manure so that all of the ammonia in the solution could be accounted for.

Problems were encountered in the first trials at 2.0 L/min airflow and 9.0-pH solution. Initially, a lack of airflow failing to capture the ammonia emitted from the solution was thought to be the problem. Increasing the airflow to 4.0 L/min, showed a difference in the amount of ammonia remaining in the solution versus the amount of ammonia emitted, which lead to the conclusion that the error was not in the airflow rate. As a result, the airflow rate was returned to 2.0 L/min showing a low ammonia capture efficiency of 17-27% with no repeatability. Increasing the solution pH to 12.0 to emit more ammonia from the solution resulted in a capture efficiency of 29%. The next step was to increase the test length to 240 minutes. This increased the capture efficiency to 70-90%, but repeatability was not achievable. A phosphate buffer was added to the solution to stabilize the pH throughout the testing, but similar poor results were found. A second boric acid trap was added to catch the excess ammonia, if any, but did not increase the capture efficiency. Next, sulfuric acid was used instead of boric acid, and no difference was found between the two solutions (boric vs. sulfuric).

Direct titration of the boric acid solution was compared to indirect titration through the distillation of the sulfuric acid solution. Analysis determined there were some losses due to direct titration, but was not enough to warrant the use of the indirect method. The phenate method of ammonia capture was also compared, but from these tests, repeatability could not be attained. Overall, the methods were not sensitive enough to prove differences from initial to final concentrations in emissions.

Because the laboratory containing the setup increased in room temperature due to outdoor climatic conditions, the entire setup was moved into an air-conditioned lab to better control the temperature. However, the temperature and air movement within the air-conditioned lab were still highly variable. Thus, the entire setup was moved into an incubator with more precise air temperature control and more control over air movement. The capture setup was tried for 18 hours to see if overall emissions and repeatability improved. There was a significant difference in the amount of ammonia found in distillation compared to the amount of ammonia found in the boric acid traps, thus prohibiting the determination of an accurate capture efficiency and test repeatability. After these discouraging results, the gas bubbler technique was abandon and an alternate method was investigated.

Microtunnel Design and Development

After being discouraged in using the bubblers, a switch was made to use ammonia and hydrogen sulfide analyzers, which are more accurate and can determine instantaneous concentrations. In addition, the tunnel method was tried to better mimic the airflow moving across a manure surface instead of using flasks.

The microtunnel is a square aluminum tube 1.4 cm by 1.4 cm with a 1.0 cm by 1.0 cm square hole cut in the middle of the tube on the bottom. This tunnel was placed on top of an Erlenmeyer flask full of manure.

Air was blown through the tube exhausting into the room. Each analyzer pulled 0.5 L/min of air from a tap on the tunnel. Ammonia and hydrogen sulfide concentrations along with air and manure temperatures were recorded.

The first tests compared an open vs. closed ended tunnel. The open-ended tunnel worked best by allowing higher airflows without pushing manure out the bottom of the tunnel by reducing backpressure within the tube. Differences in emissions throughout the tube were smaller, concluding that there was better mixing of air through the open-ended tunnel.

Next the location of the air tap to ensure complete air mixing was determined. Four taps were drilled into the top of the tunnel and plugged with a rubber stopper when not in use. The tap located 3.9 cm from the end was the most repeatable and did not allow diluted air to enter from the end of the tunnel. In addition, the type of tap was also analyzed comparing a circular holed tap placed in the middle of the tube to a slit tap from the top of the tube to the bottom. Testing showed better repeatability and lower concentrations using the slit due to a better representative sample being taken.

Finally the overall tunnel length was determined. Getting laminar flow at typical airflow rates is nearly impossible, leading to complete air mixing. The tube was lengthened to see the effects on hydrogen sulfide and ammonia concentrations. The longer tube had slightly lower concentrations, but the variability (standard deviation) was lower. Thus, the longer tube was assumed to have better air mixing properties throughout the tube instead of having the odorous air concentrated in the middle of the tube. This setup was used for the rest of the experiments.

Experimental Design and Testing Protocol

Observed data over a 16-hour period from the hydrogen sulfide and ammonia analyzers determined that concentration values stabilized (became constant) during the second 10 minutes of testing. Thus, a ten-minute warm up period was used to allow the analyzers and manure to stabilize followed by a ten-minute period for gas concentration recording. Airflow rates ranged from 2-10 L/min. Each analyzer required an input of a 0.5 L/min, so 2.0 L/min was determined to be the minimum rate. A Teflon bag air collection was added to the end of the 20-minute test for evaluation by an odor panel. Finally, solid phase micro extraction (SPME) fibers were exposed in the tunnel for the final ten minutes of testing. SPME Analysis was performed by Iowa State University.

Preliminary Results

Preliminary testing has shown dairy manure to crust within minutes of placing in the flask impeding emission. Solids removal by filtering prevents crusting from occurring. Manure collection and handling techniques proved important, showing differences in gas concentration depending on how the manure was collected and if the manure samples were fresh, refrigerated, or frozen. Samples were tested 24-48 hour from collection, and never refrigerator nor frozen before testing.

Correlations have been determined between airflow rates and ammonia, hydrogen sulfide, or odor concentrations in preliminary testing. Little or no correlation has been observed with the SPME fibers and airflow rates so far with these results. Additional testing with manure collected from various dairy and swine operations is currently being analyzed. In the future, the results will also be compared with emission modeling papers.

Validation of Odor Dispersion Measurements and Modeling

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Abstract

Nuisance issues related to odor dispersion from livestock and poultry production systems are a factor limiting growth and viability of these industries. The typical state or local regulatory approach to handle odors, assuming one exists, is the sliding scale method based on animal units where separation distance requirements depend upon facility size and manure management practices. However, this method does not address the influence of multiple sources in a community and the effect of receptor location relative to localized weather patterns.

Regulatory agencies, county and state planners need tools that can be easily implemented to help site new production systems or to approve/disapprove the expansion of existing production systems. This research effort is devoted to the evaluation of low-level odor measurement methods development and the subsequent use of field collected odor dispersion data to help develop an odor-based siting model. It is felt that an approach to siting that considers localized meteorological patterns, terrain, existing sources, location of receptors, operation size, and manure management practices would be a substantial improvement to the siting of operations that currently focus only on a distance-only separation requirement. The development of a process-based odor "footprint" siting method was the focus of this research project.

Introduction

Odor and gas dispersion in a community containing livestock and poultry production units is a complicated process that depends on many factors such as production system design, animal density, season, localized weather patterns, terrain, and any other sources in a community. The National Research Council (NRC, 2003) suggested that one of the two major ways to deal with the effects of airborne emissions from animal feeding operations was to replace the current emission factor approach with process-based modeling. While models that describe odor dispersion have been developed using a wide variety of techniques, to date no standardized approach for describing odor dispersion in a community has been developed. A method describing odor dispersion would be very useful for communities to aid in the siting of new production units or the expansion of current production units. This would also be of value to producers of all livestock and poultry commodities to have a method for determining, in advance, the likely improvement in community air quality resulting from implementation of best management practices for air pollution control. There is a great need for model development and validation in order to provide rural communities and the livestock and poultry industries with the tools needed to incorporate science and objectivity into the odor management decision-making process. The objectives of the research project were to:

1). Compare and standardize ambient level odor measurement methods from livestock and poultry production systems for evaluation of atmospheric dispersion models (ADM) for odor and

2). Incorporate existing odor dispersion modeling techniques into one consistent tool capable of handling multiple sources in a community of multiple receptors, and incorporating localized weather patterns, terrain, production size, and manure management techniques.

Source Emission of Odors

Odor must first be quantified to determine emission rates from a source. The most common and frequently reported measure of odor is detection threshold concentration. Diluting air samples with a known amount of odor-free air and presenting the dilutions to a panel of people using an olfactometer, which is an air dilution device, determines this value. Detection threshold concentration is the volume of odor-free air required to dilute a unit volume of odorous sample air to the point where it can be detected by 50% of a trained group of panel members (Nòren, 1987). Odor units (OU) are defined as the volume of dilution air divided by the volume of odorous sample air at detection and are thus dimensionless. However, the odor concentration of a sample is often expressed as odor units per cubic meter (OU/m^3) for calculating the conveyance of odor emission rates (European Committee for Standardization, 2002). If this convention is followed, then odor emission rates (OU/s) from a livestock building or manure storage unit are the product of the ventilation airflow rate (m^3/s) through the barn or over the storage unit and the odor concentration (OU/m^3) in the exhaust air (Lim *et al.*, 2001). In terms of quantifying source emissions, the use of olfactometers to describe "dilutions-to-threshold" has become the agreed upon standard in both the U.S. and Europe.

Models that describe the dispersion of odors in agricultural communities would help in siting of new animal production systems. However, to validate odor dispersion models one must be able to quantify the downwind, i.e. ambient level, concentration as it relates to odor sensation. One of the major challenges in assessing the odor impact in a community containing animal and poultry production systems is how best to quantify the low-level ambient odors present downwind from an odor source.

Downwind (Ambient) Level Odor Characterization

Before odor dispersion models can be utilized appropriately, not only must source emission rates for odor be quantified, but the ambient odor concentration must be quantified as well. The real difficulty is in quantifying the low-level concentration of odors, resulting from compounds that the human nose can detect in the parts-per-trillion range (O'Neill and Phillips, 1992).

Several methods have been proposed for quantifying ambient-level odor concentration. One method, used by Jacobson *et al.* (2003), uses trained human sniffers to assess ambient-level odor strength downwind from odor sources. This method uses the human nose calibrated to known concentrations of n-butanol, and assigned intensity levels ranging from 0 to 5, with 0 being undetectable to 5 that implies an extremely strong and annoying odor. These intensity scales are further correlated against dynamic dilution olfactometry to assign a detection threshold value from which dispersion modeling results can be compared.

Hoff and Bundy (2003a) employed a technique where a scentometer is used by a trained sniffer to quantify the detection threshold directly by mixing, *in situ,* known parts of filtered fresh air with ambient air downwind from an odor source. The "dilutions-to-threshold" measured from a scentometer are then used directly to assess modeled downwind odor concentrations.

Hartung *et al.* (2003) recognized the extreme lack of downwind odor concentration data for use in calibrating odor dispersion models. They developed a procedure to measure the downwind odor concentration using trained sniffers calibrated to known concentrations of n-butanol, similar to the method used by Jacobson *et al.* (2003). In addition, an SF_6 (sulfur hexafluoride) tracer was used at the odor emission source and SF_6 concentrations were measured simultaneously with odor concentration measurements. Downwind odor concentrations were assigned intensity scales from 1 to 6 corresponding to n-butanol scales which in turn were correlated to odor detection thresholds. The results indicated that well-planned experiments and appropriately trained panelists could produce high quality data sets for calibrating odor dispersion models.

DeFoer and Van Langenhove (2003) introduced a relatively new concept in quantifying downwind odor strength with the use of "sniffing units". The sniffing unit is intended to quantify the maximum odor perception distance from animal farms. Sniffing units (SU) were defined between 0.5 and 2 SU/m^3 and represent various levels that a human nose would define as a "no-effect" odor level. A factor of two was proposed to translate an SU to a dilutions-to-threshold level. In other words, 2 SU/m^3 corresponds to an olfactometer level of 4 OU/m^3. They proposed that the no-effect level was equal to 1 SU/m^3 which would correspond to an olfactometer level of about 2 OU/m^3, which has been proposed by others (Misselbrook *et al.*, 1993) as the "barely detectable" odor threshold level. The main objective was to devise a procedure whereby the no-effect distance separation from an odor source could be determined, and they assigned a level of 1 SU/m^3 as this threshold or boundary.

Dispersion Modeling

Current siting requirements for new livestock and poultry production systems in the U.S. are based mainly on animal units and distance to the nearest neighbor. This strategy has resulted in negative impacts to the animal and poultry industries. Separation distance alone does not account for existing odor sources in a community, the influence of localized meteorological factors on odor transmission, nor the use of improved odor management practices. A better approach would be to provide the animal industry and community planners a procedure or tool for making prudent decisions on where a facility of a given size could be placed in a community with an existing odor load. In this manner, decisions could be made on not only separation distance, but also as it relates to long-term meteorological conditions, size of production facility, odor control measures implemented, and existing odor levels (multiple-odor sources) in a community.

Dispersion modeling has been used to predict the concentration of odors and other animal housing contaminants downwind from production site sources since the early 1980s (Janni, 1982; Carney and Dodd, 1989; Ormerod, 1991; Chen *et al.*, 1998; Jacobson *et al.*, 2003; Hoff and Bundy, 2003a*;* Koppolu*, et al.,* 2002). The use of dispersion models to assist in the determination of setback distances based on air quality criteria requires knowledge of odor emission data from animal buildings and associated manure storage units, long-term meteorological data, and the tolerance level of neighbors for livestock odors.

Atmospheric dispersion models (ADMs) are mathematical tools that predict the movement of pollutants for air quality management. ADMs are used for regulatory purposes and in policy making. They can also be used to evaluate control technologies. Most models associated with gas dispersion use some form of the Gaussian plume theory (Turner, 1994). Although arguments for and against this modeling framework have persisted over time, this approach provides a reasonable and consistent procedure that could be applied to many different production strategies.

Models describing the dispersion and transport of air pollutants can be classified in various ways. For example, Zannetti (1990) categorized dispersion models as either plume-rise, Gaussian, semi-empirical, Lagrangian or stochastic. The primary components of interest in modeling contaminant dispersion are the total mass of pollutant emitted into the atmosphere, the spatial and temporal distribution of the pollutant, transport and transformation processes in the atmosphere, and deposition processes. Air pollution dispersion phenomena are influenced by atmospheric processes, which are commonly grouped with respect to their spatial scale. Macroscale (> 1000 km), mesoscale (between 1 and 1000 km) and microscale (< 1 km) are the various scales proposed by Orlanski (1975). Topographic considerations are also taken into account when using most modern dispersion models.

Using ADMs for the assessment of odor impacts has some significant challenges. These can be attributed, in part, to differences in odor perception by the human nose (receptor) and by variations at the source of the odor. Odor is characterized by four factors, namely frequency, intensity, duration, offensiveness (FIDO). Odors are a result of complex combinations of compounds. Schiffman *et al.* (2001) reported 310 chemicals as being present in the air around livestock facilities. Such a complex mixture of chemicals can result in unpredictable interactions leading to masking, enhancing, or synergistic effects with respect to the total odor response. Insensitization (odor fatigue) or diminished response of a subject to an odor following prolonged or repeated exposure makes it difficult to determine the likelihood of occurrence of an odor nuisance. A summary of some of the odor dispersion modeling approaches follows:

STINK: Smith (1993) developed a program called STINK as a research tool with which to estimate near source concentrations from a ground level area source of width X and finite length Y.

OFFSET: The University of Minnesota has released a tool for siting livestock facilities. The Odor From Feedlots Setback Estimation Tool (OFFSET; Jacobson *et al.*, 2003) was developed based upon odor concentration predictions from a dispersion model called INPUFF-2. The OFFSET tool, a paper-based product (Figure 1) useable by producers or county planning officers, has been implemented in several Minnesota counties for advising county planning processes.

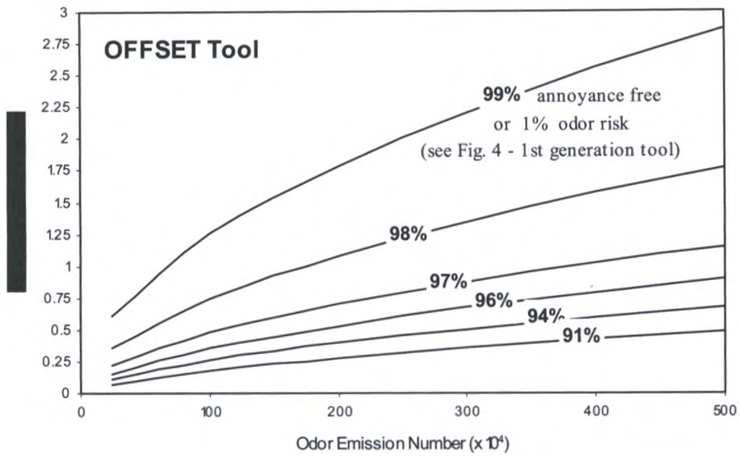

Figure 1. The OFFSET tool, which is referred to as the 1st generation tool

INPUFF-2, a USEPA model (Petersen and Lavdas, 1986) (Bee-Line Software Co., Asheville, NC), which is a Gaussian puff model, is one approach used to simulate the dispersion of odors from animal production sites. Puff models are well suited to predict agricultural odors because odor moves as a series of puffs rather than flowing as a continuous stream (McPhail, 1991). Gassman (1992) suggested that puff models represent real odor perception since the cycling of the odor actually increases odor perception to an observer so that threshold and peak concentrations will greatly exceed average concentrations for a short time period. The inputs to the model include locations of odor sources and receptors, odor source emission information (emission rate, source height, source area, emission temperature and velocity, etc.), and weather information (Pasquill-Gifford stability class, temperature, wind direction, wind speed, mixing height, etc.).

CAM: Iowa State University researchers have developed a Community Assessment Model (CAM) for predicting odor dispersion in a community of multiple odor sources and multiple receptors (Hoff and Bundy, 2003a). This model was based on Gaussian plume theory adjusted for predicting the volumetric flow rate of the downwind plume (m^3/s) resulting in the maximum ground-level odor concentration. For predicting downwind odor strength, a knowledge of the source emission rate of odors (OU/s), and the volumetric flow rate of the plume at any given downwind distance (m^3/s) yields an estimate of downwind odor strength (OU/m^3). The power of CAM is not in the complexity of the model, rather that multiple sources and multiple receptors in a community can be evaluated on a month-by-month basis to assess the annual odor load of each receptor from each source.

AERMOD: AERMOD is a near-field steady-state plume model. AERMOD is EPA's newly developed regulatory model, which incorporates planetary boundary layer concepts. AERMOD (Cimorelli *et al.*, 1998) is based on the convention that the plume is split into an updraft and a downdraft, which are associated with a vertical upward or downward velocity, respectively. The source contributions at any receptor are considered to consist of a direct source, an indirect source and a penetrated source. Terrain in AERMOD is accounted for by using the concept of a horizontal plume state and a terrain responding state. Factors determined from micrometeorolgical similarity theory and measured data are used to estimate the vertical profiles of wind speed, lateral and vertical turbulence fluctuations, potential temperature gradient, potential temperature, and the horizontal Lagrangian time scale. These are then used to compute the concentration at a receptor. A University of Nebraska research team has been working with AERMOD to address odor and gas emission issues associated with Nebraska livestock facilities. The team has had success with AERMOD for defining downwind concentrations of selected volatile fatty acids (Koppolu, *et al.,* 2002) and odor levels.

Model Validation

Most odor modeling research within communities has been strictly theoretical with little field verification of a model's predictions. The exception to this is the effort by the University of Minnesota to validate INPUFF-2 predictions. This effort included both short-range and long-range verification (Zhu *et al.*, 2000 and Guo *et al.*, 2001).

The short-range verification effort involved a panel of seven odor sniffers who recorded odor observations at 100-meter intervals between 100 and 400 meters directly downwind of the odor source. These individuals were trained to recognize odor intensity on a scale of 0 to 5 (based upon an n-butanol standard) and record those intensities downwind of 28 livestock facilities (Wood *et al.*, 2001). The team also developed an empirical relationship between odor intensity and an olfactometry-based measure of odor units. The odor sniffer data was compared against INPUFF-2 predictions of odor units. A scaling factor was applied to the odor source to adjust the model output to fit the empirical observations.

The long-range verification effort in Minnesota (Guo *et al.*, 2001) involved residents within a rural community recording odor observations over a five-month period. A community consisting of an area of 4.8 x 4.8 km was identified with 19 non-livestock residents and 20 livestock operations to complete this evaluation. Rural residents were asked to record odor intensity, timing, and duration of observations on a scale similar to that used in the short-range verification effort. Simultaneously, the INPUFF-2 model was used to predict the timing, duration, and intensity of odor for each rural resident's sensing event. Observations and predictions were compared. The INPUFF-2 model accurately predicted the low intensity observations but was less accurate in predicting high intensity observations. Since identification of recommended setback distances by OFFSET was based upon a low intensity odor threshold, it was concluded that the INPUFF-2 model and the resulting OFFSET tool provided satisfactory predictions.

Dispersion Modeling and Spatial Planning

The ultimate goal of odor dispersion modeling for this project was to develop a tool that could be used to accurately assess community odor loads for new and expanding livestock and poultry production systems. As an example, OFFSET represents a conservative non-directionally dependent siting tool for limiting the percentage of exposure time to annoying odors. This tool can be considered a 1st generation model as shown in Figure 2. An improvement on this procedure would be to include regional meteorological data to develop a directionally sensitive siting tool. This procedure could be considered a 2nd generation siting tool (Figure 2). A further refinement would be to further incorporate localized meteorological data, topography and facility specific information for site-specific planning. This procedure could be considered a 3rd generation tool (Figure 2).

In all, the goal would be to develop a procedure that could be used in any localized area that has associated with it historical weather data to develop an odor "footprint" based on an accepted odor load. Further, this footprint could be modified to incorporate the influences of multiple sources in a community, much like the efforts developed in CAM. The ultimate goal would be to prepare a user-friendly interface with a state of the art and appropriate model, to accomplish this task. This would allow producers or planning officials to model site-specific conditions for a proposed facility (including topography) and define the risk-based odor exposure footprint for an individual facility (Foot Print tool – 3rd generation, Figure 2).

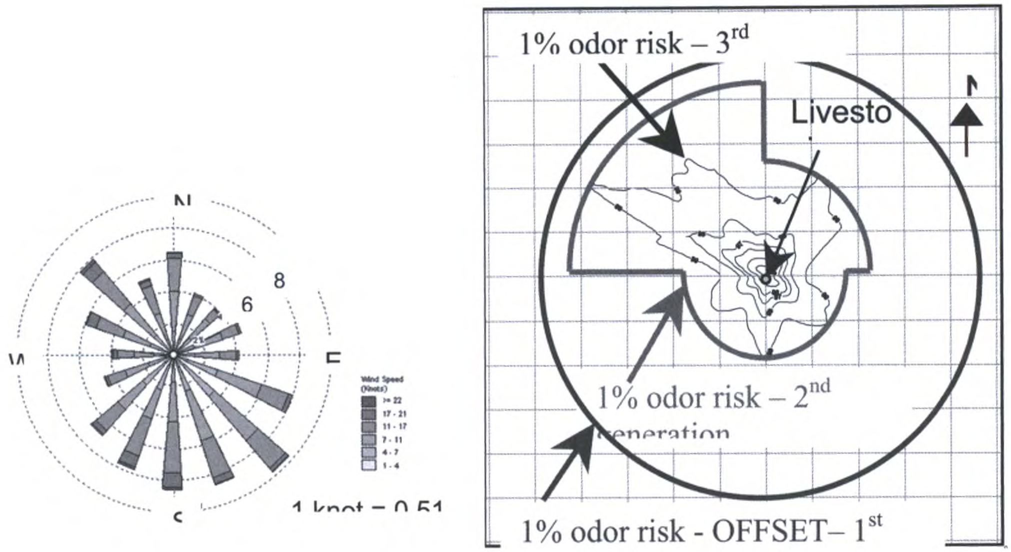

Figure 2. Predicted odor risk based on 3 generations of Footprint Tools for a livestock facility (St. Paul, MN weather conditions as indicated with accompanying wind rose).

Methods

Comparison of Ambient Level Odor Measurement Methods

Many of the techniques used to assess ambient level odor concentration were identified and discussed previously. It is felt that the methods cited have merits in their own right, but to accurately assess an odor dispersion model a standardized and consistent approach that truly reflects the sensation of odors at a receptor is required. The overall success of any odor dispersion model depends on the successful quantification of odor strength and odor perception in the community.

Two methods were used to assess proposed techniques for assessing low-level odor measurements typical of ambient downwind conditions. The first method was to use a controlled atmospheric chamber (Air Dispersion Laboratory [ADL], Iowa State University) to generate an odor from the headspace of pig slurry and distribute this odorous air to a series of panelists using three odor measurement techniques, namely, 1). direct olfactometry from sampled air, 2). olfactomtery *via* n-butanol defined levels of intensity evaluated by trained human sniffers, and, 3). Nasal Ranger scentometry (St. Croix Sensory, Inc) using trained human users. Pig slurry was mixed in various dilutions, stirred, and ventilation air entering the ADL entrained this odor providing a uniform exposure of odor to the various measurement methods. This pre-test of the proposed odor measurement methods was used to provide a base-line assessment of the correlations between measurement methods. The second method used was to conduct actual field measurements from a well-instrumented source. For this method, a series of source emission and simultaneous downwind odor concentration measurements from an instrumented deep-pit swine finishing facility in central Iowa was conducted. Measurements of ambient odor concentration were made based upon the three methods mentioned previously. Simultaneously, on-site meteorological data (MET) was collected to accurately define the atmospheric stability, a requirement crucial for accurate dispersion modeling. Figure 3 outlines the odor emission site for the field measurement experiments conducted.

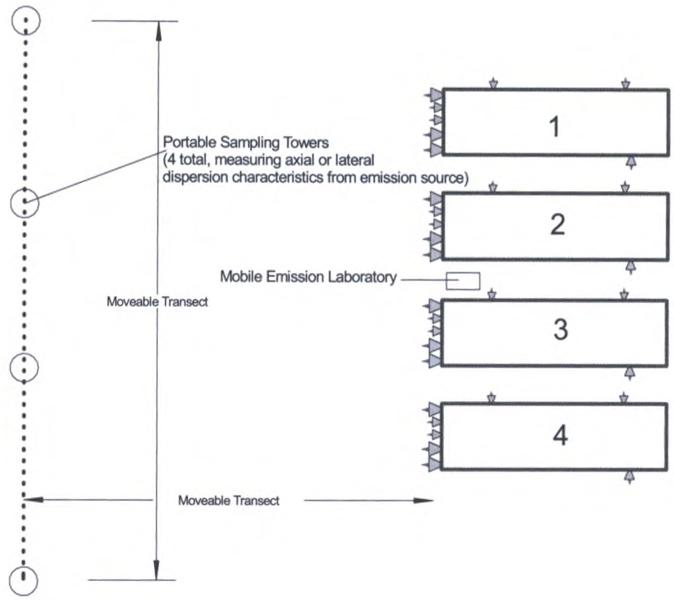

Figure 3. Top view of emission site currently instrumented and available for simultaneous emission and downwind odor measurements

Each barn of the emission site (Figure 3) is 13.7 m wide x 61.0 m long, with 19.8 m separation between buildings. Each barn houses 960 finishing pigs with a deep-pit manure handling arrangement. A mobile emission laboratory (Figures 3, 4) was used to collect emission data continuously at 10-minute increments from barns 2 and 3.

Figure 4. Mobile emission laboratory at the central Iowa emission site

The central Iowa emission site shown in Figures 3 and 4 is surrounded by fields, in a flat agricultural terrain, with a spacing of 400 m to the east and 800 m to the north and very few odor sources within 3,500 m of this site. In Iowa, predominant winds during summer are from the south to south-south-east, thus the 800 m of available space to the north is well suited for odor dispersion work. In addition, significant fall and spring weather patterns originate from the west quadrants, making the 400 m separation to the east ideal as well.

In-Field Source Emission and Ambient (Downwind) Measurements

Simultaneous source emission and downwind odor concentration data was collected in three intensive sessions between June 2004 and November 2004 where twelve distinct atmospheric events were monitored encompassing several distinct atmospheric stability classes. Each measurement event consisted of the arrangement of four grid points downwind from the emission site, arranged in sequential downwind distances. At each sampling grid point, two panelists using intensity-based measurements, two panelists using a Nasal Ranger scentometer, and two 10-L Tedlar bag samples for subsequent dynamic dilution olfactometry evaluation were collected. In addition, exhaust air samples from the barns were collected in 10-L Tedlar bags for assessing source strength. Each measurement session consisted of a 10-minute sampling interval, and this measurement session was repeated within 15-minutes resulting in two measurement sessions per downwind event. The Tedlar samples were transported and analyzed for odor strength within 24 hours using a venturi-type dynamic dilution olfactometer (AC'SCENT® International Olfactometer, St. Croix Sensory, Inc., Stillwater, MN) at either the Iowa State University or University of Minnesota Olfactometry Laboratories. Odor detection threshold, defined as the concentration that the panelist first detects a difference in the air sample compared to two clean samples, was measured in accordance with ASTM Standard E679-91 using trained panelists.

All field odor sampling events were coordinated in time with data acquisition from two on-site MET stations. One permanent 10-m high MET station fixed at the emission source and a second mobile MET station were used to assess atmospheric stability. The mobile MET station was used to measure parameters needed to evaluate atmospheric stability and to calculate on-site dispersion coefficients. These include total wind speed (3 components) at 10Hz for eddy covariance calculations, relative humidity (RH) and temperature (T) at two heights, total insolation, and net radiation for energy balance calculations.

Results and Discussion

Controlled Chamber Odor Measurement Results

A summary of the comparison between odor measurement methods using the generated odors in the Iowa State University ADL is given in Figure 5. The results shown in Figure 5B indicate the correlation measured between the Nasal Ranger scentometer and the intensity method used by the University of Minnesota. In general, for the limited data set shown the relationship was consistent between methods. For example, if the Nasal Ranger measurement recorded a 25 OU/m3, this would correspond to an intensity measurement of 2.0.

In-Field Odor Measurement Results and Downwind Dispersion

A summary of the comparison between in-field odor measurement results from the emission site shown in Figure 3 is given in Figure 6. As shown in Figure 6B considerable variability was found between the Nasal Ranger and intensity-based odor measurement methods. On average, the in-field Nasal Ranger measurement of 25 OU/m^3 corresponded to an intensity-based measurement of about 3.0, much different than the controlled chamber intensity-based measurement of 2.0 for a Nasal Ranger measurement of 25 OU/m^3.

A

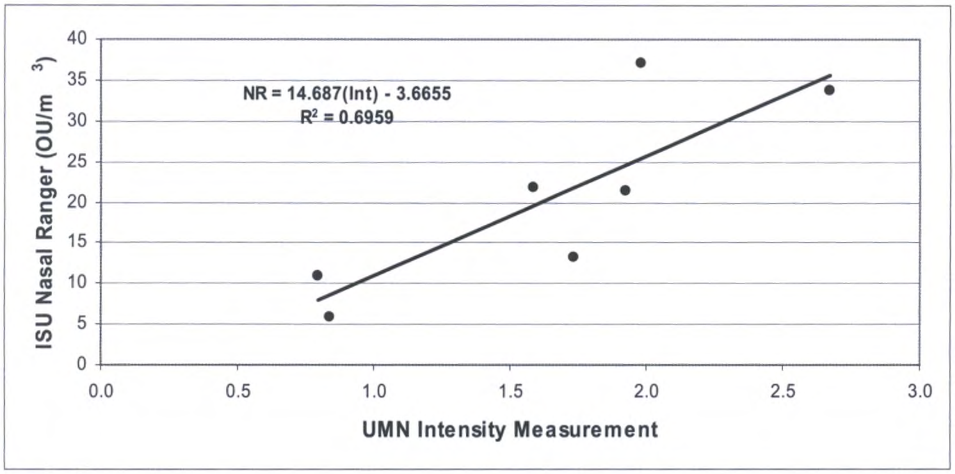

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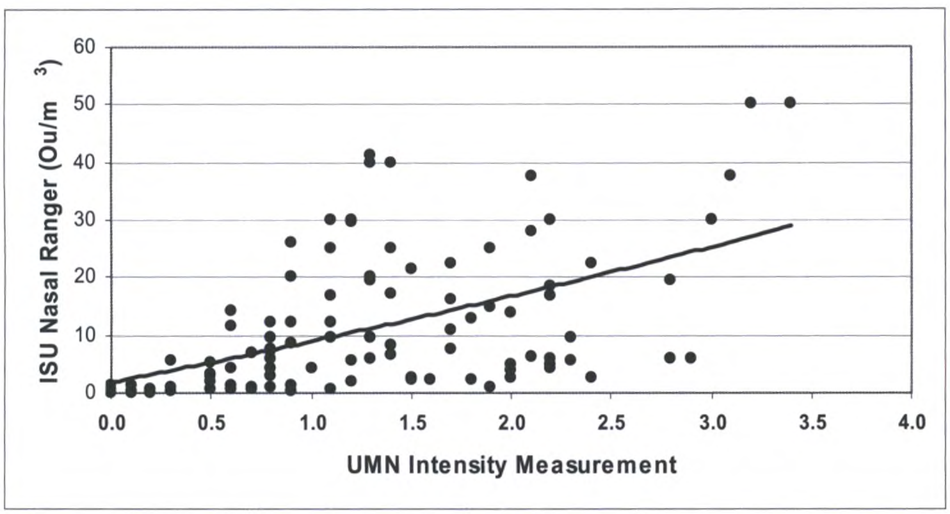
Figure 5. (A) Panelists positioned in the ADL measuring generated odors using and (B) the correlation between the Nasal Ranger and the intensity-based measurements

A

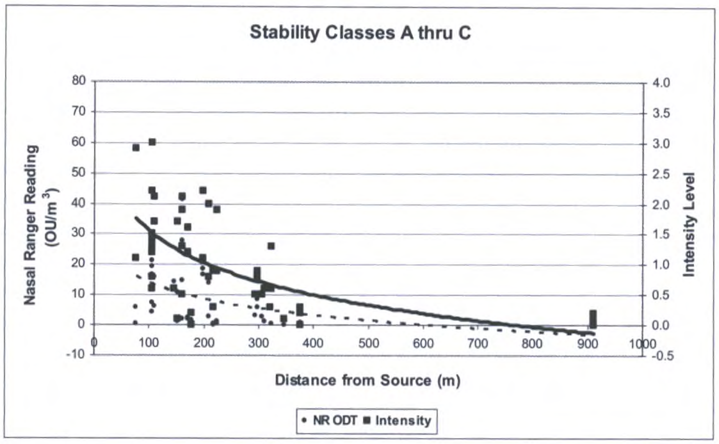


B

Figure 6. (A) Panelists positioned at one of four downwind grid points at the emission site shown in Figure 3 and (B) the correlation between the in-field Nasal Ranger and the intensity-based measurements

In-Field Odor Dispersion Results for Modeling Calibrations

Twelve distinct odor measurement events were measured using the emission site shown in Figure 3. Figure 7 shows the results obtained from these measurement events. The odor dispersion curves presented give both the Nasal Ranger and intensity-based measurements recorded for two distinct atmospheric stability conditions. Figure 7A summarizes data collected for relatively unstable atmospheres (Classes A to C) and Figure 7B summarizes the results for relatively stable atmospheres (Classes D to F). Significant variation at any downwind measurement was recorded, although decay curves are evident and were measured with both the Nasal Ranger and intensity-based methods. The distinction between odor dispersion with distance and atmospheric stability is clearly evident.



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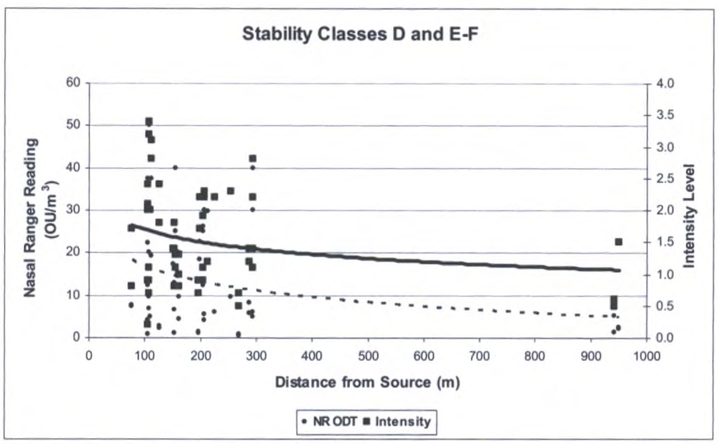

B

Figure 7. In-field downwind odor measurements for all measurement events classified as (A) Stability Classes A, B, or C, and (B) Stability Classes D, E, and F

Conclusions

An on-going research project designed to develop a livestock and poultry siting tool based on odor dispersion is being developed. This paper discussed measurements conducted designed to assess the various low-level odor measurement techniques being used today in both a controlled laboratory setting and using in-filed measurements at various atmospheric stability conditions. These results are currently being evaluated for use in the calibration of a comprehensive odor dispersion model designed to assess livestock and poultry siting decisions, taking into account terrain, local historical weather patterns, operation size, existing sources, and the location of receptors relative to the sources.

The field data collected to date shows a fairly good agreement between low-level odor measurements using the Nasal Ranger scentometer and the intensity measurement method using n-butanol as the calibration gas. Downwind odor measurements based on stability class followed traditional dispersion decay trends with atmospheric stability although a great deal of variability, as expected, was measured. Odor dispersion models that account for stable atmospheres is a necessity in order to capture the longer-range odor concentrations expected during these periods. The authors would like to thank the USDA-NRI research program for funding this on-going research project.

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Best Management Practices

Vegetative Environmental Buffers to Mitigate Odor and Aerosol Pollutants Emitted from Poultry Production Sites

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Abstract

In Iowa at a commercial poultry facility, we are assessing the ability of a multi-row vegetative environmental buffer (VEB) to mitigate odor, ammonia, and particulates. In 2004 and 2005 Eastern redcedar (*Juniperus virginiana),* hybrid willow (*Salix* X) and limber pine (*Pinus flexilis*) were planted in rows parallel to a pullet facility. Monitoring of microclimate conditions and ammonia, odor, and particulates (PM_{10} and $PM_{2.5}$) in perpendicular transects from the facility exhaust fans through the VEB is providing detailed data on flow, dispersion, capture, and throughput of the emissions. Tree species health is also being monitored. The expected impact is adoption of VEB systems as air quality best management practices to mitigate the air pollutant risks and to sustain the poultry industry, communities, and the environment.

Introduction

The poultry industry of today is facing environmental challenges relating to air and water quality. Kliebenstein (1998) suggests that the sustainability of industries within agriculture will be shaped by their collective ability to improve environmental impact technologies. Whereas odors from animal production are ubiquitous with agriculture and historically tolerated by rural communities, structural changes in the US poultry industry such as increased farm size and increased concentration of animal manure have caused more pervasive and offensive odor problems. With urban developments pushing into rural areas more people continue to be significantly affected by odor. Poultry odor nuisance may prove to be the most damaging to both rural communities, the poultry industry, and to state economies. Innovative best management practices to mitigate odor problems are needed.

The central thesis of this study is that vegetative environmental buffers (VEBs) can play significant roles in biophysically mitigating odor in a socio-economically responsible way thereby reducing social conflict from odor and dust nuisance (Malone and Van Wicklen, 2001; Malone and Abbot-Donnelly, 2001; Tyndall and Colletti, 2000).

Several recent important livestock producer outreach sources (MWPS, 2002; NPPC, 1995; Lorimer, 1998; OCTF, 1998; Jacobson et al., 1998) list tree systems (VEBs) as odor control practices but provide little physical, chemical, biological, or economic quantification as to their effectiveness.

A multi-state (Iowa, Delaware, and Pennsylvania) project is underway to quantify the efficacy of vegetative environmental buffers (VEBs), which are tree and shrub shelterbelts arranged in specific designs near and within poultry facilities, to provide cost-effective best management practice to facilitate the mitigation of odor, particulates, and ammonia, associated with intensive poultry production. Farm level monitoring and assessment of VEBs in these three states is backstopped by laboratory studies of tree/ammonia and tree/particulate interactions to lead to selection of effective and tolerant species. The farm level data will be used to simulate odor, particulate, and ammonia flow and dispersion with and without VEBs. Outreach activities will occur with collaboration of researchers, state poultry associations and government agencies. Integration of results including costs will guide producer best management decisions and inform policy decisions.

Methods

VEB Establishment

A multi-row VEB was established in two phases on a commercial poultry (pullet) farm in Northcentral Iowa. In the late summer 2004, 6 to 7 ft tall balled and burlapped Eastern redcedar (*Juniperius virginiana),* and a few test limber pine (*Pinus flexilis*) were planted in rows parallel to a pullet facility. The closest row, 25 ft from the exhaust fans, was only a partial row planted entirely with Eastern redcedar. All rows were spaced 10 ft apart and all conifers were planted 10 ft apart. Because of the limited supply of Eastern redcedar and hybrid willow (*Salix* X) in 2004 an additional planting occurred in the spring of 2005 to complete the VEB. The first three rows of the VEB closest to the building and fans (25 ft, 35 ft, and 45 ft) contain Eastern redcedar. The third row (45 ft) was planted to hybrid willow (1 ft cuttings) and Eastern redcedar and the most distant row (55 ft from the fans) was planted entirely to hybrid willow. The willows were planted 6 ft apart. The limber pine were included because of their drought tolerance capabilities, but because of the high cost per tree only a few test trees were planted in the conifer rows.

VEB Monitoring

VEB monitoring began in May 2005. In general, a Mobile Emissions Laboratory (MEL) was used to monitor semi-continuous ammonia and continuous PM10 data and this monitoring was supplemented with portable monitoring towers (PMT) installed every two weeks to capture odor, ammonia, hydrogen sulfide, and wind profiles upwind, within, and downwind of the installed VEB. The measurement methods along with some preliminary results are described below. Additionally, the effectiveness of the VEB to capture particulates was assessed by destructive sampling of tree foliage on the same schedule as the MEL/PMT sampling.

Mobile Emissions Laboratory (MEL)

A Mobile Emissions Laboratory (MEL) was installed on-site to monitor fan operation, PM10 inside the barn, ammonia concentration inside the barn, upwind, within, and downwind of the VEB. The MEL is a self-contained laboratory housing all data acquisition and gas/PM10 sampling hardware. Ammonia (NH_3) was measured using a chemiluminescence-based analyzer (Model 17C, TEI, Inc). Ammonia was measured semi-continuously at five locations including inside the barn, 1 m away from the exhaust fan, 1 m in front of the VEB, in the center of the VEB, and 1 m downwind of the VEB. These locations relative to the installed VEB are shown in Figure 1 with the MEL shown in Figure 2. Gas samples at each of these locations were sampled sequentially in 10-minute sampling blocks. A gas sampling system consisting of solenoids and relays routed gas samples from each location to the NH_3 analyzer for a total of 10-minutes each. Within this 10-minute sampling interval, the first 7-minutes were allowed for analyzer stabilization with the final 3-minutes used for analysis.

Dust sampling was conducted inside the pullet house at approximately 2m from the exhaust fan location using a Tapered Element Oscillating Microbalance (TEOM) Method (Model 1400a; Rupprecht & Patashnick Company, Inc). An installed pre-treatment head allowed particulates at and below 10 um to be sampled. PM10 sampling was continuous.

Portable Monitoring Towers (PMT)

The semi-continuous gas monitoring was supplemented with measurements conducted using three Portable Monitoring Towers (PMT) designed to capture velocity, gas, and odor profiles upwind, within, and downwind of the VEB. When installed, the PMTs were positioned at the locations shown in Figure 1. Each PMT consisted of a 9 m retractable tower with a 3-cup anemometer and a Teflon lined gas sampling tube at 1, 3, and 8 m from the ground surface. A typical PMT set-up is shown in Figure 3. For each sampling line, a vacuum box and sampling pump were used to pull sampled air into 10-L Tedlar bags, with duplicates sampled at each PMT location and within each PMT elevation. The Tedlar samples were then transported immediately to the Iowa State University Olfactometry Laboratory where odor concentration was assessed using the triangular forced-choice method (Model AC'Scent; St Croix Sensory, Inc). The PMT sampling was conducted to quantify the dispersion characteristics upwind and downwind of the VEB in an attempt to discretize between true VEB scrubbing performance and natural dilution of exhausted odorous air with downwind position. To help as well with this effort, two 6m tall diversion curtains (Figure 3) were raised during PMT sampling to minimize cross-wind effects and channel exhausted air through the VEB test section. A view as seen from the exhaust fan bank of the PMT set-up with the diversion curtains deployed is shown in Figure 4.

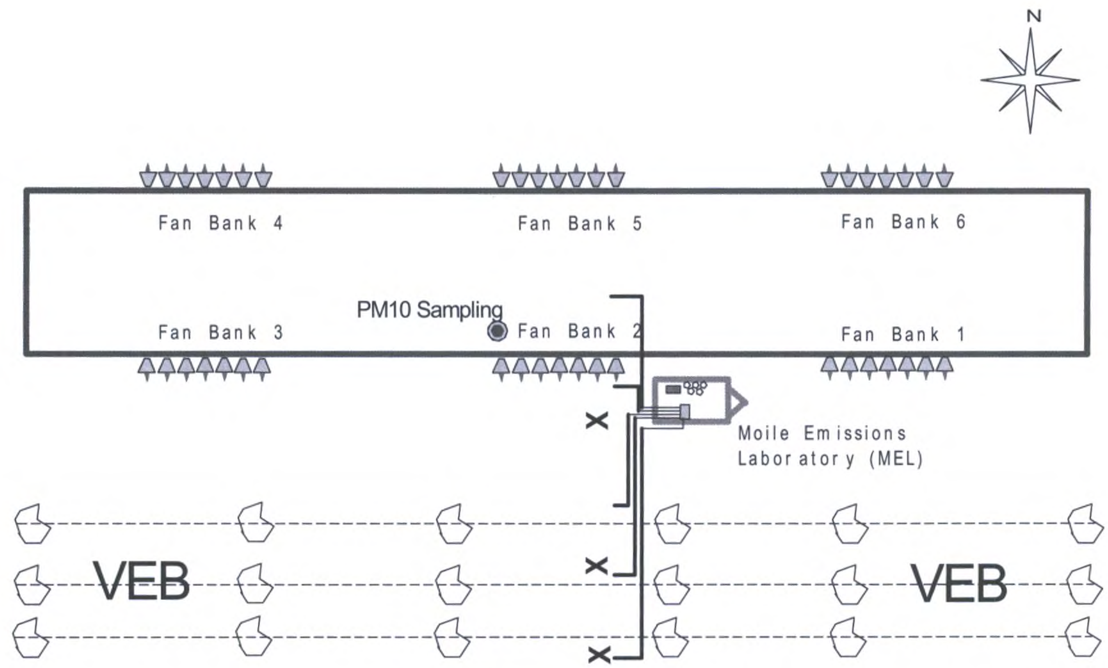


Figure 1. Pullet house monitored with the MEL, NH_3, PM10, and PMT sampling locations (x) relative to installed VEB

Figure 2. MEL shown installed on-site

Figure 3. PMT set-up with sampling diversion curtains extended

Assessing VEB Effectiveness

To assess effectiveness of the "young" VEB in trapping particulates, samples of species included in the VEB (Eastern redcedar and limber pine) as well as control specimens located at some distance from the VEB were collected 12 times on a biweekly schedule May-October 2005 (coordinated with the air quality MEL sampling regime). In the VEB, foliage samples were collected from the top and bottom halves of 2 sample trees located 35 feet from building fans and 3 sample trees located 45 feet from the building on each sample date.

Samples were washed with purified water and a dispersal agent using a flat-bed rotational shaker, and successively filtered to allow separation of coarse particulates, PM_{10} and $PM_{2.5}$. Quantity of particulate deposition was determined gravimetrically for each size class. Surface area for the foliage was determined for samples that were air-dried following the washing procedure. Samples were scanned using a digital scanner and processed using the ROOTEDGE image analysis program.

Figure 4. PMT set-up with diversion curtains shown from the exhaust fan bank looking through the VEB (Figure 1)

Results and Discussion

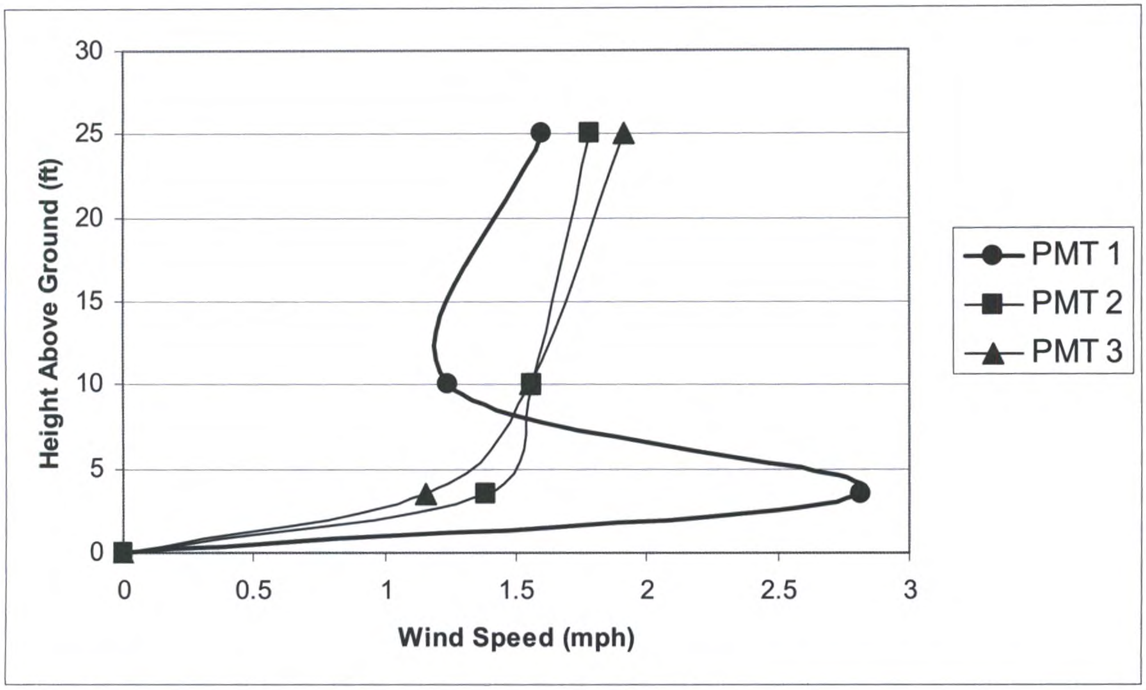
Emission Monitoring

The monitoring conducted to-date represents data collected from an immature VEB as shown in Figure 4. The monitoring results collected in 2005 will serve as base-line data to be compared as the VEB matures. Monitoring began in May 2005 and continued through December 2005, with monitoring planned again for 2006 and into 2007. To demonstrate the sampling procedures and available data for analysis, one typical monitoring day will be discussed during a day when the PMTs were implemented. Figure 5A&B show a very typical result from the PMT monitoring. The boundary layer wind speed profiles shown in Figure 5A clearly indicate the influence of the exhaust fan (PMT 1). Also, the corresponding odor concentration profiles shown in Figure 5B indicate the dilution effects as downwind distance from the exhaust fan increases. The challenge is to discretize the dilution effect that naturally occurs from entrained ambient air from the dilution or scrubbing influence of the VEB. Work continues in the development of methods to quantify the actual VEB effect.

VEB Particulate Capture

Preliminary data on total quantity of particulates captured by trees in the VEB indicate capture rates that are favorable according to levels reported in the literature (Table 1 shows data for one sample date). Control samples indicate relatively low levels of particulates under nearby ambient conditions away from the tunnel fans.

Additional data processing will allow determination of total leaf area within the VEB and estimation of total particulate capture, as well as information for each particle size fraction.



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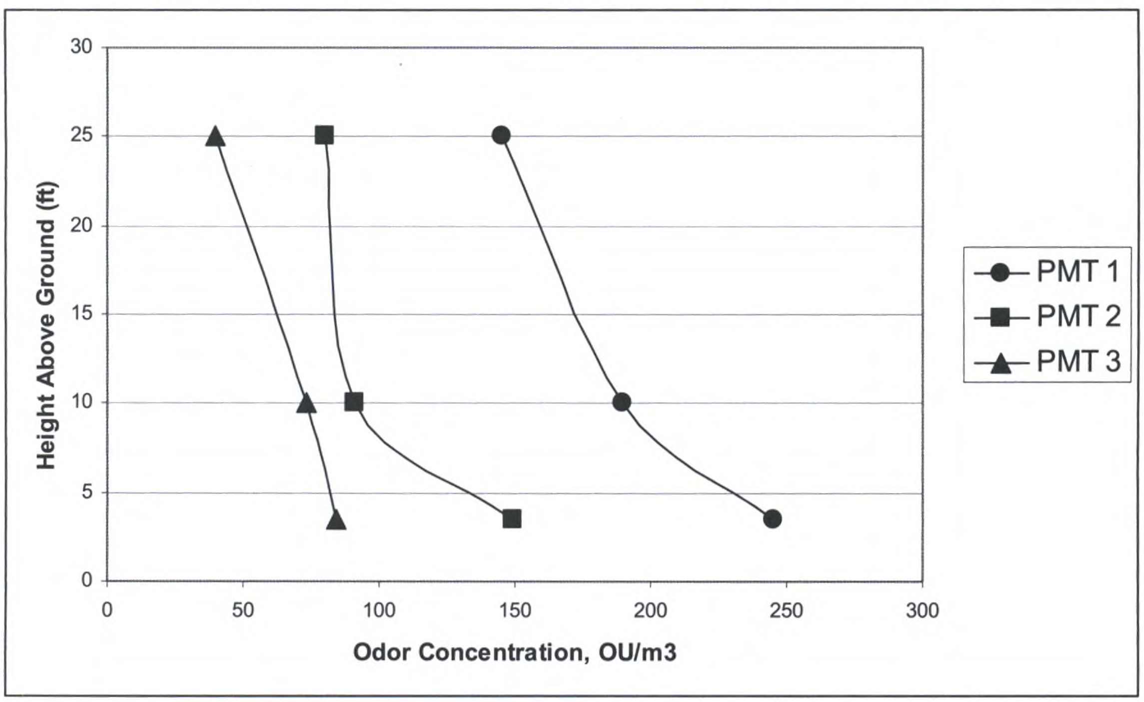

B

Figure 5. Boundary layer (A) wind speed and (B) odor concentration profiles measured between the exhaust fan and upwind of the VEB (PMT 1), within the VEB (PMT 2), and 1 m downwind of the VEB (PMT 3). The source odor concentration was 280 OU/m3 and the exhaust ventilation rate from the fans exhausting between the diversion curtains was 37.8 m^3/s during this PMT sampling event.

Table 1. Particulate capture (mg/cm^2) by Eastern redcedar and limber pine placed at 35 and 45 feet from building fans at a pullet facility for one sample date (October 10, 2005). Foliage was sampled from the top and bottom halves of VEB trees, and from one location on control trees.

Sample type	Total particulate matter, VEB samples (mg/cm^2)			Total particulate matter, control samples (mg/cm^2)	
Species / Tree part	Eastern redcedar (35')	Eastern redcedar (45')	Limber pine (45')	Eastern redcedar	Limber pine
Top	3.06	0.70	0.24		
Bottom	4.02	1.49	0.38	.06	.02

VEB Health

After one complete growing season in front of the banks of exhaust fans, it is apparent that the Eastern redcedar trees planted only 25 ft from the fans are highly stressed and may only survive and not thrive. Careful assessment of the health of these trees will be made early in the growing season of 2006.

Figure 6. A fall 2004 photo of particulate capture by Eastern redcedar from a three-month old VEB at a pullet facility in Northcentral Iowa. The trees are 25 feet downwind of the exhaust fans. Trees were balled and burlapped 6-7 ft tall trees when planted.

Conclusions

The initial results of the three-row VEB located 25 to 35 ft from the exhaust fans of a pullet facility are encouraging in terms of the ability of the VEB to capture particulates. Both visually (see Figure 6) and quantitatively the particulate capture is evident. The emission data suggest odor concentration dilution effects downstream of the fans. However, the separation of the natural, ambient dilution effect from the efficacy of the VEB to reduce odor by capture and transformation cannot be determined at this time.

Our initial evidence suggests that VEBs can be effective in mitigating poultry odor and other emissions by intercepting and diluting odor and particulates before these pollutants reach people downwind. With time we expect to quantify the efficacy of VEB in terms of retaining and transforming odor, ammonia, and particulates.

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Attenuation of Biogenic Emission of CH_4 and N_2O from Agriculture Field

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Abstract

Methane (CH_4) and nitrous oxide (N_2O) are the two important biogenic greenhouse gases emanating from the agricultural fields. These gases are radiactively very active and hence, contribute significantly to enhanced greenhouse effect, despite of their low atmospheric abundance. These gases are generated in the soil by bacteria and emitted to the atmosphere from the soil surface and through plants. Although CH_4 is produced in the highly anoxic conditions, but N_2O is produced in both oxic and partial anoxic condition. In addition, their emission is regulated by various edaphic, plant and environmental factors. Since, India has a large area under cultivation, its contribution to GHE was surmised to be major. However, the estimates made by Indian scientists for CH_4 emission from paddy fields in Methane Campaign 1991 and MAC 1998 clearly indicated that Indian paddy fields contributed only 4.03 Tg CH_4 yr^{-1} which was found ten times less than the projected figure of EPA (37.8 Tg CH_4 yr^{-1}). Despite of this fact, we investigated a few options to mitigate CH_4 and N_2O fluxes from cultivated fields by modifying edaphic factors and also using the inhibitors. These effects may not necessarily stop the global warming, but would certainly help in postponing the danger of climate change coupled with other mitigation actions for various sources of methane and nitrous oxide.

The Undercutter Method of Summer Fallow Farming to Reduce PM10 Particulate Emissions

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Abstract

Wind erosion is a major problem in the dryland winter wheat (*Triticum aestivum* L.) - summer fallow production region of the Columbia Plateau in eastern Washington and north-central Oregon. Several locations within the Columbia Plateau have failed to meet federal clean air standards for PM_{10} emissions during wind storms. Alternatives to traditional intensive tillage during summer fallow were evaluated over a 13-year period at Lind, WA. The undercutter method of summer fallow farming employs a wide-blade V-sweep for primary spring tillage plus fertilizer injection, followed by as few as two non-inversion weeding operations. Tillage is reduced from the traditional eight operations to as few as three operations using the undercutter method. Averaged over years, there were never any differences between treatments in precipitation storage efficiency in the soil or in wheat grain yield. The undercutter method consistently increased surface residue, surface clod mass, and surface roughness compared to traditional tillage. The undercutter method appears to reduce soil loss and PM_{10} emissions during high wind events. Due to the recent surge in the cost of diesel fuel and decline in the cost of glyphosate herbicide, the undercutter method of summer fallow farming has significantly higher net returns to the farmer compared to traditional tillage. Results show a 'win-win' situation for both wheat farmers and the environment.

Abbreviations: DMT, delayed minimum tillage; HRSW, hard red spring wheat; MT, minimum tillage; TT, traditional tillage; WW, winter wheat; SF, summer fallow.

Introduction

Drought, tillage, low production of crop residue, nonagregated soils with low organic matter content, and high winds often combine to leave soil vulnerable to wind erosion in the low precipitation (less than 300 mm annual) winter wheat – summer fallow (WW-SF) region of the Inland Pacific Northwest (Papendick, 2004). This region encompasses more than 1.5 million cropland hectares and is by far the largest cropping region in the western United States.

The main purpose of summer fallow is to store a portion of over-winter precipitation to enable successful establishment of winter wheat planted deep into moist soil the following August. Tillage during the spring of the fallow year is employed to break soil capillary continuity from the subsoil to the surface and create a 10- to 15-cm deep dry soil mulch to conserve water in the seed zone. Wheat farmers generally do not practice no-till (i.e., chemical) summer fallow because of increased evaporative loss of seed-zone soil moisture during the dry summer months compared to tillage. Elimination or reduction of tilled summer fallow by increasing the intensity of cropping, especially using no-till planting methods, has been shown to greatly reduce soil particulate emissions. However, to date, long-term efforts by farmers and researchers have not yet identified alternative cropping systems that can compete economically with winter wheat – summer fallow in the Inland Pacific Northwest (Nail et al., 2005; Schillinger and Young, 2004).

Traditional tillage practices are intensive and involve eight or more separate tillage passes over the field during the 13-month fallow period. Such intensive tillage operations often bury surface crop residue, pulverize soil clods, and reduce surface roughness. Blowing dust from excessively-tilled fields leads to recurrent soil losses and reduces air quality. Development and adoption of agronomically feasible, profitable and more environmentally friendly fallow management methods are needed. This paper describes research efforts that have led to a successful undercutter method of summer fallow farming to reduce dust emissions that also increase economic returns to wheat farmers compared to traditional intensive tillage practices.

Materials and Methods

Study Description

A 6-year tillage management study involving soft white winter wheat – summer fallow was conducted from August 1993 to July 1999 at the Washington State University Dryland Research Station at Lind, WA. Annual precipitation averages 244 mm. The Shano silt loam soil is more than two meters deep with no rocks or restrictive layers. The experimental design was a randomized complete block with three tillage treatments replicated four times. Individual plots were 18 by 46 meters, which allowed use of commercial-size farm equipment. Winter wheat stand establishment failed due to dry seedbed conditions during one year and plots were replanted to hard red spring wheat (HRSW). Wheat and fallow phases of the study were present each year. The three tillage management treatments were:

1) Traditional tillage (TT) – standard frequency and timing of tillage operations using implements commonly utilized by farmers.
2) Minimum tillage (MT) – standard frequency and timing of tillage operations, but herbicides were substituted for tillage when feasible and a non-inversion undercutter V-sweep implement (Fig. 1 and Fig. 2) was used for primary spring tillage.
3) Delayed minimum tillage (DMT) – similar to minimum tillage except primary tillage with the undercutter V-sweep was delayed until at least mid May. A complete description of field operations and timing for each treatment are described by Schillinger (2001).

Field Measurements

Volumetric water content measurements in the 180-cm soil profile were made immediately after grain harvest in late July (beginning of fallow), in mid March prior to primary spring tillage, and again in late August just before planting winter wheat. In addition, seed-zone water content was measured at time of planting. Soil surface cloddiness and surface roughness were determined at the end of fallow. Surface residue remaining from the previous crop cycle was measured several times throughout the fallow period. Winter wheat stand establishment was measured 21 days after planting. See Schillinger (2001) for detailed description of measurement techniques used in the experiment.

Economic Assessment

Standard enterprise budgets were constructed to assess the profitability of the three tillage systems (Nail et al., submitted; Janosky et al., 2002). Input prices are included at both 1998 and 2005 levels to show the potential for recent increases in diesel prices and decreases in glyphosate prices to favor the profitability of conservation tillage systems (Nail et al., submitted). Costs are based on the actual sequence of operations conducted on the research plots (Schillinger, 2001). Machinery costs assume farm-scale machinery typical of the region. Fertilizer, herbicide, seed and other input rates are those used during the experiment (Schillinger, 2001). Total costs include a prevailing market return for the farmer's land, machinery, and labor. Under such total cost budgeting, a "fair or normal profit" would be zero. Grain yields are those measured from the experiment. All cost and revenue figures are presented on a rotational hectare basis; specifically, for winter wheat - summer fallow, one half hectare of winter wheat and one half hectare of fallow. This ensures a standard \$ ha^{-1} basis for comparison to differing crop rotations. Consistent with the experiment, it was assumed that winter wheat would fail once every five years necessitating replanting to hard red spring wheat. Consequently all costs are weighted 80% WW-SF and 20% for HRSW-SF.

The 1993-1997 5-yr average market prices of \$144.02 Mg^{-1} for WW and \$187.22 Mg^{-1} for HRSW were retained in order to hold constant all factors except input price changes (Janosky et al., 2002). These averages are only slightly higher than recent 5-yr average wheat prices. Government payments are not included in the net revenue results as the emphasis is on market profitability rather than on varying government payments.

Validation of Wind Erosion Control

In a separate study in 2005, soil loss and dust emissions were assessed from a Shano silt loam near Lind, WA. Adjacent summer-fallowed fields were managed using either traditional tillage or the undercutter method of conservation tillage. The treatments, each 2-ha, were cropped to winter wheat in 2004 and, following harvest, remained undisturbed (wheat stubble and straw on the soil surface) until the spring of 2005. Treatments were:

1) Traditional tillage – tandem disk to a depth of 13 cm on 2 May, injecting aqua N fertilizer on 6 May, and rod weeding on 10 May and 21 July.

2) Conservation tillage - undercutting + aqua N fertilizer injection to a depth of 13 cm with overlapping 71-cm-wide V-blades on 5 May, and rod weeding on 10 May and 21 July.

Winter wheat was grown along the south and west boundary of the experimental sites during the 2005 growing season to create a non erodible boundary; this boundary minimized the influx of airborne and eroded sediment for estimating soil and PM_{10} loss from each experimental site. Crop residue and soil physical properties were periodically measured throughout the spring and summer of 2005. Crop residue cover and biomass, stubble density and biomass, stubble height, near-surface soil water content, particle size distribution, aggregate size distribution, and roughness were determined after each tillage operation or high wind event. High wind events, defined by a period of time when winds are in excess of 6.4 m s^{-1} at a 3 m height, commonly occur during the spring and autumn in the Columbia Plateau (Saxton et al., 2000). These events, which result in soil erosion and elevated atmospheric dust concentrations, are the major cause for exceedances of the National Ambient Air Quality Standard for PM_{10} within the region (Sharratt and Schillinger, 2005).

Soil loss from traditional and conservation tillage was assessed by measuring the influx and efflux of soil from the experimental sites. High-volume air samplers, E-samplers, and BNSE airborne sediment collectors were deployed at the windward and leeward positions at each site to assess sediment moving into each site from the non erodible boundary (wheat field) and that leaving each site. High-volume samplers and E-samplers were positioned at a height of 1.5, 3, and 6 m above the soil surface. Creep collectors were used to measure surface creep and BSNE collectors were positioned at a height of 0.1, 0.2, 0.5, 1.0 and 1.5 m above the soil surface to assess saltation and suspension.

Results and Discussion

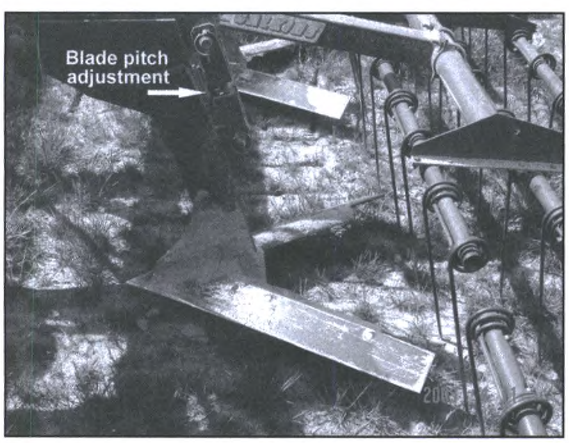

Figure 1. Haybuster™ undercutter V-sweep blade with attached 3-bar spring-tooth harrow. As a primary spring tillage implement, the undercutter completely severs capillary pores to halt liquid water movement toward the soil surface as required to retain seed-zone water in summer fallow.

Figure 2. Primary spring tillage plus liquid aqua nitrogen injection with an undercutter V-sweep implement. Note that the majority of residue remains standing and undisturbed.

Residue and Surface Soils

Two to three-fold increases in surface residue cover was consistently retained with MT and DMT compared to TT at the end of the fallow cycle throughout the 6-year study (Fig. 3, Fig. 4, data not shown). When wheat straw production was low, the minimum quantity of surface residue (390 kg/ha) required for highly erodible soil for government farm program compliance could not be achieved or was marginally met using TT, whereas ample surface residue was always present with MT and DMT. On average, twice the surface soil clod mass and a rougher surface was achieved with MT and MDT compared with TT. See Schillinger (2001) for further details.

Agronomy and Grain Yield

Seed-zone water content at the end of fallow in the 6-year study was not affected by tillage treatment. This suggests that finely divided soil particles with the tillage mulch may not be as important for retarding evaporative water loss during the hot, dry summer as previously thought. Rather creating an abrupt break between the tilled and non-tilled layer with primary spring tillage, which severs capillary channels from the subsoil to the surface, appears to be the dominant factor regulating over-summer evaporative water loss.

Winter wheat seedling stand establishment (Fig. 4) was somewhat reduced in the MT and DMT treatments (data not shown) due to the larger quantities of surface clods compared to TT, as many seedlings were unable to elongate around clods that rolled into the furrow during planting. There were no differences in wheat grain yield among treatments during any year or when averaged over five years (Table 1).

Economics

Table 2 reports costs and net returns for the three tillage systems at both 1998 and 2005 input prices. As noted earlier, the costs are weighted averages of those for WW at 80% and for HRSW at 20%. Table 2 compares net returns over variable and over total costs of each tillage system for the two input price levels. The use of 2005 input prices instead of 1998 prices increases the total cost of TT, and reduces the returns over total costs of TT, by \$2.36 (rotational ha)$^{-1}$. The differences over input price levels for total cost and net returns over total cost are identical because revenue, based on experiment average grain yields and common crop prices, is constant for both input price levels. In contrast to TT, the 2005 input prices boosted profits for both MT and DMT. Total costs decreased and net returns over total costs increased by \$6.37 and \$6.30 (rotational ha)$^{-1}$ for MT and DMT, respectively (Table 2).

Table 1. Yearly and five-year average yield of wheat as affected by traditional tillage, minimum tillage, and delayed minimum tillage during the preceding fallow cycle. ns = not significantly different at the 5% probability level.

Tillage system	1995	1996	1997	1998	1999	Avg.
	Mg ha^{-1}					
Traditional tillage	1.79	3.52	5.13	3.89	2.32	3.33
Minimum tillage	1.91	3.76	5.20	3.89	2.69	3.49
Delayed minimum tillage	1.79	3.73	4.94	3.58	2.48	3.30
	ns	ns	ns	ns	ns	ns

Figure 3. A rodweeder (2-cm square rotating rod) was operated 10 cm below the soil surface two to three times during late spring and summer to control weeds in summer fallow. Ample residue in the minimum tillage treatment (shown here) provides protection from wind erosion.

Figure 4. Winter wheat seedlings in the minimum tillage fallow treatment. Percent residue cover after planting wheat achieved or exceeded 30% residue cover all six years of the experiment with minimum tillage and delayed minimum tillage, compared to traditional tillage that attained 30% cover only two out of six years.

Table 2: Net returns for three tillage systems for winter wheat - summer fallow farming using 1998 and 2005 production costs.†

Tillage system	Revenue	Net Returns Over Cost Using Input Prices†† Variable	Net Returns Over Cost Using Input Prices†† Total	Net Returns Over Cost Using 2005 Input Prices Variable	Net Returns Over Cost Using 2005 Input Prices Total
	---------$ (rotational ha)$^{-1}$---------				
Traditional tillage	247.73	100.90ab	-36.36b	91.95b	-38.72b
Minimum tillage	259.67	103.97a	-25.07a	105.86a	-18.70a
Delayed minimum tillage	245.56	93.29b	-27.88a	96.16ab	-21.58a

†Weighted average of 80% soft white winter wheat and 20% hard red spring wheat.

††Average net returns followed by the same lower case letter are not significantly different at the 0.05 probability level. $LSD_{.05}$ for the average net returns over variable costs per rotational hectare of the three tillage systems computed using 1998 input prices is \$9.96 ha^{-1} and \$10.02 ha^{-1} for 2005 prices. $LSD_{.05}$ for the average net returns over total costs per rotational hectare of the three tillage system computed using 1998 input prices is \$6.78 ha^{-1} and \$6.89 ha^{-1} for 2005 prices. For 2005 prices, net returns over total costs for minimum and delayed minimum tillage systems are statistically superior to traditional tillage at $P<0.000001$. For 1998 prices, net returns over total costs for minimum and delayed minimum tillage systems are statistically superior to traditional tillage at $P<0.01$.

The MT and DMT systems averaged significantly higher net returns over total costs under both 1998 and 2005 input prices compared to TT (Table 2). For 2005 prices, MT and DMT systems' net returns over total costs are statistically superior to those for TT at the $P<0.000001$ probability level. For the 1998 price scenario, the MT and DMT systems' statistical advantage over TT occurs at the $P<0.01$ probability level. The statistical advantage at 2005 prices of conservation tillage versus TT is preserved for returns over variable costs for MT, but not DMT (Table 2). The statistical advantage for DMT occurs for returns over total costs because fixed costs are relatively higher for TT than DMT and this widens the profitability gap. These results confirm that recent price shifts for diesel and glyphosate have strengthened the economic advantage of conservation tillage for winter wheat - summer fallow farming in this region.

Validation of Wind Erosion Control

Conservation tillage using the undercutter appeared to suppress erosion and dust emissions from the soil surface in comparison to traditional tillage during high wind events in 2005. The horizontal flux of suspended sediment was markedly reduced by an order of magnitude near the soil surface as indicated for the 7 - 23 June high wind event in Fig. 5. The largest reduction in flux occurred near the soil surface where protection afforded by larger aggregates and more wheat residue in conservation tillage effectively trapped mobile soil particles during high winds.

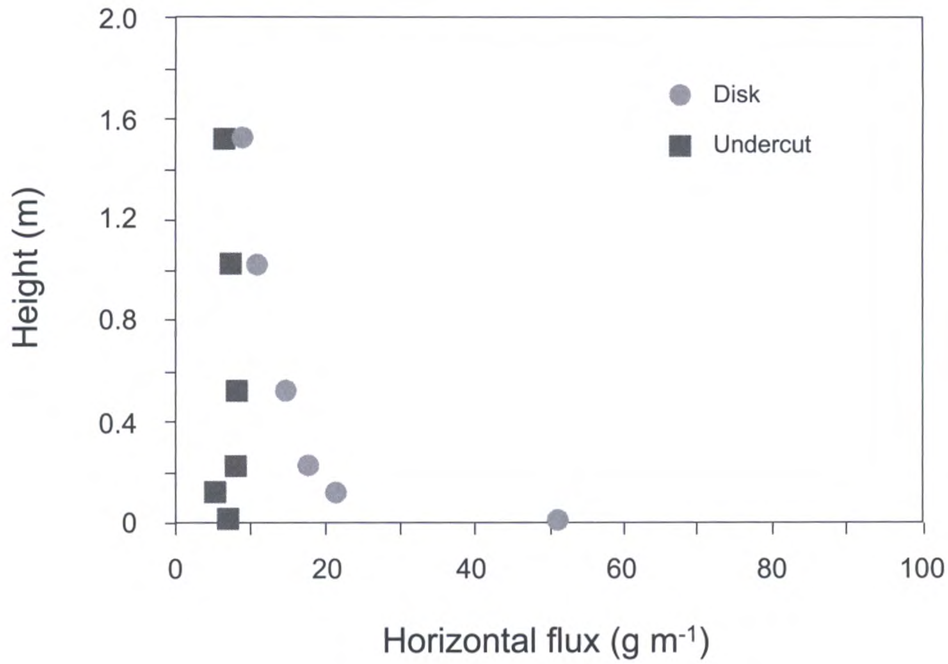

Figure 5. Horizontal soil flux as a function of height above soil surface with traditional (tandem disk) tillage and conservation (undercutter) tillage. Measurements were made during the 7- 23 June 2005 high wind event.

