



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

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Abstract

Soil and crop management practices can influence CO₂ emission from crop and grasslands and therefore on global warming. We examined the effects of two irrigation systems (irrigated vs. non-irrigated) and six management practices [no-till malt barley (*Hordeum vulgare* L.) with 67 or 134 kg N ha⁻¹ (NTBFN), no-till malt-barley with 0 kg N ha⁻¹ (NTBON), conventional-till malt barley with 67 or 134 kg N ha⁻¹ (CTBFN), conventional-till malt barley with 0 kg N ha⁻¹ (CTBON), no-till pea (*Pisum sativum* L.) with 0 kg N ha⁻¹ (NTPON), and undisturbed alfalfa (*Medicago sativa* L.) and grasses with 0 kg N ha⁻¹ (UAGON)] on soil surface CO₂ flux and soil temperature and water content at the 0 to 15 cm depth. Weekly CO₂ flux, soil temperature, and soil water content were monitored during the crop growing season from May to November 2005 in Lihen sandy loam (sandy, mixed, frigid, Entic Haplustolls) in western North Dakota. Irrigation increased CO₂ flux by 27% compared with non-irrigation by increasing soil water content during dry periods. Similarly, tillage increased CO₂ flux by 58% compared with non-tillage by increasing soil temperature. The CO₂ flux was 1.5 to 2.5-fold greater in tilled than in non-tilled treatments following heavy rain or irrigation. Nitrogen fertilization increased CO₂ flux compared with no N fertilization in 2 out of 17 measurements while cropping system did not influence CO₂ flux. The CO₂ flux in undisturbed alfalfa and grasses was similar to that in no-tilled crops. The CO₂ flux was linearly related with soil temperature and daily average air temperature at the time of CO₂ measurement. Tillage followed by heavy rain or irrigation during the crop growing season drastically increased CO₂ flux in the coarse-textured soil previously managed under Conservation Reserve Program (CRP) planting for more than 20 yr.

Introduction

Global warming due to increased concentration of greenhouse gases, such as CO₂, in the atmosphere is a major concern. One of the significant sources of CO₂ emission is from agricultural practices, which contribute about 25% of total anthropogenic emissions (Duxbury, 1994; 1995). Soil can act as both source and sink of atmospheric CO₂. The CO₂ fixed in plant biomass through photosynthesis can be stored in the soil as organic C by converting plant residue into soil organic matter after the residue is returned to the soil. While management practices, such as tillage, can increase CO₂ emission from the soil by disrupting soil aggregates, increasing aeration, incorporating plant residue, and oxidizing soil organic C (Reicosky and Lindstrom, 1993; Beare et al., 1994; Jastrow et al., 1996), no-tillage and increased cropping intensity can increase C storage in the soil (Lal et al., 1995; Paustian et al., 1995). Respiration by plant roots and soil microflora and fauna also contribute a major portion of CO₂ emission from the soil (Rochette and Flanagan, 1997; Curtin et al., 2000). The CO₂ emission from the soil to the atmosphere is the primary mechanism of C loss from the soil (Parkin and Kaspar, 2003) and provides an early indication of C sequestration in the soil when changes in soil organic C due to management practices are not detectable within a short period (Fortin et al., 1996; Grant, 1997).

Reduced tillage is regarded as one of the most effective agricultural practices to reduce CO₂ emission and sequester atmospheric C in the soil (Kern and Johnson, 1993; Lal and Kimble, 1997; Curtin et al., 2000; Al-Kaisi and Yin, 2005). Decreased tillage intensity reduces soil disturbance and microbial activities, which in turn, lowers CO₂ emission (Lal and Kimble, 1997; Curtin et al., 2000). Moreover, no-till with surface residue can further reduce CO₂ emission compared to no-till without residue (Al-Kaisi and Yin, 2005). In contrast, increased tillage intensity increased CO₂ emission by increasing aeration due to increased soil disturbance (Roberts and Chan, 1990) and to physical degassing of dissolved CO₂ from the soil solution (Reicosky and Lindstrom, 1993; Jackson et al., 2003). The effect of tillage on CO₂ emission, however, was short-lived (<24 h) (Reicosky and Lindstrom, 1993; Ellert and Janzen, 1999). Rewetting of

dry soil due to irrigation or rainfall increased CO₂ flux by increasing microbial activities, C mineralization, and respiration (Sparling and Ross, 1988; Van Gestel et al., 1993; Calderon and Jackson, 2002).

Cropping system can influence CO₂ emission by affecting on the quality and quantity of residue returned to the soil (Curtin et al., 2000; Al-Kaisi and Yin, 2005; Amos et al., 2005). Increased above- and belowground biomass production can increase the amount of residue returned to the soil (Sainju et al., 2005), thereby increasing CO₂ flux (Franzluebbers et al., 1995; Curtin et al., 2000; Al-Kaisi and Yin, 2005). Increased belowground biomass production can also increase root and rhizosphere respiration, thereby increasing CO₂ flux (Amos et al., 2005). Similarly, residue quality, such as C/N ratio, can alter the decomposition rate of residue (Kuo et al., 1997; Sainju et al., 2002), thereby influencing CO₂ emission (Al-Kaisi and Yin, 2005). Nitrogen fertilization had little effect on CO₂ emission from the soil surface (Rochette and Gregorich, 1998; Wagai et al., 1998; Amos et al., 2005).

Management practices can affect soil temperature and water content (Curtin et al., 2000; Al-Kaisi and Yin, 2005) which directly influence CO₂ emission rates (Bajracharya et al., 2000b; Parkin and Kaspar, 2003; Amos et al., 2005). For example, tillage can dry the soil but no-till can increase soil water content and reduce soil temperature because of residue accumulation at the soil surface (Curtin et al., 2000; Calderon and Jackson, 2002; Al-Kaisi and Yin, 2005). Similarly, cropping system can reduce soil temperature by providing shade with increased biomass growth but can reduce soil water content due to increased evapotranspiration (Amos et al., 2005).

Little is known about the combined effects of irrigation, tillage, cropping system, and N fertilization on CO₂ flux in transitional land that has been converted from Conservation Reserve Program (CRP) planting to agricultural use. The CO₂ flux from the conversion of no-till to conventional till, from dryland to irrigated, or from less intensive to more intensive cropping system could be different from the flux obtained during the reverse trend of land conversion. We hypothesized that the individual and combined uses of irrigation, tillage, cropping system, and N fertilization have differential effects on soil surface CO₂ flux in the transitional land. Our objectives were to: (1) determine the effects of irrigation, tillage, cropping system, and N fertilization on soil surface CO₂ flux during the crop growing season in land converted from CRP to agricultural use, and (2) compare CO₂ fluxes in the agricultural system and undisturbed alfalfa and grasses.

Materials and Methods

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

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



Figure 1. Soil surface CO₂ flux measurement with Environmental Gas Monitor chamber and data logger

In May 2005, malt barley (cv. Certified Tradition, Busch Agricultural Resources, Fargo, ND) was planted at 3.8 cm depth at the rate of 90 kg ha⁻¹ in the irrigated treatment and 67 kg ha⁻¹ in the non-irrigated treatment with a no-till drill. Similarly, pea (cv. Majorete, Macintosh Seed, Havre, MT) was planted at 200 kg ha⁻¹ in irrigated and non-irrigated treatments. In irrigated malt barley, half of N fertilizer as urea (or 67 kg N ha⁻¹) was banded at planting and other half was broadcast at 4 wk after planting. In non-irrigated malt barley, all N fertilizer was banded at planting. The P fertilizer as triple super phosphate and K fertilizer as muriate of potash were banded at 25 kg P ha⁻¹ and 21 kg K ha⁻¹, respectively, each to malt barley and pea at planting. The amount of N, P, and K fertilizers to malt barley and pea were based on soil test and crop requirement. No fertilizers were applied to alfalfa and grasses in UAGON treatment which resembled undisturbed CRP planting. However, as with other management practices, half of UAGON treatment in split-plot arrangement received irrigation and other half did not. Appropriate types and amounts of herbicides and pesticides were used to control weeds and pests during growth and after harvest of malt barley and pea. In irrigated plots, water was applied at amounts ranged from 10 to 25 mm of rain from 17 June to 14 July 2005 (a total of 87 mm) based on soil moisture content. In July and August, malt barley and pea were hand harvested from an area of 0.5 x 4 m² and straw (leaves and stems) was returned to the soil.

Immediately after planting, soil surface CO₂ flux was measured weekly in all treatments during and after the crop growing season from May to November 2005 until the ground froze. All measurements were made between 9 A.M. and 12 A.M. of the day to reduce variability in CO₂ flux due to diurnal changes in temperature (Parkin and Kaspar, 2003). The CO₂ flux was measured with an Environmental Gas Monitor chamber attached to a data logger (model no. EGM-4, PP System, Haverhill, MA) (Figure 1). The chamber is 15 cm tall, 10 cm in diameter, and has capacity to measure CO₂ flux from 0 to 9.99 g CO₂-C m⁻² h⁻¹. The chamber was placed at the soil surface for 2 min in each plot until CO₂ flux measurement was recorded in the data logger. A flag was placed as a marker in the plot where CO₂ flux was measured throughout the study period. At the time of CO₂ measurement, soil temperature near the chamber was measured from a depth of 0 to 15 cm from a probe attached to the data logger. Similarly, gravimetric soil water content was measured near the chamber by collecting soil sample from the 0 to 15 cm depth with a probe (2.5 cm diameter) every time CO₂ flux was measured. The moist soil was oven-dried at 110°C and water content

was determined. Daily average air temperature and total rainfall during the study period were collected from a meteorological station located at 0.8 km from the study site.

Data for CO₂ flux, soil temperature, and soil water content were analyzed by using the Analysis of Repeated Measures procedure in the MIXED model of SAS (Littell et al., 1996). Irrigation was considered as the main plot, management practices as the split plot, and date of measurement as the repeated measure treatment in the analysis. Means were separated by using the least square means test when treatments and interactions were significant. Statistical significance was evaluated at $P \leq 0.05$ unless otherwise mentioned. Regression analysis was done between CO₂ flux and average daily air temperature, soil temperature, and soil water content to determine their relationships.

Results and Discussion

Climate

Rainfall and air temperature influence soil water content and soil temperature, which in turn, directly influence CO₂ flux from the soil surface (Bajracharya et al., 2000; Curtin et al., 2000; Parkin and Kaspar, 2003). Daily total rainfall was higher in May and June 2005 during the critical growing period of crops, and then declined from July to November as the crop matured (Figure 2A). Since air temperature is more correlated with CO₂ flux than soil temperature (Parkin and Kaspar, 2003), daily average air temperature was also measured to determine its influence on soil CO₂ flux, besides measuring soil temperature and water content. As expected, daily average air temperature increased from May to August and then declined (Figure 2B).

Effect of Irrigation

Soil temperature at the 0 to 15 cm depth followed a trend similar to that of air temperature (Figure 3B) but was less variable (Figure 2B). Irrigation resulted in a significant ($P \leq 0.001$) interaction with date of measurement on soil temperature, water content, and CO₂ flux (Table 1). Soil temperature was lower in irrigated than in non-irrigated treatment in Days 176, 195, 202, 208, and 217 of the year (Figure 2B). In contrast, soil water content at the 0 to 15 cm depth was higher in irrigated than in non-irrigated treatment in Days 176, 202, 208, and 217 of the year. It is not surprising to observe lower soil temperature and higher water content with irrigated treatment in these days when irrigation was done to meet the crop water requirement during periods of higher air temperature and lower rainfall (Figures 2A, 2B, 3C). Increased water content and evaporation from the soil surface probably reduces soil temperature, as wet soil is slower to change in temperature than dry soil (Bajracharya et al., 2000b; Parkin and Kaspar, 2003). Increased rainfall increased soil water content of both irrigated and non-irrigated treatments in May and June when no irrigation was done.

The CO₂ flux increased from 31 kg CO₂-C ha⁻¹ d⁻¹ in Day 122 to 427 kg CO₂-C ha⁻¹ d⁻¹ in Day 188 of the year, after which it declined (Figure 3A). The flux was higher in irrigated than in non-irrigated treatment in Days 158, 176, 179, and 188. Increased soil moisture content rather than a difference in soil temperature (Figures 3B and 3C) probably increased CO₂ flux with irrigation during these periods. Increased CO₂ flux after irrigation or after a heavy rain in dry soil resulting from increased C mineralization have been known (Howard and Howard, 1993; van Gestel et al., 1993; Curtin et al., 2000; Calderon and Jackson, 2002). Since a major portion of soil CO₂ flux is contributed by respiration from plant roots, rhizosphere, and microbial flora and fauna (Rochette and Flanagan, 1997; Curtin et al., 2000), it is likely that irrigation of dry soil increased microbial activities and CO₂ emission. Averaged across measurement dates, irrigation decreased soil temperature by 0.8°C and increased soil water content by 6% and CO₂ flux by 15% (Table 2).

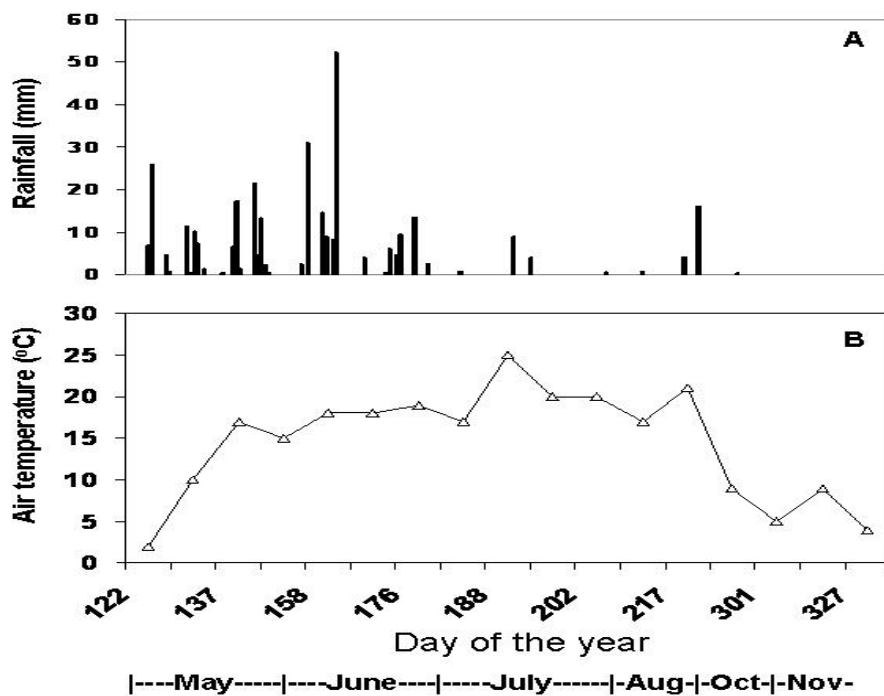


Figure 2. Daily (A) total rainfall and (B) average air temperature at the time of CO₂ flux measurement from May to November 2005 in western North Dakota.

