AMMONIA EMISSIONS FROM DAIRY LAGOONS IN THE WESTERN U.S.

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ABSTRACT. *Ammonia (NH_3) emissions from dairy liquid storage systems can be a source of reactive nitrogen (N) released to the environment, with a potential to adversely affect sensitive ecosystems and human health. However, little on-farm research has been conducted to estimate these emissions and determine the factors that may affect these emissions. Six lagoons in south-central Idaho were monitored for one year using open-path Fourier transform spectrometry, with NH_3 emissions estimated using inverse dispersion modeling (WindTrax software). Lagoon physicochemical characteristics thought to contribute to NH_3 emissions were also monitored over this period. Average total emissions from the lagoons ranged from 12 to 43 kg NH_3 ha^{-1} d^{-1}, or 5.4 to 85 kg NH_3 d^{-1}. Emissions from the settling basin on one dairy were 30% of the total emissions from the liquid storage system, indicating that basins are important sources of on-farm NH_3 emissions. Emissions generally trended greater during the summer, when temperatures were greater. High wind events and agitation of the lagoons created temporary increases in NH_3 emissions irrespective of temperature. Lagoon physicochemical characteristics, such as total Kjeldahl nitrogen (TKN) and total ammoniacal nitrogen (TAN), were highly correlated with emissions ($r = 0.52$ and 0.55, respectively). Regression models were developed to predict on-farm NH_3 emissions and indicated that TKN, TAN, wind speed, air temperature, and pH were the main drivers of these emissions. An on-farm N balance suggested that lagoon NH_3-N losses represented 9% of total N lost from the facility, 65% of total lagoon N, and 5% of dairy herd N intake. A process-based model (Integrated Farm System Model) estimated values for N excretion and NH_3-N loss from the lagoon within 5% of that measured on-farm. More on-farm research is needed to better refine both process-based models and emission factor estimates to more accurately predict NH_3 emissions from lagoons on dairies in the western U.S.*

Keywords. *Ammonia, Emission, Inverse dispersion, Manure.*

Dairy cattle require nitrogen (N) in the form of essential amino acids, which are the building blocks of protein. However, N use efficiency in dairy cattle is fairly low, and a lactating cow may excrete more than 64% of the N ingested in feed, with the majority (38%) excreted as urinary N (Niu et al., 2016). Most urinary N is in the form of urea, which when excreted into the environment can be rapidly hydrolyzed to ammonium by the urease enzyme found in feces and soil, subsequently converted to ammonia (NH_3), and lost via volatilization. Typical NH_3 emission rates from animal housing range from 5% to 60% of N excreted, with an additional amount of NH_3 lost during manure handling and storage as well as land application (Rotz and Leytem, 2015).

A few on-farm studies have evaluated NH_3 losses from dairy liquid storage, with only six studies examining emissions over the course of a year (table 1). Literature values of on-farm NH_3 emissions from dairy lagoons range from 2.5 to 68 kg ha^{-1} d^{-1} (1.4 to 129 g d^{-1} head^{-1}), illustrating the large range in emissions that can occur due to management practices and climatic variables. The generation and emission of NH_3 from manure handling and storage is influenced by multiple factors, such as exposed manure surface area, airflow across the manure surface, manure mixing, manure storage loading rate, ambient temperature, and manure age, temperature, moisture content, and pH (Rotz, 2004; Rotz et al., 2014).

The U.S. Environmental Protection Agency (EPA) estimates that 55% of atmospheric NH_3 in the U.S. originates from livestock manure (USEPA, 2014). Ammonia can cause animal health hazards at high concentrations (NRC, 2003) and human respiratory problems as a precursor to fine particulate formation (Kampa and Castanas, 2008). Excess environmental NH_3 can also degrade terrestrial and aquatic ecosystems through acid deposition and eutrophication (Kirchmann et al., 1998). At present, there are large uncertainties in the national NH_3 emissions inventories for livestock waste. To address these uncertainties, the U.S. EPA

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Table 1. Summary of average on-farm ammonia (NH_3) emissions from dairy wastewater storage reported in the literature.

Reference	Country	Method	Season	Type of Storage	NH_3 (kg ha^{-1} d^{-1})	NH_3 (g d^{-1} head^{-1})
Dore et al., 2004	U.K.	Tracer	Summer	Slurry tank	5.9	1.5
McGinn et al., 2008	Canada	Inverse dispersion	Summer	Lagoon	51	109
Mukhtar et al., 2008	U.S.	Flux chamber	Summer and winter	Primary lagoon	4.5	1.4
				Secondary lagoon	2.5	5.7
Rumburg et al., 2008	U.S.	Tracer	Annual	Lagoon	3.3	151
Bjorneberg et al., 2009	U.S.	Inverse dispersion	Annual	Lagoon	9.1	10.4
Flesch et al., 2009	U.S.	Inverse dispersion	Summer/fall	Lagoon	4.7	41
			Summer/fall	Lagoon	7.2	36
Leytem et al., 2011	U.S.	Inverse dispersion	Annual	Lagoon	20	20.3
Leytem et al., 2013	U.S.	Inverse dispersion	Annual	Lagoon	68	129
Minato et al., 2013	Japan	Floating chamber	Annual	Slurry tank	5.8	1.6
Neerackal et al., 2015	U.S.	Floating chamber	Summer/fall	Lagoon	4.3	
			Summer/fall	Lagoon	6.0	
Grant and Boehm, 2015	U.S.	Vertical radial plume	Annual	Settling basin	4.5	13.9
Todd et al., 2015	U.S.	Inverse dispersion	Summer	Lagoon	32.7	17

established the National Air Emissions Monitoring Study in 2005 with the purpose of gathering information that would be used to develop or improve emissions estimating methodologies (EEMs) for livestock, including lagoons and basins at dairy operations. However, the study only included three dairy lagoons, and all dairies handled most of their manure as a liquid, providing little variation in manure handling systems. The lagoon EEMs developed to predict NH_3 emissions were based on animal type, surface area, farm size, ambient temperature, ambient humidity, wind speed, and solar radiation (USEPA, 2012). The EPA Scientific Advisory Board review of these EEMs suggested that the use of these empirical relationships to predict NH_3 emissions from dairy lagoons was inadequate. They did not reflect the processes known to drive NH_3 emissions, and they could not represent conditions beyond those of the original data set. Thus, more work was needed to develop EEMs that would be applicable across the industry (USEPA, 2013).

Many dairy operations in the western U.S. house cattle in dry-lot systems where the majority of manure may be handled as a solid, with manure from the milking parlor and holding areas going into liquid manure handling systems. The emissions from these lagoons are likely to be quite different from lagoons where most of the manure produced is handled as a liquid. Therefore, to better characterize NH_3 emissions from lagoon systems typical of western dairy production, our goal was to study seasonal trends in on-farm NH_3 emission and determine the lagoon physicochemical characteristics and meteorological conditions that affect emissions over the course of a year. In addition, we completed a whole-farm N balance on one dairy to determine the proportion of NH_3-N originating from the lagoon system compared to the whole-farm N losses, and we compared measured values with estimates from a process-based model.

Materials and Methods

Farm Descriptions

During September 2010 to November 2015, six dairy lagoons located on private farms in southern Idaho were selected for monitoring of NH_3 emissions (table 2). These farms were selected to represent the manure handling techniques typically found on western U.S. dairy farms and based on the ability to separate the lagoon emissions from the rest of the farm. They were also situated in areas where there were no NH_3 sources directly upwind that could contribute to measured NH_3 concentrations. This enabled us to select lagoons that would not have interference from internal or external NH_3 sources. In addition, farms were selected to represent a variety of sizes ranging from <500 to >5,000 cows. Within the state of Idaho, 68% of total dairy farms have <200 cows, 19% have 200 to 1,000 cows, 7% have 1,000 to 2,500 cows, and 6% have >2,500 cows (USDA-NASS, 2012).

