Ammonia emissions from broiler production in the San Joaquin Valley[1]

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ABSTRACT Ammonia is the primary basic gas in the atmosphere and has the most important role in the neutralization of atmospheric acids generated by fossil fuel combustion. The reaction product forms a NH_4^+ aerosol, which is a major component of atmospheric particulates. These NH_4^+ particulates are part of atmospheric haze and may be transported long distances from the production site before returning to the surface by dry deposition or scavenged by precipitation. Animal production produces a significant component of anthropogenic NH_3 emissions and the National Academy of Sciences concluded that NH_3 emissions estimates from animal feeding operations have not been characterized sufficiently, leading the US Environmental Protection Agency to institute studies in the United States to obtain NH_3 emissions from animal feeding operations under the US Environmental Protection Agency Air Consent Agreement. The objective of this study is to obtain additional broiler NH_3 emissions estimates using a backward Lagrangian stochastic technique. This technique uses NH_3 concentrations measured upwind and downwind of the farm, wind observations, and atmospheric dispersion model calculations to obtain whole-farm emissions. Ammonia emissions were low at bird placement and increased steadily after about the third week of growth. At the end of the flock (47 d, ~297,000 birds), cumulative emissions for the flock cycle period were 0.016 kg of NH_3·bird^{-1}·flock^{-1}. Between-flock emissions, including bird harvest, cleanout, temporary storage of litter outside of the buildings, and downtime (buildings closed), added another 0.003 kg of NH_3·bird^{-1}·flock^{-1}. Emissions from this broiler farm were less than from some eastern US broiler farms but were comparable to broiler farms in Europe. Based on the results of this study and a similar winter study at this same farm, total flock wintertime and summertime (flock cycle plus between-flock) NH_3 emissions from this farm represented 7.8 and 8.3% of feed N as NH_3-N, respectively, or an annual average of 8.1%.

Key words: ammonia, emission, broiler, inverse dispersion, animal feeding operation

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INTRODUCTION

Ammonia is a colorless gas under standard conditions and is the primary basic gas in the atmosphere. It therefore has the most important role in the neutralization of atmospheric acids generated by the oxidation of SO_2 and NO_x from fossil fuel combustion. The reaction product forms an aerosol, NH_4^+, which is a major component of atmospheric particulates and is scavenged by precipitation (Asman et al., 1998). Other organic N compounds exist in the atmosphere, but their concentrations are generally negligible by comparison (Van der Eerden, 1982). Most NH_3 emitted to the atmosphere is of anthropogenic origin (Bouwman et al., 1997) mainly from agricultural production, with a large percentage of the total from animal production. Other anthropogenic NH_3 emissions include motor vehicles that have been shown to be comparable to agricultural emissions in some geographical locations (Fraser and Cass, 1998). Most agricultural sources tend to be scattered both temporally and spatially and most of the emitted NH_3 may be absorbed by surrounding cropping and natural ecosystems (Harper and Sharpe, 1995; Harper et al., 2004b; Harper, 2005); however, confined animal production tends to be concentrated in relatively small geographical areas and may increase localized N loading.

Ammonia emissions are the result of complex physical and chemical processes (Freney et al., 1983; Harper, 2005) with the emission rate related to 4 factors: NH_4^+ concentration of the source, temperature of the source, pH (hydrogen ion concentration where $[H^+] = 1 \times 10^{-pH}$ of the solution) of the source, and the effectiveness of turbulent transport of the NH_3 away from the source. Any measurement procedure that alters these

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factors from the natural ambient state will provide biased measurements of the ambient NH_3 emission rates. For NH_3 in particular (a diffusive gas in which turbulent transport away from the source is the rate-limiting process), any measurement technology that interferes with the turbulent transport process can result in very large errors. Mass-flow (biological) gas emissions such as CH_4, CO_2, or NO_2 are less soluble in solution and are less affected by turbulent transport.

Measurements of NH_3 emissions from animal feeding operations have been variable, leading the National Academy of Sciences to state that "...current data in the literature were not consistent and more (NH_3 emissions) work needed to be performed" (NRC, 2003). Indeed, some reports calculated that more N was lost from a farm system through gaseous NH_3 than was gained as N entering into the system (Eklund and LaCosse, 1995), plainly an impossibility, assuming the budget is performed over a meaningful averaging interval. The US Environmental Protection Agency (**USEPA**) in the National Emissions Inventory—Ammonia Emissions from Animal Husbandry estimates that 23% of input N leaves poultry production houses as NH_3-N (USEPA, 2004). It is crucially important for the poultry industry to either validate or refute these emission estimates given the recent emphasis on regulations such as the Emergency Planning and Community Right-to-Know Act and the USEPA Air Consent Agreement with their possible penalties for systems not in compliance. Also, accurate estimates of emissions and downwind concentrations would be beneficial for growers for evaluating nuisance or health complaints, or both, from neighbors.

The objective of this study was to obtain summertime NH_3 emissions measurements from poultry production in the San Joaquin Valley. This summer data can then be compared with a previous study of winter emissions at the same farm (Harper et al., 2009b) and with other studies of poultry emissions in different locations (in the humid eastern United States).

MATERIALS AND METHODS

Poultry Farm

Ammonia emissions were measured at a broiler facility in the San Joaquin Valley of California. This farm consisted of sixteen 12 × 152 m (40 × 500 ft) tunnel-ventilated, dropped-ceiling houses, each with an average initial placement of 19,295 birds [0.10 $m^2 \cdot bird^{-1}$ (1.04 $ft^2 \cdot bird^{-1}$)]. Birds on this farm are typically grown to an age of 47 d and an approximate weight of 2.7 kg (6 lb). For this particular study, bird placement occurred on July 30, 2009 [day of flock (**DOF**) = 0, wk 1], with a small number (3 houses) having been placed on July 28. Harvest of the birds occurred on September 14, 2009 (DOF = 47, wk 7). Each house was equipped with ten 1.2-m (48-in.) fans [each providing a ventilation rate of approximately 595 $m^3\ min^{-1}$ (21,000 $ft^3 \cdot min^{-1}$) at 0.25-cm (0.1-in.) pressure] and two 0.9-m (36-in.) fans [255 $m^3 \cdot min^{-1}$ (9,000 $ft^3 \cdot min^{-1}$) at 0.25-cm (0.1-in. pressure)]. The 1.2-m fans were located in the side walls (5 on either side) immediately next to the end wall. Air intake was on the opposite end of the house through an evaporative cooling pad system [approximately 1.2 × 18 m (4 × 60 ft on each side)]. The 0.9-m fans were located one on each end wall.

Rice hulls were used as bedding material in the houses. The standard practice on this farm was to clean out the brooding end of the houses after each flock (the end opposite the 1.2-m ventilation fans). The nonbrooding ends were decaked after each flock and a layer of rice hulls was placed over the decaked litter. The houses were completely cleaned out after each third flock. This study was conducted on litter that was 2 flocks old [i.e., being used as the third (and thus final) flock of the sequence].