Table 1. Analysis of variance for soil surface CO₂ flux and soil temperature and water content at the 0 to 15 cm depth.

Source	Soil CO ₂ flux	Soil temperature	Soil water content
Irrigation (I)	**	*	*
Management practices (M)	***	***	***
I × M	NS†	**	NS
Date of measurement (D)	***	***	***
I × D	***	**	***
M × D	***	***	**
I × M × D	NS	*	*

* Significant at $P \leq 0.05$.

** Significant at $P \leq 0.01$.

*** Significant at $P \leq 0.001$.

† Not significant.

Table 2. Effects of irrigation and management practices on soil surface CO₂ flux and soil temperature and water content at the 0 to 15 cm depth averaged across dates of measurement.

Irrigation	Management practices†	Soil surface CO ₂ flux (kg CO ₂ -C ha ⁻¹ d ⁻¹)	Soil temperature (°C)	Soil water content (g kg ⁻¹)
Irrigated		194a‡	14.8a	97.3a
Non-irrigated		169b	15.6b	91.9b
	NTBFN	164b	15.0c	97.1a
	NTPON	155b	15.2b	96.5a
	CTBFN	250a	15.0c	95.3a
	UAGON	150b	15.3b	86.9b
	NTBON	145b	15.2b	97.9a
	CTBON	242a	15.5a	94.7a

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

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

Effect of Management Practices

Differences in soil temperature, water content, and CO₂ flux between management practices at some measurement dates resulted in a significant ($P \leq 0.01$) management practice \times measurement date interaction (Table 1). Soil temperature was higher in UAGON than in all other practices, except in NTBON, in Day 176 (Figure 4B). Soil temperature was also higher in CTBON than in most of the other management practices from Day 195 to 217. While soil water content was consistently lower in UAGON from Day 137 to 158 and from Day 202 to 217, it was higher in NTPON from Day 144 to 166, higher in NTBON in Days 176, 179, 202, and higher in NTBFN in Days 208 and 217 than in most of the other management practices (Figure 4C). Increased soil temperature and decreased water content in UAGON was probably due to greater water requirement of alfalfa and grasses. Similarly, increased soil temperature and reduced water content in CTBON was likely a result of drying up of soil due to tillage and reduced plant growth. The plant biomass and yield were lower in CTBON (data not shown), probably due to low N availability, since N fertilizer was not applied in this treatment. Tillage can result in drying up of soil due to increase in water vapor flux (Kessavalou et al., 1998), and decreased shading with reduced plant growth can increase soil temperature (Amos et al., 2005). In contrast, increased soil water content and decreased soil temperature in NTBON, NTPON, and NTBFN was probably due to less soil disturbance, followed by residue accumulation at the soil surface. Increased soil water content and decreased soil temperature in no-till than in conventional till have been reported by several researchers (Curtin et al., 2000; Calderon and Jackson, 2002; Al-Kaisi and Yin, 2005).

Management practices, especially tillage, increased CO₂ flux more than the irrigation treatment by ≥ 159 kg CO₂-C ha⁻¹ d⁻¹ at certain measurement dates (Figure 4A). Drastic increases occurred in tilled treatments immediately following heavy rain or irrigation in Days 130, 158, 188, 195, 202, and 208 when CO₂ fluxes in CTBFN and CTBON were 1.5 to 2.5-fold higher than in other treatments. The flux was also higher in NTBFN and NTPON than in UAGON in Day 179 but was higher in UAGON than in NTPON in DAY 195. Similarly, the flux was higher in NTBFN than in NTBON and UAGON in Day 188 and was higher in CTBFN than in CTBON in Day 202.

The CO₂ flux of as much as 300 kg CO₂-C ha⁻¹ d⁻¹ following tillage and heavy rain in dry soil in northern Great Plains had been known (Curtin et al., 2000). Our results of CO₂ flux of as much as 600 kg CO₂-C ha⁻¹ d⁻¹ in CTBFN and CTBON following heavy rain or irrigation seems to be extreme but could be due to tillage in the soil that had been under CRP management for more than 20 years before the study was initiated. Tillage can result in an immediate short-term outburst of CO₂ due to decrease in partial pressure of CO₂ in soil air, followed by disturbance in soil aggregation and pores, and sudden release of CO₂ from the soil solution (Reicosky et al., 1995; Ellert and Janzen, 1999; Rochette and Angers, 1999). The effect of tillage on the CO₂ flux was short-lived, as suggested by others (Reicosky and Lindstrom, 1993; Al-Kaisi

and Yin, 2005), because fluxes were lower after tillage in the initial periods of study in Days 122, 137, and 144, regardless of management practices (Figure 4A). However, fluxes triggered in tilled treatments in Days 130, 158, 188, 195, 202, and 208 immediately after heavy rains and irrigation in dry soil. Under raindrop and irrigation impact, the soil surface may have sealed in the tilled treatment, temporarily trapping CO₂. As the water drained and soil surface dried, trapped CO₂ may have released in a brief, intense burst, as suggested by Curtin et al. (2000). Another possibility would be that tillage accompanied by rainfall and irrigation could have increased microbial activities, thereby increasing C mineralization and CO₂ flux. Better soil structure along with increased surface residue cover could have reduced surface sealing in the no-till treatment. Changes in soil organic C content due to short-term burst of CO₂ from tillage would be minor because only a small portion of soil organic matter mineralizes during the initial phase of tillage (Curtin et al., 2000; Al-Kaisi and Yin, 2005). Tillage also may have indirectly influenced CO₂ flux during these periods by increasing soil temperature (Figure 4B). Although CO₂ flux can be influenced by a range of factors including soil temperature, water content, wind speed, and CO₂ concentration gradient, greater fluxes in this study than those obtained by Curtin et al. (2000) could also be due to a difference in soil texture. Soil texture in our study was sandy loam and in their study was silt loam. The CO₂ flux increases with decrease in clay content (Parkin and Kaspar, 2003). Nevertheless, our study shows that tillage accompanied by heavy rain or irrigation can cause a sudden outburst of CO₂ during the crop growing season in the summer from areas previously under CRP management.

The CO₂ flux was similar under malt barley and pea in NTBON and NTPON, suggesting that cropping system did not influence CO₂ emission. Al-Kaisi and Yin (2005) reported no significant difference in CO₂ flux between corn (*Zea mays* L.) and soybean (*Glycine max* L.). However, lower CO₂ flux in UAGON than in other treatments on some measurement dates was probably a result of decreased soil water content (Figures 4A, 4B, 4C). The greater CO₂ flux in NTBON than in NTBON and in CTBON than in CTBON in Days 188 and 202 suggests that N fertilization increased CO₂ flux when soil water content is adequate. Limited or no increase in CO₂ flux with increased N fertilization rate had been reported by several researchers (Rochette and Gregorich, 1998; Wagai et al., 1998; Amos et al., 2005).

Averaged across measurement dates, CTBON and CTBON increased soil surface CO₂ flux by an average of 58% compared with other treatments (Table 2). These treatments also increased soil temperature and reduced soil water content, except in UAGON. Eliminating the short-term burst of CO₂ immediately after tillage (Al-Kaisi and Yin, 2005) which was not measured in this study, tillage can increase CO₂ flux by an average of 90 kg CO₂-C ha⁻¹ d⁻¹ (246 kg CO₂-C ha⁻¹ d⁻¹ in tilled vs. 156 kg CO₂-C ha⁻¹ d⁻¹ in non-tilled treatments) or 19.3 Mg CO₂-C ha⁻¹ compared with no-tillage from May to November (214 d). If the loss was estimated for the entire year using the same rate of CO₂ emission, it would be 32.8 Mg CO₂-C ha⁻¹ yr⁻¹. When CO₂ flux from tilled treatments during the peak emission periods in Days 130, 158, 179, 188, 195, and 202 were eliminated, the loss of CO₂ flux in tilled vs. non-tilled treatment averaged 4.7 Mg CO₂-C ha⁻¹ yr⁻¹. The value could, however, be overestimated, because CO₂ flux rates are lower in winter than in summer (Follett, 1997; Bajracharya et al., 2000a). Furthermore, half of this CO₂ loss would be contributed by respiration from plant roots and soil microflora and fauna (Curtin et al., 2000). Soil C sequestration rate in northern Great Plains using no-till management compared with tilled treatment has been estimated at 5 to 6 Mg ha⁻¹ yr⁻¹ after 13 to 14 yr or about 400 kg C ha⁻¹ yr⁻¹ (Curtin et al., 2000). These data show that tillage can cause a drastic loss of soil C as CO₂ emission within a short period of time, but it takes a long time to sequester C using no-till practices.

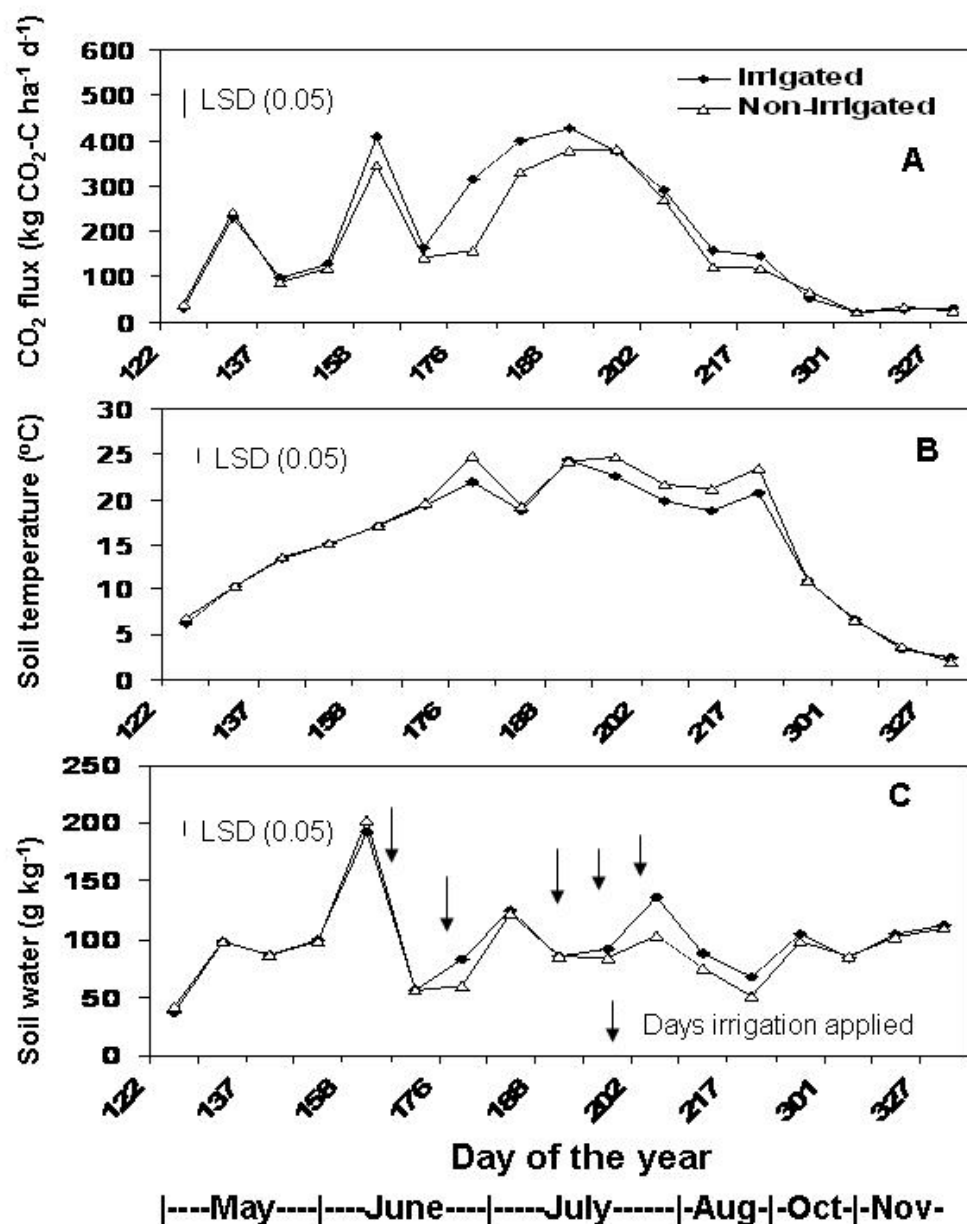


Figure 3. Effect of irrigation on (A) soil surface CO₂ flux and soil (B) temperature and (C) water content at the 0 to 15 cm depth averaged across management practices from May to November 2005 in western North Dakota. LSD (0.05) is the least significant difference between treatments at $P < 0.05$.

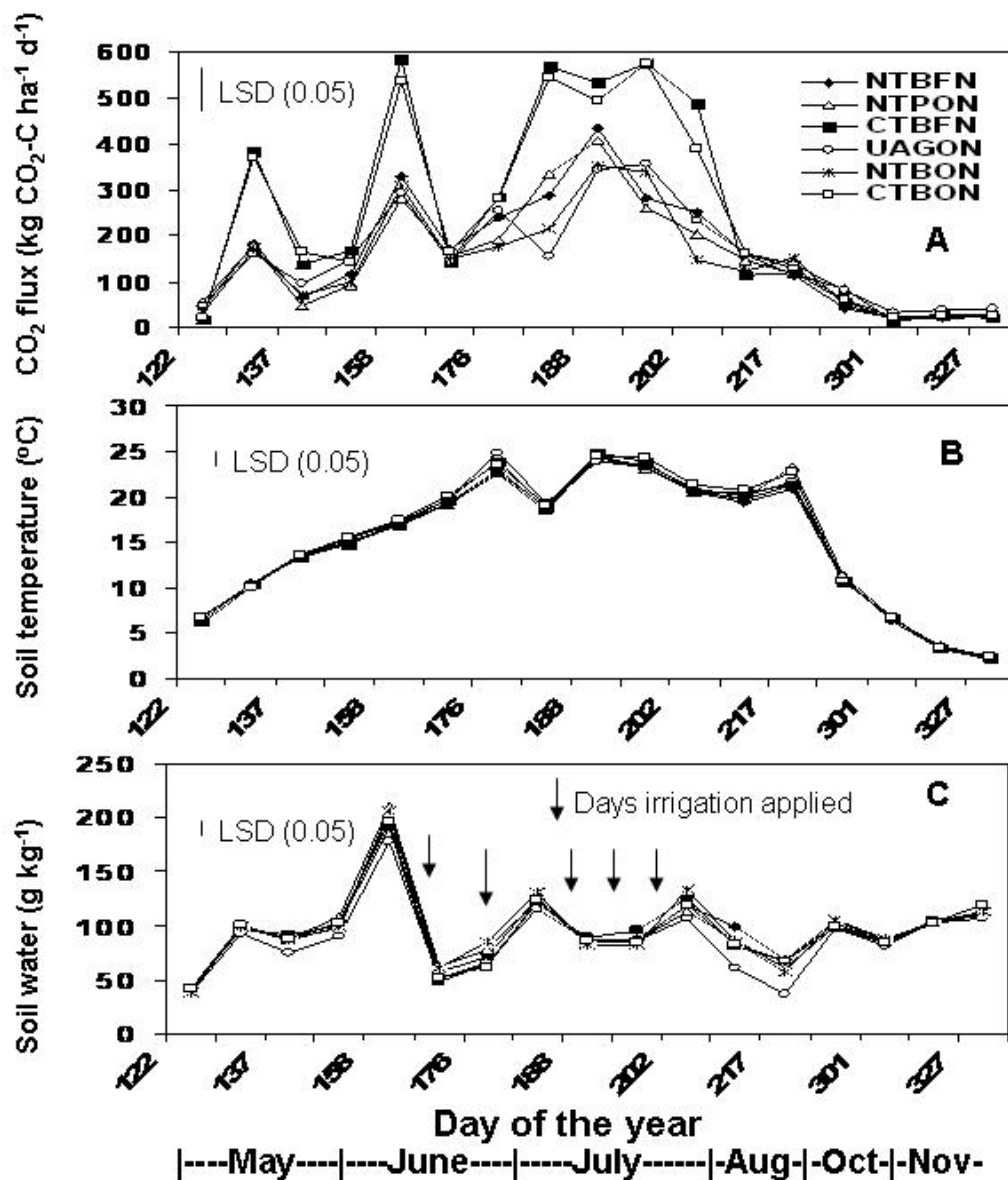


Figure 4. Effects of management practices on (A) soil surface CO_2 flux and soil (B) temperature and (C) water content at the 0 to 15 cm depth averaged across irrigation systems from May to November 2005 in western North Dakota. Management practices are CTBFN, conventional-till malt barley with 67 or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

