INVERSE-DISPERSION CALCULATION OF AMMONIA EMISSIONS FROM WISCONSIN DAIRY FARMS

T. K. Flesch, L. A. Harper, J. M. Powell, J. D. Wilson

ABSTRACT. *Ammonia (NH_3) emissions were determined from three commercial dairy farms in the north-central U.S. The dairies employed similar management, having naturally ventilated free-stall barns where barn waste is scraped and transferred to outdoor lagoons. Three potential emission sources were distinguished at each farm: barns, lagoons, and sand separators. A backward Lagrangian stochastic (bLS) inverse-dispersion technique was used to measure emissions. Total farm emission varied from 15 to 330 kg NH_3 d^{-1} depending on the farm and season. Inter-farm variability was largely explained by farm size (animal population). Emissions showed variability on seasonal and daily scales: summer rates were roughly ten times those of the winter, and mid-day rates were approximately three times those at night. The lagoons emitted 37% to 63% of the farm total during summer and fall, but they were frozen in winter and their emissions were immeasurably small. The yearly per-animal emissions from the three dairies were estimated at 20, 19, and 20 kg NH_3 $animal^{-1}$ $year^{-1}$. Regarding the measurement technique, bLS proved well-suited to our study. With modest resources we were able to measure emissions from the variety of sources at each farm and quickly move between farms. Overall agreement in measured emissions at the three farms, together with a general harmony of our measurements with those from previous studies, provides a measure of confidence in the measurement strategy.*

Keywords. *Air quality, Ammonia emissions, Atmospheric dispersion, Dairy farm.*

Ammonia (NH_3) emitted to the atmosphere has important environmental implications. When reacting with acid gases (e.g., sulfur dioxide), ammonia forms particulates that degrade air quality. Ammonia and its reaction products can also deposit downwind of an emission source and dramatically alter the nitrogen (N) balance of an ecosystem. The largest global source of atmospheric NH_3 is animal husbandry (Asman, 2002). A corollary of the modern trend to larger and more efficient confined animal feeding operations (CAFOs) is the creation of large and concentrated NH_3 sources. In some jurisdictions this has led to the possibility of regulation and oversight of agricultural operations in terms of gas emissions. However, a full understanding of the impact of CAFOs and effective means of mitigation are hindered by a lack of information on the magnitude of NH_3 emissions across the variety of management systems.

One of the problems is the difficulty in measuring CAFO emissions. Measuring gas emissions from any source is a difficult problem (Denmead and Raupach, 1993), and the chemical properties of NH_3 add to this difficulty (Harper, 2005). Furthermore, emissions from CAFOs often originate from a variety of distinct sources, such as barns and waste lagoons. One could concentrate on characterizing each of these sources in isolation using, for instance, a mass balance or gas tracer technique for barns (e.g., Sharpe et al., 2001; Kaharabata and Schuepp, 2000) and micrometeorological or chamber techniques for outdoor sources (e.g., Denmead et al., 1998; Aneja et al., 2001). Each of these traditional techniques, however, requires specialized knowledge and equipment, and the effort needed to characterize all of these components would be considerable. Another possibility is a large-scale mass balance approach measuring the total horizontal flux of gas passing from and downwind of the farm. This requires many wind and concentration measurements to determine the flux, which must be summed over a vertical plane standing downwind of the farm (Phillips et al., 2000). Moreover, to observe fully a realistic farm plume would require instruments exposed many meters above the ground.

The "inverse-dispersion" technique provides an economical alternative for measuring emissions. Here one uses a mathematical model of the dispersion of target gas from an emission source to a downwind location, so that a downwind concentration measurement can establish the emission rate (e.g., Flesch et al. 2004). This has the advantage of requiring only a single concentration measurement and basic wind information, with substantial freedom to choose convenient measurement locations. A disadvantage is that in its most practical form the technique entails the assumption of idealized wind conditions. However, with careful selection of measurement locations, it can provide a simple means of calculating emissions even in non-ideal conditions (Flesch et al., 2005a, 2005b). The technique, for example, has been used to measure emissions from dairy barns (e.g., McGinn et al.,

Submitted for review in July 2008 as manuscript number SE 7585; approved for publication by the Structures & Environment Division of ASABE in December 2008.

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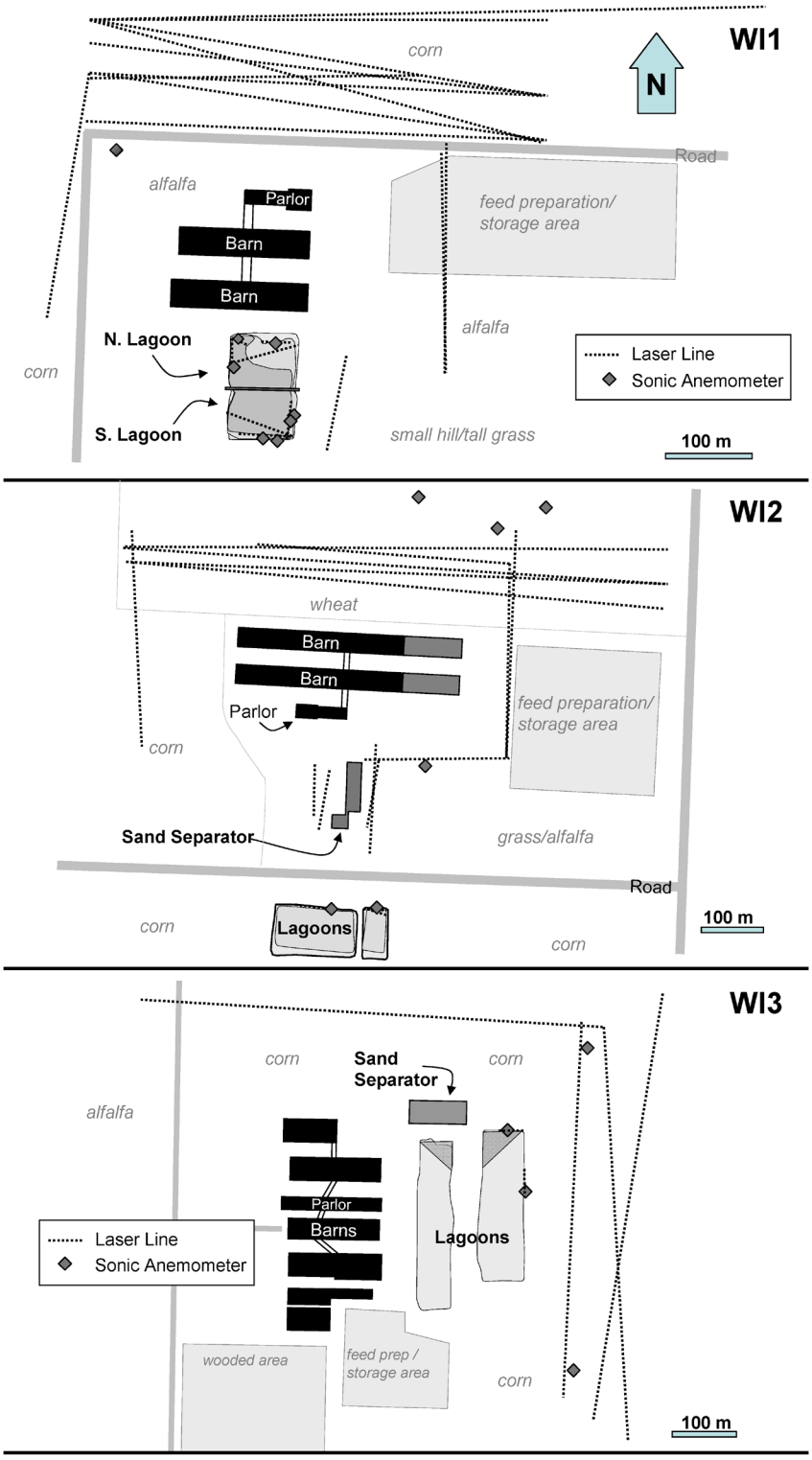

Figure 1. Map of WI1 (top), WI2 (middle), and WI3 (bottom) showing laser lines (dotted lines) and sonic anemometer locations (diamond symbols) used in the study. Different lagoon outlines represent the different seasonal levels, and the different barn outlines at WI2 show the barn's extension during the study. Surrounding summer crops are given.