Five of the dairies were dry-lot dairies where cows were housed in pens and the majority of manure was stored as a solid. In these systems, manure from the milking parlor and holding areas flows into a lagoon system, which typically consists of one or more settling basins to separate out some of the solids, followed by a larger lagoon. These lagoons are typically emptied in the spring and fall, with the manure spread on surrounding cropland, while the sludge remaining in the ponds is generally not removed on a regular basis. The settling basins are cleaned out on an infrequent basis; in

Table 2. Characteristics of the dairies used to determine on farm lagoon ammonia emissions in south-central Idaho.

	Dairy					
Parameter	D1	D2	D3	D4	D5	D6
Housing type	Dry lot	Dry lot	Dry lot	Freestall	Dry lot	Dry lot
Size of operation (head)	1,000 to 5,000	5,000 to 10,000	1,000 to 5,000	5,000 to 10,000	1,000 to 5,000	<1,000
Lagoon water source	Parlor washwater	Parlor washwater	Parlor washwater	Flush system from barn and parlor washwater	Parlor washwater	Parlor washwater
Surface area (m^2)	26,628	47,398	19,621 to 23,237	4,005 to 13,220	1,300 to 3,373	2,101
Depth (m)	2.4 to 2.7	1.5	1.2 to 1.9	0.9 to 1.6	0.3 to 1.3	0.3 to 0.9
Monitoring period (mm/dd)	9/10 to 6/11	12/10 to 6/11	6/12 to 5/13	5/12 to 5/13	7/13 to 11/14	11/14 to 11/15
Days monitored	20	18	70	41	159	277
No. of 15 min data points	346	575	1,060	1,342	4,382	7,219

many cases, they are not cleaned out more than once a year at the most. For comparison, one freestall dairy was included, with the lactating cows housed in naturally ventilated barns. The manure from the barns was removed by flushing the alleyways behind the freestalls three times a day with recycled water, which is a common practice for freestall dairies in this region. The washwater from the milking parlor and holding area on this dairy also flowed into the lagoon system. The dairy manure handling systems varied by farm and are described below:

D1: A dry-lot dairy with manure storage comprised of three settling basins and a main lagoon. The main lagoon was monitored in this study.

D2: A dry-lot dairy with manure storage comprised of four settling basins and a main lagoon. The main lagoon was monitored.

D3: A dry-lot dairy that was recently converted to a heifer operation. However, during the last quarter of the study, there were lactating animals on the farm. The lagoon system consisted of five settling basins and a main lagoon. The main lagoon was monitored.

D4: A freestall dairy using a flush system with the manure storage system consisting of a screen separator, three settling basins, three main lagoons, and a satellite lagoon. The satellite lagoon was monitored.

D5: A dry-lot dairy comprised of a concrete settling basin and three lagoons. The final lagoon in the series was monitored.

D6: A dry-lot dairy comprised of one settling basin and a main lagoon. The main lagoon and settling basin were monitored.

Equipment Location at the Dairies

The prevailing wind at all locations was from the west, with occasional wind from the east. The concentration of NH_3 was measured using open-path Fourier transform infrared spectrometry (OP/FTIR; Griffiths et al., 2009; Shao et al., 2010). The pathlength of the OP/FTIRs on all dairies was aligned north to south to be perpendicular to the prevailing wind direction. At all locations, one sonic anemometer (model 81000 ultrasonic anemometer, R.M. Young Co., Traverse City, Mich.), at a height of 3 m, was located in an area where there were minimal flow disturbances from any upwind structures to capture a more idealized wind flow of the area, as suggested by Flesch et al. (2005a). In all instances, there was cropland directly upwind of the lagoons. At both D1 and D2, the lagoon was situated directly north of the open lots, and one OP/FTIR was placed on the east and west berms of the lagoon, while the sonic anemometer and weather station were placed at the northeast corner of the lagoon. There was cropland to the west and north of the lagoon, while the settling basins were located to the east of the lagoon. At D3, the lagoon was situated directly north of the open lots, and one OP/FTIR was placed across the eastern edge of the lagoon (~25 m from the eastern edge), while the sonic anemometer and weather station were located at the southwestern corner of the lagoon. There was a highway directly to the west, with cropland to the west of that, cropland to the north, and the settling basins were located to the east of the lagoon. At D4, the lagoon was located 750 m to the west of the freestall barns and was surrounded by cropland on all sides. The OP/FTIR was located across the eastern edge of the lagoon (~42 m from the eastern edge), while the sonic anemometer and weather station were located on the southeast corner of the lagoon. At D5, the lagoon was located to the north of the open lots. One OP/FTIR was placed along the eastern berm of the lagoon, while the sonic anemometer and weather station were located at the northeast corner of the lagoon. The lagoon was surrounded by cropland on three sides. At D6, the lagoon was located to the south of the open lots, with the settling basin located to the northeast of the lagoon. One OP/FTIR was located on the eastern berm of the lagoon, while the sonic anemometer was located at the southeast corner of the lagoon and the weather station was at the northeast corner. The lagoon was surrounded by cropland on the three remaining sides.

Ammonia Concentration and Wind Measurements

Initially, lagoons were monitored seasonally (D1 and D2), but as more resources became available monitoring times were increased to better capture annual variations in emissions (D3 to D6). The number of days monitored at each dairy ranged from 18 to 277 (table 2). One OP/FTIR (Air Sentry, Cerex Monitoring Solutions, Atlanta, Ga., or ABB-Bomem MB-100, MDA, Atlanta, Ga.) was located either across the downwind edge/corner (D3 and D4) or on the downwind bank (D1, D2, D5, and D6) of the lagoon, with the sensor at 1.7 m height and pathlengths ranging from 130 to 240 m. At D6, the position of the OP/FTIR enabled monitoring of either the settling basins or the lagoons depending on wind direction. Spectra were acquired continuously and averaged over 5 min intervals. Background concentrations were measured at or near (within 2 km) each dairy for several days at the onset of the study as well as at a remote (non-agriculturally impacted) location for comparison and were less than 0.008 ppm-v. Experiments performed with the OP/FTIR units demonstrated that background NH_3 concentrations were very stable and did not fluctuate daily (CV = 19% over a 4 d period with 1,004 measurements). In addition, the on-farm concentration data at most locations were filtered for wind direction to isolate times when there was no upwind source of NH_3 present to verify that background concentrations were consistent over time. Quantitative determinations of NH_3 concentrations were performed by partial least squares regression of the OP/FTIR spectra (Griffiths et al., 2009; Shao et al., 2011, 2013), and the detection limit of NH_3 was <0.005 ppm-v. Concentration data were processed to produce 15 min average mixing-ratio concentrations at the source areas (C).

The wind environment at the dairy was described by simple Monin-Obukhov similarity theory (MOST) relationships defined by u^*, L, z_0, and β, as provided by three-dimensional sonic anemometers, where u^* is the friction velocity, L is the Obukhov stability length, z_0 is the surface roughness length, and β is the wind direction. See Flesch et al. (2004) for details on how these parameters were calculated from a sonic anemometer. Wind parameters were calculated for each 15 min period (corresponding to C observations). A meteorological station was also located at each lagoon to record barometric pressure, air temperature, wind direction, and wind speed (all at 2 m) during the experimental period.

Emissions Calculations

WindTrax 2.0 software (Thunder Beach Scientific, Nanaimo, British Columbia, Canada) was used to determine lagoon emission rates, combining the backward Lagrangian stochastic (bLS) inverse-dispersion technique described by Flesch et al. (2004) with an interface allowing sources and sensors to be conveniently mapped. This technique has been used in several controlled-release studies to determine emissions from barn and lagoon source areas and was shown to provide estimates of emissions within 15% of actual emissions (McGinn et al., 2009; Gao et al., 2010; Ro et al., 2013). For a detailed description of the bLS technique, see Flesch et al. (2005a, 2005b, 2007). The lagoons and settling basins were mapped using available satellite imagery and on-farm GPS data. Emission estimates (kg NH_3 ha^{-1} d^{-1} and kg NH_3 d^{-1}) were calculated at 15 min intervals using $N = 50,000$ trajectories and fixed background concentrations. Emissions from the settling basin at D6 were determined using the method stated above; however, the lagoon and cattle housing were also included as source areas in the model and set at an average emission rate to account for any potential emission contributions from those sources (Flesch et al., 2009). The lagoon emission rates were determined from the data generated during the same periods, and the emission rates from the housing were calculated using the approach of Bonifacio et al. (2015) and set at 49 kg NH_3 d^{-1}.

