



## **Biofilters Used to Reduce Emissions from Livestock Housing – A Literature Review**

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### **Abstract**

This paper reviews the literature on the development of biofilters for use on confined livestock facilities. More specifically the paper addresses the history, current and accepted design parameters, and status of research. Biofilters offer one potential solution to reduce nuisance odors and pollutants emitted from confined livestock housing. Typical biofilters used in agriculture consist of an organic media mixture of wood chips and compost. Barn exhaust air is blown through this media, allowing the microorganisms contained within the media to break down the gaseous compounds. Biofilters have been used in agriculture in Europe since the 1970's and in the United States since the 1990's. Much advancement has been made in terms of biofilter design since their introduction to agriculture. Design parameters have been established to minimize cost and maximize effectiveness. Some of these design parameters include bed thickness, media type, empty bed contact time, moisture control, and media bed orientation. These design parameters also influence factors such as pressure drop, microbial ecology, air channeling, foot print size, and gas/odor removal efficiency. Overall, properly designed biofilters used in agriculture have been shown to reduce hydrogen sulfide emissions by up to 95% and ammonia emissions by up to 65%. Biofilters are an effective air pollution control option for agriculture.

### **Introduction**

Solutions are needed to deal with the problem of odorous compounds and airborne contaminant emitted from animal agriculture. Odor and airborne contaminant emissions have always been a part of the livestock and poultry industry (NRC, 2003). In the past, however, sparse human population and livestock spread out over a large land area made odor less of an issue. Now, with the advent of farms containing large numbers of animals confined to a smaller area; the odorous compounds and contaminants are becoming concentrated. Additionally, many areas zoned agriculture are being inhabited by more non-farm rural residents. This trend results in odor issues for livestock producers that in the past were minor or nonexistent. The urgency of the situation is clearly evident in the number of state, national, and global environmental air pollution regulations being developed.

Solutions are needed to reduce the nuisance value of these odors. One of the more recent practices to reduce odor emissions is the use of biofilters. Biofiltration is not a new technology, but is an adaptation of natural atmosphere-cleaning processes. Biofilters use microorganisms to convert gaseous contaminants to carbon dioxide, water vapor, and organic biomass. They are effective in reducing odorous components found in low concentrations in large quantities of air; thus making biofilters effective to reduce emissions from livestock facilities (Nicolai and Janni, 2000). Biofilters use a porous solid medium to support microorganisms and allow access to the contaminants in the airflow. Air passes through the biofilter media, which is surrounded by a biofilm. The contaminants are sorbed from a gas to an aqueous biofilm where the microbes live. This biofilm is where the microorganisms break down the contaminants. Media consists of relatively inert substances which ensure large surface attachment areas and additional nutrient supply. Examples of media include peat, soil, compost, wood chips, straw, or a combination of two or more (Nicolai and Janni, 2001).

### **History of Biofilters in Agriculture**

Microbial reactions in soils have been occurring naturally for many centuries, but only since the 1950's have such techniques been used to treat waste gases. Extensive biofilter research has been conducted only in the past 35 years, thereby limiting the quantity of information.

Biofilters were used in wastewater treatment plants, chemical manufacturing facilities, composting, and other industrial air pollution schemes before being adapted to agriculture. They were first applied to

livestock facilities in Germany in the late 1960's to reduce odor emissions from livestock facilities (Zeisig and Munchen, 1987). In the 1980's biofilters were used to reduce odor from livestock operation in the Netherlands and Sweden (Scholtens et al., 1987; Noren, 1985). These biofilters were based on the Zeisig design and decreased hydrogen sulfide and ammonia by 50%. When moisture content was at an optimum level, the reduction increased to 80%. Scholtens et al., (1987) reported an average filter efficiency of about 70% ammonia removal from biofilters on piggeries and calf sheds in Sweden. Scholtens and Demmers (1991) reported that even though biofilters are effective in treating exhaust air from livestock buildings, they were rarely used in intensive livestock farming in Europe. Their absence was based on 1) the construction and operating costs of treating large quantities of air, 2) the cost of handling effluent water needed to remove nitrogen absorbed in the biofilter material, and 3) the daily performance fluctuation.

Investigation of biofilters for livestock facilities in North America began in the mid 1990's. Nicolai and Janni (1997) investigated the feasibility of treating pit gases from a swine gestation barn. Young et al. (1997) investigated with three pilot-scale biofilters using swine odor and found average reductions of odor intensity, irritation intensity, and unpleasantness were 61%, 58%, and 84% respectively. Odor reductions between 78-80% were measured on two biofilters treating pit gasses from a swine finishing barn (Hartung, 1997). Ammonia reduction was mainly influenced by airflow and moisture content.

### **Biofilter Types**

Biofilters are categorized by the configuration (open or closed) and flow sequence (up-flow, down-flow, or horizontal flow). Deviny et al., (1999) discussed the differences between an open and an enclosed biofilter. A closed system controls both the biofilter outlet and inlet gas streams whereas an open system discharges treated gas from the biofilter directly to the atmosphere. Industrial applications may place biofilters in closed vessels with deep layers of media to save space. These systems may be either up or down flow, depending upon the moisture application system.

Open biofilters are more commonly used for animal agricultural. They are less expensive than closed systems, have relatively thin layers of media to reduce backpressure on the air handler, are outdoors, and are usually quite large in terms of surface area exposed to the atmosphere. Horizontal media beds (up or down flow) and vertical media beds (horizontal flow) are used, depending upon surface area and space availability (Lefers and Nicolai, 2005).

Up-flow open biofilters are the most common biofilter used in livestock production since they are generally more economical. Most livestock building ventilation fans exhaust at or near ground level. Therefore to reduce air ducting costs, the air entering the biofilter media should be at this height. Spreading the media out horizontally instead of vertically also saves on construction costs as well. Spreading the media out horizontally may not always be an option, depending on land surface area space available near the production facility. In this case vertical media beds (horizontal flow) may be an option. By leaving the biofilters open to the atmosphere, producers are able to reduce pressure drops, saving on fan electricity bills.

### **Microbial Activity in Biofilters**

Since it is the microorganisms in the biofilter provide the actual work of breaking down the odorous compounds (or oxidation potential), it is important to understand the transformations and interaction of these microorganisms. Our knowledge of microorganisms' ecosystem is fragmentary and poor (Deviny, 1999). Biofilters may be self-inoculating, inoculated with activated sludge or compost, or induced with bacteria species. Most biofilters used in agricultural settings use compost as the source of microorganisms.

Ding et al., (2000) studied the microbial ecology and performance of a biofilter. Their research indicated that introduction of new pollutant gasses into an already operating biofilter did not adversely affect the removal efficiency of the biofilter for the pollutant gasses already being treated. The research also showed that changes in the microbial community occurred with changes in the inlet gasses. Thus, the research demonstrated the ability of the microbial community to respond to changes in the inlet gasses without affecting treatment of those gasses.

Sakano and Kerkhof (1998) studied the changes in a microbial community structure of a biofilter used to treat only ammonia. They found that the overall diversity of the heterotrophic microbial population had

decreased by the end of the experiment. The community structure of the heterotrophic population had also shifted between subdivisions. This experiment also showed the ability of microbial communities in biofilters to adapt to changes (or lack thereof) in inlet gasses.

Joshi et al. (2000) studied the gaseous ammonia removal of laboratory-scale biofilters. Their work showed that 95% of ammonia was degraded to  $\text{NO}_3$  and  $\text{NO}_2$  in the quarter of the biofilter closest to the inlet gas stream after eight days of operation.

Sadowsky et al. (1999) investigated the culturable bacteria in biofilter material that was treating exhaust air from a swine barn. They found the media material did not contain significant numbers of enteric coliforms or salmonella and concluded that sterilization and special disposal of biofilter material is not necessary. This is similar to results found by Pillai et al. (1996), who reported that while air samples at sludge application sites contain relatively high numbers of heterotrophic bacteria (averaging  $10^5$  CFU/m<sup>3</sup>), no sites contained airborne fecal coliforms or salmonella.

Further work is needed to understand the relationship between microbial community dynamics and biofilter performance. Research needs to be done to determine how fast microbial communities change in response to changes in inlet gas concentrations. However, this research should not affect biofilter design dramatically, since a sufficient amount of media to host the number of microorganisms needed to effectively remove odorous compounds is established by the EBCT (Devinny et al., 1999).

### **Biofilter Residence Time**

In order to get maximum odor reduction from a biofilter, the air passing through the filter must contact the filter media for a given amount of time. This amount of time is known as the residence time or empty bed contact time (EBCT). It is defined as the empty bed filter volume divided by the air flow rate (Devinny et al., 1999).

The correct residence time is an important variable to know in the design of an efficient biofilter. Zeisig and Munchen (1987) showed sufficient odor reduction at 5 seconds for pigs and 3 seconds for chickens. Pearson (1990) used 20 seconds while doing his cost estimates for biofilters on swine and poultry operations. Nicolai and Janni (1998) did a study to see if there was any statistical difference between a 4 second and 8 second residence time for a swine nursery barn, and concluded that 4 seconds was adequate. In a further study by Nicolai and Janni (1999), it was determined that there was no significant increase in odor reduction with residence times of 6 seconds or more for either dairy or swine. However, odor reduction was less than optimal at less than 4 second residence times. A recommended design residence time for a biofilter on a dairy and swine facility was given at 5 seconds for adequate odor and hydrogen sulfide reduction (Nicolai et al., 2004).

### **Pressure Drop**

Agricultural ventilation fans generally are designed to operate at less than 62 pa (0.25 in H<sub>2</sub>O) (Nicolai and Janni, 1998). If the pressure drop through the biofilter can be kept less than 40 pa (0.16 in H<sub>2</sub>O) it may not be necessary to replace existing fans in a livestock building when installing a biofilter (Phillips et al., 1995). Thus, pressure drop becomes a very important factor in the design of a biofilter.

In a study of various types of packing media for biofilters (heather, mixture of heather and coconut fiber, mixture of heather and fibrous peat, bean straw, coconut fiber, linseed straw, and screened wood chips), Phillips et al. (1995) concluded that wood chips over 75mm screen size appeared to be the most promising because they gave one of the lowest pressure drops and were the least compressible.

The compost/kidney bean straw mixture used by Nicolai and Janni (1997) had a maximum pressure drop of 47 pa reported after six months of settling. In a 2001 study conducted by Nicolai and Janni on different mixtures of wood chips and compost, findings indicated that pressure drop increased as the percent of compost in the mixture increased. Further research by Nicolai and Janni (2001) indicated that the pressure drop is related to the percent voids space in the biofilter media and the airflow/surface area relationship in the biofilter itself. A recommended mixture of 20 to 30% compost and 70 to 80% woodchips by weight was given as optimal for an agricultural biofilter (Schmidt et al., 2004). The results from this study allow sizing of a biofilter and provide guidance in selecting fans with adequate performance characteristics for

the expected pressure drop. Nicolai and Janni (2002) developed a graph that shows the relationship between media void ratio, airflow, and pressure drop.

### **Biofilter Moisture**

Biofilter moisture has been cited as one of the two key factors influencing biofilter performance, along with contact (residence) time (Schmidt et al., 2004). In fact, lack of control over media moisture has been cited as the cause of up to 75% of all biofiltration problems (Reyes et. al., 2000, and Boyette, 1999). Dry media causes channeling and leads to a decrease in biological activity. Media drying is especially a problem in the summer, when an increase in temperature raises the water capacity of the air, causing a faster evaporation rate of water from the media (Schmidt et al., 2004). Biofilters with higher moisture contents and longer retention times were shown to have the best removal of both ammonia and hydrogen sulfide (Sun et al., 2000).

Nicolai and Janni's (2001) study on biofilter media mixture ratio found that media moisture content influenced odor, hydrogen sulfide, and ammonia removal efficiency. This study recommended a moisture range from 35% to 65% for efficient pollutant reduction using compost and wood chips. Optimum biofilter moisture content for a compost/wood chip mixture has been given as 50% (Schmidt et al., 2004). Similarly Boyette (1999) recommends 50 to 55 percent moisture is a good target range for a compost-based biofilter media.

Moisture addition can be needed during mild and warm weather (Nicolai and Janni, 1997). A Korean study on a biofilter using pine chaff and perlite as filter media found that removal rate of odor drastically reduced when the moisture content dropped below 60% (Chang et al., 2004). Lack of a sensor to keep stable moisture contents was cited as the reason efficient odor removal was not achieved.

A number of moisture sensors and systems have been tried to measure biofilter moisture. Classen et al. (2000) used a weight-based method of calculating moisture content on three pilot-scale biofilter units by continuously weighing each biofilter. This method was found to maintain the proper moisture level on each of the three biofilter units within 4%. However, the weight-based method could only be used because dust was excluded from the biofilter. Most agricultural applications have dust loading from the air being treated by the biofilter, which contributes to the problem for a weight-based method. Also decomposition of media resulting in losses in bed weight could be assumed to be caused only by losses of moisture. However, periodic re-calibration could compensate for media degradation and washout (Reyes et al., 2000). An additional problem with a weight based system is caused by the uneven distribution of moisture in the filter. The weight system reports the average moisture content in the bed but some sections may be too dry resulting in air channeling.

Reyes et al. (2000) demonstrated that a TDR probe could be used to monitor biofilter media moisture content on a real time basis. However, this experiment was done using 60% compost and 40% perlite, a mixture that is not used for most agricultural biofilters because of excessive pressure drops.

Researchers at the University of Illinois (Funk et al., 2005) found soil and hay moisture meters inadequate for sensing agricultural biofilter moisture content because of poor media to probe contact. They also found that relative humidity sensors were of some value. An electrical capacitor type moisture sensor for determining moisture content in biofilters is being developed, with preliminary tests indicating it could be useful for monitoring biofilter moisture (Funk et al., 2005).

Lefers and Nicolai (2005) have used a watermark moisture sensor and control system to control moisture in a laboratory-scale biofilter with promising results. This research showed that a watermark sensor may be used effectively to control moisture in a lab scale biofilter. Further testing in a full scale agricultural biofilter is needed.

Moisture content has also been shown to have an effect on the pressure drop through biofilters (Nicolai and Janni, 2001). Pressure drops are higher when moisture content is higher. However, this effect is only noticeable on low compost to wood chip ratios, and is not a major factor on pressure drop through media that consists of high compost to wood chip ratios.

### **Biofilter Media Depth**

The biofilter media depth will also have an effect on pressure drop. Less media depth results in reduced pressure drop which is linearly correlated with media depth (Sadaka et al., 2002). Media depth also effects odor reduction efficiency. Nicolai and Janni's (1999) study on biofilter retention time showed decreasing media depth below 0.15 m reduced odor and hydrogen reduction below 65%. As a result they recommended minimum depth of a compost/wood chip media biofilter between 0.15 m and 0.3 m, with an estimated ideal minimum depth at 0.25 m. Findings based on research done on the spatial structure of microbial communities in peat biofilter media have indicated that 75% of inlet concentrations of aromatic compounds were degraded between 0.3 and 1 m in depth (Khammar et al., 2005). For biofilter design in agriculture, media depth is recommended to remain between 0.25 m and 0.45 m to reduce the possibility of large pressure drops, compaction, excessive drying and air channeling (Schmidt et al., 2004).

### **Effect of Temperature on Biofilters**

The microorganisms in the biofilters are living beings, and as such must be maintained at a certain temperature to sustain life. The microorganisms that are most effective in the degradation of odor compounds are mesothermic and have an optimum operating temperature between 30 and 40 degrees C (Janni and Nicolai, 2000). In a study on an agricultural biofilter exposed to atmospheric conditions in Alberta, Canada, it was found that differences in treatment temperature had no apparent influence on odor removal (Clark et al., 2004). Treatment temperature generally ranged between 15 and 30 degree C because the biofilter was insulated to prevent heat loss during the winter. However, data in the experiment did suggest that higher operating temperatures accelerated the establishment of microbial populations and the start of effective biofiltration. In a study conducted in Manitoba, Canada, an open biofilter was evaluated for treatment of odor during sub-zero ambient temperatures (Mann et al, 2002.). This study found that at ambient temperatures below -20 degrees C, biofilter bed temperatures were maintained at around 16 degrees C without the need for insulation or supplemental heat. Heat was provided by the warm exhaust air from the swine barn, and was adequate enough to ensure microbial survival during even the coldest winter days. There was also no apparent reduction in biofilter effectiveness during the winter months.

Biofilters on manure storages or unheated buildings may freeze in winter, which temporarily reduces the effectiveness of the biofilter (Schmidt et al., 2004). However, as the biofilter again thaws in the spring, the microorganisms again become active and the removal efficiency is restored.

### **Degradation of Biofilter Media**

The same microorganisms in the biofilter that treat the odorous compounds in the exhaust air also degrade biofilter media as a result of their activity (Wani et al., 1998). Because of this, the biofilter settles and becomes compacted over time, reducing component interchange surface and increasing the resistance to flow. Media dry matter may even increase over time as a result of growth of microorganisms and chemical accumulation in the media due to mineralization (Sun et al., 2000). Eventually, the biofilter suffers too much from aging effects and the media must be replaced. No long term studies have been completed on compost/wood chip biofilters treating air from livestock buildings to determine the length of media life, but it is estimated that most biofilter media will remain effective without causing a large pressure drop for 3 to 10 years or more (Schmidt et al., 2004).

### **Horizontal Airflow Biofilters**

Most agricultural biofilters are vertical airflow biofilters which may result in a large footprint. Because of space limitations for some installations, vertical flow (also called horizontal bed) biofilters may not a viable option for agricultural applications.

An alternative to vertical flow biofilter is to use a horizontal airflow (vertical bed) biofilter. These filters have a smaller footprint since the bed is spaced vertically instead of horizontally (Garlinski and Mann, 2003). Resistance to airflow in the horizontal direction was shown to be approximately 0.65 times the resistance to airflow in the vertical direction (Sadaka et al., 2002). This means a lower pressure drop for vertical bed biofilters, which could in turn result in lower energy consumption.

However, the disadvantage associated with vertical bed biofilters is the compost will settle and become denser over time causing a increase in pressure drop (Deviny et al., 1999). A larger pressure drop through

the media is caused by the natural settling of the filter bed (Zeisig and Munchen, 1987). Compaction of a biofilter can cause serious problems because the odorous gases flow through the space yielded by the compaction (Choi et al., 2003). This channeling can lead to an increase in local drying, which allows untreated air to exit from the biofilter, reducing overall efficiency. Significant media settling has occurred with horizontal-airflow biofilters when wood chip-based media is used (Garlinski and Mann, 2004). The settling results in more air leaving through the low pressure drop areas at the top of the filter. Non-uniform airflow through the biofilter media will cause the residence time to be affected, reducing the odor removal efficiency of the vertical bed biofilter.

A solution to the settling that occurs in vertical bed biofilters is to taper the media bed of the biofilter so that the media wall is thicker at the top than the bottom. This leads to a decrease in channeling and a more uniform airflow through the biofilter over a period of time, which results in improved efficiency. Lefers (2006) showed that a taper angle of 9.6 degrees provided the most uniform airflow through the biofilter bed after media settling.

### **Biofilter Costs**

Pearson (1990) reported on studies conducted in Europe that showed biofilters were effective in their performance, but needed to be more cost effective before being adopted by the livestock industry. The report demonstrated that biofilters were the most effective and the least cost device for odor control compared to bioscrubber, mopscrubber, packed tower, or sprayscrubber. In spite of this, the report reasoned that even the biofilter was still out of the price range of most producers. The report estimated that the biofilter would add 6% to the production cost of pigs and 20% to the production cost of poultry using heather as the biofilter media. However, the report also speculated that differences in construction could reduce the costs involved.

O'Neill et al. (1992a) evaluated twelve odor abatement systems for costs to treat odor emissions from mechanically ventilated livestock buildings. The seven systems judged to be effective include chemical or biological treatments of air leaving the building. Chemical treatments were found to be prohibitively expensive. Biological treatments were much lower cost and the lowest cost treatment system was found to be a biofilter.

In the United States, the cost for biofilters used in an agricultural application was investigated. Nicolai and Janni (1997) reported on a biofilter that was effectively developed for full scale mechanically ventilated production swine units with deep manure pits below a slatted floor. This research estimated the construction and operating cost of a swine biofilter at \$0.28 per piglet produced. The biofilter media used for this study was 50% red kidney bean straw and 50% compost by weight. This biofilter had an odor threshold reduction by an average of 78%, hydrogen sulfide reduction by an average of 86% and ammonia reductions of an average of 50%.

Nicolai and Janni's (1998) research indicated a total installation cost of about \$0.22 per piglet or \$0.062 per cfm when installed on a swine gestation barn. Operating costs were also estimated at \$275 per year for effective rodent control program and \$125 a year for water sprinkling of biofilter media and using higher power ventilation fans. The biofilter media for this study consisted of a mixture of 50% yard waste compost and 50% brush wood chips by weight and effectively removed 82% of the odor and 80% of the hydrogen sulfide. The report concluded that while these costs could be affordable by most swine producers, it was more than the value producers were currently spending to control odor.

Higher biofilter cost on European pig units may be attributed to the location of the building exhausts ventilation fans. Many livestock buildings utilize a chimney system through the roof to exhaust the air, adding cost to duct the air back down to a biofilter located on the ground. In the U.S. most facilities exhaust air near ground level through wall or pit fans. A shorter and simpler duct to move the air to a biofilter results in a lower cost.

### **Conclusions**

A review of the literature reveals considerable advancements have been made in biofilters for agricultural applications in the last 35 years. When designed correctly, biofilters have demonstrated to be an effective odor (up to 95%) and air pollution control option for livestock producers.

There remains research to be conducted that would further improve biofilter efficiency in agricultural applications. As an example, research is needed in testing and developing a moisture sensor and moisture control system for use on full scale agricultural biofilters. Also, research is needed to determine the length of time needed before the media requires replacement.