2006), cattle feedlots (e.g., Flesch et al., 2007), animal pastures (e.g., Laubach and Kelliher, 2005), and manure stockpiles (e.g., Sommer et al., 2004).

The objective of this study was to measure NH_3 emissions from modern dairy farms typical of the Wisconsin region of the north-central U.S. Measurements took place at three farms and over three seasons (winter, summer, and fall). The majority of this article is devoted to a description of the inverse-dispersion technique: a general overview with details of our measurement and analysis strategy. We also summarize our emission measurements and consider how emissions differed between farms and seasons.

Study Farms

Emission measurements were made at three commercial dairies: one each in northeast, east-central, and south-central Wisconsin. These are designated WI1, WI2, and WI3, and each is a modern and relatively large CAFO (>800 milking cows). Only milking cows were present at WI1, while WI2 and WI3 had a mix of milking cows, dry cows, and heifers. The dairies use a parlor milking system with cows housed in naturally ventilated, free-stall barns (side-wall curtains are raised and lowered to control ventilation). Sand is used for bedding. Animal waste (and sand) is routinely scraped from the concrete barn floors to a central channel and then moved underground to outdoor storage lagoons. At WI1 (fig. 1), the barn waste moves to the lagoons by gravity flow. Farms WI2 and WI3 (fig. 1) employ a flushing system to move the waste using recycled lagoon water. These latter two farms also have a sand separator channel where sand in the waste is deposited prior to entering the lagoons, is gravity drained, and is then recycled for bedding. There is a near-continuous stream of waste flowing through the exposed channel, and sand is removed at least once a day. The barns, lagoons, and sand separator channels are all potential sources of NH_3 to the atmosphere.

The study farms were selected as being representative of modern dairies in the region. A further selection criterion was that they offered an appropriate setting for application of the inverse-dispersion technique. This required the farms be located on relatively open terrain and be isolated from other NH_3 sources. At WI1 and WI2 there were no trees or buildings (other than the study barns) immediately around the farm, and the ground was relatively level. At WI3, the terrain was more rolling and (for some wind directions) only a small woodlot stood immediately upwind of the farm.

Inverse-Dispersion Technique

Consider the open-sided barns of our study farms (fig. 2a). These barns have an unknown ammonia emission rate Q (kg h^{-1}). We measure the time-average NH_3 concentration above the background level ($C - C_b$) at downwind point M. There is clearly a relationship between Q and ($C - C_b$). In theory, this connection can be determined with an atmospheric dispersion model that describes the dilution of gases as they are mixed and transported downwind. The model calculates the ratio of the concentration rise to the emission rate $(C/Q)_{sim}$ at M, so that the barn emission rate is given by:

$$Q = \frac{(C - C_b)}{(C/Q)_{sim}} \quad (1)$$

This is the basis of the inverse-dispersion technique. It requires a single C measurement (assuming C_b is known) with flexibility in the choice of the measurement location M. The accuracy of the technique rests on an accurate calculation of $(C/Q)_{sim}$.

The most realistic dispersion models utilize the average wind and turbulence statistics of the atmosphere to calculate $(C/Q)_{sim}$. For an idealized landscape (i.e., horizontally homogeneous) these statistics can be provided with relative ease. Monin-Obukhov similarity theory (MOST) states that the wind properties in the surface layer (below a height of approximately 50 m but above a plant canopy) are determined by a few key parameters (Garratt, 1992): the friction velocity u^*, the Obukhov stability length L, the surface roughness length z_0, and the wind direction β. These can be determined from simple surface observations (e.g., from a 3-D sonic anemometer), and in this ideal environment one can accurately calculate $(C/Q)_{sim}$ with a relatively simple model (e.g., Flesch et al., 2004).

Figure 2a is not an ideal landscape. The barns will interact with the ambient wind to create a complex pattern of wind vortices, jets, and sheltered zones. Accounting for these complications in a dispersion model is beyond practical capabilities. Instead, we focus on the question of whether idealized calculations can be used at sites that are *not* ideal. Consider the idealized analog of the barns in figure 2b, where the barns are treated as surface area sources, with no disturbance to the ambient winds. In what situation would the actual (C/Q) in figures 2a and 2b be similar? For a location M near the barns, we should expect large differences in (C/Q) because of large differences in the two wind fields in the immediate lee of the barns. However, the field studies of Flesch et al. (2005a) and McBain and Desjardins (2005) illustrate the principle that as M is moved downwind of the barns, the difference in (C/Q) between the two cases is reduced. The results of Flesch et al. (2005a) suggest that if M is beyond about 10 barn heights (h) from the barns, then using an idealized dispersion model that ignores the local wind complexity around the barns will result in only a small error in $(C/Q)_{sim}$. However, this criterion needs to be interpreted as a broad suggestion. In a barn tracer-release experiment, McGinn et al. (2006) found that idealized calculations gave good results even with M closer than $10h$ from the barns.

There is another complication at a real dairy farm. For a single C observation, the inverse-dispersion technique can give only a single emission rate Q. If the farm is a compound source, then the calculation of a whole-farm Q requires assumptions about the proportion of emissions from the different sources, e.g., assuming equal emissions from lagoons and barns. An inaccurate "allocation" causes errors in the Q calculation. However, as location M is moved farther from the farm there is decreased sensitivity to these errors. An important measurement scale for this problem is the separation distance between sources (x_s), e.g., the distance between a barn and lagoon. Flesch et al. (2005b) showed an example where once M was farther than $2x_s$ from a multi-component site,

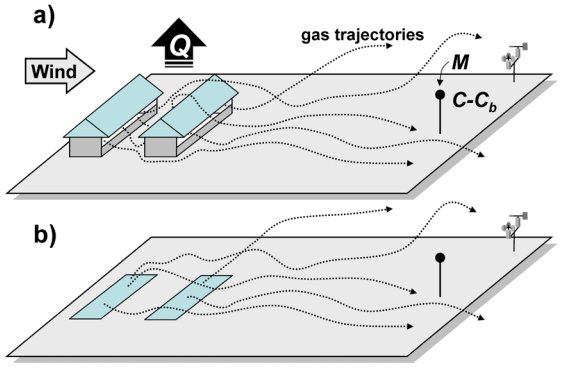

Figure 2. Illustration of the inverse-dispersion technique to measure the gas emission rate (Q) for: (a) the naturally ventilated barns in this study, and (b) an idealized analog where barns are treated as surface sources that do not modify the ambient winds. A concentration rise above background ($C - C_b$) is measured at point M, and Q is deduced with the aid of a dispersion model and wind information.