Good emission estimates depend on using data that do not violate the MOST assumptions (i.e., low winds, extreme stabilities, wind profile errors). Data were filtered by removing periods when $u^* \leq 0.10$ m s^{-1} (low wind conditions), $|L| \leq 5$ m (strongly stable or unstable atmosphere), and $z_0 \geq 1$ m (associated with errors in wind profile; Ro et al., 2013; Flesch et al., 2014).

Due to the location of the concentration sensors and other source areas on a farm, for some wind directions, measurements of downwind concentrations may not sample enough of the farm plume, which can lead to uncertainty in emission estimates (Flesch et al., 2005b). Additionally, there could be cross-contamination due to emissions from other source areas on the farm. Therefore, we filtered out data when the wind was either not within ±40° perpendicular to the OP/FTIR pathlength or from areas where there could be other NH_3 sources (such as cattle pens or manure piles) to ensure that the concentration sensors were measuring gases only from the source areas of interest.

Lagoon Sampling and Analyses

An intensive lagoon sampling campaign was conducted simultaneously with the emissions monitoring to determine spatial and temporal changes in lagoon properties that could be associated with NH_3 emissions. Measurements included total Kjeldahl N (TKN), total ammoniacal N (TAN), pH, and temperature. Total Kjeldahl nitrogen measures the amount of organic nitrogen (N) plus ammonium (NH_4^+) present in the lagoon water to give an indication of not only the NH_4^+ present that could be converted to NH_3 and lost via volatilization but also the amount of N that could become available as NH_4^+ following mineralization within the lagoon. Total ammoniacal N is a measure of NH_4^+ plus NH_3 present in the lagoon water, which is N that is readily available for loss as NH_3 depending on the pH of the water. As the pH increases, there is greater conversion of NH_4^+ to NH_3, which can then be volatilized from the lagoon water. A detailed description of the sampling protocols can be found in Leytem et al. (2017), although a brief description follows.

Lagoons were sampled (500 mL) every 2 to 3 weeks on a grid, with the number of sampling points (4 to 10) related to the size of the lagoon and distributed as evenly as possible across the lagoon surface. Lagoon depth was determined with a sampling rod that was marked for depth. The rod was allowed to sit on the top of the sludge layer at the bottom of the lagoon to determine the depth of the water column. This rod was connected to a container with a retractable lid to collect samples at specific depths. When lagoons were deeper than 1 m (D1 to D4), samples were collected from the surface (0.15 m below surface) and 0.3 m above the top of the sludge layer at each sampling location; otherwise, only surface samples were collected.

Immediately after collection, a 125 mL subsample was taken and mixed with 1 mL of concentrated sulfuric acid to stabilize the sample for TKN analysis. All samples were transferred to the laboratory in coolers, stored under refrigeration at 5°C, and processed within 24 h for TAN and within 36 h for TKN. In addition to collecting samples for analysis, the temperature and pH were determined *in situ* with a YSI 556 Multiprobe System (YSI Inc., Yellow Springs, Ohio) at each sampling location and depth; these measurements were typically made in late morning or early afternoon. All samples were allowed to come to room temperature and were thoroughly mixed prior to subsampling and analysis. Analyses were performed as follows: TAN according to Standard Method 4500-NH_3 (Eaton et al., 2005) and TKN according to EPA method 351.2 (USEPA, 1993). Because there were no significant spatial differences in lagoon characteristics with sampling position or depth (Leytem et al., 2017), the data were averaged to produce one daily value. The CV of TAN at each sampling time ranged from 0% to 40%, with 93% of values being less than 10% and only one value greater than 18%. The CV of TKN ranged from 1% to 46%, with 93% of values being less than 10% and only one value greater than 18%. The higher CV values were associated with periods when the lagoons were being emptied or agitated in some other way (filling, irrigation, etc.).

Nitrogen Balance at Dairy D6

An on-farm N balance was determined at D6 over the course of the year with quarterly sampling (November, April, July, and October). On each sampling date, data were collected from the producer on herd size, average daily milk production, milk protein content, target DMI, and dietary CP. Feed weights were recorded for each pen (2× daily feeding of lactating cows), and rejected feed was estimated for the lactating cows and sampled. Samples of the TMR, alfalfa (young stock), and grain (calves) were collected at each feeding. The TMR was sampled by compositing ten samples from each feed bunker when feed was placed. Straw samples were also collected to determine the amount of N added with bedding. Samples were brought back to the laboratory, thoroughly mixed, subsampled, and then freeze-dried, with wet and dry weights recorded to calculate moisture content. Samples were then ground and analyzed for total N via combustion of a 25 mg sample in a FlashEA 1112 N/protein analyzer (CE Elantech, Inc., Lakewood, N.J.).

During solid manure cleanout, the number of truckloads was recorded as well as the weight of four different truckloads to estimate manure weight per truck. Manure samples were collected from each pile (approx. 15 samples per pile during loading), composited, transported to the laboratory in a cooler, weighed, and frozen. All manure samples were freeze-dried, ground, and analyzed for total N (see method above for feed). Lagoon samples were collected just prior to fall emptying of the lagoon (see methods above for collection and analysis), and the lagoon liquid volume removed was estimated using the pump flow multiplied by the hours the pump ran, as well as a volume estimate based on the surface area of the lagoon and the change in lagoon depth before and after pumping. These two estimates were averaged to determine the final volume.

Total N consumed annually as feed was calculated by multiplying the total N in each ration with the amount of each ration fed (rejected feed N was subtracted from the total N provided to the lactating cows) quarterly and then averaged. Total N excreted by the lactating cows was estimated using the following equation:

$$N_E \text{ (g d}^{-1}) = 30 + 0.67 \times N_I \text{ (g d}^{-1}) \quad (1)$$

where N_E is N excreted, and N_I is N intake (Kebreab et al., 2010). The total N excreted by the dry cows, heifers, and calves was estimated using literature values (ASABE, 2005). The total N leaving the facility as solid manure was calculated by multiplying the dry weight of manure removed with the total N content of the manure. The total N leaving the facility as liquid manure was calculated by multiplying the volume of water pumped from the lagoon by the lagoon water TKN concentration. The N lost from housing and manure storage was calculated as:

$$\begin{aligned} N_{loss} &= \left[N_E + \text{straw N (kg N year}^{-1}) \right] \\ &- \text{N exported with manure (kg N year}^{-1}) \end{aligned} \quad (2)$$

To calculate the amount of NH_3-N lost from the lagoon as a percentage of total N to the lagoon, we assumed that 10% of the manure N excreted from the lactating cattle went into the liquid manure handling system based on estimates by Saggar et al. (2004) and the time the cows spent in the holding area and milking parlor each day.

For further evaluation of the whole-farm N balance, this farm production system was modeled with the Integrated Farm System Model (IFSM). IFSM is a simulation model that integrates the major biological and physical processes of a dairy farm (Rotz et al., 2016). Nutrient flows through the farm are modeled to predict potential nutrient accumulation in the soil and losses to the environment. The quantity and nutrient content of the manure produced is a function of the quantity and nutrient content of the feeds consumed. Nitrogen volatilization occurs in the barn, during manure storage, following field application of manure, and during grazing (Rotz et al., 2014). Following the prediction of losses, whole-farm mass balances of N are determined as the sum of all N imports in feed, fertilizer, deposition, and legume fixation minus the exports in milk, excess feed, animals, manure, and losses leaving the farm. The NH_3 emission model in IFSM has performed well in predicting NH_3 emissions from freestall dairy barns, manure storages, field-applied manure, and excretions during grazing (Rotz et al., 2014).

Table 3. Input data for the IFSM analysis of dairy D6.