Inverse Dispersion Technique

Due to the chemical properties of NH_3, and the interaction between environmental variables and NH_3 emissions, the most appropriate techniques for measurement and calculation of emissions are noninterfering techniques (Denmead et al., 1982; Harper et al., 2000; Harper, 2005). In this study, we used a noninterfering inverse-dispersion analysis to calculate emissions. Using averaging intervals of 15 min, NH_3 concentration is measured upwind and downwind of the farm, together with meteorological information, and this information is used in conjunction with a digital (global positioning system) map of the farm indicating the locations of the NH_3 emissions sources (e.g., houses) and the gas detector locations. A backward Lagrangian stochastic (**bLS**) dispersion model calculates the farm emission rate best explaining the measured downwind concentration (Flesch et al., 2004). The bLS technique has been examined in several studies and in several different environmental and geographical (terrain and structural) settings. In this study, we will assume a measurement uncertainty of 10% (see Table A1, Appendix; Harper et al., 2009a). Error limits are based on published studies using tracers to verify accuracy of the technique (see Appendix, Harper et al., 2009a). Table A1 in the Appendix summarizes several tracer studies on the accuracy of the bLS technique for calculating emissions. Accuracy is indicated by the gas recovery, which is the percentage ratio of the bLS calculated emissions to actual emissions (i.e., $Q_{bLS}/Q_{release} \times 100$). These studies had an average recovery of 99% with a SD of 5%. We conclude that with appropriate instrument placement and data filtering (as discussed in detail in these studies) one can expect a nominal bLS accuracy of 100 ± 10% (±2 SD – a span that includes 95% of a Gaussian distributed population). This would be the accuracy of an average of multiple measurement periods.

NH_3 Concentration Measurements

Ammonia concentrations were measured upwind and downwind of the farm using open-path, line-averaging laser spectrometry (Boreal Laser Inc., Spruce Grove, Alberta, Canada). These open-path lasers (**OPL**) use the principle of a single-line, fixed-wavelength radiation absorption within the infrared spectrum. Such a narrow absorption line, specifically designed for NH_3, avoids mutual absorption interference of other gases such as CO_2, CH_4, and water vapor. Aerosols and dust did not affect absorbed radiation in the wavelength of NH_3. The almost monochromatic nature of the laser emission enables tunable diode laser absorption spectroscopy to isolate individual gas absorption lines. Wavelength modulation spectroscopy techniques are used to maximize sensitivity. By modifying the diode injection current, the laser wavelength can be scanned across the gas absorption line, resulting in a harmonic distortion signal whose size is dependent on the amount of gas present and not on total light intensity, which may have been modified by dust, rain, or fog. The OPL were factory-calibrated immediately before the study and further reference-calibrated in the field during the study to make sure they were providing the same concentration for the same gas concentrations. Calibration drift was checked during the study, but previous studies using the lasers also had not shown drift. Simultaneous path-average NH_3 concentrations were measured (Figure 1) approximately 50 m from the houses on the north side of the farm (the prevailing upwind side) and 125 m from the houses on the southeast side (prevailing downwind side). Path lengths were 436 m for the laser north of the farm and 451 m southeast of the farm. The average path heights were 1.7 and 3.3 m above ground (north and southeast lasers, respectively).

Wind Measurements

A 3-dimensional sonic anemometer (CSAT-3, Campbell Scientific Inc., Logan, UT) was used to measure the wind and turbulence concurrent with the NH_3 concentration measurements. Wind statistics (at 2.7 m height) were measured in a pasture north of the farm such that there were no obstructions upwind of the sonic anemometer. This location was selected so as to give

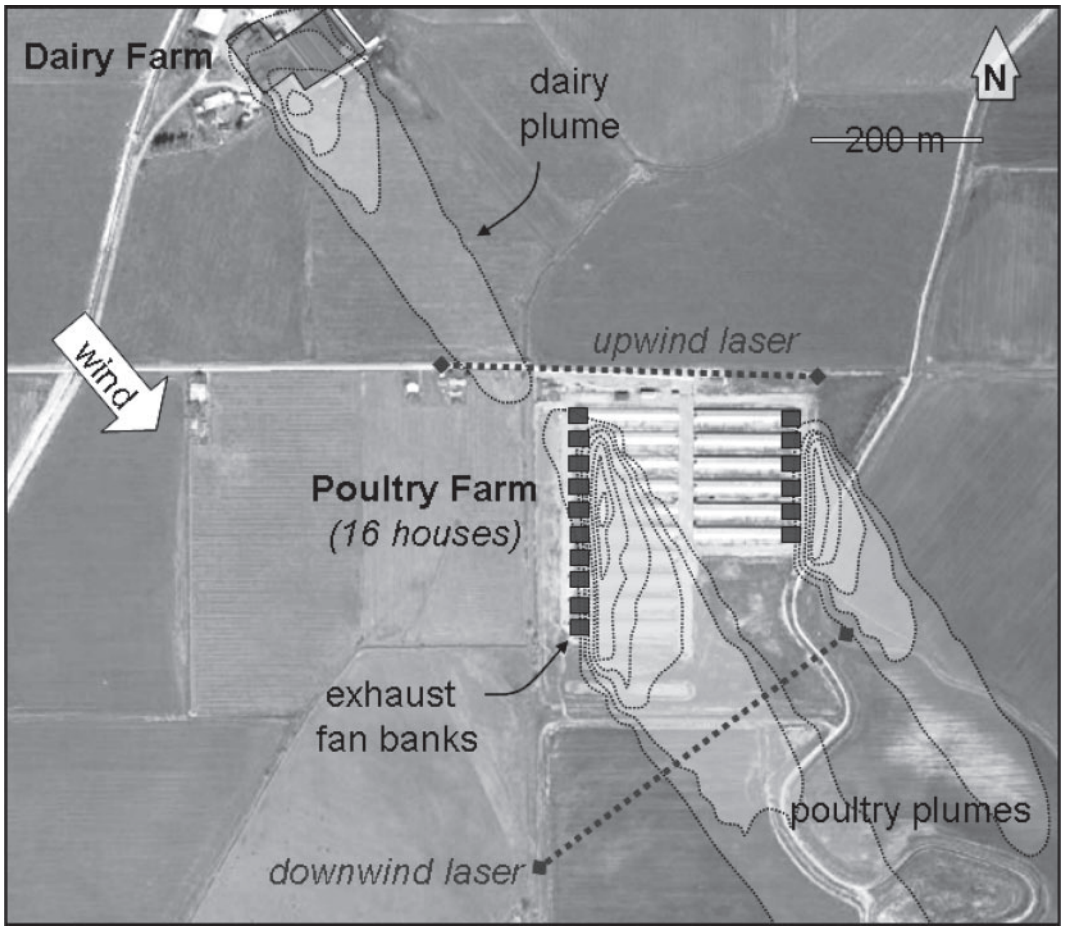

Figure 1. The study farm consists of 16 poultry houses, with each house having a bank of exhaust fans at one end (considered the emission sources). Prevailing winds during the summer were from the northwest and upwind and downwind laser paths are shown. Hypothetical plumes of NH_3 are shown emanating from the poultry houses and a dairy farm upwind of our site (dashed lines represent concentration contours in the air downwind of the farms). Open-path laser path lengths were 431 m north and 451 m southeast of the farm.