Loss of soil from the field site during the 7 - 23 June event ranged from 22 kg ha^{-1} for conservation tillage to 190 kg ha^{-1} for traditional tillage (Table 3). A reduction of soil loss using conservation tillage was also apparent during a high wind event earlier in the season (12 May – 7 June). However, these losses do not account for the sampling inefficiency of the BSNE for Shano loam, the predominant soil type at the field site.

Table 3. Loss of soil, determined from BSNE catch, from traditional and conservation tillage plots during high wind events in 2005 near Lind, Washington.

	Soil Loss	
Date	Traditional	Conservation
	---------------- kg ha^{-1} ----------------	
12 May – 7 June	88	13
7-23 June	190	22

PM_{10} concentrations were 50% less in conservation tillage compared to traditional tillage. Reduction in emissions is apparent by the smaller PM_{10} concentration above the soil surface in conservation tillage (Fig. 6). The reduction in PM_{10} emissions is more apparent nearer the soil surface and corresponds to the reduction in horizontal flux of larger soil particles. For the 23 May – 7 June event, emissions were reduced using conservation tillage to a height of about 6 m, which is the typical height of a dust plume during a high wind event on the Columbia Plateau (Fig. 6).

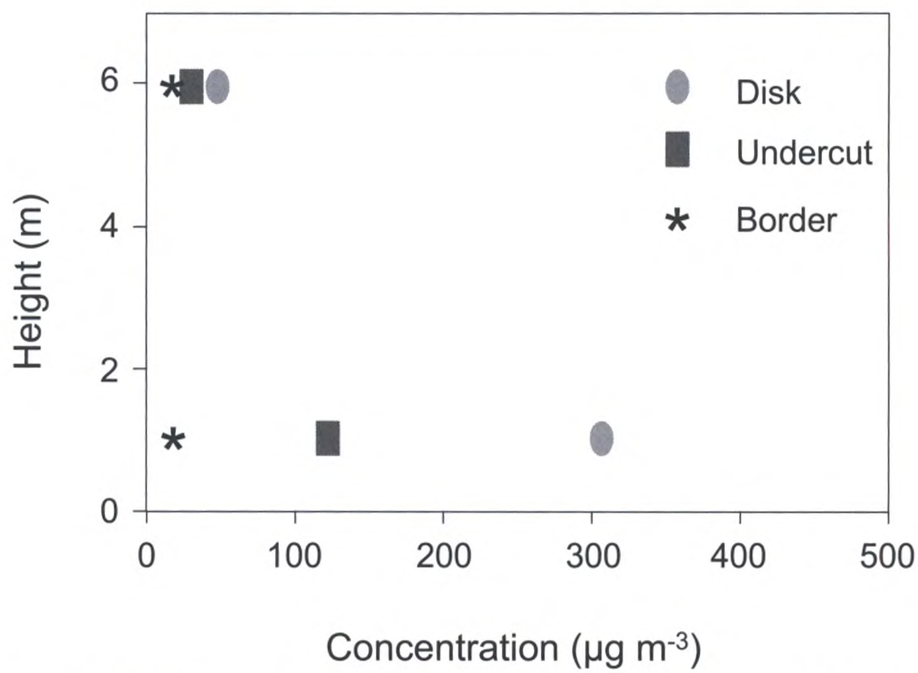

Figure 6. PM_{10} concentration at various heights above the soil surface during the 12 May – 7 June high wind event. Concentrations were measured in traditional tillage and conservation (undercutter) tillage plots and at the upwind border of the experimental site.

Conclusions

Minimum tillage and delayed minimum tillage during fallow resulted in equal wheat grain yield and superior profitability compared to traditional tillage. Minimum tillage and delayed minimum tillage also provided more surface residue for better protection against wind erosion. This research showed that, with judicious use of herbicides, tillage operations during fallow can be effectively reduced from eight with traditional tillage to as few as three with delayed minimum tillage. If minimum tillage and delayed minimum tillage fallow management were widely practiced in the Columbia Plateau, it is reasonable to expect a sharp reduction in wind erosion and suspended dust emissions, with associated benefit to air quality. Minimum tillage and delayed minimum tillage, as outlined in this paper, will also benefit the economic livelihood of wheat farmers in the dry regions of east-central Washington and north-central Oregon.

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Assessment of the Ammonia Emission Abatement Potential for Distinct Geographical Regions and Altitudinal Zones in Switzerland

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Abstract

Abatement measures for a further reduction of agricultural ammonia (NH_3) emissions will have to be introduced in most countries of Europe in the foreseeable future. A detailed study on possible abatement measures and their potential in Switzerland was therefore conducted. As abatement strategies must be adapted to the prevailing production systems, the study differentiated between nine regions (3 geographical regions x 3 altitudinal zones), using data about current farm management from a recent stratified survey. Model calculations were performed with the N-flux model DYNAMO both for a scenario considering the implementation of all technically possible measures and one with measures considered realistically feasible.

Under the present situation, extended grazing of dairy cattle and measures concerning manure application offer the most efficient reduction potential. If all technically possible emissions abatement measures would be realized Swiss NH_3 emissions could be reduced by approximately 33%. The full realization of measures considered technically feasible would reduce NH_3 emission by up to 19%. The realistically feasible abatement is at least twice as high in the valley zone than in the mountain zone.

Introduction

Ammonia (NH_3) emissions are highly relevant from an environmental point of view due to their contribution to eutrophication and acidification of vulnerable ecosystems and their involvement in the formation of secondary aerosols. The UN Convention on Long-range Transboundary Air Pollution (UNECE 1999) as well as national policies therefore set emission ceilings. For example, the convention requires Switzerland to reduce NH_3 emissions from 59.3 to 51.9 kt nitrogen (N) between 1990 and 2010 while the Swiss Federal Council, based on considerations of critical load exceedence in vulnerable ecosystems, has set a long-term goal of 25-30 kt N. While emissions were already reduced by about 20% between 1990 and 2000, mainly due to a reduction of livestock numbers, a further reduction by 30-40% would still be necessary to meet the long-term goal.

As agriculture is the major emitter of NH_3 (over 90% in Switzerland) abatement measures for a further reduction of agricultural ammonia emissions will have to be introduced in most countries of Europe in the foreseeable future. This will require reliable and differentiated recommendations to farmers about the most promising and appropriate emission abatement measures and model calculations regarding the abatement potential. As a follow-up activity to the new Swiss NH_3 emission inventory (Reidy and Menzi 2006) a detailed study on possible abatement measures and their potential was therefore conducted. The project aimed to provide farmers as well as policy makers with relevant data and recommendations.

The emission abatement potential of measures depends on their efficiency under farm conditions as well as on their applicability. The applicability of different measures depends on the animal housing and manure management system as well on several other factors such as topography, climatic conditions and soil type. It can also be influenced by the skills of farmers and their social environment. Especially in Switzerland with its very variable topographical and climatic conditions the applicability of different measures and the emission abatement potential can therefore be expected to vary considerably from region to region and from farm to farm. To account for this, our evaluation differentiated between geographical regions and altitudinal zones and practical recommendations will be provided for different farm types.

Methods

Model calculations on the abatement potential were done with DYNAMO, an empirical mass-flow model following the N flow approach (Reidy and Menzi 2006). 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 included animal houses and hardstandings, manure storage and application, grazing, mineral fertilizer use as well as crops and meadows. It is therefore well suited to take into account interactions between different emission stages and farm management parameters, e.g. the effect of measures in animal houses on emissions from manure storage and application.

For relevant current farm management parameters average values from the survey conducted in the framework of the emission inventory project (Reidy and Menzi 2006) were used. This stratified survey provided data from 1950 farms and differentiated between nine regions: the three geographical regions Eastern, Central and Northern/Southern Switzerland; for each region the valley zone, the hill zone and the mountain zone. Statistical data on livestock numbers and farming surface were taken from the official farm census for the year 2000.

Overall, about 60 different abatement measures were considered in the study. Their efficiency in reducing emissions was compiled from literature and Swiss research data. The applicability of each measures in the nine regions was assessed together with experts in the fields of feeding, grazing, animal housing, manure storage, manure application and fertilization. For each measure a technically possible and a realistically feasible reduction potential were defined. At both levels no economic restrictions were considered. The abatement potential for both levels was calculated for individual measures as well as for different scenarios of combined measures.

Results and Discussion

Under the present situation, extended grazing of dairy cattle and manure application measures offer the most efficient reduction potential. At the national level they could contribute a reduction of NH_3 emissions from agriculture of about 9.5% and 11% for grazing and manure application measures respectively, using the assumption of what would be technically possible. Using the assumptions of what would be realistically feasible, the reduction potential for grazing and manure application measures would be about 6% and 8%, respectively.

While there were no major differences between the different regions regarding the abatement potential for feeding, housing and manure storage measures, the abatement potential of grazing and manure application measures showed considerable regional differences. Figure 1 illustrates this using the example of the emission abatement potential for slurry application with trailing hose systems.

On a more aggregated scale, the realistically feasible abatement potential of low emission manure application measures ranged from 9% to 12% in the valley zone, from 4% to 6% in the hill zone and from 0% to 1.0% in the mountain zone.

If all technically possible emission abatement measures would be realized, Swiss NH_3 emissions could be reduced by approximately 33%. The full realization of measures considered realistically feasible would reduce NH_3 emission by up to 19%. The realistically feasible abatement is at least twice as high in the valley zone than in the mountain zone.

The applicability and the potential of measures for a specific farm do not only depend on climatic and topographic conditions, but are also strongly influenced by existing infrastructure and farm structure. The conclusions about the most appropriate emission abatement measures can therefore differ considerably from those derived for different regions. For example, for a farm with open slurry storage, covering slurry stores, especially if theses are newly constructed, is a high priority measure, while it is of secondary importance on the regional or national scale because around 80% of the stores are already covered.

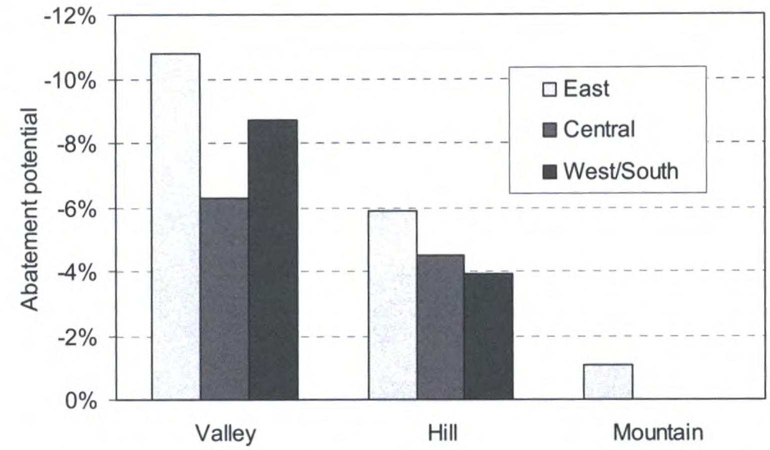

Figure 1. Estimated long-term ammonia emission abatement potential for slurry application with trailing hose systems in different regions of Switzerland. Results are given for the scenario "realistically feasible". Calculations were based on farm structure and livestock number data of 2000." The abatement potential in the central and west/south mountain region is zero.

Conclusions

There still is a considerable potential to reduce NH_3 emissions from Swiss agriculture. Nevertheless, even if all measures considered realistically feasible were fully implemented on all farms, the long-term goals could not be achieved without a considerable further reduction of animal numbers.

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Evaluation and Management of Ammonia Emissions from Poultry Litter

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Abstract

Ammonia volatilization from poultry litter results in high levels of ammonia in poultry facilities, which negatively impacts bird performance and worker health. Ammonia emissions from the houses also cause atmospheric ammonia contamination. Although there is a tremendous concern about these emissions currently, little quantitative data exists on the magnitude of ammonia fluxes from poultry litter. Hence, knowledge on the magnitude of these emissions, factors affecting emissions, and methods to control ammonia losses are needed. The objectives of this study were to: (1) measure ammonia volatilization from poultry litter in broiler houses and following land application, (2) evaluate the factors that affect ammonia losses from poultry litter, and (3) determine the impact of best management practices on ammonia volatilization. Four tunnel ventilated houses were equipped with ammonia sensors, anemometers, and data-loggers which continuously recorded the ammonia concentrations and ventilation (wind speed) at each of the windows in each house. In addition, ammonia, nitrous oxide and methane fluxes from the litter were measured using flux chambers. Nitrogen losses following land-application were evaluated using ammonia wind tunnels. Results of the study indicate that although ammonia emissions from one broiler house containing 32,000 birds can exceed the CERCLA reporting threshold of 100 lbs/day, losses are typically much lower than this value. An ammonia scrubber that attaches onto the exhaust fans of animal rearing facilities was developed and tested. The scrubber, which utilizes a dilute alum solution to capture ammonia and dust, was shown to significantly reduce ammonia concentrations in exhaust air from broiler houses. The aluminum in the solution, which is converted to aluminum hydroxide prior to land application as the pH of the solution is increased by ammonia, immobilizes soluble phosphorus (P), which reduces the risk of non-point source P runoff. Ammonia emissions from land applied poultry litter totaled 34 kg N/ha (15% of the total N applied). When the poultry litter was incorporated, ammonia losses were virtually zero (N loss was not different from unfertilized control plots). These studies indicate that there are several best management practices that can be utilized to reduce ammonia loss from poultry litter.

Introduction

Ammonia volatilization from poultry litter results in high concentrations of ammonia in poultry houses, which has been known to cause poultry production problems including increased susceptibility to viral diseases, reduced growth rates, reduced feed efficiency, decreased egg production, and blindness (Carlile, 1984). High ammonia levels in animal rearing facilities have also been shown to be detrimental to the health of agricultural workers (Donham, 1996). Carlile (1984) recommended that ammonia concentrations be kept below 25 ppm in poultry houses to reduce the incidence of negative performance associated with this gas. Keeping ammonia levels this low is not always easy for growers to do, particularly during the winter time when they are trying to minimize ventilation, because of the high cost of propane gas to heat the houses. As a result, ammonia concentrations can exceed 100 ppm in poultry rearing facilities.

Ammonia volatilization from manure also has a negative impact on the atmosphere, with respect to both acid precipitation and atmospheric N loading to aquatic systems (Ap Simon et al., 1987; van Breemen et al., 1982; Schroder, 1985). Ap Simon et al. (1987) indicated that animal manure is the dominant source of atmospheric ammonia in Europe. Van Breemen et al. (1982) indicated that ammonia plays an important role in acid precipitation. Another problem attributed to ammonia is N deposition into aquatic systems (Schroder, 1985).

Poultry producers have used a variety of manure amendments to reduce ammonia volatilization. Moore et al. (1995a, 1996) found that alum (aluminum sulfate) and phosphoric acid were the most effective compounds for reducing ammonia loss. Additions of alum to litter reduce the pH, which shifts the NH_3/NH_4 equilibrium toward NH_4. Moore et al. (1997) showed alum additions in commercial chicken houses reduced ammonia fluxes by 97% during the first four weeks of the growout and 70% during the entire flock. Moore et al. (1999, 2000) later showed that broilers grown in houses where the manure was treated with alum had better weight gains, improved feed conversions, lower mortality, and lower condemnations. Energy use (primarily propane use) was also lower in the winter when alum was used, due to lower ventilation requirements to remove ammonia (Moore et al., 1999, 2000). Several other benefits of treating poultry litter with alum have been noted. One of the primary benefits is alum reduces phosphorus runoff and leaching (Moore et al., 1999, 2000). Alum additions to litter have been shown to greatly reduce pathogens responsible for food borne illnesses (Line, 2002), including *Campylobacter* and *Salmonella.* Heavy metal and estrogen runoff from fields fertilized with poultry litter has also shown to be significantly lower when the litter has been treated with alum (Nichols et al., 1997; Moore et al., 1998). Moore and Edwards (2005) showed in a long-term study that aluminum uptake by crops, Al availability in soil and Al runoff are not affected by alum. Due to these improvements in productivity, alum is routinely used by the poultry industry; currently over 600 million chickens grown annually in the U.S. with alum.

While alum clearly is one best management practice (BMP) than can be used to control ammonia, other BMPs need to be developed and tested. The objectives of this work were to: (1) measure ammonia volatilization from poultry litter in broiler houses and following land application, (2) evaluate the factors that affect ammonia losses from poultry litter, and (3) determine the impact of best management practices on ammonia volatilization.

Methods

Four tunnel ventilated broiler houses (12.8 x 146.4 m) were equipped with ammonia sensors (Polytron I), anemometers, and data-loggers which continuously record the ammonia concentrations and ventilation (wind speed) at each of the windows. It should be noted that these electro-chemical ammonia sensors can became saturated with ammonia over time which can cause them to malfunction, hence, they must be replaced regularly. Thirty-two thousand broilers were placed in each house and were grown to 56 days of age. Fluxes of ammonia, nitrous oxide, methane, and carbon dioxide were also measured from the litter were also measured using flux chambers equipped with an Innova multi-gas analyzer. Total and inorganic N in the litter is also be evaluated throughout the year, as well as feed consumption, N content of the feed and N removed in birds, in order to construct a simple mass balance estimate of ammonia loss. Ammonia emissions following land-application were evaluated using wind tunnels from plots where litter had either been surface applied or incorporated. Litter was applied to small plots cropped to bermudagrass at a rate of 5.6 Mg litter/ha, which had 4% N (equivalent to 224 kg N/ha). Ammonia concentrations and air flows entering and exiting the wind tunnel were measured using anemometers that had been calibrated using the FANS system connected to data-loggers and phosphoric acid traps which were later analyzed for ammonium.

Results and Discussion

Ammonia Emissions from Broiler Houses

The average daily ammonia concentrations in the four houses during two flocks in early summer are shown in Figure 1. Although the average daily ammonia levels were always below 25 ppm, many times during the night they exceeded 80 ppm. High ventilation rates during the day result in lower ammonia. The average ammonia concentration in the houses was 16 ppm.

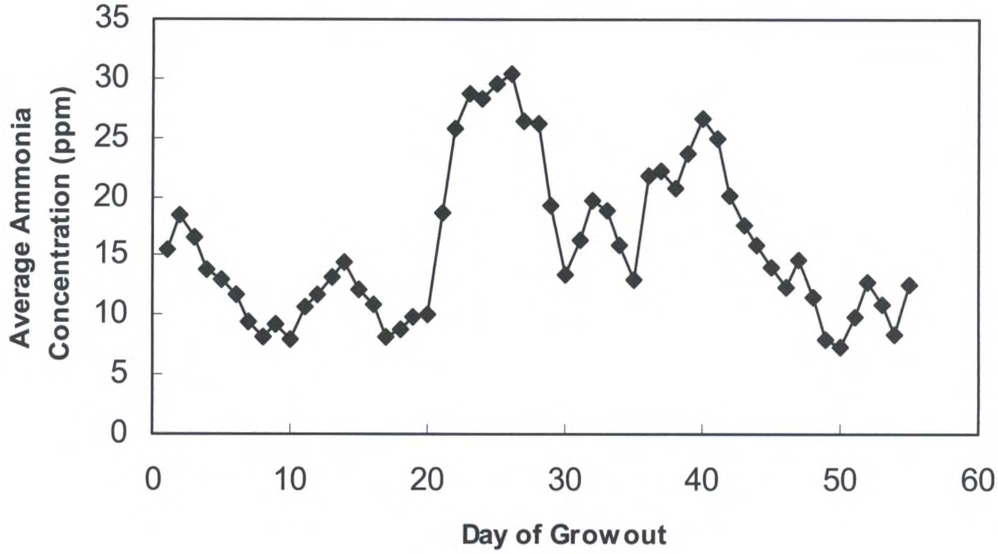

Figure 1. Average ammonia concentration in four broiler houses during two flocks.

The average daily ventilation rate increased as bird age increased (Fig. 2). Early in the flock, ventilation remained relatively constant at about 0.5 million m^3/day, but increased to 5 million m^3/day, by the time the birds were nearing 8 weeks of age.

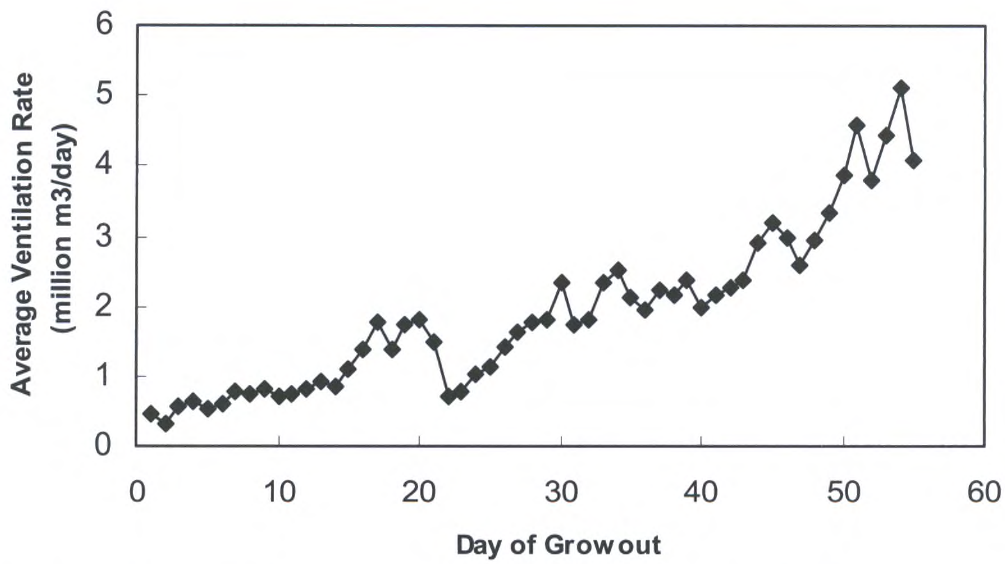

Figure 2. Average ventilation rate of four broiler houses during two flocks.

Ammonia emissions from the four houses started out relatively low (less than 5 kg/day), then increased until the 5th or 6th week of the growout, where it peaked at around 25 kg/day (Fig. 3). The average amount of ammonia emitted was 14.9 kg/day.

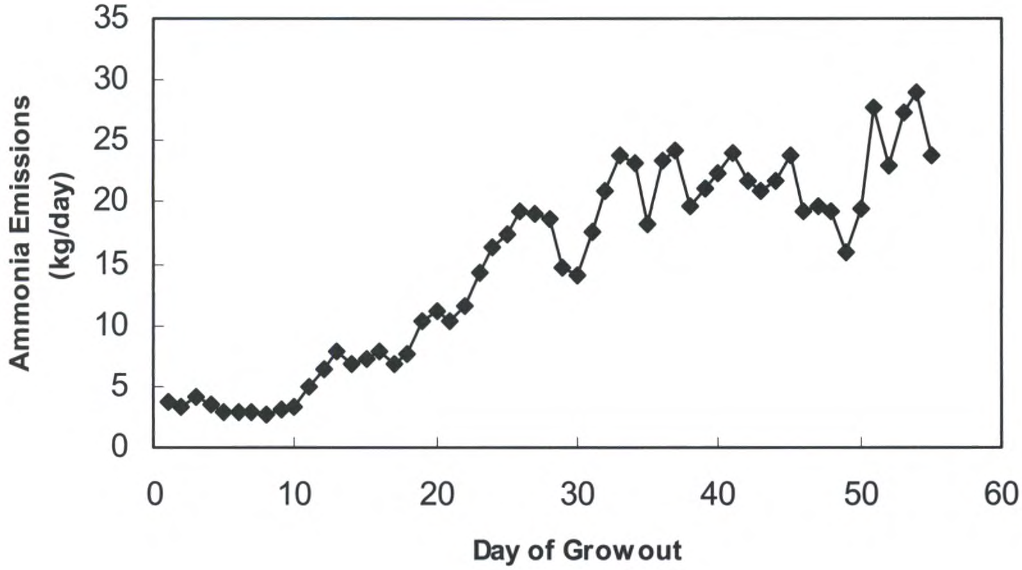

Figure 3. Average ammonia emissions of four broiler houses during two flocks.

Ammonia fluxes using small chambers were highly correlated to measured emissions, although the values tended to be higher (Fig. 4). One possible reason the fluxes could have been higher than actual emissions was the flux measurements were concentrated near the center 2/3[rds] of the house (near the water and feed). Emissions are probably lower near the walls of the houses.

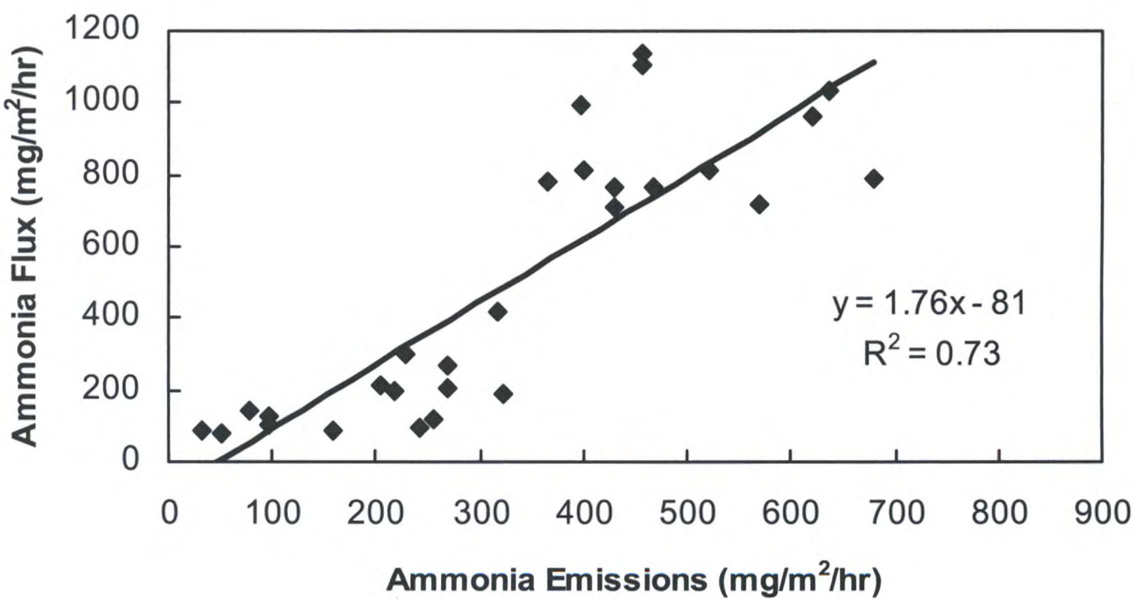

Figure 4. Relationship between ammonia fluxes using small chambers and ammonia emissions.

Ammonia fluxes stayed relatively low for the first three weeks of the growout, then increased sharply until week 5 (Fig. 5). The average ammonia flux was 501 $mg/m^2 \cdot hr$ or about 22.5 kg/house/day. Nitrous oxide fluxes were low (10-20 $mg/m^2 \cdot hr$) and varied little throughout the growout (data not shown). These rates were equivalent to around 0.5 kg N_2O per house per day.

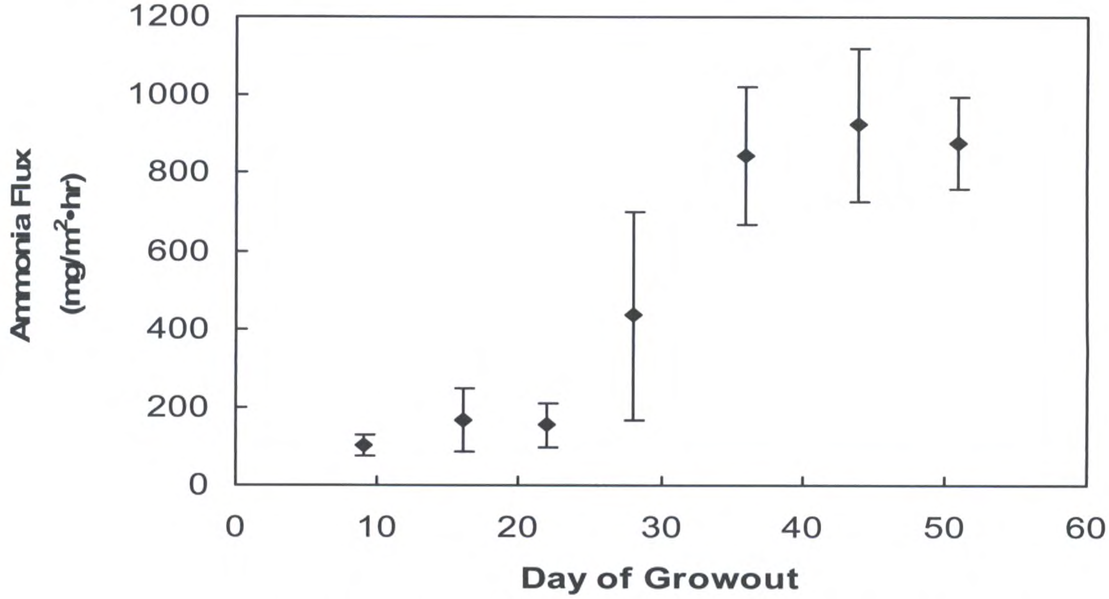

Figure 5. Ammonia fluxes from broiler litter in commercial houses as a function of bird age.

Development of an Ammonia Scrubber for Animal Rearing Facilities

A wet "scrubber" was developed to remove ammonia, particulate matter and pathogens from the air exhausted from animal houses. The prototype was constructed of wood. The scrubber has a 380 liter reservoir in the bottom filled with a dilute alum solution that is sprayed from the top of the box downward.

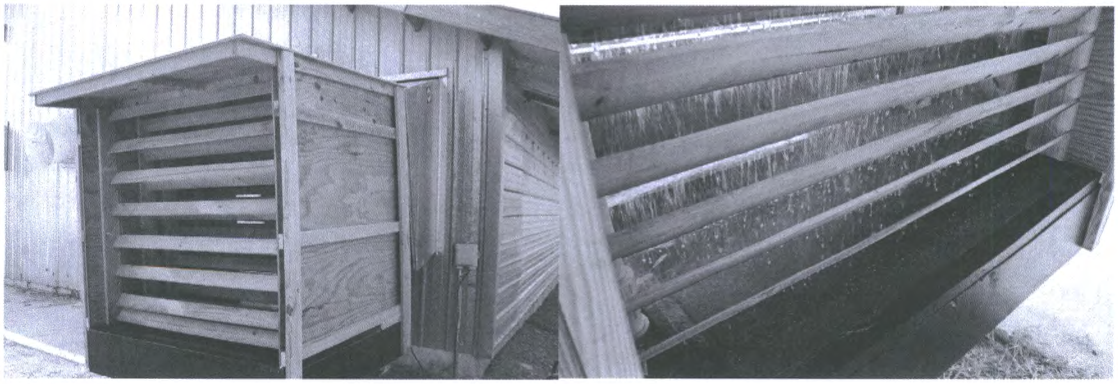

Figure 6. Ammonia and particulate scrubber for animal rearing facilities.

The amount of ammonium removed by the scrubber is dependent on the ventilation rate of the fan and on the ammonia concentration in the house. During the first test of the scrubber, which occurred in late spring, over 10 lbs of N were captured in a 24 hour period. This probably represents the best case scenario. During winter, when ventilation rates were lower, it took 3-4 days to capture this amount of N. When we first started using the scrubber, the trapping solution consisted of a 10% alum solution. One of the products created when ammonia reacts with the alum solution is the mineral ammonio-alunite [$NH_4Al_3(SO_4)_2(OH)_6$], which can be removed from solution. One of the other benefits of this technology is the reduction in soluble P in the soil once the spent scrubber solution is land-applied. Reducing soluble P should result in less P runoff and leaching. Therefore this technology provides benefits to both air and water quality.

Ammonia Emissions Following Litter Application

Ammonia emissions from small plots cropped to bermudagrass were near zero prior to litter application (Fig. 7). Once litter had been applied, losses from the surface application ranged from 1 to 7 kg N/ha/day. The peak after 12 and 13 days occurred after a small rain event. The total amount of N lost via volatilization during the two week period following application was 34 kg N/ha for the surface applied litter. This represents approximately 15% of the total N applied to the plots and over 100% of the NH_4-N applied (22 kg N/ha).

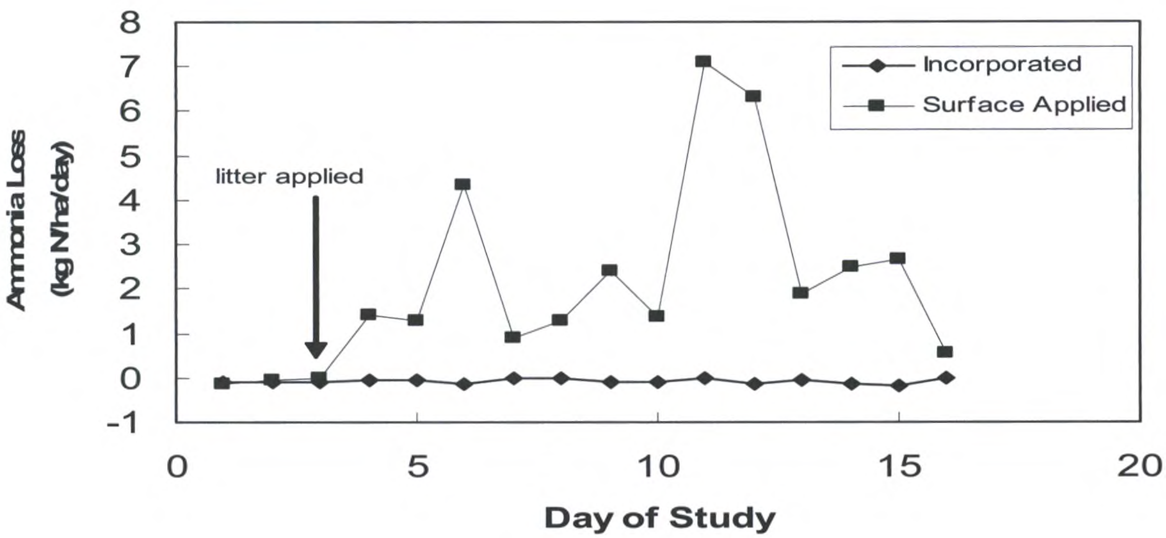

Figure 7. Ammonia emissions following poultry litter application to pastures

Earlier work by Pote et al. (2003) showed that incorporating litter into pastures significantly reduced phosphorus and nitrogen runoff compared to surface application. Litter incorporation has also been shown to increase forage yields by about 25% (Pote et al., 2003). The data in figure 7 indicate that improved yields may be due to improved nitrogen use efficiency.

Conclusions

Ammonia volatilization from poultry litter results in high levels of ammonia in poultry houses, which negatively impacts production. Average emission rates measured in this study were 14.9 kg/house/day (32.8 lbs/house/day). Ammonia emissions can be reduced with the addition of acidic compounds to litter, such as alum. Likewise, ammonia and particulate matter can be scrubbed from air exhausted from these facilities, allowing for the reuse of N as a fertilizer. Ammonia emissions from land-applied litter decreased to zero when litter was incorporated. These data indicate that there are several BMPs that can be used to reduce ammonia loss from poultry litter.

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Economics

Designing Policy Instruments for Controlling Air and Water Emissions from Agriculture

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Abstract

Impacts of agricultural production on water quality have long been a leading environmental policy concern and have stimulated a large literature on the economics of designing policy instruments to control water quality impacts of agriculture. This literature has been particularly concerned with how to contend with the measurement problems that emerge from the nonpoint character of water pollution from agriculture. More recently, there is growing interest in the air quality impacts of agriculture, particularly wastes from confined animal operations. Because air and water quality impacts of agriculture are produced jointly, initiatives to protect one environmental medium have impacts on the other. This paper explores the design of policy instruments to simultaneously address air and water emissions from agriculture, drawing on and extending results from prior literature.

Introduction

While agriculture has long been recognized as a source of air and water pollutants, government regulation of agriculture for environmental protection has evolved slowly by comparison to other economic sectors (Shortle and Abler 1999). Since the enactment of the 1972 Clean Water Act (CWA), the principal approach to water quality protection in the US has been effluent limits on industrial and municipal sources of water pollution. While these controls have done much to improve the quality of the nation's surface waters, water quality goals in many rivers, lakes and estuaries have not been met, often because significant nonpoint sources of water pollution, principally agricultural, remain largely unregulated (Ribaudo 2001). Similarly, the air quality regulations developed under legislation such as the Clean Air Act and the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) have been largely focused on easily identifiable and regulated point sources such as power plants and factories, and to highway vehicles (Ribaudo and Weinberg 2005).

Policy makers have taken increasing notice of the air and water quality impacts of agriculture, and there are increasing pressures to tackle these problems, especially those resulting from water pollution. The most important federal water quality initiatives are new rules beefing-up regulation of Confined Animal Feeding Operations (CAFOs) under the National Pollutant Discharge Elimination System (NPDES), and EPA's Total Maximum Daily Load (TMDL) Program. NPDES permits are required under the CWA for point sources of water pollution. The permits specify technology-based effluent standards and are the primary regulatory tool for controlling point sources of water pollution under the CWA. EPA's TMDL Program is intended to guide compliance with Section 303(d) of the 1972 CWA, which requires states and certain other jurisdictions to identify waters that do not meet water quality standards even after point sources of pollution have installed the minimum required levels of pollution control technology. The law requires that these jurisdictions establish priority rankings for impaired waters and develop TMDLs for these waters. A TMDL specifies the maximum amount of a pollutant that a water body can receive and still meet water quality standards, and allocates pollutant loadings among point and nonpoint pollutant sources. While TMDLs were required by the 1972 CWA, there was little progress in developing them until EPA's recently initiated TMDL Program. Nutrients from agriculture are a major cause of remaining water quality problems and, accordingly, a major target in TMDL development (EPA 2006)

Concern for air quality impacts of agricultural production tends to be less intense than those for water quality, reflecting the greater relative importance of agriculture to the nation's water quality problems. Yet, there are also new federal policy developments in this area, primarily addressing air emissions from animal feeding operations (Aillery et al. 2005). Most notably, EPA is developing new regulations for fine particulates. Ammonia from animal feeding operations (AFOs) is a major precursor of fine particulates, making AFOs in non-attainment areas with high concentrations of animals a likely target of regulations (Aillery et al. 2005, EPA 2000).

Important in the new policy developments addressing agriculture's environmental impacts is an emphasis on air and water quality impacts of wastes from intensive, confined animal production systems. This speaks to the multi-media character and policy significance of intensive, confined animal production systems. Also important is that the multi-media problems associated with animal wastes are being approached through uncoordinated single-medium regulations for protecting air and water quality. This single-medium approach fails to account for complex air-water interrelationships at various scales and may lead to unintended environmental consequences and unnecessary costs. For example, reducing ammonia losses to air by injecting animal waste directly into cropland can increase the amount of nitrogen that enters water resources (Ribaudo and Weinberg 2005). Accordingly, policies designed to mitigate either air or water pollution problems should take into account the cross-media linkages to avoid unintended and adverse effects on environmental quality. More fundamentally, an explicit multi-media approach of air and water regulations is essential to cost-effective achievement of environmental goals.

Our interest in this paper is in the design of policy instruments to cost-effectively address air and water emissions that are jointly produced in agricultural production. We begin with a review of the existing literature on the design of policies to control nonpoint pollution. The relevance of this literature follows from the fact that both air and water emissions from agriculture are largely of the nonpoint type (i.e., diffuse and stochastic, making routine accurate metering by source prohibitively costly). The nonpoint policy design literature has been largely generated in response to the challenges of nonpoint water pollution control; indeed, the literature on the economics of air quality policy for agriculture is very thin. However, the theory and some key empirical findings from the water quality research are applicable to air quality protection in a single medium context. A limitation of the applied and theoretical nonpoint policy literature is that is focuses on management for the protection of a single medium. We discuss the particular challenges involved in addressing the nonpoint cross-media problem, and discuss policy tools that for addressing the multi-media problem. We focus in particular on pollution trading.

Nonpoint Pollution Policy Design Issues

Several features of nonpoint pollution problems complicate the choice of policy instruments. First, agricultural nonpoint emissions are generated diffusely over a potentially broad land area, prohibiting accurate and cost-effective measurement given existing monitoring technologies. Second, agricultural nonpoint emissions and the fate and transport of these emissions within airsheds and watersheds are highly stochastic due to stochastic environmental processes, such as weather, that move nutrients and other chemicals off of farms and transport them to air and water resources. Taken together, these first two features result in substantial uncertainty about the decision makers who are responsible for nonpoint pollution and about the degree of each farm's or household's responsibility. One implication is that the emissions-based instruments that economists usually advocate for cost-effective pollution control are eliminated from the set of nonpoint pollution control instruments (Shortle and Horan 2001). Other constructs must therefore be used to monitor performance and as a basis for the application of pollution control instruments. These could include polluting inputs such as fertilizers and pesticides, pollution-reducing practices such as nutrient management and pollution control equipment, or estimates of emissions based on observations of on-farm management practices. In each case, monitoring is likely to be costly and imperfect, creating the potential for moral hazard with respect to imperfectly-monitored practices (e.g., shirking on a contract to undertake a particular practice because the implementation of the practice cannot be verified) and substitutions to non-monitored practices having adverse environmental consequences.

A third characteristic of nonpoint emissions is that many site-specific factors such as hydrology, climate, and location often play key roles in determining the processes that move and transport emissions, as well as the eventual environmental and economic impacts of these emissions. The result is extreme spatial variation in the feasibility, effectiveness and cost of technical options for reducing emissions. This greatly limits the applicability of the uniform technology-based regulatory approaches that are often used to control point sources (see e.g., Shortle and Horan 2001).

An array of innovative policy instruments (e.g., taxes and/or subsidies on practices affecting nonpoint pollution, taxes on ambient concentrations of pollutants in environmental media, taxes and/or subsidies tied to farm level environmental performance indicators, contracts for adoption of best management practices) have been proposed to meet the unique challenges nonpoint pollution control. Economic research indicates that no approach offers a panacea, but some have more merit than others (Shortle and Horan 2001). Among the later is pollution trading. This approach has significant merit as means for minimizing the costs of achieving environmental objectives, and has also achieved a high degree of political acceptance, and is the focus of the remainder of the paper.

We begin with a discussion of single-medium trading mechanisms that could be used to address air or water pollution from agricultural sources. Single medium trading is the primary focus of the literature on the design of trading mechanisms, and it is the focus of current trading initiatives. We then take up the challenges of multi-media trading.

Single Medium Trading

Pollution trading is a mechanism for allocating pollution loads among alternative sources in order to achieve an overall pollution load target (e.g., TMDL, mean annual loads, etc.) set by environmental authorities. Water pollution trading involving both point and nonpoint sources is being promoted by the US EPA, has been adopted by several states and some multi-state regional water quality authorities, and is being actively considered by still others as means for achieving water quality goals, especially within the context of EPA's TMDL Program.

The development of water quality trading is part of a broader trend towards the use of market-based strategies to address environmental and natural resource problems. Trading has already become, for example, a major tool for air quality protection, and is of great interest as a mechanism for managing greenhouse gas emissions as well as water quality and other environmental resources (OECD 2002). This broad interest in trading has a variety of origins, but economic arguments, and increasingly empirical evidence, about the potential cost-savings from trading by comparison to traditional command-and-control approaches have been particularly compelling (OECD 2002; Hahn 2000; Tietenberg 2006; EPA 2001).

The economic appeal of the mechanism is that trading can ensure that environmental quality goals are achieved cost-effectively because individual polluters will respond to the market in a way that allocates load reductions at minimum costs. Cost minimization essentially requires allocating greater pollution abatement to sources with lower costs than to sources with higher costs. Trading achieves this outcome in theory by creating incentives for high cost sources to pay low cost sources to reduce their discharge (subject to restrictions that water quality is equal or better as a result of the trade). As high cost (HC) sources can pay low cost (LC) sources an amount less than the amount it would cost HC sources to make the reduction, but greater than the amount actually incurred by LC sources, trading is to their mutual benefit. But this market system can only work to provide economic and environmental benefits if the markets are properly designed and implemented. Initial experiments in water quality trading have perhaps suffered because of poor market design (Ribaudo et. al. 1999).

The fundamental element of any trading system is a tradeable permit. Permits define legally allowable emissions; tradeable permits allow sources to adjust their legal allowances through market transactions. The number of permits must be set appropriately for environmental quality goals to be attained. Moreover, rules governing trading in well-designed programs facilitate trading to promote the economic objective of cost-minimization, while assuring that environmental quality goals will be met after trades. The key challenges in designing trading programs are to assign the correct number of permits and to design rules that foster the dual environmental and economic objectives.

Load Limits and Permits

For a trading program to reliably satisfy environmental goals, it is essential to specify the maximum load from point and nonpoint sources consistent with the goals, and to cap the number of available emissions permits to satisfy this maximum load. The simple creation of trading as an option for reducing effluents is not enough to lead to trading or to trading that achieves water quality goals. As emphasized recently by King (2005), markets are not an alternative to water quality regulations. Markets are fundamentally and most appropriately viewed as a mechanism for allocating emissions among sources within the context of a regulatory restriction on total loads. They should not be the determinant of the total load. In the current context where point sources generally face strong regulatory restrictions while nonpoint sources often do not, an essential key to success in trading is meaningful restrictions on nonpoint sources. Tight restrictions on point sources simply cannot produce the trades needed to achieve water quality goals where nonpoint sources are the major cause of water quality problems.

The main challenge for trading with nonpoint sources is the immeasurability of individual pollutant loads. Point source permits can be based on actual emissions as these are generally meterable. But nonpoint source emissions are unobservable. This problem is addressed in theory and practice by basing nonpoint permits on modeled or estimated nonpoint emissions. This means that farmers or other nonpoint sources must make observable and measurable management changes, either in production (i.e., nutrient management) or on the landscape (i.e., plant buffer strips), the water quality impacts of which are then estimated by a simulation model to gauge compliance.

Trading Rules: The Type of Trading

Within the context of an overall restriction, a key issue is the type of trading to implement to achieve the target. The most straight forward design for achieving water quality goals is the *cap-and-trade* model. A cap-and-trade program begins with an explicit determination of total allowable discharges. Permits for the total allowable discharges are then allocated among polluters. Methods for the initial allocation include auctions, lotteries, and "grandfathering" (Tietenberg 2006). The initial allocations can then be traded, to determine equilibrium allocations among sources. The main alternative to cap-and-trade is *credit trading*. In a credit trading program, polluters generate credits by reducing discharges below a baseline, typically defined as a legal limit on emissions.[3] Credits generated by one source may sold to another to offset emissions in excess of the legal limit. The earliest air pollution trading programs in the U.S. were of this type (Tietenberg 2006, Ellerman 2005). The subsequent SO2 and NOx trading programs are of the cap-and-trade type.

Cap-and-trade systems allow planners to dispense with the knotty issues involved in defining individual baselines for credit generation, and focus instead on total allowable level of pollution. An emerging literature indicates that cap-and-trade programs promise both better environmental and economic performance than credit-trading (Shabman et al., 2002; Dewees 2001), although the US EPA's water quality trading policy calls for the credit trading approach.

Trading Rules: Nonpoint Risk and Trading Ratios

Point source permits based on actual emissions and nonpoint source permits based on estimated emissions are fundamentally different things, and so trading them on a one-for-one basis would be like trading apples for oranges. A trading ratio is generally used to account for these differences. In water quality markets the trading ratio is usually only applied to trades involving point and nonpoint sources, and it is defined as the required reduction in emissions from a nonpoint source that are needed for a point source to increase emissions by one unit. Essentially, the trade ratio is used to define equivalence between point and nonpoint loads.

Well-designed trading ratios are influenced by the uncertainty about actual loads stemming from the measurement problem, but also by the inherent riskiness of nonpoint emissions resulting from the inherent variability or stochasticity of nonpoint loads (often due to weather-related events). Accordingly, nonpoint pollution cannot be controlled deterministically. This nonpoint risk has important implications for the design of the trading ratio. There are two opposing perspectives on this issue.

The most common perspective in practice is that *diverting controls from point sources to nonpoint sources is risky*. This perspective comes from the view that the appropriate policy objective is to maintain a particular level of control of emissions. Point source controls are viewed as relatively certain, since point source emissions are not highly stochastic and they are fairly easily measured. In contrast, nonpoint controls are highly uncertain due to the stochastic and unobservable nature of nonpoint emissions. Trades that involve point sources purchasing nonpoint permits are therefore seen as reducing the certainty of controls, creating risk. The best policy response in this case is to increase the trading ratio (Horan 2001). On the one hand, a larger ratio provides a margin of safety as point sources must purchase more nonpoint permits in order to increase their emissions. On the other hand, a larger ratio increases the cost of purchasing nonpoint permits, thereby discouraging trades between point and nonpoint sources. Typical ratios used in practice are greater than unity, and range between 2:1 and 3:1 to address this margin of safety issue (Horan 2001).

The second perspective on nonpoint risk comes from the economic theory on point-nonpoint trading (Shortle 1990; Malik et al. 1993). Here, the appropriate policy objective, consistent with TMDLs and other water quality goals, is to reduce the probability of water quality damages from point source and nonpoint source emissions. Given this objective, it turns out that *failure to control nonpoint emissions is risky*. The reason is that highly variable nonpoint emissions result in highly variable damages, and it is this variability in damage costs that are risky. Since risk is socially costly, the appropriate policy response is to reduce the trading ratio in order to encourage more nonpoint controls and thereby reduce this important source of risk. Economic welfare theory indicates that this perspective is the correct one, in which case the large ratios used in practice are counter-productive in two important ways: (i) they increase rather than decrease water quality risk, thereby increasing the economic damages from water quality impairments, and (ii) they discourage trades involving nonpoint sources, which can only increase aggregate control costs. Economic simulations find optimal trading ratios to be much lower than those found in most trading programs, in large part because of these risk effects (Horan et. al. 2002a,b; Horan et al. 2004).

Although we have discussed the choice of permit levels and trading ratio separately, they are in fact a joint decision. For instance, Horan and Shortle (2005) have shown that if the trading authority only has power to choose the trading ratio (and not permit levels), then the economically optimal trading ratio may be much different than the ratio the agency should choose if it has control over both choices.

Multimedia Trading

Fully realizing the benefits of trading for air and water quality programs cannot be achieved through the application of single media trading schemes. The interdependence of the problems requires the development of multi-media trading mechanisms that can effectively address complementarities and tradeoffs, facilitating the achievement of beneficial trades while assuring the achievement of air and water quality objectives. Essentially, multimedia trading would create property rights for sources that emit pollutants affecting different environmental media simultaneously. Like any other trading system, multimedia trading must satisfy four basic conditions outlined by the OECD (2002): (i) a quantitative environmental performance target to be achieved individually or collectively; (ii) a defined spatial and temporal flexibility given to regulated agents in the choice of location of resource extraction or pollution emission; (iii) enforcement capacities to ensure that actual performance of agents matches their obligations and initial allocation of permits/property rights (OECD 2002). And like single media trading, the nonpoint character of agricultural emissions pose problems that can be addressed with careful attention to program design. The primary new challenges, as we describe below, really stem from the interdependence of the air and water systems.

Joint Determination of Air and Water Quality Goals

Current policy frameworks generally call for limits on emissions to air and water to be determined independently. To the extent that air and water emissions are jointly produced by the management decisions of nonpoint source and point source emitters, and to the extent that there are cross-media interactions from the emissions (e.g. air emissions may be deposited into water resources, while volatilization of pollutants may occur from water resources), a multi-media approach will call for integrated goal setting. The need for integrated goal setting will hold with even more force if costs and benefits are considered in goal-setting (as is required under the Clean Water Act but not the Clean Air Act).

Multiple Markets

Single media trading schemes generally imply participation in a single market. But cost-effective multi-media approaches will imply a market for each medium, with rules governing trades both within and between markets. Environmental groups are often uncomfortable with the notion of trading between markets, for fear that such trades could improve environmental quality in one market at the expense of environmental quality in the other. But this concern can be alleviated with a coordinated approach to program design. Moreover, it should be stressed that an uncoordinated approach involving separate markets can run a greater risk of reducing environmental quality in one or more of the media. The reason is that air and water emissions are jointly produced by the management decisions of an emitter – particularly agricultural nonpoint sources. If the markets are not developed jointly, then trades in one market could lead to investments that could improve the environmental quality of one media at the expense of the other, reducing the ability to cost-effectively attain goals in either market. Indeed, it is well-established that uncoordinated environmental programs can lead to unintended, adverse economic and environmental consequences (e.g., Weinberg and Kling 1996). On the other hand, a coordinated approach can lead to cross-subsidization of emissions reductions, as a trade to reduce air emissions can lead to a change in management practices that simultaneously reduces both air and water emissions (see Horan et al. 2004 for a discussion of a similar issue in a different context). But whether or not this cross-subsidization can occur will depend on whether the trading rules are designed to take advantage of this possibility, so that a polluter can attain credit in both markets from a single management action.

Conclusion

Multi-media pollution trading has great potential as a policy approach for addressing air and water quality problems, but its usefulness ultimately depends on how the program is designed. Research, that utilizes the lessons learned from single-media markets, is needed to fully comprehend the design issues that will be pertinent for the multi-media case.

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Reducing Ammonia Emissions from Animal Operations: Potential Conflicts with Water Quality Policy

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Abstract

Animal waste from confined animal feeding operations is a potential source of air and water quality degradation from evaporation of gases, and runoff to surface water and leaching to ground water. The multi-media nature of pollution from animal waste poses challenges to farmers and to environmental protection agencies. Failure to account for the multi-media nature of animal waste in policy design and implementation can lead to unintended consequences in terms of costs to farmers and degradations of environmental quality. This paper assesses the potential economic and environmental tradeoffs between water quality policies and air quality policies that require the animal sector to take potentially costly measures to abate nitrogen pollution (nitrates and ammonia). We found that implementing ammonia emission restrictions on top of existing Clean Water Act requirements for Concentrated Animal Feeding Operations could increase the cost of meeting water quality requirements, degrade water quality, and impose costs on farmers that could have been avoided if policies were coordinated from the start.

Introduction

Animal production generates byproducts such as organic matter, urea, ammonia, nitrous oxide, phosphorus, methane, carbon dioxide, pathogens, antibiotics, and hormones. Without proper management, these materials can degrade surface water, ground water, air quality, and soils. Mitigating pollution can be difficult when more than one environmental medium is affected by a single pollution source. The correction of a single problem without simultaneously addressing others may not increase societal welfare as much as anticipated, and may even decrease it. U.S. environmental laws typically address only a single environmental medium, and coordination between policies is rare.

Nitrogen emissions from confined Animal Feeding Operations (AFOs) are a good example. Animal waste can contain significant amounts of nitrogen. Nitrogen moves freely between the soil, air, and water, and there is a high degree of interdependence between the forms and paths it takes. Emissions to air and water are linked by the biological and chemical processes that produce various nitrogen compounds. Nitrogen enters the system in animal feed. Some of the nitrogen is retained in the animal products (meat, milk, eggs), but as much as 95 percent is excreted in urine and manure (Follett and Hatfield, 2001).

Manure can collect in or under the production house for a few hours or several years, depending on the collection system. Production houses are ventilated to expel gases that are emitted, including ammonia. The manure is eventually removed from the house to a storage structure (lagoon, tank, pit, or slab) and stored anywhere from a few days to many months. Losses of nitrogen to the air and water can occur during this time, depending on the system and the extent of contact with rain and wind. The stored manure is eventually transported to fields where it is applied. Losses to air and water from the field vary, depending on application method, timing, and rate.

The form nitrogen takes in its journey from animal to field depends on a host of factors, including storage technology, manure moisture content, temperature, air flow, pH, and the presence of micro-organisms. Reducing nitrogen movement along one path by changing its form will increase nitrogen movement along a different path (National Research Council, 2003). For example, reducing ammonia emissions from a field by injecting waste directly into the soil increases the amount of nitrate available for crops, but also the risk to water resources (Oenema et al., 2001; Abt Associates, 2000). Ignoring the interactions of the nitrogen cycle in developing manure management policies could lead to unintended and adverse effects on producers' costs of managing manure and environmental quality.

Current Clean Water Act (CWA) regulations require that animal feeding operations meeting certain size and discharge characteristics (known as Concentrated Animal Feeding Operations or CAFOs) obtain a National Pollutant Discharge Elimination System (NPDES) permit (U.S. EPA, 2003). The NPDES permit for CAFOs requires a nutrient management plan covering the land on farms receiving manure. The plan must specify an application rate for manure nitrogen or phosphorus based on the agronomic needs of the crop. Research has found that most CAFOs over-apply manure nutrients, meaning that implementing a nutrient management plan could significantly increase the cost of land-applying manure (Ribaudo et al., 2003). Increased hauling costs make up a significant percentage of the total cost of meeting the land application requirements (Ribaudo et al., 2003). In this analysis we assume that CAFOs must implement a nitrogen-based nutrient management plan that contains an application standard based on the nitrogen needs of crops receiving manure.

Atmospheric emissions of pollutants are regulated by the Clean Air Act (CAA) and the Comprehensive Environmental Response Compensation and Liability Act (CERCLA). AFOs are not explicitly covered by either, but concerns over ammonia emissions are prompting discussions about how these laws might be used to regulate such emissions. In this analysis we assume that all or some animal feeding operations are required to reduce ammonia emissions.

When a production activity pollutes more than one environmental medium, addressing a single problem can lead to further resource misallocations (Lipsey and Lancaster, 1956). In this paper we examine the economic and environmental consequences of adding ammonia reduction requirements on top of the CWA requirements already placed on animal feeding operations.

Methods and Results

This study uses three separate but related analyses to capture a broad range of economic decisions (and consequences) that result from farmers' meeting environmental regulations. Data from the 1998 Agricultural Resources Management Survey (ARMS) of hog producers were used to estimate the tradeoffs that occur at the farm level when air and water policies are introduced. The broader, national scale impacts of controlling runoff and emissions, including welfare impacts on both producers and consumers and regional shifts in production, were examined with a national model of the agriculture sector. A case study of the Chesapeake Bay watershed was used to demonstrate the challenges facing farms and resource management agencies when hypothetical ammonia emission reductions are required for farms meeting the CAFO regulations in a region where land for applying manure is relatively scarce.

At the heart of all three analyses are nitrogen loss coefficients that are derived from a mass-balance accounting of nitrogen in manure. We used as our starting point the manure management "trains" (MMTs) or paths developed by EPA (2004). This inventory of current animal production and manure management systems takes a mass balance approach that is central to our study. Nitrogen in manure excreted in the production facility is accounted for through storage and application to fields. However, these MMTs did not include management practices for reducing ammonia emissions. We adapted MMTs for systems incorporating recognized ammonia reduction technologies such as lagoon covers and field injection by using reduction efficiencies reported in the published scientific literature to redirect nitrogen along the different paths (table 1). In all three analyses, CAFOs were first required to meet CWA requirements for land application of manure. Then, restrictions on ammonia emissions were applied to all AFOs.

Farm-Level Analysis

To examine the effect of potentially conflicting policies on a farmer's production decisions, we constructed a hog farm economic model. A positive mathematical programming model with calibrated cost functions captures the essential farm-level tradeoffs between ammonia air emissions and nitrogen water discharges for hog operations that are large enough to be considered CAFOs. Farmers maximize profits given input prices, output prices, regulatory requirements, and available cropland by choosing a manure management technology, the amount of land on which to spread manure, the acreage of each crop to plant, the amount of commercial fertilizer to purchase, and the number of hogs to produce. We assume that the baseline manure storage system (pit or lagoon) would not change. Farmers would meet environmental requirements by making adjustments within these systems. Water quality impacts are assumed to be directly related to the amount of nitrogen applied to cropland that is in excess of crop needs, after accounting for losses to the atmosphere. Air emissions are derived from total animal production and the type of storage/handling technology employed by the operation. For operations using pits for storage (slurry), ammonia nitrogen emissions are constrained to 10 percent above the minimum obtainable if all manure is injected into the soil. For lagoon operations, ammonia emissions are constrained to 20 percent above what is obtainable if lagoons are covered.

Table 1. Examples of manure management systems and nitrogen losses

Animal	System	N excreted	Losses from building	Losses from storage	Losses from field	Total losses to air	Total available for crops
		lbs N/head/year					
Hogs	Lagoon - uncovered	18.3	4.9	9.5	0.8	15.2	3.1
Hogs	Lagoon-covered	18.3	4.9	0.5	2.8	8.2	10.1
Hogs	Deep pit-surface apply	18.3	6	0	2.6	8.6	9.7
Hogs	Deep pit-incorporate	18.3	6	0	0.4	6.4	11.9
Dairy	Flush barn – surface apply	220	44	125	11.2	180.2	39.8
Dairy	Flush barn – incorporate	220	44	125	2.8	171.8	48.2
Dairy	Daily spread – surface apply	220	15.2	2.2	37.7	55.1	164.9
Dairy	Daily spread – incorporate	220	15.2	2.2	8.3	25.7	194.3
Poultry	Surface apply	0.9	0.18	0.03	0.17	0.38	0.51
Poultry	Incorporate	0.9	0.18	0.03	0.04	0.25	0.65
Poultry	Alum – surface apply	0.9	0.03	0.04	0.21	0.28	0.62
Fed beef	Solid storage - surface apply	102	0	20.8	13.8	34.6	67.4
Fed beef	Solid storage - incorporate	102	0	20.8	0.7	21.5	80.5

The model is calibrated with data from the 1998 USDA-ARMS survey of hog operations. In the model, nitrogen in waste may be released into the atmosphere as ammonia or preserved in the manure storage and handling system until applied to cropland. We consider two technological options for reducing ammonia emissions: injection of manure into the soil and covering lagoons.

Meeting a nitrogen fertilizer application standard was estimated to reduce hog enterprise profits by 6.9 percent, while reducing excess nitrogen applications to the soil by 100 percent. Almost 70 percent of the manure that had been applied on the farm was moved off the farm to adjacent land, increasing hauling costs by $205 million (table 2).

When air quality-based ammonia emission controls are also required, profits are reduced an additional 8.9 percent. Only part of this is due directly to ammonia management costs. Covering tanks and lagoons and injecting waste into the soil reduces ammonia emissions but increases the quantity of manure nitrogen farmers must deal with. As a consequence, the cost of meeting the CWA requirements increases by 12.8 percent as more land is needed for spreading manure and hauling costs increase. This impact on water-quality control costs might not be anticipated when ammonia control policies are developed.

Table 2 . Production, profits, emissions, and technology adoption under nitrogen application standard (NAS) to protect water and ammonia nitrogen emission standards (ANS).

	1. Base	2. NAS		3. NAS+ANS	
			% chg.		% chg.
Hogs (mil. cwt.)	119.10	117.96	-0.96	115.61	-2.93
Total profits (mil. $)	3,700	3,487	-5.77	3,187	-13.87
Hog enterprise profits (mil. $)	3,047	2,837	-6.89	2,568	-15.72
Ammonia N - storage (1000 tons)	327.5	325.3	-0.68	198.8	-39.29
Ammonia N - field (1000 tons)	33.8	34.9	3.38	52.1	54.15
Ammonia N – total (1000 tons)	361.3	360.2	-0.30	250.9	-30.55
Excess N - soil (1000 tons)	137.7	0.0	-100.00	0.0	-100.00
Application rate (factor of agronomic rate)	7.3	1.0	-86.38	1.0	-86.38
Manure transport costs (mil. $)	0.0	205.6	-	231.9	-
Manure N on-farm (1000 tons)	183.6	51.8	-71.81	42.3	-76.96
Manure N off-farm (1000 tons)	0.0	127.7	-	235.7	-
Cover lagoon (% farms, all farms)	0.00	0.00	0.00	36.42	-
Inject manure (% land, all farms)	25.56	22.55	-11.78	37.46	46.54

National Effects

Having examined the implications of addressing nitrogen concerns over water and air quality for one farm-level sector, we now take a larger view. Here, potential changes in commodity prices and shifts in production among regions are estimated assuming adoption of land application standards for manure generated on CAFOs and reductions in ammonia emissions from manure generated on *all* animal feeding operations. Tradeoffs are not limited to the farm, but extend to regions and to consumers.

We use the U.S. Regional Agricultural Sector Model (USMP model) to assess secondary price and quantity interactions across crop and animal production (USMP; House et al., 1999) at the national and regional levels. The USMP model accounts for production of major crops (corn, soybeans, sorghum, oats, barley, wheat, cotton, rice, hay, and silage) and confined animals (beef, dairy, swine, and poultry) across 10 geographic regions, comprising approximately 75 percent of crop production and more than 90 percent of livestock and poultry production in the United States. USMP is a comparative-static, spatial, and market equilibrium model that incorporates agricultural commodity, supply, and demand, environmental impacts, and policy measures.

Various adjustments to crop rotation, tillage, production, and technology can be made to meet the nitrogen application or ammonia emission constraints. The composition of cropping or animal production could change to alter the amount of manure nutrients demanded (by the crop sector) or supplied. Storage, handling, or application technologies can reduce ammonia emissions and alter nitrogen content of manure. Our model selects the optimal combination of technology, crop, and animal changes across the sectors and regions in order to minimize the net cost to society of meeting different environmental constraints (measured as changes in net returns and consumer surplus). Storage, handling, and application technologies, available in the model for meeting the CAFO nutrient standards and for reducing AFO emissions of nitrogen, are consistent with those in the farm-level analysis.[1] We also consider treatment of poultry litter with aluminum sulfate (alum) to reduce nitrogen storage losses and to decrease the bioavailability[2] of phosphorus. Our baseline for comparison is the USDA 2010 baseline projections for prices and production.

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[1] We assume a crop producer willingness to accept manure of 30 percent, meaning that up to 30 percent of available cropland in each region will utilize manure nutrients. Alternative levels of manure utilization have been considered, but are not included here.

[2] Bioavailability of phosphorus refers to the amount of phosphorus in runoff that is available for aquatic and terrestrial plant growth.

Meeting the Clean Water Act requirements raises the storage, handling, treatment, and application costs for the national agricultural sector by $534 million. Nitrogen runoff declines about 12 percent, due to CAFOs meeting a nitrogen application standard and an overall decline in farm production resulting from higher production costs. Because of the decline in production, prices rise for consumers. The total cost to producers and consumers from the CWA requirements was estimated at $850 million per year. An unintended benefit is a 10-percent reduction in overall ammonia emissions from the animal sector, brought about by the decline in production as well as a change in the mix of animals produced and regional production shifts.

When ammonia emission restrictions are required for all AFOs, an important environmental tradeoff emerges (figure 1). A 10-percent reduction in ammonia emissions for each farm results in an increase in total nitrogen runoff, even though the total number of animals declines. The reason is that ammonia restrictions are applied not only to CAFOs that must continue to meet nitrogen application standards, but to smaller AFOs that do not have to meet such standards. The increased nitrogen content of manure due to emission-reducing management measures results in higher nitrogen application rates and higher nitrogen losses to surface water. The water quality gains from the CWA requirements for CAFOs are reduced as more stringent ammonia reductions are required. The increase in nitrogen runoff could be avoided if CWA requirements were extended to all farms that must reduce ammonia emissions. The consequence, however, would be higher costs for producers and higher prices for consumers.

Chesapeake Bay Case Study

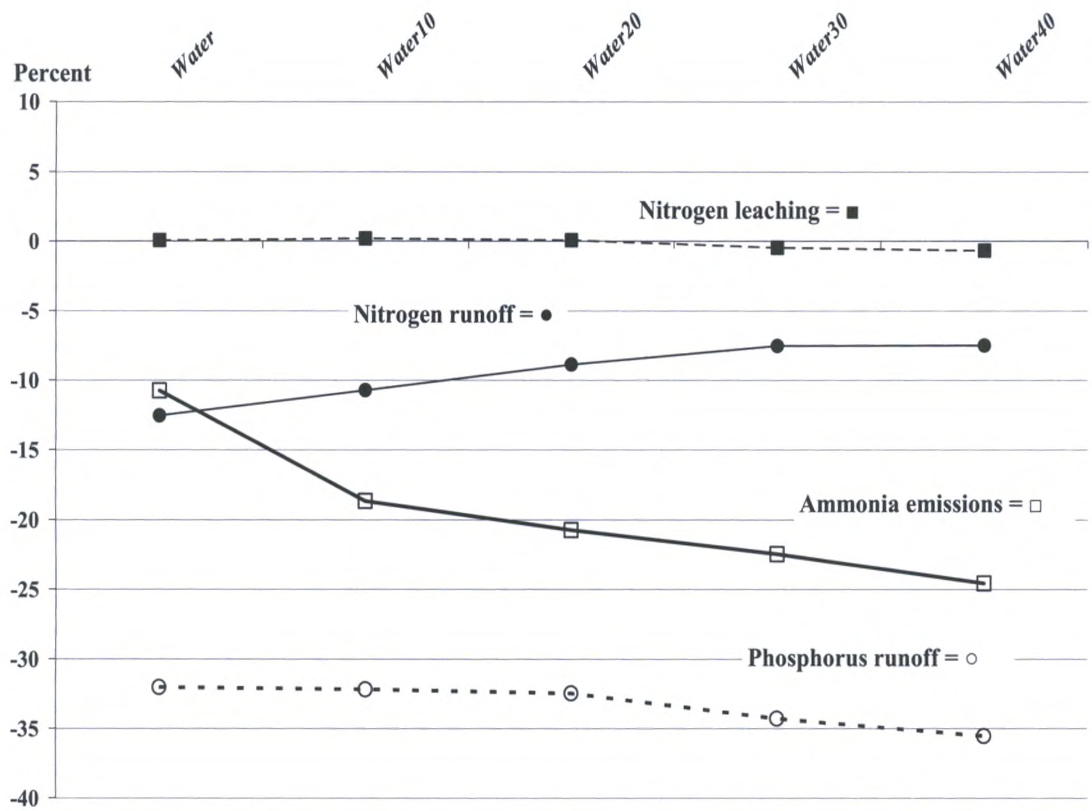
Acquiring enough land for spreading manure to meet a nutrient management plan is an issue in regions with high concentrations of animals relative to land available for spreading. The Chesapeake Bay Watershed (CBW) is one such region. Nutrient discharges to water in the region have resulted in eutrophication and related ecological shifts that have harmed wildlife and aquatic resources. Manure from confined animal feeding operations is a primary source of nutrient runoff and local air emissions (Follett and Hatfield, 2001).

Ribaudo et al. (2003) found that if cropland application remains the primary disposal method for manure, implementing nutrient management regulations would pose significant challenges. More than 30 percent of CBW crop farms would need to accept manure in order for the region's annual manure production to be land-applied according to a nitrogen-based management plan. Any change in manure handling that increases the nutrient content would only exacerbate the problem and raise the costs of hauling and applying manure.

In this analysis we used a regional modeling framework designed to capture spatial consideration in manure production and land availability for manure spreading (Aillery et al., 2005). The model and its results reflect a regional planning perspective in evaluating key cost determinants and alternative policy strategies at a watershed scale. The model does not restrict ammonia emissions directly. Instead, it requires producers to implement technologies for reducing emissions, including lagoon covers, injection, and treating poultry litter with alum. We assume that the operators of 30 percent of the cropland in the watershed are willing to use manure as a nutrient source (nationally, the willingness to accept manure is in the 10 to 20 percent range).

The annual cost of meeting a nitrogen-based application standard for water quality is estimated to be about $30 million if applied only to CAFOs (Case A in figure 1). When ammonia restrictions are also required for CAFOs, the cost of air emission-control practices totals about $9 million (Case B). However, with ammonia restrictions CAFOs must bear additional land application costs because the nitrogen content of manure increases. Roughly twice as much land on which to spread manure is required, resulting in a $9 million increase in annual hauling and application costs. The annual cost to CAFOs of reducing ammonia emissions is therefore estimated to be about $18 million.

Extending ammonia reduction requirements to all AFOs increases the total costs of reducing emissions by $32 million per year (Case C). However, as seen in the other analyses, the nitrogen content of manure produced by non-CAFOs increases because of the measures taken to reduce ammonia emissions (roughly doubles in this analysis). Unless this manure is spread according to a nutrient management plan, the risk of nitrogen runoff to the Bay is greatly increased.



Water – nitrogen application standard land receiving manure

Water10 – nitrogen application standard plus a 10 percent reduction in ammonia emissions

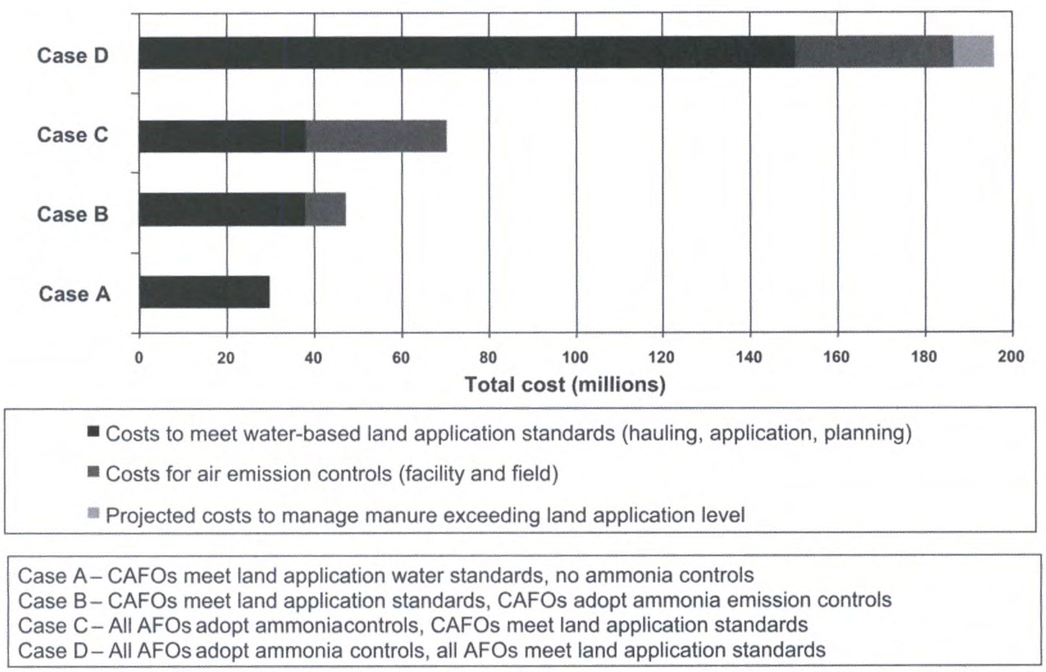
Water20 – nitrogen application standard plus a 20 percent reduction in ammonia emissions

Water30 – nitrogen application standard plus a 20 percent reduction in ammonia emissions

Water40 – nitrogen application standard plus a 20 percent reduction in ammonia emissions

Figure 1. Changes from baseline in ammonia and nutrient losses to the environment, U.S.