Input Parameter	Value
Animal type	Large Holstein
Target milk production (L cow^{-1} year^{-1})	11,711
Number of lactating cows	213
Number of young stock (>1 year)	93
Number of young stock (<1 year)	44
Protein feeding level in ration	15.5%
Primary manure collection	Flush
Percentage of manure into primary collection	10%
Secondary manure collection	Scraper
Percentage of manure into secondary collection	90%
Bedding type	Straw
Amount of bedding per mature animal	13 kg d^{-1}

We used IFSM v4.3 (Rotz et al., 2014) to estimate the annual N budget for D6, including the amount of NH_3-N lost from the housing and manure storage areas. The model input data used to simulate D6 are listed in table 3. The farm was simulated for 25 years using historical weather data (1989-2012) for Jerome, Idaho, and the long-term average simulated N flows were compared to those measured on farm. The Jerome weather station is located about 20 km from the dairy in the same farmed landscape as the dairy.

Statistical Analysis

Linear regression was performed with SAS (v9.3; SAS, 2008) to relate daily NH_3 emission estimates to both meteorological parameters and lagoon physicochemical characteristics. As the lagoon characteristics changed predictably and slowly over time (Leytem et al., 2017), daily lagoon physicochemical characteristics were calculated by linear interpolation between sampled days.

Mixed-effects models were developed to predict NH_3 emissions (kg ha^{-1} d^{-1}) using independent variables describing lagoon and environmental characteristics: TAN (mg L^{-1}), TKN (mg L^{-1}), lagoon pH, wind speed (m s^{-1}), and mean air temperature (°C). To avoid multicollinearity, two pools of independent variables were created for which the correlation of any pair of independent variables, within a pool, was smaller than 0.5. For both pools, all possible models (i.e., models resulting from all combinations of independent variables) were fitted, and the model with the smallest Akaike information criterion (AIC; Sakamoto et al., 1986) was selected. The final selected models (the best model from each pool) were subjected to a 10-fold cross-validation for determination of the mean square prediction error (MSPE) with independent data (Hastie et al., 2009). In short, the data were randomly divided into ten folds of similar size. Ten training sets were created by leaving each one of the ten folds out. The ten testing sets were the folds that were left out of each of the ten training sets. The following linear mixed-effects model was used as the framework:

$$y_{ij} = x_{ij}^T \beta + \alpha_i + \varepsilon_{ij} \quad (3)$$

where y_{ij} is the jth record (j = 1, ..., m_i) of NH_3 emissions at the ith dairy (i = 1, ..., 6), x_{ij} is the corresponding vector of independent variables to be selected, β is the vector of fixed regression coefficients, α_i is the random effect of the ith dairy [assumed $N(0,\tau)$], and ε_{ij} is the error [assumed $N(0,\sigma^2)$]. Random effects were assumed to be mutually independent and independent of errors. Models were fitted using maximum likelihood with the lme4 package in R statistical software (Bates et al., 2015). Predictions used to calculate the MSPE, in each fold of the 10-fold cross-validation, were computed only with the fixed regression coefficients, that is:

$$\hat{y}_f = X_f \hat{\beta}_{-f} \quad (4)$$

where $\hat{y}_f$ is vector of predictions in the fth fold, X_f is the corresponding matrix of independent variables in the fth fold, and $\hat{\beta}_{-f}$ is the vector of regression coefficients estimated with a training set without the fth fold.

Our objective in developing these linear mixed models was to investigate the potential use of empirical modeling for predicting NH_3 emissions from lagoons using prediction equations that could be easily used in practical conditions. It was beyond the scope of our study to identify causal functional forms, for example, with the development and use of physically based mechanistic models. Biologically, several variables explain NH_3 emissions potentially through nonlinear functional forms, and these processes are often affected by numerous input variables that were not available in this study. Therefore, our objective was to investigate the use of linear mixed models that are empirical in nature and could be used in practical situations. These empirical models can potentially aid in the development of more detailed mechanistic models.

RESULTS AND DISCUSSION

AVERAGE LAGOON EMISSIONS AND TEMPORAL VARIATION IN LAGOON EMISSIONS

The six lagoons ranged in size from 1,300 to 47,398 m^2, with depths ranging from 0.3 to 2.7 m (table 2). No crust was present on any of the lagoons during any of the monitoring periods; however, there was crust formation on the surface of the settling basin at D6 during much of the year. The average wind speed ranged from 3.6 to 5.3 m s^{-1} (table 4), while the range in daily wind speed at the dairies was 1.4 to 10.9 m s^{-1}. Average ambient air temperatures ranged from 7.9°C to 18.3°C, with daily values of -1.4°C to 31.5°C. Background NH_3 concentrations ranged from 0.004 to 0.008 ppm-v, while the range in NH_3 concentrations measured at the lagoons ranged from the detection limit to 5.8 ppm-v. The average NH_3 emissions on an area basis ranged from 12 to 43 kg ha^{-1} d^{-1}, which falls within the range reported in the literature (table 1). The greatest NH_3 emissions on an area basis were from the lagoon receiving water from the freestall flush dairy (D4), which was expected, as there would be a greater concentration of manure N. Total average NH_3 emissions ranged from 5.4 to 85 kg d^{-1}, with the largest lagoon (D2) having the greatest NH_3 emissions.

At all dairies, NH_3 fluxes tended to be greatest in summer and fall, when temperatures were high (figs. 1 and 2). Several studies have reported this same trend, with elevated NH_3 emissions in summer when temperatures were greater (Bjorneberg et al., 2009; Flesch et al., 2009; Leytem et al., 2013). Mukhtar et al. (2008) reported a 93% increase in emissions from lagoons at dry-lot dairies from winter to summer. The reductions in NH_3 emissions with decreasing temperatures are, in part, due to slowing of the biological and chemical reactions that lead to NH_3 production from urine and feces (Flesch et al., 2009). There were large spikes in NH_3 fluxes during the spring at D2, D3, D4, and D5 during high wind events (>7 m s^{-1}). Generally speaking, NH_3 emissions are driven by chemical reactions in the liquid, diffusion transfer to the liquid surface, and transfer away from the surface (Sommer et al., 1991). With increased wind speed, there would be increased surface mixing as well as a surface-to-air NH_3 concentration gradient that would promote NH_3 emissions. At D5 and D6, there were also spikes in NH_3 emissions when the lagoons were being emptied in the late fall (October) and at D5 during some freeze-thaw events in late December to early January. VanderZaag et al. (2009) also reported that NH_3 emissions increased in manure storage tanks during agitation.

There appeared to be a diurnal trend in NH_3 emissions during the summer and fall, with emissions peaking in the afternoon (~15:00 h; fig. 3), while there was no discernable trend during the winter and spring. However, the variation in hourly NH_3 emissions during all seasons was greater than the diurnal changes in NH_3 fluxes. Several other studies have reported diurnal fluctuations of NH_3 at lagoons, with peak

Table 4. Average wind speed, air temperature, ammonia emission rates, and lagoon characteristics (±SD) of six lagoons located in south-central Idaho (TAN = total ammoniacal nitrogen, TKN = total Kjeldahl nitrogen, and DL = detection limit).

	Dairy					
Parameter	D1	D2	D3	D4	D5	D6
Wind speed (m s^{-1})	4.3 ±2.3	5.3 ±1.4	4.0 ±1.8	4.3 ±2.4	4.3 ±1.5	3.6 ±2.0
Air temperature (°C)	7.9 ±13.9	14.2 ±7.1	18.3 ±7.4	16.0 ±6.0	15.7 ±7.9	14.5 ±6.9
Average background NH_3 (ppm-v)	0.004 ±0.001	0.008 ±0.005	0.004 ±0.003	0.004 ±0.003	0.004 ±0.002	0.005 ±0.003
NH_3 concentration range (ppm-v)	DL to 3.0	DL to 5.8	0.01 to 2.5	0.01 to 2.4	DL to 5.1	DL to 1.6
Average NH_3 emissions (kg ha^{-1} d^{-1})	16 ±13	18 ±6.1	12 ±4.9	43 ±24	24 ±8	26 ±12
Average NH_3 emissions (kg d^{-1})	6 ±4.9	85 ±29	25 ±11	47 ±21	7 ±2.0	5.4 ±2.6
TAN (mg L^{-1})	114±5	151 ±17	77 ±29	573 ±50	92 ±50	177 ±79
TKN (mg L^{-1})	253 ±27	451 ±131	142 ±21	756 ±80	266 ±30	378 ±51
pH	7.7 ±0.3	7.9 ±0.1	8.2 ±0.2	7.9 ±0.2	8.3 ±0.3	8.2 ±0.4
Lagoon temperature (°C)	16 ±2	16 ±2	15 ±5	17 ±3	16 ±4	15 ±4