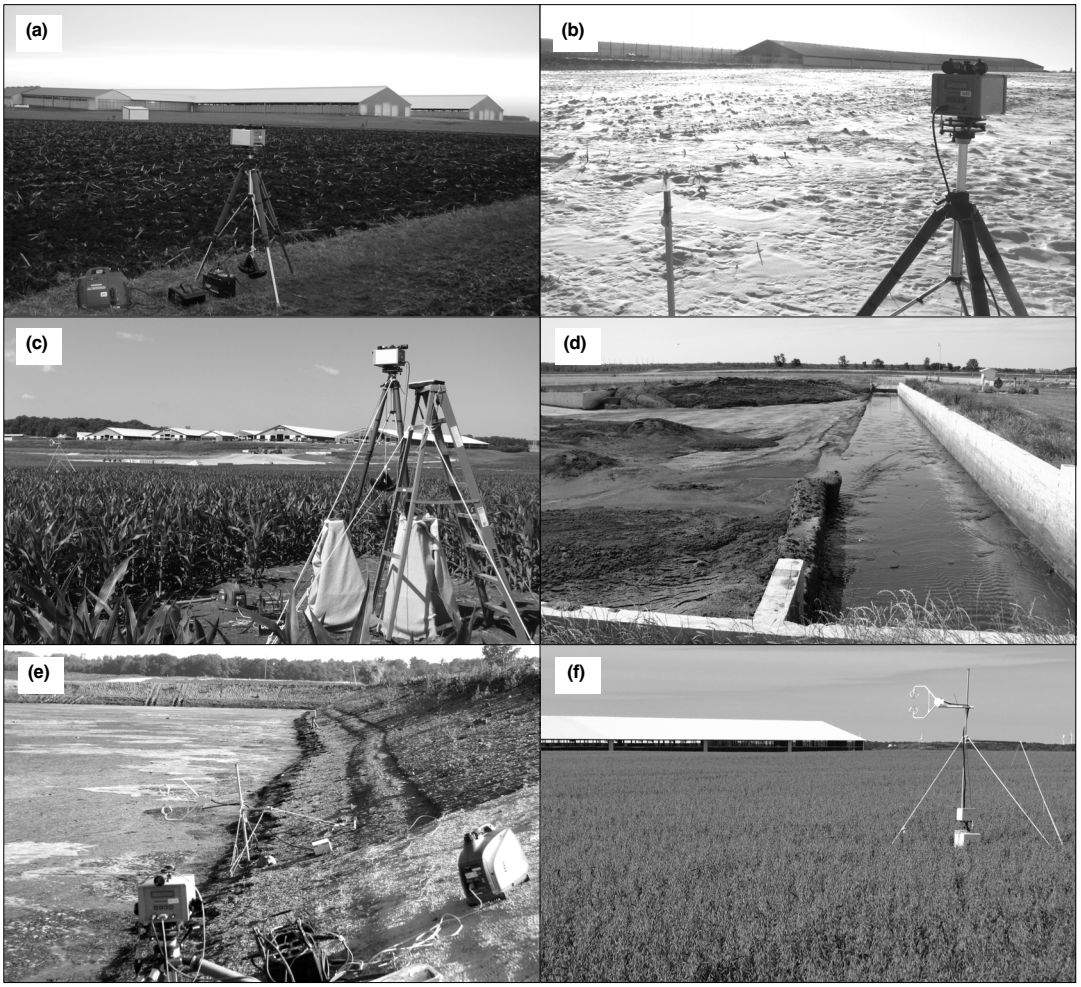

Figure 3. Photographs from the study: (a) laser measuring barn emissions at WI1-Fall, (b) measuring barn emissions at WI2-Winter, (c) measuring whole-farm emissions at WI3-Summer (laser on ladder to get above the corn), (d) sand separator, (e) laser and sonic anemometer at lagoon edge, and (f) sonic anemometer measuring ambient winds.

where x_s is the maximum of the source separation distances, the error in $(C/Q)_{sim}$ caused by an incorrect allocation was less than 10%.

Three broad requirements are thus needed when applying an idealized calculation to estimate farm emissions. First, the farm should be relatively isolated on the landscape so that wind disturbances associated with farm structures are local and there is a downwind return to a measurable ambient wind state. Isolation also ensures no nearby confounding gas sources. Second, the measurement location M should be many barn heights h downwind of the farm (we adopt 20h as a preferred configuration). And third, when calculating total emissions from a multi-component site, M should be multiple "source-separation" distances x_s from the farm.

Measurements and Analysis

Field Observations

On-farm measurements took place between December 2006 and November 2007. Each farm was visited three times (winter, summer, and fall), each time for a campaign lasting 10 to 14 days. Ammonia concentrations were measured with open-path lasers (GasFinders, Boreal Laser, Inc., Edmonton, Canada) calibrated on-site using calibration tubes flooded with NH_3 standards. The lasers give the line-average concentration between the laser and a retro-reflector, which in this experiment were separated by 30 to 1000 m (figs. 1 and 3). Laser signals were processed to give 15 min average concentrations along the laser line (C_L). Mixing ratio concentrations (ppm_v) were converted to absolute concentrations ($g\ m^{-3}$) using the average air temperature for each observation and the average atmospheric pressure corresponding to each farm's elevation. Note that whereas the above discussion of the inverse-dispersion technique assumes a *point* concentration measurement, the extension to a *line-average* concentration is not only trivial, but more importantly, is beneficial for the accuracy of the technique (Flesch and Wilson, 2005).

The farm wind environment (average wind and turbulence statistics) was approximated using standard MOST formula based on the characteristic parameters (u^*, L, z_0, and β) provided by a three-dimensional sonic anemometer (CSAT-3, Campbell Scientific, Logan, Utah). The anemometers were placed at locations chosen to represent (broadly speaking) the winds sampled by trajectories from the farm source(s) to the detecting laser line. Wind parameters were calculated for each 15 min period (corresponding to a C_L observation). See Flesch et al. (2004) for details of how these parameters were calculated from a sonic anemometer.

bLS Dispersion Model

A bLS dispersion model gives $(C_L/Q)_{sim}$ for each 15 min observation of C_L. We used the WindTrax software (Thunder Beach Scientific, Nanaimo, Canada), which combines the bLS model described by Flesch et al. (2004) with an interface allowing sources and sensors to be conveniently mapped. In the bLS model, thousands of trajectories are calculated upwind of the laser line for the prevailing wind conditions. The important information for our inference of emission rate is the set of trajectory intersections with ground ("touchdowns"):

$$(C_L/Q)_{sim} = \frac{1}{n}\sum\left|\frac{2}{w_0}\right| \quad (2)$$

where n is the number of computed trajectories, w_0 is the vertical velocity of the trajectory at touchdown, and the summation covers all touchdowns occurring within the designated source area. (The units of Q are kg m^{-2} s^{-1} in this equation. Hereafter, we multiply the areal emission rate by the source area and report Q as an area-integrated emission rate with units of kg h^{-1}.) The touchdowns map the concentration "footprint", i.e., the ground area where emissions influence C_L (see fig. 4 for examples).

The study farms are represented as a collection of surface area sources corresponding to the positions of barns, lagoons, and sand separators (mapped with a GPS). We calculate $(C_L/Q)_{sim}$ using n = 60,000 to 1,000,000 trajectories, with n chosen to keep the stochastic uncertainty of this type of model suitably small (i.e., to keep the standard deviation, given by ten subgroups of trajectories, to <10% of the average). Background NH_3 concentrations were assumed to be C_b = 0.000, 0.010, and 0.005 ppm_v for winter, summer, and fall, respectively. These values were estimated during periods when the lasers measured "fresh air" uninfluenced by the farms (some estimation was necessary because C_b was usually below the measurement threshold of the lasers).

Not all observation periods give good Q measurements, and we followed the filtering process of Flesch et al. (2005b). Three criteria identify periods when a MOST description of the wind is likely to be inaccurate, i.e., the calculated $(C_L/Q)_{sim}$ is likely to be inaccurate and that period was not used:

- $u^* \leq 0.15$ m s^{-1} (low winds)
- $|L| \leq 10$ m (strongly stable/unstable atmospheric stratification)
- $z_0 \geq 1$ m (associated with unrealistic wind profiles).

For some wind directions, a source plume only "glances" a laser line. This leads to three problems: the plume edge is associated with greater $(C_L/Q)_{sim}$ uncertainty due to the difficulty of modeling lateral dispersion; emission measurements are weighted toward unrepresentative areas at the source edge; and slight errors in wind observations (particularly wind direction) can result in dramatic errors in $(C_L/Q)_{sim}$. We therefore do not use periods where the laser touchdowns cover less than 50% of diagnosed source area (WindTrax calculates the fraction of source pixels displayed as touchdowns). An exception was the lagoons, where short laser lines at the lagoon edge give good results with lesser coverage.

The bLS calculation of $(C_L/Q)_{sim}$ assumes that NH_3 is a passive tracer with no deposition to the downwind surface, and no chemical transformation between the emission source and the laser line. Given the short distances between sources and lasers (typically <200 m), we feel this assumption is realistic.