One solution would be to require all farms that must meet ammonia restrictions to also develop and implement a nutrient management plan to protect water quality (Case D). However, coordination of policies would increase overall annual costs for managing manure in the watershed to $187 million, and create an additional problem: what to do with excess manure. If all farms were to follow both a nutrient plan and reduce ammonia emissions, there would be inadequate farmland in the watershed under an assumed 30 percent willingness to accept manure to spread all manure produced at agronomic rates. Manure would have to be moved outside the watershed, or directed to other uses that do not require land application, such as energy or fertilizer production. If we use the rates Delaware pays to haul manure out of the State, moving the excess manure out of the CBW would add an additional $9 million per year to total manure management costs in the watershed.



Source: Aillery et al., Economic Research Service, 2005

Figure 2. Annual costs of meeting nitrogen application standards with alternative emissions controls on CAFOs and AFOs, Chesapeake Bay Watershed

Conclusions

Addressing the pollution problems generated by production activities can be difficult when more than one environmental medium is affected by a single pollution source. This paper illustrates the potential tradeoffs between air and water quality when nitrogen losses from animal feeding operations are policy targets. Nitrogen in manure can take a number of forms; reducing one form of nitrogen to protect one environmental medium can increase the amount of another form moving to a different medium.

Should ammonia emission standards induce farmers to adopt manure management practices that reduce air emissions, the manure applied to land would have a higher nitrogen content. Depending on how the air quality regulations are applied, this could have two impacts on CAFOs and water quality. First, those farms identified as CAFOs might need to increase the amount of land they are spreading on to meet nutrient application standards if they are also required to reduce ammonia emissions. This would be particularly costly in a region where animal concentrations are high and cropland available for spreading manure is relatively scarce. In our analysis of the costs of spreading manure in the Chesapeake Bay watershed, nitrogen content of manure increases substantially if ammonia restrictions are introduced, which would increase the costs of meeting nitrogen application standards. The higher cost of meeting water quality regulations might not be considered in an assessment that focuses on the cost of air quality regulations.

Second, a failure to coordinate water and air policies could lead to an unanticipated loss of water quality benefits. If air quality regulations were to result in States requiring ammonia reductions on smaller farms as well as current CAFOs, the water quality benefits of the CAFO regulations could be diluted by excess nutrient applications on the smaller farms. This was the case in both our regional and national analyses. Without regulations for spreading manure at agronomic rates, farms reducing ammonia emissions would be more likely to over-apply manure, thus increasing the potential for nitrogen discharges to surrounding waters. It would be difficult to achieve ammonia emission reductions and still maintain water quality gains of the CAFO regulations if water quality regulations were not extended to smaller operations. Doing so would increase the costs to producers and consumers, but provide greater overall environmental improvements.

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Valuation of Air Emissions from Livestock Operations and Options for Policy

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Abstract

Animal husbandry is a major emitter of methane, which is an important greenhouse gas. It is also a major emitter of ammonia, a precursor to fine particulate matter, which is arguably the number one environmentally related public health threat facing the nation. This paper presents an integrated process model of the engineering economics of technologies to reduce methane and ammonia emissions at dairy operations in California. Three policy options are explored including greenhouse gas offset credits for methane, particulate matter offset credits for ammonia, and expanded net metering policies to provide revenue for sale of electricity generated with methane. Individually any of policies appear sufficient to provide the economic incentive for farm operators to reduce emissions. This paper reports on initial steps to develop fully the integrated process model to provide guidance for policymakers.

Introduction

Animal husbandry is a major emitter of methane and ammonia in the United States. Methane, which is a greenhouse gas (GHG) with 23 times the potency of CO_2, constitutes nearly one tenth of all US GHG emissions. Although methane has a shorter residence time than CO_2, its radically higher effect makes it an attractive target for policy measures, especially in the near term. Ammonia, on the other hand, is a precursor to fine particulate matter ($PM_{2.5}$), arguably the number one environmentally related public health threat facing the nation.

The main technology to control methane emissions in animal husbandry involves using methane digesters that generate and collect methane from manure. The generated biogas can then be burned and converted into heat or electricity. Electricity generation through methane digesters reduces farmers' need to purchase electricity and also can create surplus electricity that is available for sale back onto the electricity grid. Control of methane is also a potential offset for CO_2 emissions with prospective value of tens of dollars per ton at forecasted levels of costs for CO_2 control in the regional programs under design in the US (RGGI, 2005). Control of ammonia, in contrast, has potential to be tied to particulate control policies offering offsets or emission reduction credits. However, a large fraction of the benefits from the control of methane and ammonia in animal husbandry accrue outside of existing markets and cannot be appropriated by individual dairy operations choosing whether to invest in methane and ammonia control technology. For example, reductions in GHG emissions from livestock operations are currently not economically rewarded. As a consequence, dairy operations face only limited incentives for emission control using methane digesters. This, in turn, can result in less than overall optimal adoption of emission control technology by the dairy industry.

In this study, we seek to examine the full potential for methane and ammonia control in animal husbandry. Our objectives are to identify: (1) the methane and ammonia emission reduction potential of manure process control; (2) the cost thresholds that determine sensible adoption of different emission control technologies; (3) the benefits from emission control that accrue outside the dairy industry; and (4) the policies or institutions that are necessary to achieve these benefits. This information will be essential to future public policy that may give shape to either the formation or new markets for emission reductions or to direct financial and technical assistance to methane and ammonia control in agriculture.

We select the California dairy industry for our application. California is a particularly well-suited study area, since it is the number one ranked dairy state in the US and represents about one fifth of all US cows and milk production. California dairy also generates nearly $5.4 billion in cash receipts and almost a billion dollars in exports, which makes it one of the economically most important agricultural sectors in California.

The quantity of manure generated by California cows is massive—over 70 billion tons each year—and amounts to more solid organic waste than is generated by the state's 35 million residents (US EPA 2006).

Problems associated with the dairy manure in California are heightened by the increase in the average dairy size and their concentration in areas with rapidly growing population and multitude of air quality problems. While California had about 4,000 dairies in 1992, the total number had dropped to 2,100 by 2004. During the same time period, the total number of cows increased from roughly 1.2 million to 1.7 million, meaning that the average number of cows per dairy more than doubled from about 370 in 1992 to over 800 in 2004. California dairy farming is concentrated in the Central and San Joaquin Valley regions, where the five largest dairy counties in the US (Tulare, Merced, Stanislaus, San Bernardino, Kings) are situated. These counties have roughly 1.1 million cows in total, which is about 12% of US dairy cows. Tulare County by itself has approximately 440,000 dairy cows (4.5% of all US dairy cows), which is more than the total number of cows in any US state outside California except for Wisconsin, New York, Pennsylvania, and Minnesota. These counties, as well as many other California counties with significant dairy presence, are also non-attainment areas for particulate matter and ozone, meaning that they do not meet the minimum federal air quality standards (EPA Green Book). Population growth in the top-five dairy counties in California was over 20% between 1990-2000, which is well over the state average of 13.6% (US Census), meaning that the human population exposure to pollution is worsening.

California has initiated several programs to encourage the treatment of manure using methane digesters. These programs include Dairy Power Production Program, Self-Generation Incentive Program, and the net metering Assembly Bill. The Dairy Power Production and Self-Generation Incentive Programs provide cost-share funding for capital investments towards new installations of methane digesters.[3] Assembly Bills 2228 (signed into law in 2002) and 728 (signed into law in 2005) require the state's three largest investor-owned utilities (PG&E, SCE, and SDG&E) to offer net metering to dairy farms that install methane digesters. These initiatives encourage the dairy industry to adopt methane digesters, but do so without considering all the costs and benefits associated with the reductions of methane and ammonia emissions.

In this paper, we develop an integrated model to examine the control of methane and ammonia in dairy farming. We pay special attention to comprehensive accounting of both private and social benefits and costs of methane and ammonia control. The analysis focuses on the interaction of methane and ammonia with climate, energy, and public health polices, including the potential use of offsets for GHG policies or regional air pollution policies. The model is designed to provide policymakers a tool to understand the technical and economic relationships in order to realize the benefits of managing air emissions and waste discharges from agriculture.

In the rest of this paper, we first explain the air pollution issues in dairy operations. Then, we describe the integrated process model of manure management, which constitutes the core of our analysis. The description of the model includes a depiction of baseline emissions, control technologies for ammonia and methane, and the potential electricity generation, green house gas reductions, and health benefits from the adoption of control technologies. Thereafter, we utilize the model to evaluate different policy options in California . Conclusions and discussion of results close the paper.

Air Pollution Issues in Dairy Operations

Methane

Methane is produced from the decomposition of livestock manure under anaerobic conditions. According to EPA, in 2003 roughly 545 CO_2 equivalent tons of methane were emitted from human related activities in the U.S. Approximately 28% of these emissions were from animal husbandry, either from enteric fermentation during digestion by ruminant animals or from manure management.[4] Enteric fermentation, which accounts for about ¾ of methane emissions from animal husbandry, occurs when microbes in the animals fore-stomach convert feed into digestible products and create methane as an exhaled byproduct.

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[3] Some federal programs can also provide cost-share funding for methane digesters include Environmental Quality Incentives Program (EQIP), Conservation Innovation Grants Program (CIG), and Conservation Security Program (CSP). (CEC 2006; NDESC 2005).

[4] For more information see www.epa.gov/methane/sources.html (accessed 2/14/06).

The rest of methane emissions from livestock operations come from manure management (US EPA 2005)[5], which accounts for roughly 7% of total anthropogenic methane emissions in the US. Methane from manure management is produced during the anaerobic decomposition of organic material in manure. Methane production is particularly abundant when lagoons and holding tanks are used for liquid manure management. When manure is deposited on fields in a dry form, methane emissions are much less.

The main approach for controlling methane emissions from manure management is to capture the methane and burn the biogas as a way to generate electricity for on-farm use and, potentially for sale in the market. Combustion of methane for electricity generation results in emissions of carbon dioxide (CO_2), another important GHG, but burning one ton of methane (equivalent to 23 tons of CO_2 if allowed to vent) yields 2.75 tons of CO_2 and thus a large reduction in the net contributions of greenhouse gases from the farm. In addition, the electricity supplied from this activity substitutes for other forms of electricity generation including fossil fuels and thereby potentially leads to a net reduction in GHG.

Several methane digester systems are currently being implemented on dairy farms in California. Over 30 dairies have applied for California Energy Commission's cost-share grants for the installation of methane digesters, and at least a dozen installations are in already operation (Sustainable Conservation 2005, 2006). As of February 2006, assessment data are available for four methane digesters that were co-financed by the California Energy Commission: Blakes Landing, Castelanelli Bros., Cottongwood, and Meadowbrook Dairies. Table 1, which is compiled using project evaluation reports to California Energy Commission (CEC 2005a-d), summarizes information about these dairies and their methane digesters.

Generally, dairies can generate more electricity using a methane digester than they consume. Therefore, financial benefits to the dairy from a methane digester depend on the electricity output from the digester, on-farm usage of electricity, and the retail and regeneration credit prices of electricity. The effective financial benefit to the dairy operation from generating a kilowatt of electricity using the methane digester varies between the net-generation credit and the retail price of electricity (weighted by the relative volumes of on-farm electricity purchase offsets and net generation credits). For example, the Castellanelli dairy reports an average agricultural and residential energy usage of about 56,736 kWh/month, which would cost about \$6,240 at the retail rate of \$0.11/kWh. This is the amount of monthly cost savings at the dairy from the methane digester, given sufficient methane digester capacity to generate this much energy. In addition, the surplus energy output generates revenue if it can be sold to the grid for a positive price. The amount of compensation for net-generation is not yet well established. The two dairies, for which regeneration credit pricing has been described (Castelanelli and Meadowbrook dairies), suggest that a roughly \$0.06/kWh regeneration credit is realistic.

Ammonia

Animal husbandry operations are a source of approximately half of US ammonia emissions, contributing roughly 2.5 million tons of ammonia emissions per year.[6] Dairy farms are responsible for a little over 20% of the emissions from animal husbandry.[7] The amount of ammonia emissions from livestock farms depend on how animal waste is managed and will vary substantially depending on concentrations of ammonia, temperature, pH and how long the waste is stored before applied to land as fertilizer. Ammonia concentrations and therefore emissions tend to be higher with higher temperatures and higher pH and lower the longer waste is stored before land application.

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[5] Enteric fermentation and manure management contribute methane approximately equal to 115 and 39 $TgCO_2$ equivalent emissions, respectively. All GHG emission resulting from human activities total 6,072 $TgCO_2$ equivalents (U.S. EPA 2005).
For more information see www.epa.gov/methane/sources.html (accessed 2/14/06).
[6] Total ammonia emissions in the US are about 4.8 million tons per year. "PM Overview and Sources," WESTAR PM EI Workshop, Denver CO, March 2004, OAQPS, US EPA.
[7] US EPA 2004. National Emission Inventory - Ammonia Emissions from Animal Husbandry, (January 30), http://www.epa.gov/ttn/chief/net/2002inventory.html.

Table 1: Examples of Dairy Methane Digester Systems in California

	Blakes Landing	*Castelanelli Bros.*	*Cottonwood*	*Meadowbrook*
Cows (lactating)	247	1,600	5,351	2,133
Gas production, cf/day				
• Total	20,000	70,751	241,990	67,912
• Per cow	84	44	45	31.84
Electric Output				
• Generator, kW	75	160	300 (700 planned)	160
• Total, kWh/year	229,220	1,132,595 (~50% gas flared)	2,334,095 (~55% gas flared)	931,144
• KWh/cow/day	2.54	1.94	1.14 (for 300kW)	1.20 (design 1.68)
• Retail rate, $/kWh	$0.10	$0.11 (regeneration credit $0.058	$0.115	$0.069 regeneration credit (not final)
Capital Costs	$336,362	~$800K (design $773K)	~$2.7M (design $1.29M)	~$800K (design $524K)
O & M, per month	~$100-800	~$600	~$5,000	~$560
Manure collection	Covered lagoon	Covered lagoon	Covered lagoon	Plug flow digester
Agricultural and residential energy usage (kWh/month)	9,941	56,736 (summer 107,353)	N/A	42,778

Source: Compiled using California Energy Commission's 90-day evaluation reports (CEC 2005a-d).

Numerous methods have been discussed for reducing ammonia emissions using different strategies for different sources of emissions including livestock housing facilities, manure storage facilities and land application of manure. [8] One of the more effective approaches for use with housing facilities is the use of filters or biofilters to remove emissions from ventilation exhaust systems. These systems, which have been shown to remove approximately 74% of total emissions at a relatively low cost per animal, are also the main focus in our analysis. The effectiveness of other approaches such as impermeable barriers to prevent air movement out of livestock housing facilities and dietary manipulation is currently being studied. Other approaches that focus on the manure storage are currently being tested including urine/feces separation, acidification and the use of additives to prevent ammonia production and volatilization. Among these approaches urine/feces separation appears to promise the largest percentage reductions in ammonia emissions. As much as 35% of total ammonia emissions have been estimated to take place during or after land application of manure. One way to reduce these emissions is to inject the manure into the ground, or through the use of urease inhibitors following land application.

[8] The approaches to reducing ammonia emissions discussed in this paragraph are all described in greater detail in Iowa State University Extension (2004).

A Process Farm-Level Model of Animal Waste Management

Model Structure

In this paper, we develop a conceptual integrated model for methane and ammonia emissions from concentrated animal farm operations[9]. The integrated model framework includes methane and ammonia emissions from a baseline with no emissions controls and from a variety of emissions management strategies, including electricity and heat recovery as well as various ammonia emission control strategies. Such a model is transparent and useful for conducting comparative analysis. The model also considers the costs associated with these strategies and their benefits, such as GHG credit revenue and air quality (ozone and $PM_{2.5}$) impacts. Table 2 provides a complete list of the components of the conceptual model and identifies which are currently available.[10] The model is developed using software called Analytica, which provides a graphical representation of relationships in the model (Figure 1) and easily incorporates quantitative measures of uncertainty. This latter capability is particularly important because there is considerable uncertainty and variability in emission factors estimation, technology performance and control costs.

Baseline Emissions

The model includes estimates of baseline emissions of methane and ammonia in the absence of specific controls.[11] These estimates will vary depending on characteristics of the farming operation and where it is located.

Methane emissions include both emissions from enteric fermentation and from the decomposition of animal waste under anaerobic conditions. Animal and feed characteristics have a significant impact on methane emissions. This paper focuses on methane emissions from dairy operations, however the model includes enteric fermentation for six types of animals (non-dairy cattle, dairy cattle, swine, sheep, goats and horses). Methane emission factors also vary by region as a result of temperature and altitude differences and the methane emission factors for enteric fermentation by region are obtained from AP-42 (USEPA, 1998).

The amount of methane produced during waste decomposition is affected by the climate (temperature and rainfall) and the condition (oxygen level, water content, pH and nutrient availability) in which the manure is managed. Manure decomposes more rapidly when the climate encourages bacterial growth. For liquid manure systems, methane production increases with temperature. In our current model, methane emission factors by climate region are obtained from *Revised 1996 IPCC guidelines for National Greenhouse Gas Inventories* (IPCC, 1996).

The emission factors for ammonia used in the model come from EPA (2004), a study that developed ammonia emission factors by animal type for 18 different manure management trains (MMTs). Zhang et al (2005) are developing a processed based ammonia emission model.

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[9] NRC has suggested using a process-based model farm approach that incorporates "mass balance" constraints for some of the emitted substances of concern, in conjunction with estimated emission facotrs for other substancces, may be a useful alternative to the model farm construct deinfed by EPA (NRC, 2003). However, in this paper we use emission factor approach to demonstrate our concept. After careful calibration, this simple conceptual model could be useful for policy analysis and for identifying data gap and research needs. Outputs from more sophisticated process based approaches could be incorporated/adopted in the integrated conceptual model.

[10] At this moment, some components are created as place holders. We plan to refine the model components and fill the data gap as our research advances. One of the advantages of such an integrated model is that we are able to identify the information needs.

[11]Other types of livestock will be added.

Table 2. Conceptual Model Components

- Baseline Enteric Methane Fermentation*
- Baseline Methane Emission*
- Baseline Ammonia Emission*
- Ammonia Emission Control*
    - Dietary Manipulation
    - Filtration and Biofiltration*
    - Impermeable Covers
    - Permeable Covers
    - Urine Feces Segregation
    - Acidification
    - Additives
    - Control Land Application
    - Manure Amendments
- Methane Production and Energy Recovery*
    - Covered Lagoon
    - Plug Flow Digester*
    - Complete Mix Digester
    - Gasifier
    - Gas Turbine Electricity Generation*
- Ammonia Control Cost*
- Methane/Electricity Production Cost*
- Heat Recovery Cost Saving
- GHG credit revenue*
- Air Quality Externality*
    - $PM_{2.5}$ wrt ammonia emission control*
    - $PM_{2.5}$ wrt NOx emission from energy recovery facility*
    - Ozone wrt NOx emission from energy recovery facility*

* indicates that the component currently has data available in the model

Ammonia Control Options

Ammonia emissions to the atmosphere are an environmental concern because they can contribute to odor, to eutrophication of surface water and to nitrate contamination of ground water. Ammonia emissions also contribute to the formation of fine particulates, which have a negative impact on animal and human health. Strategies to reduce ammonia emissions include both preventing ammonia formation and volatilization and downwind transmission of ammonia after it is volatilized. Iowa State University (2004) provides information on relative costs and effectiveness of nine different ammonia control practices, which are listed in Table 2. For example, the ammonia emissions can be reduced by 40 to 50 percent using biofiltration at the animal housing area. According to Iowa State University (2004), the biofiltration costs for a 700-head farrow-to-wean swine facility are estimated at $0.25 per piglet, amortized over a 3-year life of the biofilter. In the model, this cost ($0.25 per animal) is assumed to apply to biofiltration applications at dairy operations as well.

Options for Methane Capture and Electricity Production

A biogas recovery system is one of three manure management techniques that can be used to capture methane. (The other two are gasification systems and composting.) Biogas recovery systems, sometimes known as anaerobic digesters, can provide renewable energy and alleviate some of the environmental problems associated with manure from large animal operations. During anaerobic digestion, bacteria break down manure in an oxygen-free environment. One of the natural products of anaerobic digestion is biogas, which typically contains between 60 to 70 percent methane, 30 to 40 percent carbon dioxide, and trace amounts of other gases, with combined heating value of 600 BTU per cubic ft (natural gas is about 1100 BTU per cubic ft). The biogas recovery systems offer a number of environmental benefits, including odor control, GHG reduction, ammonia control and water quality protection.

There are three different types of biogas recovery systems have been commercialized for managing manure. These systems range from the simple covered lagoon to the more complex plug flow and complete mix digesters. Which system is most appropriate depends on how the manure is collected and on the total solids content of the collected manure. For example, the suitable total solids content for these three systems are 0.5 to 3 percent, 3 to 10 percent and 11 to 13 percent, respectively (USEPA, 2002).

At this time, our model only considers using the plug flow digester biogas recovery system. Other recovery systems (covered lagoon and complete mix digesters) and other energy technology (gasifier) will be added in the future.

The amount of electricity generated from the plug flow digester biogas recovery system depends on daily manure production, number of animals, solid content of the manure, a fixed biogas production coefficient, the methane content of the biogas and the efficiency of the electricity generator. We conducted model verification by comparing the electricity generation using our model with numbers reported in the literature. Our estimate, 104 kW for a farm with 1000 cows is in the range of reported values.

We develop a capital cost function, using data collected from four dairy farms and reported in Table 3. We first convert the cost to 2004 dollars. We estimate the cost function using the following functional form:

$$y = ax^b$$

where, the left hand side dependent variable, y, is the average cost per cow and the right hand side variable, x, is the number of cows. In the equation, a and b are cost function parameters. Actually, b is the estimate of the scale elasticity. In our case, coefficient estimate b equals–0.76, which means that every one percent increase in farm size (in terms of number of cows), the average capital cost decreases by 0.76 percent. Cost function parameters a and b are assumed to be normally distributed, using their estimates and standard errors. We amortized the capital cost by assuming 7% compound interest rate and a twenty-year lifetime. Annual operation and maintenance cost is assumed to be 20% of annual capital cost by default and can be changed easily in the model.

The GHG credit is calculated based on the difference between baseline methane emissions (in CO_2 equivalent) and CO_2 emissions from biogas combustion (including both biogas CO_2 and CO_2 from biogas methane combustion). As noted above, we assume methane has global warming potential 23 times that of carbon dioxide. We also assume that combustion of one ton of methane yields 2.75 tons of CO_2. GHG credit revenue is equal to the product of the number of credits and the credit price.

Table 3. Capital Cost of Plug Flow Biogas Systems with Electricity Generation on Select Farms

Farm	Installation Year	Animal Production	Installed Cost	\$2004/head
Haubenschild[a]	2002	1,000		\$373
Craven[b]	1997	650	\$253,000	\$458
AA Dairy[b]	1998	550	\$240,300	\$506
Haubenschild[b]	1999	480	\$295,800	\$699

Source: (a) Nelson and Lamb (2002); (b) Moser and Matocks (2006).

Health Effects of Air Emissions

The air quality impacts of farm operations considered in the model include reduced emissions associated with ammonia emission controls and additional NO_x emission from the biogas combustion for electricity generation. Ammonia is a precursor of fine particulate matter. Once it is emitted, it could react with nitric acid to become ammonium nitrate, a secondary pollutant, in the air. NO_x is a precursor of both ozone and particulate matter. To evaluate the health impact of particulates and ozone due to the ammonia control and new NO_x emissions, we will have to analyze both emissions transport and air chemistry and changes in exposures and impacts on human health. The first task requires the development of pollutant source-receptor relationship, which is how much secondary pollutant concentration will change at the receptor site due to emission change of primary pollutant at source site. The second task requires estimates of changes in exposure and related health impacts due to the change of the secondary pollutant exposure.

In the current model, for task 1, we need source-receptor relationships for ozone with respect to NO_x emissions, $PM_{2.5}$ with respect to NO_x emissions, and $PM_{2.5}$ with respect to reductions in ammonia emissions. The authors of this paper have done research to quantify the source-receptor coefficients at the state level for the first two (Palmer et al, 2005; Shih et al, 2004). The authors could not find any farm level empirical source-receptor coefficient. So for ozone with respect to NO_x emissions, we average the eight-hour ozone source-receptor coefficients in the source-receptor coefficient matrix (for the entire study domain) as our default in the current model. We do the same thing for $PM_{2.5}$, using twenty-four hour source-receptor coefficient matrix.

We were unable to locate any source receptor coefficients for $PM_{2.5}$ with respect to ammonia control. The literature offers a range of perspectives on this issue, with some papers arguing that ammonia control has no effect on $PM_{2.5}$ concentration (LADCO 2006) for a specific region, while other research suggests that ammonia control has positive effects (Erisman and Schaap, 2004). The differences in these findings depend on whether the region being studied is ammonia limited or not. These differences in the literature suggest that there is a huge uncertainty about as well as variability in this coefficient among different regions/locations.

In this model, we develop a simple box model to estimate the source receptor coefficient for $PM_{2.5}$ with respect to ammonia control. We assume that emitted ammonia reacts with nitric acid completely to become ammonium nitrate and this ammonium nitrate is uniformly mixed within the box (after considering deposition since emissions from farm operation tend to be near the ground surface).[12] We then calculate the average change of ammonium nitrate concentration within this box due to one unit of ammonia emission reduction. Given limited time and resources, we use the simple box model approach to produce the upper bound estimate of the $PM_{2.5}$ with respect to ammonia source-receptor coefficient. We then use a uniform distribution between 0 and this upper bound to characterize this coefficient in our model.

To estimate the health benefits, we develop simple composite health benefit coefficients for ozone and $PM_{2.5}$ exposure using TAF (ORNL, 1995). The health benefit coefficient is defined as benefit in dollars per pollutant concentration change per year. The health effects considered include the number of days of acute morbidity effects of various types, the number of chronic disease cases, and the number of statistical lives lost. The pollutant concentration-response functions are found in the peer-reviewed literature, including epidemiological articles reviewed in EPA's Criteria Documents that, in turn, appear in key EPA cost-benefit analyses (Palmer et al, 2005). We first estimate pollutant concentration change at a receptor by multiplying emission reduction from the source by the relevant source-receptor coefficient. We then multiply the concentration change with the health benefit coefficient to get the health benefit estimate.

[12] In personal correspondence, Professor Ted Russell of Georgia Tech has pointed out that this assumption is not strictly correct because the reaction is an equilibrium and also there is a limited amount of nitric acid in the atmosphere and ammonia would not be able to convert to ammonium nitrate in a fully efficient manner (100 percent). The effect of this assumption is to overestimate the source-receptor coefficient which would serve as an upper bound for the reduction in $PM_{2.5}$ that would result from a reduction in ammonia. We plan to refine this estimate through more comprehensive 3-D air quality simulation model in the future.

Policy Simulations and Results

The integrated assessment model is used to study the atmospheric emissions from animal husbandry and their environmental consequences, and to investigate potential policies to improve the environmental and economic performance of the industry. In the ongoing program of research we investigate two types of policies – performance-based policies that would require specific technologies or management practices, and market-based policies that could provide economic incentives to reduce emissions. Some policies would involve the agricultural extension service in its traditional role of outreach, education and technical assistance. Other policies could require mandated practices. However, the policies we describe for this presentation involve the creation of new markets that allow farm operators to internalize social benefits from more efficient management.

We illustrate the model by exploring three policies. One is the creation of GHG credits to account for the social benefit of reduction in methane emissions. The second is the creation of particulate matter ($PM_{2.5}$) offset credits to account for the social benefit of reduction of ammonia emissions. The third is expanded net metering of electricity to provide financial payments to farm operators for electricity provided back to the electricity grid. Underlying parameters in the model such as population of farm operations, temperature and background emission inventories exhibit large variability, and several parameters in our model are very uncertain or based on nonlinear processes. In the future, we plan to account for this variability and uncertainty using simulation-based methods such as Monte Carlo analysis. To illustrate the model in this exercise we rely primarily on mid-point values for many parameters, often erring intentionally on the side of cautious choices that may under-estimate the potential benefits of the policy options, partly to guard against bias due to omitted features of the problem at this juncture. We vary two fundamental parameters to give a flavor for the potential sensitivity of the results.

Greenhouse Gas Policies

There are two pathways that offer the potential to avoid GHG emissions. One is to change management practices including diet and capture of methane. The second is to use the methane byproduct for electricity generation. Change in management practices could be mandated by fiat but the regulatory burden of enforcement would be enormous and the economic impact on the farm sector would be severe. A market-based approach could lead to a more efficient technology choice at much less cost to government and with positive economic benefits for the industry.

We model a market-based policy that provides a payment for emission offsets under GHG cap and trade programs. One such cap and trade program is in place in the EU, another has been approved in seven states in the northeast US, and others are under consideration in California and elsewhere, as well as at the federal level.[13] In various ways these programs are expected to allow for the use of offset credits awarded for emission reductions achieved outside the emission sources that are directly regulated by the program. One tenet of this approach is that offsets qualify only for emission reductions that would not have happened anyway, for example, those that are additional to current laws, regulations or practice. A key feature of offset programs is the documentation of baseline emissions, and the certification of changes in practices that would lead to emission reductions. To this end the model calculates emissions under the baseline (in absence of a policy) as well as changes under various policies and management strategies.

In our central case we model a specific management practice using a plug flow digester for a farm operating in a warm climate such as California with a size of 500 head. We consider offset credits valued at \$11 per ton of CO_2 equivalent. This value is midpoint to values that might emerge given current policy.[14] Under the creation of an offset market for these emission reductions the economic value of avoiding additional reductions at facilities regulated under the emission cap flows through to the farm operator. Electricity generation with the captured methane leads to residual emissions of CO_2, which are accounted for in the net emission reductions.

[13] See: http://europa.eu.int/comm/environment/climat/emission.htm; http://www.rggi.org/; and http://www.climatechange.ca.gov/.

[14] Emission allowances under the EU ETS are currently trading at about \$30 per metric ton CO_2. The northeast Regional Greenhouse Gas Initiative Memorandum of Understanding includes a trigger price of \$10 per short ton in order to expand the offset market to include states outside the region.

The costs of the digester that we account for include installation and operating costs, and a generator that combusts methane to produce electricity, but it does not include opportunity costs such as the alternative use of land for the digester. The value of the electricity depends on its potential use on farm or resale onto the grid. Whether independent power producers can realize the value of sale back onto the grid depends on the whether distribution companies pay for the power. Net metering policies require payment to independent power producers at an avoided cost. We assume that net metering is not available to the farm operator in our central case, and vary this in sensitivity analysis.[15] In the absence of net metering policy the farm operator can capture only the value of electricity at the farm, equivalent to displaced purchase from the grid, but extra electricity generation capability is unutilized. We assume a weighted value of $0.06 per kWh for electricity generated.[16] In addition, we note that electricity generation results in an increase in emissions of NO_x, which is a precursor to PM and ozone. The social cost of the increase in NO_x is accounted for below.

Table 4. Costs and benefits to farm operator of methane and ammonia capture under market-based policy scenario.

Climate	Warm			Cold		
Farm Size (head)	400	500	1,000	400	500	1,000
Baseline CH4 (CO_2e tons)	769	961	1,923	364	455	911
Digester Cost	29,680	31,350	37,160	29,680	31,350	37,160
CO_2 in electricity generation (tons)	332	414	829	332	414	829
Ammonia Control Cost	120	150	300	120	150	300
Electricity revenue	21,910	27,380	54,770	21,910	27,380	54,770
GHG credit revenue	4,811	6,014	12,030	358	448	896
Health benefit-ozone	-263	-328	-656	-263	-328	-656
Health benefit-PM25	12,030	15,040	30,070	12,030	15,040	30,070
NET Benefits	8,689	16,606	58,754	4,236	11,040	47,620

Monetary estimates are dollars per year (2006 dollars). The example excludes transportation costs and heat recovery value, and the potential GHG credits from reduced generation of fossil fired facilities. Electricity revenue excludes the benefits of net metering.

Table 4 reports that methane capture for electricity generation at a farm in a warm climate with size of 500 head imposes costs of $31,350 as an annualized cost. The electricity savings on the farm operation total about $27,380, which is not sufficient to justify the investment. However, the additional revenue from GHG offset credits would yield $6,014, which is sufficient to tilt the balance producing net economic benefits of $2,014 per year.

One important aspect of the incentive structure of a GHG offset market that is made apparent in the integrated assessment model is the consequence of changing diet. We do not model offsets for diet

[15] There was a CA law passed in 2002 to encouarge net metering for farms that use digesters (see http://sfgate.com/cgi-bin/article.cgi?f=/c/a/2004/05/14/BAGJG6LG3R15.DTL). PG&E has offered a limited net metering policy for biogas facilities called NEMBIO that became available in August 2003. Initially, this opportunity is available to farms that generate less than 1 MW and limited to the first 5 MW that apply (on a first come first serve basis). In 2005, AB 729 extended these limits to authorize up to 3 digesters with up to 10 MW of capacity to be eligible for net metering and the cap on total MWs of biogas digesters eligible for net metering was extended to 50. (see http://www.dsireusa.org/library/includes/incentive2.cfm?Incentive_Code=CA02R&state=CA&CurrentPageID=1)

[16] Based on representative statistics we calculate that about 54% of the electricity generating potential would be used on-farm, displacing retail electricity purchases that average $0.11/kWh for agricultural customers in California. The remaining generation potential would be unutilized. Hence, the weighted value of the electricity, in the absence of net metering, is $0.06/kWh.

management, although such a credit could be attractive. However, we do note that changes in diet would affect ultimate methane production. If the farm operator receives payment for offsets from methane capture from manure, the operator would lack the incentive to change diet to reduce enteric methane because this would also reduce methane that is available for capture in manure. Indeed, an unintended consequence of the GHG offset market associated with capture for electricity generation might be an increase in enteric methane along with methane in manure. Policy may need to link these management practices, perhaps making aspects of diet management a pre-requisite for GHG credits for capture of methane from manure.

Electricity generation creates another potential source of value external to the electricity market that is not included in this example. In the face of a cap and trade program for CO_2 the electricity generation may qualify for additional offset credits associated with the avoided emissions from fossil-fired power plants. The avoided emissions are not equivalent to the average emissions of electricity on the grid. Instead the proper measure is the change in generation at other facilities due to the methane-powered electricity. To identify this measure with confidence requires solution of an electricity market model, which is a component of our ongoing research project. For a proxy, it might be reasonable to assume that the displaced emissions comes from a gas-fired facility since natural gas is typically the marginal generation technology, especially in California. A short cut for regulators might be to associate the avoided emissions with the avoided generation source that determines the payment under a net metering program. In any event, this potentially substantial source of GHG credit revenue is not included in the results presented above.

Ammonia/Fine Particulate Related Policies

A second external effect of management practices is due to emission of ammonia, which is a precursor to fine particulate matter. Management practices could reduce the emission of ammonia, but at a cost to the farm operator. One way to provide positive incentive for improved management would be to account for the reduction in $PM_{2.5}$ that is associated with reductions in ammonia. NO_x and sulfur dioxide (SO_2) are regulated directly through a variety of programs and they are important precursors to $PM_{2.5}$, but they require ammonia for the conversion to $PM_{2.5}$. In areas that are not in attainment with the National Ambient Air Quality Standards any new source must obtain offsets of emission reductions at another source. Those offsets have potentially significant economic value, depending on the air quality management district, ranging from hundreds of dollars to tens of thousands of dollars per ton, varying by year due to changes in local economic conditions and other factors.

We consider the creation of offset credits for ammonia in the nonattainment districts in California. Using the model we solve for the expected changes in health effects due to reductions in $PM_{2.5}$ and increases in ozone that may be likely to occur were ammonia reductions to be achieved. Emission reductions would be achieved through the use of biofilters, which impose a cost of $120 per year. Table 4 indicates the $PM_{2.5}$ benefits would be substantial and would dominate the change in ozone, and these values sum to $14,712 per year in our central case. The net benefit of this management strategy would be $14,592 per year.

Important Uncertainties

There are numerous uncertainties revealed already in our preliminary modeling. An important variable is the availability of net metering and the net generation credit price. In the main analysis, we assumed that net generation of electricity is not rewarded financially. If we assume instead that the farm operation can sell its surplus electricity back onto the electricity grid at $0.06 per kWh, annual net benefits in our central case increase from $16,606 to $28,936.

The climate (temperature) in the location of the farm affects methane and ammonia emissions in the absence of control strategies. Table 4 indicates that differences between cold and warm climates cause the net benefits of the GHG offset management strategy including electricity production for a farm operation with 500 head to vary from $11,040 to $16,606.

One of the most important policy considerations is the size of the farm. We characterize a range of size from 400 to 1,000 head. This range provides opportunities for net benefits to vary by nearly an order of magnitude. For a 1,000 head farm operation in a warm climate, we find annual benefits can total $58,754.

From a scientific standpoint, one item with great uncertainty in this analysis is the characterization of atmospheric dispersion of ammonia and its contribution to ultimate particulate formation. The relevant values will vary significantly with geography and region of the country, with assumptions about background pollution, etc. Nonetheless, the proper accounting for ammonia reductions as offset credits for associated PM reductions could offer significant economic benefits to the farm operation and significant social benefits as well.

Conclusion

Animal husbandry is a major emitter of methane, an important greenhouse gas, and ammonia, a precursor to fine particulate matter, arguably the number one environmentally related public health threat facing the nation. Technologies are available to dramatically reduce these emissions, but their adoption by dairy operations has been limited. In this paper we explore market-based policies to provide farm operators with financial incentives to reduce emissions by adopting methane and ammonia control technology. We develop and exercise an integrated process model of dairy operations. Three policy options are explored including greenhouse gas offset credits for methane, particulate matter offset credits for ammonia, and expanded net metering policies to provide revenue for sale of electricity generated with methane. We find that taken individually, any of policies appear sufficient to provide the economic incentive for farm operators to reduce emissions. The magnitude of the benefit depends of the scale of the system, location in specific climate region and technology adopted and also on important assumptions in the model regarding ammonia to PM source receptor coefficients. This paper reports on initial steps to fully develop the integrated process model to provide guidance for policymakers.

In future work we plan to explore additional features of the policies discussed here. We plan to link the model with a dispatch model of the California electricity sector to estimate the CO_2 emissions displaced by expanded generation from methane digesters. We also plan to explore the effect that scaling up of these operations and the use of multi-farm digesters and associated transportation costs. We also could develop an optimization model for siting such an energy facility, taking into account its environmental cost and benefit and integration with the existing power grid. Farm level source-receptor coefficients for specific locations could affect our estimation results and this deserves further investigation. Finally, we could extend the integrated model by considering a water quality impact component. This research is expected to provide further insights about how to reduce the financial burden for the agriculture industry to improve productivity as well as environmental quality.

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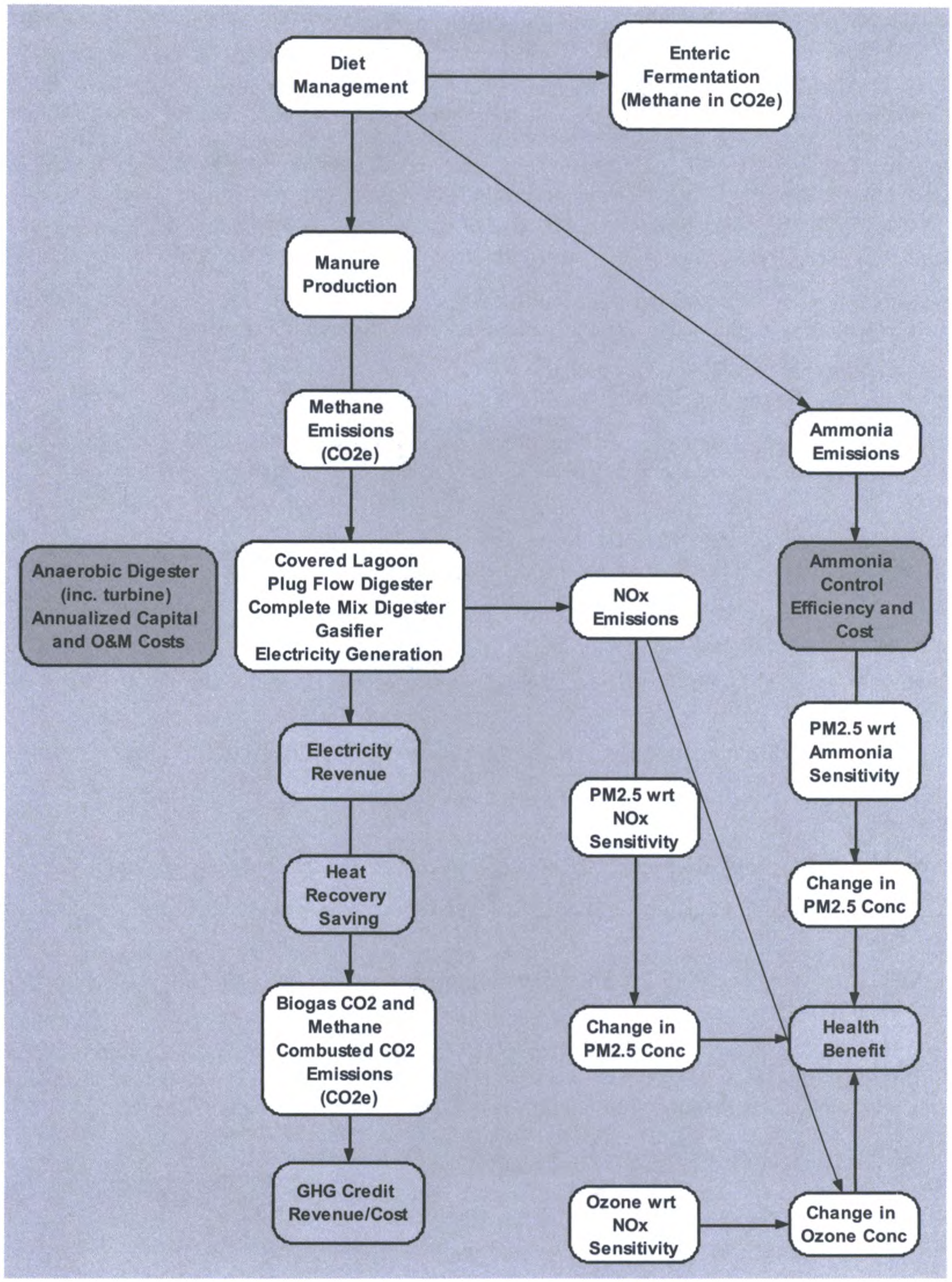

Figure 1. The Influence Diagram of the Integrated Animal Waste Management Model

Shelterbelts and Livestock Odor Mitigation: a Socio-economic Assessment of Pork Producers and Consumers

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Abstract

Pork production in the United States is expanding, especially in the Midwest. With expected economic benefits from expansion come potential environmental and social costs from odor. Scientific evidence suggests that shelterbelts – living tree barriers – can be cost-effective, biologically active buffers that reduce odor, and complement other odor control strategies used by producers. Previous research has suggested that some consumers accept partial responsibility in environmental degradation and exhibit an inertest in purchasing "environmentally friendly" products including pork meat. Market mechanisms that allow price premiums for environmentally friendly pork meat to flow back to producers can reduce financial constraints faced by producers and directly link producers to consumers through joint socially-valuable efforts. Surveying both pork consumers and producers in three different states - Iowa, North Carolina and Washington State - we have examined attitudes regarding market-based incentives for odor control and identified producer/consumer values regarding odor management in general and odor management involving the use of shelterbelts. Results from consumer willingness to pay (WTP) surveys indicate strong consumer interest and WTP for "environmentally friendly" pork products. Across all states, 82% of the respondents indicated a positive WTP for pork products that originated on farms that made odor management a priority. The maximum mean WTP was $0.14/ pound of pork meat purchased. Consumer acceptance of the use of shelterbelts specifically for on-farm odor mitigation was significantly higher than other listed odor control technologies. The mean cost that pork producers across all three states were willing to pay to plant and maintain shelterbelts for odor management was $0.14/ head produced. Additionally the producers all expressed interest in raising pigs with extra odor management if the prices received covered additional costs. With regards to variables that strengthen the behavioral intention represented by the consumer WTP figures, consumers surveyed expressed low to moderate environmental values when it came to making food purchasing decisions in general and moderate label reading behavior but expressed strong attitudes about odor management at the swine farms that produce their pork. Consumers expressed high concern about air quality around hog facilities in general (Iowa consumers being the most concerned). Non-meat attributes such as production methods also figures high in consumer importance. The results of this research will support cooperative approaches to solving odor problems that include natural odor control strategies, and help to sustain two vitally important parts of agriculture – pork production and rural communities.

Introduction

Pork production in the United States is expanding, especially in the Midwest. With expected economic benefits from expansion come potential environmental and social costs from odor. Scientific evidence suggests that shelterbelts – living tree barriers – can be cost-effective, biologically active buffers that reduce odor, and complement other odor control strategies used by producers. Previous research has suggested that some consumers accept partial responsibility in environmental degradation and exhibit an inertest in purchasing "environmentally friendly" products including pork meat. Market mechanisms that allow price premiums for environmentally friendly pork meat to flow back to producers can reduce financial constraints faced by producers and directly link producers to consumers through joint socially-valuable efforts. Surveying both pork consumers and producers in three different states - Iowa, North Carolina and Washington State - we have examined attitudes regarding market-based incentives for odor control and identified producer/consumer values regarding odor management in general and odor management involving the use of shelterbelts.

Consumer Surveys

Analysis of the statistically representative consumer surveys indicates strong consumer interest and willingness to pay (WTP) for "environmentally friendly" pork products across all three states examined – Iowa, North Carolina and Washington State (total completed interviews 349: Iowa=145; North Carolina=77: Washington=127). Across all three states, 82% of the respondents indicated a positive WTP for pork products that originated on farms that made extra odor management a priority. The maximum mean willingness to pay was $0.14/ pound of pork meat purchased. North Carolina consumers expressed the highest mean WTP at $0.16/ pound of pork meat. The mean WTP between the states does not vary significantly. However, there are statistically significant (p = .05) differences between the states with regards to overall environmental and social concerns within agriculture. On an aggregate socio-environmental concern index (which factored in concern for air and water quality, family farms, and antibiotics in food) consumers from Iowa and North Carolina had significantly higher scores than consumers from Washington. Multivariate regression analysis failed to find acceptable models of WTP using demographic information and variables that show strong attitudes towards environmental quality. However, logistic regression examination of likelihood to express a positive WTP for environmentally showed that up to 33% of the variation (based on pseudo R squared statistics) between those WTP nothing and those WTP some positive value can be explained by Key variables such as gender, overall environmental values, and attitude about on-farm reduction of hog odor. Females are 3.2 times more likely to have a positive WTP than males and those consumers with strong attitudes about hog farms making efforts to reduce odor from their farms are 7.4 times more likely to express a positive WTP.

Consumers expressed high concern about air quality around hog facilities in general with Iowa consumers the most concerned. When purchasing meat products, meat quality factors are the most important attributes (freshness and flavor) with price coming in second, yet non-meat attributes such as farming/production methods also are importance. The respondents also showed high familiarity with other kinds of differentiated pork products (i.e. organic or natural pork) but indicated low levels of purchasing such products. Consumer acceptance of the use of shelterbelts specifically for on-farm odor mitigation was higher (statistically significant) than other listed odor control technologies. The order of acceptance is shelterbelts, organic manure additives, mechanical air filtration, organic feed additives, and chemical feed/manure additives – the latter two scoring very low in acceptance. Overall and across the three states the socio-demographic variables of income and education show significant but low positive correlations with WTP and number of people in household under the age of 18 show significant but low negative correlations. There seems to be no differences between urban and rural respondents. Also, proximity to pork production and experience with swine odor shows limited influence on WTP. Continued analysis will examine within state characteristics.

Producer Surveys

The analyses of the producer survey of marketing and management (total completed interviews 587: Iowa=410; North Carolina=141: Washington=36) is ongoing. Across all three states, the vast majority of those interviewed were the owners and/ or key managers of the production. Almost all of the production operations were under individual, corporate, or family ownership. Just under 30% of those interviewed has less than 5,000 head at their primary facility, 67% were over 5,000 head and about 5% had over 50,000 head. Forty one percent of the producers raised other livestock and 76 % also raised crops. Eighty three percent operated feeder operations and 93 % utilized confinement building systems. The vast majority (92%) of producers had their primary facility within one mile from their nearest neighbor; almost 60% were within one half mile. With specific regards to use of and opinions of shelterbelts almost 60% of the producers have trees/shrubs planted in and around their facilities; thirty-two percent were planted specifically as an odor mitigation technology, the remainder as general landscaping. Out of those who do not use shelterbelts 64% said they were interested in planting trees for odor mitigation. In response to an open ended question asking about possible advantages to using shelterbelts producers mentioned odor reduction, "out of sight, out of mind" benefits, and improvement of facility aesthetics most often. Producers also weighed in on possible reasons why some hog producers do/would not use shelterbelts and listed excessive cost, labor requirements, and interference with building and site ventilation as the top three reasons respectively. Still, 73% of the producers surveyed somewhat to strongly agree that shelterbelts bio-physically remove odor from the air and 67% somewhat to strongly disagree that shelterbelts are not worth the expense.

Across all three states, the mean willingness to pay for planting and maintaining shelterbelts (for effective hog odor control) is \$0.14/hog produced. There are statistically significant (p = .05) differences between the states on mean WTP. North Carolina producers are WTP \$0.07/ hog produced, Iowa producers \$0.14/hog produced and Washington State \$0.24/hog produced.

Less than half (41%) of the producers interviewed are involved in market the hogs they raise; of that forty percent, 12% are involved in a marketing coop and 9% direct market pork to consumers. Twenty percent are involved in a marketing contract with a specific packer. Fifteen percent of the producers are involved in growing differentiated (specialty) pork with differentiated meat quality (15%), animal welfare (12%), and environmental quality (10%) being the three main attributes. Price premiums and steadier demand were listed as relevant outcomes to selling such products. Overall, across all three states 51% of the producers are interested to very interested in producing differentiated pork. Seventy percent stated that they are interested to very interested (48%) in producing pork specifically with "extra odor control" as long as the prices received covered additional odor management costs. Contracting this differentiated product with either a packer or the owner of the hogs (for those who are contract feeders), or direct marketing to consumers or through a marketing coop are the four top preferred ways to arrange for the creation of such products respectively.

Aesthetics Focus Group Information

Focus groups with pork consumers and pork producers were performed in Iowa and North Carolina during the summer of 2004. Part of the focus groups for both consumers and producers involved a short presentation about the bio-physical aspects of odor mitigation using shelterbelt systems. There was also a photo elicitation session were participants rated the visual quality of a static scene showing a facility with varying degrees of shelterbelts present (i.e. no trees to fairly extensive shelterbelt systems).

With regards to the producers, a total of 15 hog producers took part in the discussions. All of the participants in North Carolina (n=6) were large scale producers and for Iowa (n = 9) there was a mix of large and small producers. Producers from both states had very similar opinions about odor issues in their respective states. For example all the producers felt that odor is not the problem that the media is making it out to be. Producers also shared similar opinions about shelterbelts in general with most of them neither agreeing nor disagreeing that it is more pleasant to work at a facility that has trees. They did tend to disagree that shelterbelts might harbor rodents and other pests. North Carolina producers tended to be slightly pessimistic about the financial expense and labor requirements of shelterbelts. Collectively they neither agreed nor disagreed that shelterbelts actually help filter air of odor. They also agreed that shelterbelts useful in terms of providing general shelter from summer and winter winds, controlling wind erosion and in providing wildlife habitat. In terms of their opinions about the aesthetic appeal of shelterbelts planted in and around hog facilities, producers tended to view a "basic buffer" (one that has some clear shelterbelts but not a whole complex of them) as being preferred. This was explained by concerns for cost and maintenance of the more complex systems actually impacting the visual component of the trees.

For the consumers, there were a total of 27 people taking part in the discussions (Iowa n = 13; North Carolina n = 14). Overall, there were some interesting differences between the consumers in Iowa and those in North Carolina. There were high preferences for more trees in Iowa landscape in general and specifically that shelterbelts improve the aesthetics of confinement livestock production *landscape* – that is the production site as well as the overall, broader landscape that the facility is a part of. The Iowa consumers also expressed a high appreciation for "visual" response to odor issues. For the consumers in North Carolina, shelterbelts per se were not as important in the landscape largely because NC is heavily forested. They did however have strong appreciation of the idea that shelterbelts were actually a technology and they liked the innovativeness of their use.

General Conclusion

Results indicate that pork consumers are likely to pay more for meat originating from farms with higher air quality management. Moreover, consumers indicate a preference for the "natural look and feel" of shelterbelts (of trees) relative to other bio-chemical-mechanical odor control technologies. Shelterbelts are generally accepted as add-on aesthetic or public-relations technology by producers rather than highly effective in controlling odor. Producers and consumers agree that shelterbelts can and should play a role in mitigating swine odor. Some pork producers are willing to explore new ways to capture the extra money that consumers are seemingly willing to spend for "fresh air pork" through innovative marketing strategies while others value the addition of shelterbelts to farms. Shelterbelts should provide a suite of benefits for the pork industry, producer, consumer, and communities. Ultimately, the results of this research will support cooperative approaches to solving odor problems that include natural odor control strategies, and help to sustain two vitally important parts of agriculture – pork production and rural communities.

The Role of Agricultural Emissions in European Air Quality Policy

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Abstract

For Europe several hundred thousands of premature deaths, increased hospital admissions, and millions of lost working days have been associated with increased levels of air pollution. The costs of these health impacts are large. In addition, substantial damage is estimated from acidification and eutrophication of ecosystems and agricultural crops.

Ammonia emissions from agricultural activities have been recognized as a significant contributor to the acidification and eutrophication of ecosystems. They also play a critical role in the formation of secondary particulate matter. Model estimates suggest that in the year 2020, after substantial reductions of emission from the other sources, approximately half of the European damage due to acidification, eutrophication and particles will be associated with ammonia emissions.

The Regional Air Pollution Information and Simulation (RAINS) model (e.g., Schöpp et al., 1999) has been used to explore cost-effective emission control scenarios to inform the policy discussion of the European Commission on the ambition level of the EU Thematic Strategy on Air Pollution (CEC, 2005). The analysis that served development of this strategy (e.g., Amann et al., 2005) outlined the likely development of emissions of SO_2, NO_x, primary PM, NH_3, and NMVOC and their impact on air quality in Europe, and explored the scope for cost-effective measures that achieve further environmental improvements. On this basis, the European Commission has proposed to aim for 2020 at a reduction in the loss of life expectancy from the exposure to PM by half, and to significantly cut the area where deposition exceeds the critical loads for acidification and eutrophication. To achieve these environmental targets, the strategy indicates a need for reducing ammonia emissions by about 27 percent beyond the current commitments (Figure 1; CASE "B" is close to the final proposal made by the Commission), which comes at an estimated costs of about 2.6 billion €/year. These costs represent about 36% of the total thematic strategy costs for all economic sectors.

The results of the analysis and the proposed strategy indicate that agriculture needs to face a challenge of reducing significantly emissions of ammonia; without this reduction the ambitious environmental targets of the European Union cannot be met. In the final discussion stages of the thematic strategy three principal scenarios were evaluated that reflected three different ambition levels with respect to environmental targets (e.g., Amann et al., 2005). The results for these scenarios in terms of necessary emission reduction are presented in Figure 1. The gray range indicates the scope of further reduction beyond the current legislation baseline case (top end of the gray range) up to the maximum technically feasible reductions (bottom end). The current baseline shows a stark difference in level of emission reductions foreseen for various pollutants with ammonia emissions lower by only few percent compared to the 2000 levels. And in contrast to other pollutants most of this reduction is due to decline in livestock numbers rather than application of specific ammonia reduction measures; exceptions are Netherlands and Denmark. The overall technical potential for emission reduction in agriculture is relatively small (typically does not exceed 40 percent) compared to the other air pollutants where especially application of end-of-pipe measures brings high reductions. To reach the targets set by the thematic strategy, however, large part of this potential would need to be explored also in agriculture since emission of ammonia were estimated to be reduced by about 30 percent compared to 2000. This, in turn, is associated with high costs that represent a significant part of the total strategy cost as indicated earlier.

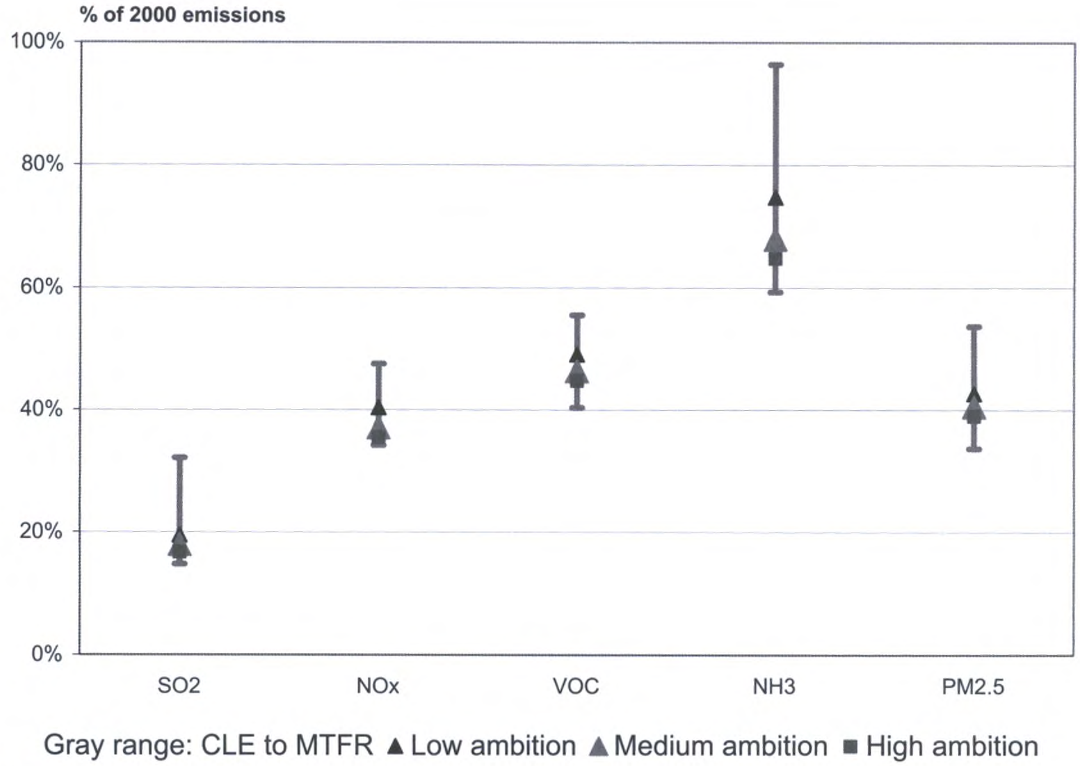

Figure 1. Emission reductions in EU-25 in relation to the levels in the year 2000.

The proposed thematic strategy, mid-term review of the Common Agriculture Policy, reform of the European sugar sector puts European agricultural policy in a very prominent place and lead to requests for further analysis of impacts of recent developments on emissions of various pollutants. In fact, not only ammonia since there are strong interactions between policies targeting ammonia and greenhouse gas emissions from agriculture. The RAINS model framework has been recently extended to include Kyoto gases and allow for such analysis (Klaassen et al., 2004).

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Particulate Matter

The Impact of Winter NH_3 Emission Reductions on Inorganic Particulate Matter under Present and Future Regulated Conditions

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Extended Abstract

Atmospheric ammonia contributes to the formation of aerosol nitrate, an important constituent of inorganic fine particulate matter, and atmospheric ammonia is a significant source of acidification and excess nutrient loading to sensitive ecosystems. Recent regulation by the US Environmental Protection Agency requires large-scale emission reductions of NO_x and SO_2. These regulated emission changes will decrease the concentration of nitrate and sulfate and hence alter the composition of inorganic particulate mater.

The complexity of the inorganic aerosol system makes prediction of the importance of ammonia after these emission reductions non-trivial (Ansari and Pandis, 1998). Sulfate has a low vapor pressure and condenses to the aerosol phase. Any available ammonia gas will neutralize the sulfate to form ammonium sulfate aerosol. If additional ammonia is available in excess of what is needed to neutralize the sulfate, it can form aerosol ammonium nitrate in accordance with thermodynamic equilibrium.

The inorganic aerosol system has three states, listed in order of increasing ammonia concentration: (1) acidic, where there is insufficient ammonia to neutralize the sulfate, (2) ammonia limited, where all of the sulfate is neutralized, but the formation of ammonium nitrate is limited by scarce ammonia, and (3) nitrate limited, where ammonia is present in excess such that the formation of ammonium nitrate is limited by scarce nitric acid. Reductions in nitrate and sulfate will cause the system to become more nitrate limited, decreasing the sensitivity of inorganic $PM_{2.5}$ to NH_3. However, if the aerosol is initially acidic, then reductions in sulfate can cause some ammonia to become available for ammonium nitrate formation, which could increase the sensitivity of $PM_{2.5}$ to NH_3 emissions.

The change in ammonia deposition is tightly linked to the future state of the inorganic aerosol system. Aerosol ammonium has a longer lifetime (7-10 days) compared to gas-phase ammonium, which rapidly dry deposits (lifetime of one day) (Seinfeld and Pandis, 1998). Decreases in aerosol nitrate and sulfate may cause increases in gas phase ammonia and more ammonia will be deposited closer to emission sites.

To better understand the future role of ammonia as an atmospheric pollutant, we use the Community Multiscale Air Quality (CMAQ) Modeling System version 4.4 to simulate current and future regional air quality and deposition (Byun and Ching, 1999; Byun and Schere, 2005). CMAQ is an Eulerian grid model that simulates advection, dispersion, gas-phase chemistry, aerosol thermodynamics, cloud processes, and wet and dry deposition. The model domain consists of the continental United States, discretized into a 36km x 36km horizontal grid with 14 vertical layers. The meteorological inputs for January 2001 are derived from the Fifth Generation Penn State/NCAR Mesoscale Model (MM5) (Grell et al., 1994). We use the same meteorology to drive all of the emission scenarios and CMAQ simulations.

Ammonia emissions are from Gilliland et al. (*in press*). Other emitted species are from the National Emission Inventory 1999 version 3.0 grown to 2001 levels. The 2010 and 2020 year scenarios include regulated reductions in SO_2 and NO_x emissions as described in the Clean Air Interstate Rule and Clean Air Visibility Rule (USEPA, 2005). The emission changes from 2001 to the future scenarios for SO_2, NO_x, and NH_3 are listed in Table 1.

Table 1. Domain-average emission changes for January for each scenario.

Species	2010	2020
NH_3	+4%	+13%
SO_2	-28%	-35%
NO_x	-30%	-40%

To calculate the sensitivity of $PM_{2.5}$ to NH_3 emissions, we simulate both the base case emission scenario for 2001, 2010, and 2020, and we simulate each of these scenarios with a 10% reduction in NH_3 emissions. The emission reduction is calculated by multiplying by a fixed factor across all NH_3 emission sources, locations, and time periods. CMAQ is run for January 2001 for each of these six scenarios. The inorganic $PM_{2.5}$ (iPM) is the sum of NH_4^+, SO_4^{2-}, and NO_3^- concentrations. The sensitivity of inorganic $PM_{2.5}$ to NH_3 emissions is then calculated as

$$\Delta iPM \Big/ \Delta NH_3 \equiv \frac{(iPM_{1.0} - iPM_{0.9})}{(iPM_{1.0})}$$

where $iPM_{1.0}$ is the monthly-averaged inorganic $PM_{2.5}$ concentration for the base case emission scenario, and $iPM_{0.9}$ is the monthly-averaged inorganic $PM_{2.5}$ concentration for the emission scenario with a 10% decrease in ammonia emissions.

To evaluate the change in reduced-form nitrogen deposition due to changes in future scenario NO_x and SO_2 emissions, we examine the wet deposition, dry deposition, and the sum of wet and dry deposition from gas-phase NH_3 and aerosol NH_4^+ for the entire month at each grid cell. The difference is calculated between the 2001 and 2010/2020 scenarios.

When looking at select non-attainment sites for winter conditions in January, we find that the concentration of inorganic $PM_{2.5}$ is less sensitive to NH_3 emission reductions in the future scenarios. However, the reductions are relatively small and the inorganic $PM_{2.5}$ sensitivity to NH_3 emission reductions remains greater than zero. At locations harboring sensitive ecosystems, we find increases in total reduced-form nitrogen deposition near NH_3 emission sources. More of the total ammonia is in the gas phase which deposits more readily than aerosol ammonium.

Disclaimer

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Concentrations of Nitrogen and Sulfur Species in the Gas and Particle Phases Downwind of Two Dairy Operations

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Abstract

Measurements of gas and particle phase nitrogen and sulfur species were made downwind of two dairy operations in eastern Colorado in 2005. The primary $PM_{2.5}$ ionic species, Cl^-, SO_4^{2-}, NO_3^-, Na^+, NH_4^+, K^+, Mg^{2+} and Ca^{2+}, gas phase concentrations of HNO_3, NH_3, and SO_2, and species' size distributions were measured using cyclone/annular/filter pack systems and multi-stage impactors. A PILS (Particle Into Liquid Sampler)/IC (Ion Chromatograph) system was deployed for high time resolution (10-15 min) measurements of $PM_{2.5}$ composition to provide a better understanding of changes in aerosol concentrations and their relation to transport.

Preliminary analysis of data from the summer and winter 2005 campaigns reveals high concentrations of gas phase ammonia (from ~10-1000 $\mu g/m^3$) downwind of dairy operations. At both study locations more than 90% of the total N(-III) (particulate ammonium + gas phase ammonia) was present as gaseous ammonia. Approximately 40-60% vs. approximately 15% of the total N(V) (particulate nitrate + gas phase nitric acid) were present as nitric acid in summer and winter, respectively. The predominance of gas phase species was favored by hot, dry conditions. Additional observations of particulate species size distributions and temporal variability in species concentrations are also reported.

Introduction

Emissions of ammonia from confined animal feeding operations are a potential concern in terms of their impacts on fine particle concentrations of ammonium nitrate and sulfate. The combined presence of NH_3, H_2SO_4, and HNO_3 can form particulate ammonium sulfate ($(NH_4)_2SO_4$) and particulate ammonium nitrate (NH_3NO_3) through gas to particle conversion reactions (Richardson et al., 1987; Seinfeld et al., 1998). Ammonium nitrate formation is a reversible reaction, with an equilibrium constant dependent on temperature and relative humidity. Relatively high humidities and low temperature favor particulate ammonium nitrate formation. In the absence of sufficient ammonia to fully neutralize sulfate, formation of particulate ammonium nitrate is not favored (Lee et al., 2004). If excess ammonia is available, however, ammonium nitrate can form.

Methods

Commercially available URG cyclone/annular denuder/filter pack systems were used for $PM_{2.5}$ sampling (Figure 1). Ambient air is drawn through a cyclone (D_{50}=2.5 μm, 10 lpm), along two 242mm denuders in series, each coated with chemicals that absorb the gaseous species of interest. The remaining air stream is then filtered through a 2-stage filter pack using a Teflon filter for particle collection followed by a nylon filter to collect any volatilized nitrate. Particle samples were analyzed for Cl^-, SO_4^{2-}, NO_3^-, Na^+, NH_4^+, K^+, Mg^{2+} and Ca^{2+} and denuders were analyzed for gas phase concentrations of HNO_3, NH_3, and SO_2.

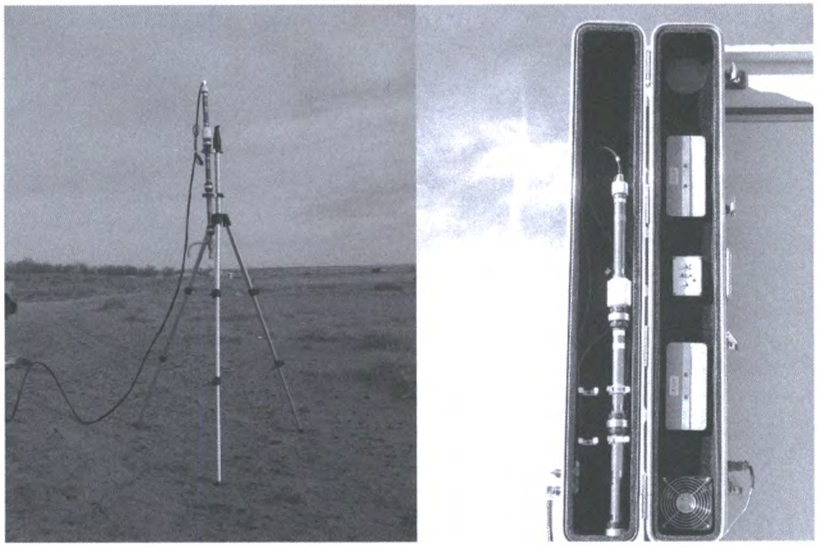

Figure 1. Typical URG sampler setup at upwind (left photo) and downwind (right photo) dairy locations.

Particle size distributions of key species were measured using a Micro-Orifice Uniform Deposit Impactor (MOUDI). The 8 main impactions stages of the MOUDI (Figure 2) collect the following aerodynamic diameter size ranges: (stage 1) 18 –10 µm, (stage 2) 10-5.6 µm, (stage 3) 5.6 – 3.2 µm, (stage 4) 3.2 – 1.8 µm, (stage 5) 1.8 – 1.0 µm, (stage 6) 1.0 – 0.56 µm, (stage 7) 0.56 – 0.32 µm and (stage 8) 0.32 – 0.18 µm and after-filter (<0.18µm). The MOUDI flow rate was 30 L/min. Total sample flow was measured by a pressure-corrected dry gas meter reading. Analysis of the collected size-resolved particulate matter will be focused on quantification of the primary ionic species (Cl^-, SO_4^{2-}, NO_3^-, Na^+, NH_4^+, K^+, Mg^{2+} and Ca^{2+}).

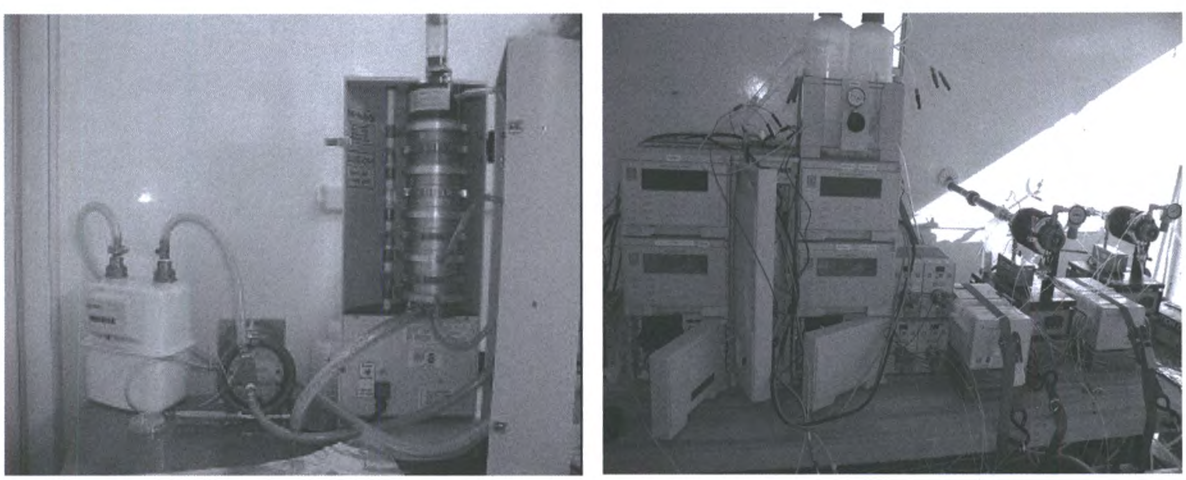

Figure 2. MOUDI (left photo) and (PILS/IC) system (right photo)

Semi-continuous measurements of particle composition were made using a PILS (Orsini et al., 2003). The overall principle of PILS/IC is to collect particles that comprise the $PM_{2.5}$ aerosol mass into a small continuous flow of high purity water. This is done by condensing steam onto a denuded particle stream, followed by impaction of the droplets formed into a liquid stream. The liquid stream is drawn into two ion chromatographs for measurement of aerosol anions and cations. LiBr is used as an internal standard to account for steam dilution of the particle collection stream. High time resolution measurements using PILS/IC systems with switching between "gas + particle" and "particle only" sampling allows quantification of concentrations of major ionic species (Cl^-, SO_4^{2-}, NO_3^-, Na^+, NH_4^+, K^+, Mg^{2+} and Ca^{2+}) of

$PM_{2.5}$ and gaseous species (HNO_3, NH_3 and SO_2) with 10-20 minute time resolution (Figure 2). The "gas+ particle" system operates without denuders.

Results and Discussion

The average MOUDI size distributions measured at dairy operations in 2005 and 2006 are shown in Figure 3. Nitrate, ammonium, and sulfate were dominant ionic species in the fine particle mode. Nitrate is present as both fine mode ammonium nitrate and in the coarse mode, likely as calcium nitrate. Formation of particulate ammonium nitrate (NH_4NO_3) occurs by gas to particle conversion of NH_3 and HNO_3. Coarse mode calcium nitrate likely results from a reaction between gas phase nitric acid (or its precursors) and soil dust particles.

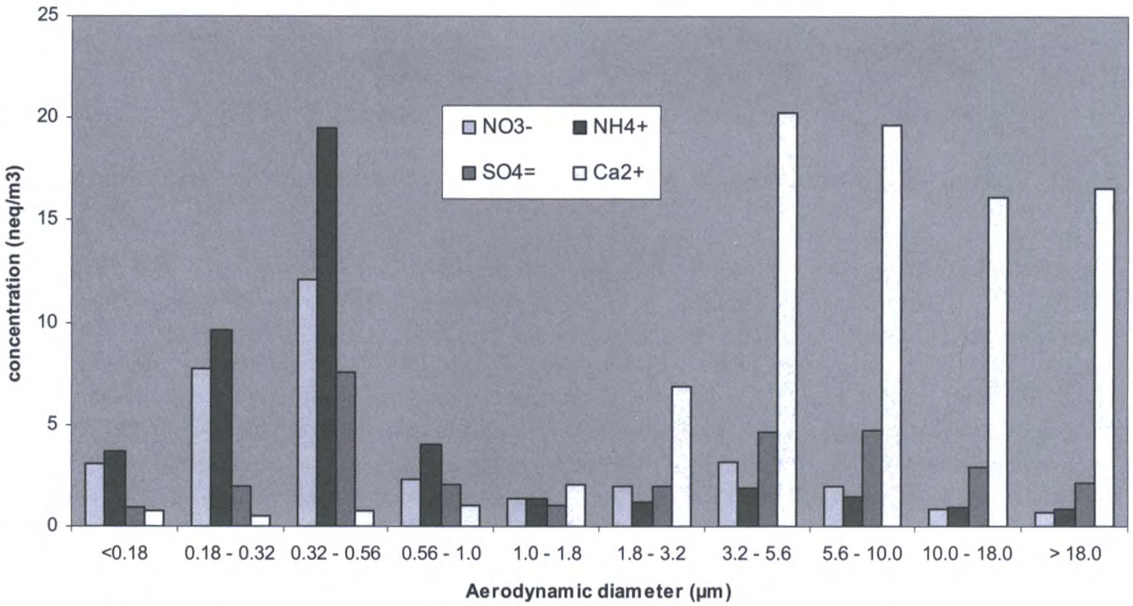

Figure 3. Average ion species' size distributions observed at two dairy operations.