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## Isotopic and Back Trajectory Analysis on Rainfall Chemistry for Developing Source-Receptor Relationships in Fine Particulate Matter Formation

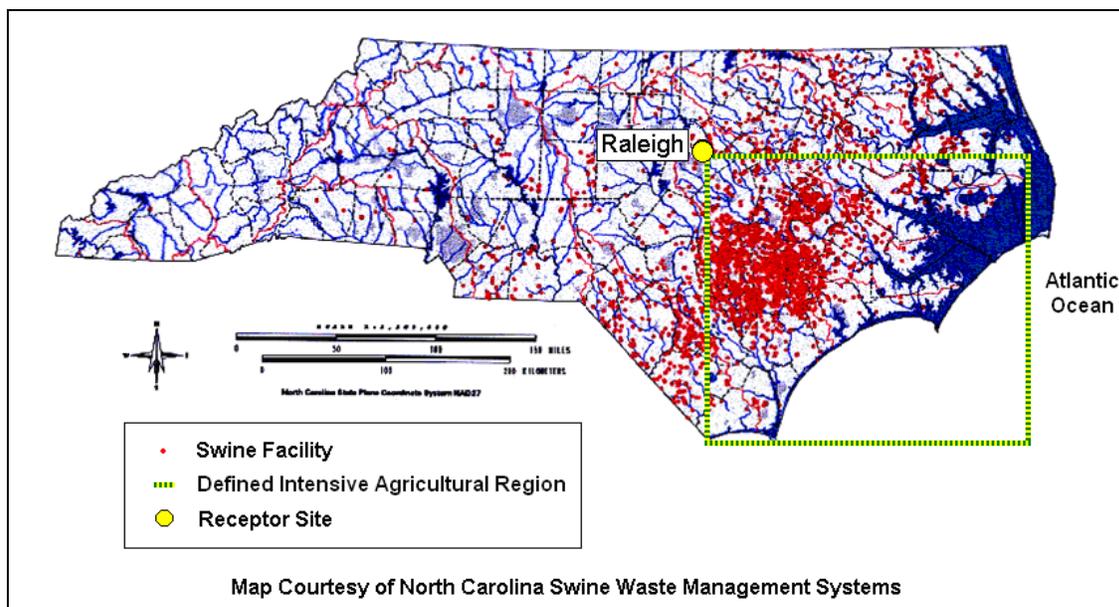
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### Abstract

Nitrogen oxides ( $\text{NO}_x = \text{NO} + \text{NO}_2$ ), nitrate ( $\text{NO}_3^-$ ), ammonia ( $\text{NH}_3$ ), and ammonium ( $\text{NH}_4^+$ ) have important effects on rainfall acidity and fine particulate matter ( $\text{PM}_{2.5}$ ) formation. The agricultural corridor (figure 1) of southeast North Carolina, which includes approximately 10 million hogs, acts as a major source for these pollutants. Previous studies indicate, for example, that up to a 50% increase in ammonium concentrations in Raleigh rainfall occurs during periods when air masses cross the animal agricultural area. The biological activity associated with farming in the region also produces pollutants with heavily depleted nitrogen isotope ratios (low delta  $^{15}\text{N}$  values), suggesting that it may be possible to verify that a biological source of pollutants is in fact being detected at a receptor.



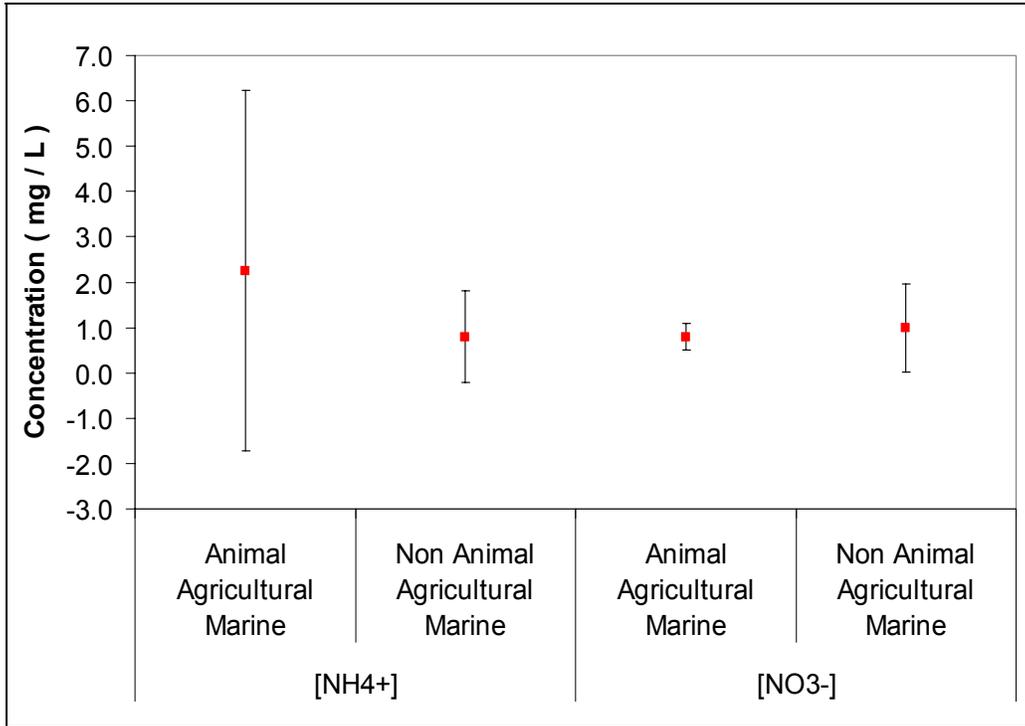
**Figure 1: North Carolina Depicting Animal (Hog) Agricultural Region**

The Hybrid Single Particle Lagrangian Integrated Trajectory (HYSPLIT) model was used to determine potential sources of nitrogen in rainwater collected at an urban site in Raleigh, North Carolina, during the first three quarters of 2004. The delta  $^{15}\text{N}$  isotope ratio signatures of each sample were used to further differentiate between sources of the same compound. This study examined the importance of pollutant sources such as animal agricultural activity (figure 1) and the role of coastal meteorology on rainfall chemistry as well as their implications on fine particulate matter formation.

### Preliminary Results

Samples that transited across the dense crop and animal (swine) agricultural region had no significant statistical difference in nitrate concentrations overall (figure 2), but had lower delta  $^{15}\text{N}$  isotope ratios (denoted delta N-15 in figure 3) in the nitrate ion averaging  $-2.1 \pm 1.7$  per mil than those which averaged  $0.1 \pm 3.0$  per mil from the nonagricultural counterparts. This indicated that it was possible to determine that

transport of pollutants from the farm region did occur, without usable information about concentration. However, the limited data set does not offer any conclusive evidence of similar patterns in ammonium ions. An increase in PM fine mass concentrations was also found to correspond to air transport over the hog farm regions. For the dates when air was transported to Raleigh from the agricultural area, PM<sub>2.5</sub> concentrations at the Raleigh (Wake County Department of Air Quality) site averaged  $15.1 \pm 5.8 \mu\text{g}/\text{m}^3$  yearly compared to a yearly average of  $11.7 \pm 5.8 \mu\text{g}/\text{m}^3$  for air which was not (figure 4).



**Figure 2: Average Pollutant Concentrations in Rainfall in Raleigh, N.C. for the Easterly Wind Sector**

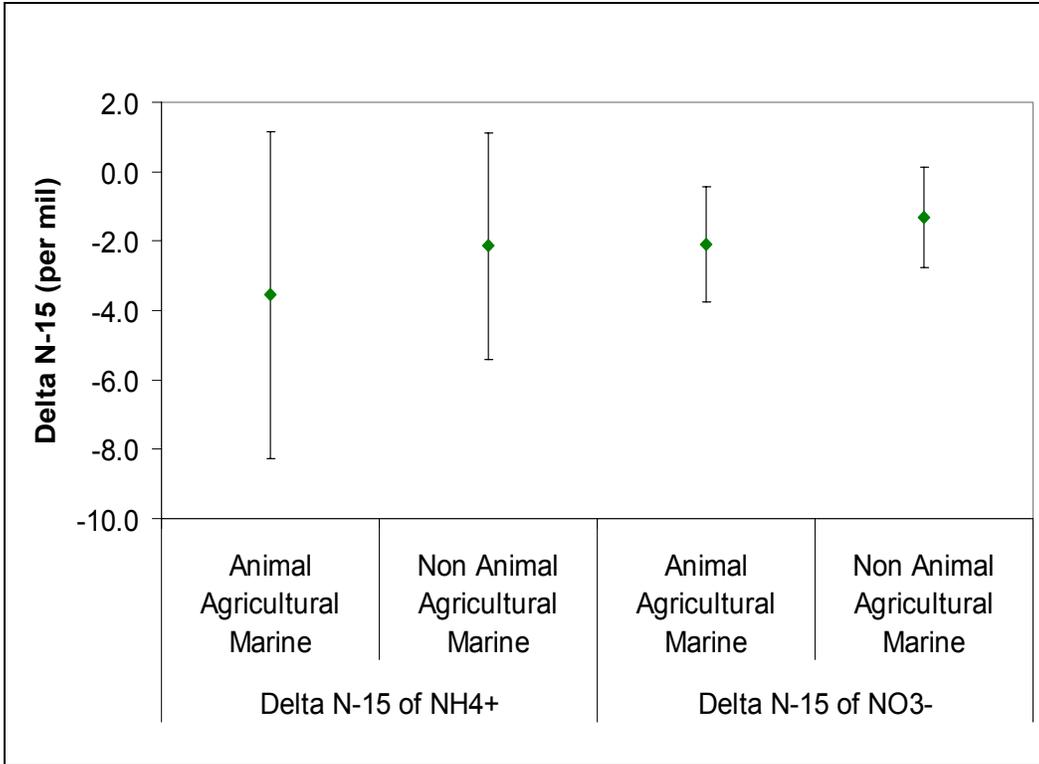


Figure 3: Average Delta <sup>15</sup>N Values in Rainfall for the Easterly Wind Sector

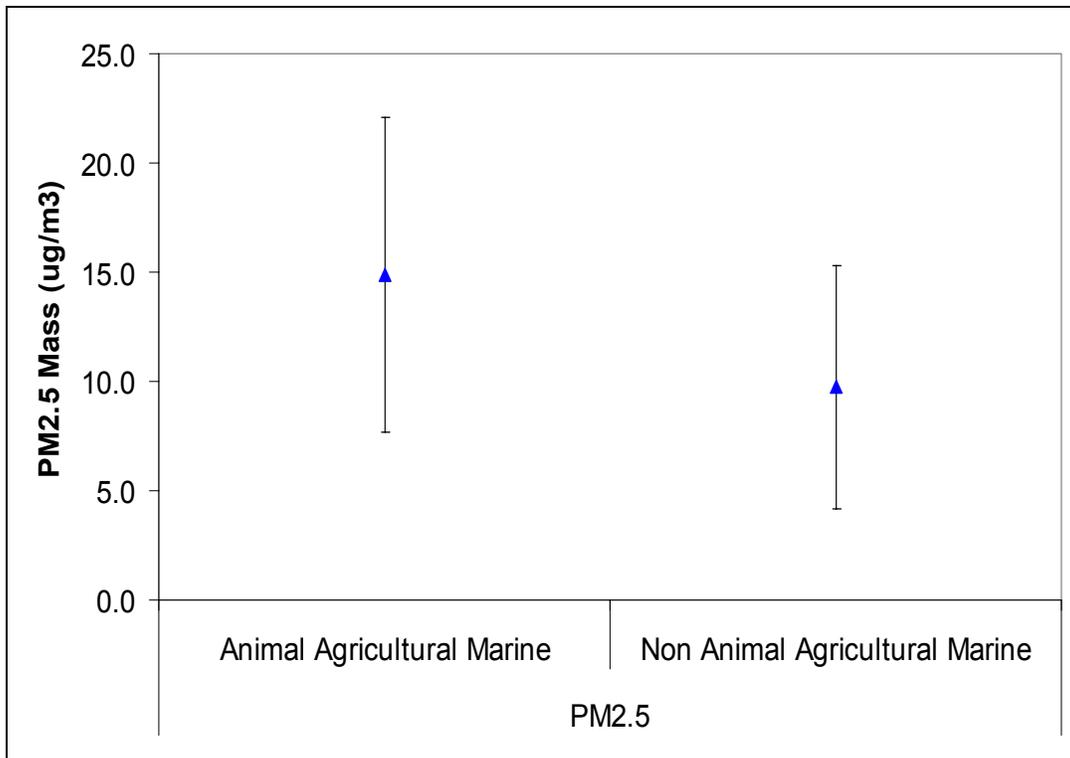


Figure 4: Average Fine Particulate Matter (PM<sub>2.5</sub>) Mass for the Easterly Wind Sector

### Future Work

Similar correlations in regional fine particulate matter isotope and chemical composition have also been investigated, though results are not yet available. Studies linking the agricultural corridor's regional air quality with the air found in Wilmington, NC (a frequent receptor of such air according to HYSPLIT forward trajectory results) may provide evidence that increases in pollutants in Wilmington are in fact associated with transit across the farming region. Previous trajectory and pollutant information hinted at the association between such transport and increases in ammonia.

### Conclusions

Results indicated that concentration, isotope, and fine particulate matter mass data were to some degree dependent on the trajectory which air at the receptor had traversed. Ammonium concentrations, nitrate isotope ratios, and fine PM mass concentrations support previous claims that there is a strong source – receptor relationship between pollutants emitted from the agricultural corridor and the air received in Raleigh. Low numbers of samples stymied the attempts to study several additional methods of verifying the results; however these may be remedied by continued collection of samples in the region.

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## **Estimation of Wind Speed Frequency Distribution Application in Predicting Wind Erosion**

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### **Abstract**

Wind erosion is a particular serious problem on many lands, and human impact on the global environment and it is an issue of international concern. To adequately predict the consequences of various land management strategies on wind erosion, new technology is needed. Wind transport moisture and heat in the atmosphere and therefore has some effect on the crop production. It also influences rates of evapotranspiration and directly exerts pressure on crops along its path. Wind may encourage soil erosion when the speed exceeds some critical threshold value for a given soil environment. Crops may be buried by wind blow or dust while the stems and leaves of the tall crops surface abrasion by sand particles. This was part of the surface energy balance project over a bare soil at a humid tropical site at Ile- Ife, Nigeria.



## Investigation of Source Regions and Temporal Variation of Ammonia Over the Eastern Mediterranean Atmosphere

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### Abstract

Approximately 1500 aerosol samples were analyzed by Unicam 8625 UV-VIS Spectrometer to measure the ammonium ion contents of the samples collected between 1992 and 2000. Samples were collected on Whatman-41, cellulose filters at the Turkish Mediterranean coast by means of Hi-Vol sampler. Major ion content, namely,  $\text{SO}_4^{2-}$ ,  $\text{NO}_3^-$  and  $\text{Cl}^-$ , of the samples were also detected by Ion Chromatography (IC). Besides, trace element contents of the collected samples were determined by using Energy Dispersive X-Ray Fluorescence Technique (EDXRF). Long and short term trends of the measured species were investigated. High concentrations of  $\text{NH}_4^+$  were detected particularly for summer season. Gaseous ammonia is the predominant atmospheric base and plays an important role in atmospheric chemistry. Ammonia, while not hazardous at atmospheric levels, participates in aerosol formation with  $\text{HNO}_3$  and hence can neutralize the acidity of gaseous and particle phase acids. The reason of high background concentration of  $\text{NH}_4^+$  during summer might be attributed to the increasing use of fertilizers in this season. In contrast to  $\text{NH}_4^+$ , high sulfate and nitrate concentrations were measured in the summer season, which could be explained by the increase of solar flux during this period of the year. Trace element contents of the analyzed samples were also showed very clear seasonal variation, for instance, elements originated from crustal sources were high in summer season due to the dryness of the soil; however, marine originated elements were high in winter season, due to the dominance of bubble bursting process over the sea surface. Finally, some statistical techniques such as Positive Matrix Factorization (PMF) and Potential Source Contribution Function (PSCF) were utilized to find the types and locations of sources affecting atmospheric composition of particles in the Eastern Mediterranean atmosphere.



## Forecasting the Arrival of Soybean Rust in the Continental United States Using a High Resolution Mesoscale Model and a Dispersion/Deposition Model

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### Abstract

On November 10, 2004, the United States Department of Agriculture announced that soybean rust (*Phakopsora Pachyrhizi*) has been detected for the first time in United States. This fungal disease has invaded all major soybean production regions of the world and caused billions dollar in losses each year in South America, Africa, and Asia. The introduction of soybean rust into the United States has the potential of causing \$2 billion dollars of yield losses if a major epidemic occurs. Extending a pollen forecasting technique, we have developed a method that accurately forecasted the arrival of soybean rust in the continental United States. This paper describes the methodology used to forecast and diagnose soybean rust transport mechanisms. The results show that this methodology is capable of predicting the locations of soybean rust infestations months in advance based meteorological conditions and disease source distribution. This forecasting methodology indicated that the cloudiness and heavy rainfall associated with tropical storms is the key factor in determining the spore viability and the abnormally large number of hurricanes was likely responsible for the invasion the disease. The technique used to forecast the arrival of soybean rust in the United States can also be used to forecast the spread of soybean rust in the United States.

### Introduction

Asian soybean rust (*Phakopsora Pachyrhizi*) was confirmed on soybean plants in Louisiana on November 10, 2004 (see Fig. 1) and has since been discovered in five other states: Alabama, Mississippi, Florida, Georgia and Arkansas (USDA/APHIS). The continental United States is the last major soybean production region affected by the disease. Soybean rust was first detected in the Northern Hemisphere in Cali (at 3.7°N, 76°W), Columbia in August 2004. The disease was first recorded in Japan in 1903, and identified for the first time in the Western Hemisphere in Hawaii in 1994. In 1996, the disease moved from Asia into Uganda, and by 2001, it had spread throughout much of Africa (Pivonia *et al*, 2004). In 2001, soybean rust was found in South America, and it has spread throughout the soybean growing areas of Brazil, Paraguay, and Bolivia. By 2003, rust had also spread to a northern, non-soybean growing area of Argentina. Severe outbreaks in the last few years in South America have heightened concern for the spread of the disease to the North American soybean. This aggressive fungal disease can reduce soybean yields by as much as 80 percent in individual field if growing conditions are favorable for its spread (Yang *et al.*; 1991). A conservative prediction indicated losses greater than 10% in nearly all the U.S. growing areas with losses up to 50% in the Mississippi delta and southeastern coastal states (Yang *et al.*; 1991).

Knowing when and where the first infection occurs provides the opportunity to control the spread of the pathogen. Although, soybean rust infections can be managed by the proper application of fungicides, the application of the fungicides is time critical and expensive. Spraying fields unnecessarily is expensive and spraying at the wrong time provides no protection to the fields. Correct application requires knowledge of which fields are infected and when.

Since the risk associated with the spread of soybean rust is high a method of forecasting which fields are infected and when would provide growers the necessary information to properly apply fungicides to protect their fields. Two approaches to the prediction of the long-distance dispersal of the soybean rust pathogen can be taken. The first is a statistical, based climatological atmospheric movement. Trajectories of SBR movement using climatological conditions of spore were used by Isard, *et al*, 2003. Although this approach can give a general climatological map of disease movement, it cannot predict the extreme events, which is more typical of disease dispersal (Nagarajan and Singh 1990; Nagarajan et al 1976). The second approach

makes use of numerical model to predict the air movement, based theoretical governing equations. This method can predict extreme events since the model equations predict the actual atmosphere rather than a long-term average atmosphere created via the statistical approach described earlier. The obvious disadvantage with modeling approach that the model's predictions are not always accurate. In principle a third approach is to combine the modeling and statistical approaches. This paper describes the second dynamic modeling approach.

We extend the methodology developed by Pasken and Pietrowicz (2005) and Pietrowicz and Pasken (2002) for forecasting pollen concentrations, to forecast the spread of soybean rust on hemispheric scale. We have combined a global-scale long-term forecast model with their pollen forecasting methodology to create a forecast of when soybean rust would arrive in the United States from the southern hemisphere. Initial experiments conducted in 2003 indicated that risk of SBR migrating to the continental United States via atmospheric dispersion from South America was small and if SBR were to be found in the continental United States, it would have to enter by crossing the Central American land bridge. The situation changed dramatically during 2004. Forecasts created during the summer of 2004 showed significant spore concentrations in the southern United States during August and September. These maps projected that SBR would be transported from South America to the southern United States during the intense 2004 hurricane season. This forecast was verified November 10, 2005 when the United States Department of Agriculture announced that soybean plants in Louisiana had been infected with *Phakopsora pachyrhizi*. Although our forecast cannot be fully tested without ground truth measurements of SBR spore concentrations, these forecasts were verified by comparing forecasts with soybean dispersion computed from meteorological data collected after the forecast, most favorable pathway frequency computations and from USDA data.

### Methodology

We have chosen to forecast the dispersion of SBR via a coupled model system. Meteorological models are used to create the data needed by the transport/dispersion/deposition model. Two meteorological models are used. The first is a global-scale long-term coarse-resolution model, which provides the boundary conditions for a second, higher-resolution model. Long-term prediction of the state of the atmosphere is a very challenging problem and most often is done at much coarser spatial and temporal scales than required by the dispersion models. Meso- $\beta$  or  $\gamma$  scale models provide the necessary high-resolution spatial and temporal forecasts, but only over short periods. Coupling these two modeling systems provides a means to create high spatial and temporal resolution meteorological data over a 60-to-80 day period. We have chosen to couple the global-scale climate model from the Scripps Institution of Oceanography's Experimental Climate Prediction Center (ECPC) model to provide the long-term lateral boundary conditions with the National Center for Atmospheric Research's MM5 mesoscale model. The high spatial and temporal scale meteorological data is then used as input to the National Oceanographic and Atmospheric Administration's (NOAA) HYSPLIT\_4, one of the most advanced modeling system for trajectories, dispersion and deposition available.

### Meteorological Models

As noted, our coupled modeling system requires a large-scale global model to provide the lateral boundary conditions needed by the regional scale model that produces the high spatial and temporal resolution seasonal meteorological forecast. We adopted large-scale model from the ECPC to provide 6-hourly forecasts up to 16 weeks into future (Roads et al. 2001). We coupled the global scale ECPC model with MM5 to generate the high spatial and temporal distribution forecast of meteorological parameters to produce the meteorological features affect transport, diffusion and deposition of particulates, such as soybean rust. MM5, developed by the Pennsylvania State University and the National Center for Atmospheric Research (NCAR), was chosen because of the flexibility the model provides in forecasting the meteorological fields (Dudhia *et al.*; 2001). The MM5 model has seven Planetary Boundary Layer (PBL) schemes, six cumulus parameterizations, three land surface models as well as many other physics options. MM5 supports the two-way nesting of domains allowing covering large-scale transport phenomena across Americas and at the same time to resolve fine-scale source distribution of soybean fields. The outer most domains were configured with 23 vertical layers with 127x97 grid points centered at 1.0°N, 76.0°W with a horizontal resolution 120 km. The inner nested domain covers the Central America including southeastern U.S. with a resolution of 40km. The lateral boundary conditions were imposed by considering an 11-grid-

point nudging zone adjacent to each lateral boundary where the weighting of the observed data was reduced linearly away from the boundaries to the interior of the model domain.