Measurement Strategies

Our primary objective is to calculate whole-farm emissions inclusive of those from the barns, lagoons, and sand separator. A secondary objective, when possible, is to calculate those component emission rates. Emissions from manure land applications were not measured. At Wisconsin dairy

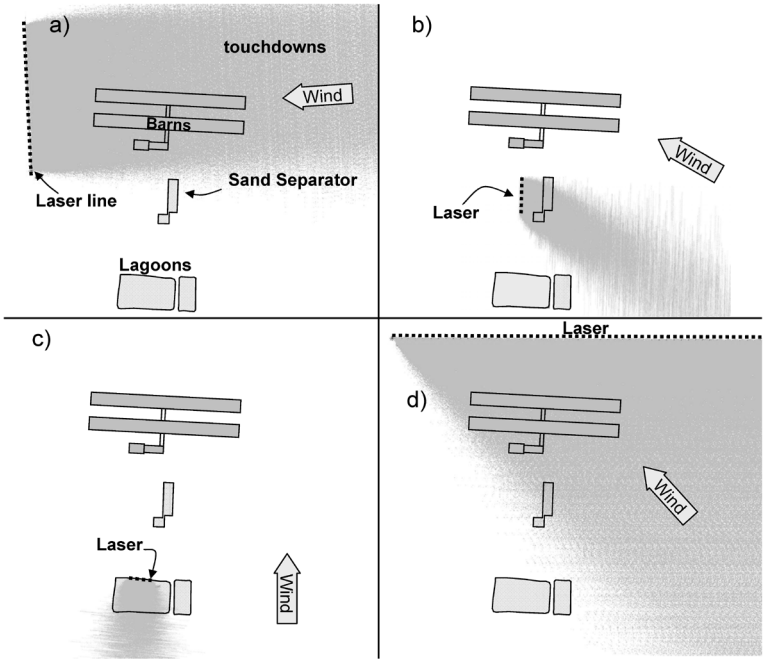

Figure 4. Examples of bLS touchdown fields for laser detectors at WI2 (touchdown dots upwind of laser lines map the ground area where emissions influence C_L): (a) laser influenced only by barn emissions, (b) laser influenced only by sand separator emissions, (c) laser influenced only by lagoon emissions, and (d) laser influenced by both barn and sand separator emissions.

CAFOs, the lagoons are emptied and the manure land-applied during spring and fall. With the exception of one farm (WI3-Fall), no lagoons were emptied during our measurements, and no manure was applied on land adjacent to the farms during the month prior to our visit. Five measurement strategies were used in this study:

Isolate barns. Often we could place a laser downwind of the barns, and the C_L increase was attributable solely to barn emissions (fig. 4a). Our criterion for laser placement was that its light path be distant from the barns by at least $10h$ (barn heights) and preferably $20h$. Ambient winds were measured away from the farm structures (fig. 3f).

Isolate sand separator. The sand separator channel and draining pad is a structure that disturbs the wind (fig. 3d). We took the channel depth h_{ss} as a characteristic height (h_{ss} ~ 1.8 m) and placed laser lines at least $10h_{ss}$ downwind of the separator. Winds were measured over the adjacent open ground, and emissions were calculated for periods when the C_L increase was solely due to separator emissions (fig. 4b).

Isolate lagoon. We followed the ideas of Wilson et al. (2001), Flesch et al. (2007), and McGinn et al. (2008). A laser line and anemometer were placed at the lagoon edge at heights from 0.8 to 1.2 m above the surface (fig. 3e), and emissions were calculated when the sensors were within the lagoon plume (fig. 4c).

Blended plumes. Sometimes a source plume cannot be isolated, and the increase in C_L is the result of two or more sources. This situation occurred when laser lines were placed to measure the barn plume but where certain wind directions also placed the lines in the lagoons/separator plumes (fig. 4d). Here our procedure is to treat the lagoons/separator as known sources, entering their "known" emission rates (taken from the direct measurements) in the dispersion model. Their contributions to the laser C_L were then calculated and subtracted before calculating barn emissions.

Whole-farm emissions. At WI3 the source components could not be isolated due to the compact farm layout (fig. 1), and we calculated only whole-farm emissions. Because of site restrictions, we could not place the lasers the desired distance from the farm (farther than one barn-to-east lagoon separation distance x_s), a criteria to minimize sensitivity in the Q calculation to errors in the assumed emission allocation. Fortunately, the winter Q was found to be relatively insensitive to the assumed barn-lagoon/separator allocation over a range of reasonable possibilities. In summer, our strategy was to reduce the effective x_s by directly measuring emissions from the east lagoon and treating it as a known source, so that x_s was reduced to the barn-to-west lagoon distance.

Our specific goal is to calculate the average daily emission rates for each farm-season. This is complicated by having a non-continuous observation record due to data filtering, off-line equipment, etc. Here we will assume that the appropriate average rates can be calculated from ensemble-average daily (24 h) emission curves. For each farm-season, we group the emission observations by time-of-day and average this data into twelve 2 h blocks to cover the 24 h day (missing blocks are interpolated from available blocks). The resulting averages are then integrated (i.e., summed) over the 24 h to give daily emission rates.

RESULTS

DETAILED LOOK AT WI2-SUMMER

Each of our study farms has a unique layout of emission sources, and this layout varies with season (e.g., the lagoons are not sources when frozen). Thus each farm-season campaign required a unique measurement and analysis plan. We lack the space to describe each in detail. Instead, we provide here those details for one farm and one season (WI2-Summer). The other campaigns use variants of the following analysis.

WI2-Summer Lagoons

The two lagoons at WI2 (fig. 1) were connected by free-flowing pipe with occasional pumping between them. We treat them as identical NH_3 sources on a per surface-area basis. Measurements were made on the larger west lagoon. A laser line and anemometer were positioned along the north edge of the lagoon, and emissions were measured during southerly winds.

Figure 5a shows lagoon emissions (Q_{Lag}) during four days in late June. A strong daily cycle is evident, with mid-day emission rates consistently three to four times the nighttime rate (fig. 5b). What explains this cycle? Ammonia emissions are generally related to four factors (Freney et al., 1983): NH_4^+ concentration of the medium, pH of the medium, temperature of the medium, and the effectiveness of turbulent transport of NH_3 away from the medium (a function of the windspeed). Of these factors, lagoon temperature and windspeed will have a daily cycle. Studies of N-rich liquid surfaces often show a dominant windspeed relationship for NH_3 emissions (e.g., Denmead et al., 1982; Harper et al., 2006; Flesch et al., 2007). Indeed, we see a positive correlation between windspeed (u^*) and Q_{Lag} (fig. 5c) with a Pearson correlation coefficient r = 0.74. (We use friction velocity u_* and windspeed interchangeably in this discussion; u_* is a scaling velocity that is roughly proportional to windspeed, although the connection is influenced by atmospheric stability.) A path-coefficient statistical analysis calculates the "direct" and "indirect" effect of the correlated properties of windspeed and time-of-day on Q_{Lag}. It indicates that time-of-day is the dominant direct effect, and we conclude that lagoon emissions follow a consistent daily cycle that is somewhat modulated by windspeed. We calculate an average emission rate of 103 kg NH_3 d^{-1} from the lagoon daily cycle (fig. 5b).

In the analysis that follows, we extrapolate our lagoon observations to estimate Q_{Lag} for non-measured periods (so we can subtract the lagoon contribution to C_L when a laser "sees" a blended barn-lagoon plume). A regression model is used for this purpose, with ambient u^* (measured north of the barns) and time-of-day as predictors:

$$Q_{Lag} = 1.06 + 3.98\,\tau_{Lag} + 3.62\,u^* \qquad (3)$$

where τ_{Lag} is a time-of-day variable, i.e., a sine wave ranging from zero at 0100 to unity at 1300 local standard time (LST):

$$\tau_{Lag} = 0.5 + 0.5\,SIN\left(2\pi\frac{HR-1}{24} - \frac{\pi}{2}\right) \qquad (4)$$

and HR is the hour-of-day. The units of Q_{Lag} are kg h^{-1}, and the units of u^* are m s^{-1}. The accuracy of equation 3 in describing lagoon emissions (r^2 = 0.88) can be assessed from figure 5d. We claim no generality of this relationship other than to these particular lagoons for this study period.