Formation of particulate ammonium nitrate is not favored thermodynamically during warmer periods. While approximately 40-50% of total N(V) (particle nitrate + gas phase nitric acid) was present as nitric acid in July 2005, total N(V) tended to be associated mainly with particles during colder periods in November 2005 (Figure 4). More than 90% of the total N(-III) (particle ammonium + gas phase ammonia) was present as gaseous ammonia both in July and November 2005. The gas-particle phase distribution of sulfur species ranged between 70-90% gas phase in both warm and cool periods.

The PILS was deployed for several days at a dairy operation in November 2005 (Figure 5). Variability in major ion concentrations by wind direction is shown in Figure 6. High particle nitrate was observed during southerly flow, consistent with oxidation of NO_x emitted in urban areas to the south (Fort Collins and Denver are both south of the dairy) followed by particle formation resulting from reaction with gaseous ammonia.

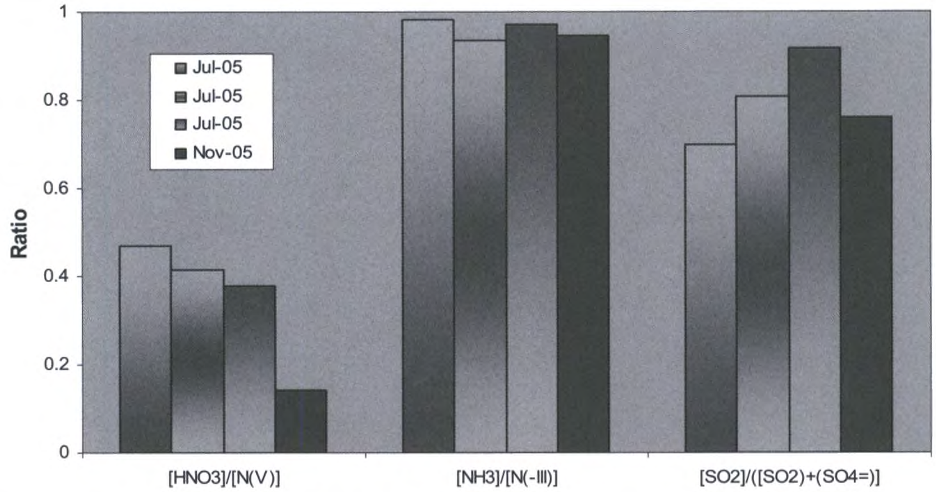

Figure 4. The ratio of HNO_3, NH_3, and SO_2 to total N(V), N(-III), and ($[SO_2]+[SO_4^{2-}]$)

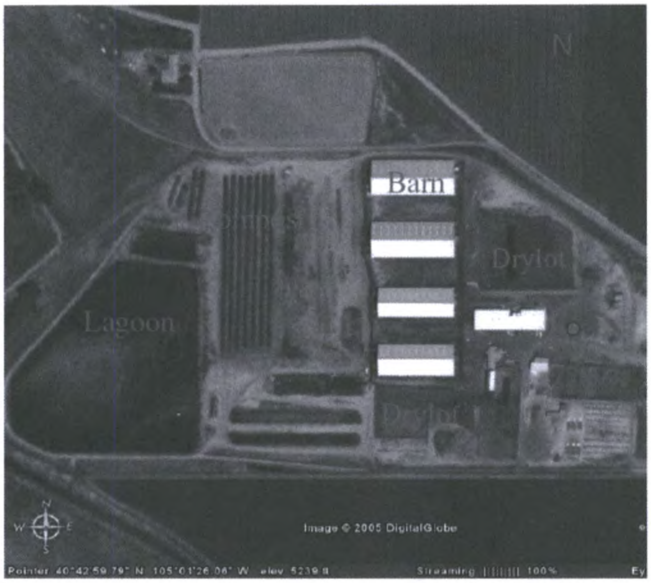

Figure 5. PILS measurement location (red dot) at a dairy studied in November 2005.

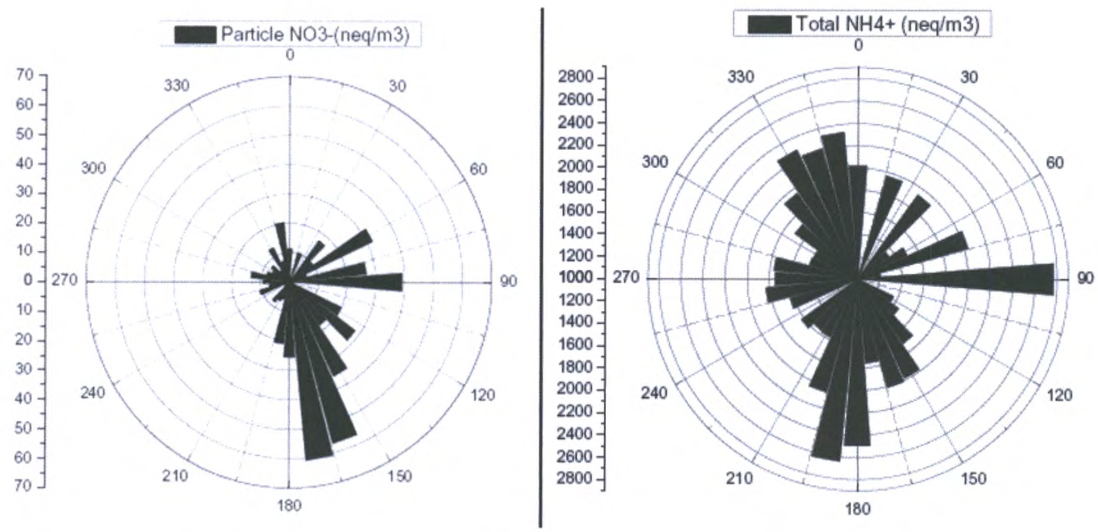

Figure 6. Particle nitrate and total ammonium from PILS with wind direction

PILS total N(-III) (gaseous ammonia + particulate ammonium) concentration variations with wind direction reveal varying contributions by different sources at the dairy. Higher concentrations were observed, for example, with flow from the northwest, consistent with emissions sources located in a large drylot and from barn areas. Comparison of total N(-III) measured by PILS with simultaneous N(-III) measurements with the URG sampler reveal considerable loss of gaseous ammonia within the undenuded PILS system, likely resulting from capture of ammonia by wet interior surfaces of the steam condensation chamber. Future efforts will make use of an alternative steam sampler design that is more efficient at total N(-III) collection.

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Agriculture and Air Quality – Airborne Particulate Matter

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Abstract

Air quality impacts from agricultural activities have generating considerable interest in the past decade. This increased interest stems from several reasons, including urban sprawl, which has led to closer proximity between urban populations and rural, farming areas; intensification of some agricultural activities, such as the increasing size of animal feeding operations; better understanding of secondary air pollutant formation, including secondary aerosols; and continuing revisions of national ambient air quality standards, including those for particulate matter (PM), reflecting better understanding of the effects of exposure to air pollutants. The U.S. Environmental Protection Agency has recently completed a review of the national ambient air quality standards (NAAQS) for PM and has proposed significant revisions to the primary and secondary NAAQS for PM. These changes, which include revision of the 24-hr PM2.5 standard, have resulted from new understanding of the role that fine particles (typically nominally smaller than 2.5 micrometers in aerodynamic diameter) play in health effects associated with short-term exposure. New standards are also proposed for a new indicator of thoracic particles – those particles with aerodynamic diameters between 2.5 and 10 micrometers (PM10-2.5) to target "urban" dust – i.e., resuspended road dust and industrial and construction sources, but excluding "rural" dust including windblown dust and PM from agricultural fields. This proposed PM10-PM2.5 standard would replace the current PM10 standard. A new, sub-daily fine particle standard is also under discussion. The agricultural sources of ambient particulate matter are reviewed, as well as their chemical and size characteristics, and what is known of their emission inventories.

Thoracic Particles

Thoracic particles (also commonly known as coarse fraction particles, or approximately PM10-2.5) are airborne particles that are generated from mechanical processes. Thoracic particles associated with agricultural activities are usually geologic in origin, and include windblown dust from fields, dust from rural roads, as well as dust generated by specific farming practices such as harvesting, plowing, or planting. Animal feedlots may also represent a source of thoracic particles. In these cases, the particles likely also contain bioaerosols, endotoxins, or allergens.

Windblown dust. Dust from dry fields especially during windy periods remains a concern for many agricultural areas in the US, particularly in arid regions. Windblown dust represents an aesthetic nuisance at best, and in worst cases has led to traffic fatalities due to significantly impaired visibility (Sundram et al., 2004). There is little current information to suggest that inhalation of windblown dust from agricultural fields represents a significant health risk, although reports of "dust pneumonia" were somewhat common during the Dust Bowl Era. There is some limited work that has suggested a link between exposure to coarse fraction particulate matter and respiratory ailments, although not specifically to agricultural coarse fraction PM. There have been several significant efforts to understand those conditions that lead to windblown dust, through monitoring and modeling activities. Some of these modeling efforts have included the Columbia Plateau PM10 Project in Central Washington, and the Wind Erosion Prediction System developed by scientists at the USDA. Modeling of windblown dust requires meteorological inputs, up-to-date land use information, terrain data, knowledge of surface treatment, and significant soil characterization including moisture, clay content, and amount of aggregation. Compounding the complexities of assembling this information is the need for detail at high temporal and spatial resolutions. Most modeling efforts have been conducted for very specific regions, and all are empirical or semi-empirical in nature. One significant exception is the WEPS, which takes into account detailed historical information such as the biomass both above and below ground, soil surface treatments, and precipitation history, and which tracks the resulting soil conditions in a chronological fashion.

Specific farming activities. Dust from a number of specific farming activities, such as almond harvesting or field plowing, has been studied, especially in California, in an effort to determine the contributions from these activities to particulate pollution there.

Animal feeding operations. Thoracic particles can be emitted from animal feeding operations (AFOs) through several mechanisms including animal activity, air circulation fans in housing units, entrainment of mineral and manure dust, and spraying of liquid wastes onto fields. These activities have not been extensively characterized for their PM emission potentials, although some studies, especially in Europe, are reported in the peer-reviewed literature, especially for dust potential in animal housing.

Measurement issues. Researchers particularly at TAMU have reported on oversampling problems that arise in the EPA Federal Reference Method (FRM) for particulate matter (Capareda et al., 2005). Oversampling occurs as a result of the confluence of particle size distributions and performance characteristics of the FRM PM10 sampler. PM10 samplers ideally would collect particles smaller than 10 micrometers in aerodynamic diameter with 100% efficiency, and particles larger than 10 micrometers would not be collected at all. In reality, however, some larger particles are also collected, and the PM10 performance is thus based upon collection of particles of 10 micrometers at 50% efficiency. Parnell and coworkers argue that agricultural PM is larger in particle size than typical urban PM10, so that the entire particle size distribution from agricultural activities is shifted toward larger particles. They have shown that this results in oversampling of PM10 when a FRM PM10 sampler is used. These researchers have developed a correction procedure that uses the ratio of PM10 to TSP to correct for the larger size distributions for various types of dust.

Emission inventories. Several compilations of emission factors have been produced, including the EPA document, AP-42, and the recent studies sponsored by the Western Governors Association – Western Regional Air Partnership (WGA-WRAP) Fugitive Dust studies.

Health questions. The EPA has recognized that exposure to thoracic particles generated from agricultural activities does not appear to represent a significant health risk, and so agricultural PM10-PM2.5 has been recommended for exemption from regulation. However, there remains other potential health issues that should be examined and which may be difficult to delineate from co-pollutants (other pollutant species that occur at the same time or from the same source). These include potential exposure to pathogens from blowing dust containing manure particles, endotoxins on bioaerosols from animal feeding operations (e.g., Rosas et al., 2001), as well as pesticides or metals carried on dust particles. In addition, there is some evidence to suggest that PM carries odor-causing species, thus making it difficult to separate odors from PM.

Fine Fraction PM

Primary fine particles from agricultural activities include smoke from field burning, as well as secondary aerosols formed in the atmosphere, for example from reactions involving gaseous ammonia or hydrogen sulfide from animal feeding operations.

Biomass combustion. Fire is used as a tool for managing field residues after harvest, weeds and pests, and wastes. Fire is also used in forestry management for the same reasons. Smoke from burning biomass may cause visibility problems and respiratory problems especially for certain vulnerable groups. As suburban areas sprawl toward farming areas, these problems become more commonplace. Smoke from biomass burning includes a number of air pollutants, including not only fine particles, but carbon monoxide, and a host of volatile and semivolatile organic compounds that include polycyclic aromatic hydrocarbons (PAHs) and methoxyphenols (MPs). Some PAHs have been found to be carcinogenic, while some MPs are respiratory irritants. While there has been some work to characterize both the chemical composition and emissions of PM and some PAHs and MPs in both vapor and condensed phases for some commonly burned biomasses (e.g., sugar cane residue, wheat residue, certain types of trees), this is an area that requires significantly more work. For example, recent studies conducted on Kentucky Blue Grass field stubble burning have shown significantly different PM emission factors and PAH and MP composition than similar studies conducted on wheat stubble burning.

Secondary agricultural sources. One of the least understood sources of agriculturally related PM is secondary aerosol – particles that result from condensation or chemical reactions between agriculturally related gaseous species in the atmosphere. Probably the most important PM precursors from agriculture are ammonia and hydrogen sulfide (e.g., Baek et al., 2004). Significant sources of these two precursors are animal feeding operations of all kinds. Dairy farms, poultry farms, and pig farms in particular can be significant sources of precursors to PM.

Ammonia and hydrogen sulfide are emitted from wastewater lagoons, manure spraying operations, as well as manure piles (e.g., Aneja et al., 2000; Aneja et al., 2001). Emissions from lagoons have been relatively extensively studied, particularly as improvements in ammonia analyzing capabilities have been realized (e.g, Mount et al., 2002). Still, the high degree of variability between operations makes it difficult to quantify the extent of ammonia and hydrogen sulfide emissions from AFOs.

Measurement issues. Measurement issues for PM2.5 from agricultural activities are similar to those from other sources, and include organic artifacts from adsorption and/or desorption from filter media (e.g., Pang et al., 2002), and equilibrium issues especially with desorption of ammonium nitrate during sampling. In addition, researchers at TAMU also showed that there was a shift in the 50% cutpoint diameter for the PM2.5 FRM from the expected 2.5 micrometers to 2.7 $\pm$ 0.4 micrometers, so that the FRM form PM2.5 also oversamples agriculturally derived PM2.5 (Capareda et al., 2005).

Emission inventories. Emission factors for PM2.5 from fugitive dust including dust from paved and unpaved roads, construction sites, and windblown dust, as originally listed in AP-42, were developed using high-volume cyclone/impactor systems that have been shown to have a positive bias by as high as a factor of 2, compared to FRM-derived factors. Moreover, PM2.5/PM10 have sometimes been assumed to be constant but in fact for a variety of western soils this was not found to be true – rather, the ratio decreased with increasing PM10 concentrations. Emission factors from agricultural field burning have been quantified for a few crops, including wheat and Kentucky Blue Grass (Dhammapala et al., 2006, and references therein).

Health questions. Potentially hazardous compounds in agriculturally generated PM2.5 can include PAHs and methoxyphenols in biomass smoke (e.g., Roberts and Corkill, 1998). Because smoke from agricultural operations often impacts less populated areas, and because of the relatively short duration, but high concentration exposures, it has been difficult to demonstrate a statistically significant link between exposures and health effects, although recent epidemiological studies have been conducted in eastern Washington (Wu et al., submitted; Sullivan et al., submitted). Anecdotal evidence continues for an association between respiratory problems and smoke exposure, especially among people with asthma, and the practice remains controversial in various parts of the country (e.g., http://www.safeairforeveryone.com/).

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Modeling Approach to Estimate PM_{10} Emissions from Pig Husbandry

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Abstract

The level of measured dust emissions from livestock operations is generally determined by the number and weight of the animals as well as characteristics of the housing system, specified by management, animal species, feeding practices, bedding material, ventilation, as well as climatic inside and outside conditions. Modeling can be used to quantify the impact of management, seasonal, and diurnal variations. This approach combines measurements of the main influencing variables and estimated model parameters. The objective of the study was to estimate and model PM_{10} emissions from mechanically ventilated pig facilities.

Investigations were performed in mechanically ventilated pig houses for weaning, fattening, and sows in Italy and Germany. Thereby, the measurements included inside and outside concentrations of airborne PM_{10} particles (scatter light photometry), ventilation rate (calibrated measuring fans), and indoor air climate at a measuring frequency of 60 sec. Furthermore, feeding times and animal related data like weight and the (group) animal activity were recorded. Based on the sampled data, a multiple-input, single-output transfer function model was taken to describe the dynamic response of dust to variations of the measured variables.

As the results indicate, the average PM_{10} emission rate was influenced largely (> 100 %) by the specific housing system and/or management in the facility. Based on the three input variables ventilation rate, animal weight, and animal activity as well as model parameters, received on-farm, the minutely measured emission rate was simulated with a mean percentage error of 21 to 39 %. The simulated and measured average emission rate differed by about 4 % to 19 %.

Further recommendations of the study were to improve continuous and accurate measurements on the activity level in animal houses, to optimize the amount of measuring days in relation to the model accuracy, and to relate the modeling approach to intermittent measurements.

Introduction

Concentrations and emissions of dust particles from livestock operations are generally determined by the characteristics of the respective housing system, specified e.g. by management, feeding practices, bedding materials, number of animals, animal species, ventilation, as well as climatic inside and outside conditions (Takai et al., 1998).

Health impacts of aerosol particles depend mainly on their biological and chemical composition. Thereby, the origins of aerosols found in animal husbandry are mainly feed, bedding materials, skin, and excrements. Thus, they consist of a complex mixture made of organic dust as well as of relatively high concentrations of microorganisms, endotoxins, and NH_3 (Donham et al., 1986; Aarnink et al., 1999; 2004; Schneider et al., 2001; Seedorf, and J. Hartung; 2002; Takai et al., 2002). Such kinds of aerosols, which contain biologically active components and hence are likely to cause infections, allergic, toxic, or pharmacological reactions, are labeled in general as bio-aerosols (Cox, and Wathes, 1995). Aerosol particles emitting from animal husbandry systems can be a nuisance towards the neighborhood, but can also be environmentally harmful (IPCC, 2005).

The emission rate of PM_{10} particles is in general determined by their release and dispersion from sources into the air, as well as by ventilation to the ambient. High ventilation rates during summer are accompanied in general with low indoor concentrations but increased emission rates, especially of inhalable and PM_{10} particle size fractions (Takai et al., 1998; Aerts et al., 2004; Jacobson et al., 2004; Koziel et al., 2004). Keck et al. (2004) investigated an increased impact of open yard exercises on the level of the absolute PM_{10} emission rate from pig husbandry during summer than during winter, as well as a significant influence on PM_{10} emissions with increasing growth stage of growing-finishing pigs. High peak-to-mean ratios during control activities of the farmers can be found especially in husbandry systems which use straw as bedding materials (Hartung, E. et al., 2004).

The influence of the animal activity on the dust concentration and emission is in general derived from observed differences between day and night-time and are related mainly to control activities of the farmers and feeding operations (Takai et al., 1998; Hinz and Linke, 1998; Wang et al., 2002), or it is derived from recorded characteristic daily patterns of the measured dust emission rate (Zeitler-Feicht et al., 1991; Koziel et al., 2004). Nevertheless, Gallmann (2003) and Schneider (2005) investigated also direct correlations between the animal activity and the PM_{10} concentration, both recorded at minutely frequencies. In general, it can be assumed that the dispersion of settled dust leads not only to a higher indoor dust concentration but also to an increase of the dust emission rate.

Modeling can be used to quantify the impact of management, seasonal and diurnal variations. This approach combines measurements of the main influencing variables and estimated model parameters. Hereby, the objective of the study was to estimate and model PM_{10} emissions from mechanically ventilated pig facilities.

Materials and Methods

Dust indoor concentrations and emissions were investigated in facilities for weaning, fattening and sows in Italy (Pianura Padana, Milan, Table 1) and Southern Germany (Hohenheim, Table 2). The investigated pig houses were equipped with mechanical ventilation systems each, separately controllable ventilation fans, and calibrated measuring fans. Temperature and humidity were measured and recorded via sensors of the ventilation control system and/or by additional sensors. Further recordings included the group animal activity in one pig facility, as well as the animal weight and feeding times in all investigated houses.

Measurements in Facilities for Weaning, Fattening and Sows in Italy

The measurements in Farm 1 in Italy (weaning and sows) included inside and outside concentrations of airborne PM_{10} particles (particles < 10 µm), ventilation rate, and indoor air climate with a measuring frequency of 1 min. Dust particles were sampled continuously ones per minute in the incoming air stream (weaning) as well as inside, close to one exhaust chimney (all facilities). Farm 2 (weaning) and Farm 1 (fattening) included measurements of PM_{10} concentrations at a frequency of 1 min; ventilation rate, outside and inside temperature, and humidity were recorded with a frequency of 15 min. In each of these facilities, the PM_{10} concentration was monitored optically using calibrated scatter light photometers (EPAM 5000, HAZ-Dust™; accuracy: +/- 3µg m^{-3}).

Temperature and humidity were recorded by the ventilation control system (FANCOM). Further, the measurements of the ventilation rate were performed with a high accuracy, separately for each exhaust chimney, using calibrated measuring fans (FANCOM; accuracy: +/- 45 m^3 h^{-1}; Berckmans et al., 1991).

The floor of the compartments was covered either with grooved plastic slats (weaning) or featured slatted concrete floor elements (fattening, sows). Feeding was applied dry or liquid, ad libitum or at three to four fixed feeding times per day (Table 1). A detailed description of the facilities is provided in Costa et al. (2004) and Costa et al. (2005).

Table 1. Investigated time periods and research facilities in Italy

	Number of animals	Average animal weight	Food supply and Feeding times	Floor material
Sows Farm 1, 18 Feb-25 Mar 04	24 to 43 sows	approx. 180 kg	dry (ad libitum)	slatted concrete floor
Weaning Farm 1, 25 Sep-9 Nov 04	350 piglets	7 kg to 36 kg	dry (ad libitum)	grooved plastic slats
Weaning Farm 2, 5 Sep-10 Oct 05	250 piglets	6-7 kg to 30 kg	liquid + dry (manually) 7:40; 13:40; 17:00	grooved plastic slats
Fattening Farm 1, 15-23 Nov 05	329 pigs	71 kg to 77 kg	liquid (automatically) 9:30; 12:30; 16:30; 19:30	slatted concrete floor

Measurements in the Fattening Pig Facility in Germany

Investigations in the research facility for fattening pigs in Germany were performed during two fattening periods (18 Nov 2003 until 24 Feb 2004 and 13 Apr 2004 until 20 Jul 2004) in two compartments. Thereof, dust concentrations were measured at four time periods (1080 min to 2550 min) per fattening period, either in compartment 1 or in compartment 2 (Table 2).

Airborne PM_{10} particles (particles < 10 µm) were monitored simultaneously at one representative measuring point indoors (all measuring periods), as well as in the exhaust shaft (six out of eight measuring periods). For the optical dust monitoring, calibrated scatter light photometers (TSI DustTrak 8520 ™; accuracy: +/- 1 $\mu g\ m^{-3}$) were used.

A vertical projection of the research facility and detailed table about the measuring instruments and accuracies for the measurements of temperature, relative humidity, ventilation rate, and animal activity is given in Haeussermann et al. (2006). Thereof, the group animal activity per pen (27 pigs) was recorded with infrared sensors (Pedersen and Pedersen, 1995), the ventilation rate was measured using calibrated measuring fans (MULTIFAN; accuracy: +/- 20 $m^3\ h^{-1}$; Hartung, E., 2001). The measuring frequency for dust, ventilation rate, indoor temperature, humidity, and animal activity was one average value per minute.

The two pens per compartment featured a concrete slatted floor. Straw was supplied weekly via an occupation equipment, type "Porky Play" (straw capacity: approx. 3-4 kg). Feed was provided either liquid or in mash form (Table 2). Fresh air was coming in via two air inlet pore channels per compartment, the outgoing air was extracted under-floor (Haeussermann et al., 2006).

Table 2. Investigated time periods in the research pig facilities in Germany

	Number of animals	Average animal weight	Food supply and Feeding times	PM_{10} measurements
12-13 Dec 2003 09-11 Feb 2004 25-26 May 2004* 29-30 Jun 2004	53 pigs 53 pigs 53 pigs 53 pigs	50 kg 101 kg 61 kg 90 kg	Liquid (sensor) automatically 20 feeding times: 6:00 until 22:00)	Indoors[1] & exhaust shaft
22-23 Dec 2003 21-22 Jan 2004 26-27 Apr 2004 08-09 Jun 2004	54 pigs 54 pigs 54 pigs 54 pigs	58 kg 85 kg 35 kg 73 kg	mash (ad libitum) automatic food supply at six feeding times: 6:10 until 20:10	Indoors[1] - Indoors[1] & exhaust shaft

* no measurements of temperature, humidity, ventilation rate, and animal activity (25-26 May 2004)
[1] control corridor, height: 1.70 m

Modeling Approach and Model Validation

The aim of modeling in this study was to simulate PM_{10} emissions, based on measurements of the main influencing variables and estimated model parameters. The dynamic response of the dust concentration and emission to variations of the measured variables was described by a multiple-input, single-output transfer function model (Young, 1984). The general structure of the used model is exemplified for a single-input by the following equation (Aerts and Berckmans, 2004):

$$y(k) = \frac{B(z^{-1})}{A(z^{-1})} u(k) + \xi(k) \qquad (1)$$

where $y(k)$ is the output, either of the dust concentration, mg m^{-3} or the dust emission rate, g h^{-1} at time k; $u(k)$ is the respective input of the measured variable at time k. Thereby, input variables were selected based on their correlation with the PM_{10} concentration (Table 4 and Table 5) and via a multiple-input regression analysis. $\xi(k)$ is additive noise, assumed to be a zero mean, serially uncorrelated sequence of random variables with variance σ^2, accounting for measurement noise, modelling errors and effects of unmeasured inputs to the process. $A(z^{-1})$ and $B(z^{-1})$ are two series given by:

$$A(z^{-1}) = 1 + a_1 z^{-1} + a_2 z^{-1} + \ldots + a_{na} z^{-na} \qquad (2)$$

$$B(z^{-1}) = b_0 + b_1 z^{-1} + b_2 z^{-2} + \ldots + b_{nb} z^{-nb} \qquad (3)$$

where: a_j, b_j are the model parameters to be estimated; z^{-1} is the backward shift operator, with $z^{-1}.y(k) = y(k-1)$; y and k are defined as in Equation (1); na, nb are the orders of the respective polynomials. The model parameters were estimated in discrete time using the simplified recursive instrumental variable approach (Young, 1984). Time step of the model was one value per 60 sec.

The modeling approach was tested on four data sets of the research facility for fattening pigs, Germany (Table 2). Selection criteria for the data sets were complete data, including indoor and exhaust PM_{10} concentrations, ventilation rate, temperature, humidity, and animal activity. The data set of 26-27 Apr was not selected due to considerable differences in the relations between dust and influencing variables, compared to the rest of the data sets. These differences were mainly explained by the early measuring period (fattening day 13-14) and related disturbances. Thus, a limitation of the model was given for the period between fattening day 24 (50 kg average weight; 12-13 Dec) and fattening day 85 (101 kg average weight; 09-11 Feb).

The model parameter estimation was performed on three of four data sets at each time, the validation of each model took place at the respective data set, which was not taken to build the model (Table 6). Validation criteria were (i) the ability to simulate the average measured dust concentration and emission of the validation set, and (ii) the agreement of the minutely measured and simulated values, expressed by the coefficient of determination R^2 and the root mean squared error (RMSE).

Results and Discussion

Variation of the Dust Concentration and Emission

The average PM_{10} indoor concentration of the different pig facilities varied from 0.11 to 0.73 mg m^{-3} (Table 3). Thereby, the lowest PM_{10} concentration occurred in the research facility for weaning pigs in Italy; the highest concentration was found in the research facility for fattening pigs in Germany. Compared to these minimum and maximum average values, the mean PM_{10} concentration at the three remaining facilities (weaning Farm 2, fattening pigs and sows, Farm 1, Italy) featured a narrower range of 0.33 to 0.48 mg m^{-3} (Table 3).

According to the dust indoor concentration, also the average PM_{10} emission rate featured a high variation throughout the stables for sows, weaning, and fattening pigs, which were in a range between 0.7 and 6.0 g d^{-1} LU^{-1} (Table 3; 1 LU = 500 kg). Similar to these investigations, the mean PM_{10} emission rates reported in literature, considering facilities for fattening pigs, varied from 0.8 to 4.3 g d^{-1} LU^{-1} (Götz, 2003; Aerts et al., 2004; Jacobson et al, 2004; Koziel et al., 2004). PM_{10} emission rates investigated by Jacobson et al. (2004) in gestation and dry sow barns were slightly higher and ranged from 0.5 to 9.1 g d^{-1} LU^{-1}.

In principle, feeding operations, the level of the animal activity, the ventilation rate, the indoor temperature, the animal weight and/or the fattening day, the housing system, as well as the management in the facility exerted an influence on the level of the dust indoor concentration and dust emission rate (Table 4; Table 5).

Thereby, the difference in the average PM_{10} emission rate between the research facilities for weaning and fattening in Italy and Germany (Table 3) can be associated on the one side with the higher ventilation rate per LU at the latter, and hence mainly due to seasonal difference. On the other side, the lower average weight per animal in the weaning stable, in combination with differences in the farm management, led to a very low dust production, which was mirrored also by a relatively low indoor concentration at this farm (Table 3). Similarly, Gallmann (2003) reported a mean PM_{10} indoor concentration of 0.17 mg m^{-3} in a free ventilated kennel housing system for fattening pigs, during measurements outside the kennels, where the air quality is only minor influenced from the animals. Hereby, the ambient dust concentration outside the weaning farm in Italy averaged on 0.02 mg m^{-3} with a maximum value of 0.17 mg m^{-3}.

Table 3. Ventilation rate, PM_{10} indoor concentration and PM_{10} emission rate at the research facilities for sows, weaning and fattening pigs, Italy and Germany

	Ventilation rate [m^3 h^{-1} LU^{-1}]	Indoor concentration [mg m^{-3}]	Emission [g d^{-1} LU^{-1}]
Sows Farm 1, Milan 18 Feb-25 Mar 04	**96**[1] (63-308)[2] (20-577)[4]	**0.33**[1] (0.15-0.81)[2] (0.00-10.6)[4]	**0.70**[1] (0.26-1.75)[2] (0.00-18.0)[4]
Weaning Farm 1, Milan 25 Sep-9 Nov 04	**273**[1] (175-387)[2] (132-650)[4]	**0.11**[1] (0.06-0.17)[2] (0.00-3.23)[4]	**0.71**[1] (0.40-1.09)[2] (0.01-23.2)[4]
Weaning Farm 2, Milan 5 Sep-10 Oct 05	**245**[1] (160-328)[2] (81-827)[4]	**0.40**[1] (0.25-0.50)[2] (0.10-16.5)[4]	**2.38**[1] (1.20-3.60)[2] (0.43-24.2)[4]
Fattening Farm 1, Milan 15-23 Nov 05	**174**[1] (141-230)[2] (105-324)[4]	**0.48**[1] (0.32-0.65)[2] (0.02-5.61)[4]	**2.04**[1] (1.35-2.71)[2] (0.52-17.0)[4]
Fattening Farm, Hohenheim 12 Dec-30 June 04*	**481**[1] (235-854)[3] (189-1070)[4]	**0.73**[1] (0.35-1.26)[3] (0.06-6.41)[4]	**5.99**[1] (3.29-9.54)[3] (0.58-29.8)[4]

[1]mean [2]range diurnal means [3]range mean values data sets (*cf. Table 2) [4] total range
LU: Livestock Unit [1 LU = 500 kg animal weight]

Influences on the Dust Concentration and Emission

Correlation coefficients for the main influencing variables on the dust indoor and exhaust concentration in the research facilities for fattening pigs are demonstrated in Tables 4 and 5. In general, only low to negligible correlations occurred between the highly dynamic varying values of the minutely measured dust concentration and influencing variables like temperature, ventilation rate, animal activity, total weight, feeding times, and indoor humidity (Table 4 and Table 5). Nevertheless, the exhaust dust concentration at the research facility for fattening pigs, Germany, was explained with a regression coefficient of $R^2 = 0.61$ when taking into account all influencing variables listed in Table 4, except for the PM_{10} indoor concentration. In order to reduce the number of input variables, a sensitivity analysis was performed, taking into account either:

- indoor humidity, indoor temperature and/or ventilation rate,
- animal activity or feeding time,
- total animal weight or fattening day.

In conclusion, including the ventilation rate, the indoor humidity, the animal activity, and either the fattening day or the total weight of the animals as input variables resulted already in an R^2 of 0.61 and 0.60. Without indoor humidity, the regression coefficient was slightly lowered on $R^2 = 0.59$. In comparison, keeping the indoor temperature in the model but eliding the ventilation rate or the indoor humidity reduced the regression coefficient clearly to $R^2 = 0.56$. When replacing the animal activity by feeding times, the regression coefficient was reduced furthermore on $R^2 = 0.43$ if all other input variables were included and to $R^2 = 0.37$ if only the ventilation rate, the indoor humidity, and the weight of the animals were used in addition. Thus, in a first step, the animal activity, the ventilation rate, and the animal weight were taken into account for building a transfer function on the dust concentration and emission (Table 6).

Table 4. Correlation coefficients, research facility for fattening pigs, Hohenheim

	PM_{10} exhaust concentration	Dust indoor concentration	Tempe-rature	Relative humidity	Ventilation rate	Animal activity	total weight
PM_{10} indoor con.	0.943 **						
Temperature	- 0.435 **	- 0.420 **					
relative humidity	0.075 **	0.130 **	- 0.292 **				
Ventilation rate	- 0.314 **	- 0.332 **	0.682 **	- 0.501 **			
Animal activity	0.483 **	0.543 **	0.151 **	0.185 **	- 0.007		
Feeding time	0.145 **	0.159 **	0.008	0.027 *	0.010	0.220 **	
total weight	0.357 **	0.298 **	- 0.338 **	- 0.627 **	0.260 **	- 0.104 **	
Fattening day	0.310 **	0.250 **	- 0.270 **	- 0.659 **	0.344 **	- 0.119 **	0.995 **

significance: * $p < 0.05$ ** $p < 0.01$

A similar tendency for the correlation coefficients of the main influencing variables and the dust concentration like for the fattening pig facility in Germany, described above, was investigated at the research facility for fattening pigs in Italy (Table 5). Nevertheless, the indoor temperature and ventilation rate featured here a positive correlation with the dust concentration, the indoor humidity was correlated negatively instead. As the animal activity was not monitored, the maximum correlation resulted here in a regression coefficient of $R^2 = 0.23$, either by including all five available input variables or by taking into account the ventilation rate, the indoor humidity, the feeding times and the fattening day only.

Thereby, the influence of the ventilation rate on the dust concentration has to be regarded in general differentiated. During short time periods, an increase of the air exchange in the room can disperse settled dust and thereby result in an increased concentration of airborne particles. In addition, the effect of an increased ventilation rate on the spatial dust concentration and dust distribution depends largely on the respective airflow pattern in the building (Wang et al., 2002). A positive correlation between the ventilation rate and the dust concentration was found both in the fattening pig facility in Italy as well as in three of the four data sets of the research facility for fattening pigs in Germany when considering them separately. However, if the ventilation rate is clearly increased for a longer period, settled dust will be transported to the outside and a negative correlation is found between the ventilation rate and the concentration of airborne dust (Table 4).

Table 5. Correlation coefficients, research facility for fattening pigs, Farm 1, Milan

	Dust indoor concentration	Temperature [°C]	Relative humidity [%]	Ventilation rate [$m^3 h^{-1}$]	Total weight [LU]
Temperature	0.261 **				
relative humidity	- 0.231 **	0.184 **			
Ventilation rate	0.272 **	0.947 **	0.212 **		
Feeding times	0.237 **	0.131 **	0.046	0.127 **	
total weight / Fattening day	0.129 **	- 0.533 **	- 0.590 **	- 0.525 **	1

significance: ** $p < 0.01$

First Results on Modeled Dust Concentration and Emission

As a first result of the model validation, the dynamic modeling approach enabled to simulate the mean dust indoor and exhaust concentration in tendency (Table 6). The average emission rate was simulated with an error of 4 % to 19 %, compared to the measured mean values. Thereby, the mean percentage error of the minutely simulated dust emission was in a range of 21 % (Figure 1) to 39 % (Figure 2). Although the dynamic course of the dust concentration and emission was simulated rather good on the data set of 12-13 Dec 2003 (Figure 2), which resulted in an R^2 of 0.70 to 0.77, the difference between the measured and the simulated mean was highest for this data set. Thereby, the difference between the measured and simulated mean at this validation was mainly due to the missing information about the influence of the low animal weight when estimating the model parameters from the three other data sets (Table 6). In contrary, the simulation of the mean dust concentration and emission was clearly improved for the two validation sets in June 2004 (Figure 1: 8-9 June). For both data sets the transfer function, built upon the respective three other data sets, included the information of the total variation of the ventilation rate and the animal weight, thus enabled a rather good simulation of the mean level of the respective validation set.

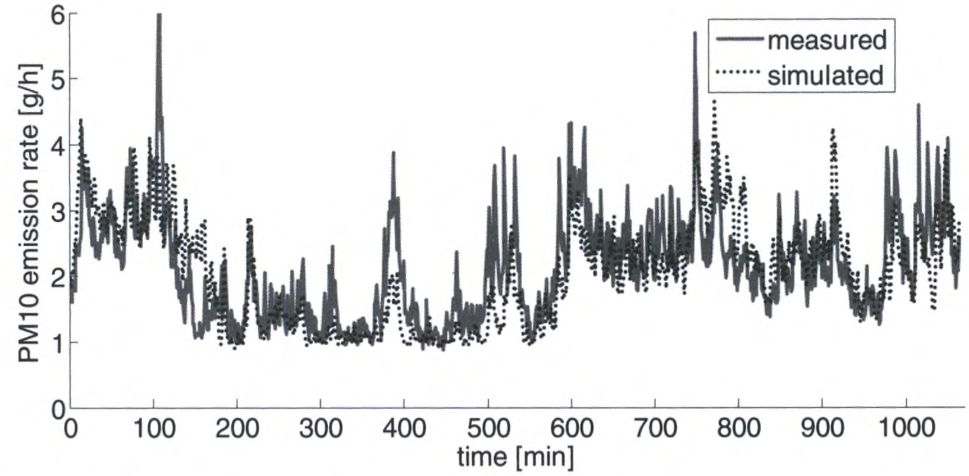

Figure 1. Measured and simulated PM_{10} emission rate, 8-9 June 2004 (54 pigs, average weight: 73 kg, mash feeding, ad libitum); input variables: ventilation rate, animal activity, and animal weight (Mean measured/simulated: 2.13/2.05, RMSE: 0.60, R_2: 0.529)

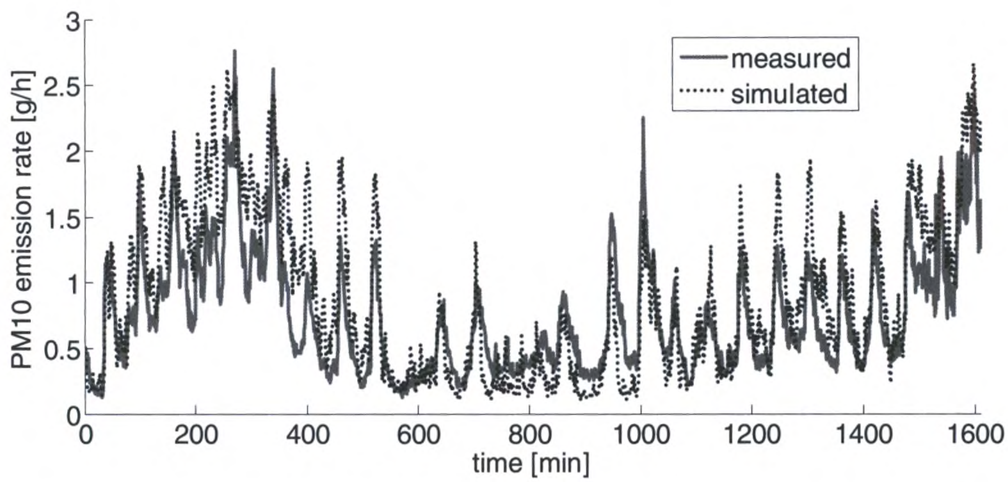

Figure 2. Measured and simulated PM_{10} emission rate, 12-13 Dec 2003 (53 pigs, average weight: 50 kg, sensor-liquid feeding); input variables: ventilation rate, animal activity and animal weight (Mean measured/simulated: 0.75/0.89, RMSE: 0.34, R_2: 0.772)

Table 6. Validation* (leave-one-out) of the transfer function model on PM_{10} concentration and emission, research facility for fattening pigs, Hohenheim (Input variables: ventilation rate, animal activity and animal weight)

	12-13 Dec 2003 (liquid feeding)	9-11 Feb 2004 (liquid feeding)	8-9 Jun 2004 (mash feeding)	29-30 Jun 2004 (liquid feeding)
Animal weight [LU]	5.3	10.7	7.9	9.7
ventilation rate [m^3 h^{-1} LU^{-1}]	mean: 371 *range: 246 – 649*	mean: 243 *range: 189 – 429*	mean: 854 *range: 796 – 907*	mean: 752 *range: 550 – 833*
PM_{10} Indoor concentration [mg m^{-3}]	**0.57[1] / 0.72[2]** *+- 0.29[3]*	**1.23[1] / 1.01[2]** *+- 0.65[3]*	**0.35[1] / 0.39[2]** *+- 0.29[3]*	**0.38[1] / 0.39[2]** *+- 0.20[3]*
	R^2: 0.702	R^2: 0.517	R^2: 0.351	R^2: 0.613
PM_{10} exhaust concentration [mg m^{-3}]	**0.37[1] / 0.52[2]** *+- 0.19[3]*	**0.79[1] / 0.58[2]** *+- 0.41[3]*	**0.31[1] / 0.30[2]** *+- 0.16[3]*	**0.30[1] / 0.33[2]** *+- 0.15[3]*
	R^2: 0.720	R^2: 0.482	R^2: 0.544	R^2: 0.442
PM_{10} emission rate [g h^{-1}]	**0.75[1] / 0.89[2]** *+- 0.34[3]*	**2.37[1] / 2.11[2]** *+- 1.07[3]*	**2.13[1] / 2.05[2]** *+- 0.60[3]*	**2.22[1] / 2.34[2]** *+- 0.84[3]*
	R^2: 0.772	R^2: 0.575	R^2: 0.529	R^2: 0.483

* Model is build out of three data sets, validation is performed at the respective fourth data set
[1] measured / [2] simulated; [3]RMSE: root mean squared error; LU: Livestock Unit [1 LU = 500 kg]

In order to improve the simulation of the PM_{10} concentration and emission rate, main future requests are:

- to optimize the database that is taken to build the model, including the total variation of the respective input variables,
- to perform measurements on input variables like the animal activity at a high time frequency and at a high spatial accuracy.

Conclusions

The average PM_{10} emission rate, measured in stables for sows, weaning and fattening pigs in Italy and Germany, featured a high variation and ranged between 0.7 and 6.0 g d^{-1} LU^{-1}, but was in general in accordance with PM_{10} emission rates reported in literature.

Main influencing variables on the PM_{10} concentration and emission were the ventilation rate, the (group) animal activity, feeding operations, the indoor humidity, the weight of the animals and/or the fattening day, as well as the housing system and the management in the facility. Taking these variables into account, the minutely measured PM_{10} exhaust concentration was explained with a regression coefficient R^2 of 0.60.

Hereby, the ventilation rate featured a negative correlation with the dust concentration in one building, while in another building a positive correlation occurred. We hypothesized that this is due to different airflow patterns in the buildings, as well as due to the length of the respectively considered time period.

The PM_{10} emission rate was modeled using a multiple-input, single-output transfer function model. Based on three input variables – ventilation rate, animal weight, and animal activity – the simulated and measured average emission rate of the respective validation sets differed by about 4 % to 19 %. Thereby, the mean percentage error of the minutely measured and simulated dust emission was in a range of 21 to 39 %.

However, further improvements of the modeling approach are necessary in order to realize an accurate simulation of the PM_{10} emission rate from pig facilities. This includes measurements on the activity level in animal houses at a high time frequency and at a high spatial accuracy.

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Particulate Matter Sampler Errors due to the Interaction of Particle Size and Sampler Performance Characteristics: Method 201a Stack Samplers

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Abstract

Agricultural operations are encountering difficulties complying with current air pollution regulations for particulate matter (PM). These regulations are based on the National Ambient Air Quality Standards (NAAQS), which set maximum concentration limits for ambient air PM. Source sampling for compliance purposes require the use of EPA approved samplers. Ideally, these samplers would produce accurate measures of the pollutant; for instance, PM_{10} samplers would produce accurate measures of PM less than or equal to 10 m (true PM_{10}). However, samplers are not perfect and errors are introduced due to established tolerances for sampler performance characteristics and the interaction of particle size and sampler performance characteristics. This paper focus on the inherent errors associated with the EPA approved PM_{10} stack sampler, sampler used in EPA's Method 201a sampling protocol. Results from this study show that if this sampler is used to sample PM with a mass median diameter (MMD) less than 10 microns then sampler would produce a concentration equal to the true concentration or if the MMD of PM being sampled was less than the cutpoint of the sampler then the sampler would begin to underestimate the true PM_{10} concentration. However, if the PM_{10} stack sampler were used in rural setting and was used to sample dust with a MMD of 20 microns and a geometric standard deviation of 1.5, then the sampler would overestimate the true PM_{10} concentration by 4.8 times. This sampler error is a consequence of the larger size PM associated with rural sources. It results in a substantial difference in the regulation of PM between urban and rural sources. To achieve equal regulation among differing industries, PM_{10} measurements MUST be based on true measurements.

Using Global Model and Satellite Data for Air Quality Studies

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Abstract

This study presents our global model analysis to estimate contributions of pollution, forest and agriculture fires, and long-range transport of aerosols to the surface PM2.5 concentrations in the U.S. and to interpret relationships between satellite observations of column aerosol optical thickness (AOT) and surface measurements of PM2.5 concentrations. Our model study for 2001 indicates that in the eastern half of the U.S., regional anthropogenic emissions, natural aerosols, and long-range transport of anthropogenic aerosols from other regions contribute about 62%, 32%, and 6% to surface PM2.5, respectively, while in the western U.S. the corresponding percentages are 28%, 60%, and 12%. Over 70 to 90% of sulfate aerosols over the U.S. are from the regional/local anthropogenic sources. Analysis of the relationship between the satellite AOT data from MODIS and surface PM2.5 concentrations from the IMPROVE network measurements reveals that the AOT and PM2.5 are much more closely correlated in the eastern than in the western U.S., mainly because of more uniform composition and less variable vertical profiles over the eastern U.S., as the model suggested.

Introduction

PM2.5 (particulate matter, or aerosol particles, with diameter less than 2.5 μm) is a key component determining air quality. At high concentrations, PM2.5 is harmful to human health, reduces visibility, and affects crop yields. Sources of PM2.5 include regional fossil fuel/biofuel combustions from industrial processes and transportation, fire emissions from agriculture activities and forests, and long-range transport from other regions. There are significant socio-economic benefits of accurate PM2.5 air quality forecasts, e.g., issuing health alert, providing guidelines for air quality management planning. Accurate PM2.5 forecasts require continuous extensive spatial and temporal monitoring of the current sates.

Remote sensing capability of atmospheric aerosol optical thickness (AOT) could lead to a quantum leap in our ability of air quality monitoring and prediction, especially for regions where surface monitoring network does not exist. In a recent EPA-NASA-NOAA prototype study, it has shown that the satellite AOT data from the MODIS instrument on the NASA EOS-Terra satellite closely track the surface PM2.5 concentrations over the U.S., providing an opportunity of "monitoring air quality from space" (Al-Saadi et al., 2005). On the other hand, studies have also shown that the correlations between AOT and PM2.5 vary with locations; they are much highly correlated in the eastern U.S. than in the western (Engel-Cox et al., 2004; Al-Saadi et al., 2005), revealing a challenge in quantitative use of satellite data for routine air quality prediction.

In this study, we use the Goddard Chemistry Aerosol Radiation and Transport (GOCART) model to investigate the origin of surface PM2.5 and its relationship to total column AOT. We address the following questions:

- What are the contributions of anthropogenic emission, biomass burning, and long-range transport to the surface PM2.5 over the U.S.?
- How are the AOT and PM2.5 correlated, and how does this relationship change with location and time?
- How can we use the satellite data for PM2.5 studies?

The GOCART Model

The GOCART model is a global scale chemistry and transport model driven by the assimilated meteorological fields from the Goddard Earth Observing System Data Assimilation System (GEOS DAS).

Spatial resolution of the model in this study is at 2° latitude by 2.5° longitude and 30 vertical layers. GOCART simulates major aerosol types of sulfate, dust, black carbon (BC), organic carbon (OC), and sea-salt, and precursor gas species (e.g. SO_2 and dimethyl sulfide). Emissions from anthropogenic, biomass burning, biogenic, and volcanic sources and wind-blown dust and sea-salt are included in the model. Other processes include chemistry, convection, advection, boundary layer mixing, dry and wet deposition, and gravitational settling. Aerosol particle sizes from 0.01 to 10 μm are simulated with parameterized hygroscopic growth which is a function of relative humidity. Details of the GOCART model are described in our publications (Chin et al. 2000a, 2000b, 2002, 2003, 2004; Ginoux et al., 2001, 2004).

Comparisons of PM2.5 from GOCART with Observations

Figure 1 shows GOCART calculated daily aerosol species concentrations at the surface that are compared with measurements at 3 monitoring sites from the Interagency Monitoring Protected Visual Environments (IMPROVE) network for year 2001. Here, PM2.5 from the model is defined as the collection of sulfate, BC, OC, and dust and sea-salt particles with diameter less than 2.5 μm. Influence of long-range transport of sulfate and dust in spring is evident from both model and observations, and the peaks in August in the western high latitude site (Gates of the Mountains) are from biomass burning. Overall comparison of monthly averaged PM2.5 values for 2001 at 168 IMPROVE sites are shown in Figure 2.

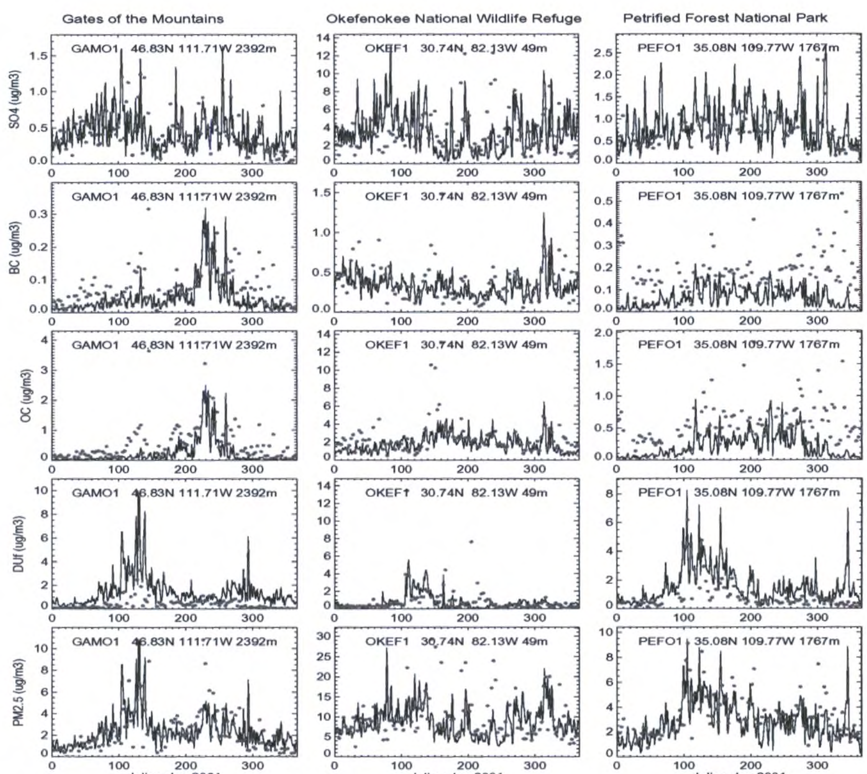

Figure 1. Daily average concentrations of sulfate (SO4), BC, OC, fine mode dust (DUf), and PM2.5 at 3 IMPROVE sites in 2001. Circles are IMPROVE data and lines are model results.

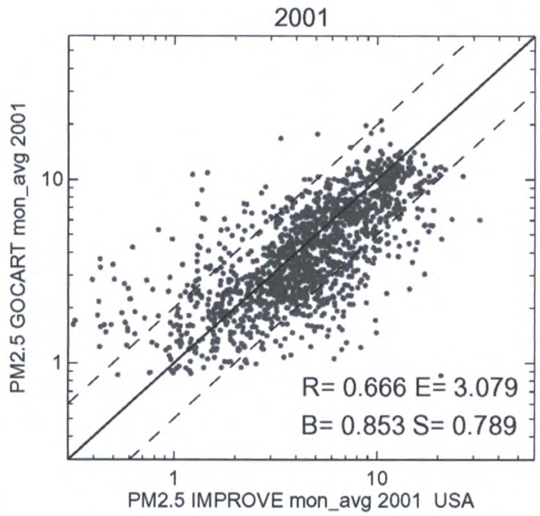

Figure 2. Comparison of monthly average PM2.5 concentrations in 2001 at 168 IMPROVE sites. R = correlation coefficient, B = mean bias, E = Root mean square error, S = skill score (see Chin et al. 2004 for definition).

Contribution of aerosols from different sources

In order to assess the contributions of aerosols over the U.S. from different source regions, we have run several model experiments in which a particular source or emission from a particular region is "zeroed out". Table 1 lists the model experiments.

Table 1. GOCART model experiments.

Exp.	Definition	Model setup
ALL	PM2.5 from all sources, including anthropogenic, biomass burning, and natural emissions	Standard model run
BKG	Background PM2.5 – from sources we cannot control, e.g. dust, volcanoes, trees, Long-range transport from other regions	Model run without North America anthropogenic emission (with biomass burning)
NAT	PM2.5 from natural sources only	Model run without all anthropogenic emission (with biomass burning)
BMB	PM2.5 from biomass burning	Differences between standard run and runs without biomass burning emissions
NAM ASA EUR	Sulfate aerosols from North America (NAM), Asia (ASA), or Europe (EUR) anthropogenic emissions	Differences between standard run with all sulfur sources and the run without anthropogenic emissions in a selected region
AFR ASA MDE	Dust aerosol from Africa (AFR), Asia (ASA), or Middle East (MDE)	Differences between standard run with all dust sources and the run without emissions in a selected region

Figure 3 demonstrates the model results of total PM2.5 and PM2.5 from background and natural sources for the four quadrants (NW, SW, NE, SE, divided at 100°W and 40°N) over the U.S. The model shows that in the western half of the U.S., about 70% PM2.5 is "background" with nearly 60% "natural" (e.g., dust, biogenic aerosols) on average. In the eastern half, local anthropogenic sources contribute to 60% of PM2.5 concentrations. Further analysis shows that the U.S. anthropogenic sources contributes to 70 – 90% of sulfate PM2.5 at the surface while dust from Asia could supply about half fine dust loading except in the SW quadrant where local deserts are the major source of dust (64%). The contribution of forest and agriculture fires varies significantly with location and seasons.

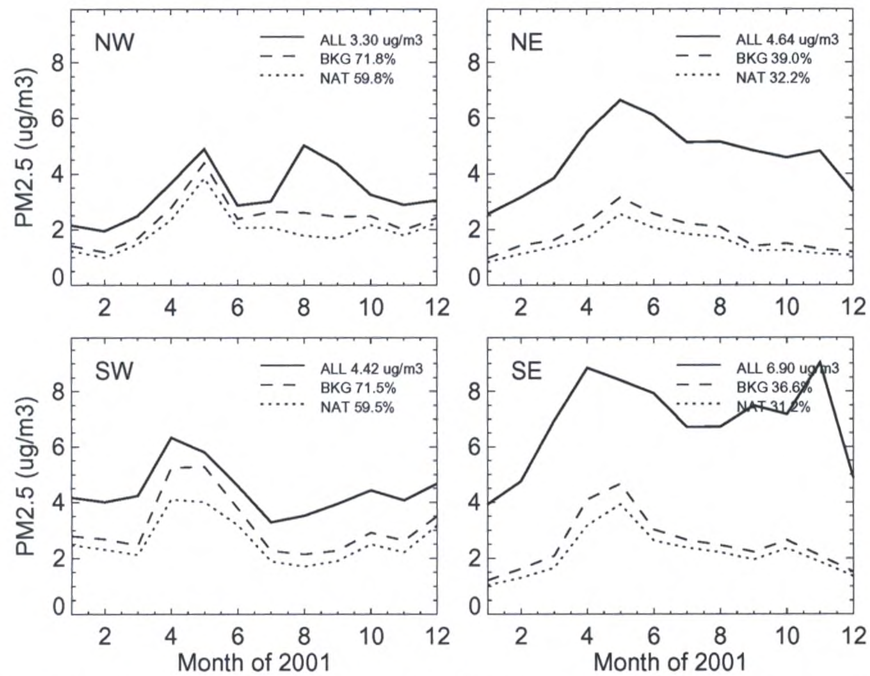

Figure 3. Monthly average surface layer PM2.5 concentrations in 2001 from the GOCART model (see Table 1 for experiment definition). Annual average concentrations and % contributions from different sources are shown. The difference between BKG and NAT is PM2.5 anthropogenic transport form other regions, and the difference between ALL and BKG is PM2.5 from North America own pollution sources (and biomass burning).

Relationship between AOT and PM2.5

As we mentioned in the "Introduction", it is important to understand the relationship between the column AOT and the surface PM2.5 in order to intelligently use the satellite data for air quality monitoring. A robust relationship between the column AOT and surface PM2.5 can be achieved only if a) the aerosol vertical profile maintains a relatively constant shape such that the change in total column AOT is nearly proportional to the change at the surface, and b) the aerosol composition stays the same such that the mass extinction efficiency, which converts aerosol mass to AOT depending on aerosol type, size, and relative humidity, remains more or less constant. Unfortunately, there is usually a significant heterogeneity in aerosol distributions and composition that make the AOT-PM2.5 relationship vary with location and season, as shown in a recent analysis using the AOT from the satellite data from the MODIS instrument and the surface PM2.5 measurements from more than 1000 sites over the U.S. (Engle-Cox et al., 2004). The concept is further illustrated in Figure 4 with the GOCART model results, which shows that the relationship between the AOT and surface PM2.5 varies from poor to excellent depending on location and time. In general, the correlation is better in summer/fall than in winter/spring and better over the eastern half of North America than over the western half.

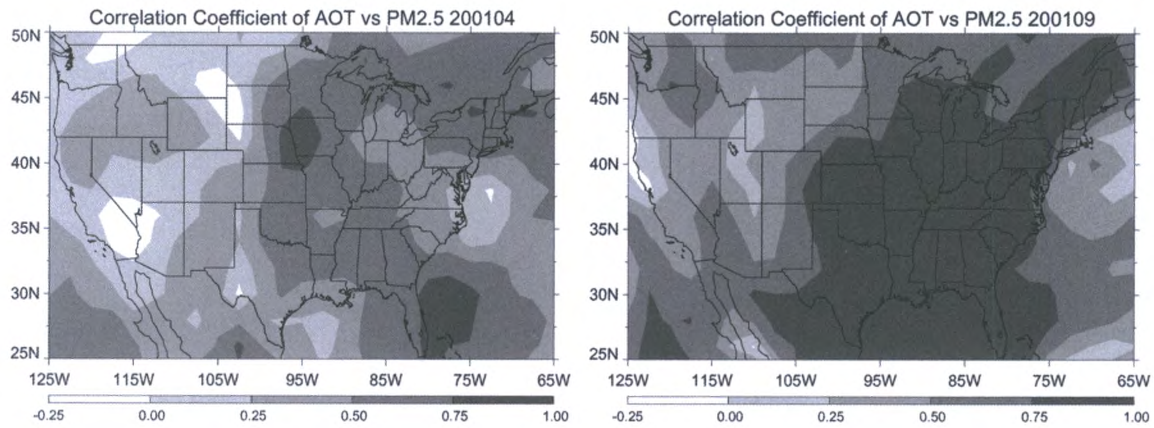

Figure 4. Correlation coefficients of daily AOT and PM2.5 for April (left panel) and September (right panel) 2001 from the GOCART model.

We further explain the cause of these different relationships by examining the PM2.5 composition and the aerosol vertical distributions at 2 sites, one located in the eastern U.S. (Ohio) and one in the west coast (Washington state). The spatial and temporal difference in the AOT-PM2.5 relationships can be explained by the differences in aerosol composition and vertical distributions. At the Ohio site, sulfate is the dominant aerosol type for both April and September and located mainly in the boundary layer. In contrast, aerosol compositions and vertical distributions are very different between April and September at the Washington State site located at the west coast. In spring, the site receives not only large amount of dust transported from Asia with plume extending to 5 km but also sea-salt aerosol from the Pacific. In late summer/early fall, on the other hand, the site is heavily influenced by biomass burning with carbonaceous aerosol dominating. Therefore, it is more difficult to quantitatively relate column AOT with surface PM2.5 in the western than in the eastern U.S. because of the larger variation of aerosol composition and vertical profiles in the west.

Conclusions

We have used the global model GOCART to investigate the relative contributions of regional anthropgenic emissions, forests and agriculture fires, and long-range transport of aerosols from other regions in the world to the surface PM2.5 concentrations at the U.S. that affect the air quality. The model study for 2001 indicates that in the eastern half of the U.S., regional anthropogenic emissions, natural aerosols, and long-range transport of anthropogenic aerosols from other regions contribute about 62%, 32%, and 6% to surface PM2.5, respectively, while in the western U.S. the percentages are 28%, 60%, and 12%. About 70 to 90% of sulfate aerosols over the U.S. are from the regional/local anthropogenic sources.

Satellite AOD data provide very helpful guidance for PM2.5 forecasts and monitoring. At the time and places when the aerosol composition and vertical distributions are stable, the column AOD can be quantitatively scaled to surface PM2.5 concentrations, such as in the eastern U.S., especially during warm months (summer/fall) that aerosols are predominantly of regional/local pollution origin and concentrated mostly within the boundary layer. However, in the western U.S. where aerosols are from multiple origins (local pollution, desert, long-range transport from Asia, biomass burning) direct link between AOD and PM2.5 is more difficult because the composition and vertical profiles have large seasonal variations.

To obtain a high quality PM2.5 prediction and assessment capability, a combination of large-scale model, satellite AOD data, and vertical profile information from either local lidar network or from future satellites (e.g. CALIPSO) is needed to quantitatively estimate the effects of long-range transport and separate boundary layer aerosols with aloft plumes.

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Public Policy and Agricultural Air Quality

History of Conflicts Between Swine Farmers and Neighbors in North Carolina: The Response of the Law to Conflict

By Ryke Longest[1]

Abstract

Conflicts between swine farmers and their neighbors have been documented in North Carolina since the early 1840s. Early disputes were caused by livestock ranging through city streets. Later 19th century disputes pitted planters against stock owners over whose duty it was to protect crops from livestock. In the early 20th century, conflicts were usually framed as zoning issues, odor and nuisance complaints. By the end of the 20th century, swine raising practices had undergone a revolution and a new set of environmental concerns came to the foreground. These recent conflicts were framed as environmental issues and challenges to agricultural exemptions from environmental and other laws. North Carolina's three branches of government were all engaged to solve these conflicts through the 1990's. Peace, but not resolution of these conflicts, was achieved through enactment of more stringent environmental laws and a moratorium on lagoon construction for swine farms. The moratorium slowed further development until technological solutions could be found to the problems identified.

Swine in North Carolina's History

During several points in American history, pork production has produced conflicts in land use and these conflicts have influenced the development of law. Prior to the introduction of swine by Spanish explorers, there were no swine in North America. Early colonial settlers in the Southern colonies of Virginia and the Carolinas kept hogs primarily on open range in swamps and forested land. This exposed the swine to predation from wolves and foxes when young and bears when older.[2] Pork was fattened by allowing them to forage on tree mast, tuberous roots and green canes found in the forests and swamps.[3]

Writing in the late 1700s, Johann David Shoepf[4] observed of North Carolina's swine raising practices:

"Their hogs likewise range throughout the year in the woods. Towards the coast in the pine forests, the cones of the pitch-pine, larger than those of the other sorts, are their favorite food; also they root up the young sprouts of these pines and eat off the bark, for which reason the pitch-pine does not spring up so readily where it has once been taken off. Farther up the country the hogs find better mast beneath the numerous oaks, chestnuts, beech-trees, and chinquapins. In winter the sows make themselves beds of pine-twigs where they litter; the owner seeks them out, brings them in nearer the house, gives them a better bed of straw, and marks the pigs. Later, to accustom them to the plantation, they are called up several times a day and fed on corn-stalks. In the autumn, after the maize-harvest, a number of hogs are brought in from the woods and placed on feed. A bushel of corn a week is allowed each head, for 5-6 weeks. The amount of corn made determines the number of hogs to be fed. Fattened hogs reach 3 to 500 pounds' weight. Live hogs sell at 3-3½ Spanish dollars the hundred. Nowhere on the whole continent is the breeding of swine so considerable or so profitable as in North Carolina. Besides what is consumed in the country, salted, exported, and lost in the woods, there are annually 10-12000 head driven to South Carolina or to Virginia.

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[1] This is an educational manuscript only and expresses the views of the author. It has neither been reviewed nor approved in accordance with the policy for issuing Attorney General's Opinions.

[2] Records of the Moravians in North Carolina, I, 131-140. (1755), available on Internet, at: http://www.ah.dcr.state.nc.us/sections/hp/colonial/Bookshelf/Moravian/extracts.htm (February 15, 2006)

[3] Byrd, William and Ruffin, Edmund, "The Westover Manuscripts: Containing the History of the Dividing Line Betwixt Virginia and North Carolina; A Journey to the Land of Eden, A. D. 1733; and A Progress to the Mines." Written from 1728 to 1736, available on Internet, at: http://docsouth.unc.edu/nc/byrd/byrd.html. (Feb. 15, 2006)

[4] Schoepf, Johann David , "Travels in the Confederation [1783-1784]" (Translated from the German and Edited by Alfred J. Morrison, Philadelphia, William J. Campbell, 1911), available on Internet, at: http://www.ah.dcr.state.nc.us/sections/hp/colonial/Bookshelf/Travels/Default.htm (February 15, 2006)

The North Carolinians therefore should not look a-skance, if their neighbors rally them for being pork-makers, for when the talk gets on their swine-breeding they themselves use the expression, 'We make pork.' But in these circumstances, a hog costing them next to nothing except for what goes into the fattening, the North Carolinians can send their salted hog-meat to market at a third or a half cheaper than their neighbors in the northern states where harder winters and more restricted pasturage make the maintenance dearer."

Colonists of the eighteenth century were mostly subsistence farmers, who relied on hominy, cornbread, and pork for staple foods.[5] Ninety-five percent of North Carolina's colonists were engaged in agriculture or related industries.[6] Since the colonists lacked paper money, gold and silver, commodities were the chief means of commercial exchange. Sixteen commodities were rated as money within the North Carolina colony in 1715, including pork.[7] About one-eighth of the pork and beef shipments from the English continental colonies were from North Carolina.[8] These statistics likely understate North Carolina's contribution to colonial swine production, since swine were frequently raised in North Carolina and driven north to Virginia or South Carolina for sale, as these colonies had the advantage in ports and commerce. Conflicts between livestock owners and others have been occupying our courts and legislative assemblies ever since.

Towns began to pass ordinances restricting the free roaming of livestock inside their boundaries. Beaufort's ordinance prohibiting swine running loose in the town was challenged by two swine owners, one of whom sued on the basis that the ordinance could not be enforced against him because he was not a town resident.[9] In ruling that the ordinance did apply, the North Carolina Supreme Court held that the farmer was liable under the ordinance, but noted that he had done "as every other farmer does, turned out his stock to range upon the unenclosed land around him."[10] Absent a valid town ordinance such as Beaufort's, such practice was legal in North Carolina.

Indeed, the University of North Carolina's neighboring village, Chapel Hill, was not immune from this trouble. "A serious trouble to pedestrians arose from the presence of numerous bovines and hogs on the streets. There was so little traffic that there was an abundance of good pasturage in the village and every family kept at least one cow, and many raised their own pork. Ladies and gentlemen were often compelled to drive animals from the sidewalks in order to pass. The more timid sometimes yielded precedence to the intruders and made a wide circuit to avoid them."[11]

Hogs in town were not a North Carolina oddity. New York's Central Park was established in the mid nineteenth century on land that had also housed hog lots.[12] In North Carolina, the highest swine populations were present in more densely settled areas in the nineteenth century. Duplin County was behind eight other North Carolina Counties in pork production in 1869.[13] Higher populations of swine were present in the

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[5] Lefler, Hugh Talmadge & Newsome, Albert Ray, "The History of a Southern State: North Carolina"124 (3d ed. 1973)
[6] *Ibid.* at 89.
[7] *Ibid.* at 157–58.
[8] *Id.*
[9] *See* Whitfield v. Longest, 28 N.C. 268 (1846)(Please note that this is the first of a number of citations to North Carolina appellate cases which are reported in a series of bound volumes, where the first number indicates the volume, the "N.C." in the citation abbreviation refers to the official court reporter and the third number is the page number upon which the case begins.)
[10] *Ibid.* at 273.
[11] Battle, Kemp P. "History of the University of North Carolina:Volume I: From its Beginning to the Death of President Swain, 1789-1868" (Kemp Plummer 1919).
[12] Taylor, Dorceta E., Central Park as a Model for Social Control: Urban Parks, Social Class and Leisure Behavior in Nineteenth-Century America; Statistical Data Included, *National Recreation and Park Association Journal of Leisure Research* (September 22, 1999).
[13] North Carolina Land Co., "A Guide to Capitalists and Emigrants: Being a Statistical and Descriptive Account of the Several Counties of the State of North Carolina, United States of America; Together with Letters of Prominent Citizens of the State in Relation to the Soil, Climate, Productions, Minerals, &C., and an Account of the Swamp Lands of the State" (From the N.C. Land Company, 1869)

county of Wake where the state capital was located as well as the important industrial and commercial center of Guilford County than in Duplin County.

English common law placed responsibility on livestock owners to keep them under control and keep them away from mischief. This rule was abrogated by statute and custom early in history of North Carolina, putting it in a free range state. In 1860, the N.C. Supreme Court held that a corn and pea crop farmer whose crop was damaged by horses and cattle of his neighbor across the Yadkin River had no claim. While the Court held that the cattle were trespassing, it further held that the offended crop owner had no recourse but to erect a lawful fence. The Court observed, "Thus the keeping under inclosure domestic animals, which is regarded as the rule of the common law of England, if it were ever recognised in our waste and thinly populated country, has been long since abrogated by various legislative acts and by constant usage to the contrary."[14]

This abrogation by the legislature of the common law rule and its acceptance by the courts arose from the recognition that England was densely settled, while North Carolina was thinly settled. As populations of people grew and land deemed waste became valuable property, the basis for the rule of law became questionable. Because livestock interests were accustomed to the older practice, they demanded free range as of right. Increasingly, row crop farmers and town residents attacked these laws and these struggles resulted in considerable legislative fine tuning.

Livestock grazing on unfenced land brought with it conflicts between row crop farmers and livestock growers throughout the state. During the end of the nineteenth century, a series of livestock fencing requirements — referred to as "fence law" or "stock law" — were passed, covering larger and larger portions of North Carolina, county by county or even swamp by swamp.[15]

The law cited in the Burgwyn case shows the extreme precision with which the legislature endeavored to carve out stock law territory. The Burgwyn Court cites the following pertinent provision within the statute: "the Legislature, at the session of 1876, Chap. 60, passed a private law for their benefit, wherein it was enacted that the river, a rail fence running from Faison's corner on the river to Mud Castle, and thence to Wheeler's Swamp, at the head of Bull Hill mill-pond, and the run of said swamp from the head of said mill-pond to the river, should be sufficient as a fence, with a declaration in the sixth section of the act that the act should not apply to stock kept north of the rail fence constituting part of the boundary, unless the fence was kept in good and lawful condition, nor to stock kept east of Wheeler's Swamp, provided a gate was kept." Such description reads more like a zoning ordinance than a piece of legislation.

As more legislators sought to protect their citizens from wandering livestock, the stock law was gradually extended to cover the entire state. Towns and counties were given the option to add the coverage of the Stock Law to their borders; where the law existed, it was a misdemeanor for the owner of livestock to allow them to run at large.[16] Cases involving the application of the Stock Law demonstrate that hogs and other livestock were getting loose and causing conflicts with row crop farmers and town residents for many years.[17] North Carolina's Stock Law produced a number of reported appellate decisions over a single animal or its monetary value. These decisions show a steady development away from English common law principles, and a form of statewide zoning engaged in directly by the legislature.