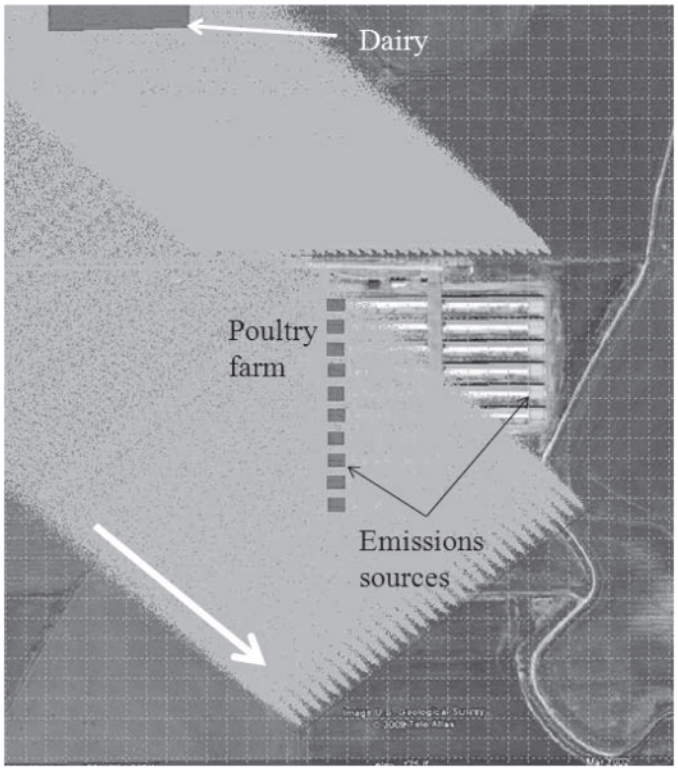

Figure 2. Upwind touchdown plumes projected (backward in time) from the laser detector light path lines (the cloud of gray dots extending upwind of the laser paths maps the ground area where emissions influence the concentration seen by the laser). A criterion in the analysis is that the touchdown cloud extending upwind from the laser should cover at least 40% of the house exhausts (emission sources) as evaluated by the inverse Lagrangian dispersion model touchdowns. Notice that (for northwest winds) the upwind laser lies within the influence of the dairy farm and would respond to (any) NH_3 emissions from it.

good measurements of the ambient winds in the area, unaffected by the farm structures (as recommended by Flesch et al., 2005b). When making bLS dispersion calculations, the wind is represented using standard micrometeorological formulae as defined by the following measured parameters: friction velocity ($u*$; characterizing the overall windspeed), the Obukhov stability length (L; a measure of the thermal stratification of the atmosphere), and the surface roughness length (z_0; a measure of the aerodynamic roughness of the terrain). Details of this wind specification are given in Flesch et al. (2004).

bLS Dispersion Model

WindTrax (Thunder Beach Scientific, Nanaimo, British Columbia, Canada) software is used to make the calculation relating downwind concentration increase above the background to the emission rate. This software combines the bLS model described by Flesch et al. (2004) with an interface allowing the farm emission sources and concentration sensors to be conveniently mapped. In the bLS model, thousands of model trajectories are calculated upwind of the laser line for the prevailing wind conditions. The important information relating the concentration to the emissions is the set of trajectory intersections with ground (touchdowns), and the needed concentration-emission rate (C-Q) relationship is determined by those touchdowns according to

$$Q = \frac{\left(C - C_{\mathrm{b}}\right)}{\frac{1}{n}\sum\left|\frac{2}{w_0}\right|}, \qquad [1]$$

where Q is the farm emission rate ($\mathrm{kg \cdot m^{-2} \cdot s^{-1}}$), C is the downwind concentration ($\mathrm{kg \cdot m^{-3}}$), C_{b} is the upwind background concentration ($\mathrm{kg \cdot m^{-3}}$), n is the number of computed bLS trajectories, w_0 is the vertical velocity of the trajectory at each touchdown point ($\mathrm{m \cdot s^{-1}}$), and the summation covers all touchdowns occurring within the designated source area. The touchdowns map the concentration footprint (i.e., the ground area where emissions influence C). For this report, the areal emission rate, Q, calculated in the above formula is multiplied by the source area and is reported as an integrated whole-farm emission rate (with units of kg of $NH_3 \cdot h^{-1}$.

The study farm is represented as a collection of surface area sources corresponding to the exhaust ends of the houses (Figure 2; the 20 m of the house end where the fans are located was designated as the emission source). The emission rate, Q, was calculated using n = 750,000 trajectories. Our assumptions about background NH_3 concentration, C_{b}, are discussed in detail in the next section.

For reasons that are easy to understand (e.g., unsuitable wind direction, such that the wind does not carry the NH_3 plume to the detector, or insufficient wind, such that the spread of the NH_3 plume cannot be reliably modeled), not all observation periods are conducive to a reliable estimation of the emission rate, Q, and therefore the filtering strategy of Flesch et al. (2005b) was used. Three criteria identify periods when the bLS dispersion model is likely to be inaccurate and such periods were not used if:

- $u* \leq 0.15$ $\mathrm{m \cdot s^{-1}}$ (low winds),
- $|L| \leq 10$ m (strongly stable/unstable atmospheric stratification), and
- $z_0 \geq 0.2$ m (which is unrepresentative of the local terrain. Note: we speculate that periods for which such improbably large occasional roughness lengths occurred may be associated with curious cattle crowding near, and thus disturbing, the anemometer in their pasture.).

For some wind directions, the farm plume only glances the downwind laser line. This can lead to inaccuracy in the Q calculations in several ways: the plume edge is associated with greater uncertainty in dispersion predictions due to the difficulty of modeling lateral disper-

sion and slight errors in wind observations (particularly wind direction) can result in dramatic errors in Q. Therefore, periods are not used in which the laser touchdowns (i.e., measurement footprint) cover less than 40% of designated source area (determined as the fraction of source pixels displayed as touchdowns in the WindTrax display; see Figure 2 for example).

The bLS calculation of Q implicitly makes the approximation that NH_3 is a passive tracer, with no deposition to the downwind surface and no chemical transformation between the farm and the laser line. Given the short distance between the farm and the downwind laser (125 m), this approximation is acceptable.

Complication from Northwest Dairy

A bLS calculation typically requires measurement of the concentration both upwind and downwind of the farm. Emissions are then calculated from the difference between the two, and this is the assumption behind equation [1]. However, the calculations are true only when the upwind concentration represents the true background C_b (i.e., uniform over the region). If the upwind concentration is in the plume of another nearby emission source, the calculation of farm emissions is more complex than indicated in equation [1]. Although there are approximately 20 farms near our study site (400 m to 3 km distant), only those located to the northwest were of concern to us because prevailing winds were overwhelmingly from the northwest.

There is a dairy farm near enough to our study farm (400 m to the northwest, Figure 1) to require a modification in the bLS analysis. During northwest winds, any emissions from the dairy will enhance the NH_3 concentration at both our upwind and downwind laser lines. Because the dairy plume is diluted as it moves downwind, the enhancement is larger at the upwind laser. The challenge is to partition the downwind laser concentration into 1) the contribution due to C_b, 2) the contribution due to the dairy, and 3) the contribution due to the poultry farm. The following is a description of our analysis strategy.

An important first step was to ascertain the regional C_b. Measurements away from the farms indicated a midday value of approximately $C_b = 0.02$ ppm_v. The nighttime C_b was usually larger due to a reduction in the atmospheric dilution (i.e., mixing aloft) of regionally emitted NH_3. With this information, we take 2 alternative scenarios for calculating emissions. In the first, we take our upwind laser concentration as the regional C_b (which tends to be greater than 0.02 ppm_v) and calculate farm emissions using equation [1]. This assumes the dairy is not a significant NH_3 source. In the second scenario, we assume $C_b = 0.02$ ppm_v and that any increase in the upwind concentration over 0.02 ppm_v is due to dairy emissions. The upwind concentration increase over C_b (0.02 ppm_v) is used to calculate a dairy emission rate, Q_{dairy}, using equation [1]. This Q_{dairy} is then used (in the bLS model) to determine the dairy contribution to the laser concentration downwind of the poultry farm. The poultry contribution to the downwind laser is then found by subtracting C_b and the dairy contribution. Underlying (and legitimizing) this sequential approach is the fact of the additivity of the NH_3 plumes from far-distant sources (i.e., background), from the dairy, and from the poultry houses.

These 2 alternative emission calculations are made throughout the flock cycle (period when birds were in the houses), and for any one observation period, the 2 alternatives should bracket the true poultry emission rate: in one scenario we assume a maximum possible C_b (our upwind concentration measurement) and in the other we assume the minimum C_b (0.02 ppm_v). Alternatively, these 2 scenarios represent the minimum (zero) and maximum possible dairy emission cases. For every observation period, we take the average of these extremes as our best estimate of the poultry emission rate.