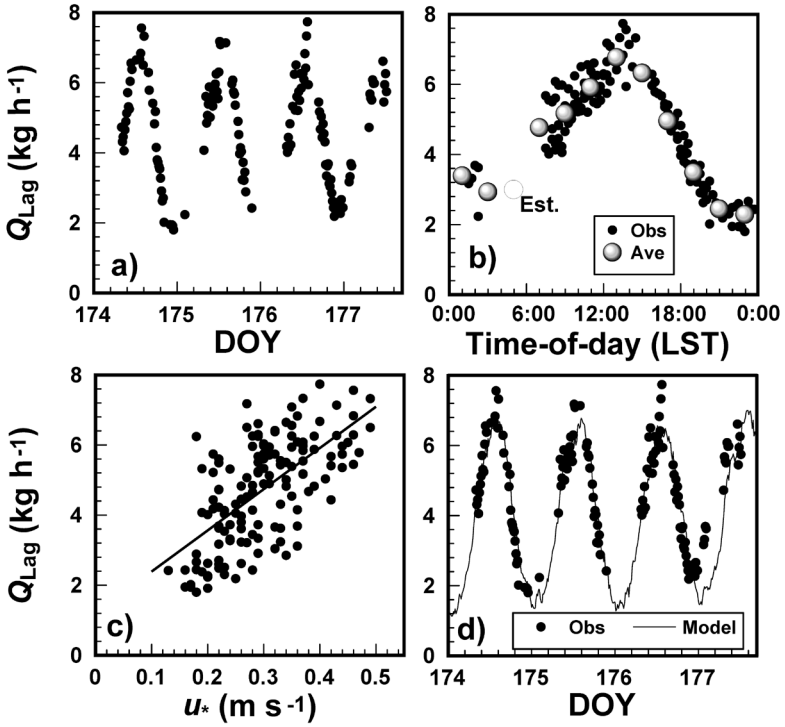

Figure 5. Lagoon emissions Q_{Lag} from WI2-Summer: (a) observations plotted versus day-of-year (DOY), (b) plotted versus time-of-day in local standard time (LST) and showing 2 h block averages (Ave, open circle is estimated), (c) plotted versus ambient friction velocity u^* (line is linear best fit), and (d) observations and regression model estimates (line) plotted with DOY.

WI2-Summer Sand Separator

The sand separator at WI2 was located between the barns (to the north) and the lagoons (to the south) and was surrounded by recently cut grass. Over four days, a laser line was alternately positioned east and west of the separator with an anemometer placed to the east (fig. 1). The NH_3 concentrations downwind of the separator showed an unexpected feature. The C_L fell to zero as windspeed fell to low levels. This was not seen with the other sources (usually the highest C_L occurs during light winds) and suggests that Q_{SS} is particularly sensitive to windspeed. Usually we do not use $C_L = 0$ data, but here such events describe an important feature, and we take $C_L = 0$ as meaning $Q_{SS} = 0$ if this occurred during light winds ($u^* < 0.3$ m s^{-1}) when the laser line was well within the separator plume (e.g., fig. 4b).

A daily cycle is clearly evident in Q_{SS} with maximum emissions in the afternoon (figs. 6a and 6b). This cycle is more variable than seen for the lagoons, and there is a clearer relationship with windspeed (r = 0.9; fig. 6c). In particular, it appears that emissions cease when $u^* < 0.2$ m s^{-1}. This suggests that air motion within the separator channel (lying below ground level) inhibits turbulent transport from the waste stream in low winds, i.e., there is a buffering layer of weak mixing over the waste stream. Stronger winds seem to weaken or destroy this buffering. We calculate average separator emissions of 31 kg NH_3 d^{-1} (averaging the daily Q_{SS} cycle in fig. 6b).

In the analysis that follows, we extrapolate our data to give Q_{SS} during unmeasured periods. We use a simple regression model based on the ambient u^*:

$$Q_{SS} = -1.08 + 6.98\,u^* \qquad (if\ Q_{SS} < 0, Q_{SS} = 0) \qquad (5)$$

The units of Q_{SS} are kg h^{-1} and the units of u^* are m s^{-1}. The accuracy of the model ($r^2 = 0.86$) can be observed in figure 6d.

WI2-Summer Barns

A laser line was positioned north of WI2 to measure barn emissions Q_{Brn} during southerly winds. For some periods, the laser line also intersected the lagoons/separator plumes. As discussed previously in the Measurement Strategies section, this complication is dealt with by treating the lagoons and separator as known emission sources in our dispersion calculations, and their contributions to C_L are subtracted before calculating Q_{Brn} (this is done automatically in the software). We used equations 3 and 5 to estimate the lagoon and separator emissions for this analysis.

Figure 7a shows Q_{Brn} over 10 days of observations. The data gaps correspond to non-southerly or light winds, or when C_L was below the laser measurement threshold (occurred mostly in the afternoon when an unstable atmosphere more effectively dispersed the barn plume). When Q_{Brn} is displayed versus time-of-day, we again see a daily cycle (fig. 7b), but with a lower daily range and less consistency than the cycle in lagoon emissions. There is a relationship between Q_{Brn} and windspeed (fig. 7c), but this is statistically weaker (r = 0.61) than the windspeed relationship for the lagoon and separator emissions. We calculate an average barn emission rate of 70 kg NH_3 d^{-1} (integrating the daily Q_{Brn} cycle in fig. 7b).

Later in this section we "gap-fill" periods of missing Q_{Brn} with a simple regression model:

$$Q_{Brn} = 0.56 + 1.21\,\tau_{Brn} + 5.92\,u^* \qquad (6)$$

where τ_{Brn} is a time-of-day variable that is a variant of equation 4, i.e., a sine wave with zero at 2300 and unity at 1100 (LST). The units of Q_{Brn} are kg h^{-1} and the units of u^* are m s^{-1}. This model ($r^2 = 0.49$) is not as statistically successful as the lagoon/separator models, but it usefully describes the average emission pattern (fig. 7d).

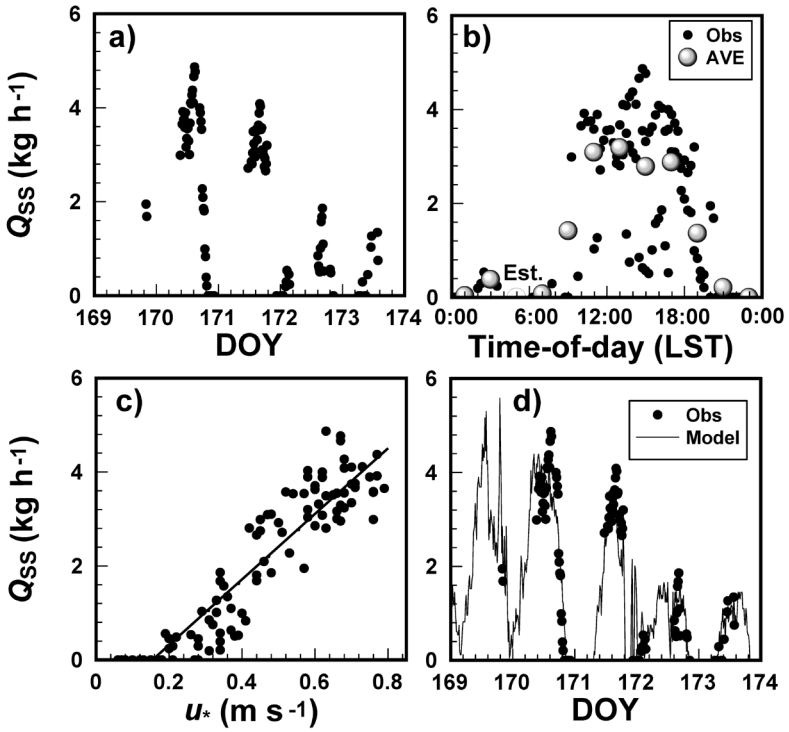

Figure 6. Sand separator emissions Q_{SS} from WI2-Summer: (a) observations plotted versus day-of-year (DOY), (b) plotted versus time-of-day in local standard time (LST) and showing 2 h block averages (Ave; open circle is estimated), (c) plotted versus ambient friction velocity u^* (line is linear best fit), and (d) observations and regression model estimates (line) plotted with DOY.