Relationship between Carbon Dioxide Emission and Temperature and Soil Water Content

A regression analysis between CO₂ flux and soil temperature at the 0 to 15 cm depth averaged across irrigation and management practices treatments and daily average air temperature at the time of CO₂ measurement revealed that CO₂ flux was linearly related ($R^2 = 0.55$ to 0.56 , $P \leq 0.001$, $n = 17$) with both soil temperature and air temperature (Figures 5A and 5B). Both soil temperature and air temperature accounted for 55 to 56% of the variability in CO₂ flux with similar slopes. Soil water content at the 0 to 15 cm depth was weakly related ($R^2 = 0.18$, $P \leq 0.10$, $n = 17$) with CO₂ flux.

The significant relationship between CO₂ flux and soil temperature is well known (Follett, 1997; Bajracharya et al., 2000b; Parkin and Kaspar, 2003). High CO₂ flux usually occurs in the summer when soil temperature is higher and soil water content and substrate C availability is adequate, while low emission occurs in the winter when soil biological activity is minimal due to near-freezing soil temperature (Kirschbaum, 1995; Follett, 1997; Bajracharya et al., 2000a; 2000b). Parkin and Kaspar (2003) reported that soil surface CO₂ flux was better related with air temperature than with soil temperature because of plant residue at the soil surface. Our results showed that CO₂ flux was equally related with both soil temperature and daily average air temperature at the time of CO₂ flux measurement. Significant effects of irrigation and management practices on CO₂ flux and soil temperature and water content (Table 1) suggests that these treatments modified soil temperature and water content which influenced microbial activities and C mineralization, thereby resulting in different CO₂ fluxes between treatments at various measurement dates.

Conclusions

Results of this study showed that both irrigation and tillage influenced soil temperature and water content and increased soil surface CO₂ flux from areas previously managed under CRP planting for more than 20 years. Tillage followed by heavy rain or irrigation drastically increased the flux during the crop growing season from May to August 2005. The CO₂ flux was influenced by soil temperature and water content which were modified by treatments. Cropping system and N fertilization had limited or no effect on CO₂ flux in the first year after the establishment of treatments. The CO₂ flux in undisturbed alfalfa and grasses was similar to that in no-till malt barley and pea treatments. Long-range studies are needed to determine the effects of treatments on CO₂ flux under various climatic and environmental conditions to estimate C loss from the soil ecosystem.

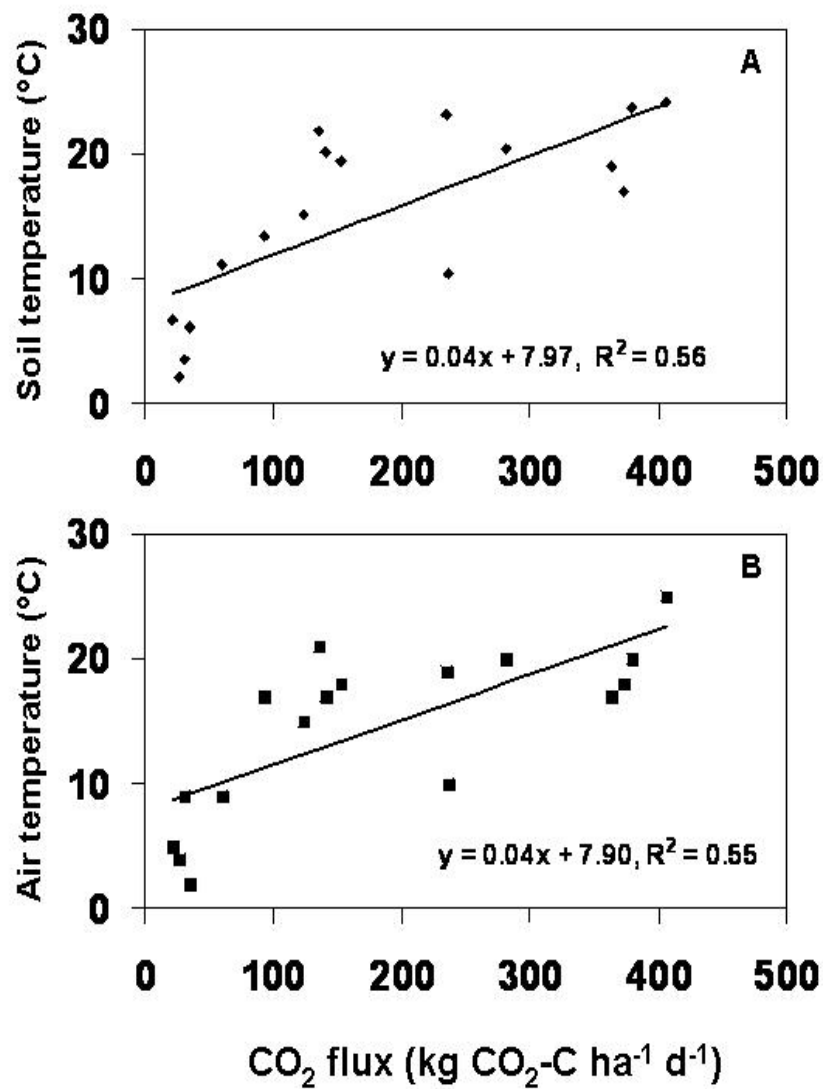


Figure 5. Relationship between soil surface CO₂ flux and (A) soil temperature at the 0 to 15 cm depth averaged across irrigation and management practices and (B) daily average air temperature at the time of CO₂ measurement.

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

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Abstract

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

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

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Summary

Investigation on precipitation quality from lowland of Vaz Research Forest in Caspian Sea region to upland areas of 1000, 1600 and 2200 meters altitude had been carried out from 1999 to 2003. There are different amount of rainfall in varying elevations, and in natural cases showed that amount of Nitrate (NO_3^-), Sulfate (SO_4^{2-}), Chloride (Cl^-), Ammonium (NH_4^+), Calcium (Ca^{++}) and Magnesium (Mg^{++}) have regular relation with altitude, and annual deposition of elements is decreased parallel with increased altitude. Nitrate wet deposition in 300, 1000, 1600 and 2200 meters were 250, 275, 193 and 104 kg/ha.yr respectively. Sulfate ion in this way including 21, 45, 28 and 2.5 kg/ha.yr wet deposition parallel with increasing mentioned altitudes. Chloride ion which originates mainly from spraying of sea-water, includes 1080, 620, 784 and 337 kg/ha.yr wet deposition regard to 300, 1000, 1600 and 2200 meters elevation respectively. Ammonium ion (NH_4^+) also including 19, 4, 8 and 4 kg/ha.yr related to 300, 1000, 1600 and 2200 meters elevation respectively. Annual wet deposition of Calcium (Ca^{++}) are 38, 22, 24 and 12 kg/ha.yr wet deposition regard to 300, 1000, 1600 and 2200 meters elevation respectively. Increased amount of NO_3^- and SO_4^{2-} in 1000 meter altitude is reversed and different with decreased amounts parallel with increasing altitude, which originating from fossil fuel consumption by native community and resident people and anthropogenic effects. Acidity (pH) of precipitation were in natural and alkaline criteria, and total average of pH for precipitation regards to each elevation were more than 7 in pH value. Measurement of Nitrogen showed that there are annual wet deposition in amount of 56, 62, 44 and 23 kg/ha.yr wet deposition regard to 300, 1000, 1600 and 2200 meters elevation respectively.

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

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

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

Introduction

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

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

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

Results

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

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

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

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

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

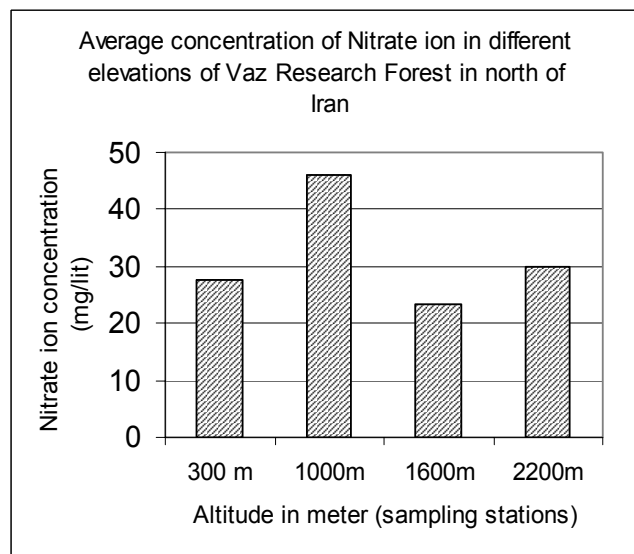
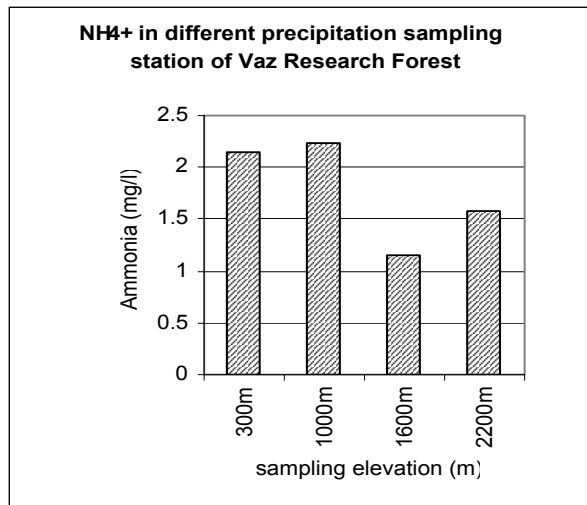
Analysis of Variance for Ammonium ion (NH_4^+) related to precipitation in Vaz Research Forest showed that there was significant difference ($p = 0.01$) for different treatments, different altitudes also between different seasons of sampling.

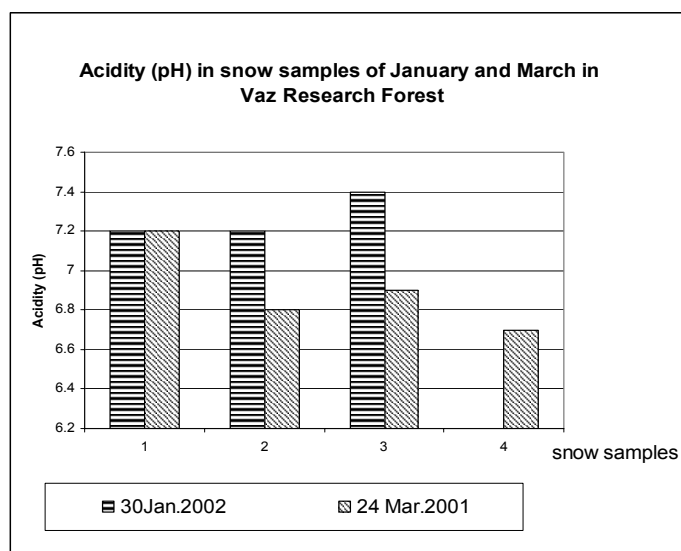
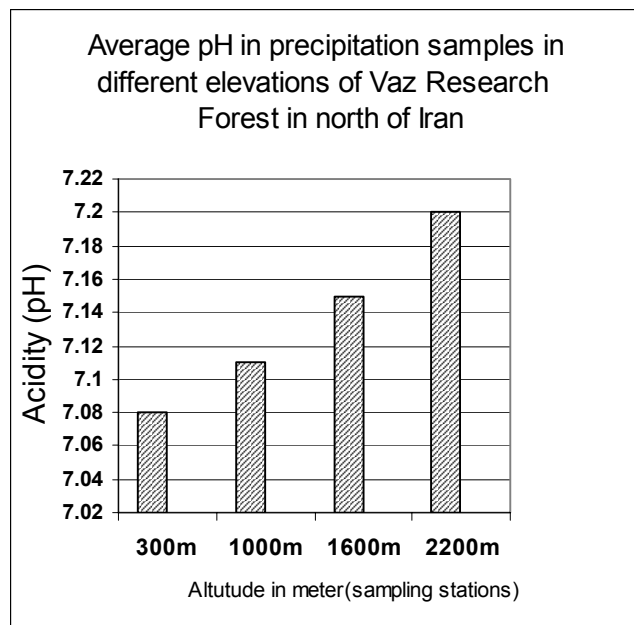
Analysis of Variance for Magnesium ion (Mg^{++}) related to precipitation in Vaz Research Forest showed that there was significant difference ($p = 0.01$) for different treatments, and between different seasons of sampling.