Calculating Average Emissions

Fifty-two days of data (47-d flock; 3 d before bird placement and 2 d after bird harvest) were taken during the study. More than 2,200 usable measurements of 15-min average NH_3 emission rates were obtained (Figure 3). This data set represents a noncontinuous time series of 15-min average emission rates, covering about 50% of the study period [missing data resulted from equipment failure, unfavorable wind conditions, and periods during which data were not collected (data downloading and instrument calibration)].

Because farm emissions increase with bird age, the emission data were analyzed on a weekly basis. To avoid a time-of-day bias (more data may be missing during either nighttime or daytime), we created ensemble-averaged daily (24 h, 0000 to 2400 h) emission curves for analysis. For each week, the emissions data are grouped by time of day. This data are then averaged into 12 two-hour blocks to cover the 24-h day. The resulting averages are then summed over the 24 h to give an average daily emission rate for that particular week.

RESULTS AND DISCUSSION

NH_3 Emission Rates

Individual 15-min emission rates during this study range from almost zero (not measurable) up to a maximum of about 15 kg of $NH_3 \cdot h^{-1}$ near the end of the flock (Figure 3). Although there were almost certainly emissions from the houses during periods of zero emissions early in the flock cycle, the downwind concentration increases above the background were so low as to be not measurable by the OPL. Two trends are identifiable in the emission time series. One is the general increase in daily emissions after wk 3 as the flock age

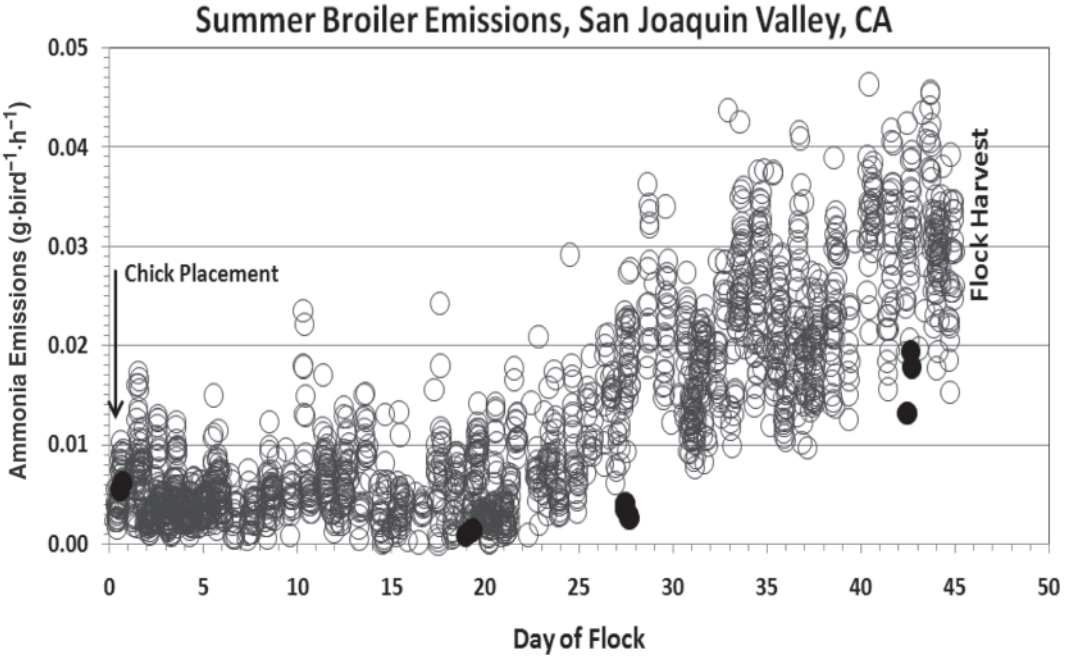

Figure 3. Fifteen-minute emissions observations collected during the production period measurements (open circles). Superimposed on the figure (solid circles) are limited measurements previously taken on this farm using a mass-balance technique (after Summers, 2005).

progressed and birds increased in size and feed consumption. But we also observed a diurnal trend in emissions, with minimum emissions at night and maximum emissions in the late afternoon (Figure 4). We attribute this trend to the diurnal temperature cycle. The diurnal difference between ambient minimum and maximum temperatures ranged from 18 to 40°C during the flock measurement period. This, in turn, corresponded to a strong diurnal difference in the ventilation rate in the houses. The observations also seem to show that emissions are sensitive to overall weather patterns. For example, during the period from DOF 29 to 32 (Figure 3), we find a short-term trend of decreasing emissions, which would appear to be at odds with the trend of increasing emissions with the age of broilers. This decline follows a decrease in ambient air temperature as a cooler air mass moved over the region.

Figure 4 shows the results of weekly calculations of the diurnal emission pattern. These weekly curves show (similarly to Figure 3) that farm emissions are relatively low for the first 3 wk of the flock but thereafter increase with time. These curves also confirm the presence of a diurnal trend, with midday maximum emissions and nighttime minimum. In some weeks, the diurnal range is large (wk 4), but in other weeks, the range is relatively small (wk 7). The lower diurnal range at

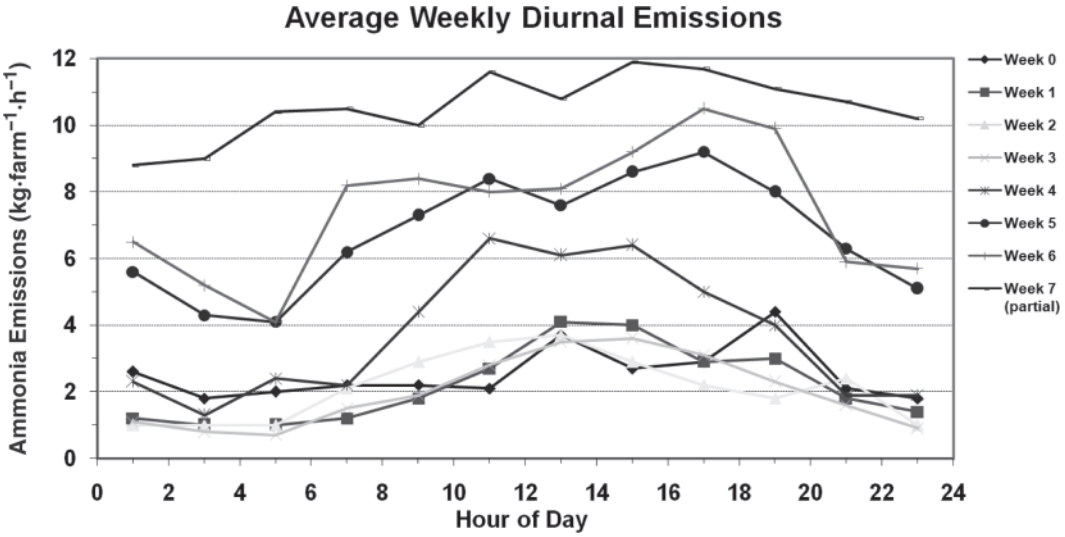

Figure 4. Weekly diurnal emissions curves based on 2-h block averages.