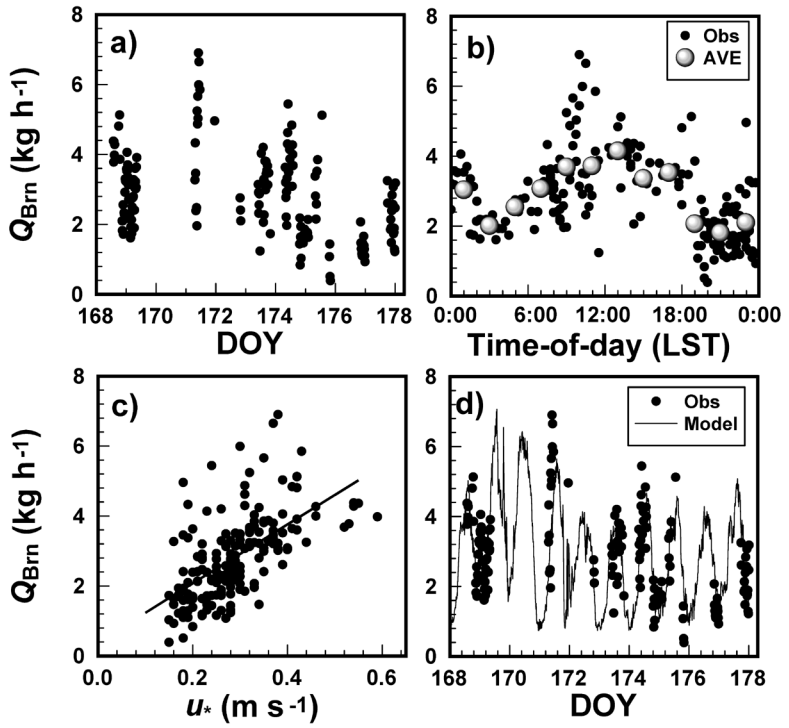

Figure 7. Barn emissions Q_{Brn} from WI2-Summer: (a) observations plotted versus day-of-year (DOY), (b) plotted versus time-of-day in local standard time (LST) and showing 2 h block averages (Ave), (c) plotted versus ambient friction velocity u^* (line is linear best fit), and (d) observations and regression model estimates (line) plotted with DOY.

WI2-Summer Whole-Farm Emissions

Total farm emissions from WI2 are defined as the sum of those from the lagoons, sand separator, and barns. We explore two alternatives for this whole-farm calculation. The first takes the component averages as determined from the direct observations described above (i.e., from the average daily cycles). This gives total emissions of 103 (lagoons) + 31 (sand separator) + 70 (barns) = 204 kg NH_3 d^{-1}.

Table 1. Daily NH_3 emissions from farms. For WI1 and WI2, we calculate emissions from barns, lagoons, and sand separator with total emissions given by their sum; for WI3 only a whole-farm calculation is made. "Obs." indicates the number of 15 min observations used in each calculation. Shaded blocks indicate either non-existence of the component emitter or that measurements were not made.

	Lagoons		Sand Separator		Barns		Total			
Farm-Season	Emissions (kg d^{-1})	Obs.	Emissions (kg d^{-1})	Obs.	Emissions (kg d^{-1})	Obs.	Emissions (kg d^{-1})	Obs.	No. of Animals[a]	Emission per animal (g $animal^{-1}$ d^{-1})
WI1-Winter	0 (frozen)	--	--	--	15	63	15		894	17
WI2-Winter	0 (frozen)	--	5	82	11	109	16		1662	9.6
WI3-Winter							28	174	3185	8.8
WI1-Summer	54	106	--	--	30	77	84		903	93
WI2-Summer	103	143	31	104	70	180	204		2198	93
WI3-Summer							330	157	3300	100
WI1-Fall	20	137	--	--	34	181	54		910	59
WI2-Fall	71	124	20	99	44	138	135		2788	48

[a] Milking and dry cows, plus heifers.

The above calculation has three weaknesses: a potential bias due to the neglect of low windspeed periods in the direct observations; different source components were measured during different days and different weather conditions; and the emission measurements are not continuous. An alternative is to use the regression models developed for each component (eqs. 3, 5, and 6) to calculate emissions over the complete 10-day observation period (DOY 168-178). This calculation includes low windspeed periods, with emissions calculated by extrapolating the regression models below the windspeed threshold for our direct observations. This calculation gives an emission rate of 95 (lagoons) + 28 (sand separator) + 70 (barns) = 193 kg NH_3 d^{-1}. This is 5% lower than the calculation using direct observations, a difference we attribute primarily to the windspeed bias in the direct observations. Given the reassuring similarity in outcome of the two calculations, we hereafter report only emissions calculated from directly measured data.

Emissions from All Farms

Below is a brief summary of the emission results for each farm-season. Results are summarized in table 1.

WI1-Winter. This farm has two barns and two lagoons. During our visit, the lagoons were initially frozen and we did not detect any emitted NH_3. We assume $Q_{Lag} = 0$ is the normal winter lagoon state (but late in our visit the lagoons began to melt, and we calculated an instance of Q_{Lag} = 3 kg NH_3 h^{-1}). Lasers placed east and north of the farm gave barn emissions during westerly and southerly winds. There is substantial variability in barn emissions over the measurement period, but this variability is not well correlated with either windspeed (r = –0.24) or outdoor air temperature (r = 0.24). There is a daily cycle in emissions (fig. 8), and from this average cycle we calculate Q_{Brn} = 15 kg NH_3 d^{-1}. This turns out to be a relatively high winter emission rate compared to the other farms. We attribute this to mild winter temperatures during our visit.

WI2-Winter. The WI2 lagoons were frozen throughout our visit, and we assume $Q_{Lag} = 0$. From a laser line just downwind of the sand separator, we calculate an average Q_{SS} = 5 kg NH_3 d^{-1}. Separator emissions were only measurable during the daytime (fig. 8). The Q_{Brn}, as measured by lasers east, north, and southeast of the barns, shows large variability but with no consistent daily cycle (fig. 8) and no strong correlation with windspeed (r = 0) or outdoor air temperature (r = –0.13). The average Q_{Brn} = 11 kg NH_3 d^{-1}. Summing the barn and separator gives whole-farm emissions of 16 kg NH_3 d^{-1}, with the barns emitting 69% of the total.

WI3-Winter. Here we calculate only whole-farm emissions from a laser-line east of the farm. The two lagoons were mostly frozen, but one corner of each was open due to vigorous pumping. We treat the barn, and unfrozen portions of the lagoons plus the separator, as separate emission sources. Whole-farm emissions are 28 kg NH_3 d^{-1}. (This calculation assumes that 67% of emissions are from the barns. If the barns give 50% of the total emissions, then the whole-farm calculation is 26 kg d^{-1}; at 75%, the calculation is 30 kg d^{-1}.) We see little evidence of a daily cycle in emissions (fig. 8) and a very weak correlation with windspeed (r = 0.2) and outdoor air temperature (r = 0).

WI1-Summer. Laser lines and anemometers were positioned at several lagoon-edge locations to measure Q_{Lag} from the north lagoon (barn waste) and south lagoon (parlor wash water). It is not surprising that the north lagoon, with its greater manure content, had about three times the emissions of the south lagoon (fig. 8). The Q_{Lag} from both lagoons show a strong daily cycle somewhat modulated by windspeed. Barn emissions, as measured from laser-lines north of the farm, show a similarly strong daily cycle (fig. 8). Whole-farm emissions are 84 kg NH_3 d^{-1} (64% from the lagoons, 36% from the barns), a six-fold increase over the winter rate.

WI2-Summer. This analysis was described earlier. In summary, WI2-Summer emissions are 212 kg NH_3 d^{-1} (49% from lagoons, 33% from barns, and 18% from the separator). This is 2.5 times greater than the summer emissions from WI1, which reflects almost exactly that WI2 has 2.4 times the number of animals as WI1.

WI3-Summer. We had the extra difficulty of making measurements in a 1.5 m tall corn field surrounding the farm (the bLS model is restricted to observations well above a plant canopy). Lasers were placed on ladders and positioned so that C_L was measured at an average of 3.5 m (E-W line) and 4.5 m (N-S line) above the corn. As part of our summer strategy, we also measured emissions from the east lagoon (to reduce the effective x_s at the farm). Whole-farm emissions at WI3 show a strong daily cycle (fig. 8) and correlation with windspeed (r = 0.69) and outdoor air temperature (r = 0.61). We calculate whole-farm emissions of 330 kg NH_3 d^{-1}, which is more than 10 times the winter rate. Direct measurements from the east lagoon indicate that it alone emits 30% of the farm total.