All sides agreed that livestock at large were capable of damaging crops. The row crop farmers disagreed with the livestock owners about who should have to pay for the fence construction and maintenance. Initially the law aided the livestock owners in this regard. Trespassing livestock who were injured or killed were protected by law. A hog farmer obtained compensation from a railroad company because spilled molasses on the railroad tracks lured his trespassing swine there to feed wherein they were run over.[18] A

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[14] Jones v. Witherspoon, 52 N.C. 555 (1860).
[15] See Burgwyn v. Whitfield, 81 N.C. 261 (1879)
[16] N.C. Gen. Stat. § 68-16 (2003).
[17] *See* State v. Tweedy, 115 N.C. 704 (1894) (indictment of town resident for shooting hog running at large in town overturned because indictment did not charge that hog was running loose unlawfully); Broadfoot v. Fayetteville, 121 N.C. 418 (1897) (challenges to ordinance and stock law under state and federal constitutions rejected by court) ; Bowen v. Town of Williamston, 171 N.C. 57 (1916).
[18] Page v. N.C. Railroad Company, 71 N.C. 222 (1874).

hog trespassing on a chicken farmer's land was protected from harm, despite having previously killed a chicken.[19] A trespassing boar which had rooted up a lawful fence several times, eluded pursuit by men and dogs and crashed down the fence altogether could not be shot by the landowner, because the boar was a public service to the community at large and was protected by statute.[20]

But the tide turned around the turn of the century as more and more people sought changes to the stock law. By 1896, it was a misdemeanor to allow stock to run at large in Wake County.[21] By 1908, mountainous counties such as McDowell were under the stock law as well.[22] The North Carolina Supreme Court recognized in the Mathis case that conditions had changed and that the law had changed with it. Justice Connor wrote:

"If the condition, in respect to the agricultural system of the people so changes as to make it conducive to their interest to require all stock to be "fenced in" and relieve the land owner of the duty to "fence it out," we can see no good reason why the Legislature may not by appropriate legislation do so, either in respect to the whole State or political divisions thereof. For the past twenty-five years, such has been the policy of the State, as evidenced by our legislation. This being true, we do not see why the Legislature, or when power is conferred upon them, the county commissioners, may not forbid stock running at large in the county, or any township thereof, and declare a mountain range, a creek or other natural political boundary a lawful fence, or the limit within which the law shall operate."[23]

By 1913, the stock law was in force over nine-tenths of the territory of the state.[24] The territories had gradually spread under the general principal that every man has a right to use his own provided he does not do so to the injury of the rights of others. In 1918, the Chief Justice observed the stock law situation and its costs as follows:

"Besides these and other arguments which have caused the extension of the "no-fence law," the "Commission for the Conservation of Food" have recently called attention to the fact that in this State last year $ 60,000 worth of stock were killed by railroad locomotives, a very small per cent of which loss occurred in the no-fence law counties, but almost entirely in that small part of the State in which the free range still obtains. At the same ratio, if stock had been allowed to run at large throughout the State, the destruction of stock and the loss of food thereby would amount annually to far over a half million dollars, for less than one-tenth of the State is now outside of the stock-law territory.

Our Legislature, in deference to the wishes of the people of any locality, have given them opportunity to declare whether they shall adopt the policy of each man fencing up his stock or of every man fencing out the stock of others. The result has been the growth of the stock law in North Carolina, until now it prevails over nine-tenths of the State; in fact, in all the State except in parts of half a dozen townships in the mountain sections where the cultivated fields are a negligible quantity and in a few counties along the Atlantic Coast, in most of which there are large areas of land not yet under cultivation, though even in this fringe of counties there are considerable areas in which the stock law prevails."[25]

Two of the last counties to be covered were Duplin and Pender County.[26] Conflicts in those counties arose over how a fence would be paid for in order to keep the open range throughout these counties. By the 1940's the entire state was under the stock law, and owners of livestock were under duty to keep them fenced upon penalty of law and liability.[27] Within a span of one hundred years, the complete reversal of the open range rule had occurred. From there on out, owners of stock were legally bound to keep them confined. This they did, with increasing density.

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[19] Morse v. Nixon, 51 N.C. 293 (1859)
[20] Bost v. Mingues, 64 N.C. 44 (1870)
[21] State v. Hunter, 118 N.C. 1196 (1896).
[22] State v. Mathis, 149 N.C. 546 (1908).
[23] *Ibid.* at 548.
[24] Marshburn v. Jones, 176 N.C. 516; 517 (1918).
[25] Marshburn v. Jones, 176 N.C. 516; 522 (1918).
[26] *See* Keith v. Lockhart, 171 N.C. 451 (1916), Marshburn v. Jones, 176 N.C. 516 (1918), *and* Faison v. Commissioners of Duplin, 171 N.C. 411 (1916).
[27] McKoy v. Tillman, 224 N.C. 201 (1944).

North Carolina's Swine Population Boom

The legal conflicts were plentiful even as the amount of livestock raised in North Carolina was declining. North Carolina was always a top tobacco producing state but gradually became an insignificant producer of livestock. In 1920, the value of all livestock in North Carolina per farm was $442, one dollar above Alabama as the lowest ranking state.[28] In 1925, North Carolina's tobacco crop was worth $87,438,000, ranking North Carolina second in tobacco production to Kentucky.[29] The agricultural landscape changed dramatically in the latter half of the twentieth century. In 1940, $5,747,918 of North Carolina's $328,695,232 in farm income came from hogs;[30] in 1969, $118,614,000 of North Carolina's $1,406,161,000 in farm income came from hogs,[31] an increase of from two to more than eight percent of total farm income in less than thirty years. The biggest jump was yet to come as new swine production techniques were developed and introduced. By 1993, North Carolina had become the third largest pork producing state; instead of allowing their pigs to forage or feeding them garbage, swine producers copied the success of poultry producers and learned to efficiently produce pigs by formulating diets and constructing confinement structures.[32]

Modern hog production depends upon three carefully controlled factors: genetic selection, feed formulation, and climate control. A geneticist selects which strain will go on each farm. Nutritionists formulate feed with vitamin supplementation to control costs and to meet specific nutritional needs of the genetic strain being raised. Automatic feeders provide pigs with a steady stream of the specially formulated meal. Hogs are confined inside a building with electrical light and heat and cooling systems. Workers remove their clothes, take a shower, and put on coveralls before entering the hog house to prevent the transmission of diseases. Hogs are given water from wells specially dug for their house. Under these conditions, swine grow quickly to market weight and once they do so, they are shipped out and the house is cleaned to be ready for the next group or "turn."

Floors inside hog houses are slatted, some partially and some fully. The slats allow manure and urine to fall through to pits beneath the houses. It is as if the entire house sat over a giant toilet bowl. There are four different types of systems for this toilet bowl to use for removing the waste from the house: deep pit, pull plug, pit recharge, and flush. Deep pits are not used much in North Carolina, but the other three types are common. In North Carolina, once these systems are activated, wastewater is sent from under the house to a lagoon.

A lagoon is an open air primary waste treatment structure. Most lagoons in North Carolina are designed for anaerobic operation, meaning that the bottom portion of the lagoon has extremely low dissolved oxygen. Anaerobic bacteria, which do not tolerate oxygen, thrive in this environment and work to break down the manure that sinks to the bottom of the lagoon. This breakdown produces gases and sludge, which accumulate in the lagoon until released or removed. Gases leave by complicated biological and chemical processes linked to life cycles of the bacteria, manure loading, and the pH of the wastewater. Sludge stays and accumulates until removed mechanically. The Natural Resources Conservation Service of the U.S. Department of Agriculture recommends that the accumulated sludge be removed every five years if it has encroached upon the treatment volume of the lagoon.[33]

Lagoons vary in design parameters such as shape, depth, and liner material. These differences are primarily correlated with lagoon age in North Carolina as lagoon designs have been modified over the years. A few farms have more than one stage in their lagoons to provide further storage and treatment. All farms need to have sufficient volume to treat the wastewater loaded into them and to store that wastewater until it can be disposed of through land application. An average sized pig of 135 pounds produces 1.37 gallons per day of urine and feces, compared to an 800 pound beef cow's production of 5.53 gallons per day. In addition to

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[28] Lefler, Hugh Talmadge & Newsome, Albert Ray, "The History of a Southern State: North Carolina" 578 (3d ed. 1973).
[29] *Id.*
[30] *Ibid.* at 647.
[31] *Id.*
[32] *Id.*
[33] Natural Resources Conservation Service Practice Standard for Waste Treatment Lagoon, CPS 359, Rev. 4, p. 15 (NRCS-NC, January 1998)

this waste volume, lagoons must also hold the fresh water added to the housing for cooling and the rainfall which flows into the lagoon. Lagoons in North Carolina are now supposed to be designed to accommodate heavy rainfall and a twenty-five year twenty-four hour storm event without encroaching into a twelve inch zone called the structural freeboard.[34]

Some wastewater from lagoons is recycled to serve as the flush water for the houses. Excess wastewater from the lagoon is applied to cropland primarily by spray irrigation for disposal. This disposal is limited by two primary factors: the nutrient requirements of crops and the ability of the soil to accept the hydrologic load. If too much wastewater is applied, it may pond up and run off the fields faster than the soil can absorb it. If too much waste is applied, the roots of the crops cannot absorb the nutrients and they may be lost to groundwater, the soil, or the atmosphere. Sprayfield crops generally include grasses for grazing and grains, which can be marketed or used by the producers. North Carolina's sprayfields do not currently produce enough crops to feed the number of livestock raised. North Carolina is still a net importer of grain for feed.

The North Carolina experience with growth was in number of swine, not in the number of swine farms. This was a national trend in pork production, but more pronounced in North Carolina.[35] Nationwide, the number of farms with swine fell from 317,087 to 103,965 between 1982 and 1997.[36] During the same period, the number of swine produced rose from 7,730,637 to 8,522,082 nationwide for a net increase nationwide of 1,191,445 measured as animal units (AU). North Carolina's growth accounted for 1,160,152 AU of that increase, or more than ninety-five percent of the net national increase. During the same period, the number of North Carolina swine farms decreased from 8,691 to 2,673.[37]

In 2002, hogs were still the number one cash receipts farm product in North Carolina, as they had been since the mid 1990s.[38] Out of the top seven crops in cash receipts for 2002, only two row crops are listed: greenhouse/nursery production was third and tobacco was fourth. Hogs almost accounted for more receipts than tobacco and greenhouse nursery combined. Within a single decade, pork had eclipsed tobacco in North Carolina's agricultural economy.

North Carolina's pork production is now significant to the economy of the nation. On December 1, 2004, North Carolina was estimated to have 9.8 million hogs out of the whole U.S. herd of 60.5 million.[39] The total number of hogs owned by operations with over 5,000 head total inventory, but reared under production contracts, accounted for thirty-eight percent of the total U.S. hog inventory, up three percent over the previous year.[40] The top ten swine producing counties in North Carolina have combined inventories of more than seven million hogs.[41] North Carolina's top ten counties account for more than eighty percent of the state's swine inventory and more than ten percent of the national swine inventory.

Many of these animals are sent for slaughter to one of the two large slaughterhouses in North Carolina. They are often also sent to other grain-rich states to be fattened prior to slaughtering.[42] In 2001, 3.8 million

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[34] *Id.*
[35] Economic Research Service, U.S. Department of Agriculture, information available on the Internet at: http://www.ers.usda.gov/data/Manure/spreadsheets/prk82.xls and http://www.ers.usda.gov/data/Manure/spreadsheets/prk97.xls (February 15, 2006)
[36] *Id.*
[37] *Id.*
[38] Agricultural Statistics Division, N.C. Department of Agriculture, Cash Receipts, information available on Internet, at http://www.ncagr.com/stats/cashrcpt/commrank.htm (Feb. 6, 2004).
[39] Agricultural Statistics Division, N.C. Department of Agriculture, Livestock, information available on Internet, at http://www.ncagr.com/stats/livestoc/anihgi12.htm (May 10, 2005) Units are hogs, not animal units.
[40] *Id.*
[41] Agricultural Statistics Division, N.C. Department of Agriculture, Livestock, information available on Internet, at http://www.ncagr.com/stats/cnty_est/ctyhogtt.htm (May 10, 2005).
[42] *Id.*

hogs left North Carolina to be finished in other states.[43] In contrast, North Carolina only had 158,000 hogs shipped into this state from other states. It is therefore true that North Carolina weans far more pigs than it slaughters.

North Carolina hog slaughter is also nationally significant. In the 1950s, Burrows Lundy moved from Pennsylvania to Clinton, North Carolina, to open the Lundy Packing Company. With the help of Lew Fetterman, Mr. Lundy moved up processing capacity from 1,000 hogs per week in the 1950s to 8,000 hogs per day in the 1980s. North Carolina is now home to the largest hog slaughterhouse in the United States (and perhaps the world) with a capacity of 32,000 hogs per day. This plant is owned by Smithfield Packing Company and is located in Bladen County near the town of Tar Heel. Smithfield Packing's parent company, Smithfield Foods, is the largest pork processor in the world.[44] Nevertheless, the star jewel in its processing crown is the Bladen County plant. The Lundy plant is also still in operation and was significantly updated after Lundy Packing was acquired by Premium Standard Farms of Missouri, a national leader in vertical integration.

Federal Water Pollution Control Act

While North Carolina was increasing its ranking in animal agriculture, the federal government began to regulate operations where large numbers of animals are confined. The Federal Water Pollution Control Act Amendments of 1972 (CWA) began a planning process for states to deal with water pollution from manure, called areawide waste treatment management plans, often referred to as the Section 208 process.[45] A few years later, this was amended to add an incentive program for agricultural polluters that used a cost sharing arrangement.[46] The Rural Clean Water Program offered financial incentives to landowners to implement best management practices (BMPs) to control nonpoint source pollution. This program was later expanded and funded under future farm bills to an alphabet soup of conservation incentive programs administered by the U.S. Department of Agriculture.[47]

Congress also took action to require states to report on their progress in water quality improvement on a watershed by watershed basis.[48] These requirements reinforced the notion that the primary role for developing plans for controlling pollution lay with states. At the same time, the states worked to implement these plans with a variety of legal mechanisms. Each state is required to "identify those waters within its boundaries for which the effluent limitations required …are not stringent enough to implement any water quality standard."[49] These waters are usually those which receive pollution from sources not regulated as point sources under the CWA. Potential sources of these pollution problems are numerous, but include runoffs and discharges from agricultural and livestock sources.

North Carolina's Incentives and Nonpoint Source Regulations

In the 1980s, North Carolina had adopted its own program to offer financial incentives to agricultural operations to undertake conservation measures. The primary program created was called the Agricultural Cost Share Program for Nonpoint Source Pollution Control. This program was created in 1986 and provides for the supervision of the program by the North Carolina Soil and Water Conservation

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[43] Shields, Dennis A. and Matthews, Kenneth H. Jr., *Interstate Livestock Movements*, USDA Economic Research Service, available on Internet, at: http://www.ers.usda.gov/publications/ldp/jun03/ldpm10801/ldpm10801.pdf .

[44] Annual Report, Smithfield Foods, Fiscal Year 2004.

[45] 33 U.S.C.A. § 1288 (2005).

[46] 33 U.S.C.A. § 1288(i), (j) (2005).

[47] The Farm Security and Rural Investment Act of 2002 authorized federal funding for the following cost share conservation programs: Agriculture Management Assistance (AMA), Conservation of Private Grazing Land (CPGL), Conservation Reserve Program (CRP), Conservation Security Program (CSP), Environmental Quality Incentives Program (EQIP), Farmland Protection Program (FPP), Grasslands Reserve Program (GRP), Wetlands Reserve Program (WRP) and Wildlife Habitat Incentives Program (WHIP). U.S. Pub. L. No. 107–171 (May 13, 2002).

[48] 33 U.S.C.A. § 1329 (2005).

[49] 33 U.S.C.A § 1313(d)(1)(a)(2005).

Commission.[50] The program is used to fund a wide variety of conservation practices, including animal waste management systems.[51] Regulation of these sources beyond incentives has proven quite difficult scientifically, logistically, and politically. Nowhere has that proved more true than in North Carolina's Neuse River.

Environmental conditions in the Neuse River are driven by complex interactions between salinity, rainfall, wind, atmospheric deposition, shallow groundwater flows, water temperatures, biology, and chemistry. The Neuse is a relatively shallow river that drains into the Albemarle-Pamlico Sound complex, the second largest estuarine system in the United States, second only to the Chesapeake Bay. The Sound's 30,000 square miles of watershed are significant habitat for a variety of birds, reptiles, turtles, fish, and shellfish. Situated at the northern range of southern species and the southern range of many northern species, it is the home for both alligators and tundra swans. Sea turtles nest on the beaches and swim in the inlets of the sound each summer and diving ducks feed in the sound each winter

While pollution and fish kills in the Neuse have been problems for many decades, recent kills have become more worrisome to state planners. Even as discharges to the Neuse from point sources have been increasingly restricted, excess reactive nitrogen in the river has led to nuisance algae blooms. Researchers spent increasing time and energy focusing on the causes of these problems in the 1980s and 1990s.

In the early 1990s, North Carolina controlled animal waste management systems through state level administrative rules on nondischarge waste treatment.[52] These rules were referred to as the 0.200 rules due to their regulatory citation number. The 0.200 rules required farms with more than 250 swine to obtain certified animal waste management plans.[53] These rules also provided that compliant nondischarge facilities would be considered "deemed permitted" and would not have to obtain individual permits unless they broke the rules.

Meanwhile, one of North Carolina's largest newspapers, the Raleigh News and Observer, ran a series of articles on the growth of the swine industry and the state of its regulation. These articles ran February 19, 21, 22, 24 and 26 of 1995.[54] This series of articles was referred to as the "Boss Hog" series and won the reporters a Pulitzer Prize in 1996 for public service. The Boss Hog articles were generally critical of the state's hog regulation. The articles contended that enforcement of environmental rules against hog farms was too lax and that the 0.0200 rules were not protective enough by themselves. Many swine farmers contended that the articles were unfair. Legislative action to expedite implementation of new rules on hog farms was killed in April of 1995 in favor of a study commission.

Oceanview Farms Case

A couple of months after legislative actions were killed, a serious new conflict arose. On June 21, 1995, the eight-acre manure lagoon at Oceanview Farms in Onslow County burst its dike, sending a tide of wastewater across neighboring roads, fields, and streams and into the New River near Jacksonville, North Carolina.

While the Oceanview facility had a certified animal waste management plan, it had not followed that plan. The farm had not been applying the waste to land as required and the lagoon was filled far beyond its specified capacity, which required that there be a minimum of one foot of freeboard to protect the physical structure from breach.[55] Larval casings for insects were found within a few inches of the dike crest, indicating that the lagoon had reached the top of the dike before rupturing. Additionally, investigators found that less than half of the land required for land application had been cleared by Oceanview. Also, the

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[50] N.C. Gen. Stat. § 143-215.74(a) (2005).
[51] *Ibid.* at (b)(5).
[52] 15A N.C.Admin.Code § 2H 0.0200 *et seq.* (1993).
[53] *Ibid.*at § 2H 0.0217.
[54] *News and Observer*. The articles were published on February 19, 21, 22, 24 and 26, 1995. Electronic copies are available online at:
http://www.pulitzer.org/year/1996/public-service/works/
[55] Plaintiff's Preliminary Injunction Brief, pp. 5-11, State of North Carolina ex rel. Michael F. Easley v. Oceanview Farms Limited Partnership, 95 CVS 1993, (Onslow County Superior Court, Sept. 12, 1995).

dike walls had been weakened by installation of piping and pumps in the walls. These factors contributed to the lagoon's massive rupture, which spilled about twenty-five million gallons of wastewater.

The Oceanview Farms case resulted in an injunction being issued against the facility, requiring significant remedial and repair actions. The North Carolina Division of Water Quality levied a $92,000 fine against Oceanview Farms in 1995, the largest ever. After the company appealed, an Administrative Law Judge reduced the fine to $75,000. Later the North Carolina Department of Environment and Natural Resources (DENR) settled the appeal and further reduced the fine to $50,000, payable over six years. The enforcement costs were not reduced and were collected for $11,820.49. While the case was closed in 2001, its repercussions still linger.

Legislative Responses to Oceanview Farms and Subsequent Spills

The spill provoked national media coverage accompanied by a swift legislative response. Other spills occurred around the same time, with a one million gallon swine waste spill on a different farm the same day as Oceanview's and an eight million gallon spill of chicken wastewater from a lagoon on July 3, 1995. On July 10, 1995, North Carolina Governor Jim Hunt ordered state water quality inspectors to do a blitz of inspections on the state's lagoons.

On July 11, 1995, the North Carolina General Assembly enacted the Swine Farm Siting Act.[56] The Swine Farm Siting Act required a 1,500 foot setback for lagoons from residences, with a farther setback from schools and a smaller one from property boundaries.[57] The Swine Farm Siting Act had a delayed effective date of October 1, 1995, which prompted a flurry of lagoon site evaluation activity between July and October of 1995.

Meanwhile, work began in earnest by a Blue Ribbon Study Commission.[58] The commission consisted of members appointed by the legislative leadership and the Governor. Based in part upon the commission's report, the General Assembly acted to strengthen permitting requirements beyond the minimum federal requirements.[59] This piece of legislation was the most comprehensive to date in North Carolina on the subject and is still referred to by its bill number, Senate Bill 1217. Senate Bill 1217 required that all operations with more than 250 swine obtain permits.[60]

Under these permits, the swine farms were required to follow a nutrient management plan with nitrogen acting as the limiting nutrient.[61] The farms are also required to use a certified applicator for waste application.[62] Senate Bill 1217 required DENR to conduct annual inspections of swine farms. It also increased setback distances, enhanced enforcement of the Swine Farm Siting Act, and increased the maximum daily civil penalty assessable against animal operations from $5,000 to $10,000 per day per violation.[63] The effects of Senate Bill 1217 were significant in North Carolina. Due to the requirements for waste management plan certification, many operations had to undertake significant upgrades of their waste management systems. The North Carolina Supreme Court has held these statutory changes to constitute complete regulation of the field.[64]

In addition to these measures, North Carolina adopted a set of operator certification requirements for those who operated animal waste management systems. These requirements included licensing upon a written

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[56] 1995 N.C. Sess. Laws, Chapter 420.
[57] *Id.*
[58] 1995 N.C. Sess. Laws Ch. 542, Sec. 4.1 through 4.7.
[59] *Ibid.* at Ch. 626.
[60] *Ibid* at Sec. 1.
[61] *Id.*
[62] *Ibid* at Section 5.
[63] *Ibid* at Section 4.
[64] "We conclude from the foregoing specifications that North Carolina's swine farm regulations, the Swine Farm Siting Act and the Animal Waste Management Systems statutes are so comprehensive in scope that the General Assembly must have intended that they comprise "complete and integrated regulatory scheme" on a statewide basis, thus leaving no room for further local regulation." Craig v. County of Chatham, 356 N.C. 40, 50, 565 S.E.2d 172, 179 (2002)

examination, a disciplinary process, and continuing education requirements.[65] Since all animal waste management systems are required to have a certified operator, the threat of disciplinary sanctions has a significant salutary effect on waste handling practices. Lastly, North Carolina required that swine farm owners register the name of the owner of their livestock, or "integrator."[66] The registered integrator is to be informed of all violations observed at the owner's farm.[67] Each facility is mandated to have a compliance inspection annually.[68]

More Fish Kills and Moratorium Building Boom

During July, September, and October, 1995, extensive fish kills occurred in the Neuse River itself, a much larger water body than the New River affected by Oceanview Farms. Millions of menhaden, as well as many flounder, croaker, and striped bass, were killed. DENR collected copious water quality samples in the areas of the fish kills. The samples showed that the water lacked oxygen only 1 to 2 meters below the surface and contained a prevalence of algal blooms. During June of 1995, record rainfalls delivered a tremendous load of nonpoint source nutrients into the Neuse River. DENR took action.

On February 8, 1996, the North Carolina Environmental Management Commission (EMC) approved a draft conceptual Neuse River Nutrient Sensitive Waters (NSW) Management Strategy. The draft contained alternative language to further discussion at public workshops. Some initial changes were incorporated into the proposed rules as a result of comments received at the workshops and written comments. The Neuse River NSW Management Strategy's action plans called for the establishment of a Neuse River Basin Coordinator to coordinate activities of agricultural agencies involved in implementing the strategy, and to ensure progress toward successful installation of BMPs. The NSW management strategy proposal required a thirty percent reduction in nitrogen input to the Neuse River by all major contributors. A final set of rules was adopted by the EMC December 11, 1997.

For agriculture, these rules provide flexibility for implementing locally determined and appropriate site specific BMPs, rather than imposing identical requirements on all agricultural land throughout the basin. Farmers collectively achieved the thirty percent reduction goal by signing on with a Local Advisory Committee (LAC). Each LAC developed the local strategy and farm plans. Farmers who did not wish to work with an LAC were required to implement the default BMP option provided under the rules. This default option combined riparian buffers and water control structures with nutrient management planning. The LACs had a more extensive matrix of options available allowing for some operations to avoid costly changes by using the average reduction of all operations. The reduction goal was thirty percent below the level determined as the average for 1991-1995. The reduction goal was met in the plans developed.

These regulatory changes were quickly followed by even more stringent legislative measures. In 1997, the Clean Water Responsibility and Environmentally Sound Policy Act was passed. This bill established a moratorium on swine farm construction and expansion, expanded county zoning power over large swine farms, directed that odor control rules be developed, strengthened the Swine Farm Siting Act, and made sweeping changes to the nutrient regulations for other dischargers. This moratorium has been amended in 1998, 1999, 2001 and 2003, with the last amendment extending the moratorium to September 1, 2007.

Like the Swine Farm Siting Act, the initial moratorium had a delayed effective date. The delayed effective date triggered a new rush of applications and building activity to beat the deadline. Some in the industry have argued that the industry's response in building in anticipation of the moratorium contributed to a pork supply glut, which led to swine price drops of 1998 and later.[69] These price drops have also contributed to the exit of many smaller producers.

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[65] N.C. Gen. Stat. §§ 90A-47 *et seq*. (2003).
[66] N.C. Gen. Stat § 143-215.10H (2003).
[67] *Ibid*. at (d).
[68] N.C. Gen. Stat. § 143-215.10F (2003).
[69] *Industry Pacesetters: Interview with Bill Prestage*, NATIONAL HOG FARMER (June 15, 2001).

Conclusion

Conflicts between swine owners and others is nothing new in North Carolina. Legislators, mayors, courts, governors and other public officials have made many adjustments in the law to accommodate the distinct problems of their age. In the 18th century, the problem with pork was viewed as a problem of waste land, what we now think of as wilderness. In the 19th Century, the problems were conflicts between swine in the streets and pedestrians in towns. Towards the end of the 19th century, legislators responded to concerns of constituents by changing the law from open range to requiring livestock owners to enclose their livestock. In the 20th century, the enclosed livestock have undergone a boom in population and created a new set of conflicts. At each stage, these conflicts have evoked a response from North Carolina's government. While North Carolina has been challenged by the pork production boom it has enjoyed, the State's response has been to do more than talk about it. North Carolina's laws have responded to the challenges as they arose, albeit with all deliberate speed.

Ecological Indicators of Air Quality: Plans and Progress

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Abstract

The H. John Heinz III Center for Science, Economics and the Environment has initiated a multi-stakeholder, technically-based process to determine indicators and analyses that can be used to monitor and report ecological responses to changes in air quality. This process builds upon the success of the Heinz Center's 2002 report, The State of the Nation's Ecosystems , in reporting indicators of ecosystem condition but will expand on it by considering ecosystem responses specific to changes in air quality. Three related areas are addressed: appropriate indicators of condition and functioning in terrestrial and aquatic ecosystems, indicators quantifying exposure to air pollutants, and analyses of the potential for observed ecosystem changes to result from air pollutant exposure. Interim results include a review of candidate indicators for both ecosystem condition and pollutant exposure as well as potential methods of analysis using statistical and process-based models. Initial review has indicated the potential importance of considering agricultural emissions in assessing ecological changes in neighboring terrestrial ecosystems. The project is anticipated to result in a major publication by the Heinz Center at the end of the third year, with recommendations for both national level indicators, analyses of available data, and a regional case study. Support for this project comes from a cooperative agreement with the EPA's Clean Air Markets Division.

Agricultural Air Quality Policy in Iowa

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Abstract

Iowa is one of the leading producers of agricultural livestock in the United States, especially with respect to pork and egg production. Air qualilty near livestock operations in Iowa remains a primary concern to rural residents. It is the responsibility of the Air Quality Bureau of the Iowa Department of Natural Resources to work closely with the general public, elected officials, and affected stakeholders to develop policy that both protects and maintains the quality of air in rural Iowa and allows the agricultural industry to continue to thrive. An example of this effort is the establishment of a health based standard for hydrogen sulfide applicable during an on-going field study of air qualilty in rural Iowa, that was developed over a four-year period.

Introduction

Iowa is one of the leading producers of agricultural livestock in the United States. According to the most recent census by USDA (USDA, 2004), Iowa is the number one producer of pork and the number one producer of eggs in the country. As such, air quality in the vicinity of animal feeding operations has remained a prominent issue in the state. Citizens living near livestock operations are concerned about odors and potential health effects from invisible gases like ammonia and hydrogen sulfide. The Iowa Department of Natural Resources, specifically the Air Quality Bureau, has the responsibility of maintaining the quality of air in rural Iowa. Developing public policy to accomplish this task has been difficult.

Historical Overview and Timeline

The issue of potential health effects due to emissions from animal feeding operations was first brought to the attention of the department in January of 2001 by a grassroots organization called the Iowa Citizens for Community Improvement (Iowa CCI). The group filed a petition for rulemaking before the Iowa Environmental Protection Commission (EPC), a panel of nine citizens who provide policy oversight over Iowa's environmental protection efforts. The petition requested that the department adopt specific fence line and ambient air quality standards for hydrogen sulfide, ammonia and odor applicable to animal feeding operations. Although the petition was eventually denied, it prompted the Governor of Iowa and the Director of the Department of Natural Resources to call upon the expertise of the University of Iowa and Iowa State University to weigh in on the issue. Specifically, the Universities were asked to provide a joint recommendation on how the department should address the impacts of air quality surrounding animal feeding operations on Iowans (Iowa State University and The University of Iowa Study Group, 2002). In February of 2002, the Universities co-authored a report entitled the Iowa Concentrated Animal Feeding Operations Air Quality Study (hereafter referred to as the University Report), which included several recommendations on how to proceed with public policy. One recommendation from the University Report was for statewide ambient air quality standards for ammonia and hydrogen sulfide:

Hydrogen Sulfide
It is recommended that hydrogen sulfide, measured at the CAFO property line, not exceed 70 parts per billion (ppb) for a 1-hour time-weighted average (TWA) period. In addition, the concentration at a residence or public use area shall not exceed 15 ppb, measured in the same manner as the property line. It is recommended that each CAFO have up to seven days (with 48 hour notice) each calendar year when they are allowed to exceed the concentration for hydrogen sulfide.

Ammonia
It is recommended that ammonia, measured at the CAFO property line, not exceed 500 ppb for a 1-hour TWA period. In addition, the concentration at a residence or public use area shall not exceed 150 ppb, measured in the same manner as the property line measurement. It is recommended that each

CAFO have up to seven days (with 48 hour notice) each calendar year when they are allowed to exceed the concentration for ammonia.

Two months after the release of the University Report, the Iowa General Assembly adopted Senate File 2293, which instructed the department to complete a comprehensive field study measuring emissions of ammonia, hydrogen sulfide, and odors to determine if these gases were present at levels that could cause material and verifiable adverse health effects in areas where people live and spend time, such as residences, commercial, educational, or religious establishments, or public use areas (Iowa Code section 459.207). In response to the new law, the department began monitoring ammonia and hydrogen sulfide using continuous monitoring techniques at ten locations throughout the state near some of Iowa's largest livestock operations. In addition, the department implemented an odor study, where field staff were trained to gauge odor levels using an instrument called a scentometer.

In July, 2002, the department moved forward with a rule recommending the adoption of the ambient air quality standards as recommended in the University Report. After extensive public comment, the EPC approved a final version of the rule in April, 2003. Specifically, hydrogen sulfide was set at 15 parts per billion (ppb), daily maximum 1-hour average, and ammonia at 150 ppb, daily maximum 1-hour average. The standards were formulated as a three-year average of the annual eighth-highest daily maximum hourly average concentration.

This initial attempt by the department to implement the recommendations of the University Report by the establishment of statewide ambient air quality standards for ammonia and hydrogen sulfide was deemed too broad and was overturned by the Iowa General Assembly.

The department modified its approach and in December, 2003 brought forth recommendations to the EPC to establish a hydrogen sulfide health effects value (HEV) and health effects standard (HES). The rule proposed an HEV of 15 ppb 1-hour daily maximum, and an HES at 15 ppb 1-hour daily maximum not to be exceeded more than 7 times a year. Recommendations to establish similar standards for ammonia and odors were not brought forth.

The Iowa General Assembly adopted House File 2523 in April 2004, providing for the regulation of air quality by establishing minimal risk levels, creating an odor panel, and making penalties applicable. However, this law was eventually vetoed by the Governor.

In September, 2004, the final version of the HEV/HES rulemaking was approved by the EPC. Based on public comments and recommendations by the Iowa Department of Public Health, the levels of the HEV and HES were changed from 15 ppb to 30 ppb, respectively. The rule became effective in September 2004.

The Regulatory Bar

The HEV represents a level commonly known to cause a material and verifiable adverse health effect. The HEV is 30 parts per billion (ppb) averaged over one hour.

The HES represents a level to determine if the baseline data from an ongoing field study indicates a need to develop regulatory plans and programs to mitigate hydrogen sulfide emissions from animal feeding operations. The HES is 30 parts per billion (ppb) daily maximum one–hour average, not to be exceeded more than seven days in one year.

The health effects standard is used primarily as a regulatory bar in order to determine if harmful concentrations of hydrogen sulfide are present near animal operations. The HES acts as a regulatory trigger that if exceeded, requires the department to take action to reduce emissions. All data obtained during the course of the on-going field study are compared to the HES. Should the HES be exceeded during the field study, the department will develop plans and programs to mitigate hydrogen sulfide emissions from animal feeding operations.

These values are applicable to animal feeding operations only, and apply only to "separated locations". These are areas where people live and spend time, such as residences, commercial, educational, or religious establishments, or public use areas. By law, monitoring sites for the field study are to be located in close proximity to these separated locations and not at the fenceline of the animal feeding operation, as would typically be done for monitoring of ambient air (Iowa Code section 459.207).

Outcome

Monitoring of ammonia and hydrogen sulfide concentrations continues to be collected at ten locations throughout Iowa near large livestock facilities, with data being compared to the hydrogen sulfide health effects standard. 2005 data indicates that the HES was not exceeded during the calendar year, therefore comprehensive plans and programs to mitigate emissions of hydrogen sulfide from animal feeding operations have not been established at this time. The Air Quality Bureau's website at www.iowacleanair.com provides links to interim data from the field study and to real-time monitoring data, as well as reports that provide more detailed graphical analyses.

The odor portion of the field study concluded in December, 2005. Tables with results and a final report detailing the methodology of the study and measurement data will be available at the Bureau's website in March 2006.

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A Life Cycle Approach to Policy Decisions on Swine Waste Management Alternatives

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The comparison of swine waste management technologies can be done in a number of ways, e.g. economics, emissions of a single chemical such as ammonia, in terms of total environmental impact, etc. The latter is referred to as the net environmental benefit or life cycle approach. Use of a life cycle concept attempts to understand the transfer or shifts in pollution that often occur in complex systems such as the swine production industry.

The Water Resources Research Institute of the University of North Carolina has funded a parallel study to the large Attorney General/Smithfield Agreement research effort. The goal of the WRRI project is to determine for a small number of swine waste management alternatives, what life cycle results would occur and compare that to the results from decision-making for ammonia emissions. Four swine waste management technologies were investigated:

1. conventional lagoon and spray irrigation (with lagoon solids removal and land application when full)
2. covered lagoon with spray irrigation, utilization of methane production for electricity, and lagoon solids removal and land application when full
3. biological aerated filter process with land application of effluents and solids, and
4. Harvestore collection and land application of raw waste.

An engineering and science approach was used to assess the energy (usually electricity) and emissions from each of these technologies. Because a life cycle approach was used, all related supply chain emissions and energy requirements were also added. Thus when electricity is used, the emissions from electrical power generation are included. When swine waste NPK are land applied, emissions and energy requirements from industrial plants for NPK are correspondingly reduced creating an environmental benefit.

Analysis of these technology comparisons was done for a number of environmental parameters, but in this summary, two are highlighted. The first is the ammonia emissions to the air and the second is the impact on global climate change potential (a combined effect of CO_2, methane, and nitrous oxides, using scientific rules for combining these emissions and expressing the effect as equivalent CO_2. These results are provided in Figures 1 and 2.

Figure 1: Ammonia emissions from life cycle evaluation of swine waste management technologies

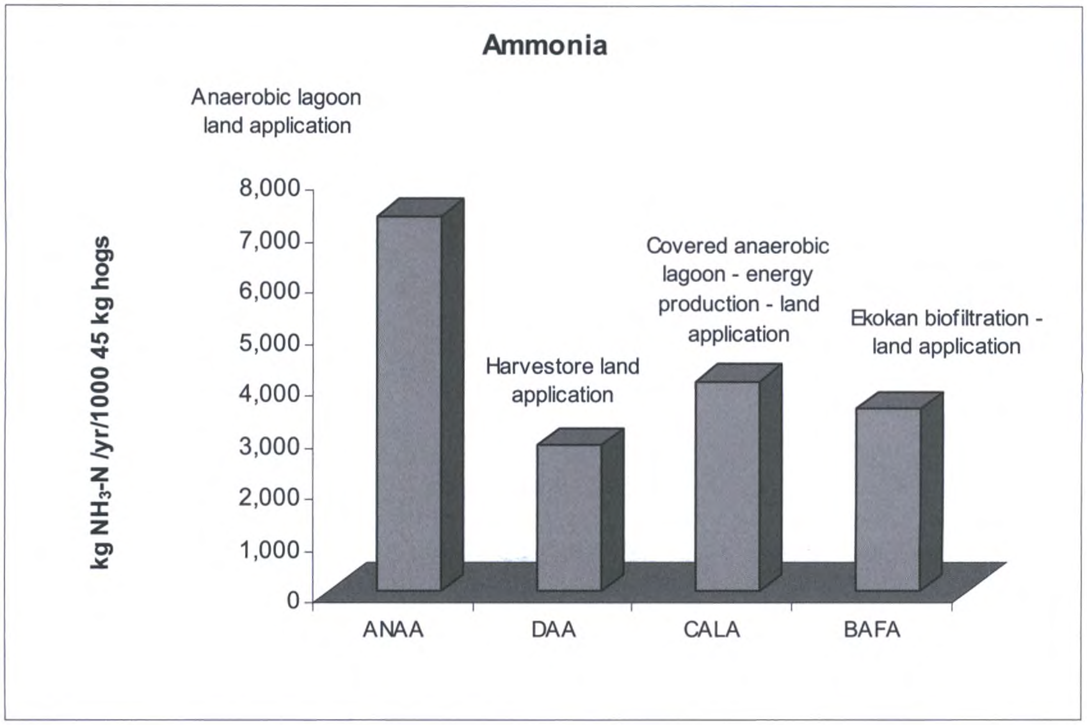
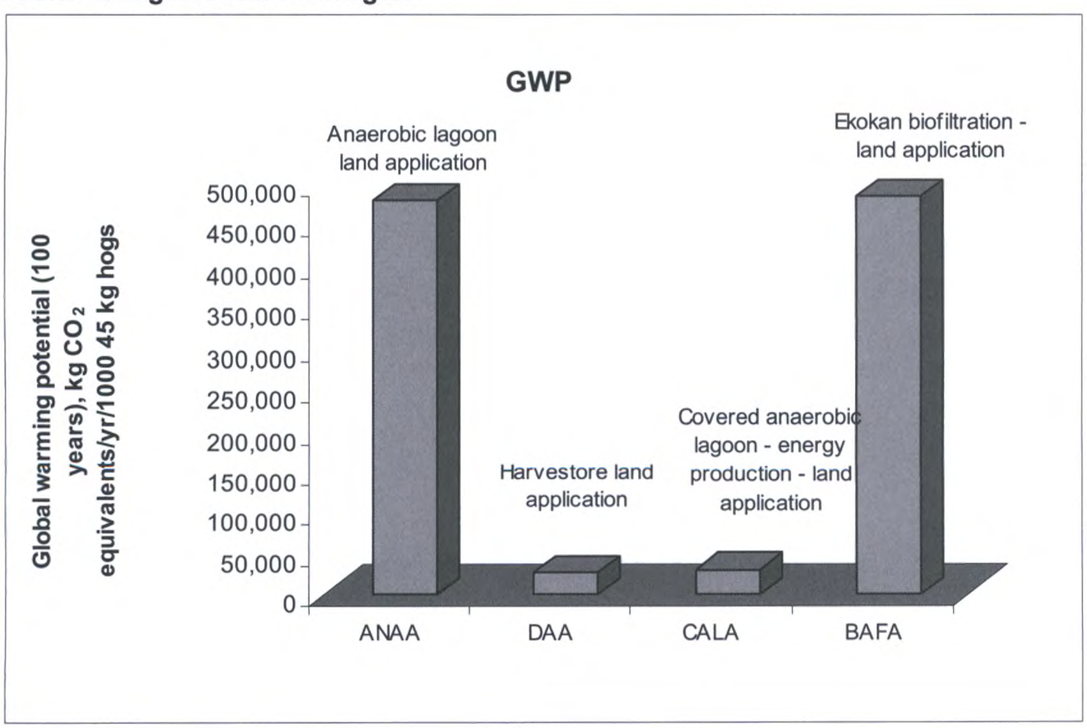


Figure 2: Global warming emissions (CO_2 equivalents) from life cycle evaluation of swine waste management technologies



The life cycle approach shows that changing swine waste management technologies results in two clear geographic transfers of pollution. First, using ammonia emissions as the criterion for technology choice (Fig. 1), the Ekokan biofiltration is slightly better than the covered lagoon and significantly better than the current lagoon-land application system. However when one looks at global warming emissions, the Ekokan system has a substantially higher impact than the new technologies of the covered lagoon or the direct land application. Thus there is a geographic shift from ammonia emissions at the swine site, to larger emissions at the power generation facilities.

The second shift is from one form of emissions to another. In this case, the shift is from ammonia in air to the constituents comprising global warming potential. This is referred to as chemical or pollution shift. In the report, other environmental shifts are documented for these four swine waste management alternatives.

Conclusions

1) A life cycle approach to swine waste management technologies selection provides the most comprehensive assessment of environmental impact.
2) Mass balance approaches provide more independent measures and lowest cost technique for determining ammonia emissions from most swine waste management technologies.
3) There are shifts in geographic impact and chemical emissions that occur when selections are made between swine waste management technologies. These shifts should be better understood when industry-wide decisions are made.

Regulation of Ammonia from Agriculture in Denmark: Concept and Methodology

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Abstract

Emissions of ammonia from Danish livestock farms are strongly regulated. Manure applications to the fields are restricted to take place in the growth seasons and within certain limits for the total load per hectare on annual basis. Farmers need to document access to fields for application of the manure. Finally the farmers need to apply to the local authorities when they intend to increase the animal production. These applications for increasing the production are treated using an official Guideline for Environmental Impact Assessment (EIA) of ammonia loads of the local nature. A structural change taking place in Denmark by 2007 will move the obligation of carrying out this assessment from the counties to the municipalities. The current Guideline for making the assessment is under review. One of the aims is to make the assessment simple to perform and updated with respect to the latest knowledge about dispersion and deposition. The present paper outlines the planned methodology behind the suggested future Guideline for EIA on local nature of ammonia emissions from livestock farming in Denmark.

Introduction

The anthropogenic emission of nitrogen compounds to the atmosphere is of great concern due to its impact on both human health and environment. The Dobris assessment showed that in the beginning of the 1990'ties the critical loads and levels for atmospheric nitrogen compounds were exceeded over large parts of Europe (EEA, 1995). The Third Assessment in 2000 showed that despite considerable improvements concerning the pressures on nature, the critical loads were still exceeded for more than half of European ecosystems (EEA, 2003). In Denmark the impact of atmospheric nitrogen on terrestrial and marine ecosystems is know to be very significant (Bach et al., 2005). Episodes of oxygen decificts in bottom waters, in worst case situations followed by death of fish and bentic fauna, are frequent phenomena in the inner Danish waters. These episode are strongly linked to the anthropogenic nitrogen loads, of which current estimates have shown that about 30% of the bioavailable nitrogen arise from atmospheric loadings (Spokes et al., 2006). It has furthermore been shown that critical loads are exceeded for more than 70% of Danish terrestrial ecosystems (Bach et al., 2005). For the most sentitive Danish terrestrial ecosystems calculations have shown that even the atmospheric background deposition exceeds critical loads (Hertel et al., 2003). Calculations also show that Danish sources contribute to 40 - 45% of the background deposition over land (Ellermann et al., 2005).

Figure 1 shows the background atmospheric nitrogen deposition to Danish land areas for 2004 and forcasted for 2020. The calculations show that a signifant part of the country has loads in the range pf 14 to 16 kg N/ha in 2004. The projection for 2020 shows that loads will still be in the range 12-15 kg N/ha over large parts of especially the western part of the country. However, close to local livestock farms this contribution may be significantly higher both in the present and future situation and lead to significant exceedances of critical loads even for the less sensitive ecosystems. Calculations with NERI's local scale plume model OML-DEP have shown that for a typical Danish livestock farm, a little more than 20% of the annual emission from barns and storages will deposited within a radius of 2km from the farm.

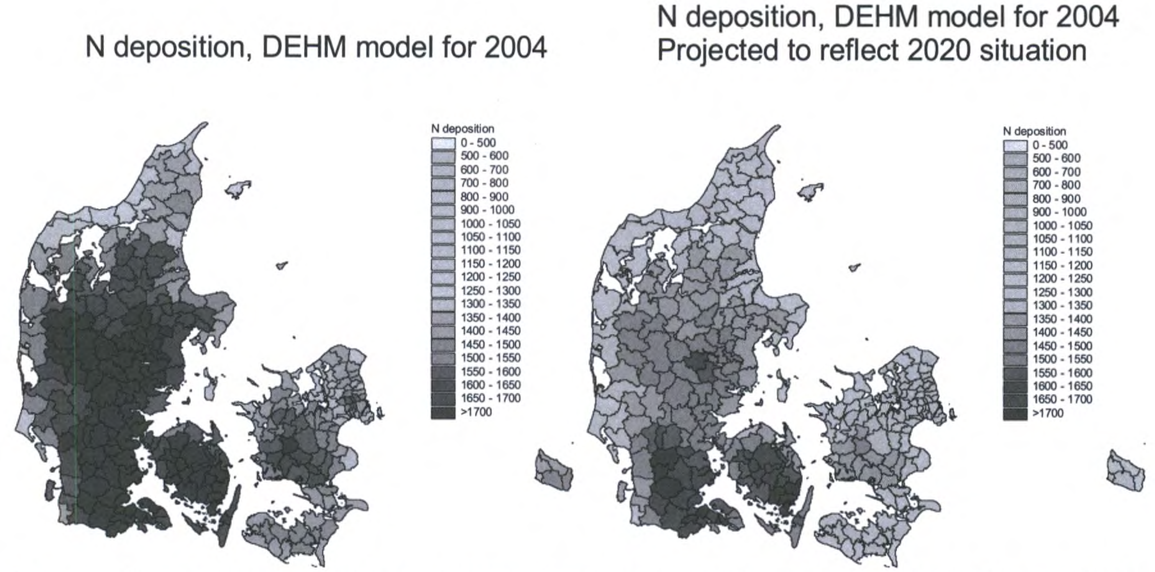

Figure 1 Computed atmospheric nitrogen loads on municipality level. The left figure is for 2004 and the right figure a projection for 2020. The projection is based on simple scaling of Danish and international contributions based on EMEP expert emissions in 2003 and 2020. Depositions are in kg/km^2 (divided by 10 this equals kg/ha)

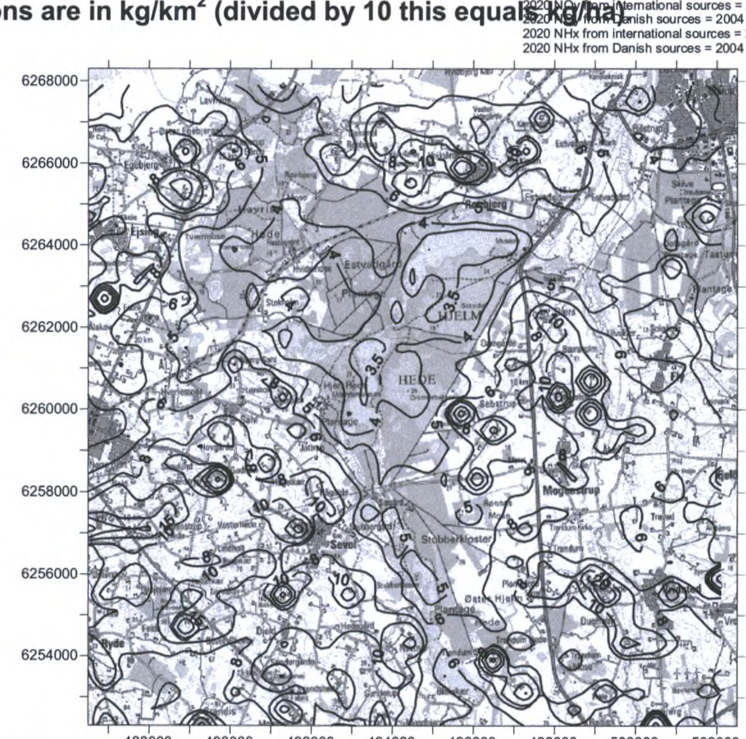

Figure 2 Atmospheric nitrogen loads (kg N/ha) from local ammonia emissions to Hjelm Heath in Jutland in the western part of Denmark. Coordinates along the axis represent UTM-32N. Calculated with the OML-DEP model within the Danish Background Monitoring Programme (Ellermann et al., 2005). The background deposition is about 11 kg N/ha for this area.

The annual emissions from barns and storages may be in the range of 20 - 40 kg N and up to even 2000 kg N. Therefore the average annual deposition in the 2km radius around the farm is espected to be in the range of 0.1 kg N/ha and up to 4 kg N/ha. Although a relatively steep gradient will be present away from the farm (see Figure 3). Similarly depositions will depend on the frequency of wind directions – in this context the prevailing wind direction in Denmark is from south-west. Similarly to the contribution from emissions arising from barns and storages, there is a significant contribution from the emissions from fields related mainly related to manure application and to a smaller extend also arising from evaporation of ammonia from crops. This contribution from the fields has also been estimated to be in the range of 2 to 4 kg N/ha as average in a 2km zone around the fields – again with a realtively steep gradient away from edge of the field. This is demonstrated in Figure 2, which shows OML-DEP calculations performed in connection with the Danish background monitoring programme (Ellermann et al., 2005) for a heath area in the western part of the country. It is evident that depositions in this area arising from local livestock farming may contribute to 3 to 10 kg N/ha on annual basis. The impact on nature areas will be governed totalle by the specific situation of livestock farms in the vicinity of nature areas. A recent study has shown that depositions of atmospheric nitrogen to Danish nature areas would typically be reduced by 1 to 2 kg N/ha/year by establishing 200m buffer zones around Danish nature areas (Schou et al., 2006).

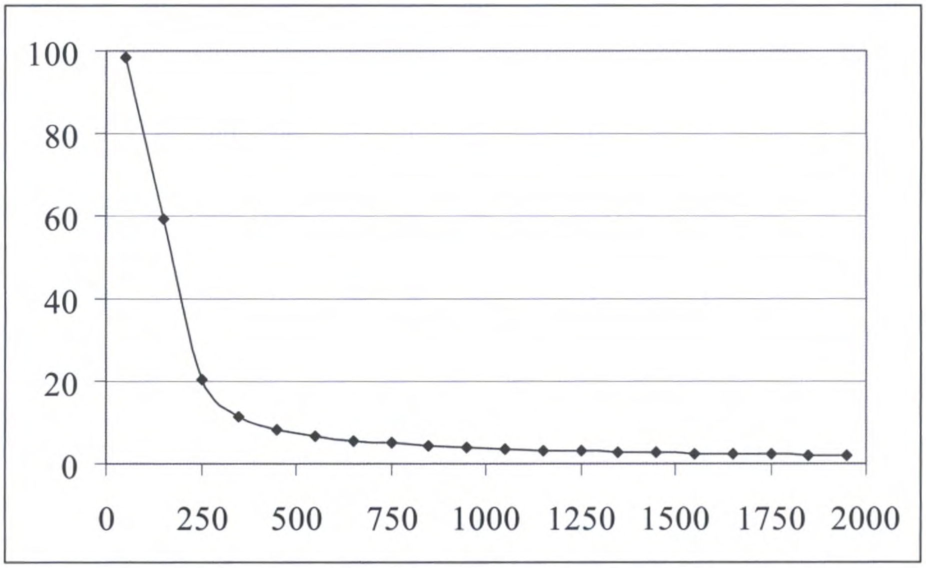

Figure 3 Ammonia deposition (kg N/ha/year) from a livestock farm with 100 animal units of cattle (68 mature cows and 69 calves), which yields an ammonia emission of 1114kg N (895 kg N from barn and 219 kg N from storage. Depositions are shown as function of the distance from the source. Calculations with NERI's local scale plume model OML-DEP.

The negative impact on the environment of anthropogenic nitrogen has been the background for three large Danish National Aquatic Action Plans in 1987, 1998 and 2004 aiming at reducing the nitrogen (and phosphorous) input into the aquatic ecosystems (Anonymous, 2004). The first two Danish national actions plans have lead to vast investments in waste water treatment plants, establishing and improving storage tanks for manure etc, and have thereby succesfully reduced the nitrogen loads of the environment. The third action plan (2005-2015) aims at reducing ammonia and odour emissions. These action plans have influenced also the atmospheric emissions in amount as well as seasonal distribution through a change in agricultural praxis (Skjøth et al., 2004).

However, beside the regulation associated to the aquatic actions plans, the emissions from livestock farms are also regulated directly. The farmers are obligished to make an application to the local authorities in connection with establishing new or increasing animal production of existing livestock farms. In connection with such an application, the local authorties perform an EIA of ammonia emissions on the local nature in the vicinity of the livestock farm. This is currently carried out by following a Guideline from the Danish Forest and Nature Agency (Bak, 2003). The applications have until now been handled by the counties, but by 2007 Danish local authorities will be restructured and the existing 9 counties in Denmark will closed down. The current obligations of the counties will be distributed between state, municipalities and three new regional centres. However, the regulation of livestock farms will in the future be handled solely by the municipalities that therefore will have to build up expertise in this field. Currently Denmark has 270 municipalities, but after the structural change, a number of these will be merged and there will remain approximately 98 municipalities. These 98 municipalities will then perform the regulation of livestock farms, and it is anticipated that this will be carried out using the suggested new Guideline.

The Danish municipalities have claimed that the current Guideline for assessment of the impact of ammonia emissions from livestock farms is too complex. At the same time new model tools have become available since the current Guideline was made. The counties have constructed a spreadsheet that has eased the used of the Guideline, but still a revision is strongly requsted in order to improve the operationality and update with state-of-the-art. The Danish Forest and Nature Agency has therefore initiated a project to form the basis for a revised version of the guideline. The present paper describes the basic methodology of this new Guideline suggested to be the tool for the municipalities in their furture handling of applications from Danish farmers for establising or increasing livestock production on their farms.

Concept for the suggested calculations procedure

The concept behind the calculation procedure in the suggested new Guideline for ammonia from animal production includes three steps with increasing complexity (see also the sketch presented in Figure 4):

A. Simple screening – a method for quick assessment of potential environmental impact as a result of airborne ammonia emitted from smaller livestock farms. This method will be used for smaller livestock farms (<75 animal units) and is solely intended for a first crude screening of farms with insignificant impact on the local nature. The screening is not carried out for cases when manure is brought to a biogas plant.
B. Standard method – the basic method for assessment of environmental impact based on nomograms and tables. This method is intended to be used for all cases that cannot be close after step A, except for situations when the applicant or others ask for more detailed treatment after step C.
C. Detailed model calculations – a detailed mapping of nitrogen loads based on model calculations and similarly detailed critical load estimates for the local nature in the nearby region of the livestock farm. This method is intended to be used only when the applicant or others may wish so on basis of a number of predefined guidelines for when such a possibility should be open.

STEP A: SIMPLE SCREENING

The annual emissions from barns, storages and manure application are calculated for a “standard” livestock farm. The emissions are based on the current animal production and using standard emission factors according to a Danish norm system. Emissions from manure application to fields are calculated as a fixed fraction of the applied nitrogen.

Influence zones around the farm and the associated fields are computed. These zones cover the area in which there is a potential risk for significant impact on the ecosystems. In the calculation of the extent of the influence zone, the frequency of various wind directions is taken into account. The radius of zone in a given direction is calculated so that it represents the distance in which the deposition is below a certain predefined level; a level at which the contribution to eutrophication is considered insignificant.

The influence zones around the sources are drawn on a map where the local nature areas are shown. Especially in cases of overlap between influence zones there is a potential risk for negative impacts of ammonia from livestock farms on the sensitive local ecosystems. In such cases the application from the farmer for establishing or increasing the animal production will subsequently be treated after the standard method in step B.

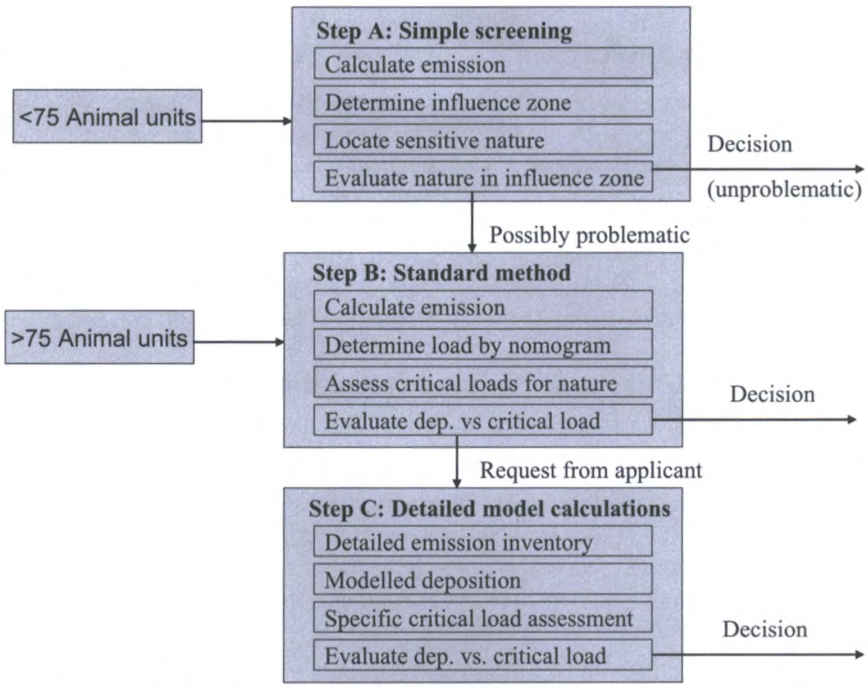

Figure 4 Sketch to illustrate the overall calculation procedure in the suggested new Guideline for assessment of environmental impact of ammonia emissions from livestock production in Denmark (Geels et al., 2006).

STEP B: THE STANDARD METHOD

Just as for the screening method the emissions from barns, storages and manure application are calculated, but in this case the emissions may deviate from the defined standard livestock farm concerning the following points: the nitrogen separation (feeding practise, production level and the time the animals spend outdoors in the fields); surface area and cover of the storage tank; time and method for application of manure on the fields, application of air scrubbers, acidification of manure or other documented ways to reduce the ammonia emission.

The annual background atmospheric nitrogen load as an average over five years is provided from standard model calculations with DEHM-REGINA at NERI on a 16.67km by 16.67km grid. An online routine on a central server selects background data on basis of user defined coordinates for the location of the farm, and the selection of background depositions is taking into account the specific land use of the area. These data will be updated on regular basis.

The deposition of nitrogen from the local sources is determined on basis of a set of standard nomograms. These nomograms are computed using the OML-DEP; a plume model with a deposition module included. A simple scaling based on the emissions from the specific livestock farm is applied, since the relationship between emission and ambient concentrations to a good approximation may be considered as linear. The calculation is performed out to the distance from the source in which the deposition is below a certain predefined level; a level where at which the contribution to the overall load is considered insignificant. This procedure is repeated for all point and area sources to account for possible overlap of influence zones.

A standard table has been compiled for the determination of typical intervals for the critical loads of the local nature areas. However, in the assessment more specific empirical or semi-empirical critical loads for the actual nature are of high importance. Such critical loads may be determined from observations of relationships between loads and impact from research or surveillance projects, but may also be based on extrapolation from laboratory studies. Another way to determine the critical load is to apply models that are based on chemical criteria for a scientifically shown relationship between loads exceeding a given critical value and unwanted effects.

The computed nitrogen depositions are drawn on maps using wind direction frequency corrected influence zones for each source. The total deposition is determined with and without the increase in animal production that the farmer has applied for permission to have. The total deposition is compared to critical loads for the nature areas in the area nearby.

STEP C: DETAILED MODEL CALCULATIONS

The emissions are determined in similar way as for step B, but in this case a detailed seasonal variation is applied. The emission variation handles separately the variation in releases from barns, storages, application of manure and fertilizer, grazing animals, evaporation from crops as well as other minor sources. This seasonal variation is computed using a procedure based on a simple growth model and primarily driven by the temperature (Ambelas Skjøth et al., 2004; Gyldenkærne et al., 2005).

The model calculations are performed with the DAMOS (Danish Ammonia Modelling System), which is based on a combination of the OML-DEP and the regional model DEHM-REGINA. OML-DEP receives initial concentrations from DEHM-REGINA and performs calculations hour by hour for a grid of receptor points in a suitable zone around the farm. The initial concentrations for each hourly calculation are taken from the up-wind direction and meteorological data are provided from a meteorological model (either Eta or MM5).

As a starting point, calculations will be performed for one year and based on the latest available data – typically this will mean from the previous year.

The OML-DEP model

The OML-DEP is developed on basis of the OML models at NERI, which are Gaussian plume models handling dispersion of pollutants from point and area sources within a distance of about 20km from the sources (Olesen, 1994; 1995; Berkowicz et al., 1986). The new feature of the OML-DEP compared with the previous versions of the OML model is that it contains a deposition description. The model is set up to perform calculations for a regular receptor net of e.g. 400m x 400m for an area of e.g. 16km x 16km. OML-DEP is a part of the DAMOS (Danish Ammonia Modelling system), which consists of the OML-DEP coupled to the regional scale model DEHM-REGINA (Figure 5).

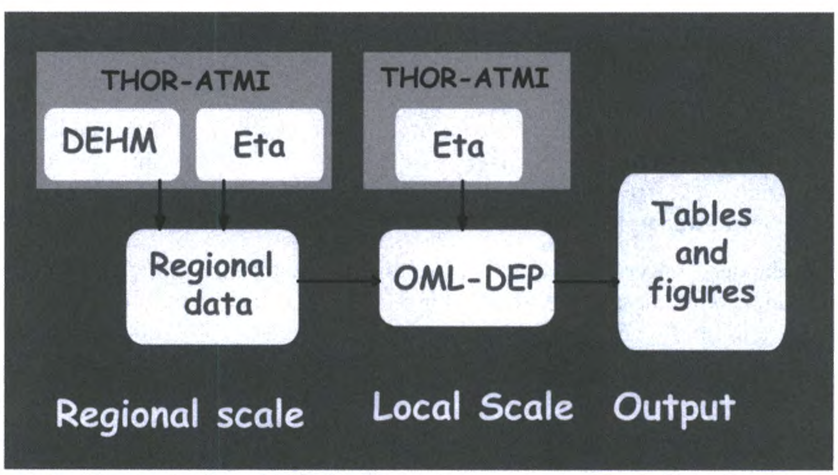

Figure 5 Sketch to illustrate the DAMOS (Danish Ammonia Modelling System) for assessment of atmospheric ammonia loads from livestock farms.

In the DAMOS system, the OML-DEP generates an initial concentration field based on upstream background concentrations from DEHM-REGINA. Calculations are based on local meteorological date from either a local mast or generated by a meteorological forecast model (at NERI the Eta and MM5 models are applied). Emission data for ammonia are obtained on basis of the Central Livestock Registry and the Basic Agricultural Registry, the farmers manure budgets reported to the National Crop Directorate and maps of the agricultural fields in the country (Gyldenkærne et al., 2004; 2005). The land use data, which are important for the deposition velocity, are obtained from national Area Information System AIS (Nielsen et al., 2001). OML-DEP was applied within the Danish national monitoring programme (Ellermann et al., 2005).

Conclusions

Ammonia emissions from Danish farms are regulated by local authorities. A structural change of the local authorities will take place by January 1st 2007. One of the consequences of this structural change is that applications from farmers for establishing or increasing animal production on livestock farms will be handled by municipalities that currently have no expertise in this field. The current Guideline is furthermore complex and new model tools have recently become available. A new Guideline for assessment of the environmental impact of ammonia emissions from livestock farms in Denmark is therefore suggested. The basic methodology of proposed calculation procedure consists of three steps with increasing complexity. First step is a simple screening based on calculation of influence zones around the sources, and comparing these with the situation of nature areas in the surroundings of the farms. Second step is the standard method that is expected to be used on the main part of the applications. In this step the load estimates are based on a nomogram method, where the curves have been produced from calculations with NERI's local scale model OML-DEP. The third step is planned only to be used only on special request by either applicant or authority. This method consists of detailed model calculations with DAMOS – the combination of the local scale plume model OML-DEP and NERI's regional scale model DEHM-REGINA for calculation of the background loads.

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Evidence of Enhanced Atmospheric Ammoniacal Nitrogen in Hell's Canyon National Recreation Area: Implications for Natural and Cultural Resources

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Abstract

Regional agricultural operations release large amounts of fertilizing pollutants to air sheds and waterways of the northwest US. To evaluate air pollution threats to historic rock paintings and natural resources along the Snake River in Hells Canyon National Recreation Area, Oregon and Idaho, USA, we monitored ambient ammonia, nitrogen oxides, sulfur dioxide, and hydrogen sulfide at five stations along 60 km of the Snake River valley floor from July 2002-June 2003 and obtained ozone and fine particulates concentration data from the Hells Canyon IMPROVE station. Ammonia concentrations were high, peaking in spring and summer; the nutrient-laden Snake River is the most likely source. Ammonium nitrate concentrations in fine particulates peaked in winter with drainage of stagnant air masses from the Snake River Basin, a national center of livestock and crop production. Other pollutant concentrations were within background ranges for remote locations. Ammoniacal nitrogen in Hells Canyon is at levels known to adversely affect western biotic communities and ecosystem processes and is potentially corrosive to clay-based pictographs. Better controls on agricultural emissions and enforcement of state water quality standards would help protect these precious resources.

Introduction

Hell's Canyon National Recreation Area (HCNRA), including the Class 1 Hells Canyon Wilderness, encompasses 71 miles of the Snake River along the northern Oregon and central Idaho border and contains one of the best-preserved collections of riverine archaeology in North America (Keyser 1992). Over 200 pictographs and carved petroglyphs ranging in age from 200 to 7,100 years are recorded in the National Register of Historic Places. In the1990's, US Forest Service career archeologists expressed concern that pictographs in HCRNA along the Snake River had deteriorated in recent decades (Schaff & Szymoniak 1996). Because air pollutants can dissolve rock and clay minerals (Van Grieken et al. 1998) and enhance the growth of biological weathering agents (Mansch & Beck 1998) of culturally modified stone, they postulated that air pollution in the Snake River corridor could be a cause of rock art weathering in Hells Canyon.

In 2000, the Forest Service sponsored a lichen study of the Snake River valley, its primary tributaries, and the adjacent Imnaha watershed, all within HCNRA (Geiser et al. 2006). Extensive bark cover of nitrophytic lichens and high nitrogen concentrations in lichen tissues indicated that nitrogen deposition was high throughout the study area but especially on the Snake River valley floor where most of the rock art is located. Higher bark pH at sites with higher lichen cover pointed to an ammoniacal as opposed to acidic deposition source. The authors concluded that despite HCNRA's remote location, deposition of nitrogen-containing pollutants was enhanced relative to other remote sites in the northwestern United States.

The present study was limited to the Snake River valley floor where the greatest threat is perceived. The objectives were to 1), measure levels of fertilizing and oxidizing pollutants that could volatilize from the Snake River or be transported in the air from local or regional sources, 2) identify which pollutants, if any, could adversely effect HCNRA cultural or natural resources at observed levels and 3), identify the most likely sources and peak transport times of these pollutants.

Methods

Sampling occurred from July 2002 – June, 2003 at 6 stations along 80 km of the Snake River valley floor in and near Hell's Canyon National Recreation Area. At five monitoring stations inside HCNRA, Ogawa passive air samplers (Ogawa & Co., 1230 S.E. 7th Ave., Pompano Beach, FL 33060, USA) for NO_x/NO,

NO_2/SO_2, and NH_3 and Maxaam samplers for H_2S (Maxaam Analytics, Inc., Centre for Passive Sampling Technology, 9331 48th Street, Edmonton, ABT6B 2R4, Canada) were placed on PVC posts about 2 m above the ground level. There were two replicate removable collection pads for each gas. Collection pads for NO_x, NO_2, SO_2, NH_3, and H_2S were replaced with clean, unexposed pads every two weeks to four weeks. Sampler components, pollutant extraction procedures and calculations of ambient concentrations followed by the analytical laboratories (USDA Forest Service Pacific Southwest Research Station, Riverside, CA 92507 and Maxaam Analytics) are described by Ogawa & Co. (1999) and Tang (2001). At the sixth station, HECA in Oxbow, OR we obtained 2001-2003 IMPROVE mean daily concentrations of ammonium nitrate and ammonium sulfate in fine particulate matter and July 2002-September 2003 hourly mean ambient ozone concentrations from a co-located portable ozone monitor (Model No. 202, 2B Technologies, Inc., PO Box 288, Golden, CO 80401, USA). Ammonium nitrate and ammonium sulfate concentrations at HECA and all other Oregon, Washington and Idaho IMPROVE stations were obtained from the Visibility Information Exchange Web System (VIEWS) (http://vista.cira.colostate.edu/views)for comparison. Frequencies and seasonality of pollution transport to HCNRA from different geographic source areas during days of peak NH_4NO_3 and $(NH_4)_2SO_4$ concentrations were estimated using 4-24 hour back trajectories at HECA available from VIEWS.

Results and Discussion

We found that ozone, sulfur dioxide, hydrogen sulfide, and nitrogen oxide concentrations in ambient air, and ammonium sulfate in fine particulates along the Snake River valley floor of Hells Canyon National Recreation Area were within background ranges expected for remote areas in the western US. In contrast, ambient ammonia and ammonium nitrate in fine particulates were seasonally enhanced. The Snake River is the most likely cause of elevated atmospheric ammonia levels detected in HCNRA, which peak in spring and summer and are most elevated close to the river, while regional atmospheric transport, especially from the Snake River Basin in winter, is the most likely source of elevated depositional ammonium nitrate detected throughout the study area by the lichen study.

Compared to other parts of the US, ammonia emissions in southeastern WA, northeast OR and the Snake River Basin of Idaho are high with total regional emissions estimated at 43,000 tons in 1998 (EPA 2005). The reaction of nitric acid with ammonia gas emitted from agricultural operations results in the formation of ammonium nitrate particles that can be transported to remote parks and wilderness, depending on the pattern of local ammonia emissions relative to the supply of nitric acid vapor (Schoettle et al. 1999). Surface winds are most likely to channel regional pollutants into HCNRA via the Canyon (Schaaf & Szymoniak 1996); gravity would drain winter time Snake River Basin inversions through the Canyon.

Local surface waters are also an important anthropogenic source of nitrogen. The Snake and Boise Rivers, which join up-river from HCNRA, are straddled by Boise, Twin Falls and Idaho Falls, many minor urban and industrial areas, and drain a major national agricultural region. The lakes above Brownslee and Oxbow dams experience severe algal blooms each summer and chlorophyll a concentrations and phosphate levels do not meet state standards for these water quality indicators nearly 100% of the time (IDEQ 2004). Anearobic microbial activity in the deep, hypoxic lake waters produce high levels of ammonia, released from the bottom of the dams into Hells Canyon reach (Meyers et al., 2003). Volatilization of ammonia from dissolved ammonium is greatly favored under high pH conditions created when algal blooms consume large amounts of CO_2, (Brady 1984); the pH of the lower Snake River ranges between 6 and 9 (IDEQ 2004). Deposition of ammoniacal nitrogen to vegetation and soils normally occurs close to the source but may be especially enhanced by humidity along the Snake River and concentration of solutes in low hanging fog.

We conclude that ammoniacal nitrogen in HCNRA is well above natural background ranges for the western US, and poses a threat to the integrity of natural and cultural resources, especially along the valley floor. Better controls on agricultural emissions and enforcement of existing water quality standards are needed to protect these precious resources.

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Agricultural Air Quality Perspectives

Overview of the National Air Emissions Monitoring Study (NAEMS)

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Abstract

Under the terms of EPA's animal feeding operations consent agreement (FR 70, 19, 4958-4977, Jan 31, 2005), agriculture is to fund and produce a national air emissions monitoring study to generate data that will characterize air emissions from all major types of animal feeding operations ("AFOs") in all significant geographic areas for each participating industry. The Agricultural Air Research Council (AARC) is a tax-free, non-profit industry organization set up to produce the study and administer the funds. EPA will supervise the study, analyze the monitoring results, and aggregate it with appropriate existing emissions data to create tools (e.g., tables and/or emissions models) that AFOs and regulators could use to determine whether they emit pollutants at levels that require them to apply for permits under the Clean Air Act or submit notifications under CERCLA or EPCRA. The swine, dairy, egg laying and broiler chicken industries have chosen to participate and have pledged funds totaling more than $15 million for the two-year study. A large team of government and university scientists and private sector experts worked for two years to select the instrumentation, protocols and quality assurance/quality control methods to be used across locations and animal species. In addition, there has been extensive internal review and input by representatives from the US EPA's Office of Enforcement and Compliance Assurance, Office of Air and Radiation, and Office of Research and Development. As recommended in the National Academy of Science 2003 report, "Air Emissions From Animal Feeding Operations," measurements also will be made during the NAEMS study to initiate a process-based consideration of the entire animal production process and its effects on air emissions. This will include continuous measurements of animal age and weight gain throughout the study; diurnal animal activity levels; manure management/handling practices; animal feeding schedules; lighting, heating and cooling schedules; fan operation; floor and manure temperatures; inside and outside air temperatures and humidity; wind speed and direction; solar radiation; feed and water consumption; manure production and removal schedules; swine mortalities; animal production schedules; and analyses for total nitrogen and sulfur in feed, water, and manure. These NAEMS observations, plus those of parallel studies funded by the dairy and swine industries will lay the groundwork for developing the more process-related emissions models recommended by the NAS. There will be an immense increase in scientific knowledge generated from the NAEMS, and experts involved are convinced that significantly increasing the number of farms to be monitored would be prohibitively expensive and would not add substantially to the value of the data collected. As EPA completes its use of the NAEMS data, participating scientists and universities will be free to publish what is likely to be a large number of scientific journal articles from the air emissions and process-based observations collected during this study, as well as papers on the methods used and models tested. The equipment purchased for the study will be made available through AARC members for other studies following the NAEMS.