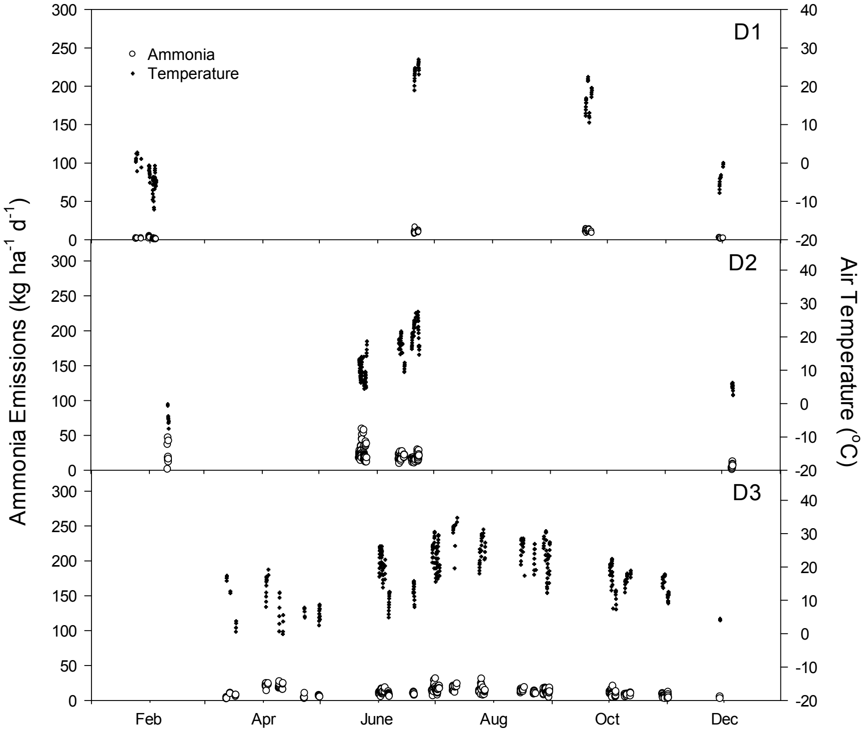

Figure 1. Measured on-farm ammonia emissions and air temperature over time at dairies D1 to D3 in south-central Idaho.

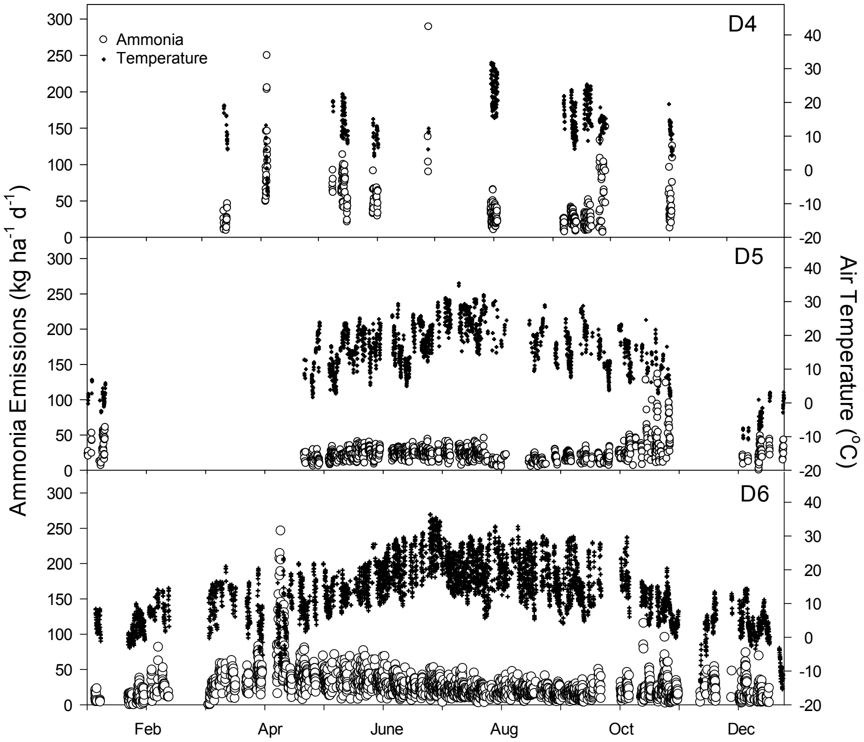

Figure 2. Measured on-farm ammonia emissions and air temperature over time at dairies D4 to D6 in south-central Idaho.

emissions occurring at mid-day (Dore et al., 2004; McGinn et al., 2008; Flesch et al., 2009; Grant and Boehm, 2015). In the present study, both the ambient temperature and wind speed increased from morning until late afternoon, with peaks around 15:00 to 16:00 h, enhancing chemical and biological reaction rates as well as enhancing diffusion and transfer. When both temperature and wind speed were binned hourly (24 h) across the year (D6), there were significant correlations ($r = 0.86$, $p < 0.001$, and $r = 0.88$, $p < 0.001$, respectively) between these variables and annual hourly binned NH_3 emissions (fig. 4), suggesting that these meteorological variables are important drivers of on-farm emissions.

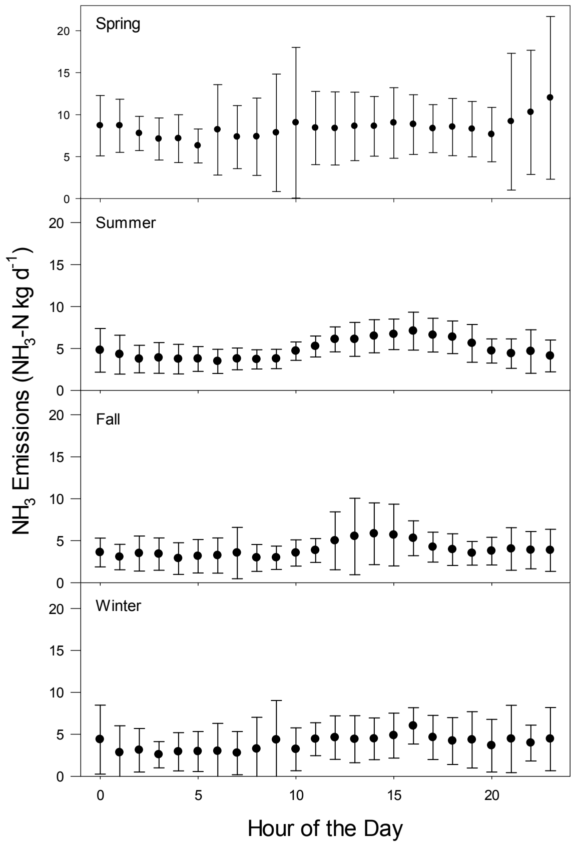

Figure 3. Hourly emissions of ammonia binned over each season (spring, summer, fall, winter) at dairy D6 in south-central Idaho. Error bars are standard deviations of mean hourly values over the season.

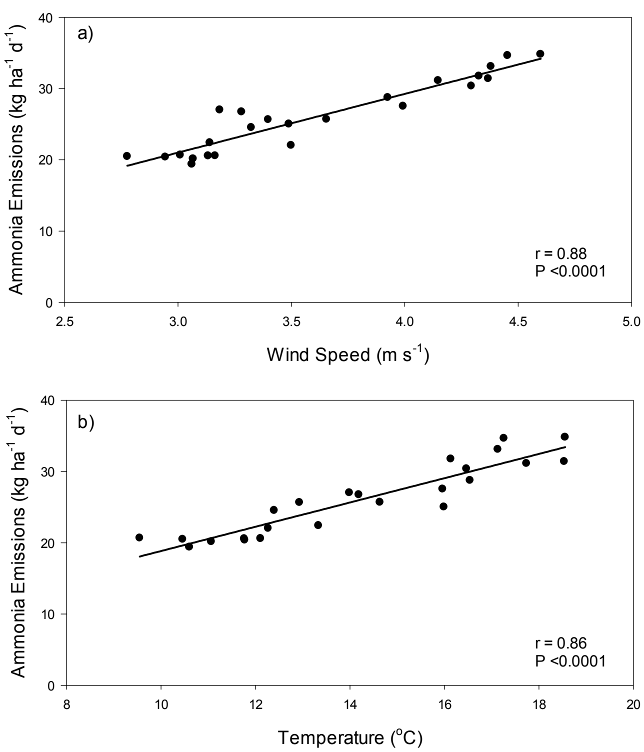

Figure 4. Correlation of hourly binned (a) wind speed and (b) temperature with ammonia emissions at dairy D6 in south-central Idaho.