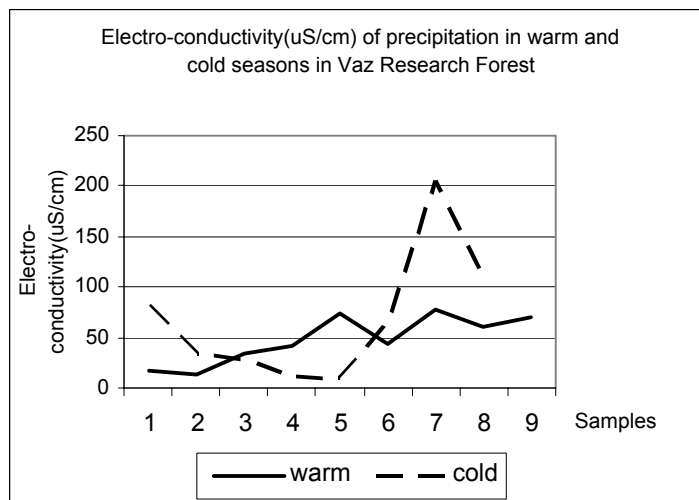
Analysis of Variance for Calcium ion (Ca^{++}) related to precipitation in Vaz Research Forest showed that there was significant difference ($p = 0.01$) for different treatments, different altitudes also between different seasons of sampling.

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

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







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

Measurement of NO_3^- and SO_4^{2-} in snow samples regards to different elevations of 1600, 2000 and 2200 meters showed that based on molarity, Nitrate molarity is 61 times of Sulfate molarity in snow samples. Weight-based comparison showed that weight equivalent of Nitrate ion is 35 times more than SO_4^{2-} .

Discussion

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

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

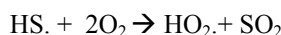
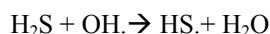
Measurement of NO_3^- and SO_4^{2-} in snow samples regards to different elevations of 1600, 2000 and 2200 meters showed that based on molarity, Nitrate molarity is 61 times of Sulfate molarity in snow samples. Weight-based comparison showed that weight equivalent of Nitrate ion is 35 times more than SO_4^{2-} .

Mineral Sulfur compounds

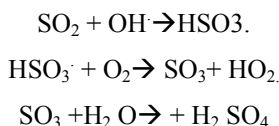
More than 60 compounds of mineral Sulfur which is released into atmosphere, can be detected. Some Sulfur content gases are produced by industrial processes and main sources of H_2S are found in urban areas and marshlands. The main source of SO_2 is anthropogenic and has a few natural sources, hence its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

SO_4^{2-} is one of the main components of atmospheric particles. The most amount of SO_4^{2-} is found in urban atmosphere, but in some forests SO_4^{2-} is higher than urban area due to transferring from urban area to forest atmosphere.

Gaseous phase of atmospheric H_2S is started as reaction with hydroxyl radical ($\text{OH}\cdot$) and continues to form SO_2 .



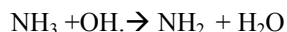
SO₂ in atmosphere reacts with hydroxyl radical and produces H₂SO₄.



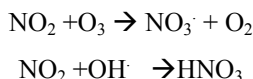
Produced H₂SO₄ forms the condensation nucleus for atmospheric water vapor and creates visible Sulfate aerosols existent throughout the troposphere (7).

Mineral Nitrogen compounds

N₂O and NH₃ emission source is high temperature combustion by Power-plants, vehicles, forest fire and solid waste burning. Gaseous Ammonia react slowly with hydroxyl radical and produces Nitrogen radical.



Probably this radical reacts with Ozon (O₃) and finally in a chain reaction lead to form Nitrogen Oxides. Solubility of NH₃ in water is high hence most NH₃ is decreased which is more than atmospheric gaseous phase disappearance. NO₂ can be oxidized both to Nitrate radical and to Nitric acid.



O₃ and Nitrate radical are two active oxidant compounds inside the atmosphere which are responsible for decreasing half-life of some rare atmospheric gases neighborhood of urban area (9).

H₂S forms the main biological source of atmospheric Sulfur. With increasing amounts of SO₂ and NO_x, in case of more oxidation it falls down as acid with rain and snow, and in case of shortage of this gases also more existence of heavy metals like Cu, Zn, Cd, Pb, Hg, Ni and alkaline ions, led to neutralizing the atmospheric acidity and cause pH reach to higher pH (10).

Nitrate and Ammonium

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

Average amount of Nitrate based on mg/lit is increased parallel with increasing elevation from sea level. Average NO₃⁻ concentration at 300 meters altitude with amount of 27 mg/lit reach to 46 mg/lit at 1000 meters altitude due to anthropogenic emission.

Natural increase amount of NO₃⁻ from 300 meters elevation to 1000 and 2200 meters elevation is 20% increased in concentration. Annual wet deposition of NO₃⁻ in altitudes of 300, 1000, 1600 and 2200 meters from sea level is 250, 275, 93 and 104 kg/ha.yr respectively. The main reason for decreasing annual wet deposition of NO₃⁻ from 300m to 1000m and 2200m is referred to varying rainfall in different elevations. Rainfall in 300m is 900 mm, in 1000m and 1600m is 600mm and in 2200m is 300mm precipitation per year. Consequently with decreasing rainfall, wet deposition of NO₃⁻ in 1600 meters reach to one third. Wet deposition of NO₃⁻ in 1000 meters altitude increases rapidly because concentration of NO₃⁻ in 1000 meters increases, although precipitation per year which in 300 m is 900 mm, decreases to 600 mm precipitation in 1000 meters altitude, which has anthropogenic origin.

Acknowledgement

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

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Abstract

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

Introduction

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

Methods

In this study controlled experimental burns of wheat stubble fields and Kentucky Blue Grass fields were performed in Eastern Washington and Idaho, United States, respectively. Wheat field burns were performed both in the Spring and Fall season of 2000 using two fuel loading groups (high and low), and four different patterns of fire ignition. Fire ignition patterns consisted of mass ignition, the conventional method of ignition, as well as backing fire (wind direction opposite to direction of fire front), head fire (wind direction same as direction of fire front), and strip fire (similar to head fire, but ignited in strips). The Kentucky Blue grass experiments, performed in Summer 2001 tested for PM_{2.5} emissions in high and low-loading (baled) fields, on both irrigated and dry land farms. In all treatments pre-burn measurements included residue loading, and residue and soil moisture content. Post-burn measurements included residue loading. During the burns meteorological variables were recorded, and video recordings were taken to evaluate the development of the fire during the analytical phase. Smoke emissions were continuously

samples at two FASS units (Fire Atmosphere Sampling Systems; Susott et al., 1991), and analyzed by the USFS Fire Sciences Laboratory, Missoula, Montana. Based on smoke emissions and fuel consumption, pollutant emission factors were calculated for CO₂, CO, CH₄, and particulate matter less than 2.5 micrometer in diameter, PM_{2.5}.



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

Results and Discussion

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

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

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

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Abstract

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

Introduction

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

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

Methods

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

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



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

Results and Discussion

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

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

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

< detected, but lower than the limit of quantification (LOQ). 0 not detected.

BF – before biological filter. AF – after biological filter.

* 5% significance level. ** 1% significance level. *** 0.1% significance level.

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

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

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

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

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

Conclusions

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

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

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Abstract

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

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

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

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

Materials and Methods

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

Short Description of the AODM

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

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

Calculating Sensation and Separation Distance

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

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

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

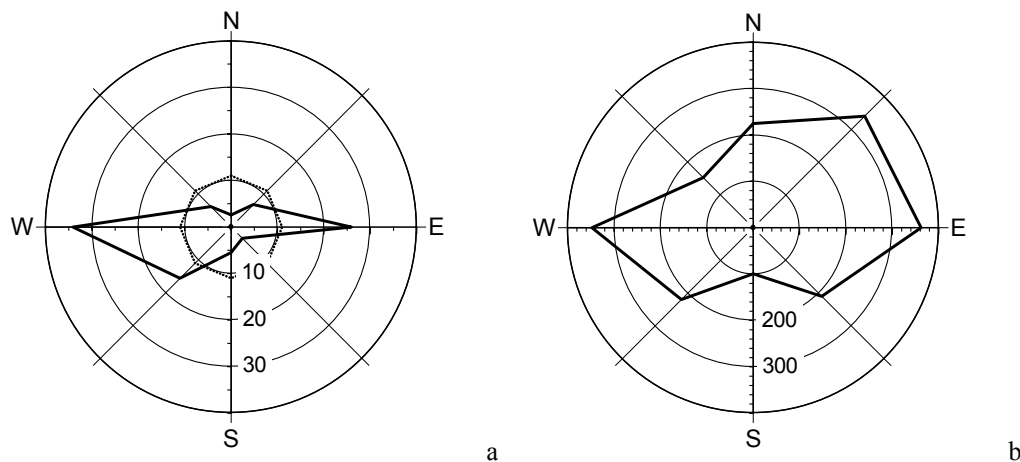


Figure 1. Polar diagram of (a) relative frequency distribution of the wind direction at Wels for eight 45° sectors. Calm conditions according to the ÖNorm M 9440 (1992/1996) with wind velocity less than 0.7 m/s evenly distributed over all directions (dotted line) and (b) separation distances (in m) calculated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

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

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

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

¹ Odour concentration threshold (OU/m³) / exceeding probability p (%)

² The land use category varies the accepted protection level

Meteorological Conditions

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

Discrete stability classes have been determined based on sun elevation angle, cloud cover and low cloud base height, and wind speed (Reuter, 1970). The cloud data are measured at the Linz-Hörsching airport, about 13 km from Wels. Within the Reuter scheme, classes 2 to 7 can occur in Austria. Stability classes 2

and 3, which by definition occur only during daylight hours in a well-mixed boundary layer, class 3 allowing also for cases of high wind velocity and moderate cloud cover, occur in 26% of all cases. Stability class 4, representing cloudy and/or windy conditions including precipitation or fog, occurs day and night (43 %). Class 5 occurs with higher wind velocity during nights with low cloud cover, a situation which is not observed frequently at Wels (6 %). Classes 6 and 7 are relevant for clear nights, when a surface inversion, caused by radiative cooling, traps pollutants near the ground. Such situations occur in 25% of all cases.

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

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

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

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

Results

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

For a certain odour threshold (e.g. 1 OU/m³ in Fig 2a) the sensitivity shows a minimum for the prevailing wind directions East, South-West, and East. The sensitivity of these wind directions lies between 7 and 12 m/%. The other directions this gradient is about 3 to 4-fold higher. For the highest protection level (1 OU/m³ / 1%) the coefficient of variation was lowest with about 36%. This means that the separation

distance S doesn't change a lot with the direction. With growing exceedance probability the direction-dependent variability of the separation distances increases.

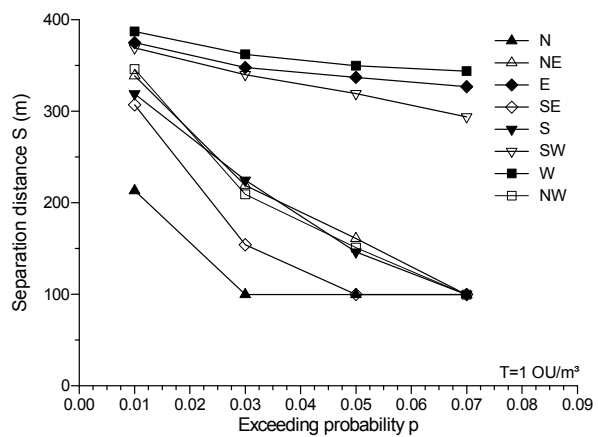
The influence of the exceedance probability is reduced for higher odour thresholds (Fig. 2b and 2c). For an odour threshold of 5 OU/m³ the sensitivity for the three prevailing wind directions is about 3 to 8m/% and about the 2-fold for the other wind directions which are more related to the valley wind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

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

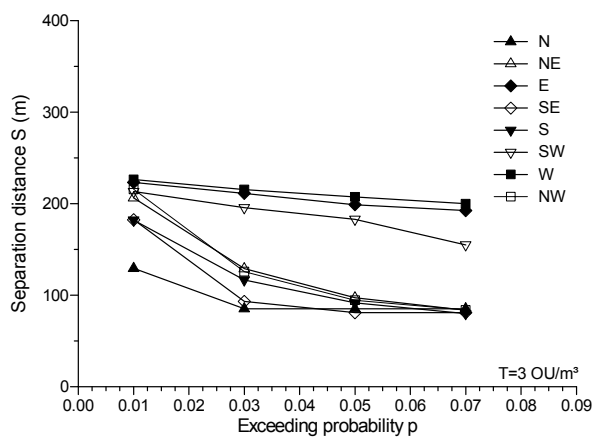
Winddirection	$\Delta S / \Delta p$ (m / %)		
	Odour threshold T (OU/m ³)		
	1	3	5
N	-17	-7	-7
NE	-39	-20	-17
E	-8	-5	-5
SE	-34	-16	-15
S	-37	-17	-15
SW	-12	-9	-8
W	-7	-4	-3
NW	-40	-21	-19

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

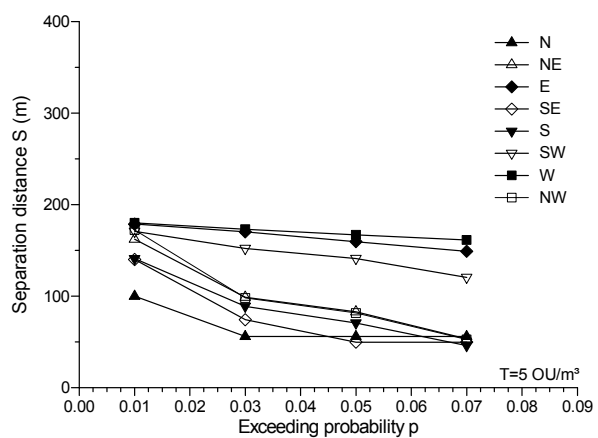
Winddirection	$\Delta S / \Delta T$ (m / OU/m ³)			
	Exceeding probability p (%)			
	1	3	5	7
N	-28	-11	-11	-11
NE	-44	-30	-20	-11
E	-49	-44	-44	-44
SE	-42	-20	-13	-13
S	-44	-34	-19	-13
SW	-50	-47	-45	-43
W	-52	-47	-46	-46
NW	-43	-28	-17	-12



a

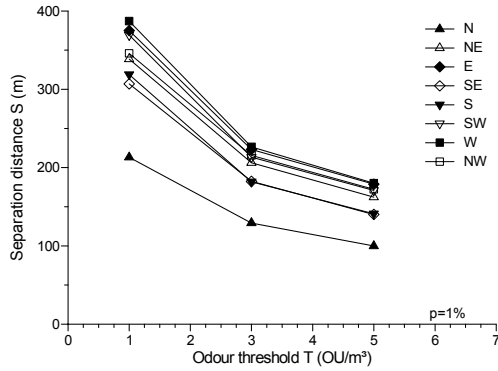


b

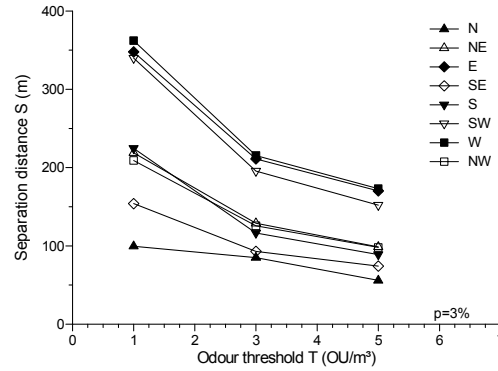


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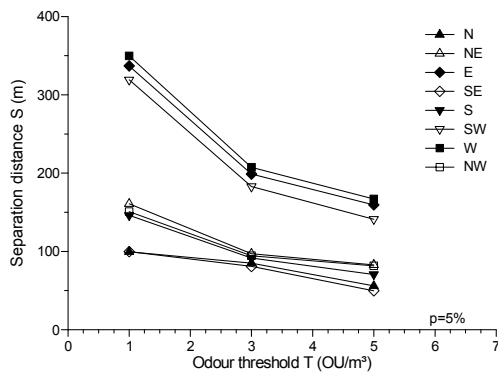
Figure 2. Sensitivity of the separation distance S (m) as a function of the exceeding probability p for various odour thresholds T of 1 OU/m² (a), 3 OU/m² (b), and 5 OU/m² (c).