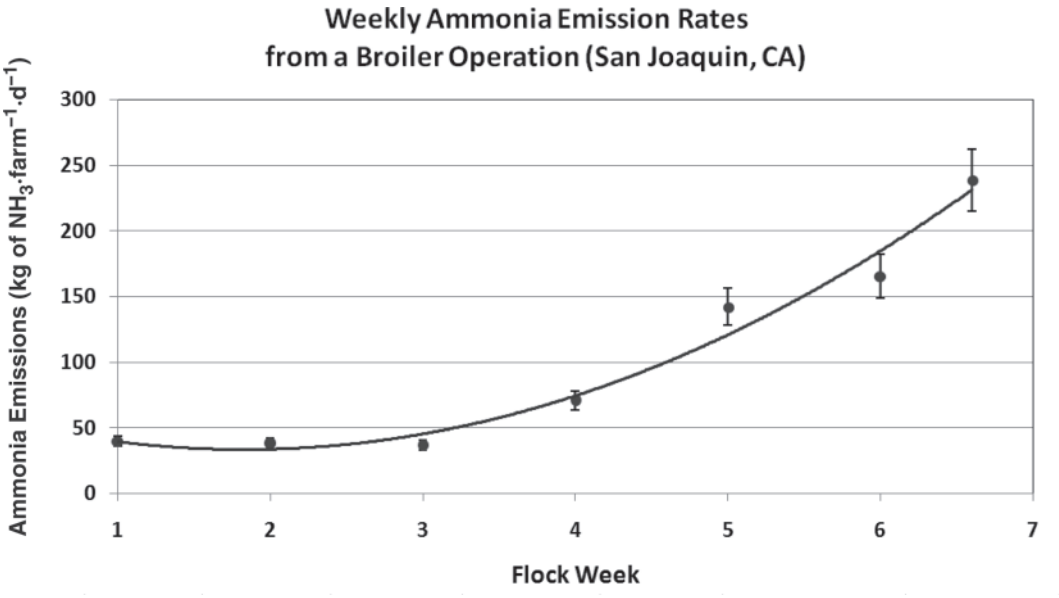

Figure 5. Weekly average summer flock emission rates for a farm of about 310,000 birds in the San Joaquin Valley, California.

the end of the flock is probably due to more similar day versus night ventilation rates to keep the birds cool as they age (earlier in the flock, there are larger diurnal differences in ventilation rates).

The weekly diurnal emission curves illustrated in Figure 4 provide the basis for our estimates of emissions over the full flock cycle. Each curve is integrated to provide a time-of-day weighted weekly average emission rate. These data are presented in Figure 5. In addition to the weekly averages, a 10% uncertainty bar is included for reference along with a fitted polynomial emission curve. As would be expected, the resulting emission curve mirrors the overall pattern seen in the 15-min time series (Figure 3), with low and steady emissions from flock wk 1 to 3 and a rapid increase in emissions thereafter. One of the interesting features in these weekly curves is the apparently low emissions in wk 6 when compared with wk 5 and 7. We attribute this to slightly cooler ambient air temperatures (about 5°C) during wk 6.

The emission curve in Figure 5 is integrated to provide total emissions during the flock growth cycle (chick placement to flock harvest), giving a farm total of 4,415 kg of NH_3·flock^{-1} or 0.016 kg of NH_3·bird^{-1}·flock^{-1}. This value does not include NH_3 emissions before chick placement or after flock harvest. Emissions occur pre- and postflock because the litter, some of which is recycled between flocks, emits NH_3 even though there are no birds in the houses. Actually, the highest emissions observed occurred after harvest (Figure 6) during housing clean-out and when the litter was placed outside awaiting removal for processing offsite. When between-flock emissions are summed, they amount to 16% of the placement-to-harvest flock emissions. Summing the flock cycle and between-flock emissions gives an overall total emission rate of 0.019 kg of NH_3·bird^{-1}·flock^{-1} (Table 1).

Comparison of Summer and Winter Farm Emissions

In 2009, the US Poultry and Egg Association funded a study (Harper et al., 2009b) on winter NH_3 emissions (February to March) at this same farm. The winter study used the same bLS methodology as this study. There were slight differences in the placement of equipment (upwind and downwind lasers), but in general, the 2 studies were carried out in an identical manner. Because farm management was similar during both winter and summer, there is an opportunity to examine seasonal differences in broiler emissions.

Figure 7 compares summer (this study) and winter emissions rates (after Harper et al., 2009b) on a weekly basis. The difference in cumulative flock emissions is not measurably different between the 2 seasons. This seasonal similarity was surprising. Although some broiler studies have shown little seasonal difference in NH_3 emissions (Wathes et al., 1997; Knížatová et al., 2009), several others have shown significant seasonal differences (Wheeler et al., 2006; Burns et al., 2007). In addition, other animal production systems generally show seasonal effects (Harper et al., 2004a,b, 2009a; Todd et al., 2008). What explains the lack of seasonal differences in some production systems?

Ammonia emissions from a source will be related to 4 physical factors: turbulence in the air surrounding the source (removal of NH_3 diffused into the water-air interface boundary layer), the vapor pressure of NH_3 in the source material (increase in vapor pressure with temperature), and the temperature and pH effects on the NH_4^+:NH_3 concentration ratio of the source material (and their effects on the dissociation of NH_4^+ to NH_3). For further discussion of chemical and physical effects on NH_3 emissions, see Harper (2005). How do these 4 factors differ between winter and summer

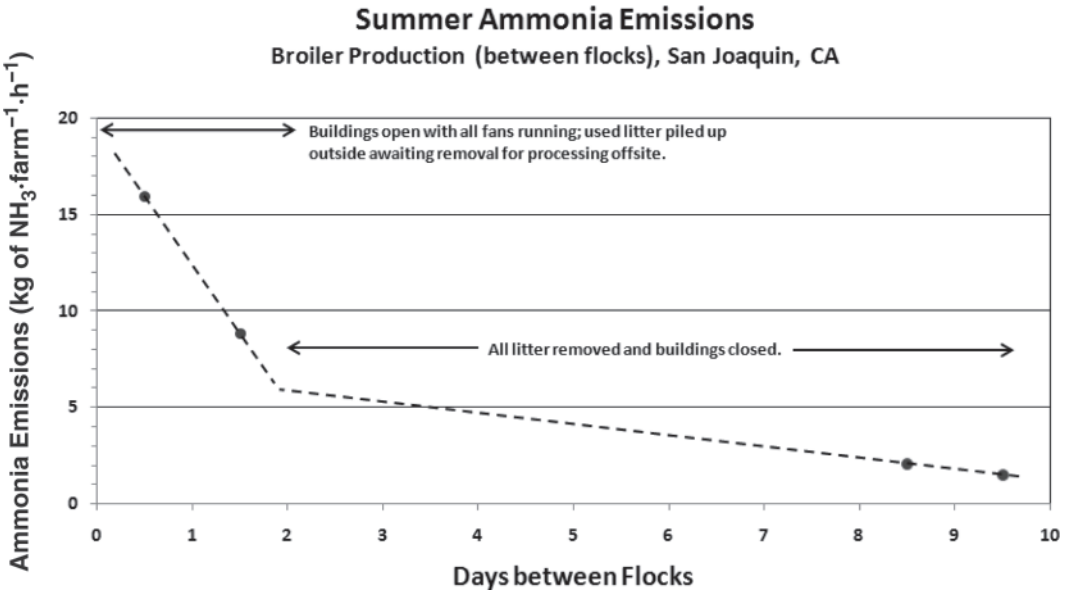

Figure 6. Emissions between flock harvest (d 0) and chick placement (d 10). From d 2 until chick placement, litter was removed from the farm site and buildings were closed. Some emissions were observed due to periodic fan cycling.

at our farm? Because management of the winter and summer poultry flocks was identical (identical length of flock cycle period, feed input, and house temperatures), the major seasonal differences would be the air turbulence in the houses (i.e., ventilation) and the humidity of air entering the houses. Figure 8A and B compares summer and winter house average ventilation rates and temperature, and Figure 9A, B, C, and D compares humidity for 4 times of the day throughout the flock cycle. The most striking difference was the much larger ventilation rates in summer (Figure 8A) needed to maintain the desired house temperatures (Figure 8B). With dramatically larger summer ventilation rates (i.e., increased turbulence in the houses), we might expect to see larger summer emissions.