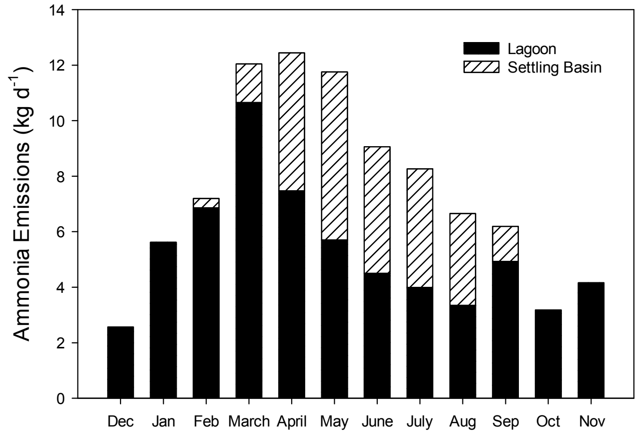

Figure 5. Emissions of ammonia from the lagoon and settling basin of dairy D6 in south-central Idaho.

The NH_3 emissions from the settling basin at D6 represented 29% of the total lagoon system emissions, even though the surface area (201 m^2) represented only 9% of the total liquid system surface area (fig. 5). Emissions of NH_3 from the settling basin were greater during the warmer months, which again would be attributed to increasing chemical and biological reactions with increasing temperatures. It appears that emissions from settling basins are important in the NH_3 emissions budget, particularly during warmer times of the year, and should be accounted for in total manure management emission budgets.

Correlation of Ammonia Emissions with Meteorological Conditions and Lagoon Characteristics

Meteorological conditions and lagoon properties are shown in table 4. The average TAN ranged from 77 to 573 mg L^{-1}, while the average TKN ranged from 142 to 756 mg L^{-1}. The average lagoon pH ranged from 7.7 to 8.3, while the average lagoon temperature ranged from 15°C to 17°C. Correlations of meteorological conditions and chemical characteristics with NH_3 emissions were performed using all available data for all lagoons. For this analysis, data from D5 and D6 were excluded for periods of freeze/thaw events as well as lagoon pumping, as they may mask the influence of these variables on emissions. There were weak correlations between meteorological conditions as well as lagoon temperature and NH_3 emissions (fig. 6). Of the measured parameters, wind speed had the best (although weak) correlation to NH_3 emissions ($r = 0.22$, $p < 0.001$), while air temperature and lagoon temperature did not show any discernable trends ($r = 0.00$ to -0.01, $p = 0.65$ and 0.05, respectively). Dore et al. (2004) also reported little correlation in NH_3 fluxes from a dairy slurry tank and air temperature or wind speed, although the slurry tank had a crust that could have reduced the influence of these parameters on emissions. Grant and Boehm (2015) found that crust formation on dairy basins reduced NH_3 emissions by 24%. Minato et al. (2013) found no relationship between temperature and daily NH_3 emissions from a dairy slurry tank. However, others have reported that NH_3 emissions from lagoons were positively correlated to both wind speed and temperature (McGinn et al., 2008;

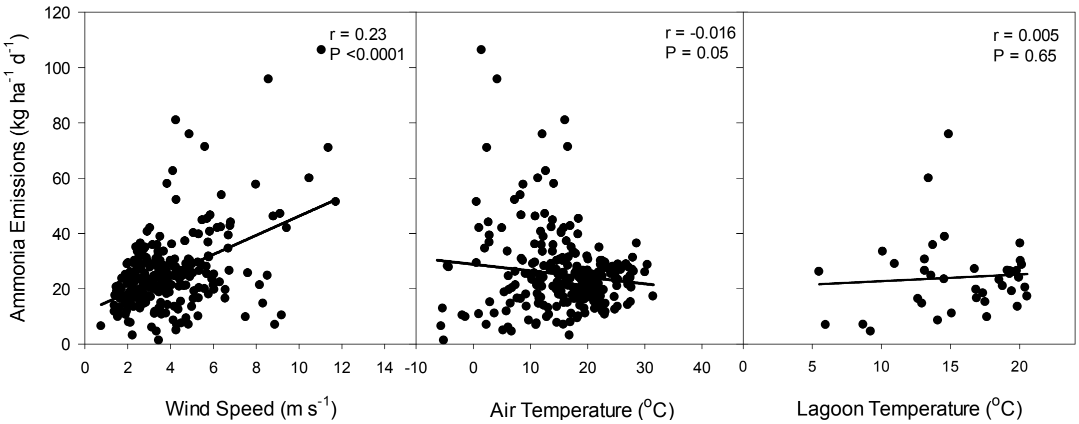

Figure 6. Linear regression of average daily ammonia emissions measured at dairy lagoons D1 to D6 in south-central Idaho with meteorological conditions. Periods of high emissions due to pumping or freeze/thaw events at D5 and D6 were eliminated from the dataset.

Flesch et al., 2009). As shown in figure 4, daily fluctuations in temperature and wind speed are drivers of NH_3 emissions from any given lagoon. However, when applied across a range of lagoons with varying climatic conditions and chemistry, these simple relationships are not as strong, implying that other factors need to be considered when estimating emissions across a broad range of conditions.

There was a much larger effect of lagoon chemical properties on NH_3 emissions, with both TAN and TKN being highly correlated to emissions with r = 0.52 and 0.55 ($p < 0.001$), respectively (fig. 7). As NH_3 emissions are driven, in part, by the amount of substrate in the liquid, this result was expected. As an example, at D6, the trend in NH_3 emissions closely followed that of TAN in the lagoon liquid (fig. 8). Controlled studies have shown that NH_3 emissions are highly correlated with the amount of TAN in liquid dairy

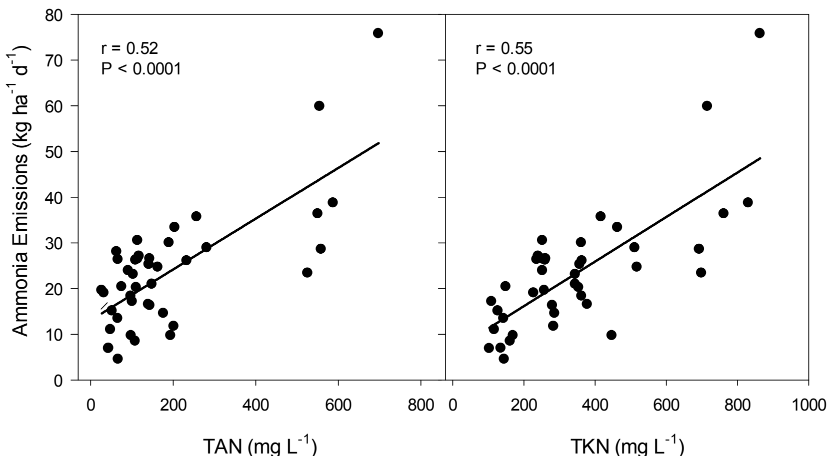

Figure 7. Linear regression of average daily ammonia emissions measured at dairy lagoons D1 to D6 in south-central Idaho with lagoon nitrogen characteristics. Periods of high emissions due to pumping or freeze/thaw events at D5 and D6 were eliminated from the dataset.