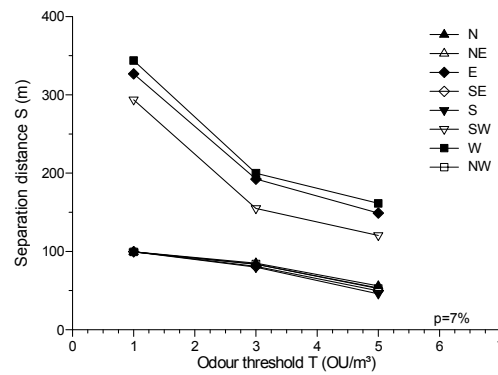
a



b



c



d

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

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

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

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

Discussion

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

The definitions of the various national odour impact criteria differ to quite an extent. Miedema and Ham (1988) and Miedema et al. (2000) found a strong relationship between the 98 percentile of the odour concentration and the percentage of the highly annoyed neighbours. They used an ambient odour concentration for an integration time of 1 hour, calculated by a dispersion model without consideration of the peak-to mean ratio. In Germany, the odour impact criterion is defined by an exceedance probability of 10% for a threshold of 1 OU/m³. To apply this odour impact criterion, the calculated odour concentration (one hour mean value of the regulatory dispersion model) is multiplied by a constant factor of 10 by using the Gaussian dispersion model or the factor 4 for the recently publishes Lagrange models (AUSTAL 2000G).

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

Further on, the perception of the odour intensity goes with the logarithm of the odour concentration according to the Weber-Fechner law (e. g., Misselbrook et al., 1993). Based upon laboratory-based experiments on perceived intensity, the Environment Agency, UK (2002), defines: 1 OU/m³ is the point of detection, 5 OU/m³ is a faint odour, and 10 OU/m³ is a distinct odour. The discrepancy between the definition of 1 OU/m³ in the laboratory by using odour free air and the situation in the field was solved by introducing the sniffing unit (van Langenhove and van Broeck, 2001; Defoer and van Langenhove, 2003).

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

Conclusions

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

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

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Measuring Hydrogen Sulfide Emissions from Manure Samples in a Laboratory Setting

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Abstract

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

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

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



Figure 1. Experimental Setup Photo.

The observed mean H₂S air concentrations and emission rates for six airflow rates are:

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

After an equilibration period of 10 min, H₂S concentration increased in the airflow. For all airflow rates, except 0.34 L/min, the H₂S concentration stabilized and became fairly constant after 40 min. Then the H₂S concentration slowly decreased from 50 min to the end of data collection. For the 0.34 L/min rate, the H₂S concentration showed a distinct peak at 50 minutes with a maximum value of 2.10 ppm before decreasing at a much faster rate than the other five airflow rates. There is a decrease in H₂S concentration with the diluting affect of increased airflow rate. Note that the mean H₂S concentration collected between 50 and 100 min is approximately twice the mean concentration collected a time greater than 400 min.

Emission rates increased during the initial period (first 30-50 minutes), followed by a slow decay curve. This initial increase and subsequent decrease could not be explained with the amount of data taken but does show proof of concept for the experimental method. The emission information takes into consideration the airflow rate in the calculation of emission rates. Preliminary observation indicates the emission rate might level off or become independent of the airflow rate at about 3 L/min, which may indicate a shift from transfer rate limitation across the surface to H₂S generation within the manure for determining the overall mass transfer rate.

In a second laboratory study, part of a single swine manure sample was used at full strength (100%), labeled Strong and part of the manure sampled was diluted with distilled water (80% water:20% manure), labeled Weak for a Strong:Weak ratio of 5:1 ratio. Adjusting the Weak sample (20% dilution) data by multiplying the observed concentrations and emissions by a factor of five resulted in significant differences (p -value ≤ 0.05) compared to the Strong sample (100% dilution) data. This indicates a non-linear or an interaction of manure strength with airflow rate and/or the air collection process and final H₂S concentrations and emission rates.

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



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

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Abstract

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



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

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Abstract

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



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

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Abstract

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

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

Introduction

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

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

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

compartments, thus allowing the comparison of dust concentrations between the different poultry housing systems.

Methods

Livestock Building

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

Measurements

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

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

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

Data Processing

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

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

Results and Discussion

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

terms of their light scattering properties and therefore their final size classification. However, recent investigations with a photo-optical instrument (similar to what was used in this study) have shown that a correction factor could help to re-adjust the predicted dust concentrations to correspond with gravimetrically obtained results (Taylor and Reynolds, 2001).

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

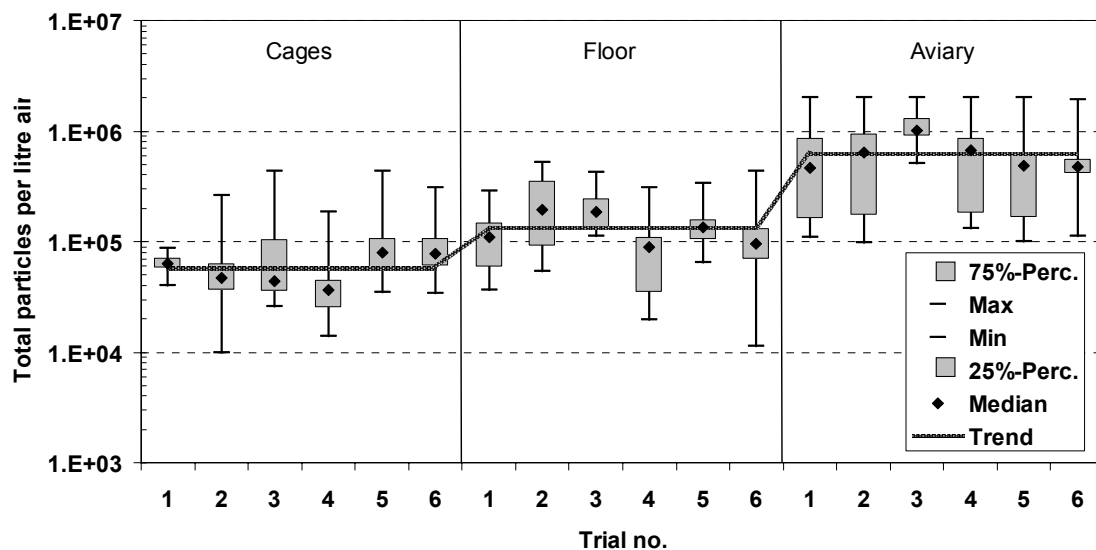


Figure 1. Median total particle concentrations in the three different compartments for laying hens. Incorporated trend line indicates the overall differences in concentrations.

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

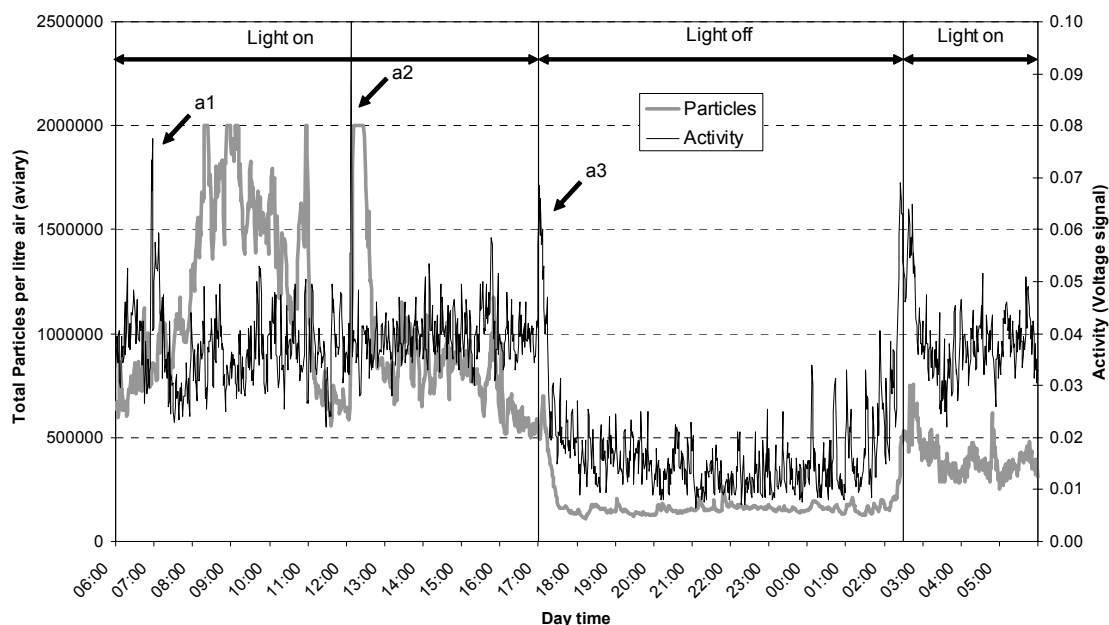


Figure 2. Example of a time-series of total particle concentration and monitored activities in the aviary system. Identified activities during the survey are marked with arrows (a1 to a3 = control walks).

Conclusions

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

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

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

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Abstract

Ammonia and hydrogen sulfide flux from a dairy manure storage pond were measured for 35-days on a 1,800-head dairy in south central Idaho. Continuous air samples were taken using a standardized wind tunnel floating on the pond and gas concentrations were averaged every 15-minutes using chemiluminescence. Diurnal variations of ammonia were found to be strongly correlated to solar radiation and temperature while variations of hydrogen sulfide were found to remain relatively constant. Average flux rates of 101.50 ug/m²/s and 7.58 ug/m²/s were measured for ammonia and hydrogen sulfide, respectively. Average emission rates for ammonia and hydrogen sulfide per milking cow were found to be 0.274 lbs/cow/d and 0.021 lbs/cow/d, respectively.



Ammonia-Based Air Permits for Idaho Dairies

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Abstract

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

Introduction

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

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

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

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

Who does the Rule effect?

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

excretion, the IDA and ICL took into account their best professional judgment to account for differences in ammonia losses from various dairy housing types, manure storage systems, and land application practices.

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

Table 1. Animal Unit (AU) or mature cow thresholds to produce 100 tons NH₃/year.

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

¹ Assumes: Expected level of N->NH₃ volatilization for: drop-hose or ground level liquid manure application

² Assumes: Expected level of N->NH₃ volatilization for: center pivot or other conventional sprinkler irrigation liquid manure application

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

Rule Requirements

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

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

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

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

During the development of this Rule, various manure treatment systems and handling practices were evaluated for their effectiveness in reducing ammonia emissions (Table 2). Many scientific studies, extension bulletins, NRCS handbooks and EPA guidance documents were referred to in preparing the

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

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

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

Conclusions

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

Table 2. Ammonia Control Practices for Idaho Dairies.

System	Component	Ammonia Control Effectiveness ¹			Compliance Method ³
		Open Lot	Freestall Scrape	Freestall Flush	
Waste Storage and Treatment Systems	Synthetic Lagoon Cover	15	20	20	1
	Geotextile Covers	10	13	13	1
	Solids Separation	3	3	3	3, 4
	Composting	4	4	4	1
	Separate Slurry and Liquid Manure Basins	6	10	-	1
	In-House Separation	0	12	0	1
	Direct Utilization of Collected Slurry	6	10	-	1, 3, 4
	Direct Utilization of Parlor Wastewater	10	10	10	1
	Direct Utilization of Flush Water	8	0	13	3, 4
	Anaerobic Digester	-	-	-	-
	Anaerobic Lagoon	-	-	-	-
	Aerated Lagoon	10	12	15	2
	Sequencing-Batch Reactor	15	20	20	2
	Lagoon Nitrification/Denitrification Systems	15	20	20	2
	Fixed-Media Aeration Systems	15	20	20	2
General Practices	Vegetative or Wooded Buffers (established)	7	7	7	1
	Vegetative or Wooded Buffers (establishing)	2	2	2	1
	Alternatives to Copper Sulfate	-	-	-	-
Freestall Barns	Scrape Built Up Manure	-	3	3	1
	Frequent Manure Removal	UD	UD	UD	-
	Tunnel Ventilation	-	-	-	-
	Tunnel Ventilation w/Biofilters	-	10	10	1
	Tunnel Ventilation w/Washing wall	-	10	10	3, 4
Open Lots and Corrals	Rapid Manure Removal	4	2	2	1, 2
	Corral Harrowing	4	2	2	1
	Surface Ammendments	10	5	5	2
	In-Corral Composting / Stockpiling	4	2	2	1
	Summertime Deep Bedding	10	5	5	1
Animal Nutrition	Manage Dietary Protein	2	2	2	2
Composting Practices	Alum Incorporation	12	8	6	2
	Carbon:Nitrogen Ratio (C:N) Ratio Manipulation	10	7.5	5	2
	Composting with Windrows	-	-	-	-
	Composting Static Pile	6	4.5	3	1
	Forced Aeration Composting	10	7.5	5	1
	Forced Aeration Composting with Biofilter	12	8	6	1
Land Application²	Soil Injection - Slurry	10	15	7.5	2
	Incorporation of manure within 24 hrs	10	10	10	2
	Incorporation of manure within 48 hrs	5	5	5	2
	Nitrification of lagoon effluent	10	10	15	3, 4
	Low Energy/Pressure Application Systems	7	7	10	1
	Freshwater Dilution	5	8	8	1, 2
	Pivot Drag Hoses	8	8	10	1
	Subsurface Drip Irrigation	10	10	12	1

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

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

³ Method used by inspector to determine compliance:

1=Observation by Inspector

2=On-Site Recordkeeping Required

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



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

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Abstract

Current estimates indicate that 25-30% of the total nitrogen loading to Chesapeake Bay is derived from atmospheric input¹. In regions having intensive agricultural livestock activities, a large and increasing fraction of the nitrogen deposition is in the form of NH_x ($\text{NH}_3 + \text{NH}_4^+$); in some locations, it has been shown to be comparable to NO_3^- deposition^{2, 3}. Despite this importance, NH_3 emission estimates are poorly constrained. A major reason is because NH_3 emission estimates rely on extrapolation of emission factors, which are based on many assumptions, and can be highly variable depending on the specific animal husbandry and manure disposal practices^{4, 5}. In particular, many of the published emission factors are derived from studies conducted in Europe, where growing methods differ considerably from those practiced in the U.S.