Although ventilation (turbulence) can increase emission rates by increasing the removal of diffusive gases from the source material, it can also dry the emission source (litter). And in the San Joaquin Valley, summer ventilation brings in dry air, which accelerates drying the litter (Figure 9A, B, C, and D). (Note: winter is the wet season in the San Joaquin Valley.) We speculate that the cause of similar seasons' emission rates at this site is drier litter in summer than in winter. The drier litter, which reduces the dissociation of NH_4^+ to NH_3, then counteracts the effect of higher summer ventila-

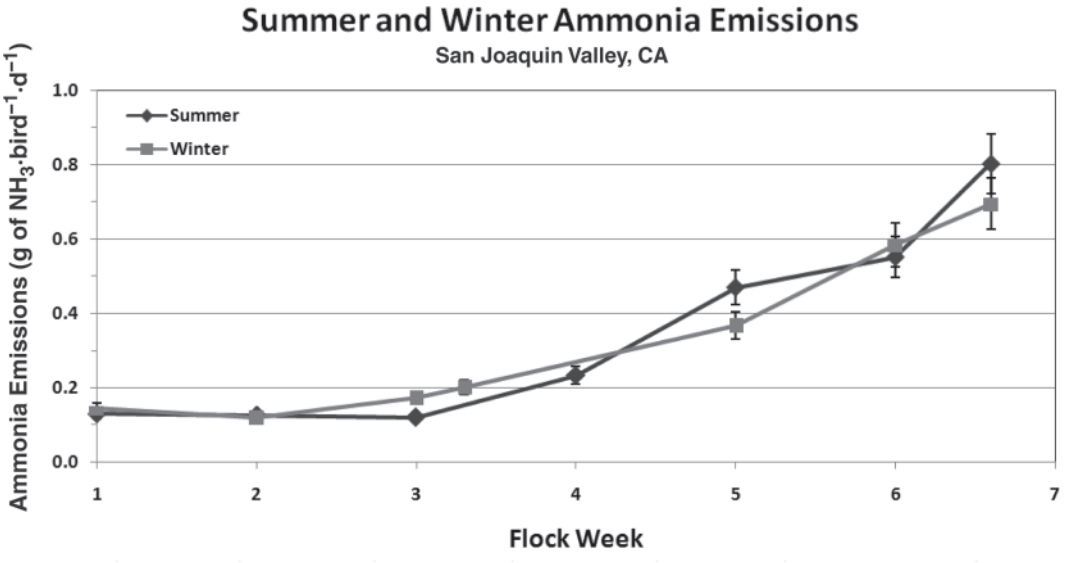

Figure 7. Comparison between winter (after Harper et al., 2009b) and summer broiler emission rates in the San Joaquin Valley, California.

Table 1. Comparison of winter and summer flock cycle emissions between humid eastern US (Kentucky) (after Burns et al., 2007; after Wheeler et al., 2006) and semi-arid US (California) broiler production

Broiler location	Winter flock cycle ($kg \cdot bird^{-1} \cdot flock^{-1}$)	Winter between-flock ($kg \cdot bird^{-1} \cdot flock^{-1}$)	Summer flock cycle ($kg \cdot bird^{-1} \cdot flock^{-1}$)	Summer between-flock ($kg \cdot bird^{-1} \cdot flock^{-1}$)
California	0.014 ± 0.001[1,2]	NA[3]	0.016 ± 0.002	0.003 ± 0.0003
Burns et al.	0.023	0.005	0.047	0.008
Wheeler et al.	NA	NA	0.054	NA

[1]After Harper et al. (2010).

[2]Measurement uncertainty is based on verified emissions studies (see Appendix, Harper et al., 2009a).

[3]NA = period data not measured.

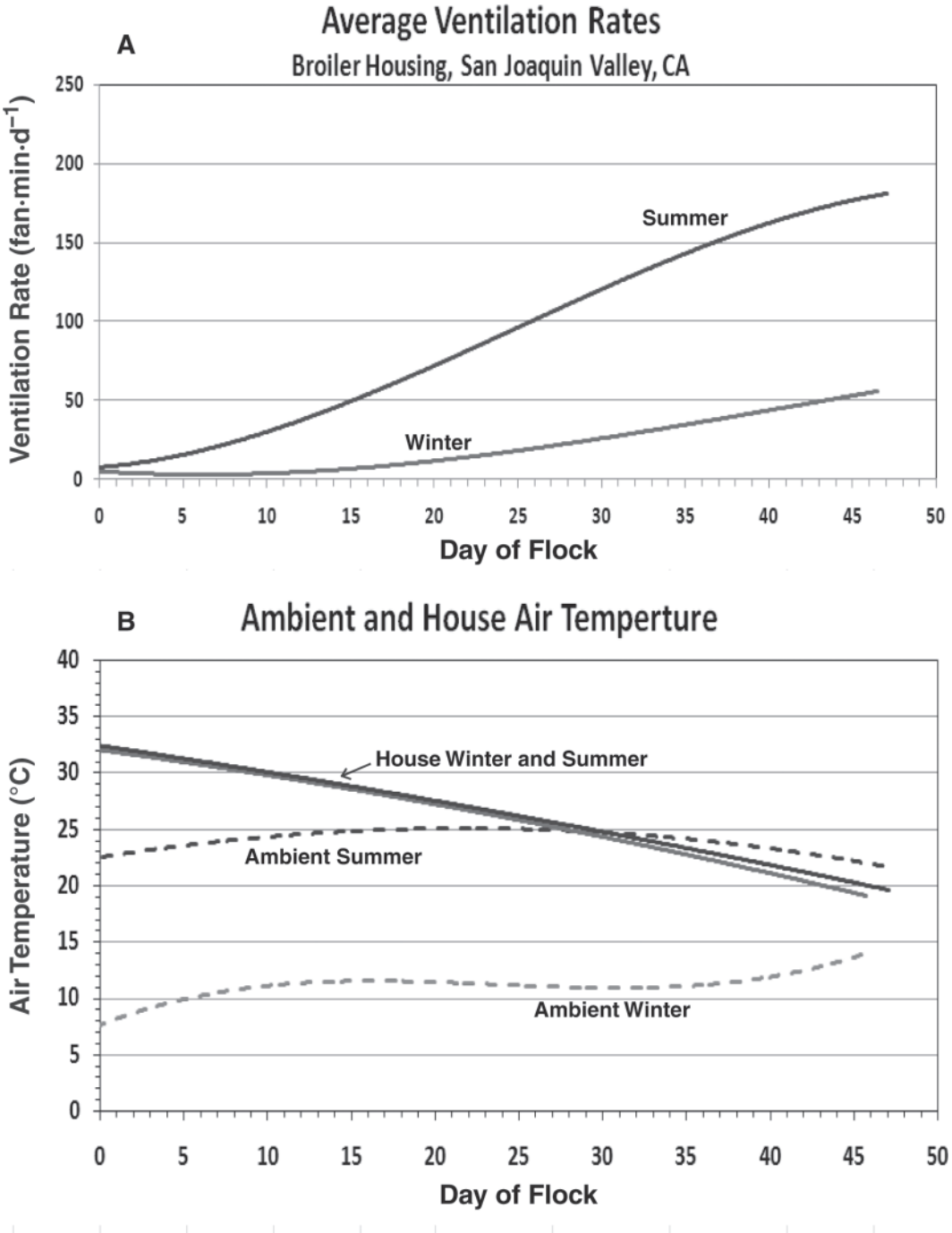

Figure 8. Comparison of summer and winter housing A) average ventilation rates and B) average temperature (from host records).