WI1-Fall. The north lagoon had been emptied before our visit and had 50% to 60% of its full surface area. Even with

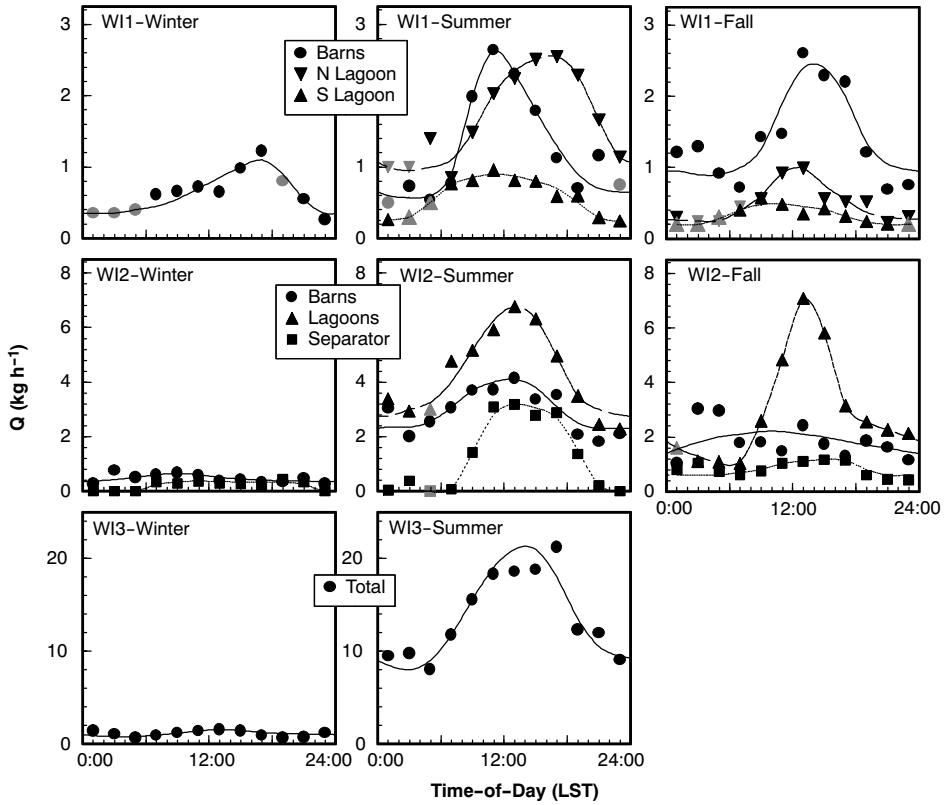

Figure 8. Farm emissions *Q* plotted versus time-of-day for the three farms for three seasons. Black symbols represent average emissions in 2 h blocks (gray symbols are estimates). Lines represent hand-drawn curves to the data.

this reduction in size, emissions from the north lagoon (barn waste) were about 50% larger than from the south lagoon (parlor wash). Total lagoon emissions are about a third of the summer values. The fall barn emissions are of similar magnitude to the summer values, and display a similarly strong daily cycle. We calculate whole-farm emissions of 54 kg NH_3 d^{-1} (37% from the lagoons, and 63% from the barns). Overall, the fall emissions are roughly two-thirds the summer rate but more than three times the winter rate. The dominance of barn emissions over lagoon emissions is the reverse of the summer situation, and the reverse of both the summer and fall situations at WI2. Perhaps lagoon emissions were reduced due to the north lagoon having been recently emptied (although there is no evidence for a reduction of whole-farm emissions compared to WI2-Fall).

WI2-Fall. Because of unfavorable wind directions, we had an abbreviated lagoon measurement period. Over a 36 h interval, we see a strong daily cycle in Q_{Lag} (fig. 8), with strong correlations between Q_{Lag} and windspeed (r = 0.62) and outdoor air temperature (r = 0.81). The barn and sand separator have a weaker daily emission cycle than was seen in the summer. Total farm emissions are 135 kg NH_3 d^{-1} (53% from the lagoons, 33% from the barns, and 15% from the separator). As was the case at WI1, the fall emission rate is about two-thirds the summer rate, but about five times the winter rate.

WI3-Fall. Fall emissions at WI3 could not be determined due to manure spreading in the fields surrounding the farm. This created a strong confounding source of NH_3 between the farm and our downwind lasers.

Discussion

Variability Between Farms

Daily NH_3 emissions from the three dairies show large variability over the study year, from 15 kg d^{-1} at WI1-Winter to 330 kg d^{-1} at WI3-Summer (table 1). This variability is well explained by two factors: size of the farm (number of animals) and the season. When emissions are expressed on a per-animal basis (table 1), there is much similarity between the farms on a seasonal basis. Emissions range from 9 to 17 g NH_3 $animal^{-1}$ d^{-1} in winter, increasing to 93 to 100 g $animal^{-1}$ d^{-1} in the summer, and then decreasing to 48 to 59 g $animal^{-1}$ d^{-1} in the fall. Accounting for farm size largely erases the inter-farm differences and highlights the dramatic seasonal effect on emissions. On average, the summer emissions are almost ten times the winter rate, and the fall rate is a little more than half the summer rate.

Not only is there large seasonal variability in emissions, but also strong daily variability. Figure 9 displays the average daily emission curves (on a per-animal basis) for each farm and season as the sum of all emission components. Two things are apparent. First, we again see the dramatic seasonal ordering of emissions and the similarity between the farms for a given season, but we also see a characteristic daily emission cycle. There is an approximate three-fold increase in mid-day emission rates compared with nighttime values (the exception being WI3-Winter). The emission components (barns, lagoons, sand separators) generally show a similar cycle. The sand separator at WI2 is the most extreme: in summer and winter, the mid-day emission rates approached the level of the barns, but nighttime emissions were immeasurably small.

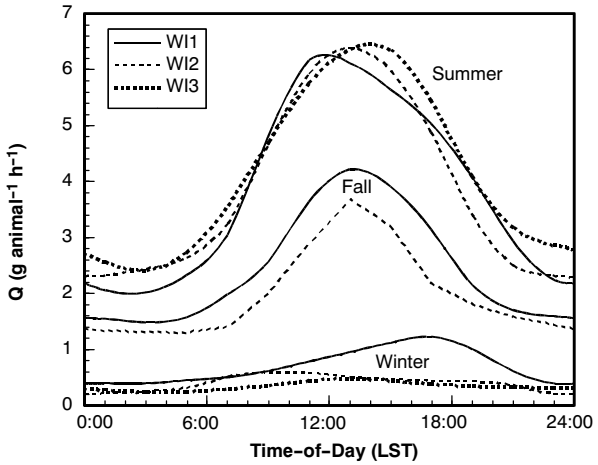

Figure 9. Average daily NH_3 emissions Q from the three farms (per animal) for three seasons. Curves are the sum of the component curves illustrated in figure 8.

The degree of consistency in the daily emission cycles at the three farms is surprising, particularly in summer and fall. Consider some of the differences between farms. One large difference is that two of the farms use sand separators and one does not. There are also differences in the allocation of emissions at the farms, e.g., in the fall, the lagoons at WI1 contribute 37% of the total emissions, while at WI2 the lagoons contribute 53%. We also found that the emission components at the farms have different sensitivities to windspeed or time-of-day, e.g., fall barn emissions at WI1 show sensitivity to windspeed, while at WI2 there was none. It appears that these inter-farm differences have little impact on the pattern of daily emissions.

Yearly Average Emission Rates

We estimate yearly whole-farm emissions using the seasonal daily emission rates. Assuming that spring and fall rates are equivalent, we calculate a three-season annual average as:

$$Q_{avg} = 365\left(Q_{winter} + Q_{summer} + 2\,Q_{fall}\right)/4 \qquad (7)$$

At WI1 and WI2, the yearly emissions are thus 19 and 45 t NH_3 $year^{-1}$. At WI3 we do not have a fall measurement, so we instead make a two-season calculation for all the farms, using the average of the winter and summer rates. This gives Q_{avg} = 18, 40, and 65 t NH_3 $year^{-1}$ at WI1, WI2, and WI3, respectively. (Interpreting WI2 yearly emissions is complicated by the change in animal numbers over the study, i.e., 68% increase.) A more useful comparison is the per-animal emissions. From the two-season calculation, we get yearly emissions of 20, 19, and 20 kg NH_3 $animal^{-1}$ $year^{-1}$ at WI1, WI2, and WI3, respectively.