Various approaches have been used to quantify NH_3 emissions from confined animal feeding operations, each with its own inherent limitations and level of uncertainty. We have developed a simple, novel, inverse plume dispersion modeling technique that employs gas-phase NH_3 concentrations obtained from a three-dimensional array of ~20 Ogawa passive samplers, for determining NH_3 emissions from poultry houses⁶. The passive samplers are deployed at 3 heights on a series of 10 m sampling towers, strategically situated in an array downwind from the house. Compared with traditional methods of NH_3 sampling (e.g., denuders, gas scrubbers) passive samplers are ideally suited for this purpose since they are inexpensive, robust, do not require electricity or other supporting equipment (e.g., pumps, mass flow controllers), and can be left unattended for extended periods. The samplers are typically deployed for 6-12 hr periods over the various stages of the grow-out cycle, and under differing meteorological regimes. A Lagrangian-based, inverse Gaussian dispersion plume model is fitted to the observed NH_3 concentration profile to determine the corresponding point source strength (i.e., total NH_3 emissions from the house). Normalizing this value to the number of birds present in the house, allows one to calculate an emission factor corresponding to each sampling period (g NH_3 -N/bird/day), which can be integrated to yield an emission factor for the entire flock. The primary advantage of this approach is that it allows for the determination of NH_3 emissions at various growth stages under “real world” growing conditions.

Our initial study involved an 11,500 bird, side-wall ventilated broiler house⁶. A total of 7 field deployments were performed at differing growth stages of two different summertime flocks, under both day and night meteorological conditions. Observed NH_3 emission factors ranged from 0.27 to 2.17 g NH_3 -N/bird/day, with an arithmetic mean of 1.18 g NH_3 -N/bird/day and flock-weighted mean of 0.74 g NH_3 -N/bird/day.

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

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Effect of Urease Inhibitor on Ammonia Emissions in Layer Houses

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Extended Abstract

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

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

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

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

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

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

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



Wet and Dry Deposition Studies at Remote, Agricultural and Urban Site of Semi-arid Region of India

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Abstract

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

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

Introduction

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

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

Experimental

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

Results and Discussion

Wet Deposition

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

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

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

Chemical Composition of Rainwater in India

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

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

$$\text{NF}_{\text{NH}_4} = \frac{[\text{NH}_4]}{2 [\text{SO}_4] + [\text{NO}_3]}$$

$$\text{NF}_{\text{Ca}} = \frac{[\text{Ca}]}{[\text{SO}_4] + 2 [\text{NO}_3]}$$

$$\text{NF}_{\text{Mg}} = \frac{[\text{Mg}]}{[\text{SO}_4] + 2 [\text{NO}_3]}$$

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

Ionic Correlation and Factor Matrix

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

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

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

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

Dry Deposition

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

Seasonal Variation in Dry Deposition Rates

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

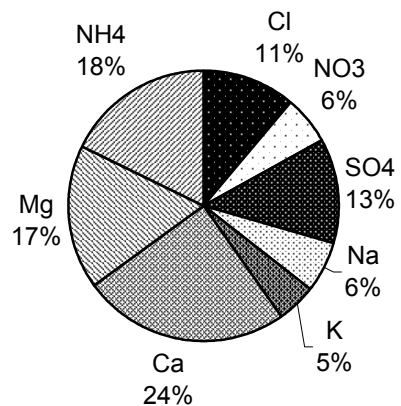
Critical Load (CL)

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

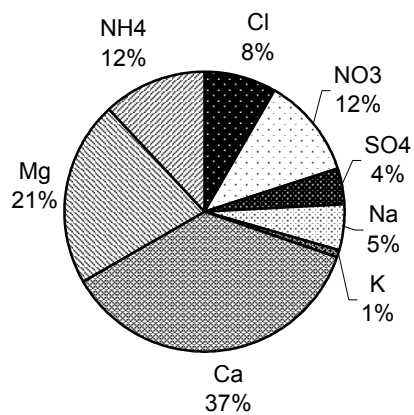
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Rampur- Rainwater composition



Gopalpura Rainwater composition



Agra- Rainwater composition

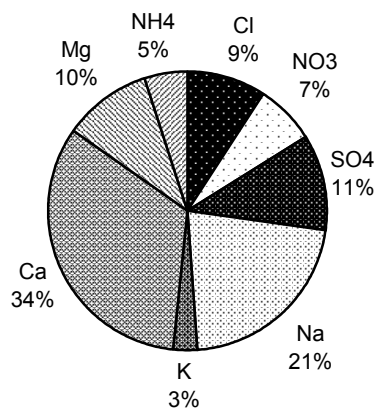


Figure 1. Chemical composition of rainwater at different sites.

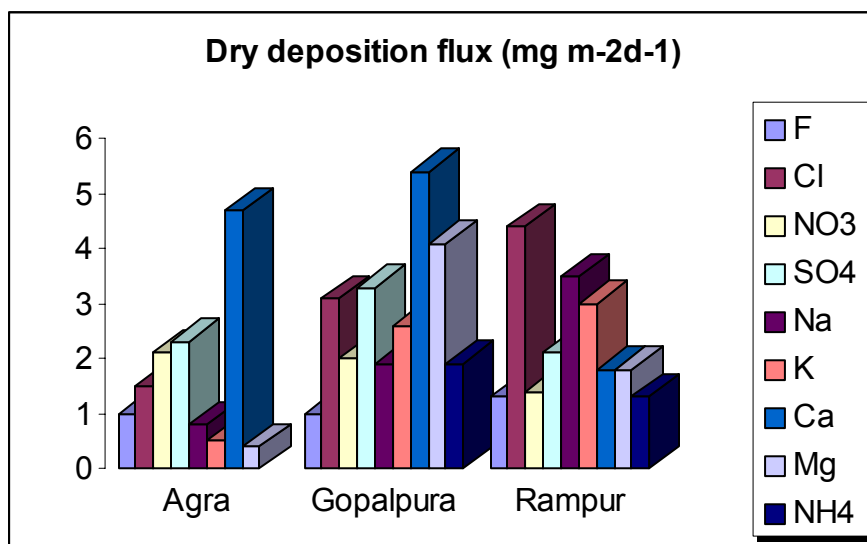


Figure 2. Dry deposition flux at different sites.

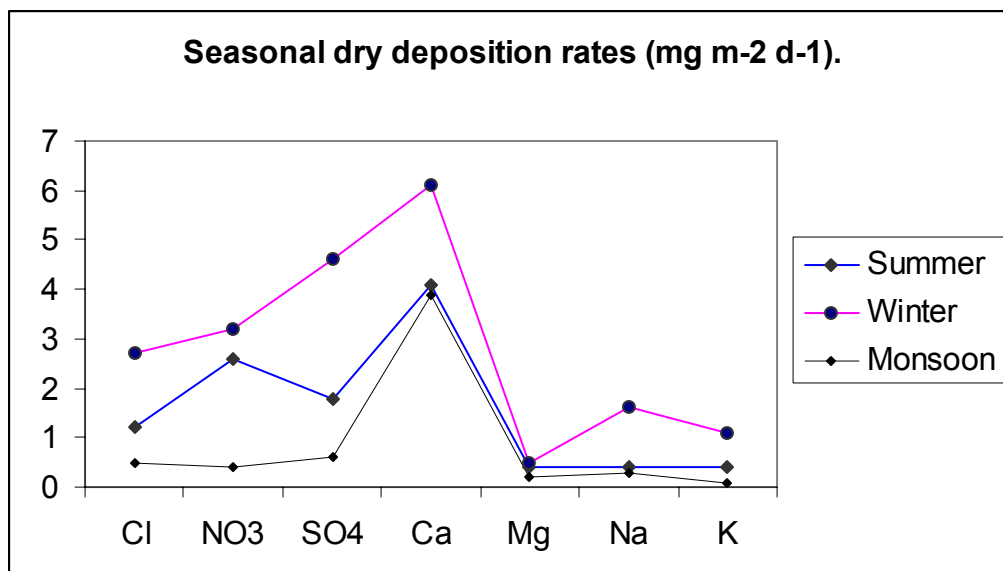


Figure 3. Seasonal Variation of dry deposition flux at different sites.



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

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Abstract

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

Influence of Agricultural Ammonia Emissions on Particulate Matter in Switzerland

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Abstract

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

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

Introduction

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

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

Methods

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

Process Based Modeling

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

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

Artificial Neural Networks

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

$$\text{PM}_{10} = f(\text{NO}_x, \text{SO}_2, \text{TA}, \text{TN}, \text{SO}_4, \text{T}, \text{H}).$$

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

Results

Thermodynamic Equilibrium Model

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

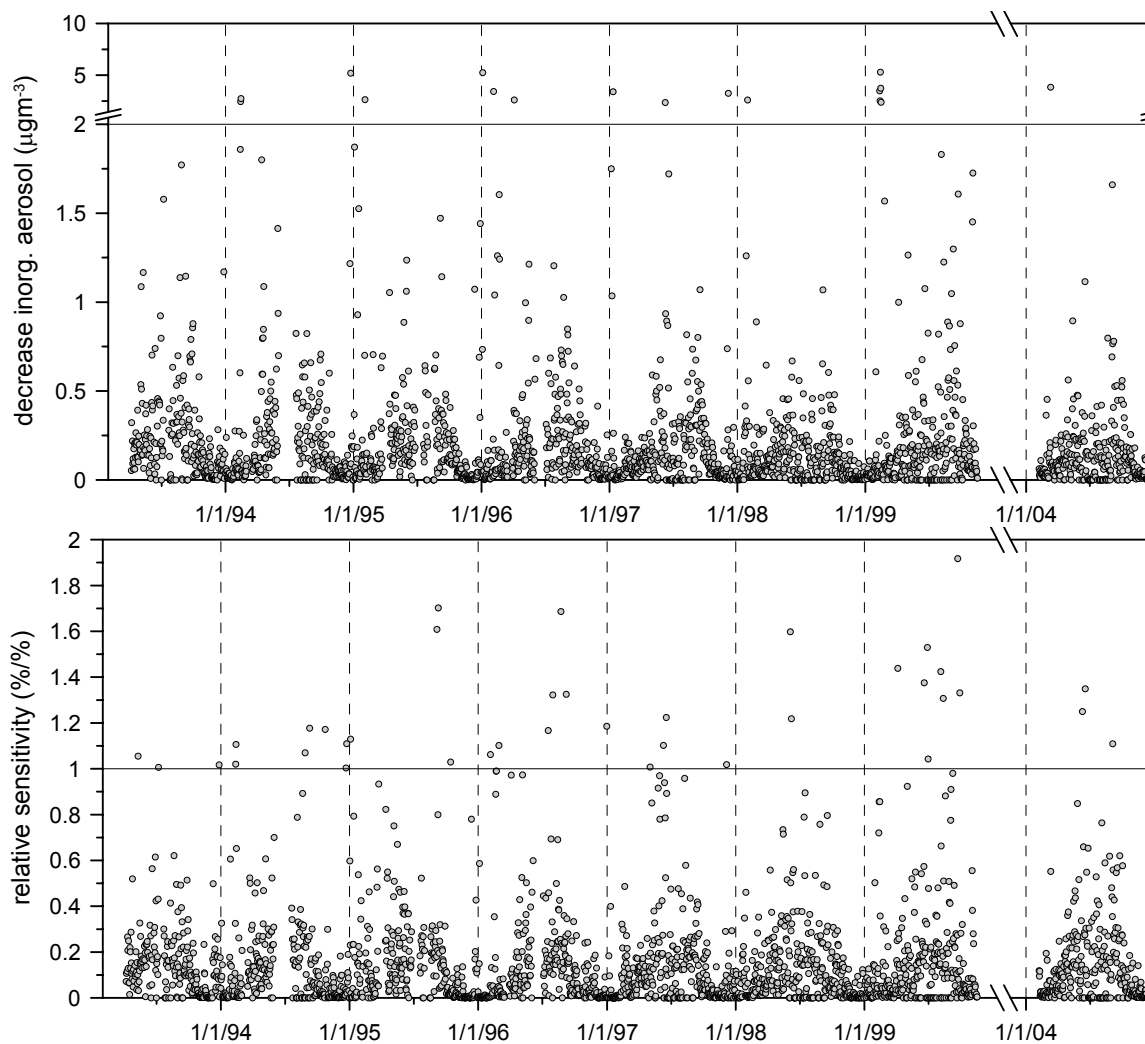


Figure 1. Decrease of inorganic aerosol mass after 10% reduction of total ammonium; absolute decrease, above, and relative sensitivity, below.

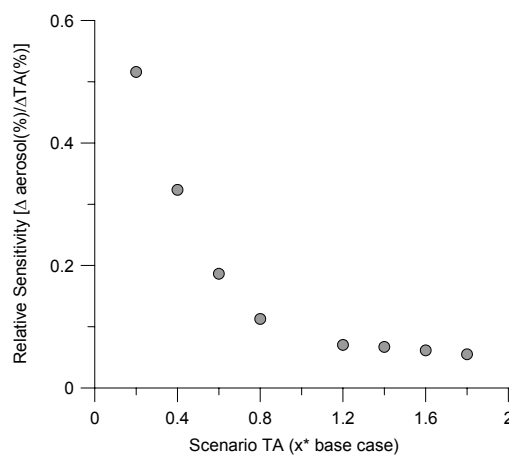


Figure 2. Relative sensitivities of inorganic aerosol mass towards reductions ($0.2 \leq x \leq 0.8$) and increases ($1.2 \leq x \leq 1.8$) of total ammonium.

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

Artificial Neural Network

Figure 3 shows the comparison between measured PM10 and neural network output. The trained network is obviously capable of simulating PM10 as a function of NO_x, SO₂, temperature, humidity, TA, TN, and sulfate.

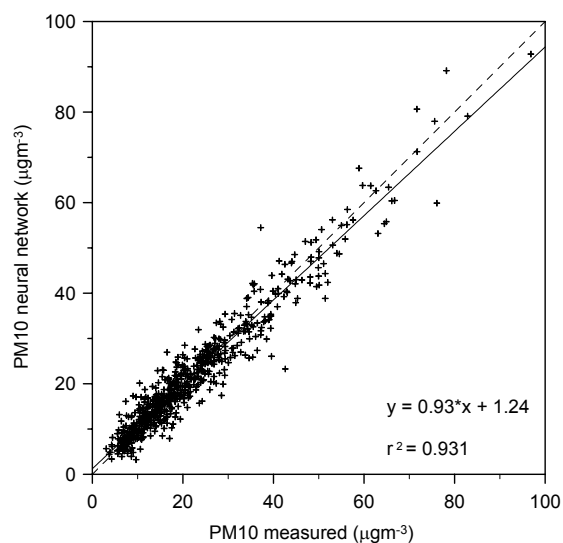


Figure 3. Comparison of measured PM10 with neural network output.