Agricultural Air Quality at NRCS

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Overview

Since 1935, the Natural Resources Conservation Service (NRCS, originally the Soil Conservation Service) of the U.S. Department of Agriculture (USDA) has worked with America's private land owners and managers to conserve soil, water, and other natural resources. More recent emphasis on atmospheric resources has expanded the Agency's and the USDA's activities in this area. This emphasis is due in part to the following:

- a heightened awareness of agriculture's role in air quality,
- the need for improving the sustainability of agriculture through conservation and preservation of air quality and atmospheric resources,
- the management of the growing agriculture/urban interface,
- the need for the development of market-based incentives in environmental and conservation programs, and
- the air quality regulatory impacts felt by producers.

In response to NRCS's increasing role, and a need to clarify this role, we are continuing to develop Department-wide national atmospheric resource policy in collaboration with our colleagues at the Agricultural Research Service, the U.S. Forest Service, CSREES, and the Economic Research Service of the USDA.

Of the 79 resource concerns identified for conservation planning prioritization in the NRCS, 12 are in air resources. Agricultural emissions of criteria pollutants and other pollutants of concern tend to be more diffuse and variable than those from other industries; therefore, the means for reduction requires a unique set of approaches. Atmospheric resource considerations include three greenhouse gases (GHGs) which have agricultural connections—carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4). NRCS seeks to develop and help implement relevant agricultural air quality technologies and practices, while enabling producers to apply them to the environmental, societal, financial, and technical benefit of all parties. With field staff in almost every county in the U.S., NRCS employees provide technical assistance based on sound science and suited to a customer's specific needs. NRCS provides financial assistance for many of its voluntary conservation activities.

Research provides a fundamental basis for any potential reductions in agricultural atmospheric emissions. However, interpretation and incorporation of those discoveries into working practice is necessary for realizing that potential. The NRCS applies a cooperative approach, providing land owners and managers with the expertise, information, tools, and financial support to evaluate and implement responsible land management decisions, while incorporating local experience from partners and program participants.

NRCS scientists are reviewing and updating accepted conservation practices and other activities which can address air quality and atmospheric change issues for agriculture. These practices and activities are delivered to producer-cooperators through conservation planning assistance, and through specific financial and conservation assistance programs.

Agricultural Air Quality and Atmospheric Change Focus Areas

Six broad air resource issues have been identified as being of greatest importance at the present time to agriculture, in general, and for special emphasis in the NRCS:

- Particulate Matter (both coarse, between 2.5 and 10 micrometers in diameter, and fine, less than 2.5 micrometers in diameter)
- Ozone (O_3) precursors, most notably nitrogen oxides (NO_x) and volatile organic compounds (VOCs)
- Greenhouse gases (CO_2, N_2O, and CH_4)
- Ammonia (NH_3)
- Odors
- Chemical drift (primarily of pesticides and herbicides)

The following examines specific activities and practices encouraged by the NRCS to address each of these issues.

Particulate Matter

Specific activities in the NRCS to reduce particulate matter (PM) emissions have been heavily focused on coarse particles, but with increasing emphasis on fine particulates, due primarily to the increasing body of research evidence showing the greatest issues associated with fine PM. Many research studies indicate that traditional agricultural activities such as tillage and other field operations generate a greater proportion of coarse rather than fine particulates, though greater investigation of the role of agricultural emissions in primarily, as well as secondarily, emitting $PM_{2.5}$ is just now being pursued in earnest.

Some examples of NRCS activities in reducing PM emissions include:

- Wind erosion estimation and control. The Wind Erosion Prediction System (WEPS) model, being developed by the USDA Agricultural Research Service (ARS), is used to predict particulate matter emissions and transport from wind erosion based on local conditions, environment, and management. WEPS includes consideration of field-scale variability, topography, and field geometry when making wind erosion estimates.
- Smoke management of prescribed burning. The NRCS works with state and local agencies to adopt and implement Smoke Management Programs (SMPs) designed to allow the use of fire as an accepted management practice, while protecting public health and welfare by mitigating impacts on resources of concern. The NRCS is also updating the prescribed burning practice standard to include smoke screening procedures effective at mitigating smoke impacts and is also partnering with other federal agencies (such as the Forest Service) to access smoke management tools (such as BlueSkyRAINS) that can be applied to agricultural smoke issues.
- Residue management, cover crops, wind barriers, implementation of a prescribed burning management plan, and alternate slash disposal methods represent a few sanctioned practices which may be applied to reducing a farm's particulate matter emissions.

Ozone Precursors

- Reducing the formation of tropospheric ozone through managing emissions of its direct precursors, including volatile organic compounds (VOCs) and nitrogen oxides (NO_x). Integrated pest management and comprehensive nutrient management planning for VOC emission reductions, and minimizing fuel combustion via reduced tractor operation (from implementing conservation tillage measures) or replacing fossil-fuel fired irrigation engines with electric motors for NO_x are examples of effective measures taken by producers to address ozone precursors.

Greenhouse Gases

There are three greenhouse gases of interest to American agriculture and NRCS: carbon dioxide (CO_2, from soil tillage, burning, and vehicle emissions), nitrous oxide (N_2O, from soil applied nutrients), and methane (CH_4, from animal production and waste management). Agricultural practices can help to lower atmospheric concentrations of these gases by:

- Sequestering carbon in soils by reducing soil tillage and disturbance

- Managing nitrogen fertilization to minimize N_2O emissions from the soil to the air
- Managing animal feeding and manures to minimize CH_4 release from production facilities.

Ammonia

Ammonia emissions (and consequent formation of fine particulate matter) may be reduced through

- Specific activities—such as splitting fertilizer applications through time or injecting fertilizers, as well as instituting an approved feed management system—demonstrate reduced ammonia emissions from agricultural operations.
- Trees that thrive on ammonia rich environments have been found to be useful in tree breaks in front of exhaust fans because they mitigate ammonia movement off-site.

Odors

- Odors from animal production and waste management, utilization, and disposal may be mitigated by activities, such as feed management and proper manure handling, storage, and processing.
- Trees may be used as windbreaks to reduce odors from animal production facilities, filtering mechanisms may be fitted onto animal buildings, or waste injection may be used, to list a few examples of effective agricultural odor emission reduction practices.

Chemical Drift of Pesticides and Herbicides

- Effective methods of reducing chemical drift from agricultural operations include the use of tree windbreaks to reduce chemical drift from fields, use of proven adjuvants to reduce overall application levels, and sensitivity to local meteorological conditions that impact volatilization characteristics.

NRCS Technical Documents

The basis for NRCS conservation practices and activities has always been based upon sound science gleaned from documented and well-respected research. Several technical document pathways exist in the Agency, establishing for NRCS personnel, as well as for the general public, the procedures that should be followed for conservation planning. With regard to air quality and atmospheric change which are relatively new areas of focus for the NRCS, new technical guidance is just now being developed and integrated into Agency procedures. These include;

The electronic Field Office Technical Guide (e-FOTG) is the primary scientific reference for NRCS, and is available online through NRCS's website. Guides are tailored to specific geographic areas, and contain technical information about the conservation of soil, water, air, and related plant and animal resources for those areas. The e-FOTG contains numerous categories of information useful for land managers and NRCS advisors, including:

- Site specifications, including NRCS Soil Surveys, Hydric Soils Interpretations, Ecological Site Descriptions, Forage Suitability Groups, Cropland Production Tables, Wildlife Habitat Evaluation Guides, Water Quality Guides, and other related information.
- NRCS Quality Criteria, which establish standards for resource conditions that help provide sustained use.
- NRCS Conservation Practices and Conservation Specifications, which define useful conservation practices and their installation requirements. The effect different practices may have on resources of concern is described by the e-FOTG.

National Engineering Handbook chapters dealing with specific agricultural air quality issues are being developed. The first chapters in final review will provide information on Odors, Tropospheric Ozone, and Greenhouse Gases. These chapters will provide NRCS personnel with a concise, scientific overview of each subject, along with recommended conservation procedures that will help address each of these issues.

NRCS is currently updating, and preparing in some cases, the air quality sections of the National Environmental Compliance Handbook. This handbook provides guidance to NRCS personnel about how to comply with federal environmental requirements when delivering technical and financial assistance.

A review of existing NRCS National Conservation Practice Standards and Conservation Practices Physical Effects (CPPE) has also begun in order to incorporate relevant and current information on agricultural air quality and atmospheric change issues. NRCS Practice Standards are technology or practice-specific documents that contain information on why and where the practice is applied, and sets forth the minimum quality criteria that must be met during the application of that practice in order for it to achieve its intended purpose(s). The CPPE documents provide guidance on how the application of that practice will affect the resources (soil, water, air, plants, animals and human) and the resource concerns associated with each of those resources. NRCS is also working to quantify the anticipated atmospheric improvement resulting from conservation efforts, including those from specific practices.

NRCS Agricultural Air Quality Tools

NRCS currently provides two tools to land managers and their advisors which maximize the conservation benefits of their practices. The tools are affixed with user-friendly interfaces, allowing many kinds of people access. NRCS is also reviewing opportunities to develop other tools that address air quality resource concerns.

The Wind Erosion Protection System (WEPS) is a process-based, continuous modeling system that employs site-specific soil, climate, wind, and other environmental conditions to predict wind erosion and the utility of practice standards for a given site.

The **V**oluntary **R**eporting of Greenhouse Gases-**C**arb**O**n **M**anagement **E**valuation **T**ool (**COMET-VR**) is a decision support tool for agricultural producers, land managers, soil scientists and other agricultural interests. Users of COMET-VR specify a history of agricultural management practices on one or more parcels of land. The results are presented as ten year averages of soil carbon sequestration or emissions with associated statistical uncertainty values. Estimates can be used to construct a soil carbon inventory for the Department of Energy's voluntary 1605(b) program.

NRCS Programs

NRCS operates a number of conservation programs that provide land owners and managers with technical and financial assistance on private lands to assist with the implementation and demonstration of developed and new conservation practices. The main programs that address agricultural air quality are the Environmental Quality Incentives Program (EQIP), which includes the Conservation Innovation Grant (CIG) program, and the Conservation Security Program (CSP).

Environmental Quality Incentives Program

EQIP provides a voluntary conservation program for farmers and ranchers that promotes agricultural production and environmental quality as compatible national goals. EQIP offers financial (cost-share of up to 75 percent of the costs of certain conservation practices) and technical help to assist eligible participants install or implement structural and management practices on eligible agricultural land. In 2005, NRCS awarded $31 million in EQIP contracts that listed air quality as either a primary or secondary concern.

EQIP activities are carried out according to an environmental quality incentives program plan of operations developed in conjunction with the producer that identifies the appropriate conservation practice or practices to address the resource concerns. The practices are subject to NRCS technical standards adapted for local conditions, and the local conservation district approves the plan.

Conservation Innovation Grants

The CIG program is authorized under EQIP as a voluntary program intended to stimulate the development and adoption of innovative conservation approaches and technologies while leveraging Federal investment in environmental enhancement and protection, in conjunction with agricultural production. Under CIG, EQIP funds are used to award competitive grants to non-Federal governmental or non-governmental organizations, Tribes, or individuals. CIG enables NRCS to work with other public and private entities to accelerate technology transfer and adoption of promising technologies and approaches to address some of the Nation's most pressing natural resource concerns.

NRCS awarded $1.6 million in 2005 for five CIG air quality projects. An additional eight awarded CIG projects address air quality as a secondary concern.

Conservation Security Program

CSP is a voluntary program that that supports ongoing stewardship of private agricultural lands by providing payments for maintaining and enhancing natural resources. CSP provides financial and technical assistance to promote the conservation and improvement of soil, water, air, energy, plant and animal life, and other conservation purposes on Tribal and private working lands. CSP identifies and rewards those farmers and ranchers who are meeting the highest standards of conservation and environmental management on their operations. In fact, the CSP motto is to "reward the best and motivate the rest."

NRCS has developed over 35 job sheets and worksheets to address the six principal air quality issues mentioned previously in this paper. These documents outline the different air quality enhancements that are eligible for payment under CSP, such as conservation tillage, windbreaks, and management of prescribed burning to address particulate matter issues and manure fertilizer injection and feed and manure management to address odor issues.

The USDA Agricultural Air Quality Task Force

The Chief of NRCS is the Chairman of the Agricultural Air Quality Task Force (AAQTF), a Federal Advisory Committee composed of individuals representing a diverse range of expertise. Recent advice to USDA as a result of this group has included:

- research priorities,
- policy on voluntary market-based incentive programs,
- agricultural burning,
- atmospheric emissions and impacts from agricultural operations, and
- other air quality related regulatory issues.

Summary

The mission of the NRCS is to provide leadership in a partnership effort to help people conserve, maintain, and improve our natural resources and environment. As such, we share a commitment to the conservation and preservation of air quality and atmospheric resources, especially as it relates to agriculture. We will continue to assist land owners and managers with technical and financial assistance on private lands to accomplish this goal and to identify, develop, promote, and implement agricultural practices and technologies that help people help the land.

For more and updated information on NRCS agricultural air quality activities, including links to the AAQTF web site, and specific programmatic information, please visit: http://www.airquality.nrcs.usda.gov/

Use of Collaborative Partnerships to Address Environmental Impacts of Agriculture

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Abstract

Environmental management in the United Stated has become an increasingly polarized process. Our decision-making is typically founded on principles of science, policy and law. Each circumstance can lead to one or more of the three disciplines dominating the others. As this occurs, the process can fracture. This is especially true if a dispute arises and the matter is "decided" by a court. The legal process creates winners and losers and often simply kicks the matter back to the same parties for solution. In contrast, collaborative processes can often provide more effective and lasting environmental results since the would-be adversaries become jointly responsible for solutions. A variety of successful partnerships have been implemented in the West in recent years. The Western Regional Air Partnership built strategies that have significantly reduced regional sulfur dioxide emissions that create haze in national parks. In March 2001, the Utah Concentrated Animal Feeding Operation (CAFO) Committee finalized a strategy to rapidly assess and mitigate impairment of water bodies by CAFOs due to non-point source water pollution. The Committee was a partnership comprised of the Utah Department of Environmental Quality, Utah Department of Agriculture and Food, Utah Farm Bureau, Utah Association of Conservation Districts, and various grower groups. The strategy was noteworthy for its innovative use of all partners to deal with the problem instead of the traditional practice of regulators writing permits, inspecting farms, and issuing violations for noncompliance. The results were stunning. Assessments were accomplished and permits issued much faster than other state programs. More importantly, farmers implemented best management practices (BMP) promptly with nitrogen, phosphorous, and BOD loadings reduced 95% by 2005. A similar partnership has now been formed to assess and mitigate agricultural air emissions in Utah. Most Utah farmers did not feel the proposed USEPA national monitoring settlement was useful to them. It was costly and would not yield data relevant to arid agricultural practices in the intermountain West. The AFO Committee created an air strategy that was formalized through a Memorandum of Understanding with USEPA Region VIII in August 2005. The MOU contains commitments for timely monitoring using approved protocols and quality procedures. A major focus of the Utah program is evaluation of BMPs so solutions can be accelerated and multimedia impacts evaluated. The Utah partnership stands in sharp contrast to the far more contentious national effort that has limited emphasis on BMPs and cross-media impacts.

Introduction

Agriculture, especially CAFOs, have come under increased scrutiny in recent years as sources of air pollution and the United States Environmental Protection Agency (USEPA) and state regulatory agencies are taking steps to evaluate the air quality impact of these activities. The USEPA initiated a national CAFO emissions inventory program using a consent decree process through the agency's Office of Enforcement and Compliance Assurance (OECA) (Federal Register, 2005a). The approach has been criticized as being too lax by some states and environmentalists and too heavy-handed by farm interests. Such polarized outcomes have become increasing common in air quality management and can stymie timely and effective results on the ground. Solutions that rely on greater collaboration and outcome responsibility by all stakeholders can be more effective than the traditional command-and-control and litigation-centric system of the last 30 years. This paper examines some principles for successful collaboration in environmental policy-making and describes examples where the principles have been successfully applied in the western United States.

Methods

Environmental Policy Making through Collaboration

The approach described here (Nielson, 2004) is based on the environmental policy-making principle of "Science for Facts, Process for Priorities" of the *Enlibra Principles* adopted by the Western Governors' Association (WGA, 1998) and the National Governors' Association (NGA, 2000). One may view environmental management as a graphical representation of an equilateral triangle with the vortices being Science, Law, and Policy (Figure 1). When applied in balance, the three forces can provide effective protection of natural resources. If one or more of the forces is used to dominate the others, the imbalance fractures our ability to effectively manage environmental protection. Different management activities require appropriate use of only parts of the triangle.

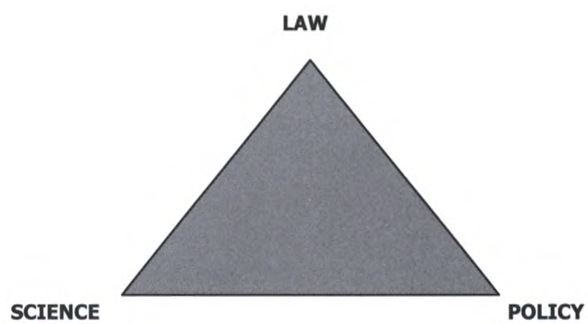

Figure 1. Environmental Policy-Making
(based on the Principle of Science for Facts, Process for Priorities)

Standard setting involves interplay between Science and Policy with Science forming the basis for the decisions (Figure 2).

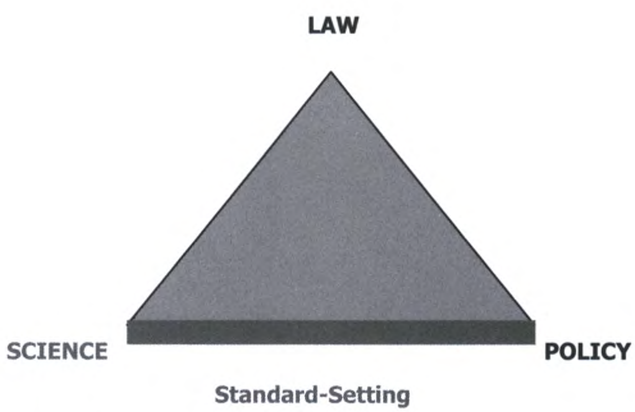

Figure 2. Environmental Policy-Making, Standard Setting

An ideal situation is represented in Figure 3 that portrays dose-response data for substances A and B. Analytical methods exist to detect even minute concentration of both compounds. Compound A has no lower limit of health effects or no no-observable-effects-level (NOEL). Compound B does have a NOEL as seen with the flat curve for health impacts at low concentrations. Science would inform us that there is not safe level of compound A and that there is some safe concentration of compound B that can be tolerated without health impacts.

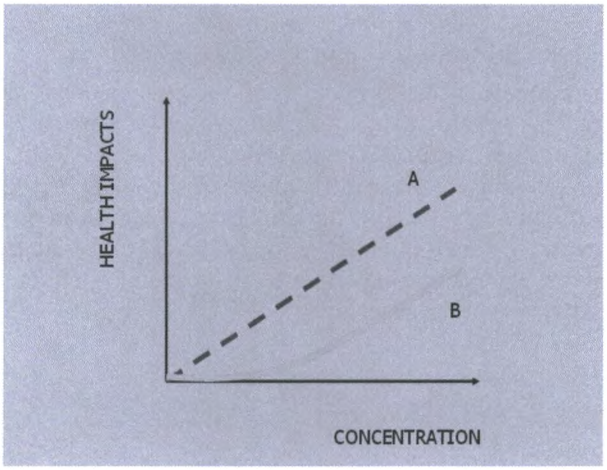

Figure 3. Standard Setting, Science Only

Sometimes the standard will be informed by Policy (Figure 4). If the pollutant has a detection limit, science may not be able to find a NOEL concentration for Compound A. An informed policy decision may determine that the standard should be at some concentration above zero pending more research or better analytical equipment.

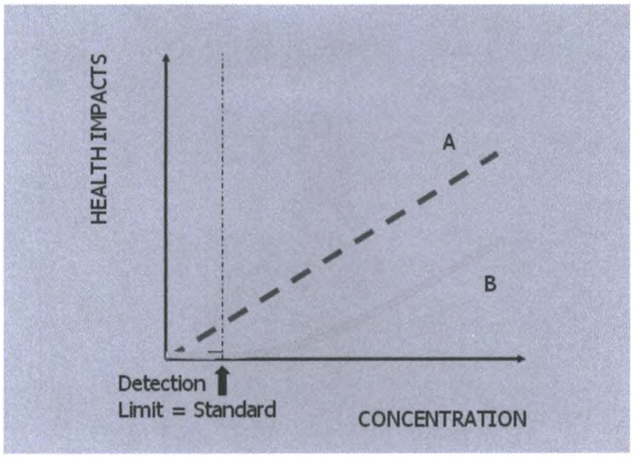

Figure 4. Standard Setting, Science and Policy

Rulemaking is a balance of Science and Law (Figure 5).

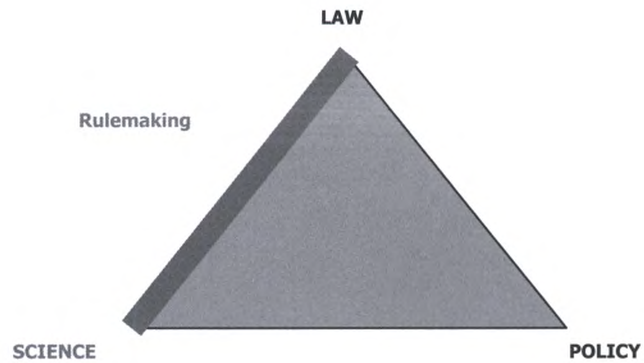

Figure 5. Environmental Policy-Making, Rule Making

A recent example of this process is the Clean Air Act requirement for USEPA to reduce hazardous air pollutants from fossil-fueled power plants (United States Code 1990a). Power plants are a significant source of air pollution (Figure 6). In 2000, the power sector generated 63% of sulfur dioxide and 22% of oxides of nitrogen in the United States (USEPA 2003 and Chu et al 2001). In 1999, power plants were responsible for an estimated 40% of 78 tons of mercury emissions that year.

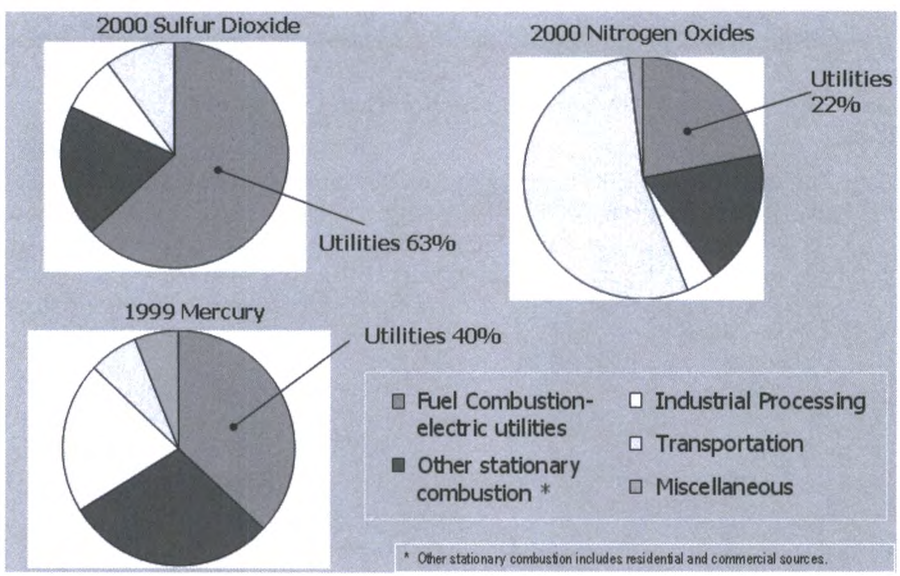

Figure 6. Coal-fired Power Plant Emissions

Mercury chemistry and control technology are complex and limited data was available as the agency sought to promulgate a rule. A stakeholder workgroup was chartered under the auspices of the Clean Air Act Advisory Committee (CAAAC). The workgroup was unable to report back with complete consensus, but it did provide a range of recommendations for a maximum achievable control technology (MACT) rule that reflected the significant uncertainty of the science as well as the control equipment engineering and economics. (Amar 2002) The outcome of this process will be examined further below.

Planning is a crucial function of environmental management and the disciplines of Law and Policy dominate this process (Figure 7).

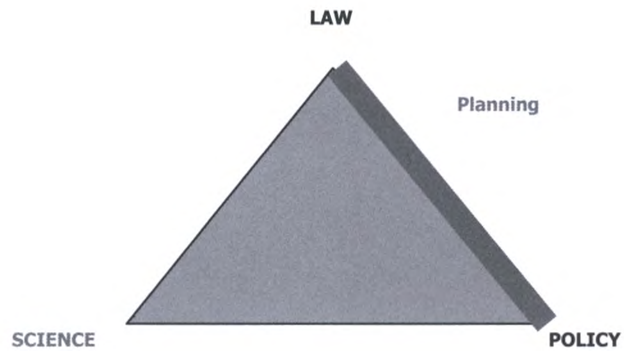

Figure 7. Environmental Policy-Making, Planning

Transportation planning provides a good example of traditional approaches. The federal statues and rules that set forth the process for transportation planning create a complex system of steps for states and metropolitan planning organizations (MPOs) that leads to a long range transportation plan with a 20-year horizon and a state transportation implementation plan with a shorter horizon for funding and design of specific projects (Code of Federal Regulations 2001). The process looks at the present and sets forth a single plan for the future, one forecast – one solution. The process can constrain outcomes.

The current model for environmental policy-making and problem solving is defined by a number of characteristics, almost all negative. Starting points for stakeholders are often defined as *extremes*. Participants use emotional symbols. Processes become *weapons*, not *tools*. Lawyers, lobbyists, dueling scientists and public relations firms become "proxy stakeholders" hired by the real interested parties. The result can be years of costly and divisive conflict. Even worse is the lack of real environmental progress as the drama unfolds.

A recent example introduced above is the mercury rule for power plants. The USEPA finally promulgated a mercury rule in 2005 (Federal Register 2005b) that disregarded the CAAAC's report and infuriated many stakeholders (Becker 2005). Not surprisingly, the agency was sued by multiple parties (USEPA 2005). The case is pending before the US Court of Appeals for the District of Columbia. This course of events has become the norm and might be represented graphically by a 3-D figure in which the triangle has grown vertically with the Courts now at the pinnacle and responsible for the outcome (Figure 8).

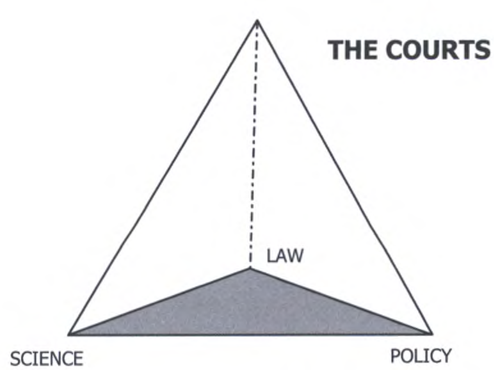

Figure 8. Environmental Policy-Making, Courts Decide

Ironically, the Court regularly throws the controversy back to the same protagonists for resolution through remands (Federal Register 2004). Even if a court's opinion is "decisive" and a rule is vacated, the original parties often just get to start all over to solve the problem.

A more productive approach uses the same three elements of Science, Law, and Policy in a more balanced and respectful manner in a stakeholder process (Nielson, 2004) under the *Enlibra* principle of "Science for Facts, Process for Priorities." This is a process that allows (even requires) stakeholders to take responsibility for solutions, not just their position (Figure 9). Under this approach, participants separate subjective choices from objective data gathering. The group must also avoid polarized positions backed by selective scientific "facts" that do not allow for environmental problem solving. A key component discussed further below is incorporating Science into the Policy-making process. Finally, communicate, communicate, communicate. Stakeholders need to agree to and take responsibility for these principles as a condition of participation.

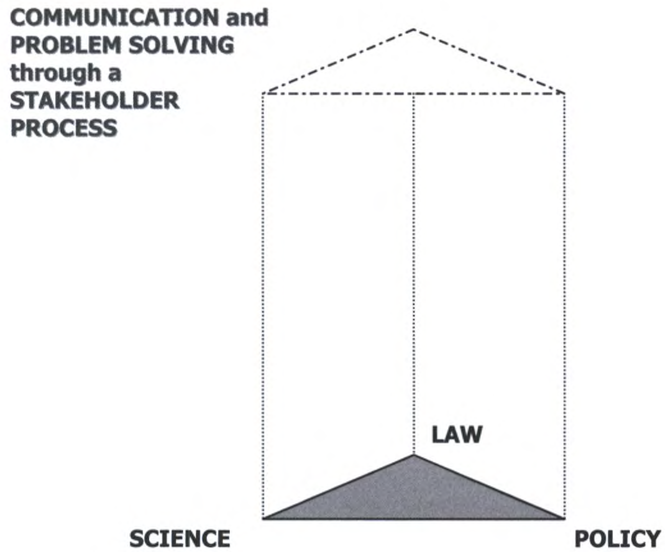

Figure 9. Environmental Policy-Making, Shared Responsibility for Outcomes

The method of incorporating Science into the Policy-making process warrants further examination. First, the process needs to include a *range* of respected scientists who are independent of pre-existing stakeholder positions. A peer review of scientific findings should be undertaken to the maximum extent possible to ensure that *all* issues are addressed appropriately. The group must make decisions based on the best information available; uncertainty can rarely, if ever, be eliminated. The public must be educated about the science used to build credibility and acceptance of any decisions that are eventually made. Once decisions are made, the group has to monitor the outcomes to ensure the desired results or modify the course of action appropriately.

Collaborative processes may not be appropriate or successful in all settings, but partnerships have provided solutions for surprisingly contentious situations. Examples are described below as well as a collaborative strategy for addressing air quality impacts from agriculture.

Results and Discussion

Regional Haze and the Western Regional Air Partnership (WRAP)

Regional haze in the West is an example where collaboration was used to achieve at effective rulemaking. The 1990 amendments to the Clean Air Act established a requirement to reduce regional haze on the Colorado River Plateau and the Grand Canyon Visibility Transport Commission (GCVTC) was established to determine solutions (United States Code 1990b). The Commission issued its consensus report in June 1996 (Grand Canyon Visibility Transport Commission. 1996). Crucial to this effort was the Commission's Public Advisory Group comprised of a diverse set of stakeholders representing states, tribes, federal agencies, local government, academia, media, industry, and public interest groups. It was this group that hammered out consensus for the report. Ironically, the USEPA omitted the GCVTC's recommendations from the proposed regional haze rule (Federal Register 1997). The strength and value of consensus among a large and diverse stakeholder group quickly convinced the agency to embrace the Commission's report. The USEPA adopted the recommendations in section 309 in its final regional haze rulemaking (Federal Register 1999).

The Western Regional Air Partnership (WRAP) was formed from many of the same stakeholders to develop implementation strategies for the rule. The work of the GCVTC and WRAP provided a suite of flexible alternatives to the more traditional regulatory approach of section 308 of the haze rule. The keystone provision was a backstop sulfur dioxide trading program for major sources of sulfur dioxide. Regional annual emission milestones were established for all major sulfur dioxide sources, not just coal-fired power plants. This program and others would achieve significant reductions of pollution causing haze in western Class I national parks and monuments while providing great flexibility to industry and regulating authorities. Extraordinary effort was made to recognize the undeniable role coal would play in future electrical generation while reducing emissions. Five states (Arizona, New Mexico, Oregon, Utah, and Wyoming) submitted state implementation plans under section 309 by December 2003. Despite the enormous effort to gain consensus, the section 309 rules were repeatedly litigated by the Center for Energy and Economic Development (CEED) and others; various parts of the rule were subsequently vacated or remanded to the USEPA (American Corn Growers vs. EPA 2002, Center for Energy and Economic Development vs. EPA 2005). Remarkably, most stakeholders, including regulated industry, remain committed to the strategy conceived by the GCVTC. Sulfur dioxide from coal-fired power plants was reduced by 202,500 tons per year or 35 percent by 2004, already ahead of the required milestones. Reductions by 2018, the end of the first planning period, are expected to be 325,000 tons or 60 percent (Cummins 2005). The GCVTC recommended a 50-70% reduction of sulfur dioxide from all major sources from 1990 levels by the year 2040. Considering all major sulfur dioxide sources (power plants, smelters, refineries, cement plants, etc.), there has already been a 40 percent reduction from 829,000 tons to 501,000 tons in 2004. While legal battles and their fallout create uncertainty, air quality progress continues, not because of rules, laws, or threats of enforcement, but because of the power of the collaborative partnership has been sustained.

Envision Utah and Transportation Planning

As discussed above, environmental planning often involves the law and policy, but traditional methods can result in only one outcome that may not be acceptable to all parties. The Legacy Highway in Utah is such an example. In 1996, the Wasatch Front Regional Council and Utah Department of Transportation expedited planning for a new freeway connecting Salt Lake County to Davis County to the north to relieve rush hour congestion and provide an alternative to Interstate Highway 15 as the only major north-south highway in that corridor. The proposal was very popular with Davis County residents and appeared headed for quick funding and construction. However, there were others who felt the project promoted urban sprawl, threatened wetlands, and would damage air quality. They argued that all corridor and mass transit alternatives were not considered in the environmental impact statement. These parties ultimately were granted an injunction halting construction in November 2001 from the 10th US Circuit Court of Appeals just as the project got underway. In late 2005, the parties and the State sealed an agreement that would allow construction of a more modest roadway with less environmental impact. Construction is scheduled to resume in March 2006 (West, 2006). The delay cost Utah taxpayers an estimated $220 million (Buttars, 2005). Further, there has been a strong backlash against the Sierra Club, Salt Lake City Mayor Rocky Anderson and others who participated in the legal action in the form of emotional statements in the media, various retaliatory tax proposals (Ewing, 2002) and proposed legislation to require expensive bonds before a stay could be requested (Tilton 2006). The result of poor initial collaboration by meeting only the minimum consultations required by law proved both costly and divisive.

By contrast, Envision Utah, a public-private partnership formed to seek alternative visions for Utah's future, undertook the largest public outreach program in the history of Utah. The concept was simple: Ask Utahns what they want their communities to look like many years into the future by providing a choice of outcomes. The effort first sought input of what issues mattered. It then convened hundreds of focus groups with maps and markers to actually draw out preferred alternatives. Finally, a set of four alternatives were presented through thousands of newspaper inserts and the internet to allow people to "vote" on major decisions for transportation, water, housing, air quality, and other matters. The alternatives were displayed graphically and with costs so it was easy to understand the choices. The results have been widely portrayed and used by a wide array of municipal, regional, and state officials, business leaders, and the public ever since.

A further outgrowth of Envision Utah and the failure of the Legacy Highway process has been a very different approach to transportation planning along the Wasatch Front. The Wasatch Front Regional Council and Mountainlands Association of Governments united to work with Envision Utah to develop a new vision that embraced more views and a broader geographic area (Wasatch Front Regional Council 2005). Utah Department of Transportation, the Utah Transit Authority, the metropolitan planning organizations, state and federal environmental and natural resources agencies have also formed an Executive Planning Team. These same groups have invited public interest groups to chart the course for future transportation infrastructure. The result is that planning a new major beltway and significant transit projects are progressing without the highly charged atmosphere that accompanied the Legacy Highway.

Utah Strategy for Water Pollution from Animal Feeding Operations

In 1999, the United States Department of Agriculture (USDA) and the USEPA released a joint unified strategy to address runoff from animal feeding operations (AFO) which allowed for a certain amount of flexibility in detail by individual states. Following the release of the national strategy, the Utah Department of Environmental Quality (UDEQ), Division of Water Quality (DWQ) organized the Utah AFO Committee to develop a workable strategy for Utah. In March 2001, the Utah AFO Committee finalized a strategy to rapidly assess and mitigate impairment of water bodies by CAFOs due to non-point source water pollution (Utah AFO Committee 2001). The Committee was a partnership comprised of the Utah Department of Environmental Quality, Utah Department of Agriculture and Food, EPA, USDA Natural Resources Conservation Service (NRCS), Utah State University Extension Service, Utah Farm Bureau Federation, Utah Association of Conservation Districts, and various grower groups. The strategy was noteworthy for its innovative use of all partners to deal with the problem instead of the traditional practice of regulators writing permits, inspecting farms, and issuing violations for noncompliance. A schematic of the process is shown in Figure 10.

The partnership approach allowed the producers to work closely with organizations with which they had an established relationship of trust. The knowledgeable committee members were able to provide information about animal waste management designs that were affordable and proven to be effective in preventing runoff into the waters of the State. The committee developed training materials and provided 19 manure management workshops throughout the state and 10 workshops to develop and write Comprehensive Nutrient Management Plans (CNMPs) (Loveless et al., 2004).

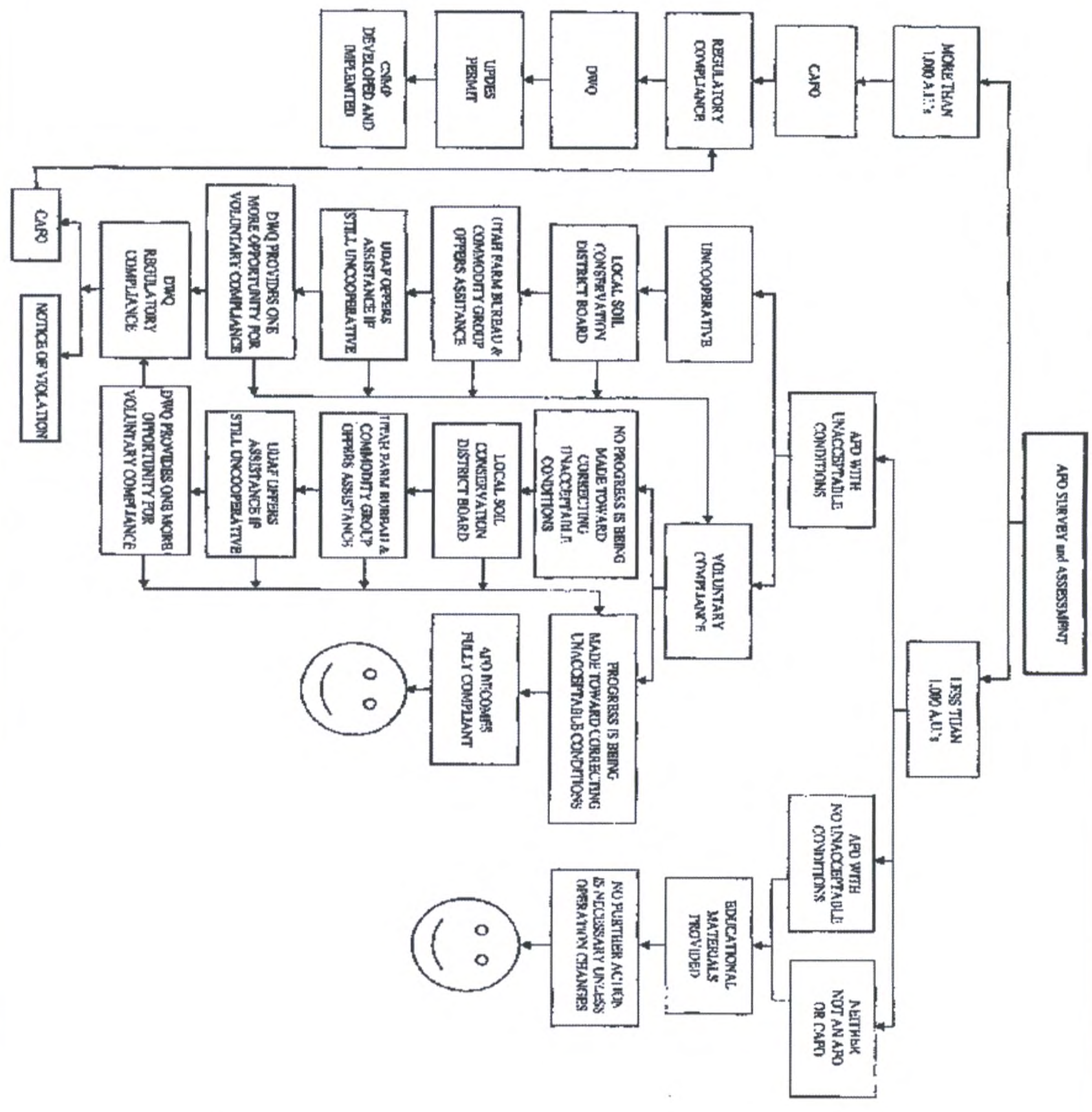

Figure 10. Utah AFO Water Quality Strategy Process

The producers also had an understanding of runoff impacts and had a similar concern for protecting the waters in the areas in which they live. With the shared goal of improving the quality of the water resources, regulators were able to understand the other concerns of the producers and help them secure loans and grants to fund construction projects while surviving in a low profit margin industry. By 2004, more than $7.1 million in federal and state funds have been utilized to further the program in the State of Utah (Loveless et al., 2004).

The biggest difference in the Utah strategy is the "potential" CAFO (PCAFO) designation. This allows smaller operations with runoff problems a flexible window of opportunity to fix problems and come into compliance while still qualifying for federal funds, an opportunity that regulated operations do not receive.

Another hallmark of the Utah program was its approach to protecting confidential grower information. Data on individual farms are closely held by the growers as well as the USDA Agricultural Statistics Office and proved to be among the most difficult issues. Ultimately, the Utah Farm Bureau Federation served as an intermediary for sensitive data and were held responsible for reporting to State regulators. This approach made regulators uneasy, but the State and USEPA Region VIII eventually agreed to move ahead and see if such a non-traditional strategy would accomplish the ultimate environmental results.

The gamble paid off handsomely. Even before the completion of the Utah AFO Strategy, work began under the direction of the Utah Farm Bureau Federation and the Utah Association of Conservation Districts to conduct on-farm assessments of every animal feeding operation in the state. By 2004, essentially all of the 2915 on-farm AFO assessments were complete. In contrast the State of Colorado reported that by 2004, 40 farms had been inspected under a traditional regulatory program (Colorado Department of Public Health and the Environment 2004).

The committee determined that 2056 of the 2915 AFOs (71%) had no water quality problems. They identified 58 CAFOs and 392 PCAFOs for a total of 450 sites. Of these, 391 have completed CNMPs, 234 have controlled runoff, 220 reduced runoff by implementing CNMPs and fell below PCAFO thresholds, and 264 are in full compliance (Petersen 2006). An example of one of hundreds of projects from the program is shown in Figure 11. The project consisted of manure bunkers and evaporation pond in the foreground plus a buffer zone before a stream in the distance.

Figure 11. Lane Sorenson Farm, Sanpete County, Utah

Assessments were accomplished and permits issued much faster than other state programs. More importantly, farmers implemented best management practices (BMP) promptly with nitrogen, phosphorous, and Biological Oxygen Demand (BOD) loadings reduced 95% by 2005 (Loveless et al., 2004, Petersen 2006). These goals were accomplished with minimal increases in the size of the regulatory agencies charged with implementing the new federal programs.

Controlling Air Pollution from AFOs: The USEPA AFO Compliance Agreement Proposal

On January 21, 2005, EPA announced a program for farmers to participate in an Animal Feeding Operations Air Quality Compliance Agreement to resolve non-compliance and fund air monitoring of AFOs (Federal Register 2005). The USEPA strategy began with an enforcement-based action to generate funds for a monitoring program from penalties and fees. The agency hopes to monitor 28 sites throughout the nation for swine, poultry, and dairy cows. The results will be used to calculate emission factors based species and other parameters that might affect emission rates. The emission rates could then be used to determine if farms comply with the Clean Air Act Operating Permit and New Source Review (NSR) permitting programs, Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) reporting, and the Emergency Planning and Community right-to-Know Act (EPCRA) reporting.

Since agricultural emission factors can be highly variable, actual emissions are not generally well defined so most operations do not know whether they are in compliance with federal environmental laws or not. The USEPA solicited farmers to pay fees and penalties regardless of their ultimate compliance status to provide protection from future enforcement action based on past emissions.

The federal enforcement-centric plan suffers from some other shortcomings. It does not include a plan to reduce emissions, measure reductions, or determine Best Available Control Technologies (BACT) or Best Management Practices (BMP) as substantive objectives of the study. The key objective is to find out who is or is not in compliance. Based on the water quality program results, meaningful emissions reductions could take 5-10 years. The single media focus could hamper or reverse successful water quality and animal waste management efforts. Finally, the proposed USEPA Consent Order creates an adversarial relationship symbolizing a breakdown of the "policy-making triangle" rather than a problem solving partnership that promotes more lasting strategies. Not surprisingly, the question of confidential grower information is also a major issue in the national program. Many states and environmentalists are suspicious when critical data is not transparent so it will be interesting to see how USEPA finally addresses this concern. Achieving the same degree of trust on a national scale is far more difficult than in a single small state. This might suggest there is an opportunity for other states to pursue their own tailored strategies to avoid or minimize some of the challenges of a single national effort. It may also be easier to produce a strategy at the state scale that minimizes economic impact on agriculture while achieving equivalent or superior environmental results.

Utah's Air Quality Requirements

The State of Utah has been delegated authority to implement all programs allowed in the Clean Air Act. Key air programs with possible applicability to agriculture are the National Ambient Air Quality Standards (NAAQS), NSR Prevention of Significant Deterioration of Air Quality (PSD) permitting program, and the Operating Permit program. In addition, Utah has a minor NSR program for sources with less than 250 tons of emissions per year that are not regulated by the federal PSD program. The minor NSR program requires Best Available Control Technology on all sources not directly regulated by any federal air quality standard with actual emissions more than 5 tons per year of sulfur dioxide (SO_2), carbon monoxide (CO), nitrogen oxides (NO_x), particulate matter (PM_{10} and $PM_{2.5}$), volatile organic compounds (VOC), or more than 500 pounds of any single hazardous air pollutant or 2000 pounds of any combination of hazardous air pollutants (Utah Code Annotated 2005).

The very low permitting thresholds for the minor NSR program could likely ensnare a large number of small agricultural operations in a permitting system designed for industrial sources. Utah officials immediately recognized the opportunity to extend the successful Water Quality AFO partnership to address air quality requirements as a viable alternative to the normal NSR permitting process.

Utah Strategy for Air Pollution from AFOs

Most Utah farmers did not feel the proposed USEPA national monitoring settlement was useful to them. It was costly and would not yield data relevant to arid agricultural practices in the intermountain West since none of the 28 federal program sites would be in the region. The AFO Committee created an air strategy that was formalized through a Memorandum of Understanding with USEPA Region VIII in August 2005 (Roberts and Nielson 2005). The MOU contains commitments for timely monitoring using approved protocols and quality assurance procedures. A major focus of the Utah program is evaluation of BMPs so solutions can be accelerated and multimedia impacts evaluated. The Utah partnership stands in sharp contrast to the far more contentious national effort that has limited emphasis on BMPs and cross-media impacts.

The Utah Air Quality Strategy has the overall goal to meet the requirements of state and federal environmental regulations while maintaining a viable agricultural industry in the State of Utah. The State wanted to commence immediate steps to evaluate and implement agricultural practices that will minimize overall environmental impact similar to the AFO Water Quality Strategy by expanding the scope of Utah's existing Agriculture - Environmental Partnership. The air strategy needed to constructively engage EPA to augment, not compete with, the federal AFO program and assist in funding, evaluating, and approving options as a means of achieving national objectives in Utah. As with the national program, Utah's first objective was to rapidly assess air quality issues and the impacts of animal feeding operations. Since no federal monitoring sites were slated for the intermountain West, the State needed to determine how to measure or quantify emissions and harmonize the methods with the national program protocols. Utah also wanted to determine air quality and cross-media (water quality and waste) BMPs for species and type of operation as an integral part of the entire process. It was also essential to quantify results and measure progress in actual reductions in air contaminants released from the very beginning of the effort. This was a feature of the water quality strategy that jump started environmental results. It is believed that such early results and broader industry penetration could provide a measure of relief from litigation. All in all, the program was viewed as providing Utah growers with a viable alternative to EPA's national Consent Order although growers were free to participate in both programs. Concurrent with the program planning effort was a successful drive to secure funding for testing and to assist growers in implementing viable BMPs.

Based on the proceeding principles, a Utah AFO Air Quality Plan was developed with specific objectives, schedules and metrics. The plan's objectives are to:

1) Quantify air emissions from Utah's confined animal feeding operations. Specifically:
   a. Identify emissions: Ammonia (NH_3), Volatile Organic Compounds (VOC), Nitrogen Oxides (NO_X), Carbon Monoxide (CO), Hydrogen Sulfide (H_2S), Particulate matter (PM_{10} and $PM_{2.5}$), Hazardous Air Pollutants (HAPs) and other air contaminants.
   b. Identify species: Swine, dairy cows, poultry (turkeys, broilers, layers).
   c. Select monitoring locations that cover range of geographic and other factors.
   d. Determine relationships of emissions under various conditions for example determine if NH3 emissions increases when H2S in controlled.
   e. Select monitoring locations with a range of BMPs in place for water and waste management.
   f. Emphasize emission evaluations during valley wintertime inversions and sub-freezing temperatures.
   g. Write test protocols to include Quality Assurance plan and statistical methods.
   h. Conduct emissions testing.
   i. Report results.
2) Develop emission factors for species, physical management practices and BMPs.
3) Determine what BMPs could be considered BACT.
4) Identify where air quality BMPs could degrade water quality BMPs for waste and vice versa.
5) Determine operations that may be subject to CAA permitting (Approval Order or Operating Permit) and CERCLA/EPCRA requirements.
6) Develop multi-media implementation plan that:
   a. Starts with a voluntary, incentive-based approach,
   b. Meets CAA regulatory requirements, and
   c. Builds on the success of the Water Quality program to maximize Air, Water and Waste benefits.

With funding from State and Federal sources anticipated by March 1, 2006, purchase of equipment and field work is anticipated to begin by mid-2006. The Utah Division of Air Quality will be program manager with USU faculty serving as principal investigators. The program results will be further leveraged by two separate, but concurrent, investigations. USU scientists and engineers have spearheaded research to characterize unusually high PM2.5 levels in the Cache Valley where the university is located and is home to a significant dairy industry. In addition, ongoing national research by USUs Space Dynamics Laboratory to develop new state of the art analytical methods of for particulate measurements may further enhance program results.

Conclusions

Collaborative methods can often lead to better, more timely, and more durable environmental results than a traditional regulatory paradigm. Stakeholder processes can be time intensive upfront and may not always be fully successful if stakeholders are not committed to solutions, but they can be highly beneficial environmentally even if the ultimate legal outcome is in question. If the process does fails for a specific problem, it can lead to relationship that could be valuable in dealing with a future environmental impasse.

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Modeling Agricultural Air Quality: Current Status, Major Challenges, and Outlook

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Abstract

Agricultural air quality is an important emerging area of atmospheric sciences that represents significant challenges in many aspects of research including measurements, modeling, regulations, emission control, and operation managements. This work presents a review of current status, major challenges, and future research opportunities of agricultural air quality modeling.

Introduction

Current air quality research focuses largely on criteria pollutants such as nitrogen oxides (NO_x), ozone (O_3), and particulate matter (PM). Limited attention has been given to non-criteria air pollutants such as nitrogen- and sulfur-containing compounds from agricultural sources (e.g., ammonia (NH_3), nitrous oxide (N_2O), hydrogen sulfide (H_2S)). Agriculture provides a major source of those compounds. For example, 90% of the atmospheric burden results from animal production and emissions from slurries and manures in the U.S. (Davison and Cape, 2003) and many European countries (Sutton et al., 1995; Van Der Hoek, 1998; Hutchings et al., 2001; Sotiropoulou et al., 2004). Growing evidence has shown that the increased size and geographical concentration of animal-feeding operations (AFOs) and agricultural crop production are increasing the emissions of odor (e.g., organic acids) and trace gases (e.g., carbon dioxide (CO_2), methan (CH_4), NO_x, NH_3, and H_2S) to the atmosphere (e.g., Kurvits and Marta, 1998; NRC, 2003; Aneja et al., 2006). Increases in the emissions of those agriculturally emitted compounds in the U.S. and abroad and their adverse impacts on the quality of the air, water, soil, the biodiversity, and the entire agro-ecosystem have raised growing public and regulatory concerns. Those concerns have led regulators and policy makers from the U.S. and other countries to begin considering mitigation strategies for agriculturally emitted air pollutants. Regulations for NH_3 emission reductions from the livestock farming have been initiated and enforced in the Netherlands to meet stringent emission and deposition of NH_3 targets (Lekkerkerk, 1998). In the U.S., although there is currently no national ambient air quality standards (NAAQs) for those agriculturally-emitted air pollutants, a reporting requirement for the large released quantities of NH_3 and H_2S from AFOs has been enforced under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and the Emergency Planning and Community Right-to-Know Act (EPCRA), as part of the Clean Water Act (CAA). In addition, mitigation measures are being taken at a state level. For example, both Minnesota and Texas have state ambient air quality standards for H_2S; the North Carolina Environmental Management Commission is one of the first agencies in the U.S. to adopt rules for odor control from swine farms in 1999.

Advanced 3-D AQMs accounting for emissions, transport, transformation, and removal of air pollutants provide a powerful tool to simulate the fate, distributions, and impact of agriculturally-emitted air pollutants. National Research Council has clearly identified a need for such three-dimensional (3-D) transport/transformation models in providing scientific basis for the development of relevant mitigation strategies (NRC, 2003). The current status of agricultural air quality modeling and the future research needs and challenges are provided below.

Summary of Review Results

The development of feasible regulations of air emissions from AFOs requires a scientific basis that is currently lacking due largely to inadequate funding from governmental agencies and little attention from scientific communities for agricultural air quality research. Knowledge gaps and critical needs for agricultural air quality research have been recently identified by National Research Council (NRC, 2003) and the USDA Agricultural Air Quality Task Force (http://www.airquality.nrcs.usda.gov/AAQTF/). Significant uncertainties lie in nearly all aspects of research including the sparseness of monitoring stations and observational data of emissions, concentrations, and deposition fluxes, the lack of accurate emission inventories and reliable measurement methodologies, poorly-quantified health-effect associated with the AFOs-emitted species, and the need for process-based emissions models and 3-D transport/transformation models to support regulation and policy-making.

Current 3-D urban-to-regional air quality models (AQMs) are designed to simulate the sources, transport, chemical transformation, and removal of major criteria air pollutants such as sulfur dioxide (SO_2), NO_x, O_3, and PM and its composition (e.g., sulfate (SO_4^{2-}), nitrate (NO_3^-), ammonium (NH_4^+)), and the important gaseous precursors of O_3 and PM such as NH_3 and volatile organic compounds (VOCs). Among all agriculturally emitted compounds, NH_3 is the only species that is simulated in 3-D urban-to-regional AQMs but large uncertainties remain in its emission inventories, chemistry, and dynamic treatments. Current urban-to-regional AQMs do not treat other agriculturally emitted species such as N_2O and H_2S. The important processes of NH_3/NH_4^+ simulated in those AQMs include advection, diffusion, aerosol thermodynamics and dynamics (e.g., gas/particle partitioning, thermodynamic equilibrium reactions, condensation/evaporation, and coagulation), dry deposition, the dissolution in cloud droplets and rain water, and subsequent scavenging and wet deposition (e.g., Byun and Ching, 1999; Binkowski and Roselle, 2003; Zhang et al., 2004; Zhang and Jacobson, 2005; Zhang et al., 2006). Growing evidence has shown that NH_3 may play an important role in new particle formation through ternary nucleation involving sulfuric acid (H_2SO_4), water vapor (H_2O), and NH_3 (e.g., Coffman and Hegg, 1995; Weber et al., 1997, 1999, 2003; Kim et al., 1998; Korhonen et al., 1999; Kumala et al., 2000). This process has recently be incorporated into CMAQ (Zhang et al., 2005), although it has not yet been included in most other AQMs.

Compared with criteria air pollutants, modeling studies on agriculturally emitted pollutants are sparse on all scales. Several modeling studies of reduced nitrogen ($NH_x = NH_3 + NH_4^+$) have been conducted on a global scale or a large continental scale at a relatively coarse resolution from 150 × 150 km to 10° × 10° (e.g., Dentener and Crutzen, 1994; Galperin and Sofiev, 1998). Urban-to-regional simulations using 3-D Eulerian or Lagrangian chemistry and transport models at finer resolutions of 5 × 5 km to 80 × 80 km have also been performed but focused primarily on European countries (e.g., Galperin and Sofiev, 1998; Metcalfe et al., 1998; Ambelas SkjØth et al., 2004) and are very limited in other regions (e.g., in the Kanto region of Japan (Sakurai et al., 2005); in the eastern U.S. (Mathur and Dennis, 2003); and in the southern U.S. (e.g., Wu et al., 2005; 2006)). To study the fate of NH_3 emissions and its impact on PM formation, a comprehensive 3-D modeling is being conducted by the lead author's air quality modeling group at North Carolina State University. Two 1-month baseline simulations have been conducted using the US-EPA's modeling system for August and December, 2002 in a Southeast U.S. domain that covers primarily the state of North Carolina at a 4-km horizontal grid spacing (Wu et al., 2005, 2006). The US-EPA's modeling system consists of 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 Community Multiscale Air Quality (CMAQ) modeling system (Binkowski and Roselle, 2003) version 4.4 with the Process Analysis technique. The initial and boundary conditions for MM5 and CMAQ simulations at a 4-km grid spacing are developed based on the Visibility Improvement State and Tribal Association of the Southeast's (VISTAS) Phase II modeling study at a 12 km grid spacing (http://www.vista-sesarm.org.asp). A comprehensive evaluation of both meteorological and chemical conditions along with a detailed process analysis for the baseline simulations are being performed (Krishnan et al., 2006; Wu et al., 2006; Queen et al., 2006). The baseline simulation results are evaluated using the observational datasets from national and state-owned networks such as the Interagency Monitoring of Protected Visual Environments (IMPROVE), the EPA Speciation Trends Networks (STN), the Clean Air Status Trends Network (CASTNet), and the North Carolina Department of Environment and Natural Resources (NCDENR). The relative importance of meteorological and chemical processes for $PM_{2.5}$ and its composition such as SO_4^{2-}, NO_3^-, and NH_4^+ and gaseous precursors such as NH_3, NO_x, and SO_2 is being examined. The likely reasons for the discrepancies between the simulated and observed meteorological variables and chemical concentrations are being identified. The sensitivity of the predicted precipitation and wet deposition amounts to the cloud microphysical modules is being studied using two cloud microphysical modules (Queen et al., 2006). The sensitivity of the predicted dry deposition amounts to important dry deposition module parameters (e.g., dry deposition velocity and resistance) is evaluated (Krishnan et al., 2006). The gas/particle partitioning of NH_3/NH_4^+ is studied using both CMAQ and several aerosol thermodynamic modules in order to understand the formation of ammonium salts under various meteorological and chemical conditions (Wang et al., 2006; Hu et al., 2006). In addition to the inaccuracies in the simulated meteorological field and the uncertainties in the model treatment for aerosol dynamics and chemistry and dry/wet deposition, the inaccuracies in the estimation of NH_3 emissions can have a large effect on the model performance on ammonium, nitrate, and $PM_{2.5}$. Sensitivity simulations have thus also been conducted to evaluate the accuracy of the NH_3 emission inventory used and the impact of emission adjustments on overall model predictions (Hu et al., 2006).

Large uncertainties in current agricultural air quality modeling lie in several aspects including (1) inaccurate emission inventories as a result of inaccurate emission factors for various source categories from animal operations and crop production and the use of different methods to generate the inventories; (2) simplified model treatments of chemical and physical processes (e.g., gas/particle partitioning, dry and wet deposition modules); (3) inaccurate meteorological predictions (e.g., fraction velocity and precipitation); (4) lack of a detailed information on terrain characteristics and land use (e.g., surface roughness and vegetation types); and (5) paucity of observational data of emissions, concentrations, and deposition amounts for model verification and evaluation. In this review, uncertainties associated with each of these aspects will be reviewed in detail. Model simulation results from several case studies conducted in several domains (e.g., the southeastern U.S. and the state of North Carolina) by the lead author's group will be presented. The deficiencies and uncertainties in current AQMs, model inputs, and measurements will be indicated along with recommendations regarding potential model improvements and data needs. Finally, the important implications of results from 3-D AQMs in developing relevant regulations and control strategies for agricultural air quality as well as future research opportunities for studying agriculture-related pollutants and their impacts on air quality, human health, and regional climate will be discussed.

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Implications of Poposed PM Coarse National Ambient Air Quality Standards (NAAQS) on Agricultural Sources

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Abstract

The US Environmental Protection Agency is currently assessing the need for a National Ambient Air Quality Standard for the coarse fraction of particulate material (PM_c), specifically, the fraction of particulate matter between 2.5 and 10 µm in aerodynamic equivalent diameter. EPA is primarily relying on epidemiological studies that examine the possible health effects of PM_c to reach a decision about developing a coarse particulate matter standard. These epidemiological studies utilize data from size-selective PM samplers to estimate the study population's exposure to PM_{10}, $PM_{2.5}$, and PM_c. Epidemiological studies typically focus on urban populations in order to obtain sufficient sample size and increase statistical certainty of study findings. This focus on urban environments has resulted in a lack of studies evaluating the effect of coarse particulate matter in rural environments. There are a number of key differences between the urban and rural environments in the United States that can lead to mistakes in applying data from urban studies to rural environments. These include differences in particle sources, affecting particle size distribution and composition, differences in the concentration of gaseous co-pollutants, and differences in PM sampler performance in the two environments. It is our contention that these differences between the urban and rural environment are significant and that the epidemiological studies cited by EPA rely on data that are not representative of rural environments, raising concerns that the implementation of a PM_c standard in rural environments will impose an unfair and unwarranted regulatory burden on the businesses and citizens in these areas.

Environmental Load of Ammonia in the Vicinity of Livestock Enterprises

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Abstract

Animal production is the main source of ammonia emissions. Ammonia emissions are part of international strategies of prevention of air pollution and its control but also part of national regulations. At the present time reliable data of emission flows and reduction efficiencies are available from stable with forced ventilation only. Reflecting to naturally ventilated stables or free land keeping measurements of the ammonia concentration in ambient air in the vicinity of such sources may give a contribution to solve this problem.

A simple and cheap method is the use of passive samplers. Units with 4 single Ferm samplers each are mounted in the vicinity of poultry houses and close to a free land calf keeping. The study runs since June 2003 and is still running. Concentration levels range between 1 μg/m³ and more than 300 μg/m³. As main parameter the ambient temperature is identified, the progress in fattening is less important. Concentration decreases faster than assumed in some models. This result can be of interest if minimum distances are required between a livestock enterprise and e.g. forests or residential districts.

Aim of the study is to provide data on farm level. These data may be used in models to estimate source strength.

Introduction

In the past ammonia in the air of livestock buildings was seen as a problem of man and animal health and welfare only. But since a couple of years ammonia is considered as pollutant and agriculture is part of large scale air pollution and control strategies e.g. the UN ECE Convention on Long Range Transboundary Air Pollution (UN ECE 1979). In different protocols national emission ceilings are given. Some countries will keep other will exceed the limits.

Emissions of air pollutants may be measured directly as fluxes but mostly indirectly as the product of air flow rate and airborne concentration. At the present time reliable data of emission flows and reduction efficiencies are available from stableswith forced ventilation only. Reflecting to naturally ventilated stables or free land keeping measurements of the ammonia concentration in ambient air in the vicinity of such sources may give a contribution to solve this problem (Gärtner et al. 2004). A simple and cheap method is the use of passive samplers. Parallel with measurements of air quality inside the building (Hinz et al. 2004) first investigations using this technique were initiated 2003 on a turkey barn and are still running. Meanwhile broiler, calves and pigs will be tackled.

In the following the procedure and results are presented at the example of a turkey and calf enterprise.

Methods

The investigations were carried out in two commercial poultry farms and the experimental plant of the FAL:

1) Fattening turkeys in a stable with natural ventilation and a veranda
2) Fattening broilers in a stable with forced ventilation and free range
3) Fattening pigs in a stable with forced ventilation
4) Calves in free land keeping with cottages shown in figure 1.

Figure 1. Calves in free land keeping- location of one sampler to detect ammonia

In this paper method and results are given for the examples of turkeys and calves only.

To measure ammonia concentration passive sampler were used. In each case 4 samplers were mounted on 7 respectively 13 pales in a height of approximately 2.5 m above ground. In calf keeping the masts stood inside or at the border of the range. At the poultry houses the measuring positions were located in different distances. On the turkey farm these distances vary from 2 m near the curtains up to 167 m from the front wall of the barn. Figure 2 a), b) schematically gives the arrangements for calves and turkeys.

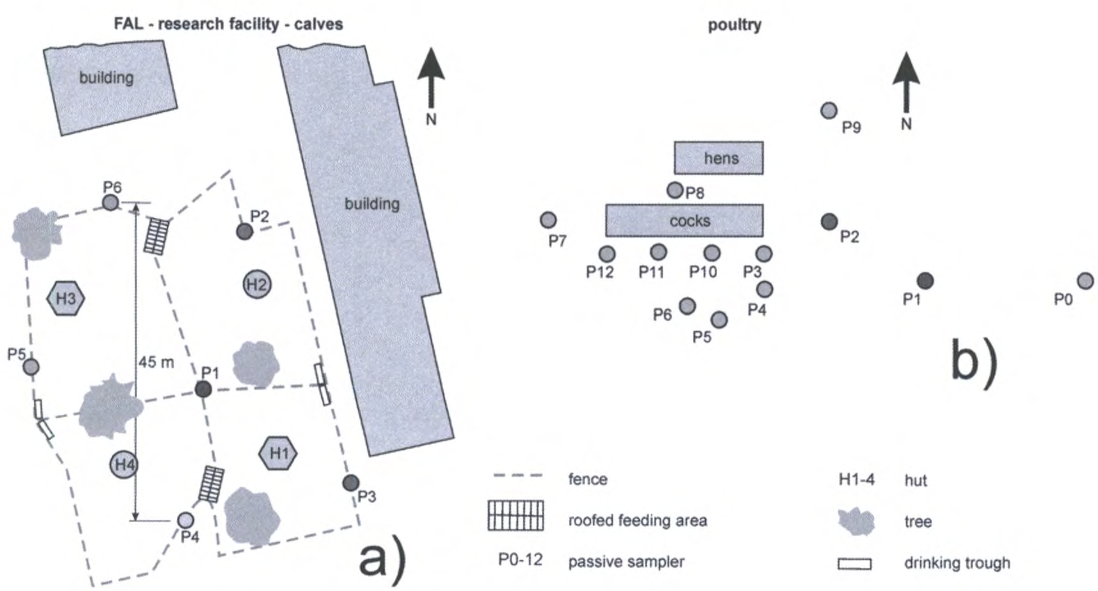

Figure 2. Arrangement of pales at calf a) and turkey b) enterprises

In the region of that turkey farm the main wind directions are west but, especially in winter must be noted east. It is to consider that in a distance of about 300 m from the turkey barn a lot of laying hens are housed. To get the information about wind speed and wind direction samplers had an ultra sonic anemometer was installed between the both houses but over roof height. All were changed periodically every 14 days. The campaign for calves was running for 9 months and is still running for the turkeys, whereby some locations had been changed with priority to pales in main wind directions.

The sampler itself was constructed according to Ferm figure 3.

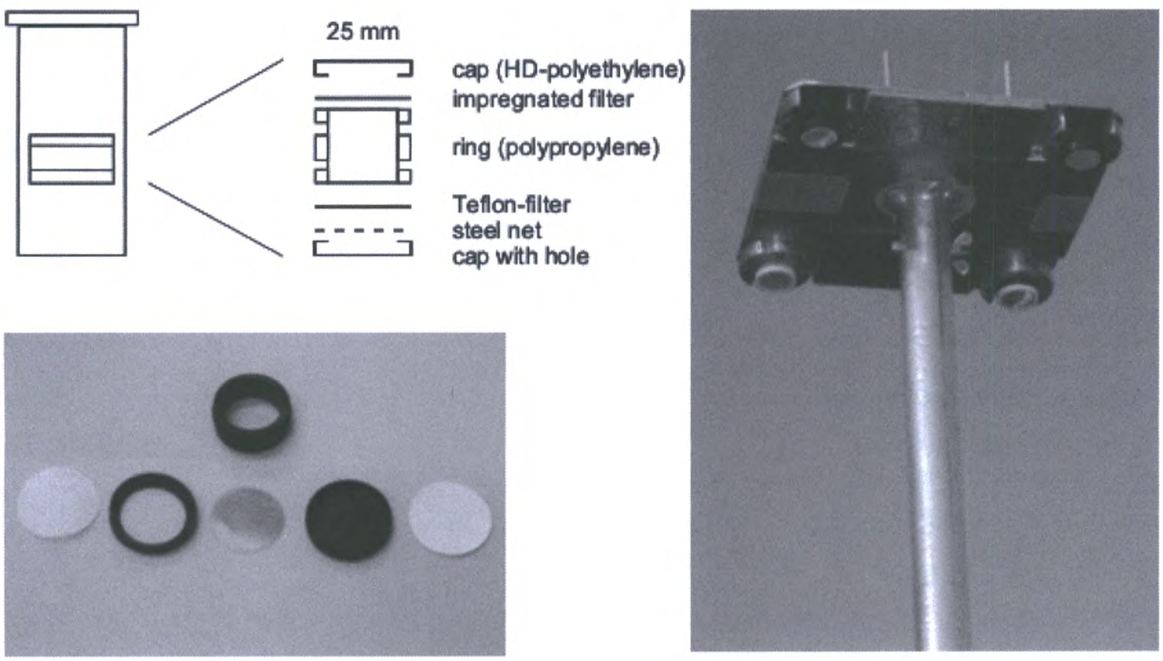

Figure 3. Passive sampler according to Ferm, details and arrangement on a pale

Ferm constructed this type of sampler short, broad and therefore sensitive for relatively low concentration in ambient air of the environment (Ferm 1991). A thin porous membrane filter was used to avoid turbulent diffusion inside the sampler. A filter covered with citric acid takes up the ammonia.

After sampling the filter is extracted and analysed. The result is an averaged concentration related to the sampling time.

Results and Discussion

Calves in Free Land Keeping

For calves in free land keeping ammonia concentration ranged between 1 $\mu g/m^3$ and 60 $\mu g/m^3$. Local dependencies were found with higher values in areas with feeding / mucking sites. The shape of the curves during the period of investigation is very similar and follows mainly the course of temperature. Figure 4 shows this finding on the example of pale 6, figure 2 a).

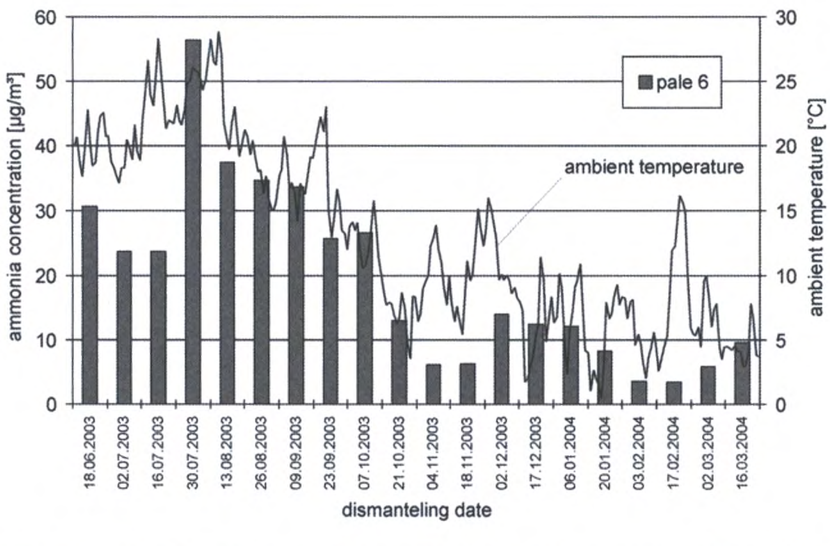

Figure 4. Course of ammonia concentration and ambient temperature at pale 6

After increasing to maximum values in July concentration and temperature go more or less continuously down to be below 10 µg/m³. This is the limit value in Germany to protect sensitive plants and ecosystems. Especially under the consideration that concentrations had been measured directly at the free range there is no relevant influence to the environment to be seen, caused by that calf keeping.

Turkey Enterprise

Depending on a much higher strength of the ammonia source and the large variety of the pale locations with respect to the distance from the houses, ammonia concentration was measured in a range between 1µg/m³ and more than 300 µg/m³ for single samples of 14 days duration each. Annual averages reached measures up to more than 100 µg/m³ for the closest pales - pale 3 and pale8. In figure 5 the annual averages and a three years average are plotted for all pales around the stables.

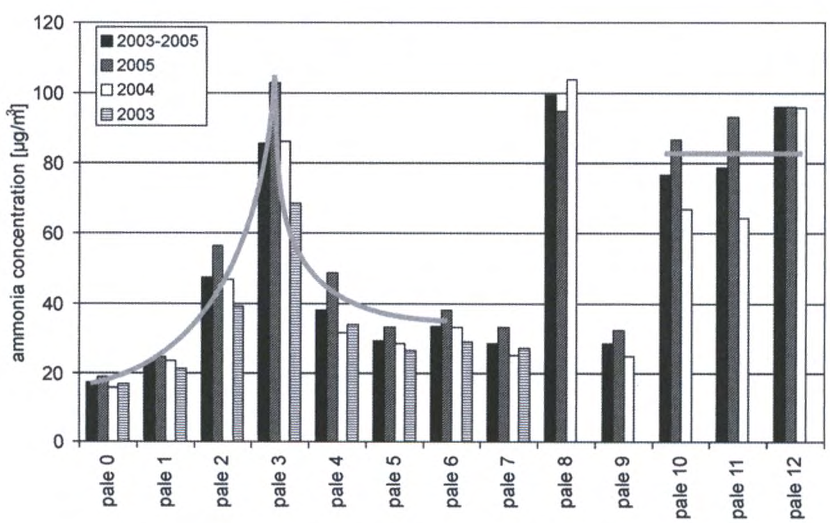

Figure 5. Averaged concentrations for all pales

There are some differences between averages but almost negligible. Comparison of pales 3 with the pales 2, 1 and 0 demonstrates a relatively sharp decrease of ammonia concentration with the distance of dispersion. 50 % reduction needs a distance of less than 40 m. A similar message gives the comparison of pale 3 with pales 4-6, which show concentration on the same level.

Sampling times of 14 days may mask influences of short-term events like mucking out at the end of a fattening period. To check this, samplers were mounted for two days only. The result is drawn in figure 6 in comparison with the normal conditions with birds inside.