The similarity in yearly per-animal emissions at the farms is striking. While the study farms differ in many details (e.g., use of a sand separator, etc.), these differences appear to be of secondary importance compared to their shared management system (i.e., naturally ventilated free-stall barns, sand bedding, regularly scraped barn floors, open lagoons). The inter-farm agreement in emissions can also be taken as an indication of the success of both the bLS inverse-dispersion technique and our sampling strategy of a series of 10-day campaigns as giving a representative emission record.

Table 2. Barn NH_3 emissions at WI1 and WI2. The average outdoor air temperature (T_{air}) is given for the study periods. Lower portion of table gives barn emission rates measured in other studies.

Barn-Season	Barn Emissions (kg day^{-1})	Barn Emissions per Animal (g $animal^{-1}$ day^{-1})	T_{air} (°C)
WI1-Winter	15	17	1.6
WI2-Winter	11	6.6	-6.4
WI1-Summer	30	33	18.8
WI2-Summer	70	32	21.2
WI1-Fall	34	37	3.9
WI2-Fall	44	16	11.9

Other Studies	Emissions per animal or animal unit[a]	Details
Isermann (1994), quoted in Amon et al. (2001)	16.6 g AU^{-1} day^{-1}	Loose housing (no animal stalls)
Demmers et al. (1998)	31.6 g AU^{-1} day^{-1}	Naturally ventilated cubicle barn (free stall) with scraped floors: Jan. to May in U.K.
Snell et al. (2003)	38 to 85 g $animal^{-1}$ day^{-1}	Four naturally ventilated barns: winter in Germany
Pfeiffer et al. (1994), quoted in Monteny and Erisman (1998)	25 g $animal^{-1}$ day^{-1}	Naturally ventilated cubicle barn (free stall) with scraped floors
Groot Koerkamp et al. (1998)	24 to 48 g $animal^{-1}$ day^{-1}	Several cubicle barns (forced ventilation): different seasons in Europe
Powell et al. (2008a)	13.4, 24.7, and 25.4 g $animal^{-1}$ day^{-1}	Heifers in tie-stall barn chamber in Wisconsin during winter, summer, and fall
Powell et al. (2008b)	6.7, 18.8, and 8.4 g $animal^{-1}$ day^{-1}	Lactating cows in tie-stall barn chamber in Wisconsin during winter, spring, and early fall

[a] Animal unit (AU) = 500 kg animal.

Comparison with Other Studies

How do our emission measurements compare with other studies? We are unaware of other whole-farm efforts like ours, but we can compare our barn and lagoon observations with literature values. Table 2 lists barn emissions from WI1 and WI2 on a per-animal basis, together with comparable values from other studies. Except for the cold WI2-Winter period, our measurements fall within the range of previous studies of naturally ventilated free-stall barns, as well as observations from forced ventilation and tie-stall barns.

Barn emissions at WI1 and WI2 are lowest in winter, as expected given other seasonal studies (e.g., Groot Koerkamp et al., 1998). It is interesting that the seasonal ordering at WI1 match almost exactly that found by Powell et al. (2008a, 2008b) in chamber measurements in Wisconsin, with winter rates about one-half to one-third lower than those in the spring and summer. However, at WI2, the winter emissions are only 20% of summer rates. We attribute this to the cold winter temperatures during our WI2 visit. The cold reduces barn emissions by slowing the chemical and biological reactions that lead to NH_3 production from urine and feces, and by reducing the ventilation rate (indirectly) as barn curtains are closed to conserve heat. The curtains at WI1 were 25% to 75% open during our visit, while at WI2 they were closed. A daily emission cycle was also found in the measurements of Powell et al. (2008a, 2008b). In these tie-stall barn studies,

Table 3. Lagoon NH_3 emissions for summer and fall. The average outdoor air temperature (T_{air}) is given for the measurement periods. Lower portion of table gives emissions measured in other studies.

Lagoon-Season		Lagoon Emissions (kg d^{-1})	Areal Average (g m^{-2} d^{-1})	T_{air} (°C)
WI1-Summer	Parlor-wash	14	3.5	17.5
	Manure	40	8.7	19.7
WI2-Summer	Manure and wash	103	7.7	22.0
WI3-Summer	Manure and wash	100	6.1	20.2
WI1-Fall	Parlor-wash	7.8	2.3	3.4
	Manure	12	4.4	3.2
WI2-Fall	Manure and wash	71	6.7	15.2

Other Studies	Emissions (g m^{-2} d^{-1})	Details
Zhao et al. (2007)	0.5 to 15	Dairy lagoon measured at noon over the year at an Ohio dairy (chamber)
McGinn et al. (2008)	5.1	Quasi-continuous summer measurements at dairy lagoon in Canada (bLS technique)
Misselbrook et al. (2005)	2.1 to 10.4	Variety of crusted cattle slurry tanks in the U.K. (chamber)
Sommer et al. (1993)	4.2, 6.3	December-June and July-September emissions from cattle slurry in open tank in Denmark (chamber)

the daytime (1000 to 1500 h) emissions were 10% to 30% greater than nighttime (1900 to 0500 h) emissions. Here we observe a much larger daily range, with daytime barn emissions roughly two to three times the nighttime rate.

There is also agreement between our lagoon measurements and those of previously reported studies (table 3). The McGinn et al. (2008) study is the most comparable to ours, as they measured emissions from a dairy lagoon over many days (day and night) to calculate average summer emissions. The average emissions of all our summer lagoons is 6.5 g NH_3 m^{-2} day^{-1}, somewhat higher than the 5.1 g m^{-2} day^{-1} reported by McGinn et al. (2008). Given the potential differences between the two studies (lagoon size, chemistry, climate, animal numbers, etc.), we consider these rates to be surprisingly similar.

The general accord between our measurements and those reported in other studies is encouraging. It provides yet another indication that the inverse-dispersion technique and our measurement strategy provide accurate emission measurements.

Summary and Conclusions

A bLS inverse-dispersion technique was used to measure NH_3 emissions from three broadly similar CAFO dairy farms in Wisconsin. Farm emission rates varied from 15 kg NH_3 d^{-1} (WI1-Winter, the smallest farm) to 330 kg d^{-1} (WI3-Summer, the largest farm). Inter-farm variability was largely explained by farm size (animal population). On a per-animal basis, the yearly emission rates at the three farms were estimated to be 20, 19, and 20 kg NH_3 $animal^{-1}$ $year^{-1}$. The emissions showed variability on two important time scales: seasonal and daily. Summer emissions were almost ten times the winter rates, and mid-day emission rates were approximately three times those at night. The lagoons were the largest emitters during the summer and fall, representing 37% to 63% of the farm total. During winter, the lagoons were frozen, and emissions were immeasurably small. The similarity in emission rates at the three study farms (on a per animal basis) suggests that our observations are representative of modern free-stall dairies in Wisconsin.

The bLS measurement technique proved well-suited to our study. With rather modest equipment and labor resources (one person on-site), we were able to easily move the necessary equipment and measure emissions from the variety of sources at each farm, and to quickly move from one farm to the next. There was no disruption to the farm management during our measurements. A key to using the bLS technique was the selection of study farms located in relatively open terrain, allowing us to place sensors in convenient locations while meeting the theoretical criteria for the bLS technique (i.e., laser lines located at least 10 barns heights, and one source separation distance, downwind of the farm). We believe the overall agreement in emissions measured at the three farms, together with the general agreement between our calculated emissions and those from previous studies, confirms the utility of our measurement strategy.

Acknowledgements

This study would not have been possible without the assistance of the three commercial dairy operators and staff. We thank them for their generous support. Funding and equipment were provided by the USDA-ARS Dairy Forage Research Center, the Canadian Foundation for Climate and Atmospheric Sciences, the University of Alberta, and the University of Georgia. Thanks are extended to W. K. Coblentz and W. Jokela for their help in administering and organizing the research, to S. R. Struss and D. Grande for their assistance in locating suitable farms, and to S. M. McGinn for welcome ideas in data analysis.

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