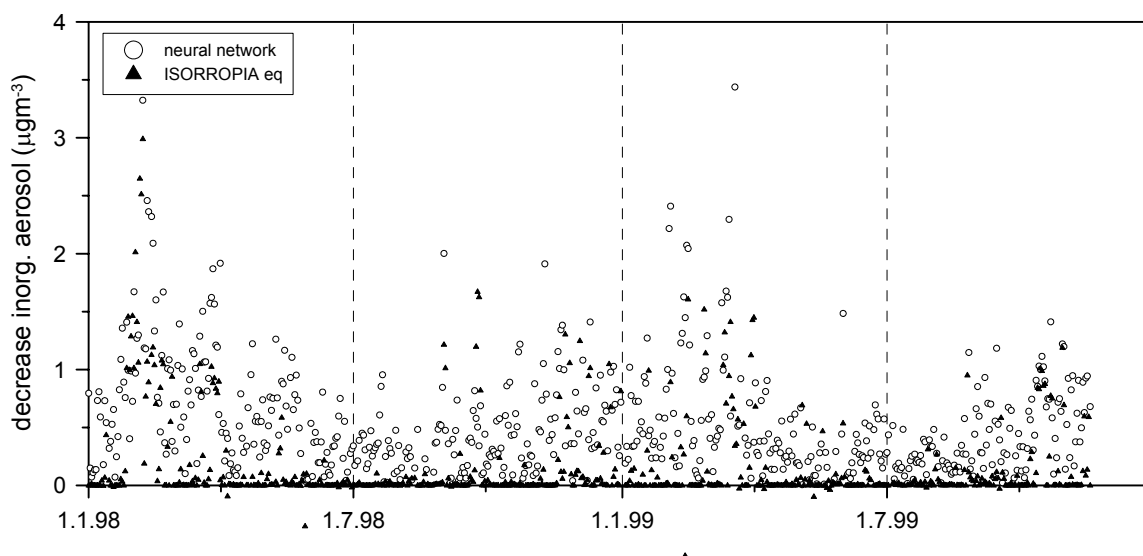


Figure 4. Decrease in inorganic aerosol mass after 10% reduction of TA as predicted by the neural network. ISORROPIA calculations shown here were performed under identical conditions as in PM10 measurements, i.e. equilibrated at 50% rel. humidity and 295 K.

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

Emission Reduction Scenarios

The change of total ammonia and total nitrate concentrations after reductions in ammonia and NO_x emissions was calculated with the 1-D photochemical model. Table 1 summarizes the change in TA concentrations on the 2nd day of the model run for three reduction scenarios 25-75%. The sensitivity of the inorganic aerosol mass to NH₃ emission reductions was obtained by combining these 1D model results with the sensitivities of inorganic aerosol mass to changes in total ammonia. The qualitative behavior of the inorganic aerosol mass to changes in ammonia emissions is similar as the one found for the relationship between aerosol mass and total ammonia: the sensitivity is higher with increasing reductions. Small reductions in ammonia emissions will only cause negligible decreases of particulate matter.

Table 1. Results of 1D-model; reduction of total ammonium and inorganic aerosol after NH₃ emission changes.

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

Discussion and Conclusions

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

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

Technical and operational measures in livestock farming have the potential to reduce ammonia emissions in Switzerland by about 30%. Given the sensitivities of the aerosol mass found here, the potential for reductions needs to be fully exploited to cause measurable decreases in PM₁₀.

On the other hand, ammonia reductions in this NH₃-saturated regime will cause nearly proportional decreases in gas phase ammonia. NH₃-emission controls thus offer a greater handle in reducing nitrogen deposition than particulate matter.

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

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Abstract

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



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

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Abstract

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

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

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Abstract

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

Introduction

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

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

Materials and Methods

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

Table 1. Main characteristics of the three production units.

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

In unit 1, a full-scale wastewater treatment system was started in 2003 to treat all raw manure produced. The treatment system combined solid-liquid separation with removal of N and P from the liquid phase. The system treated raw manure flushed from the barns in three steps (Vanotti et al., 2006). The first step flocculated solids from raw flushed manure using polyacrylamide. This step produced separated solids that were transported off-site and converted to organic plant fertilizer, soil amendments, or energy at a centralized facility. In the second step, N management to reduce NH₃ emissions was accomplished by passing the liquid through a module where immobilized nitrifying bacteria transformed NH₃ into nitrate. Subsequent alkaline treatment of the wastewater in a P module precipitated calcium phosphate and killed pathogens. The treated water was recycled to refill the barn pit recharge system, and excess water was stored in the lagoon and later used for crop irrigation. As the treatment system recovered the manure solids and replaced the anaerobic lagoon liquid with cleaner water, it converted the anaerobic lagoon in unit 1 into a treated water pond (hereafter called lagoon 1 or converted lagoon). Unit 2 remained with the traditional anaerobic lagoon treatment and was used as a control (hereafter called lagoon 2 or traditional lagoon).

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

During 2004, NH₃ emissions were measured in all three lagoons. Ammonia emissions were determined with passive flux samplers using the method of Sommer et al. (1996). The passive samplers were placed at four fixed locations perpendicular to each other around each lagoon. This layout enclosed most of the lagoon surface within a circular sampling plot (figure 1). Nine data collection periods lasting 23 h each were scheduled from February to November 2004 for the three lagoons. Further details on NH₃ emission measurements using the passive flux method are described in Szogi et al. (2006).

Even though waste treatment in all three production units was substantially different, animal production management remained the same during the study. Total annual NH₃ emissions were obtained by fitting a Gaussian curve to daily NH₃-N emission data versus day of the year using Prism 4.0 software (GraphPad Software, Inc., San Diego, CA). All water analyses were performed according to Standard Methods for the Examination of Water and Wastewater (APHA, 1998).

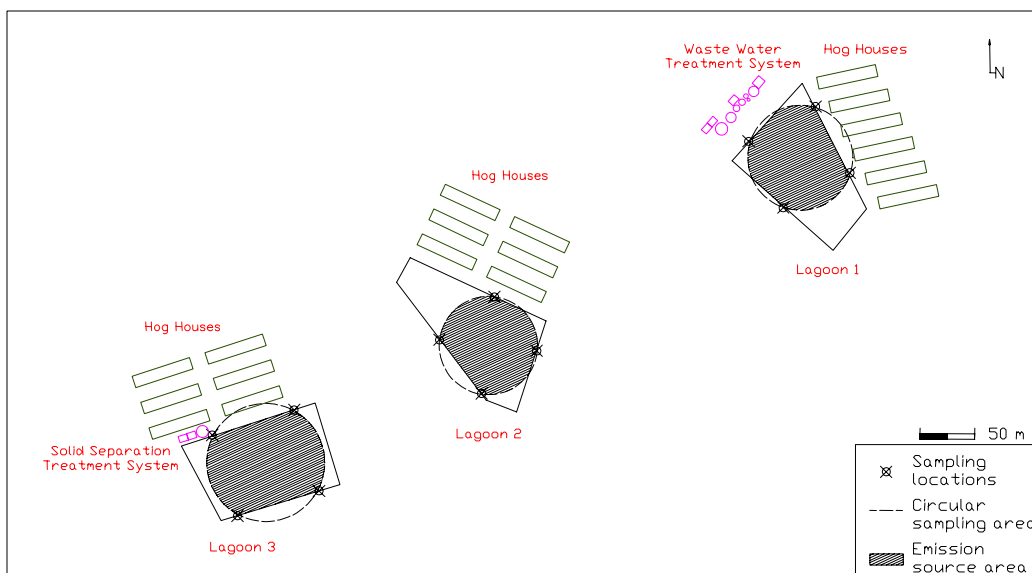


Figure 1. Schematic diagram of monitoring set-up for ammonia emission study for converted lagoon 1, traditional lagoon 2, and traditional lagoon 3 with solids separation, Duplin Co, NC.

Results and Discussion

Lagoon Water Quality

Monitoring of water quality was initiated in 2002, one year before the wastewater treatment plant started to operate in unit 1. The criterion to determine water quality improvement was the reduction in concentration of various water quality indicators (table 2). In 2002, the three lagoons were receiving flushed manure from the barns. Thus, the three anaerobic lagoons had similar annual mean pH, total ammoniacal N (TAN), total Kjeldahl N (TKN), and nitrate plus nitrite ($\text{NO}_2 + \text{NO}_3\text{-N}$) concentrations (Jan.-Dec. 2002; table 2).

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

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

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

Sampling Period	Lagoon	Treated	pH	TAN ^[2] (mg/L)	TKN (mg/L)	NO ₂ + NO ₃ (mg/L)	TS (g/L)
Jan.-Dec. 2002	1	N	8.0 (0.1)	464 (98)	506 (108)	0.08 (0.20)	4.2 (1.6)
	2	N	8.0 (0.2)	467 (118)	521 (122)	0.07 (0.21)	3.5 (0.9)
	3	N	8.0 (0.2)	469 (121)	517 (115)	0.07 (0.18)	3.6 (0.8)
Jan.-Dec. 2003	1	Y	8.1 (0.1)	186 (129)	230 (138)	4.1 (5.8)	2.7 (0.5)
	2	N	7.9 (0.1)	446 (102)	522 (127)	0.43 (1.4)	3.2 (0.7)
	3	N	7.9 (0.1)	375 (124)	439 (140)	0.41 (1.4)	2.8 (0.4)
Jan.-Dec. 2004	1	Y	8.1 (0.3)	37 (32)	76 (34)	20 (16)	2.3 (0.2)
	2	N	8.0 (0.2)	364 (88)	406 (79)	n.d. ^[3]	2.9 (0.2)
	3	Y	8.1 (0.2)	344 (141)	391 (140)	n.d.	2.3 (0.4)

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

^[2] TAN = Total ammoniacal N; TKN = Total Kjeldahl N; NO₂ + NO₃ = Nitrite plus Nitrate; TSS = Total suspended solids.

^[3] n.d. = not detected.

Reduction of Ammonia Emissions

Total annual NH₃ emissions in lagoons 1, 2 and 3 were calculated by fitting a Gaussian distribution to measured daily NH₃ emission values (figure 2). This curve was selected because it provided a good fit to the changes in daily NH₃ emissions throughout the year both in terms of R² and normality of residuals (residuals = emissions observed - emissions predicted) using the Shapiro-Wilk test statistic (Delong and Yuan, 1988). The total annual NH₃ emission for each lagoon is represented by the area under the curves in figure 2. On an annual basis, NH₃ emissions from the traditional lagoon (control) totaled 12,540 kg N/lagoon/yr (or 13,633 kg N/ha/yr) compared to 3,355 kg N/lagoon/yr (or 3,647 kg N/ha/yr) from the anaerobic lagoon 3 with solid separation treatment. Although water quality changes were modest in lagoon 3 with respect to the control (TAN, TKN and TS, Jan. – Dec. 2004, table 2), total annual NH₃ emissions reduction in lagoon 3 was 73% with respect to traditional lagoon 2. These results indicated that NH₃ emission reduction was more sensitive than the water quality to effect of the solid removal treatment. For the converted lagoon 1, reduction on annual NH₃ emissions was even larger with respect to traditional lagoon 2. Annually, NH₃ emissions from the converted lagoon totaled 1210 kg/lagoon/yr (or 1,311 kg N/ha/yr) compared to 12,540 kg N/lagoon/yr of the traditional lagoon. Lower N concentrations in the converted lagoon (lagoon 1, Jan.–Dec. 2004, table 2) substantially reduced annual NH₃ emissions by 90% with respect to those found in the traditional anaerobic lagoon. These results overall demonstrate that use of new wastewater technologies can substantially reduce ammonia emissions from confined swine production.

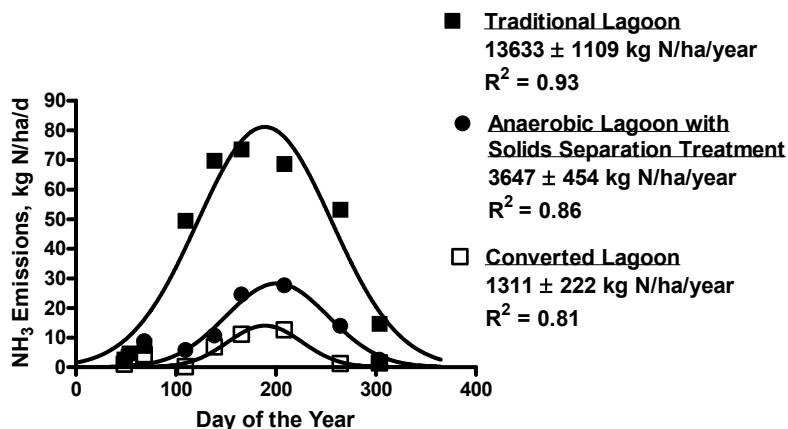


Figure 2. Change in daily rate of ammonia emissions throughout the year for the traditional lagoon, anaerobic lagoon with solids separation, and converted lagoon. The total annual ammonia (NH₃) emission for each lagoon system is represented by the area under each curve and indicated in the legend.

Conclusions

There is a need for treatment technologies that can effectively address environmental concerns associated with anaerobic lagoons. In particular, reduction of NH₃ emissions is a major environmental concern associated with confined swine production. In order to meet this need, two new wastewater treatment systems were demonstrated at full-scale in two of three 4,360-pig production units on a finishing farm in Duplin Co., NC. One treatment system combined solid-liquid separation with removal of N and P from the liquid phase, and the other wastewater treatment system consisted of solid-liquid separation before anaerobic lagoon storage. The third production unit was kept as a control using traditional anaerobic lagoon treatment.

In summary our findings indicate:

- 1) Ammonia emissions from the traditional anaerobic lagoon totaled 12,540 kg N/lagoon/year (13,633 kg N/ha/year). This result compares to lower NH₃ emissions of 3,355 kg N/lagoon/yr (or 3,647 kg N/ha/yr) from the anaerobic lagoon with solid-liquid separation, and 1,210 kg N/lagoon/yr (or 1,311 kg N/ha/yr) from the converted lagoon.
- 2) Although water quality improvements were modest in anaerobic lagoon with solid-liquid separation, total annual NH₃ emissions were reduced by 73% with respect to those of the traditional lagoon
- 3) Remarkable water quality improvements such as lower N concentrations in the converted lagoon substantially reduced annual NH₃ emissions by 90% with respect to those found in the traditional anaerobic lagoon.