MM5 allows the number of layers in the model to best suit the forecasters' needs and it includes a land surface model that includes vegetation, soil types and deep soil temperatures. We chose the 24-category database and physical parameters for the western hemisphere. MM5 has been extensively tested by the meteorological community and has a reputation for producing high quality forecasts. The MM5/PSU modeling system has been successfully used for a broad spectrum of studies ranging from theoretical to real-time studies. These results have been carefully studied to test the accuracy of the modeling system. These studies show that the MM5/PSU modeling system is the best performing modeling system for meso- $\alpha$  and meso- $\gamma$  scale studies.

### **HYSPLIT\_4 Model**

The National Oceanic and Atmospheric Administration, NOAA, and Air Resources Laboratory's HYSPLIT\_4 (Draxler and Hess, 1998), which has been under development since 1982, was used to calculate the transport/diffusion and deposition of the SBR spores. HYSPLIT\_4 uses gridded model output or a series of gridded meteorological fields from a variety of sources ; such as Eta Data Analysis System (EDAS) or output of a forecast model such as MM5/PSU, Eta, AVN or other NCEP models as input. The model uses a hybrid between the Eulerian and Lagrangian coordinates to calculate trajectories and dispersion of air parcels. Particle advection and diffusion calculations are made in a Lagrangian framework while concentrations are calculated on a fixed grid. Air concentration calculations associate the mass of the airborne particulates with the release of puffs, particles, or a combination of both, which is user specified.

The dispersion rate is calculated from the vertical diffusivity profile, wind shear, and horizontal deformation of the wind field. Air concentrations are calculated at a specific grid point for puffs and as cell-average concentrations for particles (Draxler and Hess, 1998). One of three assumptions can be used to compute air concentrations along the parcels trajectories, a puff model, a particle model or a combination of puff and particle models called PARTPUF. The combination method was used in this study.

Options in HYSPLIT\_4 allows for gravitational settling, wet and dry deposition and re-suspension of pollutants Draxler and Hess (1998). These options give more flexibility to replicate realistic conditions and allow for various tests. This model has been under development since 1982. The accuracy of trajectories and dispersion/deposition using numerical model output has been tested using data collected during several field experiments. Results indicate that no discernible differences were seen with the average error rates in the 20% to 30% range of the travel distance. These results indicate that model output can be used for the meteorological fields for trajectory models. These studies indicate that the use of trajectory models from the use of numerical forecast models, even ones with relatively large grid sizes and time steps, have results that have utility in this study.

Although SBR spores can be transported over great distances, they may not be viable once they are deposited. Spore viability is dependent on four criteria; 1) time, 2) UV-B exposure, 3) temperature and 4) precipitation. The SBR spore concentrations over time were decimated by each of these factors. To decimate the SBR spore concentrations over time, we have assumed an exponential decay, similar to radioactive decay, to reduce the number of viable spores with time. Although SBR spore viability does not typically follow a radioactive decay profile, we do not know how long each spore has been aloft so an exponential decay for all of spores suspended has been assumed.

It is known that SBR spore viability decrease rapidly when the ambient temperatures are sub-zero for more than seven days. Since the source regions for of the SBR spores are in the tropics, the 0° C temperature is about 3-4 km above the ground surface. Only those spores above 3 km above were decimated over the total duration for which the spores were suspended.

SBR spore viability is also affected by exposure to UV-B radiation. Unfortunately determining the amount of UV-B exposure is a complex since exposure is affected by the amount of cloud cover of the current location and the amount of ice coating if any. We have assumed to have an e-folding time at 60 MJ day<sup>-1</sup> total radiation, of which about 30 MJ per day is in UV portion.

In order to inoculate a soybean plant, the SBR spores require moisture. Precipitation washout of the SBR spore provides a means of removing the SBR spores from the air and provides the necessary moisture needed by the spores. We adopted half-time wet deposition of 25 mm precipitation (Isard *et al.*; 2003).

We use pathway frequency as a measure of the probability that an area or region is susceptible to infection of SBR spores. It should be pointed out that the maximum likelihood pathway frequencies are computed only from trajectories, thus do not represent actual spore concentrations, but are related to source regions. The pathway frequency can be interpreted as a region where SBR infections could occur.

### Results

Experiments with the coupled for SBR forecasting began in 2003 with the creation of maximum likelihood pathway frequency maps, similar to those produced by Isard *et al.* (2003). These pathway frequency maps indicated that there were only a few pathways to the southern United States. The forecasts of SBR concentrations over our domain of interest also indicated that it was unlikely that SBR would arrive in the Continental United States in 2003.

The same series of experiments were conducted during the summer of 2004. These new experiments indicated a different scenario for the late summer and early fall of 2004. As with the previous year two prediction experiments were carried out: one creating the maximum likelihood pathway frequency maps, and the second creating the concentration maps. Further, these forecast experiments were carried for two 40-day periods, one starting early August and the other in early September. The first forecast period was begun after the detection of SBR in Cali and the second one corresponding to the middle of hurricane season.

Figure 2a shows the prediction of spore pathway as represented by the high frequency of pathway beginning on August 7, 2004 and ending on September 17, 2004. As noted earlier the ECPC's GCM provided the boundary whereas MM5 provided the meteorological forecasts over the forecast region. Within MM5 a two-way interactive nesting is also employed, which means the information in coarse and fine MM5 domains are free to interact with each other, which results in better meteorological simulations.

The source points of spores were placed in  $2.0^\circ \times 2.0^\circ$  rectangular area centered at Cali (Fig. 2). The source points were uniformly distributed at 90 grid points separated by  $0.05^\circ$  of latitude and longitude. The source is assumed to release spores continuously during the integration and the spores can reach at least 10 m above the ground by turbulence. The majority of spores released from the sources moved westward to the Pacific Ocean following the easterly winds. These spores turned north later but did not reach the northern Mexico or United States.

The first maximum likelihood pathway frequency forecast was conducted without any environmental limitation on the spore viability. In this experiment, the spores were treated as markers on air parcel trajectories. The spore clouds mainly moved northward in large amounts across southern and eastern Mexico and the Gulf of Mexico and finally reached the continental U.S. The high frequency areas cover south and eastern Texas, Louisiana, and moderate frequency in Georgia, and Florida (see Fig 2a). This would suggest the disease in Texas where no rust was detected so far. Although the dry weather during the time period could have prevented the disease. The second maximum likelihood pathway frequency forecast included UV-B decimation and a 10-day life span. This limitation mimics the temperature effect on spore viability. Incorporating UV-B decimation and a ten-day lifetime results in a sharply reduced maximum likelihood pathway frequency (Fig. 2b). Only two distinct strands of spores moved northward directly towards the U.S. One started from the southern tip of Gulf of Mexico along east coast of Mexico and finally reached Louisiana, U.S. where the first detection of the rust and most widely spread among all five states occurred. The pathway matches closely the detected locations of the rust shown in Fig. 1. This strand also resembles the track of Hurricane Bonnie that developed over the Gulf of Mexico during August 9-12, 2004 although the model predicted track is somewhat east of the observed. The reason can be that the storm developed from the ocean sheltered the rust spores from UV radiation. The spores in the storm-free areas were killed after enough radiation exposure. The other strand directly extends from Cali to Georgia and Florida. Its frequency is lower the western one, in agreement with relatively less widespread of the rust in these two states. This strand again is associated with the cloudiness of a storm, most likely Hurricane Ivan.

The distinct strand structure when UV radiation is considered strongly suggests the key importance of UV radiation as a criterion for the spore viability. The correlation between hurricane tracks and spore pathways provides further evidence that clouds in a storm system protect spores from UV radiation.

Nevertheless the diagnosis of the single release of the spores showed the western strand occurred in early August may be responsible for the rust in Louisiana, and neighboring states and while the eastern one in early September is responsible for the disease in Georgia and Florida. The early one was likely associated with Hurricane Bonnie while the late one was likely related to Hurricane Ivan. It should be noted that the accurate timing does not have significant meaning for single events in the framework of seasonal or long-term prediction. This probably explains why the time difference in the two strands (~one month) does not correspond the time difference in disease occurrence among states.

A second forecast was begun on September 4, again for 40-days, ending before the first detection of the disease. This forecast also showed a clear preferred pathway towards the Louisiana, suggesting that this pathway is the most likely route. Based on this forecast the air parcels first moved from the Cali region and then move northeast into Ivan system following the track of the cyclone to the United States.

Although the maximum likelihood pathway frequency forecasts provide an indication of SBR contamination, these forecasts do not account for dispersion, deposition washout and other factors that would alter the number of SBR spores in an air parcel. Indeed the maximum likelihood pathway frequency forecasts merely indicated the number of air parcels that may contain SBR spores that have passed over a particular point in space. A more accurate approach is to determine the actual SBR spore concentration from the SBR spore emission rates and a dispersion/deposition model. NOAA's HYSPLIT\_4 modeling system for trajectories, dispersion and deposition was again combined with NCAR's MM5 to produce forecasts of SBR spore concentrations except HYSPLIT-4 was run in concentration mode rather than trajectory mode. Source regions for SBR spores in Brazil and Columbia were obtained from Yang *et al.* (200X). Each source region occupied the full area of the grid square and an emission rate of  $3.619 \times 10^{12}$  spores per hectare per hour (Isard, *et al.*; 2003). The number of spores deposited on the surface and spore concentrations at 5, 10, 100, 500, 1000 and 5000 meters above ground level (AGL) were computed over the 40-day period. As with the maximum likelihood pathway frequency forecasts UV-B decimation and a 10-day life span in the air is imposed on the spores.

Unlike the maximum likelihood pathway frequency chart shown in Fig. 2, the concentration forecasts are created every 3 hours at each of the seven levels. The resulting concentrations are plotted to determine the areas in the forecast domain affected by SBR spores. The resulting concentration maps allow the changes in concentrations change over time. A single concentration map does not describe the detail shown in an animation of the same region. Time-lapse movies of the concentrations at each of the seven layers for the 40-day period are available at <http://www.eas.slu.edu/People/RWPasken/sbr/movies>. These time-lapse movies clearly demonstrate that the SBR migrated from Brazil and Columbia into the United States as each of the hurricanes passed from the Atlantic to Gulf of Mexico and finally the southern Gulf Coast. Figure 3a is the number of soybean rust spores deposited per square meter on the for 06 UTC on August 11, 2004. Fig 3a shows that the SBR spores are concentrated in South America and along the Inter-Tropical Convergence Zone (ITCZ) on the west side of Central America. By 21 UTC on August 11, 2004 a strand of SBR spores from Columbia and Brazil has stretched to Haiti and the Dominican Republic. The SBR spore concentrations for 12 UTC August 15, 2004 (Fig. 3c) mark Hurricane Francis's landfall over southern Florida. The simulation indicated that as Francis makes landfall approximately 100 SBR spores per square meter were deposited over most of Florida. The Gulf Coasts of Louisiana, Mississippi, Georgia and Alabama indicates surface depositions of 1 to 5 SBR spores per square meter. It is important to note that figure show the number of spores deposited on the surface over a 3-hour period. The total accumulation over the lifetime of the event would be higher.

The importance of the fine-scale structure of the atmosphere in controlling the dispersion and deposition of SBR spores is revealed in the vertical distribution of SBR spores (Fig. 4). The atmospheric boundary layer is typically 1 km deep, while over 70% of the SBR spores were trapped below 4 km AGL. Although the Fig. 4 shows the SBR spores initially trapped below 1 km AGL and deepens over time to 4 km, the increase in depth is an artifact of the initial launch of the SBR spores at the beginning of the forecast. This is because the model starts with an atmosphere void of any spores. In reality, the SBR spores were already present throughout the 4 km deep column near the end of the forecast period. The shallow depth over which

the SBR spores are spread suggests that the determining factor for the dispersion and deposition of the spores would appear to be the depth of the boundary layer and vertical dispersion and transport in the small-scale flow.

Heavy precipitation associated with the hurricanes is indicator of the importance of the small-scale and boundary layer flow. Wet deposition of SBR spores promotes inoculation of rust (Yang *et al.*; 1991, Yang *et al.*; 1990). For example, the heavy rainfall center close to the eye wall of Ivan (to a lesser degree, Bonnie) coincides with the region where fields infected with SBR were found. This result strongly suggests the rainfall washout played a key role during the event. The heavy rainfall washed out the airborne spores and provided moisture for the subsequent disease to develop (Tan *et al.*; 1996). Rainfall amount and rain frequency are very important determinants for the disease epidemics (Yang *et al.*, 1990). In contrast, the lack of rainfall in Texas provides an explanation for the lack of SBR infection, although the minimal number of hosts in this region may be partly responsible also. Even if SBR spores present, there are no mechanisms to deposit them to the ground and the lack of moisture would inhibit their survival, and increase the UV-B radiation which would make the viability of SBR spores difficult.

If the small-scale flow were the only determining factor, the SBR spores could not be carried over large distances. If, however, the large-scale flow were also a controlling factor then the structure of the ITCZ would control the spread of the SBR spores. Typically the ITCZ would carry the spores to the Pacific Ocean. Strong meridional flows are associated with numerous strong hurricanes disrupted the climatological zonal flow, permitting air parcels to move across the weakened ITCZ. This is seen by eight named hurricanes occurring in August, a record (Shinngle; 2004). Fig. 5 is the streamlines at 850 hPa during the period of hurricane Ivan. The streamlines show a strong southerly flow from the Cali region to hurricane tracks. This meridional flow was quite persistent during much of the hurricane season including the time of Bonnie. Thus, this bridging flow carried spores to the hurricanes which transported them to the continental United States. This strong bridging flow implies that the large-scale flow is a major contributing factor in determining the dispersion and deposition of SBR spores. The streamline field in Fig. 6 further demonstrates that movement of SBR spores is controlled by the large-scale structure of the atmosphere. Fig. 6a reveals the classical climatological zonal flow and the transport of SBR spores by the large-scale flow towards the ITCZ and then over the Pacific Ocean. Zonal flow breaks down with the approach of the hurricanes, the zonal flow breaks down and becomes meridional. This allows the SBR spores from both Brazil and Columbia to be drawn into the air entering the hurricane (Fig. 6b).

### Discussion

The primary factor controlling the prediction of the arrival of SBR spores in the United States is the quality of the meteorological forecasts. Although the meteorological models predict the arrival of the hurricanes along the Gulf Coast, the timing and location of the landfall of these hurricanes were incorrect by as much as a day. For our purposes, a forecast error as large as this large is not as important as correctly predicting the occurrence of the hurricanes. Coated with water and shielded from UV radiation by the hurricanes extensive cloud deck the SBR spores are able to survive longer than would otherwise be possible. As the hurricanes passed into the Gulf of Mexico air was drawn from Brazil and Columbia into the hurricanes core. As the hurricanes makes landfall the SBR spores are washed out of the air and onto host plants. The SBR spores act as cloud condensation nuclei (CCN) and become coated with water. Since the spores are coated with water and embedded with the deep cloud cover associated with the hurricane, the spores are protected from the UV that would normally kill them. When the SBR spores are washed out as the hurricanes make landfall (Fig 6c), the water-coated SBR spores have a ready source of the moisture needed to germinate enhancing the risk of SBR infection. Thus, spores drawn into hurricanes have an enhanced probability of infecting hosts

Detailed verification of these forecasts is limited by the lack of measured atmospheric SBR spore concentrations in the continental United States, however, the detailed verification of the methodology by Pasken and Pietrowicz (2005) indicates that it is sound. To further test the methodology, concentration forecasts were recreated using observed data over the same period rather than forecasts in November 2005. These new concentration forecasts remove the meteorological forecast dependency and leave only a dependency on the quality of the dispersion modeling. These new concentration forecasts differed only in the timing of arrival of the SBR spores, suggesting that the meteorological forecast plays a major role in the dispersion and deposition of SBR spores. This is consistent with results discussed in Pasken and

Pietrowicz (2005). In addition, the lack of measured concentrations does not prevent a verification of the maximum likelihood pathway frequency and concentration forecasts in a subjective manner. The success of the long-range meteorological forecasts and the consistency of the maximum likelihood pathway frequency and concentration forecasts predicted the arrival of SBR in the United States in late August and early September. Given the time needed for germination and the time needed to verify the infection, SBR spores would have needed to arrive in the same time frame. These forecasts can be further confirmed by the presence of SBR infections in areas where the highest forecasted SBR spore concentrations and largest maximum likelihood pathway frequency values intersect.

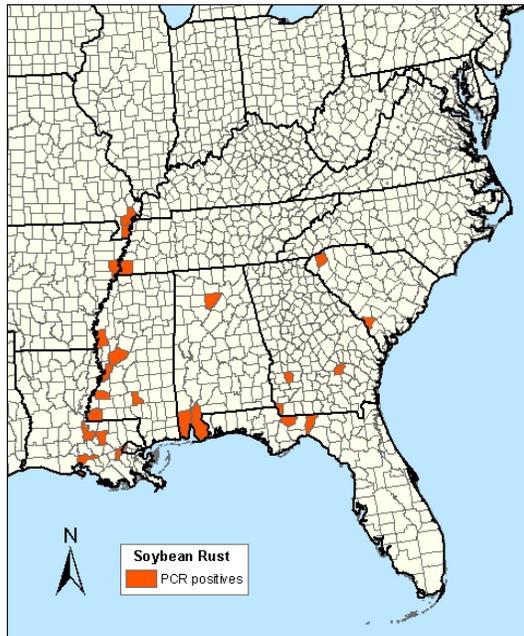
### Conclusions

Forecasting the arrival of *Phakopsora pachyrhizi* in the United States is an extremely complex problem, because it requires meteorological forecasts at several scales of atmospheric motion over an extended period, a dispersion/deposition forecast and an understanding of plant pathology. We have shown that it is possible to make a forecast of the dispersion and deposition of plant pathogens over a long period with reasonable accuracy, as indicated by the location of SBR infections and the correlations of deposition and maximum likelihood pathways.

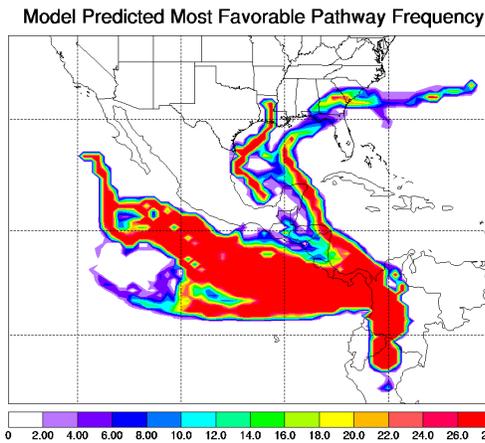
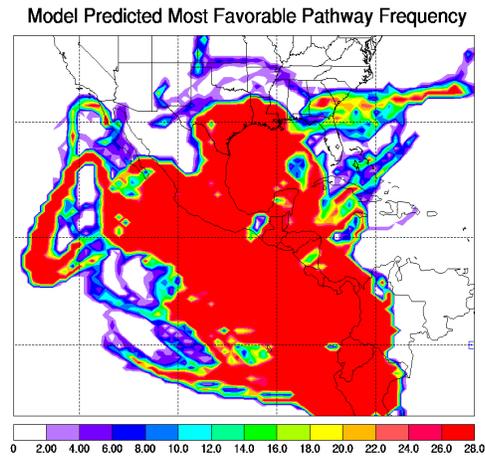
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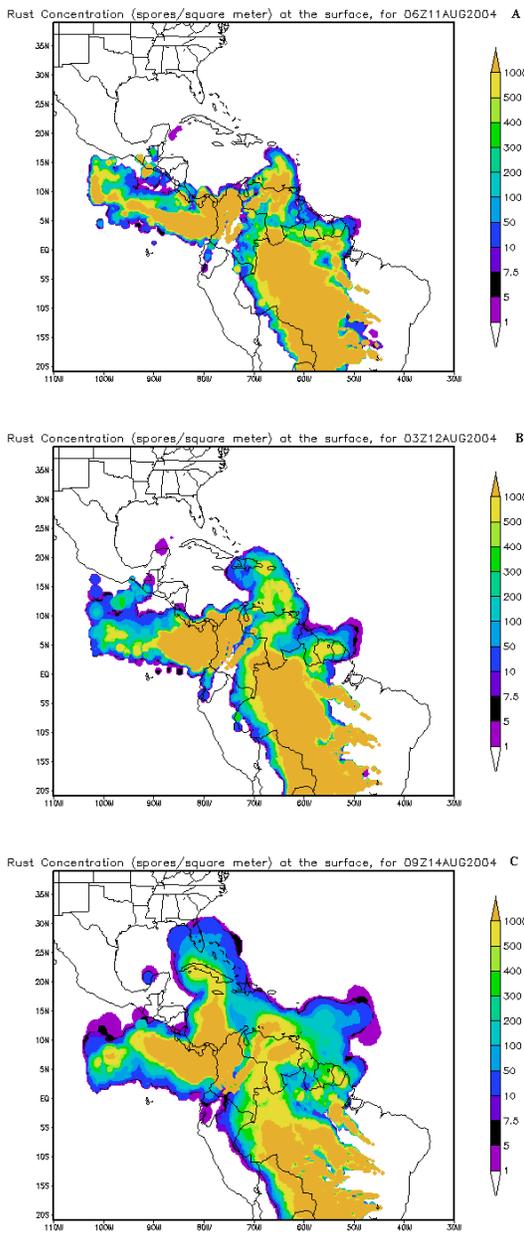
## Figures



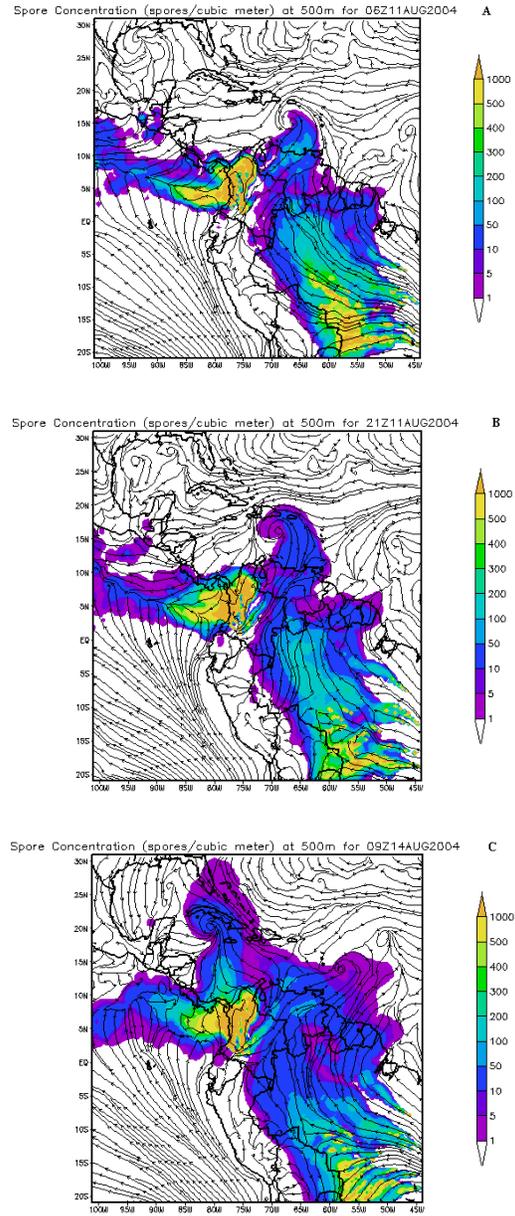
**Figure 1. Sites of soybean rust detected in November 2004.**  
<https://netfiles.uiuc.edu/ariatti/www/SBR/USmap.html>