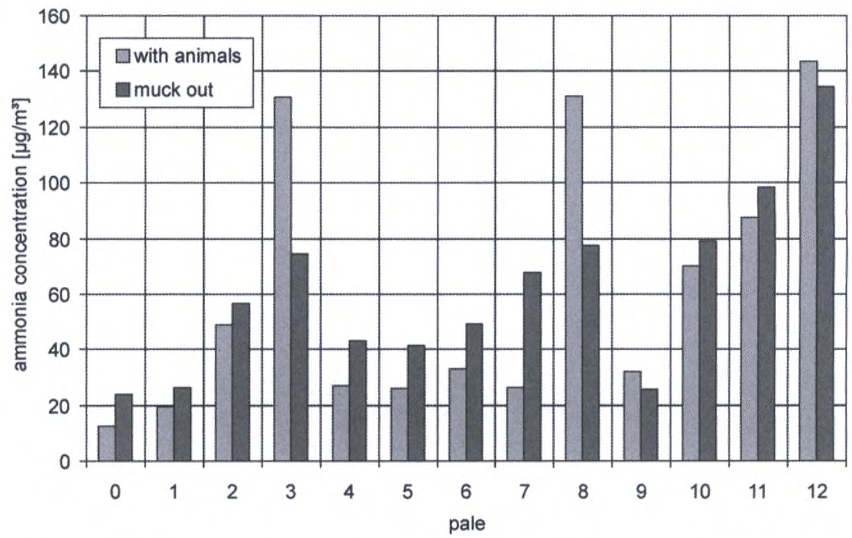

Figure 6. Ammonia concentration during fattening and mucking out

Passive sampler are able to detect short-term events. For most of the pales concentration during mucking is higher than during the attening period and will dominate the 14 days average.

Conclusion

Passive samplers according to Ferm are designed to determine low ammonia concentration in the ambient air. The investigations show that this technique is generally appropriate to measure in the vicinity of barns and other enterprises in animal production even if concentration reaches values up to 300 µg/m³ and more.

Averaging sampling times of 14 days but also short-term application for one or two days can be realized.

Local dependencies can be detected clearly.

As a main parameter the ambient temperature during the course of a year was found.

Measurements around a free land calf keeping showed concentration values between 1 µg/m³ up to 60 µg/m³ and 10 µg/m³ up to 300 µg/m³around the turkey barn in 14 days averages.

Ammonia concentration decreases relatively sharp verified by measurements done on a turkey fattening farm.

Considering local conditions and the run of a year there was no remarkable environmental impact in the investigated cases.

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Posters

Foliar Nitrogen Status and Growth of Plants Exposed to Atmospheric Ammonia (NH_3)

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Abstract

Two experiments were conducted to evaluate the potential of plant species to trap NH_3 discharged from poultry houses. The four plant species (cedar, locust, poplar, and grass) used in experiment (Exp) 1 and four species (spruce, arborvitae, poplar, and willow) used in Exp 2 were transplanted into 4- to 8-L pots and grown in four environmentally controlled chambers. Two of the four chambers received continuous anhydrous NH_3 at 4 to 5 ppm (Exp 1) or 6 to 8 ppm (Exp 2) while the other two chambers in each experiment received no NH_3. The results of Exp 1 showed that locust was the fastest growing species, supported by its total biomass, root, and root DM weights. There was a trend for all the plants exposed to NH_3 to have greater leaf DM than their non-exposed counterparts at 6 (43.0 vs. 30.8%; P=0.09) and 12 wk (47.9 vs. 36.6%; P=0.07), and significantly greater ($P \leq 0.05$) leaf N at 6 (6.44 vs. 3.67%) and 12 wk (7.05 vs. 3.51%) when exposed to NH_3. An increase in leaf N due to NH_3 exposure was also noted in Exp 2 (4.99 vs. 2.83%; $P \leq 0.05$), but with no indication with species interaction. Numerically greater leaf DM due to NH_3 exposure was consistently measured in poplar tissues at both sampling periods in Exp 1. In Exp 1, poplar, as well as locust and grass, deposited 1.5 to 2 fold greater N in their leaves than cedar tissues as a result of NH_3 exposure compared to non-exposed plants. Similar numerical trends in foliar N were also observed in poplar, willow and arbovitae in Exp 2 (P > 0.05). Locust (Exp 1) likely benefited from ambient NH_3 while poplar and willow (Exp 1 and 2) were negatively affected by atmospheric NH_3.

Introduction

Ammonia (NH_3) is the foremost gas of concern in poultry houses. Feed can be a major contributor of this gas since NH_3-N losses as a percentage of dietary feed-N range from 18 to 40% depending on type of bird (broiler, turkey, pullet, or laying hen) (Patterson and Lorenz, 1996, 1997, Patterson et al., 1998; Patterson et al., 1999), age, and management style. Both dietary and in-house management strategies are utilized to reduce the generation and emission of NH_3 inside the house, but offer no benefit once it is discharged via the exhaust fans. Field observations since 2003 on Pennsylvania poultry farms indicate that neighbors are concerned about emissions around many poultry production sites (unpublished data). However, concentrations of the airborne NH_3 once it is discharged by the exhaust fans is not well documented, and is dependent on the fan performance, outside wind speed, distance of measurement from the exhaust fan, and the presence of vegetation (Seedorf and Hartung, 2000; Pitcairn et al., 1998).

Yin et al. (1998) reported it is possible for atmospheric NH_3 to enter plants through foliar stomata and assimilate into plant cells through the glutamine synthetase and glutamate synthase pathways at up to 5.51 ppm NH_3 for short periods of time without disturbing photosynthesis or transpiration. Hence, at favorable concentrations, NH_y (NH_3 + NH_4^+) will induce plant growth while at a critical threshold NH_3 will cause necrosis, growth reduction, and increased frost sensitivity (Van deer Eerden et al., 1998). The objective of this study was to evaluate the tolerance of some plant species to NH_3 exposure released at a concentration range commonly detected near commercial poultry farms.

Materials and Methods

Four plant species were grown in environmentally controlled chambers up to 12 wk in Experiment (Exp) 1 or 6 wk in Exp 2. The plants comprised of 53 four-yr-old red cedar (*Juniperus virginiana*), 40 one-yr-old thornless honey locus (*Gleditsia triacanthos* var. *inermis*), 57 one-yr-old hybrid poplar (*Populus sp.*), and 58 uniform of clumps of reed canary-grass (*Phalaris arundinacea*) in Exp 1. In Exp 2 45 3-yr-old white spruce (*Picea glauca*), 45 3-yr-old arbovitae (*Thuja spp.*), 46 two-yr-old poplar, and 32 two-year-old streamco willow (*Salix purpurea*) were utilized. All the plants were transplanted into 4- and 8-L pots containing NX-6 pine bark media and placed onto two tables in each chamber. Two of the four chambers were treated with (+) NH_3 exposure at 4 to 5 ppm (Exp 1) or 5 to 7 (Exp 2) and the other two in each experiment received no (-) NH_3. Each plant species in each chamber of Exp 1 was divided into two groups with no fertilizer (-) or NPK fertilizer at 100 ppm (+) once a week. All plants in Exp 2 received the fertilizer.

The anhydrous NH_3 was released from a 150-L NH_3 tank via a flow meter, monitored by a photoacoustic NH_3 detector (Model 1412, Innova, DK-2750 Ballerup, Denmark) weekly and backed-up with passive dosi-tube readings (3D, Gastec Corp., 6431 Fukaya, Japan). The lighting program was set for 16 h light and 8 h dark (16L:8D) every day for all chambers and light intensity was maintained at minimum (Exp 1) or medium requirements (Exp 2). Chamber temperature (T) and relative humidity (RH) were programmed to meet the required level for plant growth. Two fertilized plants per species per chamber were sampled on wk 0, 6, (Exp 1 and 2), and 12 (Exp 1) for fresh plant biomass wt, root wt, root DM, foliar DM and N analyses. Plant height and stem diameter were recorded on all plants only in Exp 1.

Data were subjected to a split plot design analysis of Proc Mixed of SAS (SAS Institute, 1999). Model 1 was used to analyze foliar DM and N data of Exp 1 and 2, while Model 2 was used to analyze plant height and plant diameter data obtained from Exp 1. The two models are as described below:

$$X_{ijk} = \mu + A_i + C(A_i)_j + S_k + AS_{ik} + \varepsilon_{(ijk)} \quad \text{(Model 1)}$$

$$X_{ijkl} = \mu + A_i + C(A_i)_j + S_k + F_l + AS_{ik} + AF_{il} + SF_{kl} + ASF_{ikl} + \varepsilon_{(ijkl)} \quad \text{(Model 2)}$$

where X_{ijk} or X_{ijkl} is the value observed, μ is the overall mean, A is the effect of NH_3 (main plot factor), C(A) is the error for A (where C = chamber, which is nested in A), S and F are the effects of species and fertilizer (sub plot factors), respectively, AS is the interaction effect of NH_3 by species, AF is the interaction effect of NH_3 by fertilizer, SF is the interaction effect of species by fertilizer, ASF is the interaction effect of NH_3 by species by fertilizer, and ε is the residual error. Tukey's test (SAS Institute, 1999) was employed to the data showing significance ($P \leq 0.05$).

Results

The average daily T and RH ranged from 24 to 24.7 ºC and 44.3 to 44.7%, in the (-) and (+) NH_3 chambers, respectively in Exp 1, while in Exp 2 they were 24.2 to 24.7 ºC and 44.8 to 45%, respectively. Light intensity averaged 1522 lux in Exp 1 and 3049 lux in Exp 2. The concentrations of NH_3 ranged from 0.08 to 0.20 ppm in the (-) NH_3 chambers (Exp 1 and 2) and 4 to 5 ppm (Exp 1) and 6 to 8 ppm (Exp 2) in the (+) NH_3 chambers.

Experiment 1

Species had a significant effect ($P \leq 0.05$) on plant height and stem diameter in Exp 1 (Table 1). Although the significant effect of species by fertilizer at 6 wk existed ($P \leq 0.05$) it did not indicate a trend of plant height with fertilizer. Species was the only factor to influence total fresh biomass wt and fresh root wt (Figure 1) at all weeks in Exp 1 ($P \leq 0.05$). A significant effect of species was also observed on plant root DM. An increased root DM due to the interaction of NH_3 by species was only numerical particularly in locust (46.7 to 56.1%) at 12 wk (data are not shown).

There was a repeatable trend of the NH_3 treatment to increase leaf DM at 6 ($P=0.09$) and 12 wk ($P=0.07$) and a highly significant NH_3 effect on N deposition in plant foliage at both weeks (Table 2). Leaf DM content appeared to be positively impacted by the NH_3 treatment in the case of red cedar and hybrid poplar at week 6 ($P=0.07$) and upon hybrid polar at 12 wk ($P=0.06$) but not so for locust or reed canary grass. Moreover, the NH_3 by species interaction was highly discernable in leaf N levels with greater concentrations at 6 and 12 wk for locust, poplar, and grass as a result of NH_3 exposure. Total foliar N of the fresh leaf material at 6 and 12 wk was calculated by difference between control (-) and (+) NH_3 chambers to be 0.45 and 0.87 g 100 g^{-1} of fresh foliage weight for grass, 1.25 and 1.34 g for locust, and 2.67 and 6.09 g for poplar based on the leaf DM and N concentrations reported in Figure 2.

Table 1. Plant heights and stem diameters of plants treated with (+) or without (-) atmospheric ammonia (NH_3) for 12 wk, Exp 1.

Treatment	Height (cm)			Stem Diameter[2] (cm)		
	Wk 0	Wk 6	Wk 12	Wk 0	Wk 6	Wk 12
NH_3: (-)	53.8	66.0	68.5	0.72	0.70	0.70
(+)	53.4	65.1	65.4	0.74	0.70	0.71
Species (Spec):						
Red cedar (R)	48.2[b]	46.3[c]	46.9[c]	0.87[a]	0.77[a]	0.78[a]
Honey locust (L)	54.6[ab]	83.1[a]	88.0[a]	0.60[b]	0.62[b]	0.64[b]
Hybrid poplar (P)	57.8[a]	69.0[b]	68.7[b]	0.71[b]	0.71[b]	0.69[b]
Reed canary grass (G)	53.9[ab]	63.8[b]	64.2[b]	-	-	-
Fertilizer (Fert): (+)	52.7	66.0	67.8	0.71	0.67	0.68
(-)	54.5	65.1	66.1	0.74	0.73	0.73
Spec x Fert:						
R × (+)	48.6	45.0[e]	45.2	0.86	0.72	0.74
R × (-)	47.8	47.5[de]	48.5	0.88	0.81	0.81
L × (+)	54.1	85.7[a]	92.8	0.57	0.61	0.65
L × (-)	55.0	80.5[ab]	83.3	0.63	0.62	0.64
P × (+)	54.1	64.8[bc]	65.7	0.70	0.66	0.65
P × (-)	61.5	73.1[abc]	71.7	0.72	0.77	0.73
G × (+)	54.1	68.3[bc]	67.4	-	-	-
G × (-)	53.7	59.3[cd]	61.0	-	-	-
NH_3 × Spec:	…[1]	…	…	…	…	…
NH_3 × Fert:	…	…	…	…	…	…
NH_3 × Spec × Fert	…	…	…	…	…	…
Sources of Variances:	Probabilities					
NH_3	0.94	0.71	0.52	0.80	0.90	0.72
Spec	0.04	0.0001	0.0001	0.0002	0.03	0.04
Fert	0.45	0.71	0.57	0.51	0.14	0.27
Spec × Fert	0.53	0.05	0.19	0.94	0.65	0.62
NH_3 × Spec	0.83	0.95	0.21	0.91	0.42	0.71
NH_3 × Fert	0.64	0.11	0.07	0.74	0.91	0.45
NH_3 × Spec × Fert	0.52	0.83	0.82	0.93	0.98	0.82

[a-e]Means in a column with no common superscripts differ significantly ($P \leq 0.05$).
[1]Mean values are not shown because they did not have significant probabilities ($P \geq 0.05$).
[2]Measuremens were taken only on red cedar, honey locust, and hybrid poplar.

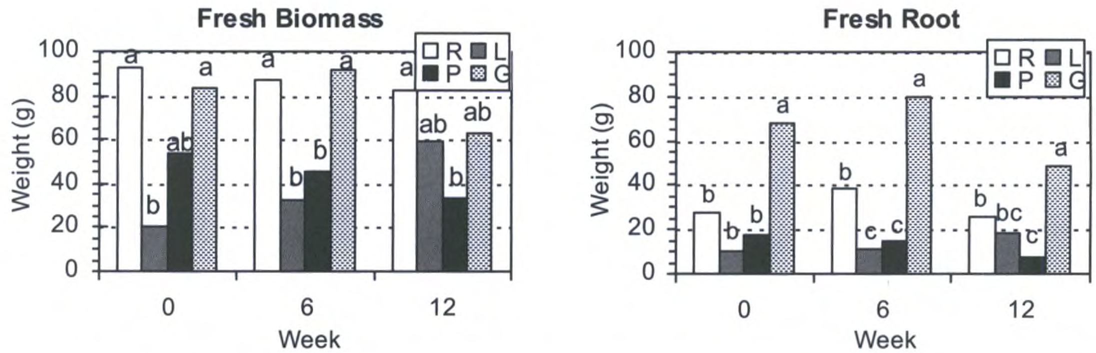
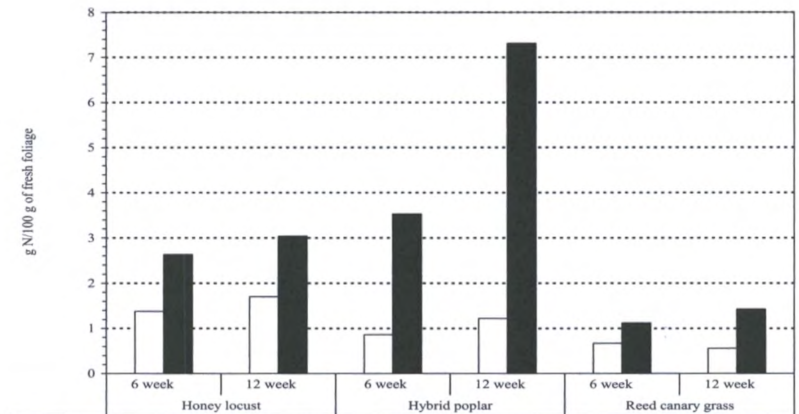


Figure 1. Fresh biomass and root weight of the plants treated with (+) or without (-) NH_3 exposure, Exp 1.

Table 2. Leaf dry matter (DM) and nitrogen (N) concentration of plants treated with (+) or without (-) atmospheric ammonia (NH_3) exposure for 12 wks Exp 1.

Treatment		DM (%)			N (%)		
		Wk 0	Wk 6	Wk 12	Wk 0	Wk 6	Wk12
NH_3:	(-)	27.1	30.8	36.6	3.84	3.67^{b}	3.51^{b}
	(+)	26.9	43.0	47.9	3.82	6.44^{a}	7.05^{a}
Species:							
	R	46.9^{a}	70.9^{a}	64.2^{a}	1.63^{c}	1.95^{c}	2.26^{c}
	L	26.9^{b}	25.8^{b}	32.8^{bc}	4.92^{a}	7.46^{a}	7.15^{a}
	P	19.1^{c}	28.9^{b}	51.0^{ab}	5.35^{a}	6.81^{a}	7.12^{a}
	G	15.1^{c}	22.1^{b}	21.0^{c}	3.41^{b}	4.00^{b}	4.61^{b}
NH_3 x Species:							
	(-) × R	47.5	57.6	64.8	1.53	1.57^{d}	1.84^{e}
	(-) × L	26.8	25.1	31.9	4.88	5.49^{b}	5.32^{bc}
	(-) × P	18.8	18.9	30.3	5.49	4.55^{b}	4.04^{cd}
	(-) × G	15.2	21.7	19.5	3.45	3.08^{c}	2.86^{de}
	(+) × R	46.2	84.1	63.5	1.74	2.34^{cd}	2.68^{de}
	(+) × L	27.1	26.5	33.8	5.00	9.44^{a}	8.99^{a}
	(+) × P	19.3	38.9	71.7	5.20	9.07^{a}	10.20^{a}
	(+) × G	15.0	22.6	22.5	3.37	4.91^{b}	6.36^{b}
Sources of Variances:		---------------------------------- Probabilities ----------------------------------					
NH_3		0.93	0.09	0.07	0.87	0.02	0.0001
Species		0.0001	0.0001	0.0002	0.0001	0.0001	0.0001
NH_3 × Species		0.92	0.07	0.06	0.61	0.0001	0.0001

[a-e]Means in a column with no common superscripts differ significantly ($P \leq 0.05$).



	6 week	12 week	6 week	12 week	6 week	12 week
	Honey locust		Hybrid poplar		Reed canary grass	
□ (-) NH3	1.38	1.7	0.86	1.22	0.67	0.56
■ (+) NH3	2.63	3.04	3.53	7.31	1.12	1.43

Figure 2. Total foliage N of the plants at 6 and 12 wk treated with (+) or without (-) NH_3 exposure (fresh leaf wt before and after NH_3 exposure at 6 wk were 8.95 and 9.0 g for locust, 10.88 and 9.33 g for poplar, and 9.93 and 13.6 g for grass, respectively; while at 12 wk they were 14.85 and 24.68 g for locust, 4.68 and 3.85 g for poplar, and 14.07 and 13.8 g for grass, respectively), Exp 1.

There was a numerical trend for NH_3 exposure to increase leaf color scores at week 6 (P=0.10) and 12 (P=0.09) and mean leaf damage scores at week 6 (P=0.06) and 12 (P= 0.09) (Table 3). There were also highly species-dependent responses to NH_3 as shown by higher leaf color and damage scores. This indicated that some species were sensitive to NH_3, e.g. cedar, poplar, and grass whereas the locust color was enhanced by NH_3 treatment and showed lower damage values than other species.

Experiment 2

The greater plant foliar N resulting from NH_3 exposure was apparent at 6 wk compared with non-exposed plants (4.99 vs. 2.83%; $P \leq 0.05$) (Table 4). A similar trend in foliar DM was also observed with NH_3 exposure (57.4 vs. 46.5%; P=0.16). Poplar and willow deposited more N in their leaves than the other two species (arbovitae and spruce). Although the probability value did not indicate a significant NH_3 by species interaction on foliar N of the plants at 6 wk, the numerical trend toward greater foliar N in those exposed to NH_3 was discernable, particularly in arborvitae, poplar, and willow (P=0.13). This was also supported by a numerical trend of foliar DM at the same period.

Higher average leaf color scores and leaf damage scores due to NH_3 exposure was also observed e.g. 3.48 vs. 3.08 (P=0.14) and 4.51 vs. 4.40 (P=0.74), respectively (Table 5). Different plant species showed different responses to NH_3 exposure with poplar and willow showing greater leaf color and damage scores than arborvitae and spruce. However, there were no NH_3 by species interactions observed for plant color and damage scores.

Table 3. Leaf color and damage scores of plants treated with (+) or without (a) ammonia (NH_3) exposure for 12 wk, Exp 1.

Treatments	Color Score			Damage Score		
	Wk 0[1]	Wk 6	Wk 12	Wk 0[1]	Wk 6	Wk 12
NH_3: (-)	2.21	2.12	2.86	1	2.60	4.42
(+)	2.21	2.50	3.22	1	4.05	5.40
Species (Spec):						
R	2.00	2.78^{a}	3.86^{a}	1	3.15^{b}	5.59^{a}
L	3.00	1.39^{c}	1.75^{b}	1	1.54^{c}	3.90^{b}
P	2.00	2.88^{a}	4.32^{a}	1	4.42^{a}	6.08^{a}
G	2.00	2.18^{b}	2.25^{b}	1	4.18^{ab}	4.07^{b}
Fertilizer (Fert):						
(+)	2.21	2.31	2.96	1	3.46	4.78
(-)	2.21	2.31	3.13	1	3.18	5.04
Spec x Fert:	...[2]	...	...	-	...	...
NH_3 x Spec:						
(-) × R	-	2.57^{ab}	3.90^{ab}	-	2.96^{b}	5.11^{abc}
(-) × L	-	1.62^{bc}	2.17^{cd}	-	1.20^{c}	4.54^{bcd}
(-) × P	-	2.44^{ab}	3.77^{ab}	-	2.55^{b}	5.17^{abc}
(-) × G	-	1.85^{bc}	1.62^{cd}	-	3.69^{b}	2.85^{d}
(+) × R	-	2.98^{a}	3.82^{ab}	-	3.35^{b}	6.06^{ab}
(+) × L	-	1.16^{c}	1.33^{d}	-	1.88^{c}	3.25^{cd}
(+) × P	-	3.31^{a}	4.87^{a}	-	6.28^{a}	6.99^{a}
(+) × G	-	2.51^{ab}	2.87^{bc}	-	4.68^{ab}	5.28^{abc}
NH_3 × Fert:	...	...	...	...	...	...
NH_3 × Spec × Fert	...	...	...	...	...	...
Sources of Variances:	Probabilities					
NH_3	-	0.10	0.09	-	0.06	0.09
Spec	-	0.0001	0.0001	-	0.0001	0.0001
Fert	-	0.97	0.43	-	0.38	0.44
Spec × Fert	-	0.44	0.13	-	0.92	0.20
NH_3 × Spec	-	0.03	0.002	-	0.001	0.001
NH_3 × Fert	-	0.33	0.27	-	0.16	0.06
NH_3 × Spec × Fert	-	0.19	0.08	-	0.75	0.06

[a-d]Means in a column with no common superscripts differ significantly ($P \leq 0.05$).

[1]Data at week 0 were not subjected to ANOVA because there were no differences within each species.

[2]Mean values are not shown because they did not have significant probabilities ($P \geq 0.05$).

Table 4. Leaf dry matter (DM) and nitrogen (N) concentration of plants treated with (+) or without (-) atmospheric ammonia (NH_3) exposure for 6 wk, Exp 2.

Treatment	DM (%)		N (%)	
	Wk 0	Wk 6	Wk 0	Wk 6
NH_3: (-)	29.9	46.5	2.62	2.83^a
(+)	30.9	57.4	2.67	4.99^b
Species:				
Arbovitae (A)	41.2^a	37.9^b	1.75^c	3.12^b
Poplar (P)	22.8^c	77.1^a	3.35^b	4.86^a
Spruce (S)	33.5^b	39.5^b	1.45^c	2.10^b
Willow (W)	24.2^c	53.3^b	4.04^a	5.55^a
NH_3 × Species:				
(-) × A	39.7	34.5	1.67^c	2.24
(-) × P	22.9	79.3	3.54^{ab}	3.55
(-) × S	33.3	37.3	1.49^c	1.55
(-) × W	23.8	35.0	3.78^{ab}	3.96
(+) × A	42.7	41.3	1.83^c	4.00
(+) × P	22.7	75.0	3.15^b	6.17
(+) × S	33.7	41.8	1.41^c	2.65
(+) × W	24.6	71.5	4.29^a	7.14
Source of Variances:	------------------------------- Probabilities -------------------------------			
NH_3	0.36	0.16	0.77	0.04
Species	0.0001	0.0001	0.0001	0.0001
NH_3 × Species	0.71	0.09	0.05	0.13

[a-c]Means in a column with no common superscripts differ significantly ($P \leq 0.05$).

Table 5. Leaf color and damage scores of plants treated with (+) or without (-) ammonia (NH_3) exposure for 6 wk, Exp 2.

Treatment	Color Score Wk 0[1]	Color Score Wk 6	Damage Score Wk 0[1]	Damage Score Wk 6
NH_3: (-)	2.02	3.08	1	4.40
(+)	2.00	3.48	1	4.51
Species:				
A	2.00	1.61[d]	1	1.56[b]
P	2.00	4.88[a]	1	6.96[a]
S	2.00	2.57[c]	1	2.29[b]
W	2.03	4.04[b]	1	7.00[a]
NH_3 x Species:				
(-) × A	2.00	1.50	1	1.40
(-) × P	2.00	4.92	1	6.92
(-) × S	2.00	2.54	1	2.27
(-) × W	2.00	3.33	1	7.00
(+) × A	2.00	1.73	1	1.73
(+) × P	2.00	4.85	1	7.00
(+) × S	2.00	2.60	1	2.30
(+) × W	2.00	4.75	1	7.00
Source of Variances:	---------- Probabilities ----------			
NH_3	-	0.14	-	0.74
Species	-	0.0001	-	0.0001
NH_3 × Species	-	0.12	-	0.98

[a-c]Means in a column with no common superscripts differ significantly ($P \leq 0.05$).
[1]Data at week 0 were not subjected to ANOVA because there were no differences within each species.

Discussion

The superior growth of locust (Exp 1) compared with other species is likely due to the unique symbioses of its roots with bacteria in fixating air NH_y (Zahran, 2001). Because the effect of fertilizer on plant heights at wk 6 did not show any impact within a species, this suggested that growth response of plants to fertilizer was unique to each species. Visual observation of red cedar in the (+) NH_3 chambers in Exp 1 showed that new shoots became dry as they emerged indicating metabolic disturbance or necrosis, which was possibly due to NH_3 exposure. The same symptoms occurred in Exp 2 with the spruce. Krupa (2003) reported that needle necrosis following needle drying was a common response of conifers to NH_3 near livestock farms. These species will become very sensitive when they are exposed to NH_3 for long periods of time (Van deer Eerden, 1982).

The greater leaf N of honey locust, hybrid poplar, and grass observed in Exp 1 and the numerical trend in Exp 2 indicated greater metabolism or detoxification of the absorbed NH_3-N (Fangmeier et al., 1994). However, greater N deposition in leaves does not necessarily mean that all the plants, especially hybrid poplar (Exp 1 and 2), grass (Exp 1), willow, and spruce (Exp 2) benefited from continuous atmospheric NH_3 exposure in a closed chamber as shown by the poor leaf color quality and damaged leaves (Tables 3 and 5). Exposure to 4 to 5 ppm of NH_3 under a 16L:8D photoperiod was seemingly sufficient to elicit the adverse effects of NH_3 on these species. Black spots and necrosis indicated by leaf-tip drying were the two consistent symptoms of foliar tissues in the current experiments and consistent with the other findings as reviewed by Krupa (2003). Visual injury of the plant foliage is associated with NH_3 exposure and increased arginine levels have been reported with yellowing of young needles in *Pinus sylvestris* (Roelofs, 1987a, 1987b). This type of injury was clearly seen among red cedar in Exp 1, spruce, and some arborvitae in Exp 2 in the (+) NH_3 chambers. Visible injury in honey locust did not seem to be as great as in other species supporting the fact that this species might have benefited from NH_3 exposure as also indicated by its superior growth.

It is important to note that Two-Spotted Spider Mites were in evidence among the locust, poplar, and cedar from week 6 on in the (+) NH_3 chambers in Exp 1, although all the plants were sprayed with a miticide at the start of the experiment and again at 3 and 6 wk. The presence of these mites might have contributed to leaf color and damage scores of the plants since one of the contributing effects of elevated leaf N due to NH_3 exposure is increased sensitivity to insect pest infestation (Krupa, 2003). Therefore, it was not surprisingly to find numerically greater damage scores of all plants in the (+) NH_3 chamber compared to those in the (-) NH_3 chambers at 6 and 12 wk in Exp 1 (Table 3).

Overall, this study demonstrated that plant foliage has the potential to absorb NH_3-N discharged from the exhaust fans of poultry and livestock barns. As an example, assume the average concentration of NH_3 in a small hen house (4,000 birds) with four 61-cm exhaust fans is 30 ppm (Wathes et al., 2003; Miles et al., 2004) (equivalent to 0.020833 g m^{-3} [Krupa, 2003]), and each of the four 61-cm fans is discharging 140,213 m^3 of air d^{-1}, and if approximately 30% of NH_3-N is trapped by the plant foliage; then the foliar biomass required to trap this amount of NH_3-N will be 47 to 108 kg d^{-1} of poplar, 215 to 231 kg d^{-1} of locust, or 331 to 665 kg d^{-1} of grass based on the data herein. With this knowledge one can calculate the foliar biomass and number of trees or grasses to plant around a commercial poultry house to reduce ammonia emissions and environmental pollution.

Conclusions

Foliar condition and nutrient concentration appeared to be the more sensitive plant indicators of continuous anhydrous NH_3 exposure in environmentally controlled chambers. Although almost all the species deposited more N in their leaves due to NH_3 exposure, only honey locust appeared to tolerate and might have benefited from the continuous 4 to 5 ppm NH_3 exposure.

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Plant Foliar Nitrogen and Temperature on Commercial Poultry Farms in Pennsylvania

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Abstract

Trees have been planted for many years in agriculture settings for windbreaks and shade purposes. A new, but largely untested benefit may be as a visual screen for combating neighbor issues, as a barrier for flies, and as a filter for fan emissions from poultry and livestock farms. This field study sought to evaluate the potential of trees planted around commercial poultry farms to trap NH_3, the gas of greatest environmental concern to the poultry industry. Four plant species (spruce, poplar, streamco willow, and hybrid willow) were planted in front of the exhaust fans of eight commercial farms from 2003 to 2004. In 2005 because temperature (T) can be a stressor for trees, T was monitored with data loggers among the trees in front of the exhaust fans (11.4 m to 17.7 m) and at a control distance away from the fans (≥48 m) during all four seasons in Pennsylvania. Poplar and spruce foliage samples were taken in August 2005 from one turkey and two layer farms for dry matter (DM) and nitrogen (N) analysis. The two layer farms had poplar plantings and all three farms had spruce. The results showed that farm type had a significant effect on plant leaf DM but not leaf N. Although poplar had less foliage DM compared to spruce (41.3 vs. 50.1%), it contained greater N (3.43 vs. 2.46%). Plant location clearly showed greater foliar DM and N levels among those plants near the fans compared with controls, 51.0 vs. 40.4% and 3.61 vs. 2.28%, respectively. Greater foliar DM may have resulted from the plants' capacity to trap NH_3-N emitted by the exhaust fans resulting in better N status in the tissues, growth and biomass of the plant, or desiccation of the plants from the proximity of the fans. However, the difference in foliar DM and N concentrations due to species, location, and the interaction of the two factors was unlikely correlate with ambient T. Summer T were similar in front of the fans and at a control distance away from the fans on all farms (25.80 vs. 25.32C-layer 1, 25.92 vs. 25.53C-layer 2, and 25.45 vs. 25.54C-turkey) suggesting no greater T stress associated with fan proximity.

Introduction

Air emissions from poultry and livestock production are numerous and may include dust or particulate matter, odors and nitrogenous compounds including ammonia. Ammonia (NH_3) emissions can be significant. Our own data using mass balance techniques on commercial pullet, laying hen, broiler and turkey farms indicates that between 18 to 40% of feed N is lost to the atmosphere mostly as NH_3-N (Patterson and Lorenz, 1996, 1997; Patterson et al. 1998, Patterson et al., 1999). Planting trees around poultry farms has been utilized for wind breaks and for shade. Recently vegetative shelterbelts have been used as a visual screen, a barrier for fly migration and to trap emissions (odors, dust, and gases) discharged by the exhaust fans from poultry farms (Malone, 2004).

Plants have the capacity to absorb aerial ammonia (NH_3) via foliar stomata and cellular assimilation through the glutamine synthetase and glutamate synthase pathways (Yin et al., 1998). Van deer Eerden et al. (1998) reported that at the right concentrations, NH_y (NH_3 + NH_4^+) would favor plant growth, but at a critical threshold it would cause tissue necrosis, reduced growth, and greater frost sensitivity.

In chamber studies we determined that multiple plant species including red cedar, white spruce, arbovitae, honey locust, hybrid poplar, streamco willow, and reed canary grass deposited almost two-fold greater N in their leaves when exposed to continuous NH_3 at 4 to 8 ppm, compared to control chambers without atmospheric NH_3 (Adrizal et al., 2006). However, only honey locust consistently grew well and showed little foliar injury compared to other species; indicating its capacity to tolerate and utilize aerial NH_3-N.

Malone (2004) planted three plant species (4.9 m high bald cypress, 4.3 m high Leyland cypress, and 2.4 m high red cedar, 9m wide) at 9, 12.2, and 14.6 m downwind of the tunnel fans on a roaster chicken farm. During the summer, the trees reduced air velocity by 99%, dust by 50-53%, and NH_3 by 29-67% down wind of the trees. One concern faced by extension personnel sighting trees for poultry farms was that heat exhausted from the poultry barns in winter may throw plant species out of dormancy, or result in temperature and/or dehydration stress on the plants. This study was designed to evaluate the potential of trees planted around poultry house exhaust fans to trap NH_3, and the impact of tree proximity to the fans on environmental temperature and its associated stressors.

Materials and Methods

There were eight commercial poultry farms involved in this study including 3 broiler, 3 layer, 1 pullet, and 1 turkey farm. Cox tracer data loggers (Model CT-1E-D-16, Sensitech, Inc. MA, USA) were used to monitor temperature (T) on all farms at two locations. The first T logger was placed away from the buildings and fans (≥48 m, control), while the second was placed near the fans (11.4 to 17.7m) among the trees. Each logger was hung inside the wall of a propylene shield and secured horizontally to a metal post at 1.5 m high from the ground matching the height of the facing fan. All the loggers were programmed to record the T every 30 min continuously for two consecutive months in each season (winter: January to February; spring: April to May; summer: July to August; fall: October to November) in 2005.

More than 2000 plants (Norway spruce [*Picea abies*], hybrid willow [*Populus sp.*], and streamco willow [*Salix purpurea*]) were planted in rows on the all poultry farms from 2003 to 2004 (Table 1). In 2005, monitoring for T began in August, and foliage samples from three plants of each species were taken from the selected farms based on row position and distance from the exhaust fans.

Table 1. Characteristics of commercial poultry farms and trees.

Farm	House type	Birds & houses	Farm issues	Trees
Broiler 1	litter	21,000/house	Visual screen, snow load, odors and dust	2 rows Norway spruce 1 row hybrid willow 1 row streamco willow
Broiler 2	litter	50,000/2 houses	Dust , odors and snow load	1 row Norway spruce 1 row hybrid willow 1 row streamco willow
Broiler 3	litter	20,000/house	Dust and odors	1 row streamco willow
Layer 1	high-rise	125,000/house	Dust, odors, flies and visual screen	2 rows Norway spruce 2 rows hybrid poplar
Layer 2	high-rise	475,000/3 houses	Dust, odors and flies	2 rows Norway spruce 1 row hybrid poplar 1 row streamco willow
Layer 3	high-rise	1,000,000/8 houses	Visual screen, dust and odors	2 rows Norway spruce
Pullet	high-rise	83,000/house	Visual screen, snow load, energy conservation, and urban encroachment	2 rows Norway spruce 1 row hybrid willow
Turkey	litter	40,000/2 houses	Dust, odors, water quality, feathers and truck traffic	2 rows hybrid willow 10 rows Norway spruce

An incomplete randomized block design was applied in this study where farms were considered as a block. The two mathematical models employed were Model 1 for the analysis of T and Model 2 for the analysis of foliar DM and N data using Proc GLM of SAS followed by Bonferroni test for plant significance (SAS Institute, 1999). The foliar percentage DM and N data were transformed to arc sin before the analysis. The two models are described below:

$$X_{ijk} = \mu + F_i + L_j + \varepsilon_{ijk} \quad \text{..... (Model 1)}$$
$$X_{ijkl} = \mu + F_i + S_j + L_k + (S\times L)_{jk} + \varepsilon_{ijkl} \quad \text{..... (Model 2)}$$

where X_{ijk} is the observed value, μ is the overall mean, F_i is the i-th farm, L_j is the j-th location where the temperature was monitored (Model 1), S_j is the j-th plant species, L_k is the k-th location where the plants were planted (Model 2). $(S\times L)_{jk}$ is the species by location effect at j- and k-th combination, and ε_{ijk} or ε_{ijkl} is the residual errors for Model 1 and Model 2, respectively.

Results

Farm type had a significant effect on plant foliar DM ($P\leq0.001$) but not on foliar N concentration (Table 2). The average DM concentrations of plants from the two layer farms were greater than that from the turkey farm.

Plant species and location showed significant effects on both foliar parameters (Table 2). Poplar was found to have less DM in its leaves than spruce (41.3 vs. 50.1%; $P\leq0.05$), but greater foliar N levels (3.43 vs. 2.46%; $P\leq0.05$). Planting location clearly showed greater foliar DM and N levels in those plants located near the exhaust fans compared with controls with 51.0 vs. 40.4% DM and 3.61 vs. 2.28% N, respectively. There was a significant species by location interaction with greater tissue DM again near the fans in both species, and a similar trend with tissue N levels, although not significantly so.

Table 2. Foliar DM and N (%) of hybrid polar (*Populus sp*) and Norway spruce (*Picea abies*) sampled at commercial poultry farms.

	DM (%)	N (%, DM)
Farm:		
Layer 1	47.2^a	3.16
Layer 2	53.7^a	2.91
Turkey	36.3^b	2.77
Species:		
Poplar	41.3^b	3.43^a
Spruce	50.1^a	2.46^b
Location:		
Control	40.4^b	2.28^b
Fan	51.0^a	3.61^a
Species × Location		
Poplar × Control	31.9^b	2.68
Poplar × Fan	50.6^a	4.19
Spruce × Control	48.8^a	1.88
Spruce × Fan	51.4^a	3.03
SEM	4.4	0.34
Sources of variances:	--------------Probabilities	--------------
Farm	0.001	0.484
Species	0.008	0.001
Location	0.001	0.0001
Species × Location	0.009	0.696

[a-b]Means in a column with no common superscripts differ significantly ($P\leq0.05$).

The temperature data presented in Table 3 showed that none of the factors (farm type or location) had an impact on the temperatures recorded throughout all four seasons ($P > 0.05$). However, temperature differences were realized in all seasons and followed the same pattern on all farms, ranging from -0.06 to 1.22 °C in winter, 13.56 to 16.96 °C in spring, 24.71 to 28.06 °C in summer, and 9.28 to 12.56 °C in fall

(Table 3). Summer T's were similar near the fans and at the control distance away from the fans on all three farms where plant tissues were sampled (25.80 vs. 25.32 °C [layer 1], 25.92 vs. 25.53 °C [layer 2], and 25.45 vs. 25.54 °C [turkey]). This indicated temperature stress was not an issue for trees near the fans.

Table 3. Average temperature recorded at two locations (control vs. downwind of the exhaust fans) on commercial poultry farms during the four seasons of 2005.

Farms (initial)	Winter		Spring		Summer		Fall	
	Control	Fan	Control	Fan	Control	Fan	Control	Fan
	(°C)[1]							
Broiler 1	0.26±7.14	0.28±7.08	14.68±7.78	14.10±7.02	28.06±7.53	27.33±7.10	10.57±7.41	12.56±8.75
Broiler 2	1.22±6.55	0.60±6.07	16.69±6.37	16.9±56.40	26.20±6.03	26.81±6.41	9.85±7.29	9.89±7.17
Broiler 3	0.16±6.27	-0.27±5.84	16.57±6.99	16.96±7.24	25.24±6.21	25.79±5.89	10.70±7.48	11.40±7.66
Layer 1	-0.06±7.54	-0.67±6.94	15.41±5.81	15.37±6.24	25.32±6.11	25.80±6.56	9.66±7.13	9.28±7.46
Layer 2	-1.20±6.50	-0.22±7.06	13.78±7.15	[2]	25.53±6.18	25.92±5.55	9.44±7.18	10.16±7.30
Layer 3	0.11±4.80	0.30±4.80	15.51±7.42	16.24±7.36	24.97±6.75	24.90±5.61	9.34±7.37	9.42±7.42
Pullet	1.38±5.11	0.99±4.95	15.96±7.54	16.74±6.24	25.51±6.90	24.71±6.47	9.79±7.57	9.50±7.31
Turkey	-2.08±6.79	-0.26±8.82	[2]	13.56±8.43	25.54±7.45	25.45±6.72	11.00±8.17	[2]

[1]Data (means ± SD) were recorded with data loggers programmed to read the temperature every 30 min for two months per season (winter: January to February; spring: April to May; summer: July to August; and fall: October to November).
[2]Technical failure of the data loggers resulted in no temperatures recorded at these locations.

Discussion

Ammonia concentration at the trees located near the exhaust fans was not measured in the current study. However, previous studies have documented ammonia losses from poultry farms (Liang et al., 2005; Miles et al., 2006; Patterson and Lorenz, 1996, 1997, Patterson et al., 1998; Patterson et al., 1999). Recently, Adrizal et al., (2006) demonstrated that plants grown in environmental chambers with atmospheric ammonia at 4 to 7ppm deposited almost 2-fold more N in their leaves compared to control plants in chambers without ammonia. Significantly great tissue DM was also observed in these studies, much like what was documented on the poultry farms herein. Although, not all plant species appeared to benefit from the ammonia exposure in the chamber studies. Hybrid poplar was the species with the greatest capacity to incorporate atmospheric ammonia into foliar tissue. However, the poplar also had the greatest tissue damage and negative color scores compared to the other species, suggesting upper limits to ammonia tolerance. The tissue injuries observed on the plants exposed to NH_3 in environmentally controlled chambers, however, were not apparent among plants on the poultry farms in the current field study. This suggests greater plant tolerance to NH_3 exposure under outdoor conditions that may included rain cleansing, and lower ammonia concentrations.

Reduced air velocity, dust, and NH_3 levels have been reported by Malone (2004) downwind of trees planted in front of poultry fans. Pitcairn et al. (1998) documented fewer numbers of nitrophilus plant species at distances downwind of the fans on livestock farms. Each of these demonstrates the importance of planting distance for the capacity of plant tissues to trap aerial NH_3-N emitted by the exhaust fans, and the sensitivity of some species. Lastly, temperature results from this study demonstrated that exhausted air temperature from buildings housing commercial poultry are not a plant stressor or affect the dormancy cycle of the trees planted within 11 to 18m from the fans.

Conclusions

Environmental temperatures monitored near commercial poultry house fans (11.4 to 17.7 m) did not differ from those at control distances from the fans (≥48 m) during all four seasons in Pennsylvania. Hybrid poplar and Norway spruce foliar N and DM concentrations were greater in the foliage sampled near the fans compared to the controls plants. Under the conditions of this study, both hybrid poplar and Norway spruce were able to trap aerial NH_3-N emissions from the fans and tolerate the concentration of ammonia realized under these field conditions.

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An Assessment of the Role of Terrain and Land Cover in the Development of Local Wind Flow Patterns: Development and Validation of the Land Use/Land Cover Dataset

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Abstract

Surface characteristics, such as terrain and Land Use/Land Cover (LULC) type, impose an important control on the land-atmosphere exchange processes leading to local wind flow patterns. In turn, these local wind flow patterns have a strong influence on the dispersion, distribution and deposition of atmospheric pollutants. As a result, a thorough understand of the role of the land surface in driving local wind flow patterns is necessary in order to accurately model the movement of these pollutants and assess air quality. In order to investigate the impacts of the land surface on these processes it is first necessary to develop am accurate LULC dataset. This research focused on the Walnut River Watershed (WRW) located in the Southern Great Plains (SGP) of the United States, a region that has been shown to be strongly affected by ozone, NO_x, and other pollutants. Specifically, this research compared two high-resolution (30 m) LULC datasets: the 1992 National Land Cover Dataset (NLCD 92) and the Kansas Gap Analysis Program dataset (GAP). While the two datasets agreed for the majority of the land area of the watershed, important differences due to classification and smoothing errors are evident. For example, small clusters of trees within riparian zones are often omitted from the LULC datasets. The differences resulted in an uncertain or inaccurate classification of approximately 27% of the total area of the WRW (Figure 1). Moreover, neither dataset achieved an overall accuracy of 80% with the NLCD 92 dataset having an overall accuracy of 68.7% and the GAP dataset having an overall accuracy of 79.2% within the WRW. Relationships between LULC type and surface characteristics, such as the normalized difference vegetation index (NDVI) and elevation, were developed using only those pixels where both LULC datasets agreed. For example, a strong relationship was found between LULC type and elevation. Using a reallocation scheme based on these relationships, the uncertain pixels within the WRW were reclassified. The resulting map (Figure 2) had on overall accuracy approaching 93%, more 24% greater than the NLCD 92 dataset and nearly 14% greater than the GAP dataset. In particular, the revised LULC dataset demonstrated an improved representation of Open Water, Wetland, and Wooded LULC types. For example, the User's Accuracy of the Wooded LULC type increased by more than 25% from approximately 71% in the case of either the NLCD 92 or GAP datasets to more than 96% in the case of the revised LULC dataset. This research represents

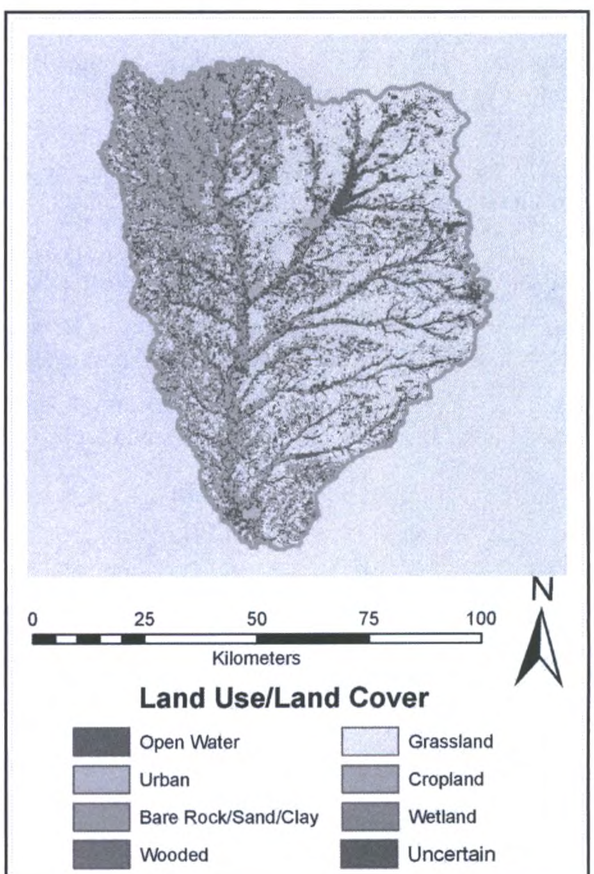

Figure 1. The LULC types within the WRW, as well as those areas with an uncertain classification, are shown.

an important first step toward understanding the influence of surface properties on both local wind flow patterns and the downwind transport and deposition of atmospheric pollutants. Future research will focus on the use of remote sensing and surface observations to quantify and analyze the wind flow patterns within the WRW and their relationship to heterogeneous surface characteristics.

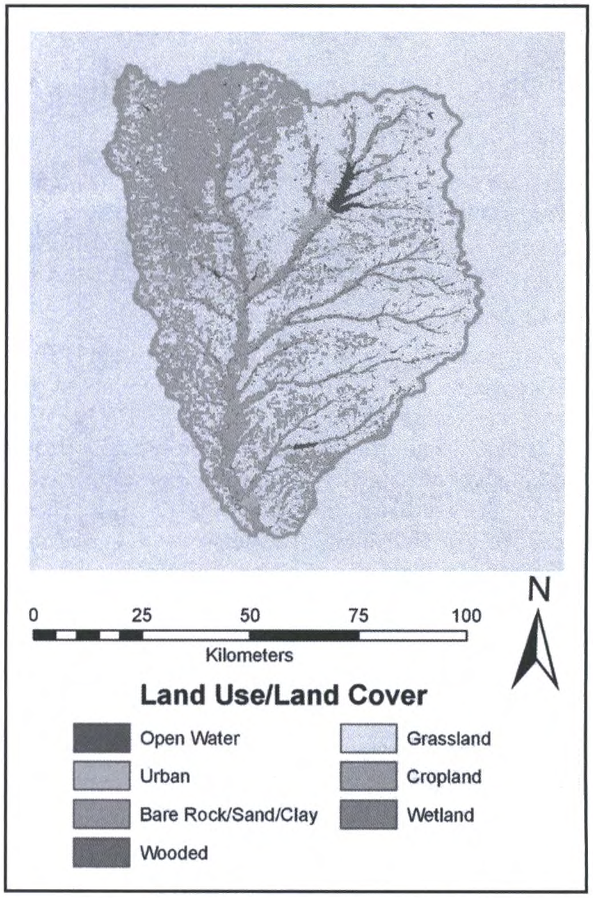

Figure 2. Using relationships between LULC distribution and surface characteristics, the revised LULC dataset shown was developed. It has an overall accuracy of 92.9%.

Dietary Modifications to Reduce Air Emissions from Broiler Chickens

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Abstract

The impact of feeding broilers reduced protein (LP) diets and control diets with industry protein and amino acid concentrations (C) on emissions of ammonia (NH_3), hydrogen sulfide (H_2S), nitric oxide (NO), nitrite (NO_2), carbon dioxide (CO_2), methane (CH_4) and non-CH_4 total hydrocarbon as well as on broiler performance and yields were determined. Broilers were housed in environmental chambers with continuously measured gas concentrations and airflows. Ross 308 male broilers were allocated to one of the chambers at hatch and grown for 42 days. Three sequential trials (flocks) were done to determine the impact of build-up litter. Broilers on the C treatment were fed based on a four phase feeding program: starter (St), grower (Gr), finisher (Fn), and withdrawal (Wd) diets. Broilers on the LP treatment were fed based on a six phase feeding program: prestarter (PreSt), St, Gr1, Gr2, Fn, and Wd diets. Formulated protein concentrations were 22.1, 20, 17.2, and 16.6% for the C St, Gr, Fn, and Wd diets, respectively while those for the LP PreSt, St, Gr1, Gr2, Fn, and Wd diets were 22.0, 18.6, 18.1, 17.3, 15.8, and 15%, respectively. Synthetic sources of lysine, methionine, isoleucine, valine, tryptophan, and argentine were used where needed to meet minimum requirements in the LP diets while only methionine and lysine were needed to in the formulation of the C diets. Body weight (BW) was determined by chamber at the start and end of each phase and feed disappearance (FC) determined at the end of each phase. Feed to gain ratio (FCR) was determined after correction for mortalities from the BW and FC data. Broilers on the C treatment weighed more at the end (42 days) over the three flocks (2.78 kg) than those on the LP treatment (2.69 kg) but FCR was similar (1.89 vs 1.91 for broilers fed C and LP treatments diets, respectively). Twenty broilers per chamber were randomly selected for yield determination in flock 3. Dress percent ((whole bird - gastrointestinal tract)/live weight) and breast yields (whole deboned breast weight/dress weight) was not affected by dietary treatment. Breast weight was also similar for both treatments (543 vs 541 g for broilers fed the C and LP treatment diets). Diet treatment affected NH_3 emissions in flock 1with the LP treatment resulting in lower daily NH_3 emissions (26.5 mg kg^{-1} in flock 1) compared to daily NH_3 emissions from broilers fed the C treatments diets (33.8 mg kg^{-1}). There was no effect of treatment on NO, NO_2, SO_2, H_2S, CO_2, CH_4, or non- CH_4 hydrocarbons in flock 1. Lowering dietary protein while maintaining minimum concentrations of amino acids resulted in substantial reductions in NH_3, NO, and NO_2 and no impact on breast weight or yield.

Introduction

Public and regulatory concerns related to air emissions from livestock and poultry operations has increased in recent years. Nuisance concerns with pollutants such as particulate matter, volatile organic compounds, ammonia (NH_3), methane, hydrogen sulfide, and odors have not subsided but there has been a refocused on human health implications. Outside of private lawsuits, state and federal regulatory agencies are seriously focusing on further determination of emissions to see if they fit current regulations or to determine if new regulations should be drafted. Of concern is the paucity of data on emissions from different livestock and poultry operations. An emission is the product of the concentration of the pollutant in question multiplied by a flow rate. Each of those factors has a unique set of measurement challenges associated with precision as well as methodology selection, that in most cases, still need to be resolved. In the absence of data, the regulatory community often will utilize the only information available – which can often produce less than adequate minimum standards. For example, regulatory agencies in California calculated poultry emissions based on emission factors for VOC from dairy cows. This was done under the misguided assumption that emissions should be same, for poultry and dairy, per unit of body weight (Mitloehner, 2005). Unfortunately, this is not the first time that policy preceded science, as we are entering an era where funding and timing of that funding relative to policy development does not go hand-in-hand.

Because there is very limited information in the published literature on the actual air emissions from broiler operations a study was done at a new air emissions chamber facility at Iowa State University to determine baseline air emissions of ammonia (NH_3), hydrogen sulfide (H_2S), nitric oxide (NO), nitrite (NO_2), carbon dioxide (CO_2), methane (CH_4) and non-CH_4 total hydrocarbon from broiler flocks grown on a "typical" industry phase feeding (control, C) and management program and on a reduced protein (LP) feeding program.

Materials and Methods

All animal procedures were approved by the institutional animal care committee. Ross 308 hatchling male broiler chickens were allocated to one of eight air emission chambers during each of three sequential 42-d flocks (50 chicks per chamber per flock). Chambers were randomly allocated at the start of each flock to one of two dietary treatment: a control treatment (C) that consisted of four feed phases: a 17-d starter (St, d 0 - 16), a 13-d grower (Gr, d 17 – 29), a 6-d finisher (Fn, d 30 - 35) and a 6-d withdrawal (Wd, d 36 – 42) and a reduced crude protein treatment (LP) that consisted of six feed phases: a 7-d pre-starter (PreSt, d 0 - 6), a 10-d starter (d 7 - 16), a 7-d grower 1 (Gr1, d 17 – 23), a grower 2 (Gr2, d 24 – 29), a 6-d finisher (d 30 - 35) and a 6-d withdrawal (d 36 – 41). Chamber body weight was measured and recorded at the start of each flock and on d 7, 17, 24, 30, 36, and 42. Feed disappearance was determined for each phase allowing for feed efficiency (feed to gain, FCR) calculations. Mortality was checked and recorded daily and feed efficiency was corrected for mortality. Diets were formulated to be isocaloric, by phase, were primarily corn and soybean based, but differed in protein content as follows: 22.1, 20, 17.2, and 16.6% for the C St, Gr, Fn, and Wd diets, respectively while those for the LP PreSt, St, Gr1, Gr2, Fn, and Wd diets were 22.0, 18.6, 18.1, 17.3, 15.8, and 15%, respectively. To maintain minimum amino acid (lysine, methionine, total sulfur amino acids, threonine, arginine, isoleucine, tryptophan, and valine) required concentrations in the diets synthetic amino acids were used. In C diets only synthetic lysine and methionine had to be used while in the LP diets lysine, methionine, threonine, arginine, tryptophan, valine and isoleucine were included in the diet. Diets were analyzed for dry matter, N, calcium (Ca), phosphorus (P), amino acids and selected microminerals.

At the end of flock 3, 20 broilers per chamber were randomly selected and sampled for yield determinations. Feed was removed 12 hours prior to sampling to ensure an empty body weight determination. Broilers were weighed individually, killed by cervical dislocation, the intestinal tract (from crop to cloaca) and abdominal fat pad removed for the determination of carcass weight. The breast was then removed and weighed. Clean wood shavings were sampled, placed in pans and weighed prior to the start of the first flock. Litter was weighed and sampled at the end of each flock for dry matter excretion determination as well as litter analysis (dry matter, nitrogen (N), calcium, phosphorus, and micromineral analysis) in order to provide estimates of volume and nutrient content excreted by broilers on each dietary treatment..

Throughout each flock, emissions of CO_2, NH_3, NO, NO_2, SO_2, H_2S, CH_4 and non-methane total hydrocarbons were made by sampling the incoming air for 20 min followed by sequential sampling of each of the 8 chambers for 15 min each. Average concentration of each gas during the last 5 min of each 15-min sampling period was recorded. This provided a total of 10-11 daily observations in each room. All sampling was automated. Analyzers employed for sample analyses included a TEI Model 17C ammonia/NO_x chemiluminescence analyzer and a TEI Model 45C H_2S/SO_x pulsed fluorescence analyzer (Thermo Electron Corp., Franklin, MA). Airflow (positive pressure system) through each room was measured every 30 sec using differential pressure transducers calibrated for the pressure difference across orifice plates.

Data were analyzed as a 2 x 3 factorial of 2 dietary treatments and 3 flocks (SAS v 8.0). Emissions data were adjusted for number of birds. Significance was accepted at or below a $P<0.05$.

Results and Discussion

Dietary treatment had an effect on final body weight (42 d) with broilers fed the LP treatment diets being 87 g lighter than those fed the C treatment diets (2.780 and 2.693 kg, respectively over the three flocks). Broilers fed the LP treatment diets were lighter at 42 d than those fed the C treatment diets in all three flocks but the weight differences were greatest in flock 1 (2.705 and 2.861, 2.841 and 2.907, and 2.534 and 2.573 kg for broilers fed the LP and C treatment diets in flocks 1, 2, and 3, respectively). Differences between the body weights of broilers fed the two diet treatments were evident as early as 24 d of age (1.072 and 1.101 kg for broilers fed the LP and C treatment diets, respectively). Feed consumption was lower in broilers fed the LP treatment diets resulting in no difference in FE between treatments C and LP (1.89 and 1.91, respectively over the three flocks). Mortality was similar between treatments (6%) with no flock or treatment effect observed. Others have observed decreases in body weight when diet protein is reduced (Neto et al., 2002; Bregendhahl et al., 2002) but it has been speculated (Burnham, 2005) that the decreases in performance observed by these researchers have been caused by not supplement back with sufficient amounts of limiting amino acids other than methionine and lysine. On a practical basis, however, bird performance can be hindered by excessive lowering of CP in diets due to a number of factors: reduced dietary potassium levels, altered dietary ionic balance, dedicated deficiency of nonessential amino acids, imbalances among certain amino acids (e.g. branched chain amino acids), and/or potential toxic concentrations of certain amino acids (Waldroup, 2000). These issues did not appear to be the cause for the lower body weight in the current study. Diet amino acid analysis showed that formulated and analyzed concentrations were similar. When looking at when differences started to appear and the proportrional differences at the different ages, it appears that the decrease in protein in Gr1 diet in the LP treatment may have been too severe. Despite differences in ending body weight the FCR was not affected.

In flock 3, when birds were sampled for yield determinations, treatment did not affect body weight, carcass weight, breast weight, or dress percent and breast yields. Numerical differences in body weight were similar to those seen in the three flock performance data. Body weight of sampled birds was 2.721 and 2.667 kg for broilers fed the C and LP treatment diets but because replication was low (one flock, four replicates of 20 birds per treatment) differences that were significant in the performance data were not significant in this portion of the study. Breast weights, however,were not different between broilers fed the C and LP treatment diets (542.9 and 540.7 g, respectively) and breast yields were 22.75 and 23.13% for broilers fed the C and LP treatment diets. Thus dietary treatment had no effect on breast weight or yield.

Dietary treatment affected NH_3 emissions with the LP treatment resulting in lower daily NH_3 emissions (26.5 mg kg^{-1} d^{-1} in flocks 1) compared to emissions from broilers fed the C treatments diets (33.8 mg kg^{-1} d^{-1} in flocks 1) . NO and NO_2 were also lower in the LP vs the C treatment. There was no effect of treatment on NO, NO_2, SO_2, H_2S, CO_2, CH_4, or non- CH_4 hydrocarbons. Lowering dietary protein while maintaining minimum concentrations of amino acids resulted in substantial (22%) reductions in daily emissions of NH_3 and no impact on breast weight or yield. Others have reported reduction in NH_3 when protein is reduced in the diet (Ferguson et al., 1998; Elwinger and Svensson, 1996). Elwinger and Svensson (1996) fed broilers diets containing 18%, 20% or 22% CP and measured NH_3 emissions from the litter bed. Total N losses in the houses averaged 18% to 20% of total N input. Schmidt et al., 2002 reported NH_3 emissions from turkey barns to be 592.6 and 12.3 mg kg^{-1} d^{-1}, in summer and winter, respectively. The data obtained in the current research on NH_3 emissions reflects a very dry environment since the work was during the fall and winter (September 2005 and February 2006). Litter dry matter content, that is typically between 70 and 73% in commercial houses was very high in the current study averaging 79 and 81% after flocks 1 and 2. The low moisture in litter may have contributed to a lower than expected NH_3 emissions but this can only be confirmed when litter concentrations are determined on whole nutrient balance can be calculated.

Acknowledgements

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Total Nitrogen Deposition on Land in the Northeastern part of Romania

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Abstract

Recent investigations on atmospheric deposition of N-containing compounds have indicated elevated nitrogen fluxes which can significantly affect both terrestrial and aquatic ecosystems (*e.g.* soil acidification, surface-water acidification and eutrophication of aquatic systems). In most of the estimations nitrogen budgets have focused on inorganic nitrogen, despite the fact that recent works revealed significant contribution from organic N-containing species to the total N-cycle.

This is the first report on a long-time series observation (27 months) on dissolved organic nitrogen (DON) deposition on land in the northeastern part of Romania (Iasi, one of the largest urban areas in Romania). Rainwater DON was determined by the persulphate oxidation method, while inorganic-N (NO_3^- and NH_4^+) using ion chromatography. An annual mean of 45.7 μM N has been estimated as volume weighted mean for DON, while the corresponding values for NO_3^- and NH_4^+ were 42.9 and 59.7 μM N respectively. DON can thus account for a significant part (up to 30%) of the total dissolved-N. Significant correlation coefficients have been observed between organic nitrogen and species as NH_4^+ and K^+ which give a first insight on its possible sources (soil and biomass burning).

Introduction

Most of the atmospheric nitrogen species are deposited to the Earth surface via precipitation. Beside dissolved inorganic nitrogen forms (DIN) the organic forms (ON) have been shown to be also important constituents of the atmospheric N deposition (wet and dry). Wet deposition is a major source of nitrogen input to ecosystems. The increased atmospheric N deposition is mainly contributing to the alteration of the normal operating functions of the aquatic and terrestrial ecosystems in different geographical locations. Anthropogenic sources have a strong impact on the global nitrogen cycle with most of the human activities responsible for the increase in global nitrogen being local in scale (*i.e.*, production and use of nitrogen fertilizers, burning of fossil fuels, power generation plants and industries).

Deposition of inorganic N-containing compounds has been found to be much higher in Romania than in any many other regions in Europe (Arsene et al., submitted to Atmospheric Environment, 2006). However, very little is known about the characteristics of nitrogen sources, rates of input and removal, and the effects of nitrogen deposition on the environment. This work reports on a long-time series observation (27 months) of total dissolved nitrogen (TDN) deposition on land in the north eastern part of Romania (Iasi, one of the largest urban areas in Romania).

Methods

Details in the analysis of inorganic species by ionic chromatographic method can be found in Economou and Mihalopoulos, 2002. The determination of DON in rainwater was made on the base of the total nitrogen measurements from which the DIN fraction concentration was subtracted. Persulfate oxidation-based conversion (wet chemical) of organic N to nitrate which is among other reliable measurement techniques (*i.e.*, UV photolysis and high-temperature catalytic oxidation techniques) was used for DON quantification.

This method was found to be more efficient at converting organic-N compounds to NO_3^- although the final results can be affected by significant non-systematic losses depending upon sampling, storage and analysis procedures (Scudlark et al., 1998). However, with appropriate optimization this method can give reliable results (Bronk et al., 2000). Comparison studies of the data obtained by different independent measurement techniques, *i.e.* UV and persulphate oxidation methods, indicates that the observation of the rainwater DON concentration is not the result of an analytical artifact (Cornell and Jickells, 1999).

Results and Discussion

Figure 1 shows the geographical location where the sampling procedure has been undertaken from May 1, 2003 to July 31, 2005 (47°20″ northern latitude and 27°60″ eastern longitude geographical coordinates).

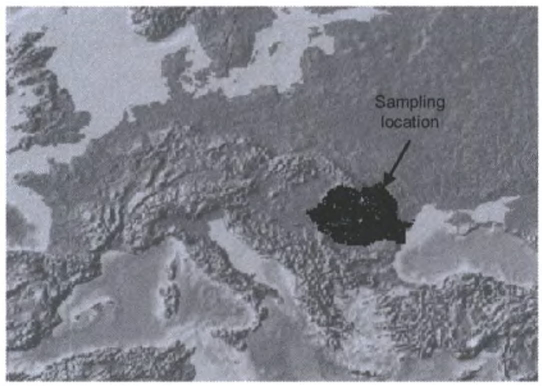

Figure 1. Location of the sampling site

Individual trajectory plots of the air masses origin were examined in details for all the period of interest. Five day backward trajectories were calculated in order to identify the most likely travel path of air masses affecting the surveyed region. The prevalence of air masses originating from the northern sector (N-NE-NW) with contribution from 40 – 60% over a year is the most obvious feature. Most of the raining events occurred in air masses originating from that sector.

The samples collected from May 2003 to July 2005 have been subjected to analysis for various nitrogen species and pH. Figure 2 shows the distribution both of pH and temperature over the study period. The mean pH value over a year is close to 5.6.

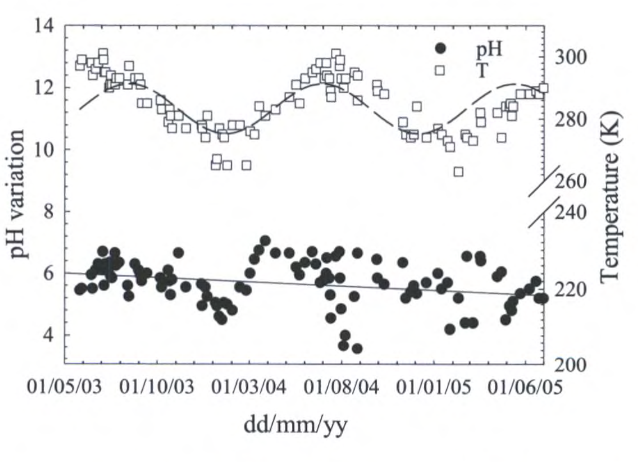

Figure 2. Variation of the temperature at the sampling site and rainwater pH distribution

Based on the volume weight mean (VWM) concentration inorganic nitrogen identified as NO_3^- and NH_4^+ appears to be the dominant fraction of N representing 70% of the total nitrogen. Ammonium concentration in the analysed rainwater samples ranged from 22.4 μM N to 155.1 μM N (mean value 59.7 μM N) while

NO_3^- ranged from 6.2 µM N to 93.8 µM N (mean value 42.9 µM N). For organic nitrogen the VWM concentration is found to be 45.7 µM N with a range from 14.2 µM N to 174.8 µM N.

Each of the quantified nitrogen-containing compound correlate well with the rain amount indicating these species are washed out of the atmosphere by rainfall. From the interspecies correlation analysis is obvious the correlation between DON and NH_4^+ which would suggest that these N species might have similar source in rainwater (Figure 3). Significant correlations have been observed between DON and K^+ as Figure 3 shows.

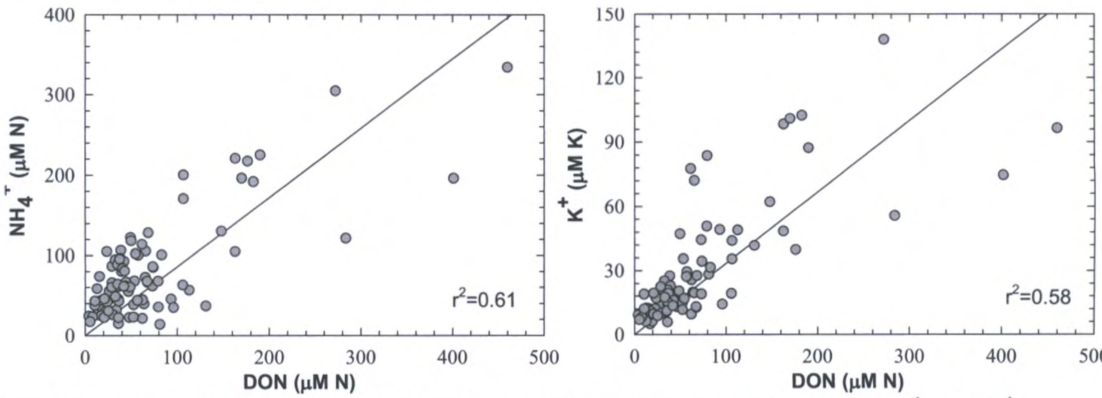

Figure 3. Correlation between concentration of DON and species like NH_4^+ and K^+

The VWM concentrations of N-containing species identified in rainwater collected at Iasi from May 2003 to July 2005 (seasonal variation) are given in Table 1. Seasons are defined as follows: winter December 1 – February 28, spring March 1 – May 31, Summer June 1 – August 31, autumn September 1 – November 30.

Table 1. VWM concentration of N-containing species in rainwater collected at Iasi, Romania.

	µM N DON	µM N NH_4^+	µM N NO_3^-	µM N TDN	DON (%) relative to TDN
Average	45.7	59.7	42.9	148.3	30.8
Winter	40.4	79.7	47.7	168.8	27.2
Spring	39.4	34.9	46.7	121.0	26.6
Summer	48.6	61.2	46.2	156.0	32.8
Autumn	54.4	62.8	31.3	148.5	36.7

The winter maximum concentration for NH_4^+ is most probable due to the application of fertilisers. In Romania during winter alternative cold and warm seasons occur so that sometimes the agricultural activities (especially application of fertilisers) peak during this season. The VWM concentration of DON was at maximum during summer and autumn but at the moment is difficult to make an assumption of the possible sources likely responsible for these peaks.

Figure 4 depicts fluxes variation of the identified N-containing species in order to better identify potential seasonal differences in N speciation.

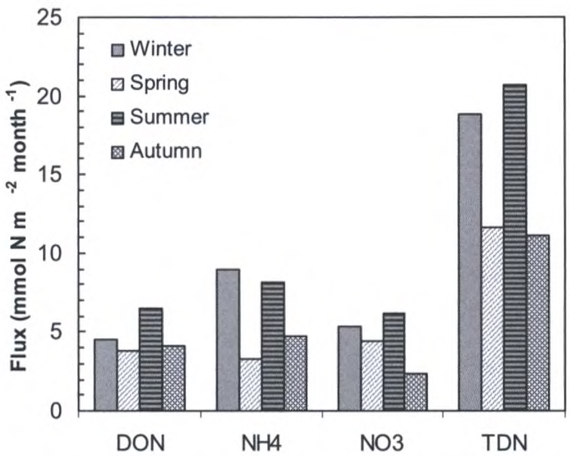

Figure 4. Fluxes of the various analysed N-containing species during different seasons

Conclusions

Rainwater collected between May 2003 and July 2005 was analyzed in order to identify inorganic and organic fraction of the nitrogen containing compounds. Inorganic nitrogen is the dominant form of N representing as high as 68 – 70 % of total nitrogen based on the monthly VWM concentrations. The results presented here suggest that DON also contributes a large fraction of total water soluble N in precipitation across Iasi region. It has been showed that DON concentrations are strongly correlated across the monitored site with those of NH_4^+, implying possible common sources. Elevated nitrogen-containing compounds concentrations like those observed in the monitored area might have a strong impact on terrestrial and aquatic ecosystems in the area so that an accurate quantification of the deposition fluxes of atmospheric N will be of a great importance for reliable predictions of the impacts.

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How Do Property-Line Concentrations of Feedyard PM_{10} Vary with Time of Day, Season and Short-Term Weather Phenomena?

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Abstract

We present particulate-matter concentration data from a long-term, quasi-continuous monitoring project around the perimeter of a commercial cattle feedyard in the Southern High Plains. Ground-level PM_{10} concentrations vary seasonally and diurnally as well as in response to short-term weather patterns. The data provide a suitable basis for annualizing PM_{10} emission factors originally derived from short-term, worst-case monitoring campaigns.

An Early Look at "Integrated Corral Management" as a BMP for Feedyard Dust Control

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Abstract

The economic value of fugitive dust control from cattle feedyards is an elusive quantity, which makes it difficult to assign cost/benefit analysis to Best Management Practices (BMPs). Integrated Corral Management is a means of leveraging existing feedyard-management approaches by putting advanced communications and software technologies into the hands of employees who are in the best position to provide feedback on environmental quality. These technologies give the feedyard manager new and more efficient options to integrate environmental considerations, including air quality, into the day-to-day, profit-oriented operation of a cattle feedyard.

Visibility Measures Can Be Used To Estimate Feedyard Dust Concentrations

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Abstract

Path-averaged measurements of total atmospheric extinction are a reasonable surrogate for mass concentration measurements of particulate matter (PM_{10} and TSP) downwind of cattle feedyards, provided that one accounts for the effect of relative humidity on the particles' refractive properties.

A Comprehensive Analysis of the Evening Dust Peak at Cattle Feedyards

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Abstract

The well documented peak in ground-level, fugitive dust concentrations that occurs near cattle feedyards in the mid- to late evening is a consequence of three primary factors: an increase in animal activity, an increase in atmospheric stability and a diurnal minimum in the moisture content of the uncompacted manure on the corral surface. We present a theoretical model to explain this phenomenon and to provide a rational basis for estimating the importance of each of those factors in predicting the ground-level, property-line concentration of PM_{10} at a given time of day. We also present preliminary data to illustrate animal behavior patterns in the cattle feedyard.