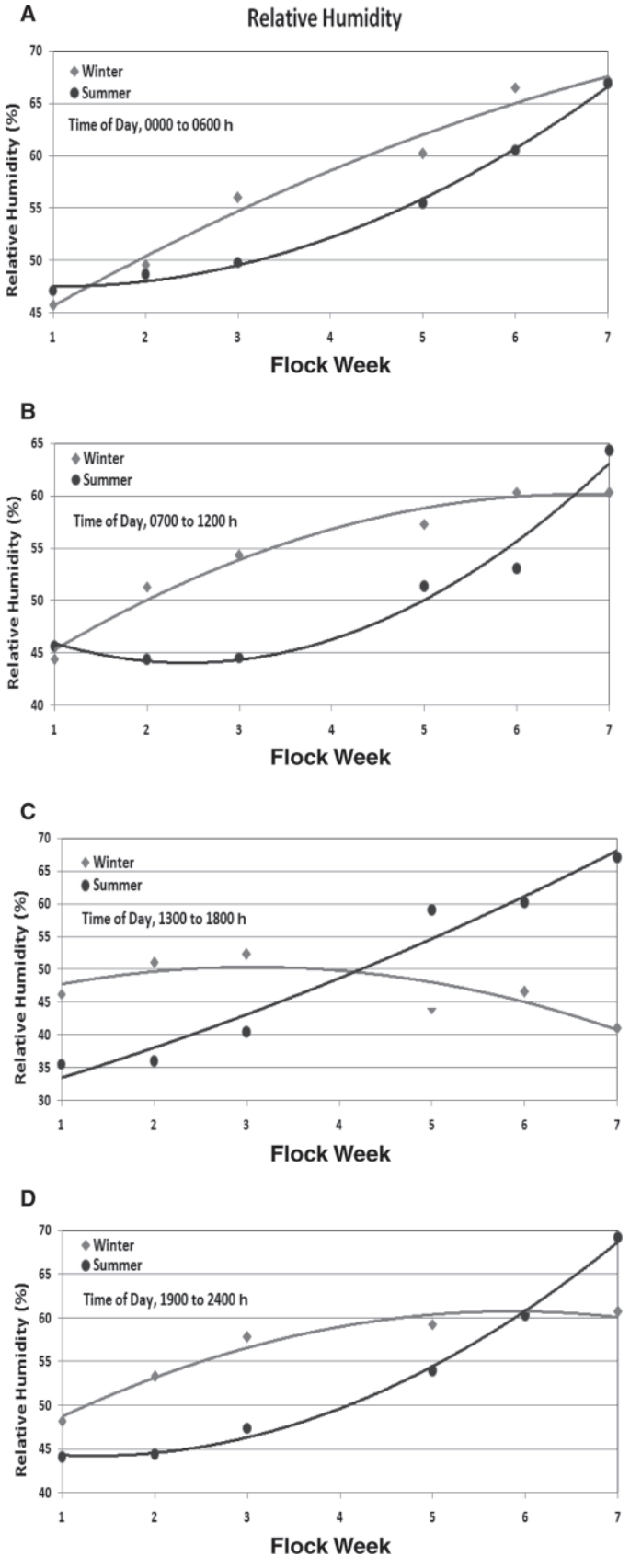

Figure 9. Comparison of summer and winter housing average weekly RH by time of day in broiler houses, San Joaquin Valley, California (from the host producer records).

tion rates. This effect was suggested by Knížatová et al. (2009) and may also have been the case in broiler studies of Wathes et al. (1997), who found that summer emissions did not increase over winter even though summer ventilation rates were much higher.

Comparison of California Emissions with Other Studies

The seasonal emission data from this California farm can be compared with other winter and summer studies from the eastern United States (Kentucky). These studies (Wheeler et al., 2006; Burns et al., 2007) used a concentration × flow integration (air mass-balance) technique. The emissions from these Kentucky farms are compared with emissions from this California farm in Table 1 (litter ages for all flocks were on the third flock).

The flock cycle emissions from the California winter flock, 0.014 ± 0.003 kg of NH_3·bird^{-1}·flock^{-1}, were 61% of the Burns et al. winter flock (0.023 kg of NH_3·bird^{-1}·flock^{-1}), whereas summer flock cycle emissions, 0.016 ± 0.002, were only 34% of their flock emissions (0.047 kg of NH_3·bird^{-1}·flock^{-1}). The Wheeler et al. winter study only looked at emissions over 3 wk during winter midflock (flock wk 3, 4, and 6 were reported) and emissions were not comparable without interpolation; however, for the 3 wk reported, the Wheeler et al. emissions were considerably larger than both the California (48% larger) and Burns et al. flocks (32% larger). The Burns et al. summer flock cycle emissions (Figure 10, Table 1) were considerably higher than their winter emissions (194%). There was no measurable difference between our California winter and summer emissions (0.014 ± 0.001 and 0.016 ± 0.002 kg of NH_3·bird^{-1}·flock^{-1}, respectively).

There were interesting comparisons in postflock emissions between the Burns et al. and these California studies. During the Burns et al. study, it was not possible to evaluate emissions during bird harvesting and removal of house litter because of the disruption of the controlled ventilation needed in the air mass-balance technique. However, emissions during their study were evaluated during other postharvest times when the houses were closed for measurement. In the Burns et al. winter study, emissions during the decaking process were very high during and for 2 d afterward (0.005 kg·bird^{-1}·flock^{-1}) and represented about one-third of the flock cycle downtime winter emissions. Summer downtime emissions for the Burns et al. flock (no decaking emissions reported) accounted for 15% of the total flock emissions (Table 1). Another study in the humid eastern United States (Pennsylvania) found that between flock downtime emissions were about 20% of total winter flock emissions (Topper et al., 2008). A study in Europe (Demmers et al., 1999) suggested that about 11% of NH_3 emissions resulted from downtime periods.

There was another (independent) study of NH_3 emissions from this same farm. Summers (2005) calculated NH_3 emissions at this California farm but using a completely different measurement technique (concentration × flow integration). Summers' measurements were much less comprehensive than these studies (only selected hours on 4 d), but because the farm management during Summers' measurements was essentially identical to that during this study, their results provide an interesting reference to our measurements and measurement technique. Summers' emission rates are illustrated (Figure 3) on the same plot as data from this study. We see that data from their limited time periods on 3 of their study days, NH_3 emissions were comparable to our measurements (considering the large diurnal variability of our observations). We take this general similarity in emission rates as an indication of the accuracy of bLS measurements at this farm. However, we also note that for a measurement day during the middle of the study, Summers' emissions were considerably smaller than our values. Summers then extrapolated from his limited data to a flock emission rate of 0.005 kg of $NH_3 \cdot bird^{-1} \cdot flock^{-1}$. Although this is considerably smaller than our 0.014 ± 0.003 kg of $NH_3 \cdot bird^{-1} \cdot flock^{-1}$, we feel that the very limited data of the Summers' study, and its time-of-day measurement bias, compromise his estimate of long-term, averaged flock emissions.

Another eastern US study (Siefert and Scudlark, 2008) from tunnel-ventilated broiler houses measured NH_3 emissions on 3 partial days (all near 1200 h). This very limited study suggested that this type of house produced 2.7 g of NH_3-N$\cdot bird^{-1}$ emissions over the last 3 wk of a 6-wk production period (1.2, 0.8, and 0.7 g of $NH_3 \cdot bird^{-1} \cdot wk^{-1}$, respectively). Our California farm emissions during this same growth period were almost 5 times higher (12.8 g of NH_3-N$\cdot bird^{-1}$). They also found, different from these California studies (and those of Burns et al. and Wheeler et al., described above), that NH_3 emissions decreased during their last 3 wk of studies by over 40%. The Siefert and Scudlark study should be interpreted with caution because the results are based on a small number of passive denuder sensor measurements (sensors prone to inaccuracy, Harper, 2005) plus there are also concerns about their choice and use of an unrealistic Gaussian plume model (Harper, 2005) to convert downwind concentration measurements to emissions.