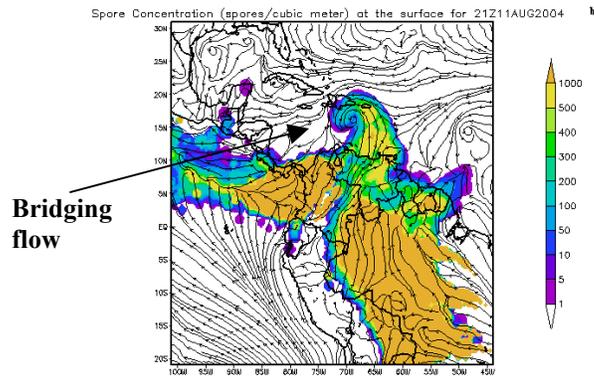
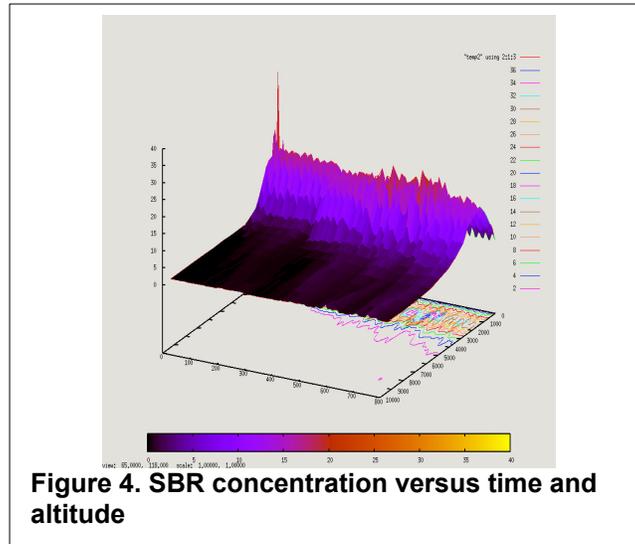
**Figure 2. Predicted most preferred pathway of soybean migration. (a): With no limitation on soybean rust viability; (b) With UV radiation and maximum life time span.**



**Figure 3. Soybean rust spore deposited on the surface and 10m streamlines for (a) 6 UTC August 11, 2004 (b) 3 UTC August 12, 2004 and (c) 9 UTC August 14, 2004**



**Figure 6. Soybean rust spore concentrations and 500 m streamlines for (a) 6 UTC August 11, 2004 (b) 21 UTC August 11, 2004 and (c) 12 UTC August 15, 2004**





## Ammonia Volatilization During and Following Dairy Slurry Application in the Field

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### Abstract

Livestock manure is known to be a major source of atmospheric ammonia (NH<sub>3</sub>). In order to determine the magnitude of these emissions, daytime ammonia fluxes were measured following the application of dairy slurry in a forage field using two micrometeorological approaches. The Relaxed Eddy Accumulation technique and the Flux Gradient technique were used in conjunction with denuder tube coated with oxalic acid as a trapping medium. The flow rate was set at 2 L min<sup>-1</sup>. The 60-cm long denuder tubes were extracted with 3 - 10 mL of deionized water. The NH<sub>4</sub><sup>+</sup> concentration in water was analyzed with an ion chromatograph (Dionex DX-600 System) equipped with a ED50 Electrochemical Detector and using a Dionex CS-16 analytical column, guard column and 38 mM MSA as the eluent at ambient temperature. The detection limit of NH<sub>4</sub><sup>+</sup>-N concentration was 0.24 µg L<sup>-1</sup>. The rate of application of the dairy slurry was 83 kg N ha<sup>-1</sup>. The highest NH<sub>3</sub> emissions occurred on the first day after application with maximum ranging between 11-13 µg N m<sup>-2</sup> s<sup>-1</sup>. The ammonia volatilization was reduced by 6 - 7 fold on the second day and was in the range of 1-2 µg N m<sup>-2</sup> s<sup>-1</sup>. By the third day, NH<sub>3</sub> fluxes were one order of magnitude lower than the second day. Over 11 days of observation, 1.5 kg NH<sub>3</sub>-N ha<sup>-1</sup> was volatilized during the daytime, which is less than 2% of the total slurry amount applied. Almost two third of the NH<sub>3</sub> emissions occurred on the first day. Under these conditions of limited ammonia volatilization, both methods gave consistent results and were able to detect small fluxes. These data will be used to verify process-based models.



## Exploration and Application of Statistical Methods to Identify Vegetation-Surface-Atmospheric Interactions Using Field Observations

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### Abstract

Vegetation affects the surface energy balance and the associated winds and regional circulation patterns. Efforts are underway to understand the magnitude and nature of the different variables that can participate in the land surface – atmosphere processes. There is also considerable variability and uncertainty associated with the specification of vegetation-soil-atmospheric parameters in environmental models. The main objective of this study is to apply statistical methods to identify significant interactions between vegetation, land-surface and atmospheric factors and their effects on latent heat flux in non urban areas. Three different statistical techniques such as (i) a resolution IV fractional factorial (FF) design based approach; (ii) the fractional factor separation (FFS) approach and a generalized least square (GLS) regression method have been used to conduct the analysis. The results from the three different methods are analyzed using data from the ten field stations from the International H<sub>2</sub>O project (IHOP 2002). Significant variability exists between the different stations due to their spatial location, vegetation, availability of soil moisture. The questions this study seeks to answer are (i) how do the three different statistical methods compare in terms of capturing the interaction patterns, and (ii) whether the complete and continuous structure of the explanatory variables provide better information than the partially grouped data? It was found that in all the 10 stations, the fractional factorial methods and Factor Separation methods yielded very similar results and confirmed the presence of at least two way interactions. However higher order interactions were difficult to identify due to the confounding of effects and absence of several combinations. This feature, i.e. inability to always the best combinations of variables settings to extract the different interactions, remains a limitation of the current analysis, which can be reduced in future studies by considering the interaction considerations in designing the field experiments. Multivariate visualization display further confirmed that utilizing continuous information as compared to discrete combinations can better capture the patterns and help explain the large variability present. There were indication that GLS procedure is also useful in terms of detecting higher order interactions and non linear relationships between the factors which were undetectable by both FF and FS methods. The shortcomings of the three approaches are addressed and the trade off between using these approaches is discussed.



## Recovery of Malodorous Livestock Odors and Gases from Tedlar® Sampling Bags Using Olfactometry and Thermal Desorption Tubes

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### Abstract

Analysis of odors with attention to improving air quality is of great interest to agricultural communities; therefore it is important that odor samples are representative of conditions at the time of sampling. The purpose of this research was to quantify the recovery of odor and odorous chemicals in commercial and homemade Tedlar® sampling bags. A standard gas generator was used to produce a standard gas mixture consisting of seven volatile fatty acids: acetic (941 ppbv), propionic (5212 ppbv), isobutyric (468 ppbv), butyric (291 ppbv), isovaleric (296 ppbv), valeric (154 ppbv), and hexanoic (454 ppbv) acids and one semi-volatile organic compound, para-cresol (9 ppbv). The standard gas was placed in the Tedlar® bags and chemical concentrations were quantified at times of one hour, one day, two days, three days, and seven days using thermal desorption tubes and a gas chromatograph/mass spectrometer (GC/MS). Odor concentration, intensity, and hedonic tone were measured by human panelists using dynamic triangular, forced-choice olfactometry.

Chemical recoveries ranged from 2 - 40% one hour after sampling and 0 - 14% after seven days, an indication that chemical concentrations decreased with time in the bags. Odor concentrations in Tedlar® bags as observed by olfactometry panelists decreased approximately 8% after one day and 14-39% after seven days. Minimal changes were observed in intensity and hedonic tone. These results suggest that Tedlar® bags influence gas concentrations with some potential to affect odor. Appropriate precautions should be taken in the selection of odor sampling bags to avoid contamination or non-representative analyses. It is apparent that bag samples will experience considerable reductions in chemical concentrations immediately upon collection. Minimizing time between the sampling event and sample analysis is still important in reducing overall chemical losses.

### Introduction

Air quality is of great interest to agricultural communities. It is therefore important that odor samples are representative of the conditions sampled when testing the quality and quantity of odor experienced in these communities. Tedlar® sampling bags are used by many because of its low cost, low permeability, and non-reactive, chemically inert qualities (Parker et al., 2003). In olfactometry analysis, odor samples are presented to a group of panelists to determine detection threshold (DT), intensity, and hedonic tone.

Zhang et al. (2001) found a poor correlation between intensity measured in the field with Nasal Rangers and odor concentration measured with Tedlar® sample bags and olfactometry. However, the odor intensity of bagged samples did correlate well with the olfactometry measurements. Parker et al. (2003) found background DTs to increase with increased holding time in Tedlar® bags and considered heating at 100°C as well as purging after heating and again prior to sampling necessary to reduce background DTs to an acceptable level. Williams (2003) considered Tedlar® bags able to remove organic acids from swine odor samples unless samples were analyzed within a few hours of collection. Keener et al. (2002) also found that Tedlar® sampling bags had background odorants that affected olfactory analysis of samples. In Keener's study, tri-packed sorbent tubes (Tenax TA, Carboxen 1000, and Carbosieve SIII) were able to recover greater than 74% of the 19 swine odorants tested.

According to Spinherne et al. (2003), volatile fatty acids (VFAs) are considered important components of agricultural, solid waste processing and disposal, industrial and municipal wastewater collection, and treatment system odors. Spinherne developed a standard gas mixture which consisted of acetic, propionic, isobutyric, butyric, isovaleric, valeric, and hexanoic acids and was considered representative of agricultural odors using a gas generator and permeation tubes (Spinherne et al., 2003). Koziel et al. (2004) compared recoveries using a standard gas mixture with SPME fibers and Tedlar, FEP (Teflon), foil, and PET (Melinex) sampling bags. The standard gas mixture used in Koziel's study included acetic acid, propionic acid, isobutyric acid, butyric acid, isovaleric acid, valeric acid, hexanoic acid, p-cresol, 2'-

aminoacetophenone, indole, and 4-ethylphenol. The SPME fibers were identified with the highest recoveries (106.2% and 98.3%), followed by the PET (71.7% and 47.2%), FEP (75.4% and 39.4%), homemade Tedlar® (47.3% and 37.4%), commercial Tedlar® (67.6% and 22.7%), and foil (16.4% and 4.3%) bags. Recoveries were greater at the 0.5 hour storage time as compared to the 24 hour storage time (Koziel et al., 2004). Koziel et al. (2005) used Tenax TA sorbent tubes with the standard gas mixture and found them able to recover 94.8% at the 0.5 hour and 88.3% at 24 hours storage time, realizing the second greatest recoveries when compared to the previous SPME and bag tests.

The objectives of this research were to quantify the recovery of odor and odorous compounds in Tedlar® sampling bags at times of 1 hr to 7 days using a standard gas mixture and GC/MS analysis.

### Materials & Methods

Two types of 10L Tedlar® bags were considered in this study, a commercial Tedlar® (SKC Inc., 863 Valley View Road, Eighty Four, PA 15330) and the WTAMU homemade Tedlar® (constructed with Dupont™ TST20SG4 transparent film, baked at 100°C for 24 hours per Parker et al. (2003)). A standard gas mixture consisting of acetic acid, propionic acid, isobutyric acid, butyric acid, isovaleric acid, valeric acid, hexanoic acid, and p-cresol was used in these trials. Bags were filled with the standard gas mixture seven, three, two, and one day prior to olfactometry testing. On the day of olfactometry testing, a one hour standard gas sample was obtained as well as an odor-free (Zero-Air) air sample in the bag type to be tested that day. This scenario was used for both bag types.

Thermal desorption tube samples (Tenax TA) were obtained from each bag by passing 600 ml of gas through the tube using an SKC Pocket Pump (SKC Inc., Eighty Four, PA 15330). Thermal desorption tube samples were also obtained directly from the standard gas generator to quantify the concentrations of the standard gas.

All tube samples were analyzed using a Perkin-Elmer automated thermal desorber (ATD) (710 Bridgeport Avenue, Shelton CT 06484-4794) and a Varian 3800/Saturn 2000 gas chromatograph (GC) equipped with a mass spectrometer (MS) (Varian Inc., Walnut Creek, CA 94598). Samples were automatically desorbed and injected into the GC/MS. Upon injection, samples were held at 250°C for 20 minutes in the splitless mode with a 1 mL/min flow of helium. The column oven began at 60°C and increased to 230°C at a rate of 6°C per minute for a total run time of 30.3 minutes. The strongest ion, quaternary ion, and other ions as molecularly described by the NIST database were used in the method for accurate identification of each compound. The GC/MS compares the area of each peak in the sample tested to associated peaks in the standard samples used to develop the method, calculating concentration in parts per billion.

Olfactometry analysis was performed at the WTAMU Olfactometry Lab per ASTM Standards (ASTM, 2001a, b, c) on day-7 of each experiment, analyzing odor in sample bags that had been maintained at room temperature for 1-hr, 1-day, 2-days, 3-days, and 7-days. All potential panelists were previously trained in all aspects of the odor analysis. Panelists were selected based on sensitivity to an n-butanol reference gas sample as described in the ASTM standards. Panelists identified with adequate sensitivity that day were eligible for the panel. Those individuals considered hyperosmic or anosmic were excluded. The panel was conducted with four panelists sniffing each sample twice, yielding a total of eight readings per sample. A machine blank was obtained and analyzed before beginning analysis of the actual odor samples.

Samples were analyzed for odor concentration (detection threshold, DT) using a triangular forced-choice olfactometer (AC'SCENT International Olfactometer, St. Croix Sensory, Inc., Lake Elmo, MN). Individual panelist DTs were calculated according to ASTM standards (2001b) as the geometric mean of the concentration at which the last incorrect guess occurred and the next higher concentration where the odor was correctly detected. Samples were analyzed for intensity using a static-scale method by comparison to five standard n-butanol solutions, following the general guidelines of ASTM (2001a). Solutions consisted of 0.25, 0.75, 2.25, 6.75, and 20.25 ml n-butanol per L of water, which corresponded to intensities of 1.0, 2.0, 3.0, 4.0, and 5.0, respectively. Intensity scores ranged from 0.5 for an odor sample weaker than the lowest n-butanol concentration to 5.5 for an odor stronger than the highest concentration, in increments of 0.5. The average intensity was calculated for the panel using the arithmetic mean. Hedonic tone (HT) was determined in a similar manner by sniffing the full strength odor sample. Panelists were asked to subjectively assign a score for hedonic tone on a scale of -4 to +4, with -4 being very unpleasant, 0 being

neutral, and +4 being very pleasant. The average hedonic tone was calculated for the panel using the arithmetic mean.

Concentrations for both the commercial and the homemade Tedlar® bag types were obtained in the GC/MS chemical quantification analyses. Percent recoveries of chemical compounds were calculated as the amount recovered at time T divided by the concentration of the standard gas on the day that bag was sampled. In the olfactometry data, percentage decrease in the panel detection threshold (PDT) was calculated by dividing the PDT of the 1, 2, 3, and 7 day samples by the PDT of the 1-hour samples.

## Results

Detectable concentrations of all eight compounds were detected in the Zero-Air samples for both bag types (Table 1). This indicates that Tedlar® bags have the potential to affect odor analyses of agricultural odors as sampled.

**Table 1. Background Chemical Concentrations in Homemade and Commercial Tedlar® Bags.**

Zero-Air Samples	Homemade Tedlar® Conc. (ppb)	Commercial Tedlar® Conc. (ppb)
Acetic Acid	82.53	29.956
Propionic Acid	118.906	119.148
Isobutyric Acid	4.626	15.444
Butyric Acid	3.074	5.104
Isovaleric Acid	1.256	4.218
Valeric Acid	1.123	1.093
Hexanoic Acid	10.66	7.075
P-cresol	0.142	0.179

Chemical concentrations found in the bags, and therefore percent recoveries, were greater for the 1 hour samples than the 1, 2, 3, or 7 day samples for both types of Tedlar® sampling bags. Recoveries ranged from 2.4% to 40.1% for the 1-hour samples and from 0.2% to 19.8% for the 7-day samples (Tables 2 & 3).

**Table 2. Concentrations and Recoveries in Homemade Tedlar® Bags**

WTAMU Homemade Tedlar®	1-hour		1-day		2-day		3-day		7-day	
	Conc. (ppb)	% Recovery								
Acetic Acid	303.971	19.0%	162.228	13.5%	100.828	10.6%	78.846	8.1%	36.487	6.4%
Propionic Acid	1463.095	21.5%	757.905	12.6%	552.680	10.0%	494.662	8.1%	90.912	1.4%
Isobutyric Acid	218.211	38.4%	183.858	34.3%	143.669	29.3%	139.068	25.3%	28.991	5.4%
Butyric Acid	64.848	17.9%	38.648	11.7%	26.160	8.0%	21.149	6.2%	3.180	1.0%
Isovaleric Acid	94.033	27.0%	62.3	18.0%	45.909	13.9%	42.087	12.0%	0.958	0.3%
Valeric Acid	8.667	4.7%	5.674	3.3%	3.387	1.9%	2.663	1.5%	0.511	0.3%
Hexanoic Acid	86.388	14.5%	42.54	9.7%	29.003	4.9%	23.252	8.2%	4.061	2.6%
P-cresol	0.264	2.4%	0.163	1.6%	0.166	1.3%	0.162	1.5%	0.000	0.0%

**Table 3. Concentrations and Recoveries in Commercial Tedlar® Bags**

SKC Commercial Tedlar®	1-hour		1-day		2-day		3-day		7-day	
	Conc. (ppb)	% Recovery								
Acetic Acid	97.085	14.2%	91.388	9.4%	70.887	8.5%	102.389	12.7%	116.723	14.1%
Propionic Acid	673.731	19.4%	548.008	13.7%	401.134	9.2%	444.378	10.3%	215.662	4.3%
Isobutyric Acid	140.718	40.1%	141.399	35.6%	125.299	33.0%	125.958	31.1%	53.502	11.3%
Butyric Acid	38.294	18.6%	29.696	12.4%	21.832	8.6%	20.326	8.0%	5.711	2.0%
Isovaleric Acid	57.771	27.7%	48.226	20.2%	34.025	13.5%	32.918	12.7%	4.324	1.5%
Valeric Acid	5.463	4.9%	3.064	2.3%	2.363	1.8%	2.255	1.6%	0.331	0.2%
Hexanoic Acid	14.297	3.7%	15.621	2.4%	6.907	1.4%	10.062	2.1%	2.544	0.5%
P-cresol	0.251	4.5%	0.273	3.9%	0.161	2.2%	0.268	3.9%	0.312	4.9%

This data indicates a portion of the chemical compounds present in air sampled with Tedlar® bags would not be available to the panelists upon olfactometry analysis, the amount depending on the length of time in the bag endured between sampling and analysis.

Olfactometry analysis indicated little difference in panel detection threshold (PDT), intensity, or hedonic tone between the homemade and commercial Tedlar® varieties (Table 4). Background odor (zero air in Table 4) was detected by the panelists in both varieties, with a greater PDT (stronger odor) in the commercial Tedlar®. Samples experienced a reduction in odor (PDT) with no great change in hedonic tone or intensity relative to time in the homemade Tedlar® bags. Air samples in the commercial Tedlar® bags however experienced a slight increase then a decrease in PDT with time, and again no great changes were apparent in hedonic tone or intensity. Percent reductions show an 8 % loss in odor in the first 24 hours (1-day) and a 14-39% loss in 7 days.

It should be noted that while we were able to determine the concentration of the chemical compounds in the standard gas, we did not directly measure the PDT of the standard gas because the gas had to be placed in the Tedlar® bags for olfactometry analysis. For this reason, the percent recoveries in the olfactometry analysis were calculated relative to the 1-hour sample. Thus, a direct comparison of the percent recoveries between the chemical concentrations (Tables 2 & 3) and the olfactometry data (Table 4) is not valid.

**Table 4. Olfactometry data for both the homemade and commercial Tedlar® varieties.**

	PDT	% Recovery	Intensity	Hedonic Tone (4)
<b>Homemade Tedlar</b>				
Zero Air	8		0.8	0.0
1 Hour	323		3.1	-2.5
1 Day	296	92%	3.1	-2.3
2 Day	271	84%	3.5	-2.5
3 Day	232	72%	3.3	-2.5
7 Day	196	61%	3.3	-2.5
<b>Commercial Tedlar</b>				
Zero Air	19		1.8	-0.8
1 Hour	191		2.8	-2.0
1 Day	298	92%	3.1	-2.0
2 Day	401	124%	2.1	-2.0
3 Day	401	124%	2.5	-2.0
7 Day	277	86%	2.3	-1.8

## Conclusions

The chemical compounds present in the standard gas could not be completely recovered once placed into either commercial or homemade Tedlar® sampling bags. The time spent in the bags affected odor as perceived by olfactometry panelists, as would be expected from the reduced recoveries of chemical components. Therefore, Tedlar® bags may not be the best sampling device for maintaining chemical integrity of air samples for odor analysis. Other methods of air sampling may provide a more chemically complete sample for analysis.

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## Spatial Variability of Emissions from Swine Confinement Operations

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### Abstract

Over the past five years the interest in gaseous emissions from animal confinement operations has increased rapidly. The increase in the size of the facilities as well as the geographic concentration of facilities has led to serious concerns regarding adverse health affects on workers and the surrounding population. Attempts to regulate the emissions by state and federal agencies have been hampered by the complexity of the required analyses and variability of emission factors found in the literature. Because of this variability, it is very difficult to make measurements that reflect the overall emissions from the facilities.

Ammonia concentrations around facilities range from less than 10 ppbv to 100 ppmv depending on factors such as season, wind speed, wind direction and distance away from the facilities. The type of storage lagoon such as covered verses uncovered affects the concentrations of gases such as ammonia, methane and hydrogen sulfide. Daily average ammonia and hydrogen sulfide concentrations were 805 ppbv and 25 ppbv, respectively over an open lagoon and the daily average methane concentration was 24 ppmv over a covered lagoon. The choice of sampling height, location of the sampler, and season also affected the concentrations measured. Some sort of “standard protocol” must be established before emission rates can to determined and applied to swine confinement operations.