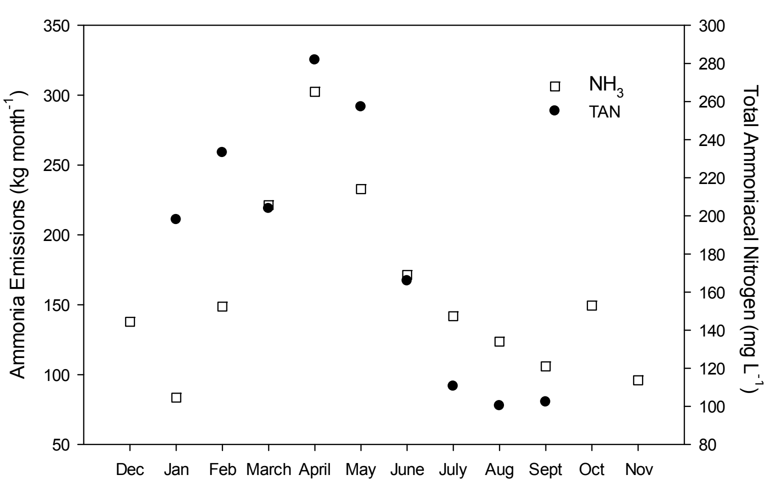

Figure 8. Ammonia emissions and total ammoniacal nitrogen (TAN) measured over time at dairy lagoon D6 in south-central Idaho.

manure (Amon et al., 2006; Sun et al., 2014). On-farm studies have also reported that NH_3 emissions from swine lagoons are highly correlated with NH_x concentrations in the lagoon water (Aneja et al., 2001; Blunden and Aneja, 2008). The present study, which included multiple lagoons with a wide range of TAN and TKN concentrations as well as a wide range of meteorological conditions, illustrates the importance of including lagoon N characteristics in NH_3 emissions estimates.

Because NH_3 emissions are driven by both lagoon characteristics and meteorological conditions, we aimed to derive a method for estimating NH_3 emissions based on these parameters. Therefore, we tested models that included ambient air temperature, wind speed, TAN, TKN, and pH to estimate lagoon NH_3 emissions. The selected models, i.e., the one "best" model (with the smallest AIC) in each one of the two pools of independent variables, are presented in table 5. The square root of the MSPE for each model, determined through 10-fold cross-validation, are also presented in table 5. A complete list of all potential models is presented in table A1 in the Appendix. The model with the smallest prediction error had TKN, wind speed, and mean air temperature as independent variables. The model with the second smallest prediction error included TAN, wind speed, mean air temperature, and pH. It is important to note that, in this case, the AIC and the MSPE (from cross-validation) provide different types of information. The AIC is calculated using the log-likelihood and a penalty for the number of parameters of a mixed model that has both fixed and random effects. The MSPE was calculated through cross-validation with a testing set comprised of data not used for model fitting. Further, predictions used for calculating the MSPE used only the fixed part of the model, i.e., the random effect of dairy was set to zero to simulate the situation in which predictions were wanted for dairies that were not part of the data used in the present study.

The AIC can be used to compare different models (i.e., model selection), with models with smaller AIC preferred. Therefore, its main use is for selecting one model from a set of competing models. The MSPE obtained through cross-validation can also be used to compare competing models, that is, models with smaller root MSPE have better predictive ability (i.e., their predicted values are, on average, closer to the observed values). However, its main use, when expressed as a percent of the observed mean, is to give a relative measure of how large the prediction error is with respect to the mean (i.e., a relative measure of how well the model predicts NH_3 emissions). In this context, the AIC and MSPE were used for model selection and evaluation, respectively. For instance, the prediction errors were substantially large when compared to the mean of the NH_3 emissions. In particular, the square root of the MSPE was approximately 48% of the observed NH_3 emission mean for both models. Moreover, diagnostic plots (fig. 9) suggested that there was considerable variation in the predictions, especially for predictions of relatively greater emissions. The residuals versus fitted values plots (fig. 9) suggest large variation in predictions and possibly a variance that increases with the predicted values. Therefore, the same models described in the previous section were fitted with NH_3 emissions transformed with a natural logarithm operation (Kutner at al., 2004). In essence, the dependent variable was $y' = \log(y)$, and the model selection, fitting, and cross-validation procedures were re-conducted with NH_3 on a natural logarithm scale. The fitted models, with associated AIC and MSPE, are presented in table 5 (models 3 and 4). The selection of variables was slightly modified, and the model with the smallest prediction error had TKN, wind speed, mean air temperature, and pH as independent variables. The model with the second smallest prediction error used TAN instead of TKN as an independent variable. The prediction errors, obtained through cross-validation, ranged from 13% to 15% of the mean natural logarithm of NH_3 emissions. The diagnostics plots show that the predicted values were in better agreement with the observed values (fig. 9, second row of plots).

The on-farm and predicted emissions for all dairies using model 2 (non-transformed data) are shown in figure 10. As indicated in the figure, the predicted emission patterns generally follow those found on farm, although some of the larger spikes in emissions were not captured with the model. The top panel of figure 11 shows all observed versus predicted values using model 2 (non-transformed data). The observed versus predicted values showed good agreement and had an r of 0.47 ($p < 0.001$). It is apparent from the figure that a series of data points is separate from the main grouping. When investigated further, these points were found to be associated with D4, the freestall flush dairy, with the model consistently overpredicting emissions from this lagoon. When the data from this dairy are eliminated (bottom panel), the r improved to 0.57 ($p < 0.001$). This overprediction of emissions at D4 was likely because there were limited data points associated with a freestall dairy in the model input data; therefore, the models developed in this study are more applicable to open-lot dairies that do not use flush sys-

Table 5. Ammonia (NH_3) emission prediction models, associated Akaike information criterion (AIC), and square root of the mean square prediction error (RMSPE) obtained through cross-validation. Emissions are either expressed in kg ha^{-1} d^{-1} or in a natural logarithm scale.

Model[a]	Prediction Equation[b]	AIC	RMSPE[c] (kg ha^{-1} d^{-1})	RMSPE[c] (% of mean)
Original scale				
1	$NH_3 = -34.7 + 0.098(TKN) + 3.38(Wind) + 0.492(T_m)$	1,667	11.9	47.9
2	$NH_3 = -78.3 + 0.112(TAN) + 3.19(Wind) + 0.437(T_m) + 7.45(pH)$	1,650	12.0	48.0
Log scale				
3	$Log(NH_3) = 2.18 + 0.003(TKN) + 0.119(Wind) + 0.030(T_m) - 0.158(pH)$	66.9	0.406	13.1
4	$Log(NH_3) = 0.027 + 0.004(TAN) + 0.110(Wind) + 0.028(T_m) + 0.161(pH)$	35.9	0.479	15.4

[a] Models 1 and 2 have the smallest AIC in pools of independent variables 1 and 2. Models 3 and 4 use the same pools of independent variables as models 1 and 2 but with the NH_3 transformed with a natural logarithm operation.

[b] Wind is the wind speed (m s^{-1}) ranging from 1.39 to 11.7, T_m is the mean air temperature (°C) ranging from 0.60 to 31.5, TKN is total Kjeldahl nitrogen (mg L^{-1}) ranging from 110 to 855, and TAN is the total ammoniacal nitrogen (mg L^{-1}) ranging from 18.2 to 676.

[c] With 10-fold cross-validation and using only fixed regression coefficients.