Ammonia Emissions and Dry Deposition Studies at Some Hog Farms in North Carolina

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Abstract

Experimental and modeling studies have been conducted on emissions, dispersion and dry deposition of ammonia taking one swine farm as a unit. Ammonia emission fluxes were measured during the years of 2002 to 2004. Measurements of ammonia flux were made at 11 swine facilities in North Carolina using a dynamic flow-through chamber system over the anaerobic waste treatment lagoons and other water holding structures, and each farm was measured during two 8-10 day periods representing warm and cold seasons. Barn emissions were also measured during those periods. Flux data was used to validate a process based Coupled mass transfer and chemical reaction model and an Equilibrium model developed by Aneja et al., (2001a) to better understand emission processes from lagoon surface. Several model performance statistics were used to evaluate the performance of the two models against the observed flux data. These indicate that the simpler Equilibrium model does as well as the Coupled model. Emission data from lagoon and barns is used to study short range downwind dispersion and dry deposition of ammonia from selected swine facilities using USEPA's AERMOD model.

Introduction

Ammonia is an important atmospheric pollutant that plays a key role in several air pollution problems. Ammonium salts remain a major component of inorganic atmospheric aerosols and thus NH_x (NH_x = ammonia + ammonium) plays a major role in the physical and chemical processes of the atmospheric nitrogen cycle (Warneck, 1999). Ammonia is gaining increasing importance, as a principle source of atmospheric aerosols (Baek et al, 2004). Gaseous ammonia undergoes dry deposition with deposition velocities ranging up to 14 cm/s (Phillips et al., 2004). Due to its high deposition velocity and its reactivity in the atmosphere, gaseous ammonia has a relatively short atmosphere lifetime, on the order of a few days or less (Warneck, 1999). NH_3 emissions from source region, primarily evolving from swine and poultry operations, are found to increase NH_4^+ concentration in precipitation at site up to ~ 80 km away (Walker et al., 2000). Results presented in NCDENR report (1999) demonstrate that ammonia (NH_3) emissions from swine production facilities can significantly enhance dry deposition of NH_4-N to adjacent forest canopies. Phillips (2003) showed that NH_3 contributed 47% of the total nitrogen dry deposited in the Neuse river watershed.

With the increasing human demand for food production, the use of nitrogen containing fertilizers and production of domestic livestock is increasing. Domestic animal waste is the major source of ammonia emissions. (Warneck,1999). Air mass trajectories suggest that wet and dry deposition of ammonia and ammonium emitted from agricultural operations in eastern North Carolina could potentially affect all river basins in the coastal plain region, as well as sensitive coastal ecosystem and estuaries (Walker et al., 2000). Such ecosystems are subject to potential environmental consequences, including aquatic eutrophication and soil acidification. High N-loading can also have detrimental effects in terrestrial ecosystems, effects that can result in the greater export of N to surface and groundwater.

Quantification of NH_3 emissions and deposition is necessary in order to assess the potential extent of such environmental effects. For reduction strategies to put in place, we need to know the total budget of ammonia and contribution of various sources. Livestock waste emission estimates used by EPA are based on emission factors recommended in a 1994 study by Battye et al.(1994) ,which are derived from European measurements, where animal practices may vary significantly from United states. Battye et al (2003) shows that livestock waste contribute from 80-89% of ammonia emissions in North Carolina. Emission rates from lagoons are difficult to measure, requiring specialized equipment. An emission model to predict ammonia emissions would reduce the expense of determining emissions on the large number of lagoons.

Measurements of ammonia flux from hog waste treatment lagoons (Aneja et al., 2001b; Aneja et al., 2000) and from fertilized and unfertilized soils (Roelle , 2001) have been made and analyzed with respect to corresponding environmental parameters, including lagoon and soil temperature, pH and TAN (Total Ammoniacal Nitrogen). Harper et al (2004) studied the ammonia emissions from hog lagoons using micrometeorological technique. Todd et al (2001) used a network of open-path fourier transform infrared (FTIR) optical ray method to measure ammonia emission rates from the hog lagoon. Studies have also been done to model ammonia emissions. Asman et al (1998) reviewed the ammonia research, process description and emission factors for ammonia emissions. Koelliker and Minor (1973) developed desorption model for ammonia emissions using two film theory. The overall mass transfer co-efficient (Halsam et al, 1924) in this model depends on wind velocity and temperature. This gives an emission of zero under calm conditions with no wind. Olsen and Sommer (1993) modeled ammonia emissions from stored slurry considering effects of wind speed and surface cover. A model to predict ammonia volatilization from flooded soils using total ammoniacal nitrogen (TAN), pH, temperature, floodwater depth and wind speed was developed by Jayaweera and Mikkelsen (1990). De Visscher et al (2002) developed a two layer model to study emissions from anaerobic lagoon. The model uses effluent concentration, water temperature, wind speed and effluent pH. Aneja et al (2001a) developed a Coupled mass transfer and chemical reactions model and Equilibrium model to simulate ammonia emissions from swine waste lagoons. This two layer model takes into account two film theory using molecular transfer of ammonia across the lagoon-air interface. It also takes into account pseudo-first order reaction of ammonia with water and acidic species (H_2SO_4, HNO_3, HCl) in the atmosphere. It incorporates air temperature, lagoon temperature, pH, wind speed, TAN and ambient ammonia concentration. This model shows exponential increase with lagoon temperature and pH and linear increase with wind speed and TAN.

This study includes comparisons of measured ammonia emission fluxes from swine waste treatment lagoon systems and modeled ammonia emission fluxes using both the Coupled and Equilibrium models developed by Aneja et al (2001a). Comparison of measured and modeled emission fluxes will help us to validate the model and also help us to make improvements in model or measuring techniques of required parameters in this model. This study will also focus on dispersion and dry deposition of ammonia downwind of swine facilities, to quantify ammonia dry deposition downwind of selected hog farms.

Sampling and Measurement

As a part of project OPEN (Odor, Pathogens and Emissions of Nitrogen), ammonia flux measurements were made at 11 swine farm operations in eastern North Carolina. The waste from the hog sheds was flushed out with recycled lagoon water and discharged back into the waste lagoon from the top, often with additional treatment using potential Environmentally Superior Technologies (ESTs). Each farm was sampled twice between 2002 and 2004, one representing the warm season and the other representing the cold season. Only the lagoon component of hog farm was investigated in this study. Fifteen minutes averaged measurements were made for ammonia flux and environmental data. This data was then averaged to one hour period for use in this study.

A flow through dynamic chamber system with a variable-speed continuous impeller was used to measure NH_3 emissions from lagoon surfaces (Aneja et al., 2000; Chauhan 1999; Kim et al., 1994). A Thermo Environmental Instrument Incorporated (TECO) Model 17C chemiluminescnce ammonia analyzer was used to monitor ammonia concentration during ammonia flux measurement periods. A 10 m meteorological tower was erected at each site to measure wind speed and direction, temperature and relative humidity. Wind speed and direction were measured at 10 m above the surface. Air temperature and relative humidity (RH) measurements were made at 2 m height. The pH and temperature probes were placed in the lagoon at depths of 15-20 cm. Lagoon water samples were collected daily from measurement sites and were analyzed for total ammoniacal nitrogen (TAN).

Mass Transport models

Two process-based models were developed by Aneja et al (2001a) to determine ammonia flux from a lagoon-air interface. The principle characteristic of these models are the two thin layers or films of air and liquid above and below the air-liquid interface for molecular exchanges between water and air, respectively (Whitman and Davis, 1923, Cussler 1996). All the resistance to mass transfer across the interface is due to the thin layer in which molecular transfer takes place. The steady state molecular diffusion equation for a horizontally homogenous thin layer in the liquid or gas (air) adjacent to the air-liquid interface is given by Arya (1999);

$$D_i \frac{d^2 C_i}{dz^2} = k_{ri} C_i \tag{1}$$

where C_i is the concentration of the diffusing material, z is the vertical distance from the interface, D_i is the molecular diffusivity, and k_{ri} is the reaction constant for ammonia in the liquid or gas phases such that $D_i = D_L$ and $k_{ri} = k_{rL}$ for liquid and $D_i = D_a$ and $k_{ri} = k_{ra}$ for air, respectively.

If we neglect chemical reactions in the two films, a simple equilibrium model can be derived from equation 2. Coupled Mass Transfer with Chemical Reactions Model (Coupled model) takes into account molecular diffusion and some chemical reactions. In the liquid film, only ammonia's reversible reaction in the water is considered, and pH is assumed constant. For the air film, the primary reactions of ammonia with sulfuric acid (H_2SO_4), nitric acid (HNO_3), hydrochloric acid (HCl), water, and the hydroxyl radical (OH) are considered (Finlayson-Pitts and Pitts, 2000; Warneck, 1999). Further details of these models are given by Bajwa et al (2006).

Results and Discussions

The Coupled mass transfer and chemical reaction model and the Equilibrium model show strong dependence on lagoon temperature, lagoon pH, total ammoniacal nitrogen (TAN) and wind speed. The modeled ammonia flux increases exponentially with the increase in lagoon temperature. Lagoon temperature affects the Henry's law coefficient, the liquid phase diffusivity of ammonia and ammonium, the dissociation constant, and viscosity and density of the liquid layer. The pH of a waste treatment lagoon controls the chemical equilibrium between ammonia and ammonium, and an increase in pH increases the fraction of ammonia in the solution. With an increase in pH, both models show an exponential increase in ammonia flux. Lagoon TAN controls the concentration of ammonia in lagoon water. Any increase in lagoon TAN gives a corresponding linear increase in ammonia flux. Since the air film is a laminar sub-layer, it is affected by meteorological and environmental parameters such as wind speed and stability. Thus wind speed may also affect ammonia emissions. Sensitivity analysis of both models shows a polynomial (nonlinear) relationship between ammonia flux and wind speed. Air temperature and ambient ammonia concentration did not show any significant effect on the ammonia flux for both the models (Bajwa et al).

Ammonia flux and most of the lagoon and environmental data were averaged over 15 minute intervals during the measurements periods for both the warm and cold seasons. But for the present analysis, those data were further averaged for 1 hour periods. Although flux measurements were made for longer periods, only those hourly data are used for which all the required meteorological parameters (wind speed and air temperature) and lagoon parameters (lagoon pH, lagoon temperature and total ammoniacal nitrogen) were available.

Various statistical measures have been proposed and utilized for evaluating the performance of air quality and dispersion models (see e.g., Irwin, 1983). Here, we have used a few simple statistical parameters, such as mean bias (MB), normalized mean bias (NMB), and normalized root-mean-square error (NRMSE) as given by Irwin (1983). These statistics were calculated for both the warm and cold seasons, separately, as well as for the combined data set.

Table 1 compares the mean flux, mean bias, normalized mean bias and normalized root-mean-square-error (NRMSE) for both the Equilibrium and Coupled models. These model performance statistics indicate slight superiority of the simpler Equilibrium model over the more complicated Coupled mass transfer with chemical reactions model. Percentages of hourly model predicted fluxes that are within a factor of two of the observed hourly fluxes are comparable for the two models; these are less than 45%. We have also examined separately the cases of gross over and under predictions (by more than a factor of 5) by the Equilibrium model.

Table 1 Statistical performance parameters for Equilibrium and Coupled models

Statistical Parameter	Equilibrium Model prediction			Coupled Model Prediction			Observed		
	Warm season	Cold season	Combined data	Warm season	Cold season	Combined data	Warm season	Cold season	Combined data
Number of Hours (N)	706	868	1574	706	868	1574	706	868	1574
Mean Flux (µg NH_3-N/m^2-min)	1150.8	944.8	1037.2	2210.8	1391.3	1758.9	1545.1	538.1	989.7
Mean Bias (µg NH_3-N/m^2-min)	-394.3	406.8	47.5	665.7	853.3	769.1	-	-	-
Normalized Mean Bias	-0.26	0.76	0.05	0.43	1.59	0.78	-	-	-
NRMSE	1.76	1.96	2.01	3.58	2.69	3.90	-	-	-
% Within a Factor of 2	38	43	41	45	41	43	-	-	-

Conclusions and Future Work

Ammonia flux measurements were made on swine waste treatment lagoons using a dynamic flow through chamber system. Hourly averages of wind speed, lagoon temperature, TAN, lagoon pH and air temperature were used as inputs into the two thin-film mass transfer models to predict ammonia flux and these predictions were compared with hourly averaged values of measured ammonia flux. Measurements made in the warm and cold season were analyzed and modeled separately to look into the seasonal differences between measured and predicted ammonia fluxes. Measured ammonia fluxes were higher in the warm season as compared to the cold season as high lagoon temperatures in the warm season lead to increased ammonia fluxes.

Both the Equilibrium model and the Coupled mass transfer with chemical reactions model predicted ammonia flux reasonably well in both seasons. Observed ammonia flux falls between predicted fluxes by the Equilibrium and Coupled models in the warm season, while both models overpredicted ammonia flux in the cold season. Equilibrium model predictions gave lower value of NRMSE and bias than the Coupled model predictions in both seasons. Average of predicted fluxes by both models were within a factor of two of observed fluxes in both the warm and cold season, except by the Coupled model in the cold season when the mean was more than twice the observed flux. Equilibrium model gave more consistent results as NRMSE and bias varied less between both seasons as compared to the Coupled model results.

Further analysis of data available from all 11 farms will be carried out to validate Coupled mass transfer and chemical reaction model and Equilibrium model. More statistical analysis of modeled and measured flux will be done to calculate over prediction or under prediction by models to measure accuracy of predicted values. We will also examine separately the cases of gross over and under predictions (by more than a factor of 5) by the Equilibrium model. AERMOD model will be used to study dispersion and dry deposition of ammonia downwind of some swine facilities.

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Odor Emission Reduction from Enclosed Growing-Finishing Pig House Using Different Biofilter Media

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Abstract

This study was conducted to determine the odor reduction efficiency of a biofilter design using different filter materials. The summary of results are as follows; 1.The airflow penetration rate of the different filter materials namely; rice straw, woodchips, rice hulls and sawdust were 0.72 m/s, 0.64 m/s, 0.48 m/s and 0.17 m/s, respectively. 2. Compared to a pig barn with no biofilter, NH_3 emission was reduced by 77 % using a biofilter media of sawdust and wood chip followed by rice hull and rice straw with a removal efficiency of 69% and 46%, respectively. Based on a two bi-weekly monitoring of NH_3 emission, wood chips as a biofilter media proved to be the most superior and consistent in reducing this gas due to removal efficiencies of 76 %, 55% and 76% for days 7, 14 and 36' respectively. On the other hand, rice hull was the most effective among biofilter media in reducing H_2S with 86.27 % removal efficiency during a two consecutive bi-weekly monitoring period. The above findings also demonstrated that H_2S could be easily trapped/absorbed effectively by all the biofilter media than NH_3. Finally, the airflow penetration rate of the different biofilter media tended to be related to odor elimination efficiency with having the slowest penetration rate of 0.17 m/s.

Introduction

A biofilter is a bed of organic material (rice straw, sawdust, rice hull, wood chips etc) where odorous air can passed through (Fig, 1). It is important however that the right moisture in the filter material should be attained for the survival of microbes that are naturally present in the different filter media. The material filters dust from the air and serves as a host for microbes that convert odorous gases into non-odorous gases. Analysis of livestock odor (there are around 411 odor causing compounds) has been done during the last 30 years and majority of them were identified and described by Wright et al (2005). Song et al. (2005) was able to quantify a reduction in daily gain and feed conversion efficiency of pigs (5 kg to 31 kg) by 22 % and 9 %, respectively when indoor $NH_{3\ \mathrm{was}}$ increased from 3.73 mg/liter to 4.91 mg/l.

The effectiveness of the biofilter depends upon the length of stay of the odorous air in the filter long enough for the odorous gases to be trapped on the medium containing the microbes to neutralize the malodor compounds by way of biochemical reaction. Two gases typically found in air from swine facilities are hydrogen sulfide and ammonia, and properly operating biofilters can remove 80 to 95% of those gases.

It is on this premise that, varying biofilter media were assessed to determine their varying removal efficiencies of malodors from an enclosed pig barn.

Methods

The biofilter was installed in a two-storey high-rise enclosed pig barn (HRHB) at the National Livestock Research Institute in Suwon, South Korea. The HRHB has an air flow capacity of 7,200 m^3/h and and the size of the exhaust fan is 630 mm. On the other hand, the biofilter had a width of 2,000 mm, length of 2,400 ppm and height of 600 mm. The duct size was 730 mm. The NH_3 gas was collected using a handy gas sampler (Gastec GV-100, Shimoto, HS-7) (Figs 1 and 2) capable of detecting 0~30 mg/ℓ and 0~1,000 mg/ℓ for NH_3 and CO_2, respectively and measured by Gas Chromatography. The bedding materials used were rice hull, rice straw, saw dust and wood chip. The periodic changes when the indoor air, which served as the control, was sucked out and allowed to pass through the different biofilter media was monitored on days 7, 21 and 36.

Results and Discussion

In terms of persistency of the different filter media in neutralizing the malodors, woodchip proved to be the most effective in neutralizing NH_3 gas as it had the lowest concentration on the 36th day at 10.7 □/□, followed by sawdust, rice straw and rice hull at 12.3 □/□, 21.5 □/□and 35.3 □/□, respectively. Therefore, woodchips were able to reduce almost 80% of the NH_3 gas inside the pig house, which means a significant improvement in the quality of air to be borne by the neighborhood.

Table 1. Periodic NH_3 concentration changes after biofilter media penetration*

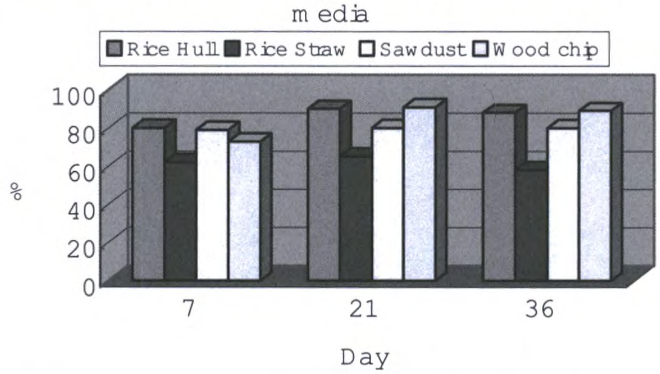
Items	Duration, days		
	7	21	36
Indoor pig house[1)]	7.6	12.3	51.9
Rice hull, □/□	2.7	19.2	35.3
Rice straw, □/ℓ	4.3	7.5	21.5
Sawdust, □/ℓ	1.8	5.5	12.3
Wood chip,□/ℓ	2.2	17.6	10.7

*Measured by gas chromatography

1) Exhaust gas concentration

Likewise, in terms of persistency rate for the entrapment of the H_2S by individual biofilter media rice hull was the most consistent during the two consecutive bi-weekly monitoring at 86.27% followed by wood chip, saw dust and rice straw with values of 84.66%, 79.53% and 62.08%, respectively. The high carbon nature of the rice hull and wood chip as compared to the other media apparently contributed to effective absorption and utilization of the microbes in the biofilter media.

Figure 1. Periodic removal rates of the different biofilter media



In terms of on the spot dust and gas entrapment, (Table 3) by the individual filter media, the absorption of NH_3 was very effective in both the sawdust and wood chip biofilter media as their recovery of the gas amounted to 3 □/□in contrast to rice hull and rice straw which is 4 □/□and 7 □/□, respectively. It is interesting to note also that H_2S, which is one of the most offensive odors, was no longer detected in the rice hull, sawdust and woodchip filter media, while in the rice straw it was barely detected. Being the least dense among the filter media, it was expected that dust level was the highest in the rice straw. Woodchip on the other hand prove to be the most effective in absorbing CO_2 although this is not given emphasis in the study, as it is not considered a malodor. However, prolonged exposure of this gas inside livestock houses can be detrimental also to the health of animal caretakers.

Table 3. Concentration of gas and dust after penetrating the different biofilter media

Items	NH_3, □/ℓ	H_2S, □/ℓ	Dust, □/ /m^3	CO_2, □/ℓ
Indoor pig house*	13	4	93	780
Rice hull	4(69.23) ***	N.D**.	53(43.01)	510(34.61)
Rice straw	7(46.15)	1(25)	99(6.45)	550(29.48)
Sawdust	3(76.92)	N.D.	38(59.21)	380(51.28)
Woodchip	3(76.92)	N.D.	41(44.08)	470(39.74)

*Reference value

**Not detected

***All values in parenthesis are reduction values relative to indoor pig house level

Conclusions

It is therefore clear that based on our confirmatory experiments, biofilter is very effective in reducing malodors and with the abundance of rice straw, rice hull, sawdust and wood chips, there is no reason why we cannot adapt this as part of a livestock waste management strategy. Various agencies in our government can promote biofilter to both the backyard and commercial sectors. Meanwhile, companies in the contract growing franchise for broilers and hogs can recommend biofilter as part of their housing innovations as the added investment which is very minimal can be offset by the improvement in the growth when malodors particularly ammonia is reduced

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A Field Survey on Concentration of Odor Compounds in Pig Buildings and Boundary Areas

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Abstract

A field survey was done to determine the levels of odor emission from pig buildings and quantify the odorants and their respective reduction rates when detected 20 meters within the boundary area. The odorants were measured from large, medium and small farms with enclosed and open housing systems and slurry and sawdust manure fermentation treatment methods. Among the odorous matters investigated, NH_3 had the highest emission level at 0.9 to 21 ppm; followed by Hydrogen Sulfide(H_2S) with a highly varied concentration of 51.9 to 6,712.4 ppb , methylmercaptan (CH_3SH) ranging from non-detectable level (ND) to 27.64, dimethylsulphide ($(CH_3)_2S$) was measured from ND to 2.6 ppb. Considering the prevailing wind direction and air velocity ranging from 0.23 to 0.73 m/s within the boundary area, the odorous matters; NH_3, H_2S, CH_3SH, $(CH_3)_2S_2$ and $(CH_3)_2S$ were 0.2 to 4.5 ppm, 0.01 to 0.06 ppb, ND for$(CH_3)_2S$, respectively. These findings suggested that the odor compound, $(CH_3)_2S_2$, had the lowest detection level in the boundary area whilst $(CH_3)_2S$ cannot be detected within a 20-meter distance only. However, with these results, other odor compounds from pig buildings has to be further investigated under more controlled environmental factors.

Introduction

Nowadays, there has been an increase in intensive farming resulting to increased concentration of malodors. Consequently, numerous researches have been done to reduce malodors. For efficient reduction of malodors from livestock facilities, elimination of odor causing compounds should be given priority. Another important aspect of livestock waste management is the extent to which these malodors can be detected at a certain distant from the pig barn. This is equally important to determine also the stability or persistency of the odorous nature of the malodorants.

The distance from the pig barn to the nearest neighbor is critical in minimizing air quality impacts, such that the closer the neighbor, the more important odor control efforts become (ISU, 2004).

Methodology

There were 16 farms, with size ranging from small to large scale operation. The building length ranged from 4 to 45 meters while the width ranged from 9.7 to 43 meters. The manure treatment strategies were described including the ventilation types. The pig house facilities were gestating and breeding, nursery and growing houses. The air velocity was also measured to serve as a variable for standardizing the possible transit characteristics of the varying odor compounds. The malodorants were trapped using a handy gas sampler (Gastec GV-100,Shimoto, HS-7) Fig.1, equipped with a collection bag unit. Malodorants were detected using Gas Chromatography. The carrier gas wa He with inlet temperature at 150 □ (Split ratio 10:1), Column : HP-1 (30 m * 0.32 mm * 4 um), Column flow : 2.4 ml/min, Oven temp. : 40 □(2 min)-10 □/min-160 □(5 min), Detector : SCD(Temp. : 800 □, H2 : 100 ml/min, Air : 40 ml/min).

Results and Discussion

During the time of collection, the environmental conditions in the varying pig farms are presented in Table 1. Air velocity ranged from 0.09 to 0.73 m/s with the boundary area usually having the highest velocity. As expected the temperature outside in the boundary areas were slightly lower than the inside temperature of the different pigggery house sections. Relative humidity was quite high in one boundary area at 90% while the other was only 69%.

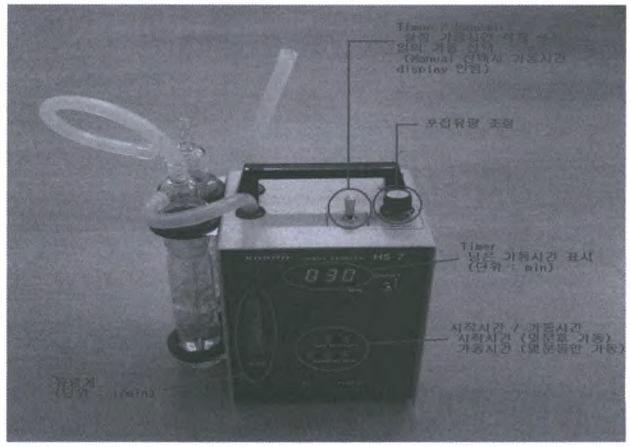

Figure 1. The portable gas sampler

As presented in Table 2, the nursery section had the highest concentration of NH_3 emission at 6.68 mg/m^3, which could apparently be attributed to the high protein diet of the piglets. Temperature inside the nursery houses are also higher as compared to the other sections of the piggery house. This could be one factor for the variation. There was a heavy concentration of H_2S in the gestation and breeding section at 2,841.73 μg/m^3 . The massive concentration of feces of the sows and the contamination with the high protein diet of the sucklings could have resulted to high emission of this odor matter. methylmercaptan $(CH_3)SH$ were detected to be almost similar in emission levels from all the piggery sections while dimethylsulphide $(CH_3)_2S$ was maximized in the nursery section.

Table 1. Environmental conditions from the source of odor matter

Item*	Pig house	Air velocity (m/s)	Inside Temperature (□)	Relative-Humidity (%)
L	Gestation & breeding	0.20	25.0	84
	Farrowing	0.18	26.0	77
	Nursery	0.09	27.0	84
	Growing□finishing	0.24	25.0	83
	Boundary area	0.23	23.0	90
M-1	Gestation & breeding	0.22	22.0	75
	Farrowing	0.09	21.3	75
	Nursery	0.13	25.2	82
	Growing-finishing	1.11	21.3	81
M-2	Gestation & breeding	0.15	-	-
	Farrowing	0.33	28.6	71
	Nursery	0.19	-	-
	Growing-finishing(sawdust)	0.15	30.0	84
	Growing□finishing(slurry)	0.33	29.0	82
S	Gestation & breeding	0.10	25.0	74
	Farrowing	0.70	26.0	73
	Nursery	0.23	25.0	82
	Growing□finishing(sawdust)	0.30	25.0	76
	Boundary area	0.73	25.0	69

Table 2. Concentration of odor matter emissions from pig sections

Item		Gestation & breeding	Farrowing	Nursery	Growing □ finishing
NH_3 (□/□)	Arithmetic mean	□3.19	2.66	6.68	4.71
	Range	1.59~5.08	1.29~4.63	0.68~15.94	1.14~8.58
H_2S (□/□)	Arithmetic mean	2,841.73	908.59	289.15	1,549.58
	Range	78.78~10,188.46	64.21~3,313.33	44.47~904.95	64.21~4,406.64
$(CH_3)SH$ (□/□)	Arithmetic mean	21.86	21.00	24.64	22.07
	Range	N D~23.79	20.79~21.21	21.00~27.64	20.57~24.00
$(CH_3)_2S$ (□/□)	Arithmetic mean	7.75	5.81	11.35	8.30
	Range	6.64~8.30	N D~ 8.30	8.58~14.12	N D~14.39
$(CH_3)_2S_2$ (□/□)	Arithmetic mean	9.23	9.23	9.65	5.46
	Range	9.23~9.65	8.81~9.65	8.81~10.91	N D~9.23

Ammonia (NH_3) and hydrogen sulfide (H_2S) are two major odorous gases emitted from animal operations (Xue and Chen, 1999). These gases are generated while manure undergoes microbial degradation. Ammonia is produced by the decomposition of nitrogen-containing compounds in the excreta, especially in urine.

In Table 3, against the prevailing wind direction and air velocity ranging from 0.23 – 0.73 m/s within a 20-m boundary area, the odor matters namely; NH_3, H_2S, ($(CH_3)SH$), (($(CH_3)_2S$) and ($(CH_3)_2S_2$) were substantially reduced to 0.15~3.42□/□, 20.34~104.43□/□, ND~21.00□/□, ND~9.65□/□ and ND for.$(CH_3)_2S_2$, respectively. Likewise the, reduction rates for the odor matters in the same order were; 78.24%, 76.61%, 24.02%, 32.94% and 100 %, respectively. These findings also suggested that the methylmercaptan $(CH_3)SH$ was highly concentrated and stable as it was still strongly detected as evidenced by its low reduction rate in the boundary area whilst dimethyldisulphide$(CH_3)_2S$ was the first odorant to be eliminated because of its zero detection level within a 20-meter distance only. Finally, dissipation of odorants from pig buildings has no relationship to their respective molecular weights but has to be further investigated under more controlled environmental factors.

Table 3. Concentration of odor matters in boundary area

Item	L	M-2	S
NH_3(□/□)	2.13	3.42	0.15
H_2S(□/□)	104.43	49.03	20.34
$(CH_3)SH$(□/□)	21.00	N D	20.79
$(CH_3)_2S$(□/□)	N D	N D	N D
$(CH_3)_2S_2$(□/□)	9.65	N D	8.39

Conclusion

Detection of odor matters within boundary areas remains a critical challenge to environmentalists as they are governed by several factors. The information in this field survey will therefore serve as a basis for devising scientific approaches to accurately quantify odor matters and recommend appropriate measures to mitigate their emission.

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The Effects of Coordinate Rotation Procedure on Eddy Covariance and Relaxed Eddy Accumulation Flux Measurements

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Abstract

Quantifying the exchange of traces gases between the atmosphere and agricultural operations (crop or animal) is crucial for advancing research on agricultural air quality. Micrometeorological techniques such as eddy covariance, relaxed eddy accumulation (REA), and the modified Bowen ratio are the most direct, unintrusive methods for measuring surface atmosphere exchange. All of theses techniques rely on three-dimensional sonic anemometry. Data from a sonic anemometer and the fluxes calculated from the results are subject to a mathematical processing step known as coordinate rotation before the fluxes can be interpreted. The objective of this research was to evaluate the two most widely used rotational schemes: (1) rotation to the natural wind coordinate (NWC), and (2) rotation using the planer fit method (PF). The traditional NWC procedure is a double or triple rotation procedure that aligns the mean wind direction with the x-axis and sets the cross-stream stress to zero every observational period. The PF method determines the same rotational angles by defining a three-dimensional 'plane' using historical three-dimensional wind data. Eddy covariance systems consisting of a sonic anemometer and a fast response open-path gas analyzer were deployed at 3-m above the soil surface at different locations in Kansas representing three different land/cattle management schemes: an ungrazed prairie, a grazed prairie, and two cattle feedlots. The effects of coordinate rotation were examined by comparing sensible heat, latent heat, and carbon dioxide fluxes using the two different rotation procedures. Additional work was conducted to examine the effect of rotation schemes on the measurement of ammonia fluxes when using REA. The differences between these two methods could be very important for estimating nutrient balances and calculating emission factors.

Ammonia Emissions from the Application of Dairy Effluent

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Abstract

The San Joaquin Valley (SJV), in the interior portion of California, is one of the most productive agricultural regions in the world where most of the state's 5 billion dollar dairy industry is located. Because of the its extent and continuing growth, agiculture is the major source of ammonia (NH_3) in California with dairies accounting for roughly 60% of statewide NH_3 emissions. Ammonia is of concern because of it's participation as a precursor to $PM_{2.5}$ as well as the passage of California State Bill 700 which has brought increased scrutiny to NH_3 emissions. The SJV experiences elevated levels of $PM_{2.5}$ usually in winter months. The dominant manure management practice at the dairies is a free stall flush system where water is flushed through barns where animals are housed and fed removing manure and depositing it eventually to a lagoon or storage pond. The effluent is applied several times a year through irrigation systems to surrounding crop land as a means of applying nutrients, water, and disposal of effluent. Dairy producers and agronomist are currently working to develop nutrient mangement plans to apply agronomic rates of nutrients while achieving maxium crop yields of forages. Losses of nitrogen occur through the volatilization of NH_3 from the application of dairy effluent. In order to develop accurated nutrient management plans these losses must be quantified.

Because free stall flush manure management systems are not common throughout the national dairy industry little research has been conducted on NH_3 emissions from these systems. Preliminary NH_3 emissions data from dairy manure applications was collected in the winter of 2005/2006 for a project to calibrate a computer simulation model of carbon and nitrogen biogeochemistry for California dairy systems. As part of this project emissions of NH_3 and nitrous oxide (N_2O) will be monitored from California dairy cropping systems focusing on manure applications.

NH_3 emissions were monitored form two manure appication methods in January and February 2006. A slurry injection method was monitored along with a more traditional application of effluent through flood irrigation. The injection of dairy slurry is rare with less than six producers known to use this practice in the state. Ammonia emissions were monitored using a USEPA Emission Isolation Flux Chamber. Samples were collected with active chemical filter packs consisting of 47mm glass microfibre filters treated with citric acid (3% in 95% Ethanol). The ammonium citrate was extracted from the filter with de-ionized water and analyzed with Nessler's Reagent and a spectrophotometer. Air was pulled out of the flux chamber with a personal air sampling pump at a flow rate of 2.0 L min^{-1} to sampling media for 15 to 25 minutes. Ultra zero air was used to flush chamber at a flow rate of 5.0 L min^{-1} for thirty minutes. Fluxes were calculated with formulas provided in the Measurement of Gaseous Emissions Rates from Land Surfaces Using an Emission Isolation Flux Chamber User's Guide.

Slurry injections were monitored two dates post-application, immediately and one week. Flux chambers were placed at the same undisturbed locations for the one week post-application samples. Three types of surface characteristics sampled by the flux chamber were evaluated, representing 25, 50, and 75% manure coverage of area, respectively . Percent total solids (TS) of slurry was 11% while the liquid effluent was estimated to be approximately 1 to 2 % TS. Ammonia emissions from the flood irrigation application of liquid effluent to a fallow field was sampled two and four weeks post-application. Two different surfaces were evaluated to characterize emissions, presence of solids and no solids on the surface of the field. All sampling was conducted from 9:30 to 16:00. Ambient temperatures ranged from 10 to 16 °C.

Ammonia fluxes from the slurry injection ranged from 1.2 to 352.1 μg NH_3-N m^{-2} min^{-1} The highest 352.1 μg NH_3-N m^{-2} min^{-1}flux from the slurry injection occurred at 30 minutes post injection, at a location where 75% of the flux chamber area was covered with manure. Surface coverage of slurry seemed to be positively correlated with ammonia fluxes. The 25% manure surface area was determined to be most

representative of the total field from on site observations and consultation with the farm manager. Soil moisture conditions may have increased the surface area of the field covered with slurry as higher than normal levels of moisture did not allow for efficient injection. Emissions of ammonia at the 25, 50, and 75% manure coverage locations decreased by 99, 87, and 65% of initial fluxes sampled immediately post-application.

Ammonia fluxes from the liquid effluent flood irrigation application ranged from 4.4 to 116.9 μg NH_3-N m^{-2} min^{-1}. Fluxes of ammonia were generally higher at the head end of the irrigation check where more solids have settled out compared to the tail end. From two weeks post-application to four weeks post-application fluxes decreased by over 90% at both locations. It is assumed that the period of highest emissions would be during and immediately post-application of the effluent which was not sampled. In order to better understand emissions characteristics from this type of application sampling must be done during this period as well as during night hours to characterize assumed diurnal variations in emissions.

Future sampling will be conducted during the Spring and Fall months which are the most common periods to apply dairy effluent in California between cropping rotations of winter forage and summer corn silage. Application of dry manure from scraped corrals or solids separation processes will be monitored in addition to the injection and liquid effluent flood applications.

Improved Temporal Resolution in Process Modelling of Nitrogen Trace Gas Emissions

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Introduction

An emerging issue in Canada is how to better quantify agriculture's contributions to atmospheric warming and pollution contamination generated by nitrous oxide (N_2O) and ammonia (NH_3) emissions. Agricultural soil emissions have far-reaching environmental impacts including stratospheric ozone destruction, global warming, ecosystem degradation, and contributions to secondary organic aerosol production causing adverse human health effects. Emission inventories are essential to predicting these emissions and their subsequent atmospheric transportation, transformation, and deposition. These inventories can also predict longer term trends and assess emission reduction strategies. The processes of soil N transformations and transport are complex because they are controlled by site specific factors (i.e. soil temperature, acidity, etc.) influenced by weather, vegetation, soil properties, and land management. Due to the high spatial and temporal variability associated with these processes, measurement-based emission inventories become expensive and emission factors can lose accuracy. Process-based models are capable of developing emission factors that account for the complex soil interactions, but current models lack temporal refinement in that they operate on a daily time step and few models consider NH_3 emissions.

Proposed model

The proposed model, AGRIN, seeks to develop a one-dimensional (vertical), time-dependent process model that predicts hourly N trace gas emissions from N applied to a bare soil in a northeastern Ontario climate. The model takes existing model theory (e.g. Molina et al. 1983; Parton et al. 1987; Hansen et al. 1990; Li et al. 1992) and experimental observations (e.g. Blagodatsky and Richter 1998; Reth et al. 2004) and develops a more refined temporal scale. The major N processes simulated include decomposition, volatilization, nitrification, denitrification, and leaching. See Figure 1 for a schematic of the model.

The soil module determines the soil heat and moisture environment which drives the rest of the modules. The decomposition module simulates the carbon and nitrogen flows through the soil organic matter pools to determine net mineralization of nitrogen. The nitrification module simulates nitrifier dynamics which control the oxidation of ammonium to nitrate while the denitrification module simulates denitrifier dynamics which control nitrate reduction to dinitrogen. Substrate allocation to each of the nitrification and denitrification modules is based on DNDC's "anaerobic balloon" (Li, 2000). N profiles are generated and the atmospheric transport module uses the surface concentrations to simulate N trace gas volatilization by applying a resistance analogy. General limitations of the model are that it's one-dimensional and does not incorporate the snowmelt or the influence freeze / thaw cycle(s) experienced in colder climates.

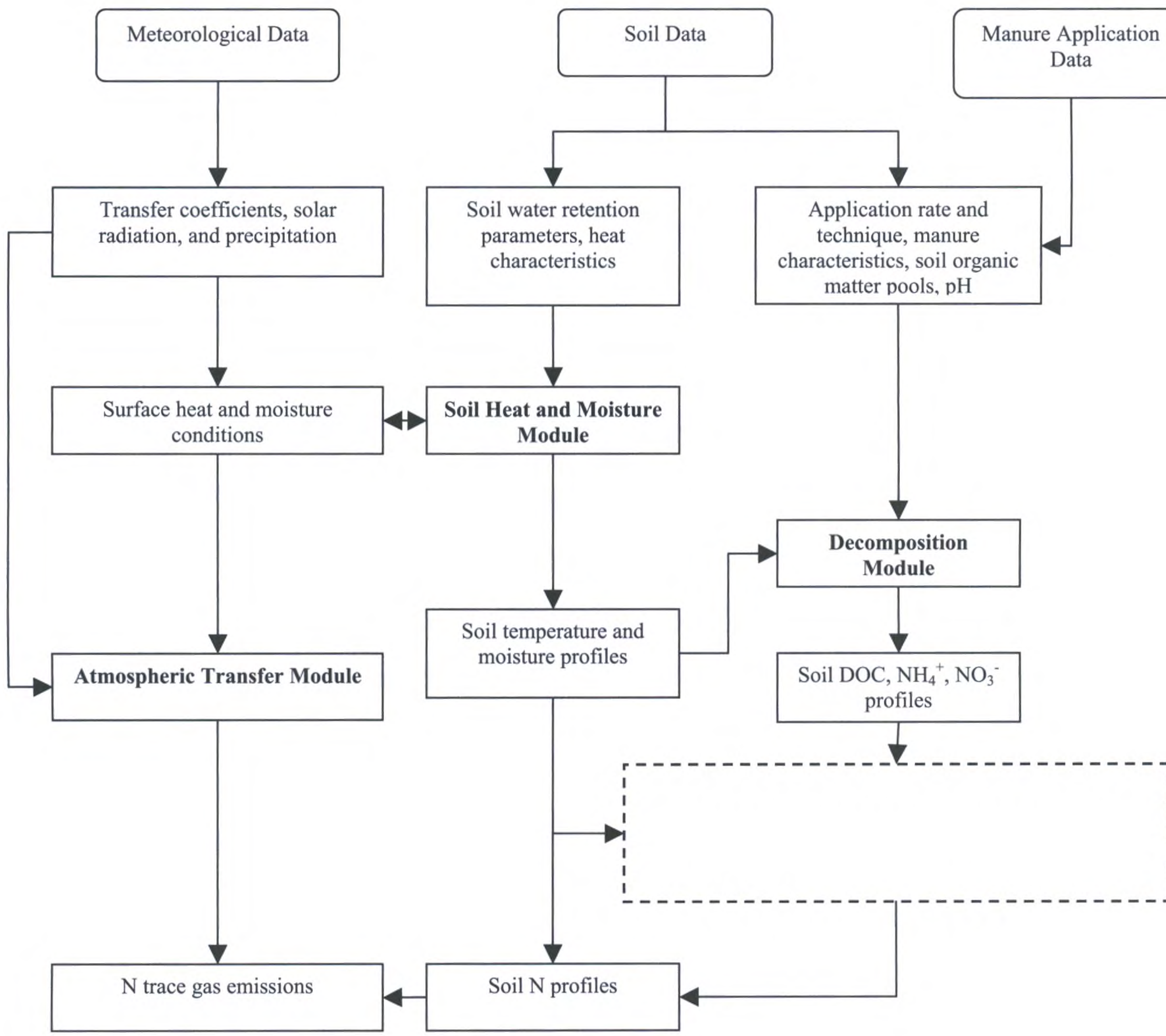

Figure 1: Schematic of AGRIN Model

The N species (organic N, NO_3^-, NO_2^-, NH_4^+, NH_3, N_2O, NO, N_2) profiles are calculated by applying a partial differential equation describing convective-dispersive movement through the soil:

$$\frac{\partial(C_s + \theta C_w)}{\partial t} = -\frac{\partial}{\partial z}\left[-\theta D\frac{\partial C_w}{\partial z} + q_w C_w\right] + S$$

where C_s is the adsorbed concentration of the species, θ is the soil volumetric moisture content, C_w is the concentration of the species in the soil solution, t is time, D is the diffusion coefficient of the species in soil, z is the depth below the soil surface, q_w is the soil water flux density and S is the source-sink term. The system of equations formed is solved numerically with the Crank-Nicholson finite difference scheme where the time derivatives are determined at the mid-point of a time step. The time step is user-specified and can be constant or dependent on convergence.

Experimental Study

A dataset was developed for model testing in August of 2005 when experiments were performed by Agriculture and Agri-Food Canada in Ottawa, Ontario. The experiments measured N_2O and NH_3 emissions following application of poultry manure and pig slurry to a bare field. Measurement techniques included flux gradient, relaxed-eddy accumulation and tunable diode laser. Ancillary measurements included soil properties, manure properties and hourly meteorological data.

Future Plans

Testing thus far has consisted mainly of soil module validation. Model calibration of the N modules is currently underway while validation of these modules will be conducted using the August 2005 dataset. Trends expected include large emissions occurring during application and subsequent emission trends associated with diurnal patterns. The goal of the model testing is to derive emission factors for N trace gases from an agricultural bare soil on a diurnal, monthly, and seasonal scale. Potential exists for the model to become a useful tool in predicting emissions on local, regional, or national scales. Other long-term objectives include generating input for short- and long-range transport models as well as to determine and assess emission reduction measures.

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Effects of Acidifying Liquid Cattle Manure with Nitric or Lactic Acid on Gaseous Emissions

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Abstract

Animal manure is a source of undesired gaseous emissions as there are ammonia (NH_3), methane (CH_4) and nitrous oxide (N_2O). They can be reduced by manure additives. Investigations were done on acidifying liquid cattle manure with nitric or lactic acid respectively on laboratory scale. The manure was stored in open vessels with 50 kg for 92 days (nitric acid) and with 75 kg for 190 days (lactic acid), respectively.

Ammonia emissions could be reduced effectively with nitric acid as well as with lactic acid. About 70% of NH_3 emissions could be avoided. Lactic acid had a better abatement effect on methane emissions than nitric acid, the reduction rates were 90 and 75% respectively. Nearly no nitrous oxide was detected from the manure acidified with lactic acid. Whereas the manure acidified with nitric acid emitted a large amount of N_2O. Nitric acid is not an advisable emission abatement technique.

Introduction

Agricultural operations are a source of gaseous emissions. International notes and regulations aim at the reduction of NH_3 as well as green house gas emissions (UN/ECE, 1999; EU, 2001). Animal husbandry and manure management is the main anthropogenic source for NH_3 (Berg et al., 2003) and emits also CH_4 at a large extent (Mikaloff Fletcher et al., 2004). N_2O emissions have their main source in soil (Bouwman, 1990). Under special conditions only when surface is dry and encrusted, manure emits N_2O at a noticeable extent (Berg & Hörnig, 1997).

Emissions from manure can be reduced by additives. Manipulating the balance between ammonia (NH_3) and ammonium (NH_4^+) by lowering the pH value of the manure is a promising possibility to reduce not only NH_3 but also CH_4 emission. A direct influence of the pH value on N_2O emission is not known. The effects of two different acids on the mentioned gaseous emissions from liquid cattle manure are investigated.

Methodology

Investigations on acidifying liquid cattle manure were carried out on laboratory scale at the Leibniz-Institute for Agricultural Engineering Potsdam-Bornim (ATB) (Fig. 1). Two trials were done: trial I with lactic acid and trial II using nitic acid. Fresh liquid cattle manure was collected from a manure pit in a commercial dairy cattle house.

In trial I 75 kg liquid manure were stored in cylindric vessels for a period of 190 days. Vessels made of plexiglas® with a volume of 92.4 l (diameter and height 0.49 m respectively) were used. The manure was acidified with different amounts of lactic acid achieving a pH value of the liquid manure of 3.8, 4.3 and 4.8 respectively at the beginning of the storage period.

In trial II nitric acid and different vessles were used. The vessels were also made of plexiglas® but with a volume of 65.4 l and different shape (diameter 0.29 m, height 0.99 m). 50 kg liquid manure were stored in these vessels for a period of 92 days. The same pH values of the liquid manure were adjusted at the beginning of the investigations (3.8, 4.3 and 4.8).

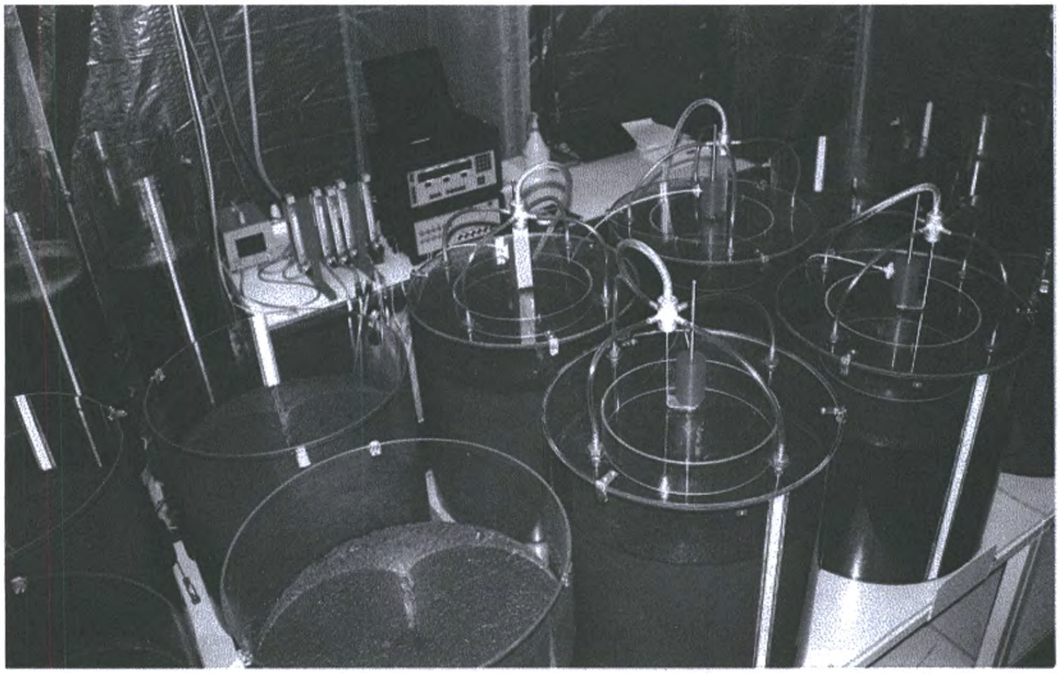


Figure 1. Experimental equipment for investigation of gaseous emissions as well as storage behavior.

Gaseous emissions were determined by using the open / dynamic chamber method. Vessels were closed and ventilated only during the measurements of the gaseous emissions with lids provided with air inlet and outlet. A compressor supplied the air inlets with fresh air. The fresh air flow rates were adjusted by air flow meters so that the air in the headspace was changed always one time per minute.

The headspace concentrations of the gases were determined by sampling exhaust air from each vessel and analyzing by a photoacoustic IR analyzer (multigas monitor). Ammonia (NH_3), nitrous oxide (N_2O), methane (CH_4), carbon dioxide (CO_2) and water vapor (H_2O) concentrations were measured in the headspace and in the fresh air. Headspace concentrations of the different vessels fresh air (background) concentrations deducted were compared with each other and the control within the trials respectively. The gas fluxes calculated from the concentrations could not be used for calculating fluxes from on-farm storage facilities. The different environment and volume to surface ratio of the vessels in the lab and on farms cause different gas fluxes.

The procedure of the measurement of the gaseous concentrations was the following: Exhaust air from each vessel headspace and fresh air were sampled sequentially at 20 minutes intervals. The first 16 minutes vessel exhaust air and the last 4 minutes fresh air was sampled. It took the photoacoustic IR analyzer 2 minutes for analyzing one sample. Thus 8 values for exhaust air were generated per vessel and 2 values for fresh air between each vessel. The first 5 values from vessel exhaust air were used for stabilization and the last 3 values for calculating a mean value representative for the measurement respectively. Fresh air between the vessels flushed the measuring chamber of the analyzer and demarcated measurements between each vessel. Fresh air concentrations were determined before and after the measurements of the first and the last vessel respectively.

Further parameters were determined:

- dry matter (DM), organic dry matter (oDM), total Kjeldahl nitrogen (TKN), total ammoniacal nitrogen (TAN) and organic acids of the manure by chemical analysis at the beginning and the end of the investigation period and between
- manure temperature and pH value
- sedimentation
- flow properties of the liquide manure by a rotational-type viscometer.

Results and Discussion

Manure Property

Properties of manure before and after storage are presented in table 1. The dry matter content of the manure was 7.8 and 8.0% at the beginning and 11.0 and 7.7% at the end of the storage period for trial I (lactic acid) and trial II (nitric acid) respectively. The long storage period of 190 days and the small volume to surface ratio of 0.398 of the vessels in trial I caused the noticeable increase of the dry matter content.

Table 1. Contents of total Kjeldahl nitrogen (TKN) and total ammoniacal nitrogen (TAN) of the manure samples before and after storage.

	TKN (g N kg-1 fresh mass)		TAN (g N kg-1 fresh mass)	
	Beginning	End	Beginning	End
Control, lactic acid	3.06	3.03	1.31	0.98
PH(b)≡4.8, lactic acid	3.02	3.58	1.29	1.53
PH(b)≡4.3, lactic acid	3.10	3.75	1.28	1.56
PH(b)≡3.8, lactic acid	2.88	3.67	1.24	1.56
Control, nitric acid	3.02	3.00	1.17	1.33
PH(b)≡4.8, nitric acid	3.35	3.23	1.19	1.27
PH(b)≡4.3, nitric acid	3.51	3.28	1.17	1.12
PH(b)≡3.8, nitric acid	3.55	3.41	1.17	1.16

Approximately 2, 4 or6% by volume of 50% concentrated lactic acid were necessary to lower the pH value of the liquid cattle manure to 5, 4.5 or 4.0. Nitric acid was much more effective, only 1, 1.3 and 1.5% by volume of 50% concentrated acid were needed to achieve the same pH values mentioned before.

The courses of manure pH values and temperatures are shown in figures 2 and 3. Lower pH values were more steady than higher ones. During trial I (Fig. 2), it was noted that in order to maintain desired pH values, lactic acid had to be added several times, at day 63 (pH(b)≡4.8), 91 (pH(b)≡4.8), 104 (pH(b)≡4.3), 119 (pH(b)≡4.8, pH(b)≡4.3), 133 (pH(b)≡4.8, pH(b)≡4.3) and 174 (all). All samples were homogenized two times, at day 84 and 154. Nitric acid was added three times during trial II (Fig. 3). Homogenizations of all samples were carried out also two times, at day 49 and 72. The mean pH values of the manure are given in table 2.

Manure temperatures were increasing in the course of the investigation period due to increasing room temperature. The range was between 10 and 22°C. Trial II began at day 75 of trial I. Thus in the following the course of the manure temperatures were nearly identically for both trials.

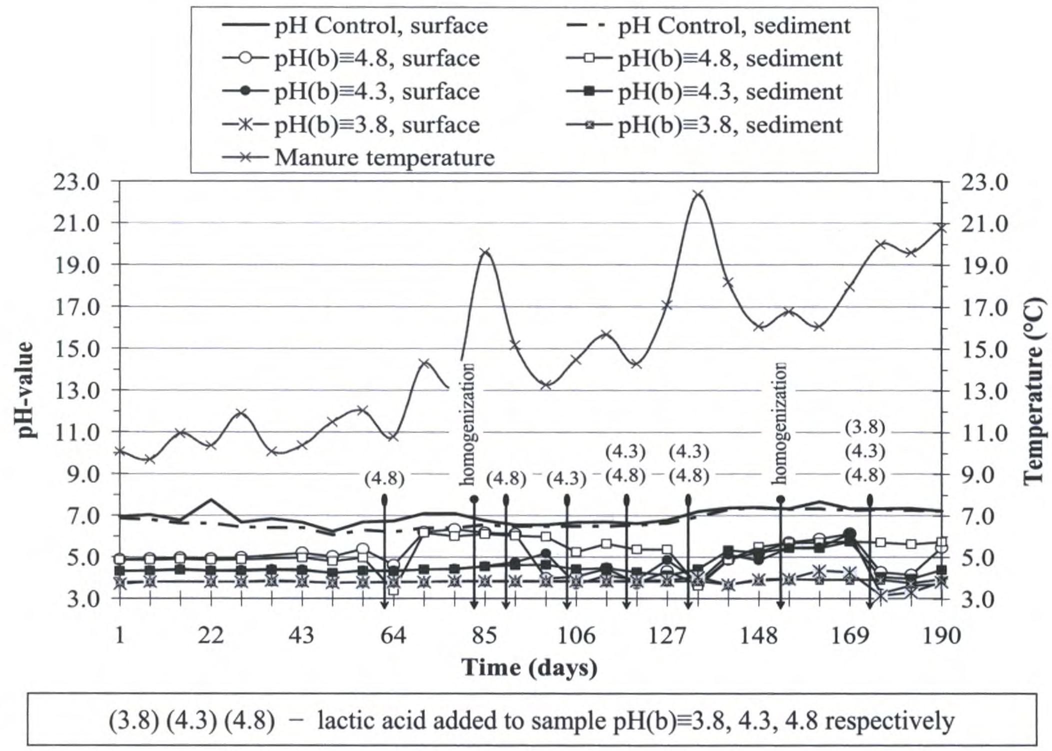


Figure 2. pH values and temperature of the lactic acid added liquid cattle manure.

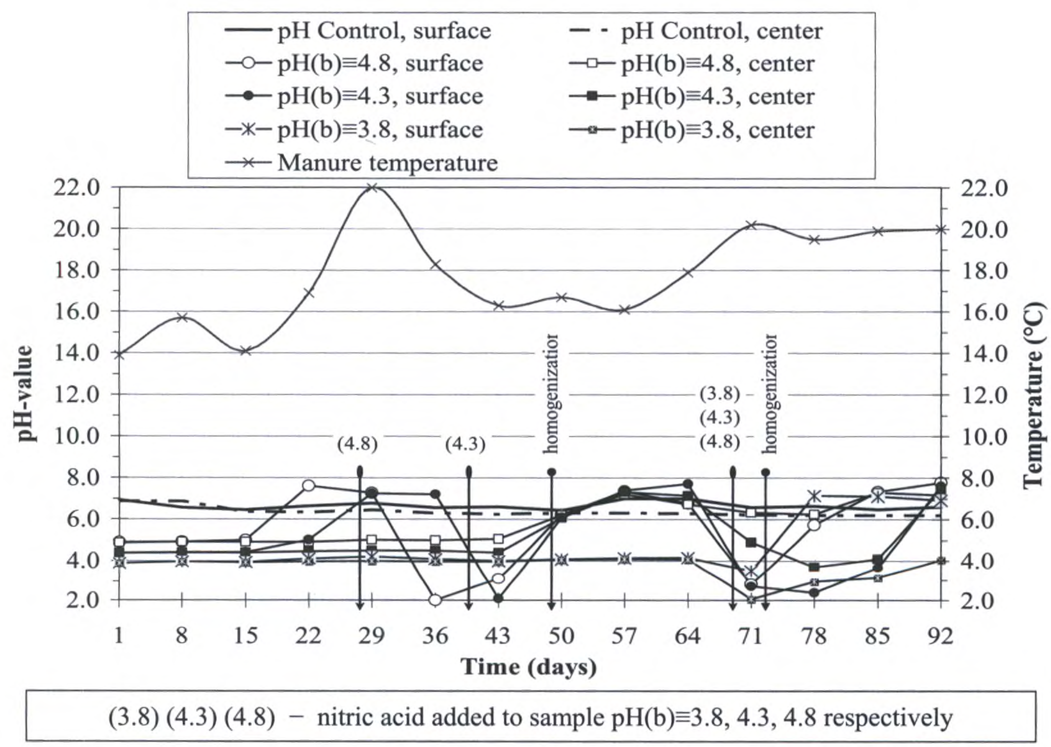


Figure 3. pH values and temperature of the nitric acid added liquid cattle manure.

Table 2. Mean pH values of the manure samples.

	pH
Control, lactic acid	6.50
pH(b)≡4.8, lactic acid	5.73
pH(b)≡4.3, lactic acid	5.14
pH(b)≡3.8, lactic acid	4.18
Control, nitric acid	6.85
pH(b)≡4.8, nitric acid	5.20
pH(b)≡4.3, nitric acid	4.49
pH(b)≡3.8, nitric acid	3.86

Ammonia Emission

NH_3 emissions were proportional to manure pH values. Lactic acid reduced NH_3 emission much more effective than nitric acid. With mean pH values of 5.73, 5.14 and 4.18 (Tab. 2) lactic acid reduced NH_3 emissions by 65, 72 and 88% respectively. Whereas nitric acid could reduce emissions with lower mean pH values of 5.20, 4.49 only by 29, 49 and 71% (Tab. 3).

Table 3. Mean ammonia (NH_3) emissions of the manure samples.

	Headspace concentration ($mg\ m^{-3}$)	Emission rate ($g\ m^{-2}\ d^{-1}$)	Emission rate ($g\ m^{-3}\ d^{-1}$)
Control, lactic acid	7.00	0.89	2.49
pH(b)≡4.8, lactic acid	2.43	0.31	0.90
pH(b)≡4.3, lactic acid	1.96	0.25	0.71
pH(b)≡3.8, lactic acid	0.85	0.11	0.31
Control, nitric acid	9.55	2.78	3.67
pH(b)≡4.8, nitric acid	6.79	1.96	2.61
pH(b)≡4.3, nitric acid	4.83	1.40	1.85
pH(b)≡3.8, nitric acid	2.76	0.80	1.06

Nitrous Oxide Emission

A slightly N_2O emission was detected from the control of trial I when it was getting dry and encrusted (Tab. 4). When the crust was destroyed by homogenization no N_2O emission occurred. But in the course of the following weeks this process started up again and N_2O was increasing again.

Manure acidified with nitric acid emitted N_2O at a large extent, although the manure surface nearly did not became encrusted. At the samples with higher pH values higher emission rates were measured than at the samples with lower pH values. Adding acid again and homogenizations did not prevented denitrification processes effectually but stimulated them.

Table 4. Mean nitrous oxide (N_2O) emissions of the manure samples.

	Headspace concentration ($mg\ m^{-3}$)	Emission rate ($g\ m^{-2}\ d^{-1}$)	Emission rate ($g\ m^{-3}\ d^{-1}$)
Control, lactic acid	0.49	0.062	0.174
pH(b)≡4.8, lactic acid	0.05	0.006	0.019
pH(b)≡4.3, lactic acid	0.04	0.005	0.014
pH(b)≡3.8, lactic acid	0.02	0.003	0.007
Control, nitric acid	0.0	0.0	0.0
pH(b)≡4.8, nitric acid	14.0	4.07	5.38
pH(b)≡4.3, nitric acid	12.5	3.62	4.78
pH(b)≡3.8, nitric acid	7.6	2.21	2.92

Methane Emissions

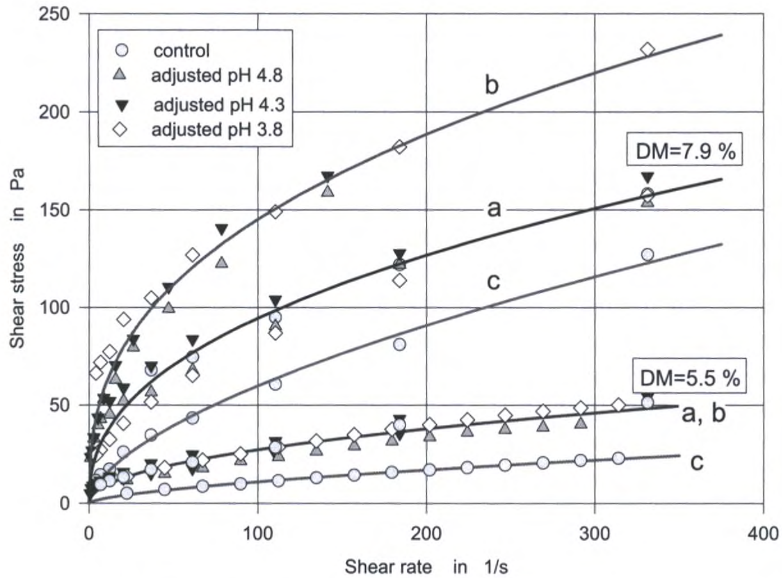
Lowering manure pH generally reduced CH_4 emission. Also here lactic acid was much more effective than nitric acid. From the samples acidified with lactic acid nearly no CH_4 emitted (reduction rates of 94, 91 and 98%). Whereas nitric acid could reduce emissions with lower mean pH values of 5.20, 4.49 (Tab. 2) only by 17, 64 and 75% (Tab. 5).

Table 5. Mean methane (CH_4) emissions of the manure samples.

	Headspace concentration ($mg\ m^{-3}$)	Emission rate ($g\ m^{-2}\ d^{-1}$)	Emission rate ($g\ m^{-3}\ d^{-1}$)
Control, lactic acid	53.8	6.84	19.1
pH(b)≡4.8, lactic acid	3.18	0.41	1.18
pH(b)≡4.3, lactic acid	4.59	0.58	1.67
pH(b)≡3.8, lactic acid	1.03	0.13	0.38
Control, nitric acid	7.90	2.30	3.03
pH(b)≡4.8, nitric acid	6.55	1.90	2.51
pH(b)≡4.3, nitric acid	2.87	0.83	1.10
pH(b)≡3.8, nitric acid	2.00	0.58	0.77

Rheological Property

At the beginning of the storage period there was only a slightly influence of acid addition on the flow properties of the manure. But after the storage period of 190 days, appreciable differences were measured. The flow properties of the untreated manure became better, whereas the acidified manure had a higher viscosity. It was caused by reduced conversion processes and a higher dry matter content.



a – control and acidified manure at the beginning of the storage period
b – manure with lactic acid at the end of the storage period
c – control at the end of the storage period

Figure 4. Flow curves of cattle slurry with lactic acid before and after storage

Conclusions

Adding nitric acid or lactic acid to liquid cattle manure gave different effects. Nitric acid could lower the pH value more efficiently. 1.3% by volume of 50% concentrated nitric acid were necessary to reach a manure pH of 4.5, whereas 3.7% by volume of 50% concentrated lactic acid were required to reach the same manure pH value. Advantages of lactic acid during acidifying were a considerably lower foam formation and a more innocuous handling than with nitric acid.

Both lactic and nitric acid could reduce ammonia emissions effectively, the reduction rate was nearly 90 and 70% respectively. In contrast, the effects on nitrous oxide and methane were quite different for both acids. From the manure acidified with lactic acid, nearly no nitrous oxide was detected, more than 90% of the methane emission could be detained. From the manure acidified with nitric acid, a large amount of nitrous oxide was emitted. The measured mean headspace concentration was about one order of magnitude higher than that of the control. The methane emission was about 25% of the control.

Ammonia emissions can be reduced effectively with nitric acid as well as with lactic acid. Lactic acid has a better abatement effect on methane emissions than nitric acid. Nitric acid is not an advisable emission abatement technique.

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Real-Time Monitoring of Air Pollution Due to Wildland Fires, Using a Mesoscale Model

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Abstract

In this paper a mesoscale model coupled with near real-time remote sensing data, has been applied to forecast air pollution due to wildland fires. Operational Multiscale Environment model with Grid Adaptivity (OMEGA) developed by SAIC (Science Applications International Corporation) is used in our current study. Satellite images have been used along with the NFDRS (National Fire Danger Rating) fuel load data to estimate the current fuel load available for burning. Emission from the fire has been calculated by estimating the area burned by the fire using real-time satellite data, and using emission factors given by EPA (Environmental Protection Agency). We have concentrated our efforts on estimating the emission of $PM_{2.5}$ and Carbon Monoxide due to wildland fires. A forest fire in the Eastern United States has been taken as a case study and the accuracy and efficiency of the model to run on real time basis has been shown. The whole processing is done using a sixteen node parallel cluster, so as to speed up the processing time for the model. A framework has been proposed to use mesoscale model along with real-time remote sending data to automatically detect fire pixels, run the model and generate the output in GIS (Geographic Information Systems) format to be distributed on the web. This will facilitate rapid distribution of forecast result which will be of immense help to persons involved in disaster management of wildland fires.

Carbon Dioxide Emissions from Agricultural Soils

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Abstract

Carbon dioxide emissions from soils of Slovakia have been determined and generalized. Absorption method of the CO_2 release determination from soil have been used. Data from 9 different soil places were collected during the three years and afterward processed and generalized.

About 10 061 thousand tons of C-CO_2 can be produced in average from agricultural soils of Slovakia during the growing season (245 days). It is about 4.2 tons C-CO_2 per 1 ha. Less productive soils release more CO_2 than high productive soils. Most of C-CO_2 is emitted from areas of Cambisols (3475 thousand tons), Fluvisols (1710 thousand tons) and Haplic Luvisols (1348 thousand tons).

Total amounts of C-CO_2 yearly released from agricultural soils of Slovakia is very close to the total C-CO_2 emissions by Slovakian industry.

Introduction

Energy and chemical substances are transformed in soils as essential soil functions in nature. Some incomes from those transformations in soil can be utilized by agriculture and another soil use practices. Mainly effects on air quality can be discussed. In this case is CO_2 released from soil into the air very important parameter of soils.

The CO_2 production in soil and consequently CO_2 emission from soil into the open air is a significant factor of air quality formation. Equilibrium between both CO_2 release from soil and CO_2 assimilation by plants can be considered as one of the most important principle of ecological stability of the nature.

Of coarse, a concrete data about CO_2 release from soil must be determined and presented if we want to evaluate some effects of soil on CO_2 contents in the open air. Accounting data of the CO_2 release from soil into the air are available from separate scientific and expert publications and from generalized (e.g. IPCC) studies. From national policies point of view also the data screening from different state territories must be available. This philosophy was adopted before our experimental activities.

Materials and Methods

In Table 1 contains data on the research sites (plots) and treatments used. For 3 years, the soils were sampled five times yearly in March, May, July, September, and November to a depth of 0.05 - 0.15 m. Non-N-fertilized treatments were used. Unified plant rotation was used in all plots (legumes, corn plants, and root crops with details depending on soil ecological conditions). The nitrification intensities in the soil samples and the CO_2 release intensities from soils were determined simultaneously by 14-days incubation tests.

Indexes of soil quality (productivity) were determined for each experimental site on the basis of soil ecological parameters by method of Dzatko et al. (1976). Indexes of soil quality can be determined for each site from Geographical Information System (GIS) database for Slovakian agricultural soils (productivity grades, 1 - 100). The indexes of productivity were identified for each experimental site to make a generalized data assessment.

Table 1: Soil types and treatments in the experimental plots

No	Soil type*	Treatments**
		N (-)
1	Cambisols	40/100
2	Plano-Gleyic Luvisols	170/300
3	Fluvi-Eutric Gleysols	40/110
4	Stagno-Gleyic Luvisols	40/100
5	Eutric Fluvisols	55/100
6	Calcaro-Haplic Chernozems	40/110
7	Fluvi Calcaric Phaeozems	6/110
8	Orthic Luvisols	90/100
9	Albo-Gleyic Luvisols	60/120

* According to WRB (1998) **; N (-): applied P_2O_5 and K_2O (kg ha^{-1}); N (+):

Results and Discussion

Carbon dioxide is produced by soil organisms through respiration. Therefore, the intensity of CO_2 release from soil was in the past considered as a "biological activity of soil". Now it is usually considered as data about carbon mineralization in soil. Moreover, sometimes the intensity of CO_2 release is simply called "loss of soil organic matter". From a greenhouse-effect point of view, this is a source of greenhouse gasses.

It is well known that the CO_2 release from soil can be in relatively large quantities (from hundreds of kilograms (C) to several tons per ha a year) (Glinski and Stepniewski 1985, Cole et al. 1990, Anderson 1995, Grant et al. 2002). From many published data it is clear that farming land produces more CO_2 than do other soils. Besides, manured soil can release more CO_2 than non-fertilized soil; drained soil, more than undrained soil; more productive soil, less than less productive soils; and so on (Richter 1974, De Jong 1981, Buyanovsky et al. 1986, Rochette et al. 1991, Reicosky and Lindstrom 1995).

This is a generalized overview, but specific data are very different under different soil conditions. CO_2 release is a very sensitive soil parameter with high variability from soil to soil and from land to land. When we want to have general data about CO_2 release from soils, we have to make detailed observations. This philosophy had been adopted before our relatively extended research and before all determinations of CO_2 emissions from soils were performed. The averaged data of CO_2 release from the soils investigated are collected in Table 3. Apart from soil types, the table also lists soil-production potentials in the form of indexes of soil productivity. It is clear from Table 2 that different soils show relatively deferent CO_2 release potentials (from about 30 to about 89 mg C-CO_2 kg^{-1} within 14 days). We can see that less-productive soils release more CO_2 than high-productive soils.

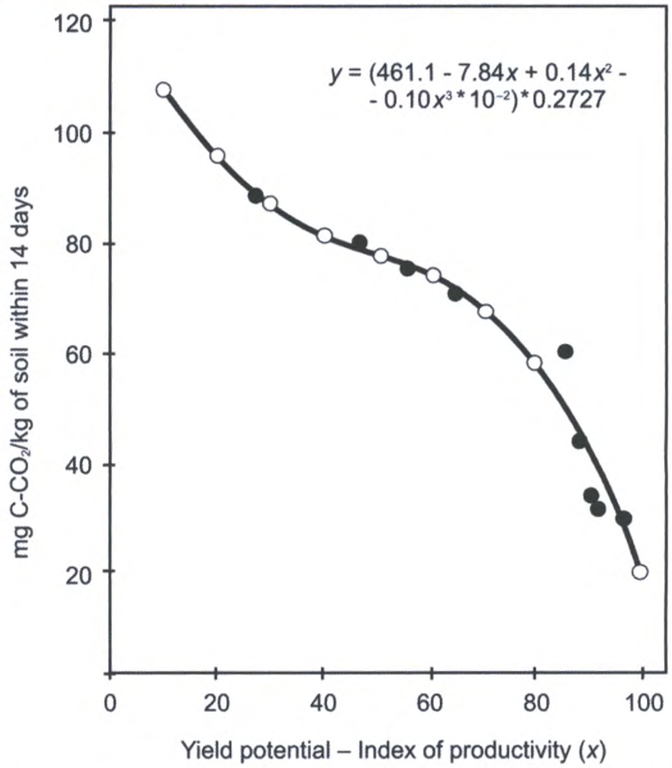
Table 2: CO_2 emissions from soils of experimental sites (non-fertilized plots)

Soil Type *	Index of soil productivity	mg C-CO_2 kg of soil
Eutric Cambisols	28	89
Luvic Stagnosols	47	80
Gleyic Eutric Fluvisols	55	75
Albic Stagni-Haplic Luvisols	64	72
Eutric Fluvisols	90	34
Calcaric Haplic Chernozems	91	31
Calcaric Mollic-Gleysols	95	30
Haplic Luvisols	85	61
Stagnic Glossisols	88	45

* WRB 1998

The regression between indexes of soil productivity and CO_2 release from soil is presented graphically in Figure 1. With this regression model, we calculated specific data for CO_2 release from the main soil units in Slovakian soil cover (Table 3). The last column of this table presents specific quantities of CO_2, which could be released from specific areas of specific soil units in Slovakia (values calculated with the help of the GIS database for the agricultural soils of Slovakia).

Figure 1. CO_2 emission from soils in relation to soil yield potentials



It can be said that on the average, about 10,061 thousand tons of C-CO_2 can be produced in the soil cover (agricultural soils) in Slovakia during the growing season (about 4.2 tons of C-CO_2 per ha within the growing season). Most of C-CO_2 is emitted from Cambisols (3, 475 thousand tons), Fluvisols (1,710 thousand tons) and Haplic Luvisols (1,348 thousand tons). This is not only because of the high intensities of CO_2 release from these soils but also because of their widespread occurrence on the territory of Slovakia. The total C-CO_2 emission from the Slovakian industry was estimated to be about 11,700 thousand tons (Ministry of Environment 2004). This amount is very close to the natural CO_2 amount produced by soils.