### Introduction

Over the past five years the interest in gaseous emissions from animal confinement operations has increased rapidly. The increase in the size of the facilities as well as the geographic concentration of facilities has led to serious concerns regarding adverse health affects on workers and the surrounding population. Attempts to regulate the emissions by state and federal agencies have been hampered by the complexity of the required analyses and variability of emission factors found in the literature (USEPA, 2001 and Wathes et al., 1998). The principles used to govern the monitoring of stack emissions from factories are not applicable to these animal production facilities. Even in tunnel ventilated buildings with as few as four exit fans, it is not well defined as to where to put the sensors, whereas it is a rather simple task to determine the location of the sensor(s) in a stack from a factory. Constant, uniform emissions from a factory are quite different than emissions from an animal confinement operation. Gaseous and particulate emissions from these animal facilities are governed by many parameters, some controllable and some not. Within the building, diet, size, season, and feeding activity all affect the concentrations of gases and particulates emitted from that facility (ISU, 2002 and Zhu et al., 1997). The type of manure storage such as depth and covering also impact the emissions (Harper et al., 2000, Sharpe, et al., 2002 and Zahn et al., 2001). The difficulty of taking measurement at the property line or around facilities is compounded by wind direction, wind speed, time of day, time of year, horizontal and vertical distance (Pfeiffer, 2004). Add in the problems associated with instrument variability and ammonia measurements in general, and then the assessment of emissions from confinement facilities becomes a very difficult undertaking (Sumner, A.L., 2004).

### Site Descriptions

The two lagoons are located in northern Missouri. The Whitetail Site with the biocover is located just north of Unionville, MO., and the uncovered lagoon at the Locust Ridge Site is located south of Lucerne, MO. The lagoon at Whitetail occupies an area of 14,880 m<sup>2</sup> at a depth of 7.3 m. It serves approximately 8800 pigs and is part of a multicomponent manure management program. The lagoon was covered with a Biocap™ geotextile cover manufactured by Amoco and distributed by Baumgartner Environics, Olivia, MN. The lagoon at Locust Ridge occupies an area of 14190 m<sup>2</sup> at a depth of 7.3 m. It serves approximately 8800 pigs and serves as a primary storage lagoon. This is significantly different than the function of the lagoon at White Tail.

The building site consists of 3 buildings housing approximately 2500 pigs and is located in Boone County, Iowa. Manure is collected in deep pits below each of the buildings. The pits are flushed yearly following the corn harvest and applied to fields. The ventilation system is classified as natural with curtains opening and closing as required to regulate the indoor temperatures. Pit fans (6 per building) are run as programmed by the manager of the site.

### **Methodology for Lagoon Sampling Systems**

Three sampling lines for each site were built using a combination of Teflon tubing, foam insulation, aluminum tape and a self-controlling heat tape system. The sampling lines ranged from 50 feet to 300 feet in length and were located such that one sampling line was over the middle of the lagoon, another on the north berm and another on the south berm. The sampling lines were constructed by wrapping aluminum tape around Teflon tubing (PFA, 1/2in ID). The Raychem<sup>R</sup> (Menlo, CA) heating system was run along this line. This combination was then wrapped with another layer of aluminum tape. The heating system was set to maintain a temperature of 40° F. The entire line, consisting of the Teflon tubing and heating line wrapped with aluminum tape, was then placed inside 6 foot long tubes of black pipe insulation ( 5/8" ID x 1/2" thickness) placed end-to-end. The entire sampling line was then placed into another series of black pipe insulation tubes (2 1/8" ID x 1/2" thickness). Duck tape was used to seal the seams forming a continuous sampling line. To minimize damage from rodents, snow and soil surface temperatures, the berm lines were suspended above ground using metal posts. The lines extended into the heated trailer where they were attached to the sampling manifold system. The manifold system consisted of a 8-port sampling valve and controller from Valco Instruments (Houston, TX), a Teflon<sup>R</sup> manifold, and 2 vacuum pumps. The valve was designed such that while one port is being sampled via the manifold on one vacuum pump, the other seven ports are linked to a common exit port that is attached to the other vacuum pump. The valve is designated SC, for common outlet configuration and MW, for low pressure. The valve was controlled by a high torque multiposition microelectric actuator from Valco linked to a Valco DVSP4 digital valve sequencer that served as a timer to rotate the valve to a different port every 15 minutes. The vacuum pump was set so that each of the seven non-sampling ports and associated sampling lines were pumped at a rate of 3 liters/minute. The continuous pumping served a dual function: 1) to allow for instantaneous measurements when the port is switched to sampling mode since there is no dead volume, and 2) to minimize the affects of NH<sub>3</sub> sticking to the tubing, hence creating a pseudo equilibrium between the air and the tubing. The port being sampled was pumped at a rate set to just exceed the requirements for the analyzers. Each port was held in the sampling mode for 15 minutes. Ports 1, 3, 5, and 8 were connected to the sampling lines from 2 meters above the lagoon, north berm, 1 meter above the lagoon, and south berm. Ports 2, 4, 6, and 7 were connected to lines in the trailer from which the air in the trailer was drawn though activated carbon resulting in a "blank" air. The purpose of the "blank" air was to allow the analyzers to return to baseline between sampling.

### **Micrometeorology**

Floating platforms were constructed using 8" PVC pipe filled with Styrofoam balls and sealed at both ends. Attached to the PVC pipes that served as pontoons was a welded aluminum frame that had a central mast extending greater than 2.0 meters above the surface of the lagoon. The entire platform was placed in the center of each lagoon and attached to cable(s) that crossed the lagoon. The cables were anchored at the berms. The platform was free to move up and down depending on the liquid levels in the lagoon. However, there was some flexibility resulting in rotational movement relative to the winds. This resulted in errors relative to the measurement of wind direction. Since wind direction was used only in selecting the proper berm for background concentration corrections, it was not a serious problem. Micrometeorology sensors were attached to the masts at heights equal to 0.25, 0.5, 1.0, and 2.0 meters above the surface of the lagoons. A Model 03001 Wind Sentry (R.M. Young, Grand Rapids, MI) was placed at a height of 2.0 meters to measure wind speed and direction. At the other heights a Model 03101-5 anemometer from R.M Young was installed to measure wind speed only. A CS500 Temperature and Relative Humidity Probe (Campbell Scientific, Logan UT) was placed at a height of 2.0 meters to measure air temperature and relative humidity. Model 107-L Temperature probes from Campbell Scientific were placed at the other heights to measure air temperature only. Data was collected on a Model 21X data logger from Campbell Scientific. A 12 volt system using a MSX20 solar panel from Campbell Scientific was used to power the system. A cell phone system consisting of a COM100 Motorola analog cellular phone package and a

COM210 phone modem from Campbell Scientific were installed for direct communications to Ames, Iowa. Data averaged over 10 minute intervals was downloaded to Ames from the data logger using this system.

### Instrumentation for Ammonia Measurements

Ammonia concentrations were determined using a Model 17C Chemiluminescence Analyzer (Thermo Environmental Instruments, Franklin, MA). The Model 17C utilizes the reaction of nitric oxide (NO) with ozone (O<sub>3</sub>) as its basic principle of detection. The sample is drawn into the instrument by an external pump and after it reaches the reaction chamber, it mixes with ozone, which is generated by the internal ozonator. The following chemical reaction then occurs  $\text{NO} + \text{O}_3 = \text{NO}_2 + \text{O}_2 + \text{Light}$ . The intensity of the luminescence, which is detected by a photomultiplier tube, is proportional to the concentration of NO in the air being sampled.

To measure the total nitrogen concentration (N<sub>T</sub>) both nitric oxide (NO<sub>2</sub>) and ammonia (NH<sub>3</sub>) must be transformed to NO prior to reaching the reaction chamber. This transformation takes place in a stainless steel converter heated to approximately 775°C. Upon reaching the reaction chamber, the converted molecules along with the original NO molecules react with ozone. The resulting signal represents the N<sub>T</sub> reading. In the NO<sub>x</sub> mode, the gas passes through a low temperature converter that converts NO<sub>2</sub> into NO without converting any of the ammonia. This gas stream passes into the reaction chamber where the signal generated is the sum of the original NO plus NO<sub>2</sub> and is referred to as the NO<sub>x</sub> fraction. The NH<sub>3</sub> concentration is then determined by subtracting the signal obtained in the N<sub>T</sub> mode from the signal obtained in the NO<sub>x</sub> mode.

### Instrumentation for Hydrogen Sulfide Measurements

A Model M101A H<sub>2</sub>S Analyzer (Advanced Pollution Instrumentation, San Diego, CA), was used to measure the concentrations of hydrogen sulfide. The M101A is equipped with a low temperature (315°C) converter that converts H<sub>2</sub>S to SO<sub>2</sub> while leaving other gases in the sample unaffected. Sample gas is drawn into the analyzer by an external pump and passes through a scrubber to remove all traces of naturally occurring SO<sub>2</sub>. The converter then converts the H<sub>2</sub>S into SO<sub>2</sub> which is measured by fluorescence. However, a recent report from Battelle (Sumner et al., 2005) indicates that other sulfur-containing compounds are also reduced and subsequently reported as H<sub>2</sub>S. What is actually being measured, reduced sulfur or H<sub>2</sub>S, is likely to receive attention in the near future. The results given in this paper are presented as H<sub>2</sub>S.

### Instrumentation for Methane Measurements

A Model 55C Analyzer (Thermal Environmental Instruments, Franklin, MA) was used for the detection and quantitation of methane and volatile organic compounds (VOC's). The instrument uses back-flush chromatography with flame ionization detection (FID) to measure methane and VOC's. A proprietary GC column achieves complete separation of methane from all C<sub>2</sub> and greater carbon containing compounds and the VOC's. Methane passes through the GC column directly for analysis while all other components are retained on the column. CH<sub>4</sub> is measured by the FID, and then the gas flow is switched to allow purging of the column and the subsequent measurement of all the retained compounds by the FID. The VOC's elute as a large, broad peak and quantitation is accomplished relative to some carbon containing compound such as hexane or propane. Results are reported as propane, hexane, or carbon equivalents.

### Instrument Calibration

Instruments were calibrated weekly using cylinder gases (Scott Specialty Gases, Plumsteadville, PA). The highest quality standards are necessary for the primary calibration gases. For ammonia, Scott's Title V<sup>R</sup> Grade standards were used. For hydrogen sulfide, EPA Protocol<sup>R</sup> Grade standards were used. Methane was an EPA Certified Grade. Lower quality gas standards were used for the weekly calibrations; however, they were verified as to their actual concentrations against the primary standards. The weekly calibrations were done at a single concentration which was chosen based on the actual real-time data such that that the concentration was in the middle of the ranges being measured. For linearity checks the primary calibration standards were diluted using a gas dilution system. Two systems were used during the study. For the first half of the study, a Model 700 (API, San Diego, CA) was used. This system was a standard dilution system with no special coatings and the mass flow controllers were calibrated but not with NIST traceable

standards. For the second half of the study a Series 4000 (EnviroNics, Boston, MA) gas dilution system was used. This system was calibrated with NIST traceable standards and all transfer lines and valves were silanized to minimize problems with ammonia and hydrogen sulfide.

### Data Retrieval

At the beginning of the project, the sites were visited weekly. Data that was stored internally by each analyzer was downloaded to a PC and brought back to Ames for processing. Problems existed with the TEI equipment because its software was compiled on an older version of Windows<sup>R</sup> that was not completely compatible with the newer PC's. In January, following programming by Mr. Scoggin at NSTL, Lab View<sup>R</sup> (National Instruments, Austin, TX) was installed along with internet capabilities at each site. By using Lab View<sup>R</sup>, the instruments could be monitored and controlled remotely and data could be downloaded from the locations to Ames. This could be done more frequently resulting in a decrease in lost data. For this reason the last 6 months of data collection were far more comprehensive than the first 6 months.

## Methodology for Natural Ventilated Buildings

### Instrumentation for Ammonia

The system used was an IonPro IMS (Particle Measuring Systems, Boulder, CO); an air cleanup unit, Model 64-01 (Parker-Hannifin Corporation, Tewksbury, MA); and an continuous oil-less air compressor Model 1000-25B (Jun-Air, Buffalo Grove, IL.). The sample is pulled into the analyzer by a small pump and forced over a semi-permeable membrane on the outside of a reaction cell. The membrane allows materials of interest to enter the detection cell, while other materials pass on, thus removing many possible interferences. Purified dry instrument air sweeps the membrane on the inside of the cell and delivers the sample to the reaction region. There the sample is ionized by a weak plasma formed by a small nickel-63 radioactive source. A dopant material is added to the flow to enhance the ionization process and increase specificity. The ionized sample molecules drift through the cell under the influence of an electric field. An electronic shutter grid allows periodic introduction of the ions into a drift tube where they separate based on charge, mass, and shape. Smaller ions move faster than larger ions through the drift tube and arrive first at the detector. The current created at the detector is amplified, measured as a function of time, and a spectrum generated. A microprocessor evaluates the spectrum for the target compound and determines the concentration based on the peak height

The analyzer is equipped with a 4-port valve that allows air to be constantly pumped through the tubing connected to each port, minimizing delay times and carryover. The on-board computer switches to the requested port at selected intervals allowing a small amount of sample to enter the cell. The analyzer is also equipped with a permeation tube, which under fixed temperature, and carrier gas flows, allows a known concentration of ammonia to enter the cell to be used as an internal standard for calibration purposes.

## Results and Discussion for Naturally Ventilated Buildings

The location of the analyzer is critical in regards to the concentrations being measured. Distance from the buildings as well as the distance above the buildings impact the resulting concentration measurements. As shown in Figure 1, there is a vertical gradient with regards to ammonia concentrations when measured between buildings in July as well as considerable variations within each height. Placing the analyzer 100 feet downwind of the buildings results in the variations shown in Figure 2. The analyzer was setup to measure ammonia concentrations at heights of 3 feet, 20 feet, and 40 feet. Data in this figure show that not only are there differences in ammonia concentration due to height, but also due to time of day. Concentrations were highest between 4:00 pm and 10:00 pm. The effect of horizontal distances on ammonia concentrations are shown in Figure 3. The sampling points were 6 feet above ground level and increasing distances up to 90 feet (all are downwind). The results as shown in this figure clearly indicate a rapid decline from 1000 ppbv ammonia at a distance of 3 feet to less than 50 ppbv at a distance of only 90 feet.

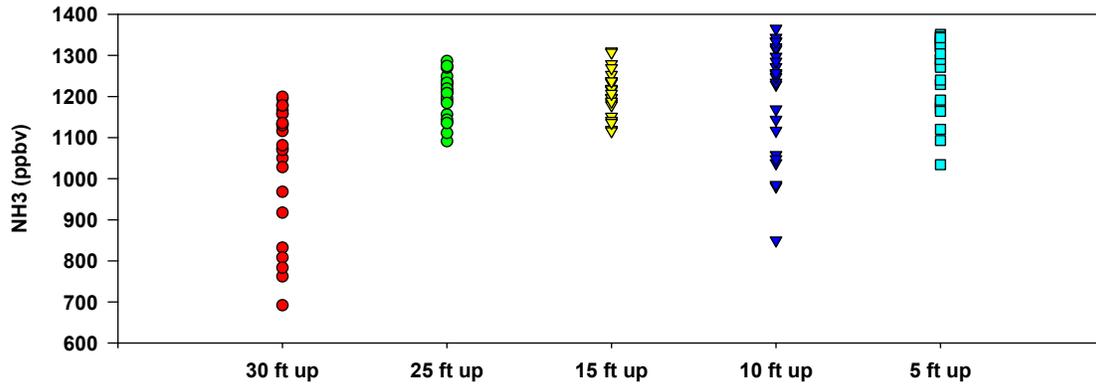


Figure 1. Variability of Ammonia Concentrations Due to Height

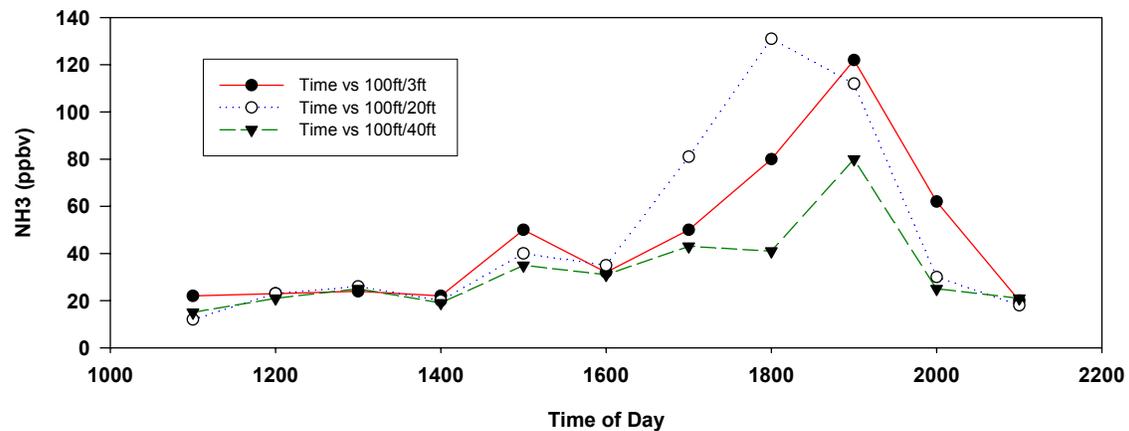


Figure 2. Affect of Height on Ammonia Concentrations 100 Feet Downwind

It is interesting that this rapid decline occurs over such a short distance. Since most property lines are further from the buildings than the 90 feet in this study one must question the validity of using the property line as the measuring point for many proposed and/or enforced regulatory standards. Now that the affects of distances have been examined, what happens when the wind changes direction? An example of the affects of changing wind directions are presented in Figure 4. The differences range from 1000 ppbv ammonia when the analyzer is downwind to less than 20 ppbv ammonia when the wind direction reverses and the analyzer ends up being upwind. Finally, what is the variability of ammonia concentrations if the analyzer is placed directly in front of a pit fan such that the pit fan's exhaust is being monitored directly? This is shown in Figure 5. Presented in this figure are concentrations over a 24 hour period in August, 2005 from an analyzer placed 20 feet from a pit fan that was running continuously at a constant velocity. There was no mechanical mixing of the deep pit contents at this time. This variability may have been due to the biological activity in the deep pit, but more likely was due to changes in animal activity. Are the pigs active and feeding, or are they resting? Disturbing the pigs prior to taking a measurement could lead to elevated ammonia concentrations that may not reflect average emission concentrations from the production facility.

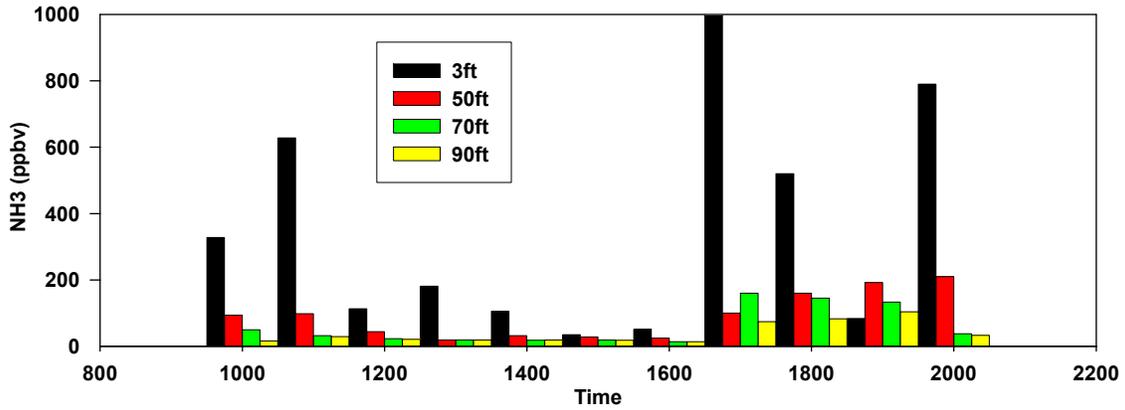


Figure 3. Affect of Distance Downwind on Ammonia Concentrations

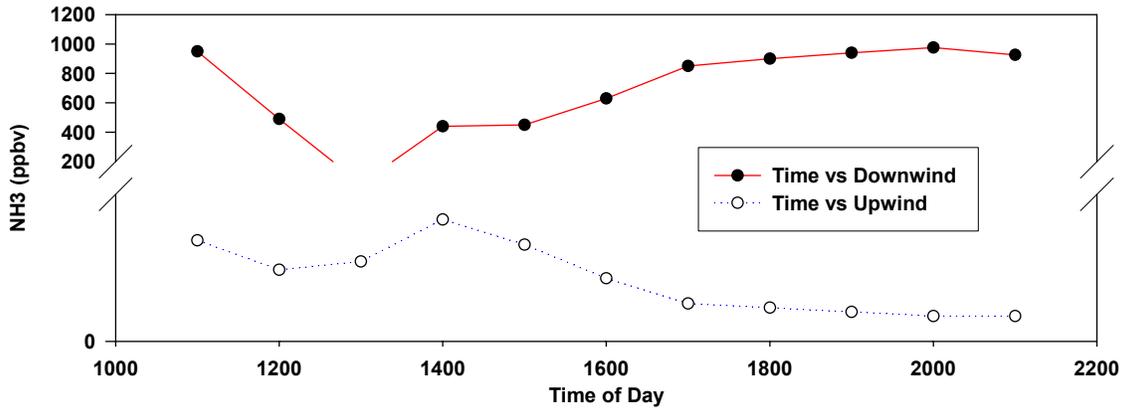


Figure4. Affects of Wind Direction on Ammonia Concentrations

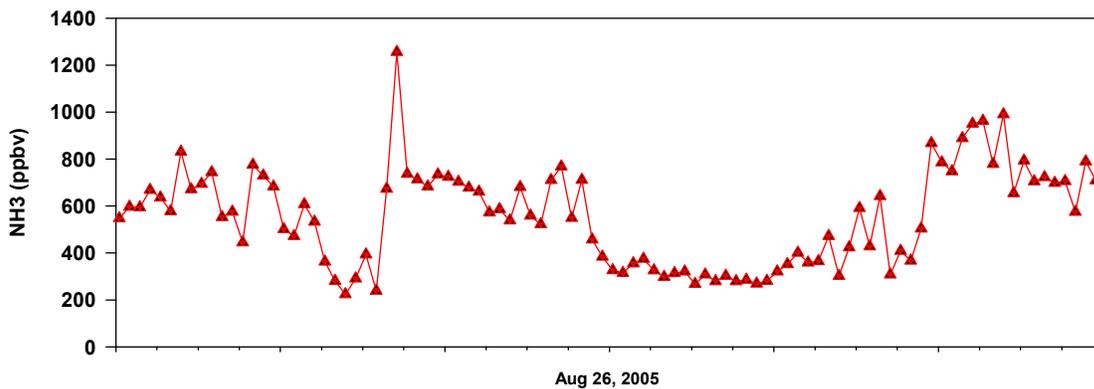


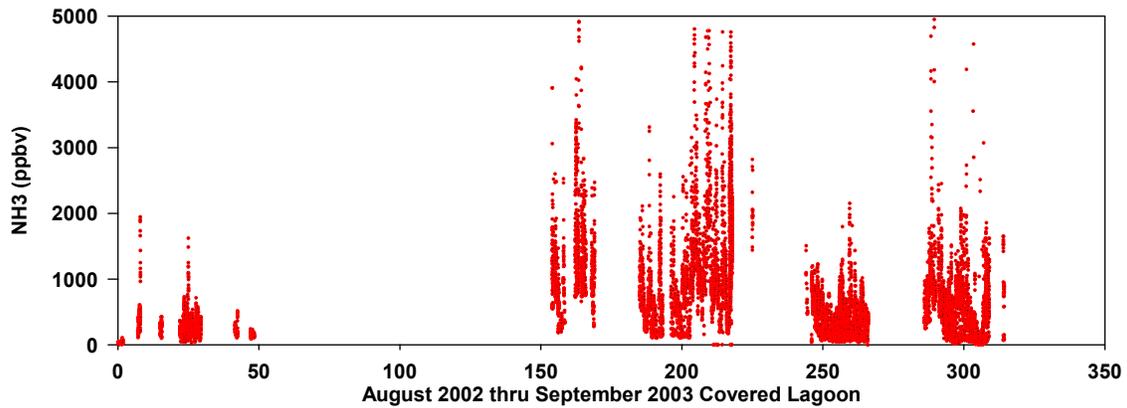
Figure 5. Variability of Ammonia Concentration in Front of a Pit Fan