Acknowledgements

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

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Abstract

Many recent nitrogen oxides (NO_x) control programs, including the EPA's NO_x SIP call and the latest CAIR programs, have adopted the "cap and trade" approach to mitigate regional NO_x emissions. An important premise of the cap-and-trade mechanism is that the marginal damages of NO_x emissions are equal across the trading domain. This paper examines the marginal ozone (O₃) damages on agricultural crops caused by NO_x emissions in different locations around the continental United States. We calculate O₃ damages using a new integrated assessment model that links pollution emissions to their chemical transformation and transport, to crops exposure, and to final impacts. We find that a fixed amount of increased NO_x emissions in a rural upwind state can cause crops losses more than 100 times as much as in a more urbanized Northeastern state. Although there are clear benefits to a "cap and trade" program which substantially reduces total emissions, the results of this paper, together with our previous studies on the marginal impacts of NO_x emissions on human health, raise questions about the fundamental assumption underlying the current "cap and trade" programs of NO_x emission control.



Air and Groundwater Pollution in Turkish Mediterranean Coast

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Introduction

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

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

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

The aim of this thesis can be summarized as follows:

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

Results and Discussion

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

$$\text{nss-SO}_4^{2-} = \text{t-SO}_4^{2-} - 0.139 [\text{Cl}^-],$$

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

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

Table 1 Summary of the statistics for the measured air pollutants (in $\mu\text{g m}^{-3}$)

<i>Parameter</i>	$\text{NH}_3(\text{g})$	NH_4^+	Cl^-	NO_3^-	SO_2 (g)	total SO_4^{2-}	nss- SO_4^{2-}	HNO_3 (g)	O_3
# of Samples	174	173	168	171	176	171	157	30	171
Mean	0.99	2.19	4.30	2.28	2.03	2.35	1.98	0.42	61.8
Median	0.64	1.99	2.01	1.94	1.97	1.87	1.46	0.35	61.9
Standard Deviation	1.08	1.17	6.94	1.77	1.06	2.01	1.87	0.28	14.54
Minumum	0.04	0.02	0.09	0.09	0.08	0.25	0.09	0.06	25.77
Maximum	6.44	6.08	58.3	8.94	5.89	12.5	11.4	1.33	98.56
Range	6.40	6.06	58.2	8.85	5.82	12.3	11.3	1.27	72.79

Table 5. OC pesticide concentrations in ground waters

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

Table 6. OP pesticide concentrations in ground waters

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

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

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Abstract

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



Mitigating Swine Odor with Strategically Designed Shelterbelt Systems: A Review

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Poster Abstract

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

Introduction

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

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

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

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

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

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

3) Physical interception of dust and other aerosols

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

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

- Volatile Organic Compounds (VOC's) –the offensive smell in livestock manure- have a distinct affinity to the lipophilic membrane (the cuticle) that covers plant leaves and needles. Studies are underway to examine the efficiency of various plants (Beattie et al., undated).
- Researchers have quantified measurable quantities of anthropocentric VOC's that have accumulated at the surface of plants (adsorption) and within the plants tissues (absorption) (Reischl et al., 1989; Reischl et al., 1987; Gaggi et al., 1985)

- Micro-organisms dominate the surface of plants (Preece and Dickenson, 1971). These organisms also adsorb and absorb VOC's and provide additional surface area for pollution collection. These organisms also have the ability to metabolize and breakdown VOC's (Screiber and Schonherr, 1992; Muller, 1992; Beattie et al, undated).

5) Enhancing the aesthetics of pork production sites and rural landscapes.

- Professionals involved with livestock agriculture generally accept that a well-landscaped operation, which is visually pleasing or screened from view by landscaping is much more accepting to the public than one which is not (Lorimor, 1998; NPPC, 1995; Melvin, 1996). It is this notion of visual screening that has made landscaping and shelterbelts a common suggestion from agricultural engineers with regards to minimizing odor problems.
- If it is made known to neighbors and local communities that a shelterbelt is being used as a pollution (air or water) control tool, it may serve as very visible proof that a livestock producer is making an extra effort to control odor.

Shelterbelts are also a technology that can be considered “production technology neutral” and “size neutral” in that swine producers of all kinds – confinement, modified confinement, hoop house, pasture - and all sizes can plant designed shelterbelt systems. Shelterbelts, very uniquely, offer a technology that both producers and rural residents and communities can appropriately use, suggesting “user neutrality”. Further, as opposed to other odor mitigating technologies that typically depreciate over time, shelterbelts may be the only odor control technology that theoretically increases in effectiveness over time. As with other “tree” based technologies used in agriculture, the effectiveness of shelterbelts in mitigating odor comes from providing complex ecological infrastructure within an otherwise ecologically simplified system (Schultze et al, 2000). As the trees grow larger, and more morphologically complex their ability to mitigate odors should become increasingly efficient. Though this improvement over time is contingent upon the health and maintenance of the shelterbelt systems and the continuance of hog production best management practices.

This research review makes it clear that the published information on the ability of shelterbelts to mitigate on-and off-farm livestock odor is limited and further bio-physical, economic, and social qualification and quantification of this technology is needed. Yet the existing evidence seems to indicate that shelterbelts can help incrementally to reduce odor pollution and sustain both the swine industry and the quality of rural life.

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Financial Feasibility of Using Shelterbelts for Swine Odor Mitigation

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Poster Abstract

This study is a farm-level financial feasibility examination of an odor mitigation technology that has been receiving increasing attention – the use of shelterbelts in and around swine production facilities. Shelterbelts are purposefully planted rows of trees and shrubs around the main sources of swine odor – the swine buildings, the manure storage systems, and crop fields that receive land applied manure. As current research suggests if shelterbelts are utilized appropriately (proper species, scale, location, etc.), they should play a significant incremental role within a “suite” of odor mitigation strategies, which collectively will reduce odor nuisance risk for all stakeholders in rural air quality. Yet despite the apparent promise of this technological approach, very little is known about the financial requirements for installation (i.e. site preparation and planting) and long term management of shelterbelt systems that have been designed for odor mitigation. Therefore, by using a series of hypothetical pork finishing farms and a number of shelterbelt design scenarios, the specific objectives of this discounted cash-flow oriented financial analysis are: 1) To assess the farm level financial nature of planting and maintaining odor mitigation shelterbelt systems over an extended planning horizon (20 years); 2) To compare various shelterbelt costs with known figures (WTP) on pork producer expenditures strictly for odor control; 3) To assess the effect of current federal programs (specifically the Environmental Quality Incentive Program and the Conservation Reserve Program) on reducing the financial burdens to swine producers for using shelterbelts. The results show that the expense in using shelterbelts in these varied cases, in most cases adds only a few pennies to overall production costs. The calculated costs for shelterbelt establishment and maintenance over a twenty-year period for all model pork finishing farms, depending upon the scenario, range from \$0.01 to \$0.82 per pig produced. The results of this study also emphasize the importance of cost share programs, particularly EQIP. Still, in most cases, both with and without cost share programming, the upfront and maintenance costs were well below known expenditures for odor management technology.

Introduction

While odors from swine production are ubiquitous with animal agriculture and have been historically tolerated by rural communities, structural changes in the US swine industry (i.e. increased farm size, increased geographic concentration of production facilities and therefore animal manure) have caused odor to become much more pervasive and offensive (Iowa CAFO Air Quality Study, 2002).

This study is a farm-level financial feasibility examination of an odor mitigation technology that has been receiving increasing attention – the use of shelterbelts in and around swine production facilities (MWPS, 2002). Shelterbelts are purposefully planted rows of trees and shrubs around the main sources of swine odor – the swine buildings, the manure storage systems, and crop fields that receive land applied manure.

As research suggests (Malone and Abbot-Donnelly, 2001; MWPS, 2002; Khan and Abbassi, 2001) if shelterbelts are utilized appropriately (proper species, scale, location, etc.), they should play a significant incremental role within a “suite” of odor mitigation strategies. Collectively, these strategies can reduce odor nuisance risk for all stakeholders in rural air quality. Yet despite the apparent promise of shelterbelts for odor mitigation, very little is known about the financial requirements for installation (i.e. site preparation and planting) and long term management of shelterbelt systems that have been designed for odor mitigation (Lorimor, 2002).

The purpose of this analysis is to show that the planting and long term maintenance costs of shelterbelts for odor mitigation are indeed within the range of (producer determined) acceptable expenses for odor management. Therefore, the specific objectives of this financial analysis of the use of shelterbelts for odor mitigation are:

- To assess the farm level financial nature of planting and maintaining odor mitigation shelterbelt systems over an extended planning horizon (20 years) for four different hypothetical model swine farms.
- To compare shelterbelt costs with known figures on pork producer expenditures strictly for odor control.
- To assess the effect of current federal programs (specifically the Environmental Quality Incentive Program and the Conservation Reserve Program) on reducing the financial burdens to swine producers for using shelterbelts.

A critical assumption to this financial analysis is that shelterbelts can be bio-physically effective in incrementally attenuating swine odor. Recent research suggests that shelterbelts can play significant roles in biophysically and socially mitigating odor in a socio-economically responsible way thereby reducing social conflict from odor and dust nuisance (Leuty, T. 2004; Malone et al., 2004 a; Malone et al., 2004 b; Malone and Abbot-Donnelly, 2001; Tyndall and Colletti, 2001).

Research has shown that because the source of livestock odor is near the ground and the tendency of the odor plume is to travel along the ground, shelterbelts of even modest heights (i.e. 20-30 ft) may be ideal for plume interception, disruption, and dilution (Bottcher, 2001, Heisler and DeWalle, 1988; Laird, 1997; Thernelius, 1997; Takle, 1983). Shelterbelts can be adapted to fit the production situation and expected/experienced odor plume shape and timing. Depending on the shelterbelt design and tree/shrub species used, this technology can provide long term, year round plume interception, with increasing effectiveness over time. More is also becoming known about how landscape aesthetics affect how people might perceive livestock odor, suggesting that landscape elements such as shelterbelts can lead to improvements and perhaps more positive interpretations of livestock odor and the farm systems that create them (Mikesell et al., 2001; Kreis, 1978).

Shelterbelt Designs for Four Hypothetical Pork Feeder Finisher Farms

Four hypothetical pork-finishing farms were developed by modeling them after real pork finishing farms in Iowa. The model farms serve as the frame for financial calculations of shelterbelts designed to mitigate odor emissions specific to those model farms (See Table 1 below for details regarding each model farm). Each model farm reflects different pork production technologies, building ventilation and manure storage. Each model farm has an accompanying model shelterbelt system designed specifically for the site specificity of each model farm and the represented production technologies. Additional assumptions underlying each model farm shelterbelt design and the farm level financial comparisons are as follows: There are two main odor management zones receiving odor mitigative attention with shelterbelts – 1) A shelterbelt component for the farm itself, which includes the immediate production site property, the swine buildings and the manure storage facilities; And 2) a shelterbelt component for a contiguous crop field that receives manure from the production facility for nutrient utilization purposes.

The swine farms are assumed to be located in central Iowa. Shelterbelt systems are designed with these wind patterns in mind.

Two planting stock cost scenarios were used: A high priced 6+ year old planting stock scenario and a low priced seedling stock scenario.

Table 1. Model Farm and shelterbelt system details.

	Farm A – Full confinement facility	Farm B– Full confinement facility	Farm C– Full confinement facility	Farm D – Hoop Barn Facility
Annual Pig Production	10,500	2,025	1,600	1,000
Building Ventilation Type	Tunnel	Mechanical	Natural	Natural
Manure Storage	Above ground concrete	Above ground concrete	Above ground concrete	Solid/bedding
Farm Acreage	5.91	1.38	0.90	0.83
Number of trees Planted	950	325	178	122
Number of Shrubs Planted	214	141	51	145
Application Field Acreage	660	160	100	25
Number of Application Field Trees	2,145	1,045	835	395

Financial Appraisal

This study used standard cash-flow analysis to examine the costs of establishing and managing a designed shelterbelt within and around the four model pork finishing facilities and shelterbelts that are planted around manure application fields. Costs examined include: site preparation, plant stock, shelterbelt establishment, shelterbelt repair and maintenance (long term management), and land rent (opportunity costs) for shelterbelts planted around crop fields. Measures of odor mitigation effectiveness from shelterbelts are not considered.

All costs were discounted using standard discounting formulation (Klemperer, 1996). Final costs of the model shelterbelts are reported in two ways, first Present Value Cost (PVC) and PVC minus any governmental program payments (PVC^{GOV}) are established for each shelterbelt design. The second way is to apply a Capital recovery factor to total costs of each model shelterbelt and divide by the number of pigs produced annually; this presents total costs on a per unit of production (per pig) and spreads the costs out across all the pigs produced in a twenty-year period. This method is appropriate because the shelterbelt systems are designed to mitigate swine odor over an estimated ownership span (ISU, 1998) and therefore costs are spread over all the pigs produced at that facility over a 20 year period.

Across all the model farms the average costs associated only with the shelterbelt systems planted in and around the facility grounds range from \$0.02 and \$0.25 per pig produced (seedling price and high price respectively). When EQIP cost share programming is factored in, the average costs to producer's fall to \$0.01 to \$0.13 per pig. Looking at the combined average costs for establishment and management of shelterbelt systems around the facility grounds and the manure application fields of all four model facilities, the total costs without cost share come to \$0.21 and \$0.82 per pig produced (seedling/high price respectively). With cost share programming (in this case EQIP for the facility grounds and CRP for the application fields) the costs fall to \$0.05 and \$0.26 per pig produced. The largest facility had the highest upfront costs and the highest present value costs over 20 years, but ultimately slightly lower costs per pig produced due to the high production volume.

When these costs are compared to known (revealed) pork producer willingness to pay for on-farm odor management (USDA, 1996), the costs are in most cases well below what producers have spent in the past. The results of this study (both with and without cost share programming) show that the relatively low costs of establishing and maintaining shelterbelts for odor mitigation allow a “feasibility excess” of as much as \$0.33 - \$0.59 per pig produced that, ostensibly, could be spent on other complimentary odor management strategies. Financially there is room for shelterbelts to be part of a “suite” of odor management technology. It is also clear that EQIP benefits become more important as upfront costs of shelterbelt implementation

increase (in this case the costs of the planting stock). This point is relevant for producers planning shelterbelt systems using older, larger planting stock.

However, a separate survey of NRCS EQIP agents revealed that while seven of the top ten hog producing states do recognize shelterbelts as an EQIP relevant odor mitigating Best Management Practice only a handful of pork producers around the country have received EQIP dollars specifically to use shelterbelts for odor mitigation.

General Conclusion

Based on the above cost information the use of shelterbelts for odor mitigation in these varied cases only add a few pennies to overall production costs. Expansion/promotion of cost share possibilities to producers likely will play a key role in future use of shelterbelts as an odor mitigating technology. Pork producers are in a legitimate quandary in that they face distinct risk of nuisance litigation at the same time that their profit margins are thin at best and very often negative on average. The costs of even pennies per pig can seem too much to bear under such economic conditions. As other studies have shown for some time now, when the government offers to help pay for agricultural externalities such as air and water pollution, farmers are much more likely to participate in promoted technological approaches (Purvis et al., 1989). This research demonstrates that shelterbelts can be a financially feasible part of overall plans to improve the sustainability of both rural communities and of the livestock industry.

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