Although European broiler emission rates may not be ideally comparable to US emission rates due to different management, housing type, and climate, annual emissions reported by Demmers et al. (1999) of 0.011 kg of $NH_3 \cdot bird^{-1} \cdot flock^{-1}$ and Wathes et al. (1997) of 0.025 kg of $NH_3 \cdot bird^{-1} \cdot flock^{-1}$ bracketed our 0.018 kg of $NH_3 \cdot bird^{-1} \cdot flock^{-1}$ (all of the European studies used mass-balance techniques). Modeled emissions by Groot Koerkamp et al. (1998) for 4 countries in Northern Europe (United Kingdom, the Netherlands, Denmark, and Germany) also bracketed our emissions ranging from 0.009 to 0.020 kg of $NH_3 \cdot bird^{-1} \cdot flock^{-1}$.

Annual Emissions Calculation

Total annual emissions for broiler production at this California farm were estimated as the average of winter and summer emission rates (Table 2). Previous studies in other livestock sectors have shown that transitional seasonal periods (autumn and spring) are about the average of winter and summer emissions (Harper et al., 2004b, 2009a). Based on an industry average of 5.5

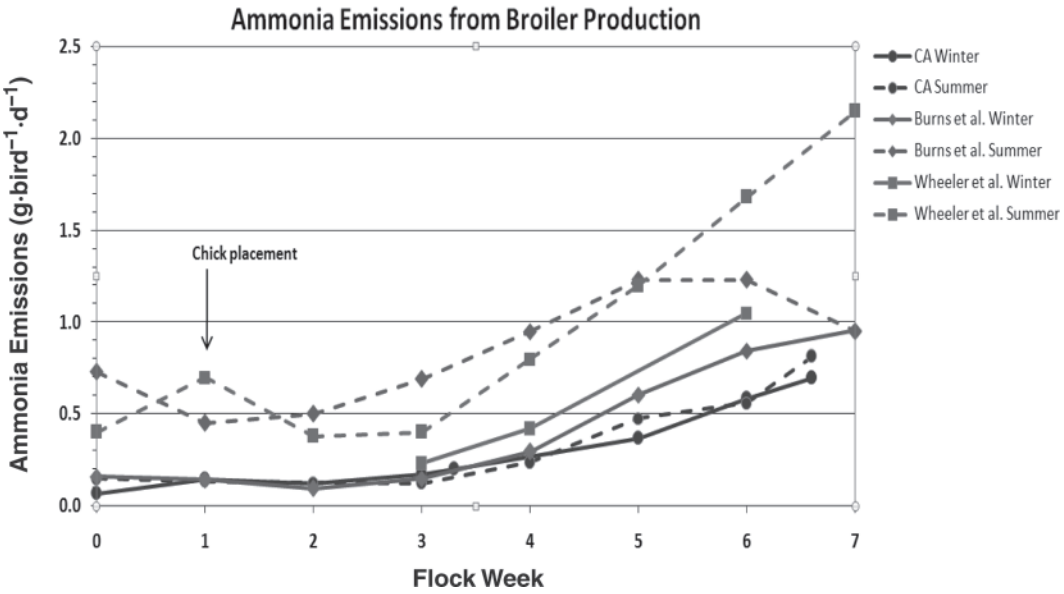

Figure 10. Comparison of emission rates from this farm winter (after Harper et al., 2009b) and summer flocks and from comparable seasonal flocks in the eastern United States during winter and summer production periods. The eastern US flock emissions were determined using a concentration × flow integration technique (Wheeler et al., 2006; Burns et al., 2007) and emissions for this present study were determined using inverse dispersion (backward Lagrangian stochastic) analysis techniques.

Table 2. Comparison of total emissions between humid eastern Kentucky (after Burns et al., 2007) and semi-arid California broiler production (after Harper et al., 2010)

Broiler location	Winter flock ($kg \cdot bird^{-1} \cdot flock^{-1}$)	Summer flock ($kg \cdot bird^{-1} \cdot flock^{-1}$)	Average annual ($kg \cdot bird^{-1} \cdot flock^{-1}$)	Total annual ($kg \cdot bird^{-1} \cdot yr^{-1}$)[1]
California	0.017 ± 0.002[2,3]	0.019 ± 0.002	0.018 ± 0.002	0.099 ± 0.010
Burns et al.	0.021	0.055	0.041	0.226

[1]Assuming 5.5 flocks per year.

[2]After Harper et al. (2010).

[3]Assuming that winter between-flock emissions were the same as summer.

flocks produced per year, annual NH_3 emissions for this farm (according to the bLS measurements as described above) are 0.08 kg of $NH_3 \cdot bird\text{-}place^{-1} \cdot yr^{-1}$. Calculated annual emission for an eastern US farm (Burns et al., 2007) was 0.23 kg of $NH_3 \cdot bird\text{-}place^{-1} \cdot yr^{-1}$.

Recent estimates from the USEPA (Table 3-7, USEPA, 2004) suggest that 23% of annual input feed N is emitted as NH_3-N from broiler production housing. From our present measured emission rates (flock cycle plus between-flock) and total input feed N, we calculate that 7.8 and 8.3% of the input feed N is lost as NH_3-N in winter and summer, respectively, or an annual average of 8.1%.

We can compare the N efficiency of this broiler farm with that measured in other animal industries. Ammonia emissions from this farm are considerably less (on a per-kilogram-of-feed-N-provided basis) than emissions from a beef feeding operation, where Todd et al. (2007, 2008) reported annual averages of 39 and 53% of feed N lost through NH_3 emissions. These broiler emissions are somewhat less than the 14.8% emissions rate reported from swine production (Harper et al., 2004a) but not different from the average annual emissions of $7.6 \pm 1.5\%$ reported from dairy production in Wisconsin (Harper et al., 2009a).

Summary

The bLS noninterference measurement technique was used to determine NH_3 emissions from a poultry operation in central California during the summer season. The complete flock measurements (flock cycle plus between-flock periods) of total farm emissions were 0.019 ± 0.002 kg of $NH_3 \cdot bird^{-1} \cdot flock^{-1}$. Emissions (on a per-bird and % feed N basis) were not significantly different between winter and summer. This lack of a seasonal difference contrasts with emissions as reported by other studies in the humid eastern United States, where summer emissions were much larger than winter values. When compared with other studies in the eastern United States (both summer and winter), the NH_3 emissions from this California poultry farm are significantly smaller. This is mostly due to the comparably lower summer emissions at the California farm (e.g., summer emissions for this flock were 0.016 compared with 0.047 and 0.054 kg of $NH_3 \cdot bird^{-1} \cdot flock^{-1}$for 2 eastern US flocks). Emissions from broiler operations in Northern Europe bracketed the California emission rates ranging from 0.009 to 0.025 kg of $NH_3 \cdot bird^{-1} \cdot flock^{-1}$. Based on 5.5 flocks per year production cycles, total annual emissions for this farm are 0.099 ± 0.010 kg·bird-place$^{-1} \cdot yr^{-1}$ or $8.1 \pm 0.8\%$ of the feed N lost as NH_3-N. Annual emissions from this California farm were considerably smaller than USEPA estimates for broiler production in the United States: $8.1 \pm 0.8\%$ of feed N lost as NH_3-N emissions compared with USEPA estimates of 23%.

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