### Results and Discussion for Lagoons

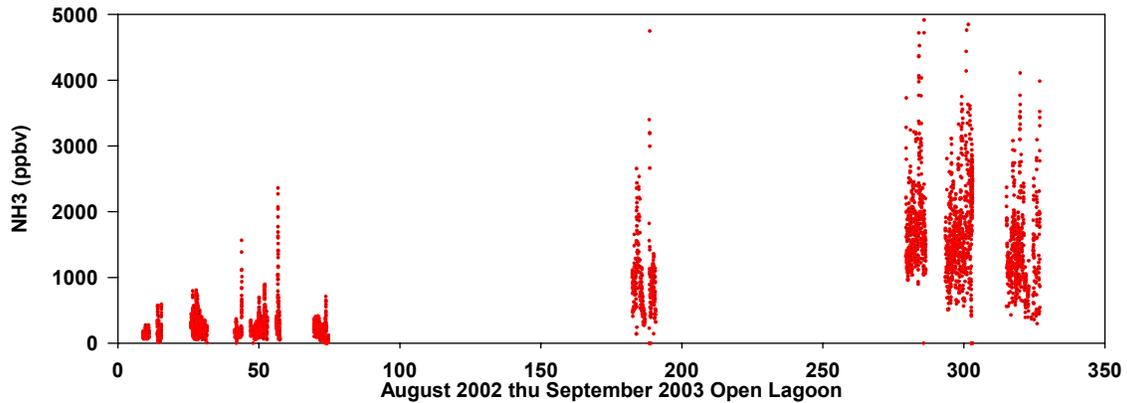
From the previous figures, one can see that choosing a sampling point that yields a representative concentration is a very difficult, if not impossible, task. Measuring concentrations of gases above and on the berms from lagoons presents the same problems. The following discussion is focused on two lagoons,

one covered and the other uncovered (open air), with emphasis on daily and seasonal variations as well as sampling height above the lagoon. Wind affects will also be discussed relative to a sampling location on one of the berms. Data for ammonia, hydrogen sulfide and methane are presented.

Figures 6 and 7 show the concentrations of ammonia two meters above the center of a covered and uncovered lagoon, respectively. The ammonia concentrations range from greater than 5000 ppbv to less than 100ppbv, a 50 fold difference. To rely on a single measurement as an indicator of the emissions from any swine production facility can only result in questionable conclusions.. Given this high range of concentrations the importance of long-term continuous monitoring as part of any research and/or regulatory investigation becomes quite evident.

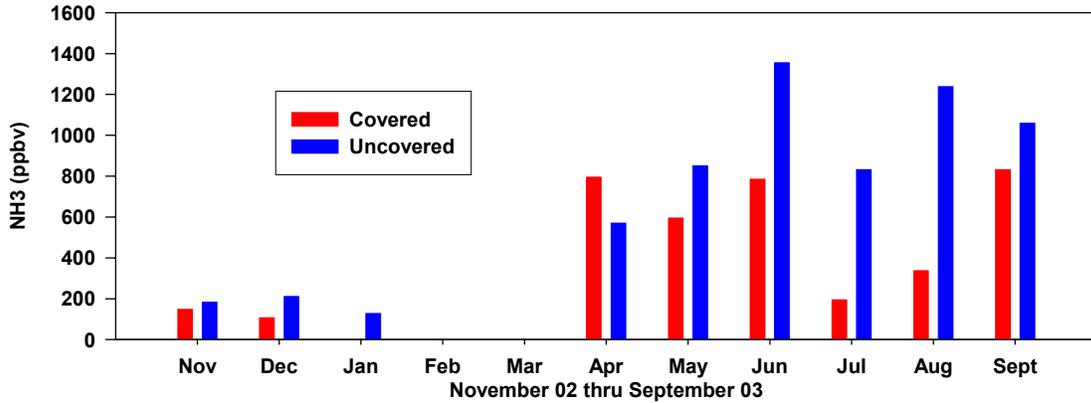


**Figure 6. Ammonia Concentrations from a Covered Lagoon**



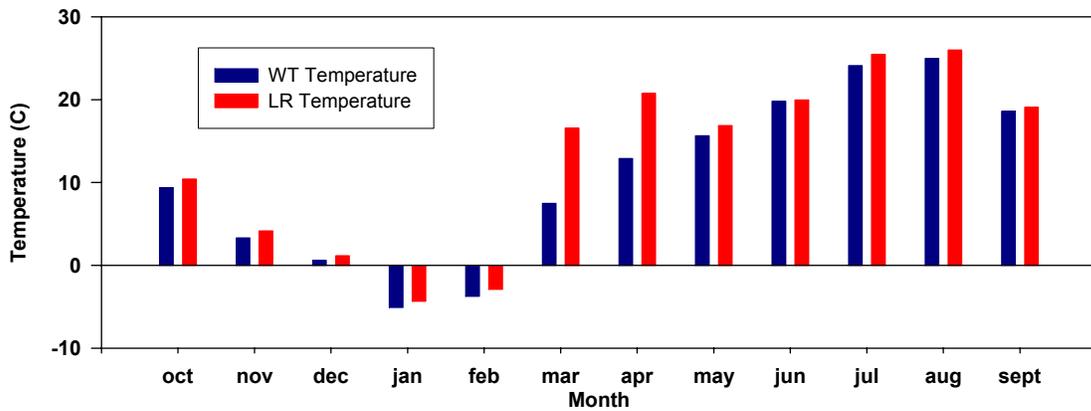
**Figure 7. Ammonia Concentrations from Uncovered Lagoon**

Due to analyzer problems and power outages the number of data points collected at the two locations were quite different. Also, there were long periods of time in which no data was collected. A direct comparison using all the data points yields averages that are significantly biased by the seasonality of the concentrations and other variables. As shown in Figure 7, the time of year has a major impact on the concentrations of ammonia over the lagoons.

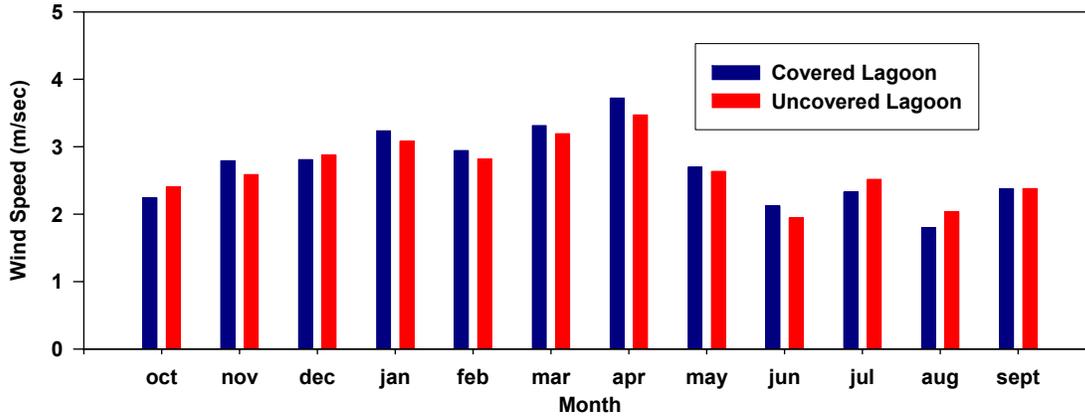


**Figure 8. Daily Ammonia Concentration Averages By Month**

In this figure, the daily average ammonia concentrations from paired days (as close as possible) averaged over each month are represented. The monthly ammonia concentration averages vary considerably, not only between months of the year, but between the covered and uncovered lagoon. The average ammonia concentrations range from less than 200 ppbv in the winter to more than 1400 ppbv during the summer. Overall, there are significant differences due to the cover with the exception of April. There does not seem to be an explanation for this reversal of concentrations in April. The volume of the liquid in the uncovered lagoon may have been supplemented with clean water as part of the management program resulting in a decrease in ammonia concentration. Concentrations for the covered and uncovered lagoons were 414 ppbv and 805 ppbv, respectively. This represents a 49% difference that is significant at a p-value of <0.01.



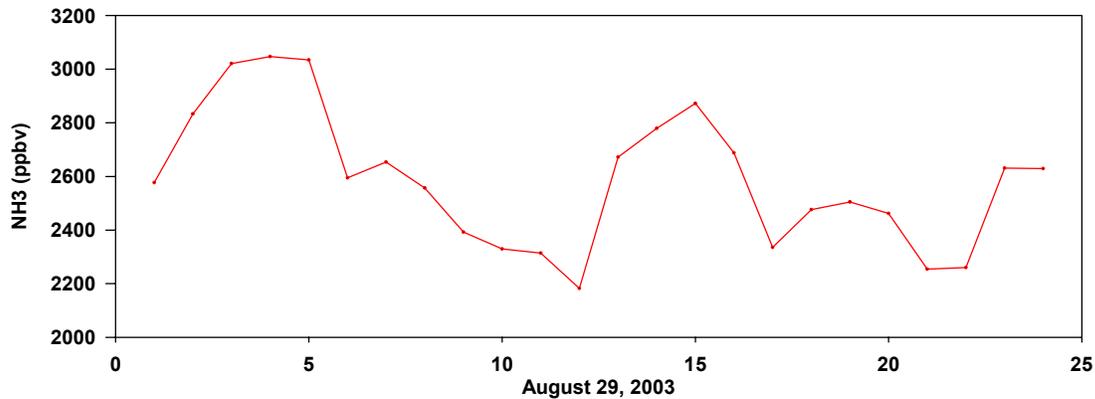
**Figure 9. Monthly Average Temperatures**



**Figure 10. Monthly Average Wind Speeds**

The two lagoons were separated by over 30 miles so it is possible that wind and temperature differences played a role in the concentration differences noted in Figure 8. However, based on the data presented in Figures 9 and 10, this is not the case.

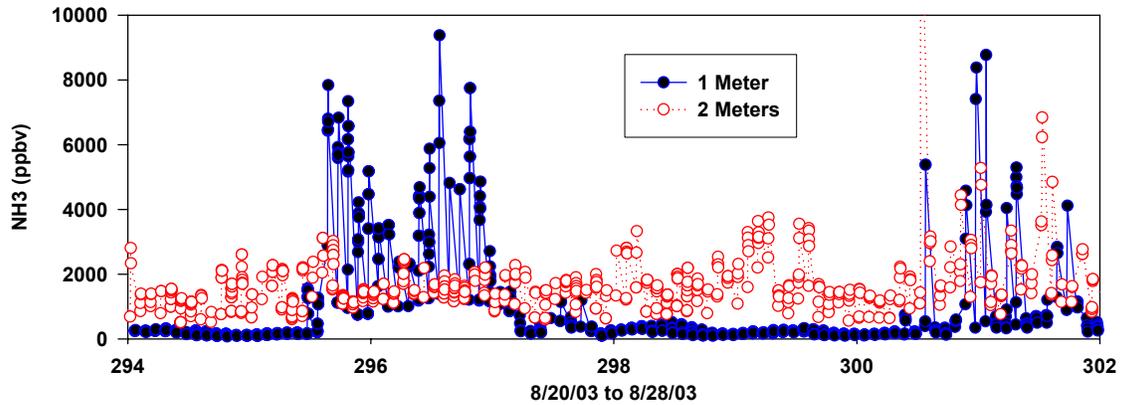
From the previous figures, the variations in ammonia concentrations due to season and covering are significant; however, the variations are not limited to long term changes. An example of this is shown in Figure 11. In this figure, ammonia concentrations are shown for a 24 hour period with analysis done every minute. The sampling location is two meters above the center of the uncovered lagoon. The ammonia concentrations varied from 2200 ppbv to over 3000 ppbv during the 24 hour time period. The values peaked early in the morning and again at 1500 hours. Wind speed was rather constant at 2.6 m/s, but the wind direction changed from 50 degrees to 130 degrees during the day. Data concerning the transferring of waste into the lagoon was not generated and this could be a key factor. Another factor that affects concentration values is the height of the sampler above the lagoon. For this study, two sampling lines were located above the center of the lagoon. One was 1 meter above the lagoon, and the other was 2 meters above the lagoon. Figure 12 shows the concentration data from these two locations. Note that the real sampling times are actually offset approximately 1 hour due the sampling schedule that included blanks and berm sampling over a 2 hour cycle that was discussed previously.



**Figure 11. 24 Hour Variation of Ammonia Over the Center of Open Lagoon**

The data presented in Figure 12 indicates that the concentrations at 1 meter can actually be less than the concentrations at 2 meters. Several factors such as low wind speed, affects of the berm, wind direction and

relative humidity could be used to explain this occurrence. The actual design of the sampling tower and attachment of the two sampling lines could also account for this fact.



**Figure 12. 1 Meter Verses 2 Meters Sampling Height**

The affect of wind direction concentrations of ammonia can be observed by looking at a plot of the south berm concentrations verses wind speed using a wind rose diagram. This is shown in Figure 13. Due to the construction and mounting of the floatation platform the wind directions are actually 80 degrees larger than those shown in the figure. For example a wind direction of 20 degrees is actually 100 degrees and a wind direction of 300 degrees is actually 20 degrees. Winds from the north cross over the lagoon before reaching the south berm and thus the concentrations of ammonia are higher. Winds coming from the south pass over open land before reaching the berm and thus reflect the lower background concentrations of ammonia. Three dimensional wind data were not taken during the study. Dissecting the wind direction into vertical and horizontal components might lead to some interesting conclusions regarding the transport of ammonia from the production facilities into the atmosphere.

The same variability that exists for ammonia also exists for hydrogen sulfide. The average paired daily concentrations of hydrogen sulfide were 8.8 ppbv and 25.0 ppbv for the covered and uncovered lagoon, respectively. This represents a 64% reduction in hydrogen sulfide concentrations that is significant at a p-value of <0.01. The hydrogen sulfide concentrations for the uncovered and covered lagoon are shown in Figures 14 and 15, respectively.

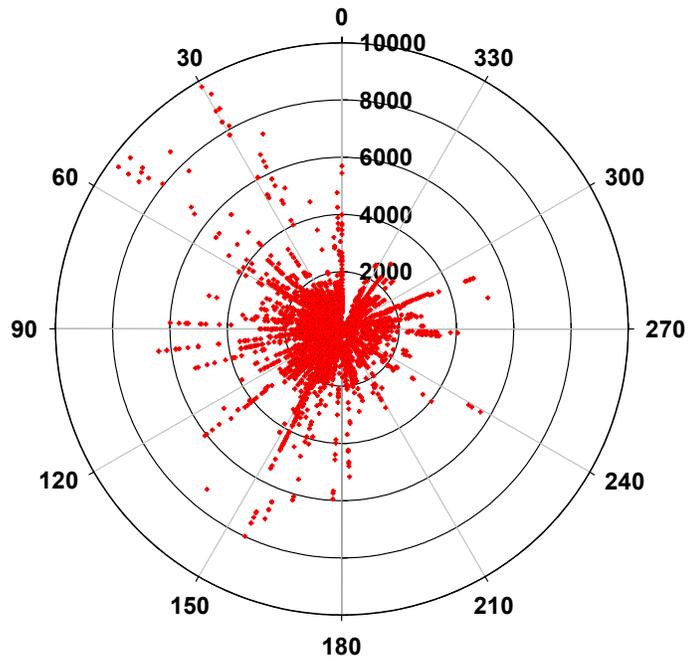


Figure 13. Ammonia Concentration Verses Wind Direction

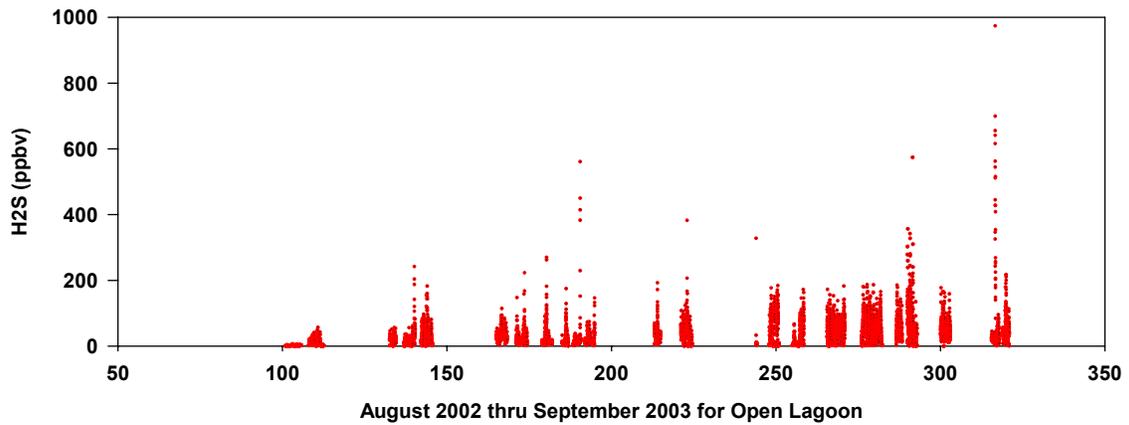
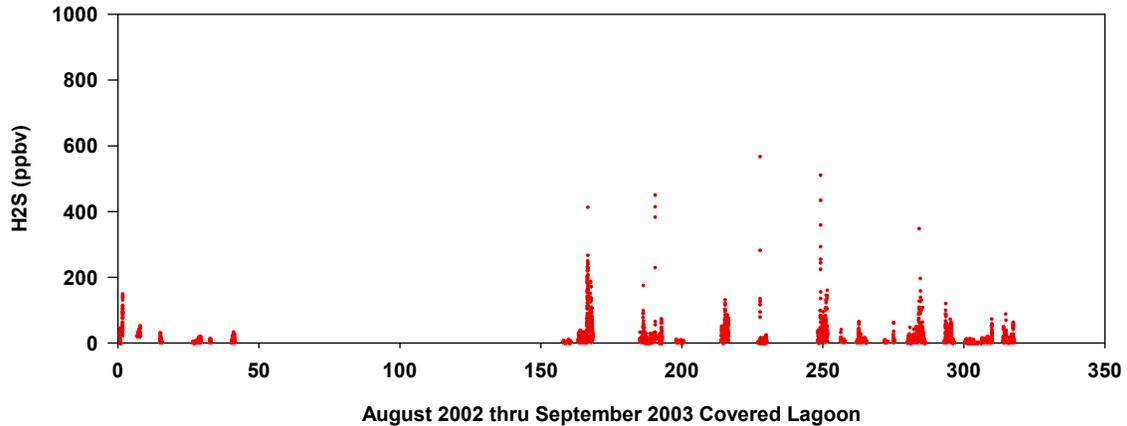
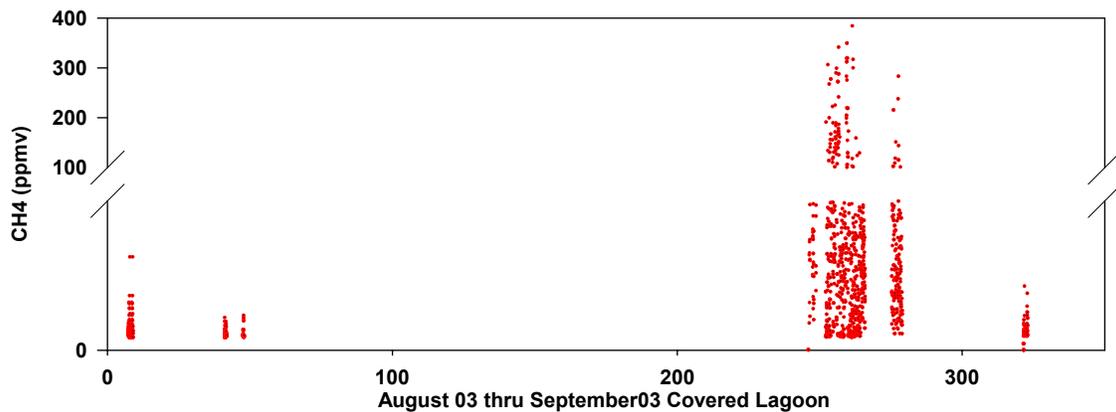


Figure 14. Hydrogen Sulfide Concentrations from Uncovered Lagoon



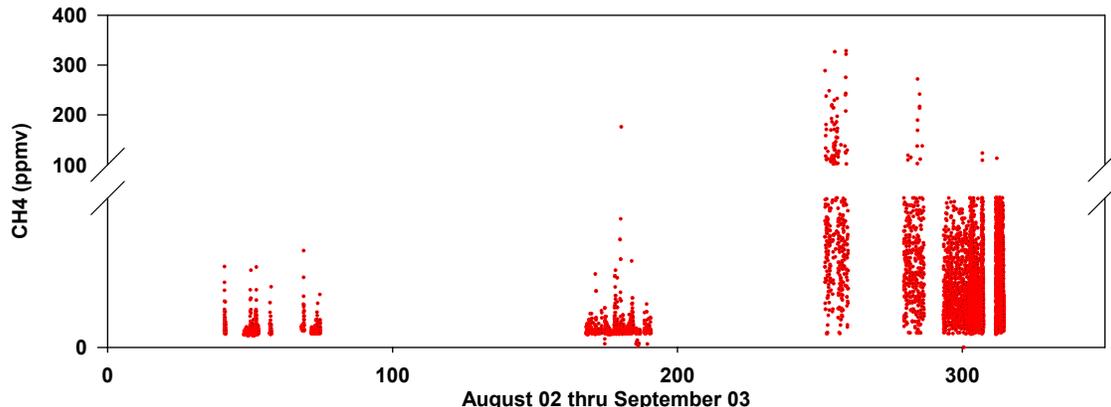
**Figure 15. Hydrogen Sulfide Concentrations Above a Covered Lagoon**



**Figure 16. Methane Concentrations Above a Covered Lagoon**

The variations in methane concentrations shown in Figures 16 and 17 were similar to those from ammonia and hydrogen sulfide with one exception. The methane concentrations were higher above the covered lagoon, than the open lagoon. This is possibly due to the increased conversion of the volatile organic compounds to methane as a result of the trapping effect of the cover and/or increased anaerobic conditions. Large bubbles that formed under the cover demonstrated the semi permeable nature of the cover. The paired daily average concentrations for methane were 13.2 ppbv and 24.1 ppbv for the uncovered and covered lagoons, respectively. This 49% increase was significant at a p-value <0.01.

The instrument used for the measurements was the TEI Model 55C Methane/Non-Methane Hydrocarbon Analyzer. Known concentrations of methane were presented to the instrument via the manifold system and a diluter connected to a 100 ppmv cylinder of methane. The results clearly indicated that the instrument was able to accurately measure methane concentrations. However, testing of the non-methane measurement features yielded disturbing results. When propane was presented to the instrument via the diluter and a 100ppm cylinder of propane the results were excellent. Propane is a typical hydrocarbon and these results were not surprising.



**Figure 17. Methane Concentrations Above an Uncovered Lagoon**

However, using this instrument for the measurement of other organic compounds associated with lagoon emissions such as acetic acid, butyric acid, propionic acid, and p-cresol may not yield satisfactory results. No response was seen when placing these compounds at high ppmv concentrations directly in front of the inlet for the Model 55C. No peaks were detected, but a steady rise in the background signal was observed. These compounds are not classified strictly as hydrocarbons because of the presence of other side groups such as COOH (acids), OH (phenol), and SH (mercaptan). These compounds are more polar than the traditional hydrocarbons and are likely to adsorb to the column packing material and not elute in a sharp band during the flow reversal process. Acids, aldehydes, ketones, etc., are important constituents of the volatile organics associated with emissions from lagoons, and it is incorrect to assume that the Model 55C will measure them. Even with its limitations, the Model 55C worked well for measuring methane concentrations.

### Summary

The measurement of gases from swine confinement operations is not an easy task. Any attempts to apply the methodology associated with the measurement of gases emitted via a smoke stack will result in failure due to the large number of variables associated with the confinement operation. The variability of measured ammonia, hydrogen sulfide, and methane concentrations that have been described in this paper are due to time of day, time of year, distance, and type of storage. Other variables such as diet, animal activity, building type, and ventilation system are currently under investigation by this author and many others. The determination of emission factors and ultimately the pounds of gases emitted from animal confinements on a per day or per year basis will require the concentration data discussed in this paper in order to reach a scientifically sound conclusion.

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## Denitrification and Nitrous Oxide Fluxes from Frozen, Manure-Amended Soil

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### Abstract

Intensive studies of nitrous oxide (N<sub>2</sub>O) fluxes indicate highly variable but significant responses to synthetic fertilizers, composts, and fresh animal manures. Effects of fertilizers on N<sub>2</sub>O emissions often vary with fertilizer form, soil characteristics, timing, and rate of application. In the upper Midwest, manures are often applied in autumn with the assumption that denitrification is negligible over-winter when soils are frozen. Little data, however, supports this assumption. Soils collected from crops fields in production were assayed to determine effects of dehydrated manure application on N<sub>2</sub>O emissions in winter. Intact cores were incubated 17 days later at field temperature (-2 °C) in the laboratory and N<sub>2</sub>O flux and denitrification measured. Nitrous oxide emissions and denitrification occurred in both fertilized and unfertilized soil. Autumn application of dehydrated manure stimulated biological denitrification, compared with un-amended soil, even at -2 °C.

### Introduction

Organic agriculture currently represents ½ million hectares of U.S. cropland and interest in conversion from synthetic to organic-based fertilizer amendments is expected to rise with the cost of fossil-fuel-based fertilizers. Agricultural fertilization worldwide contributes more anthropogenic nitrous oxide (N<sub>2</sub>O) to the atmosphere than any other single known source [Bouwmann, 1990]. Nitrous oxide is photochemically and radiatively reactive and is 296 times more effective per molecule than CO<sub>2</sub> as a greenhouse gas. A greater understanding of factors controlling N<sub>2</sub>O flux to the atmosphere for production agriculture soils in the field is needed. Data are lacking that quantify how organic fertilizer products contribute to N<sub>2</sub>O emissions in an agricultural production environment and under freezing field conditions.

Agricultural fertilization influence N<sub>2</sub>O fluxes, but the response varies with soil properties, fertilizer type, application timing, and rate [Breitenbeck, 1986]. Intensive laboratory studies indicate significant response in N<sub>2</sub>O flux when soils are amended with synthetic fertilizers, composts, and fresh animal manures. Rarely, however, are fertilization effects tested in a production agriculture environment, especially in the Red River Valley (RRV) ecoregion, where soils tend to remain frozen for five months of the year. Consequently, amounts of N<sub>2</sub>O and N<sub>2</sub> produced from denitrification in crop fields at sub-zero temperatures are largely unknown. We aimed to determine if organic fertilizer application would influence N<sub>2</sub>O flux and denitrification in frozen soils for crop fields in the RRV. We address this question on working farms for the purpose of addressing the need for data representative of production-oriented agriculture.

### Methods

The RRV region, located in the upper Midwestern U.S., is one of the most productive agricultural regions in the world. Soils are lacustrine remnants of the south-easternmost extent of glacial Lake Agassiz, and climate is characterized by long, cold winters (~5 months) and mild to warm, abbreviated summers. In general, RRV soils remain frozen the entire winter, compared to other regions of the Midwest, where freeze-thaw cycles are common. Since soils remain frozen during winter, organic soil amendments are typically applied to crop fields in late-autumn just prior to freeze-up. Conventional, synthetic fertilizers, such as urea, are applied more often in the spring just prior to planting.

To address the question of how manure might alter nitrous oxide flux during winter after autumn fertilization, two 80-acre, homogenously managed fields were selected: one managed conventionally for >50 yrs and one managed organically for >5 yrs. Both soils share similar soil properties, climate, and crop rotations and are located <1 km from each other in Clay County, Minnesota. Between field treatments an uncultivated control site with no history of fertilization was identified. Both were last planted with soybeans and were last fertilized 1.5 yr and 2 yr prior to this experiment, respectively, for conventional and

organic fields. On November 20, 2004, the organic field was amended with 67.5 kg N ha<sup>-1</sup> of a pelletized, dehydrated manure product (Creekwood Farms, Lake Mills, WI) and homogenized into the top 5 cm of soil. Soil temperatures below freezing were recorded one week following manure application and soil cores (0-15 cm depth) collected ten days after soil freezing. Five intact cores were extracted by hand using a 5-cm diameter hammer auger at random points within 8-ha field plots from both conventional and organic fields and immediately placed in cold storage (8 °C). A core from the uncultivated site was also collected. Cores remained below freezing for the duration of the experiment. Soil temperature (0-15 cm) was -2 °C at the time of core collection.

We utilized a laboratory-grade, temperature-controlled freezer (Scientemp Corp.) for the purpose of simulating field temperatures in the laboratory. The day after field sampling, intact cores were extracted from collection tubes and each was carefully placed inside a 1-L glass Mason jar equipped with a septum for gas sampling and allowed to equilibrate at -2 °C for four hours. Included were two empty jars, or blanks, which were treated as samples throughout the experiment. Prior to sampling, covers were removed to introduce fresh atmosphere into the headspace while remaining in the freezer. Jars were capped, and a 15-ml headspace sample was drawn and replaced with an equal volume of N<sub>2</sub> to maintain constant pressure. The headspace was sampled every 0.25 hr to complete a 1-hr time course. Aliquots were immediately injected into 15-ml exetainers (Labco Unlimited) and analyzed for N<sub>2</sub>O using a Varian Model 3800 Gas Chromatograph and Combi-Pal auto-sampler. In this system, sample is auto-injected into a 1-mL sample loop, then loaded onto columns and routed through <sup>63</sup>Ni electron-capture detector (ultra-pure 95% Argon/5% CH<sub>4</sub> carrier gas) detector. Following the incubations, bulk density and percent water filled pore space (WFPS) were determined for each core. The mass of N<sub>2</sub>O-N produced was calculated based on a linear time course, following correction for headspace dilution with N<sub>2</sub>.

## Results and Discussion

Average ( $\pm$  standard deviation) N<sub>2</sub>O emission for manure-amended soil at -2° C on a mass basis (dw=dry weight) was 0.68 ng g dw<sup>-1</sup>hr<sup>-1</sup> N<sub>2</sub>O-N ( $\pm$ 0.67), while average emission for soil that was not recently amended was 0.23 ng g dw<sup>-1</sup>hr<sup>-1</sup> N<sub>2</sub>O-N ( $\pm$ 0.18). Nitrous oxide emission for soil that was uncultivated with no history of fertilization was 0.10 ng g dw<sup>-1</sup> hr<sup>-1</sup> N<sub>2</sub>O-N. On an area basis (assuming N<sub>2</sub>O measured from 0 to 15 cm is emitted at the surface), manure-amended soil emitted 114.40  $\mu$ g m<sup>-2</sup> hr<sup>-1</sup>, compared to 37.65  $\mu$ g m<sup>-2</sup> hr<sup>-1</sup> for un-amended soil.

Denitrification was measured on the same soil cores the following day also at -2° C using the acetylene inhibition technique [Paul and Beauchamp, 1989], and denitrified N (N<sub>2</sub>O+N<sub>2</sub>) measured following the protocol described above. Total denitrification (N<sub>2</sub>O+N<sub>2</sub>) on a mass basis for manure-amended soil was 1.03 ng g dw<sup>-1</sup>hr<sup>-1</sup> ( $\pm$  0.75), while average denitrification for soil that was not recently amended was 0.27 ng g dw<sup>-1</sup> hr<sup>-1</sup> ( $\pm$  0.23). There was no evidence of denitrification in the uncultivated soil core. On an area basis, manure-amended soil emitted 173.01  $\mu$ g m<sup>-2</sup> hr<sup>-1</sup>, compared to 43.67  $\mu$ g m<sup>-2</sup> hr<sup>-1</sup> for un-amended soil. Water-filled pore space was similar for both organic and conventional field soils, with average WFPS of 41.4 ( $\pm$  5.0) and 39.1 ( $\pm$  5.8), respectively.

Results indicate biological denitrification at temperatures below freezing for both treatments. Moreover, denitrification was greater for manure-amended soil, compared with soil that was not recently fertilized. These rates were determined 17 days after manure application. If they are representative of winter denitrified-N losses and if these were consistent over the 5 months of winter, then 6.23 kg N ha<sup>-1</sup> of the added N (9% of the total N input) would be denitrified from frozen, manure-amended soil. Conversely, frozen soil that was not recently amended, either organically or synthetically, would denitrify 1.57 kg N ha<sup>-1</sup> over-winter. These data suggest denitrification rates increase following application of dehydrated manure in late-fall and point to the importance of microbial activity in frozen soils [Rover, 1998]. Moreover, the data suggest that denitrification measurements are essential for understanding the fate of manure N applied to agricultural soils [Calderon, et al., 2005].

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## **Ammonia Flux and Dry Deposition Velocity Estimated from an Intensively Managed Animal Agricultural Facility in North Carolina**

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### **Introduction**

Recent studies in Europe and the United States have reported increasing atmospheric concentration levels of ammonia ( $\text{NH}_3$ ) and ammonium ( $\text{NH}_4^+$ ), especially in regions of concentrated animal feeding operations (Aneja et al., 1998). These higher concentration levels have shown to be directly related to the rapid growth of intensively managed agriculture. At present, ammonia emission and subsequent wet and dry deposition are a significant waste management problem facing animal husbandry and agriculture. In NC, estimates reveal that the swine population contributes approximately 46% of the  $\text{NH}_3$ -N emissions. This estimate shows a direct link to the recent growth of the swine population in NC. NC is ranked second in the nation for swine production with approximately 10 million hogs in about 2500 hog farms located in eastern North Carolina (Aneja et al., 2000). Areas of scattered local sources (small, rural family farm operations) contribute a wide range of  $\text{NH}_3$  emission and dry deposition. Likewise, many environmental problems result from atmospheric ammonia deposition, e.g., particulate matter formation, aquatic and terrestrial eutrophication, and odor emanation.

An experimental study was conducted on the emission and dry deposition fluxes of ammonia under different meteorological conditions, using a micrometeorological technique (micrometeorological gradient and modified Bowen-ratio methods in conjunction with the Monin-Obukhov similarity theory) over natural surfaces in North Carolina where intensively managed agriculture/animal farms are located. Diurnal and seasonal variations of ammonia flux and dry deposition velocity were investigated under a wind range of wind and atmospheric stability conditions yielding hourly variation of  $\text{NH}_3$  flux and deposition velocity.

The primary objectives of this research are: (1) to measure vertical fluxes of ammonia and related dry deposition velocities from near-surface concentration gradient measurements over natural surfaces in eastern North Carolina downwind of a source; (2) investigate and evaluate the variability of ammonia flux and related dry deposition velocity on a specified natural surface (i.e. grass) with respect to the time of the day, season, and meteorological factors; (3) obtain empirical relations for dry deposition velocity of ammonia; and (4) quantify the fate of atmospherically deposited nitrogen during summer season in North Carolina terrestrial ecosystems (water and land). This particular area of emphasis is chosen because of the obvious lack of data and the expressed need for a better understanding of ammonia flux and dry deposition velocity in eastern North Carolina, where ammonia sources have increased very rapidly in recent years (DAQ, 1997).

### **Methods and Materials**

A series of seasonal field experiments of ammonia concentrations were conducted at the North Carolina State University Air Quality Educational Unit (USDA-ARS, Raleigh, NC) during Fall 2001, Winter, Spring and Summer 2002. This site is a relatively flat, uniform and smooth site with grass or short vegetation, which is located near a small swine production facility. Two chemiluminescent analyzers, TEI Model 17C (TEI, 2004), were utilized in conjunction with a solenoid for each analyzer to alternate measurements between two elevations (2m and 6m). Hourly-averaged measurements of wind speed, wind direction and temperature were also measured at the same two heights (2m and 6m) using a 7m walk-up tower in the horizontally-homogeneous atmospheric surface layer. The general gradient method is used for estimating the vertical flux and the deposition velocity of ammonia (Businger, 1986; Hicks, 1986) (see Phillips et al., 2002; Phillips et al., 2004)).

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## Results

### Seasonal NH<sub>3</sub> Concentrations: Comparison with Meteorological Conditions

The seasonal NH<sub>3</sub> concentrations reveal that the largest NH<sub>3</sub> concentrations were during the fall measurement campaign, while the highest maximum NH<sub>3</sub> concentration was observed in summer at 2m (41.71 μg m<sup>-3</sup>). The larger fall average concentrations are due to the lagoon irrigation practices used and relatively warm temperatures (25.11 ± 3.09 °C), whereas, lagoon irrigation was not applied during the other seasonal measurement campaigns. Based on the animal waste management plan used by the Dairy Educational Unit at the measurement field site, lagoon liquid irrigation was applied to a field southeast of the air quality tower where a small grain overseed (wheat) was cultivated. A relatively small difference of 1.0-0.65 μg m<sup>-3</sup> exists between spring and summer average NH<sub>3</sub> concentrations during day and nighttime at the two heights (2m and 6m). The winter season had the lowest overall concentrations collected during each seasonal campaign with averages of 1.73 ± 2.00 μg m<sup>-3</sup> at 2m during daytime; and 1.37 ± 1.50 μg m<sup>-3</sup> during nighttime.

During this measurement period frequent changes in the sign of NH<sub>3</sub> concentrations gradient and, hence, flux in the afternoon and night were observed, which are the result of the impact of multiple meteorological variables: temperature variation, wind speed variation from 0.90 to 5.0 m s<sup>-1</sup>, wind direction variation from north to northeast, to east, and to southeast, and relative humidity variation between 98 to 70%. Likewise, a diurnal relationship between temperature and NH<sub>3</sub> concentrations became apparent reflecting typical near-surface diurnal inversion patterns in which as temperature increases (decreases), NH<sub>3</sub> concentrations also increases (decreases). During the fall measurement period, not only were the northeast and southeast wind directions dominant during daytime but also, higher concentrations of NH<sub>3</sub> occurred in the eastern wind sector due to the transport of NH<sub>3</sub> from a swine lagoon located east of the air quality tower. The correlation between greater frequencies of NH<sub>3</sub> concentrations with winds southwest and west of the tower were observed during the winter, spring, and summer suggesting the effects of environmental conditions (grazed fields as a result of dairy cows located south and southwest of the measurement site), while west wind directions show the effects of horse and chicken farms west of the tower.

### Seasonal NH<sub>3</sub> Deposition Fluxes

Seasonal averages and ranges of negative deposition fluxes calculated by the gradient method are presented, which occurred when hourly concentration gradients ( $\partial \bar{c} / \partial z$ ) were positive and detectable (Phillips et al., 2004). Since, the level detection limit (LDL) defined by the TEI Model 17C manual is 1 ppb (~0.7 μg m<sup>-3</sup>); measurements collected and gradients calculated using ammonia concentrations below this LDL value have been excluded from estimated NH<sub>3</sub> seasonal fluxes and deposition velocities. Table 1 presents the statistical analysis (average and standard deviation) of seasonal NH<sub>3</sub> deposition fluxes (after limiting criteria based on  $v_{max}$  was applied to estimates of deposition velocities, see Phillips et al., 2004) based on hourly-averaged fluxes, where N equals the number of sampling days and n equals the number of sampling hours, and negative flux implies downward flux or deposition. The direction and magnitude of flux change hourly, diurnally and seasonally, suggesting the effect of environmental, meteorological, and stability conditions, as well as irrigation applications (e.g. Fall Season). Nevertheless, consistently throughout each season, deposition mostly occurred during the late afternoon, evening, and the early morning hours. The results of the seasonal statistical analysis show smallest average negative fluxes in Winter 2002, with hourly-averaged deposition fluxes ranging from -0.14 to ~0 μg-NH<sub>3</sub> m<sup>-2</sup> s<sup>-1</sup> and an overall average of -0.02 ± 0.03 μg-NH<sub>3</sub> m<sup>-2</sup> s<sup>-1</sup>. Spring and summer season-averaged deposition fluxes are about the same (-0.11 ± 0.15 μg-NH<sub>3</sub> m<sup>-2</sup> s<sup>-1</sup>), while the average of hourly fall fluxes of NH<sub>3</sub> is -0.14 ± 0.19 μg-NH<sub>3</sub> m<sup>-2</sup> s<sup>-1</sup>. The minimum (maximum deposition) fluxes in these seasons ranged from -1.16 to -0.90 μg-NH<sub>3</sub> m<sup>-2</sup> s<sup>-1</sup>.

**Table 1. Average seasonal NH<sub>3</sub> deposition fluxes and estimated deposition velocity where N=number of sampling days; n=number of measurements].**

Season	NH <sub>3</sub> Deposition Fluxes* ( $\mu\text{g-NH}_3 \text{ m}^{-2} \text{ s}^{-1}$ )	NH <sub>3</sub> Deposition Velocity ( $\text{cm s}^{-1}$ )	
	Daily	Day	Night
Summer	-0.11 ( $\pm 0.14$ ) N=23, n=135	3.94 ( $\pm 2.79$ ) N=16, n=85	0.76 ( $\pm 1.69$ ) N=18, n=50
Spring	-0.11 ( $\pm 0.15$ ) N=21, n=88	2.85 ( $\pm 2.01$ ) N=11, n=37	0.62 ( $\pm 1.04$ ) N=15, n=51
Fall	-0.14 ( $\pm 0.19$ ) N=11, n=57	2.82 ( $\pm 1.98$ ) N=8, n=34	0.07 ( $\pm 0.17$ ) N=10, n=23
Winter	-0.02 ( $\pm 0.03$ ) N=15, n=71	2.41 ( $\pm 1.92$ ) N=9, n=30	0.19 ( $\pm 0.27$ ) N=12, n=41

\* Negative flux is depositing downwards.

### Seasonal Ammonia Deposition Velocities

Estimating only the aerodynamic resistance,  $r_a$ , the maximum possible deposition velocity ( $v_{max} = r_a^{-1}$ ) has been used to validate estimated deposition velocities with measured meteorological conditions, where  $-v_{max}$  is the maximum possible emission rate (Wyers and Erisman, 1998). However, this is based on the questionable assumption that the resistance for the transfer of ammonia is identical to that of momentum. Therefore, we applied a relaxed criterion as an alternative to the assumption of  $v_d < v_{max}$ , where we consider from the bulk transfer method and considerable experimental evidence (Arya, 1977; 2001), of heat and mass being transferred more efficiently than momentum under unstable and convective conditions. Our relaxed criterion for acceptable values of  $v_d$  is that  $v_d \leq 2 v_{max}$  under unstable and convective conditions and  $v_d \leq v_{max}$  under stable conditions. Seasonal assessments of  $v_{max}$  for all hourly samples of deposition flux show seasonal variations in the  $v_{max}$  values due to seasonal influences on wind speed and stability.

Estimates of  $v_d$  that did not meet the above relaxed criterion were considered to have large uncertainty in the estimated  $v_d$ s, and excluded from any further analysis. Seasonal averages shown in Table 2 represent an average of all hourly NH<sub>3</sub> deposition velocities over the number of observation days and are further divided into average daytime and nighttime deposition velocities. They meet our relaxed criterion based on  $v_{max}$  and are considered to be more reliable than those excluded.

Summer measurements yielded the largest average daytime deposition velocity of  $3.94 \pm 2.79 \text{ cm s}^{-1}$  while winter season gave the lowest  $v_d = 2.41 \pm 1.92 \text{ cm s}^{-1}$ . The average values for daytime  $v_d$  during spring and fall seasons are about the same ( $2.8 \pm 2.0 \text{ cm s}^{-1}$ ). Conversely, nighttime estimates of  $v_d$  are much smaller, especially during fall ( $0.07 \pm 0.17 \text{ cm s}^{-1}$ ) and winter ( $0.19 \pm 0.27 \text{ cm s}^{-1}$ ) seasons. These daytime and nighttime differences are largely due to different stability conditions. The highest average deposition velocities were generally observed during unstable and near-neutral conditions and lowest values during very stable conditions (Table 2).

**Table 2. Average seasonal Stability Classification for estimated deposition velocity [where N=number of sampling days; n=number of measurements].**

	Stability Classification for Deposition (cm s <sup>-1</sup> )			
	Summer	Spring	Fall	Winter
<b>Unstable (Ri&lt;0)</b>	4.42 (±2.65) N=12, n=72	3.03 (±1.66) N=6, n=23	3.58 (±1.59) N=5, n=26	3.00 (±1.87) N=8, n=22
<b>Moderately Stable (0≤Ri&lt;0.14)</b>	2.37 (±2.32) N=12, n=22	2.14 (±1.95) N=9, n=30	0.88 (±0.60) N=3, n=4	0.50 (±0.54) N=12, n=24
<b>Very Stable (Ri≥0.14)</b>	0.07 (±0.11) N=17, n=41	0.09 (±0.10) N=13, n=35	0.03 (±0.03) N=10, n=27	0.07 (±0.09) N=10, n=25

### Turbulence and Stability Effects

Micrometeorological variables including wind speed at 10m or friction velocity, atmospheric stability, surface heat flux and moisture flux affect turbulence transfer through the surface layer. Friction velocity, a measure of mean wind shear and shear-generated turbulence near the surface in both the canopy layer and above the canopy homogeneous surface layer, is noted to be well correlated with dry deposition velocity and one of the most important variables (Arya, 1999). A regression analysis reveals that a strong relationship exists between friction velocity and estimated deposition velocities. The results of this procedure are based on the combination of seasonal data stratified with respect to stability (unstable, moderately stable and very stable categories) with corresponding regression equation and R<sup>2</sup> values. All seasons display strong correlations with R<sup>2</sup> values of 0.74, 0.54, and 0.86 based on power regression curves for unstable, moderately stable, and very stable conditions, respectively.

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## Abating Ammonia Emission from Dairy Barns Through Feed, Herd and Bedding Management

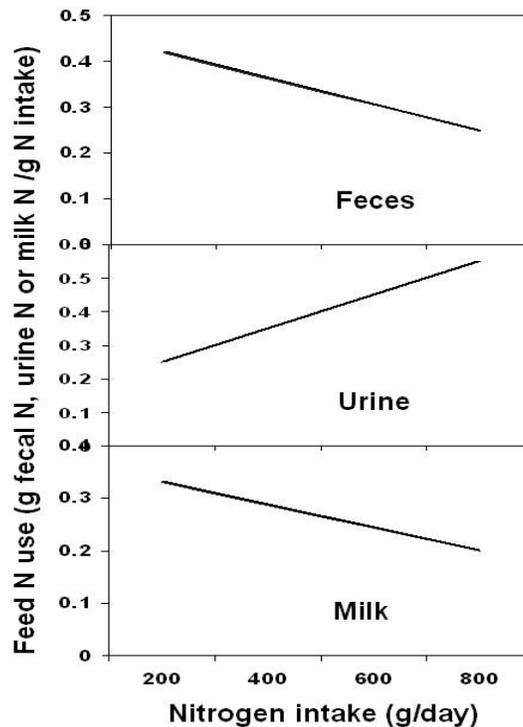
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Dairy farms are thought to emit large amounts of ammonia and therefore contribute to nitrogen (N) fertilization of natural ecosystems and provide precursors for particulates that adversely affect visibility and human health. The 2003 NRC report “Air Emissions from Animal Agriculture” (NRC, 2003) made an urgent call for process-based research that assists producers and regulatory agencies in developing strategies that improve air quality. Only about one-fourth of the crude protein (N) fed to a dairy cow is converted into milk, the remainder is excreted in manure. About one-half of the nitrogen in manure is excreted in urine, which can be converted quickly to ammonia and lost as gas. Ammonia production and loss can occur rapidly in the barn and continue through manure storage and land application. Some losses of N as ammonia gas are inevitable, but ammonia N loss can be reduced, and the fertilizer N value of manure can be maintained, through good management.

**Figure. 1. General relationships between N fed to dairy cows and N in milk, feces and urine (adapted from Castillo et al., 2000)**



Only approximately 20 to 35% of the N (crude protein) fed to dairy cows is converted into milk (Fig.1). When recommended protein levels are fed to dairy cows, manure N is excreted approximately equally in urine and feces. Feeding N to dairy cows in excess of their requirements dramatically increases urine N

excretions. About three-fourths of the N in urine is in the form of urea. Urease enzymes, which are present in feces and soil, rapidly convert urea to ammonium. Ammonium can be transformed quickly into ammonia gas. Feces contain little or no urea. For this reason urinary N is much more vulnerable to ammonia volatilization than is fecal N.

### Some Approaches to Reduce Ammonia Losses from Dairy Barns

#### Put More Feed N into Milk, Not Manure

One of the most reliable approaches to reducing ammonia emissions per unit of milk produced is to increase level of milk production per cow. On Wisconsin dairy farms, the efficiency by which the crude protein (CP) contained in feed is converted into milk (i.e., feed N use efficiency) varies according to production practices. Milk production and feed N use efficiency are highest on farms that use total mixed rations (TMR), that balance rations four times per year, and milk thrice daily (Table 1). These practices put more feed nutrients into product (milk), and less into manure.

**Table 1.** Impact of feed management and milking frequency on milk production, and feed N use efficiencies (FNUE) on 54 Wisconsin dairy farms (Powell et al., 2006)

Practice	Practice use	Milk	
		Production lbs/cow/d	FNUE %
Use TMR	Yes	74a <sup>†</sup>	27a <sup>†</sup>
	No	57b	24b
Balance rations 4 times/year	Yes	67a	26a
	No	54b	21b
Milk thrice daily	Yes	88a	33a
	No	63b	25b
Use Posilac®	Yes	82a	29a
	No	61b	25b

<sup>†</sup>within a practice, means followed by different letters differ significantly ( $P < 0.05$ ).

Over the long term, continued genetic selection of cows for high milk production potential will be a very effective means of reducing ammonia N emissions per unit of milk produced. Furthermore, increasing production per animal would decrease the number of cows needed to meet the market demand for milk.

#### Feed the Correct Amount and Type of Crude Protein to Dairy Cows

Diet formulation to eliminate excess CP usually reduces feed cost, and it is one of the most effective tools for reducing emission into the atmosphere of N containing compounds from dairy farms. Nitrogen excretion by dairy cows via urine, and therefore the amount of manure N susceptible to ammonia loss, is highly influenced by the amount and type of protein fed. As the amount of protein in feed exceeds what is required, relatively less N goes into milk and more goes directly into urine (Fig. 1).

Significant reductions in urine production can be obtained by reducing dietary protein levels. For example, if 17.5% dietary protein currently represents an industry average for lactating cows, carefully formulated diets containing 16.0-16.2% crude protein, which meets requirements for the lactating cow and still provides a reasonable margin of safety, would reduce N excretion in urine by about 20% (Broderick, 2003).

Various lactation trials have been conducted whereby Holstein cows were fed different levels of CP, fiber, corn silage, alfalfa silage, alfalfa haylage, and tannin-containing forages [alfalfa, birdsfoot trefoil low tannin (BF-T-Low) and birdsfoot trefoil high tannin (BF-T-High)]. Thus far, most tested diets have had small, if any impact on milk production, or milk protein and fat content, but affected the amount and relative N partitioning between urine and feces (similar pattern to what is depicted in Figure 1; Broderick, 2003; Misselbrook et al., 2005). Increases in urine N excretion due to excessive feeding of dietary protein and/or feeding highly soluble protein increases ammonia emissions (Table 2). Fresh and stored slurry from low CP (13.6%) diet had less than one-half the ammonia loss than slurries from the high CP (19.4%) diet (Table 2). Fresh slurry derived from BF-T-High diets had less ammonia loss than slurry from alfalfa or BF-T-Low diets. Stored slurry from BF-T-High and -Low diets had less ammonia loss than slurry derived from alfalfa.

**Table 2. Cumulative ammonia emissions for fresh and stored slurries derived from different dairy diets applied to silt loam soil (Misselbrook et al., 2005).**

Trial type	Trial components	Liquid manure type	
		Fresh	Stored
		% applied N volatilized	
CP level	13.6%	31b <sup>†</sup>	12b
	19.4%	68a	29a
Forage tannin type	Alfalfa	31a	30a
	BF-T-Low	33a	23b
	BF-T-High	25b	19b

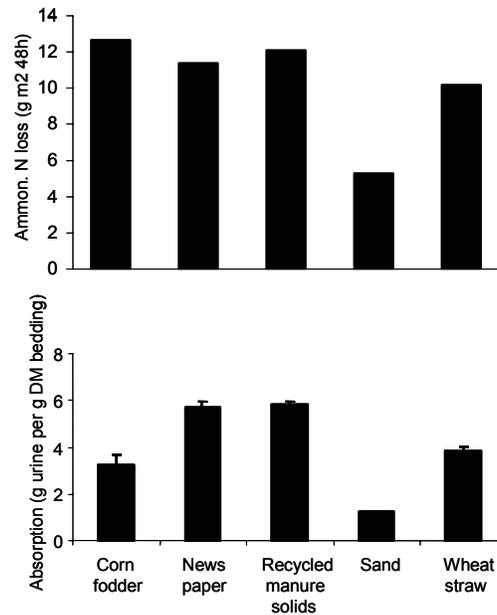
<sup>†</sup>within each trial, values with different letters are significantly different ( $P < 0.05$ )

### Bedding Affects Ammonia N Loss from Dairy Barns

Dairy cattle barns are thought to be major sources of ammonia emissions to the atmosphere. Our research is showing that the bedding material used can influence the magnitude of these emissions. The physical characteristics (urine absorbance capacity, bulk density) of bedding materials are of more importance than their chemical characteristics (pH, cation exchange capacity, carbon to nitrogen ratio) in determining ammonia emissions from applied urine and feces (Misselbrook and Powell, 2005). For example, of the bedding types commonly used in dairy barns, sand is the least and recycled manure solids the most urine absorbent. (Fig. 2). When equal volumes of urine were applied to dry bedding, ammonia emissions over 48 h were significantly lower from sand than from the other bedding types.

Preliminary results from in-barn trials show a similar pattern of ammonia emissions from beddings to that determined in the lab. Ammonia loss from composted manure solids was greater than from chopped straw and pine shavings (Data not shown). Because of warmer temperatures, ammonia emissions are 20 to 55% greater during the summer than during the winter. Our initial studies indicate that the selection of bedding type may be based not only on cow comfort and health, but also on their ability to reduce ammonia N emissions.

**Fig. 2. Urine absorbance capacity and cumulative ammonia N emissions from different beddings (Misselbrook and Powell, 2005)**



### Impact of Ammonia Loss on Plant Availability of Manure Nitrogen

In areas where dairy production is integrated with crop production, ammonia loss from manure is important because it is a direct loss of crop N that is available to the farmer. Given the high potential of ammonia N loss in manure handling, storage, and land application, only a small fraction of the N excreted by a dairy cow and applied to land may actually be recycled through crops. Furthermore, the loss of ammonia N also reduces the nitrogen:phosphorus ratio in manure, which may increase the risk of manure phosphorus applications in excess of crop needs. Many dairy farms have soil test phosphorus levels that exceed agronomic recommendations, and the runoff of phosphorus from these fields and subsequent pollution of lakes, streams, and other surface waters has become a major concern.

Reducing ammonia N losses from dairy farms and making greater use of conserved manure N will quickly make economic sense. Natural gas accounts for 75-90% of the cost of making anhydrous ammonia. As the price of natural gas continues to skyrocket, the fertilizer N value of manure, and therefore the conservation of the ammonia N contained in manure, will become more important. Reducing volatile N losses would not only conserve manure N available for field applications, but also reduce the amount of carbon dioxide, a greenhouse gas, that is generated in making fertilizer N.

### Conclusions

Substantial reductions in ammonia loss from dairy farms can be achieved by reducing in-barn losses, by covering manure storage, and by incorporation of manure in the field. The following steps can be a guide for action:

1. Remove excess protein from the cow's diet. This normally saves on feed cost, as well as reducing ammonia N emissions.
2. For new construction, floors that divert urine away from feces can reduce ammonia emissions. Slatted floors facilitate this, but there is still considerable loss of ammonia from the surface of the slatted floor.

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3. Select bedding (e.g. sand, pine shavings) that separate feces and urine, which reduce ammonia losses.
4. Cover the manure storage. When organic bedding such as straw is used, a crust will form on the surface of the slurry pit. This reduces ammonia N losses and odors. Excessive agitation during unloading of the slurry from storage should be avoided.
5. Incorporate manure in the field. However, this strategy needs to consider potential tradeoffs in situations where nitrate leaching may be a concern.

Implementation of 1, 3, 4 and 5 could potentially reduce ammonia N loss from about 50 to 14-18 kg/cow/yr, a 65-70% reduction. This means additional 32-36 kg N per cow would be available annually for application to field crops. At a fertilizer N value of \$0.75/kg N, this can mean annual savings of \$2,300 to \$2,700 per 100 dairy cows on fertilizer costs.

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## Distortion of Turbulence in the Flow Field Surrounding an Agricultural Facility

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### Abstract

The emission and dispersion of particulates from agricultural facilities at local to regional scales is a current issue in science and society. The transport of particulates in the atmosphere at these scales is largely determined by turbulence. Any models that attempt to simulate the dispersion of particles will specify or assume various statistical properties of the turbulence field. Statistical properties of turbulence are well documented for idealized boundary layers above uniform surfaces. However, an animal production facility is a complex surface with structures that act as bluff bodies and greatly distort the turbulence fields in the vicinity. As a result, the initial diffusion of plumes in the local region will be greatly affected by the complex nature of the surface. Previous LIDAR studies of plume dispersion over a facility indicate that plumes move in complex yet organized patterns that would not be explained by the properties of turbulence generally assumed in models. The objective of this study was to characterize the near surface turbulence statistics in the flow field in the vicinity of an array of animal confinement buildings. This was accomplished by erecting three towers in the upwind, within the array and downwind regions of the flow field. Changes in various turbulence statistics will be analyzed as the wind moved over the site. These include power spectra and cross-spectra, and identification of intermittent structures. The distortion of the turbulence by the structures will be quantified, as well as how it is related to various atmospheric conditions.

### Introduction

The emergence of large scale animal production facilities has contributed to concerns pertaining to the impact on air quality due to the emissions of particulates, odors ammonia (NH<sub>3</sub>), methane (CH<sub>4</sub>) and a suite of volatile organic carbons (VOC's). The focus of this investigation is particles. Confined animal production facilities present a unique challenge to estimating transport into the surrounding atmosphere. The usual approach of eddy covariance to determine fluxes, is not valid in such cases, since underlying assumptions of extensive source regions that are flat and uniform are simply not valid in these cases. These facilities contain the additional complexity of buildings distributed in a variety of configurations that act as bluff body obstructions to the mean wind flow that will affect the turbulent transport processes. To effectively model the transport processes at such facilities, requires documentation of effects of bluff bodies (or buildings) on the turbulent flow field near the facility.

The properties of the turbulence and resultant diffusion, result from both the general state of the micrometeorological conditions that contribute to the stability of the boundary layer at the surface, and the local effects of structures present at the facility. The mechanisms of the dispersion of the particulates are governed by scales of turbulence that range from very small local scales of cm, up to the atmospheric boundary layer that may extend to heights of several km.

In order to understand the transport of particles from a complex source, such as an agricultural facility, the distortion of the turbulence by the array of sources must be documented. The effects of the structures on the turbulence must be quantified and related to the properties of the undisturbed flow field.

The objectives of this study are:

1. To quantify the changes in turbulence statistics that result from the presence of the structures and spatially complex sources of an agricultural facility.
2. Integrate these findings with the observed images of spatial variations of particle density determined from lidar measurements.

### Methods

A study was conducted at a small swine facility near Ames, Iowa that began in late August and continued for 10 days into early September of 2005. The facility was comprised of three buildings with dimensions of 55 m length by 18 m wide and 6 m in height with the length of the buildings oriented in an east-west direction. They were separated by a distance of 18 m. Each building housed approximately 1300 pigs and represented the source area for particulates. At a distance of 200 m to south of the facility a 10 m tower was erected in a soybean field, with two eddy covariance systems mounted at 4 and 8 m above the soybean canopy. The same configuration was sited 200 m to the north of the buildings in a corn field. The prevailing wind direction during the study was from the S-SW, and the above towers represented upwind and downwind (from the buildings) turbulence conditions. Between the northern most and center buildings a 20 m tower was erected, onto which 3 eddy covariance systems were mounted at 7, 12 and 18 m above the ground level. The 7 m height was approximately 1 m above the apex of the building roof. This configuration allowed us the opportunity to measure horizontal and vertical turbulence characteristics of the wind flow in the upwind, among building and downwind turbulence flow fields.

Each EC system for all towers was comprised of a 3-D sonic anemometer (CSAT3, Campbell Scientific Inc.) and a LI7500 IRGA (LiCOR Inc.). Additional meteorological instrumentation included a temperature and humidity probe (HMP-45, Vaisala) on each tower. At the 20 m tower all three eddy covariance systems were wired to a single data logger (CR5000, Campbell Scientific Inc.) so that simultaneous and synchronized data acquisition could be accomplished for the vertical EC configuration. The north and south tower EC systems were also wired to a CR5000 data logger. The EC systems were programmed to sample at a rate of 10 Hz while the temperature and humidity sensors were sampled at a rate of 1 Hz. The high frequency EC data series were conditioned following common procedures in micrometeorology described in Kaimal and Finnigan, (1994).

### Preliminary Results

The initial analysis of the data begin by identifying a simple case that would include strong winds from the south, to document changes in turbulence statistics upwind, over, and downwind of the facility. This occurred on September 05, 2005 between 1100 and 1600 hours. We present the ambient wind and temperature conditions for the period from 1500-1600 hours (Table 1).

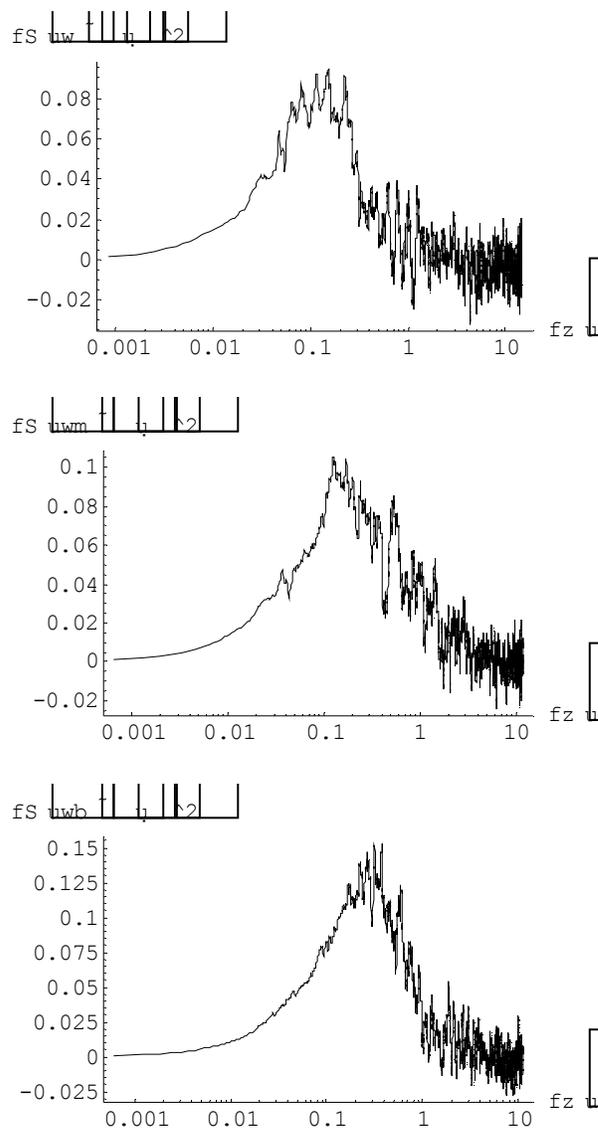
<i>Tower</i>	<i>Wind speed</i>	<i>Temperature</i>	<i>Wind Direction</i>	<i>u*</i>
	(m/s)	(deg C)	(deg)	(m/s)
South	5.3	29.0	197	0.40
Bottom	3.1	29.5	196	0.69
Middle	5.1	29.0	194	0.56
Top	6.1	29.8	193	0.39
<b>North</b>	3.0	28.7	190	0.50

**Table 1. Average wind and temperature conditions for DOY 248, for the period between 1500-1600 hours at the south, north and 20 m towers.**

Table 1 shows that air temperatures were warm with moderate winds originating from the south at  $\sim 5$  m/s. Friction velocity ( $u^*$ ) values ranged between 0.39 and 0.69 m/s in response to different surface roughness conditions. The largest intensity of turbulence was observed at the lowest height of the tower over the facility, likely due to the bluff effects of the buildings. Average air temperatures were coolest over the corn surface and warmest over the building array. A notable difference can be seen between the average wind speeds over the soybean (south) in the approach position relative to the corn (north) and is an indication of the greater surface roughness presented by the corn canopy relative to the soybeans.

A sample of the normalized cospectra plots for momentum ( $uw$ ) from the three heights at the 20 m tower is shown in Figure 1. The cospectra show the contribution to the flux by eddies of different frequency. In this case we are interested in the influence of the buildings on the turbulence structure for momentum and apply this understanding to particulate transport. The upper plot corresponds to the top location on the tower and descends accordingly with the bottom plot representing the bottom eddy covariance system. The observed spectra for the three heights show a trend where the frequencies of peak contributions grow smaller with height. This is reasonable and expected as the greater heights will encounter larger eddies relative to the bottom system. However, the magnitudes of the cospectral densities are greatest at the bottom and least at the top, indicating more momentum transfer due to the distorting effect on the turbulence from the physical building obstruction.

Note also that the reduction of the contributions at higher frequencies than the peaks, differ from the three heights. The classic 5/3 law is not observed at all heights, suggesting again the distortion of turbulence by the facility. Further analysis will examine the effects of these distortions to the  $H_2O$  and  $CO_2$  fluxes as well as compare the cospectra results from the 20 m tower to those from the upwind and downwind turbulent fields.



**Figure 1. Cospectra of horizontal and vertical velocity for 18 m height (upper), 12 m height (middle), and 7 m height (lower), above the facility.**

Future analyses will integrate the observed changes in the spectra and cospectra of turbulence, with the spatial patterns of particulates as measured by the lidar. The combination of the two sets of data will suggest how the dispersion of particulates is affected by the complexity of such a site. The results will suggest implications for modeling the dispersion of particles from these facilities.

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