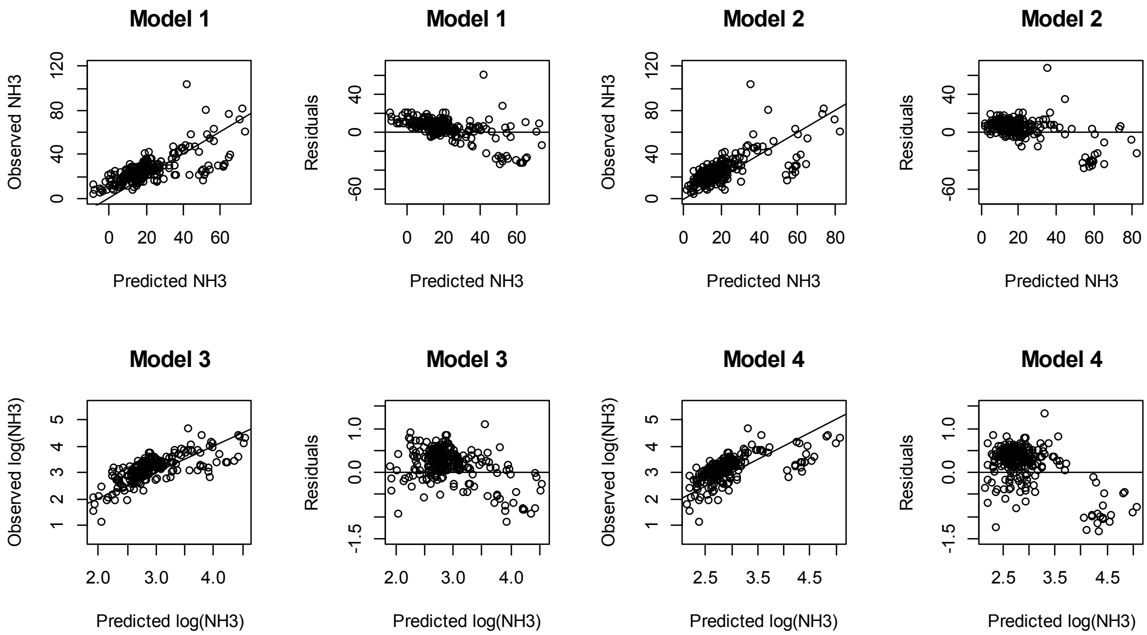

Figure 9. Observed versus predicted values, and residuals versus predicted values. The first row contains models with NH_3 emissions expressed in kg ha^{-1} d^{-1}, and the second row contains models with NH_3 emissions on a natural logarithm scale. Residuals and predictions were obtained through 10-fold cross-validation and were calculated using only the fixed-effects parameters.

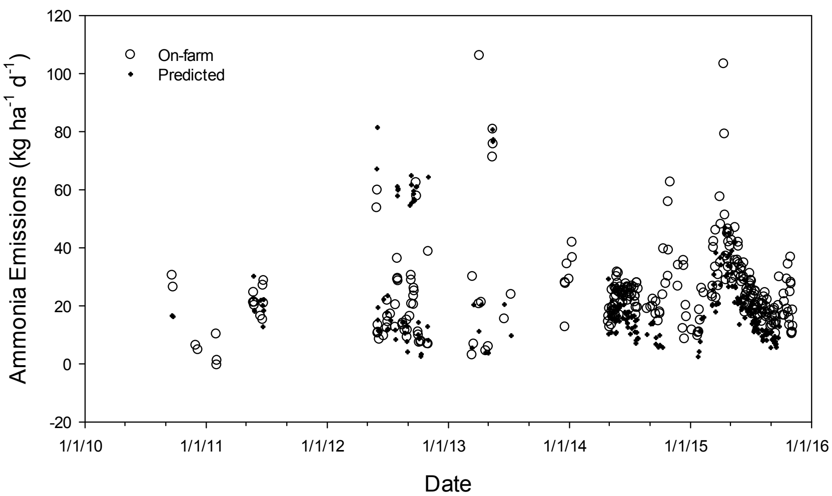

Figure 10. On-farm ammonia emissions for all dairy lagoons (D1 to D6) compared with emissions predicted using model 2 (table 5).

tems for manure management. The two data points above 60 kg ha^{-1} d^{-1}, where the model greatly underpredicted emissions, were associated with the spike in emissions at D6 during the spring during a high wind event, suggesting that although the model includes wind speed as a variable, it may not be able to simulate large spikes in emissions occurring during high wind events. It is important to point out that extrapolation outside data boundaries is always dangerous and can lead to nonsensical predictions. Minimum and maximum values for all independent variables are presented in the footnote of table 5. Model users should not use the NH_3 prediction models outside these ranges.

Overall N Balance and Model Prediction of Nitrogen and Ammonia Losses

The overall annual on-farm N balance for D6 is presented in table 6. The estimated on-farm N consumed in the feed was 64,240 kg with excreted N + straw bedding estimated at 51,292 kg. Nitrogen exported in milk N was 13,654 kg, or 21% of N intake, similar to the 19% of N intake reported by Todd et al. (2015). Nitrogen excreted was 69% of feed intake. The amount of manure N leaving the storage facilities was estimated at 26,448 kg, and the total N lost from housing and manure storage was 24,844 kg. This suggests that ~48% of the excreted N (or 39% of fed N) was lost through volatilization, leached from the lots, or remained in the lot soil. Todd et al. (2015) reported that ~56% of excreted N and 43% of fed N was lost through NH_3 volatilization from dry lots and lagoons on a dairy in Texas. Their estimates are slightly higher than ours; however, their estimates were based on data from a short period in the summer, when emissions would be greatest. The cumulative NH_3-N loss from the lagoon system was measured as 2,234 kg (settling basin + lagoon), which represented 9% of the total N lost and 65% of

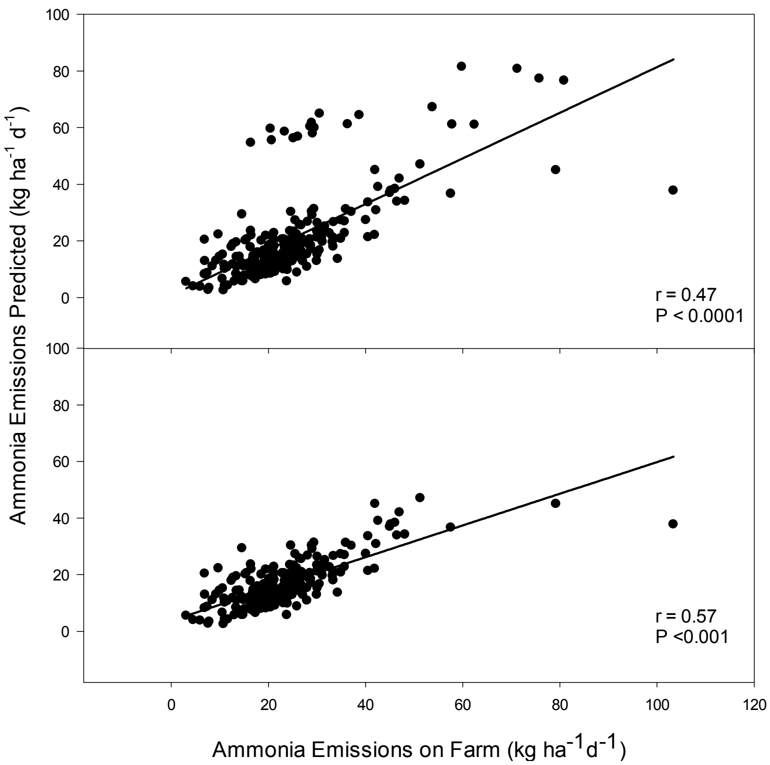

Figure 11. On-farm ammonia emissions versus predicted emissions using model 2 (table 5). The top panel includes all data, and the bottom panel excludes D4 (freestall flush dairy).

Table 6. On-farm nitrogen balance and modeled values using the Integrated Farm System Model.

Parameter	On-Farm Measurement	IFSM Simulated
N consumed in feed (kg N year^{-1})	64,240	57,859
Target CP of feed ration, lactating (%)	15.5	15.5
N exported in milk (kg)	13,654	13,688
N exported in cattle (kg)	1,550	1,838
N excreted (kg)	44,427	42,304
N applied in bedding (kg)	6,865	6,865
N excreted + straw N (kg N year^{-1})	51,292	49,169
N manure leaving storage, solid + lagoon (kg N year^{-1})	26,448	20,084
N lost in housing + manure storage (kg N year^{-1})	24,844[a]	29,085
N lost of excreted N (%)	48	69
N lost of fed N (%)	39	50
NH_3-N lost from lagoon (kg N year^{-1})	2,234	2,294
NH_3-N lost from lagoon (% of total N lost)	9	8
NH_3-N lost of total lagoon N (%)	65	67

[a] Does not account for N that could be retained in lot soils or lagoon.

the estimated N entering the lagoon. The NH_3-N lost from the lagoon system was approximately 5% of the N intake of the lactating herd. Todd et al. (2015) reported an NH_3-N loss of 2% of N intake during the summer at a dairy lagoon in Texas.

The IFSM-simulated values for N balances at D6 are also presented in table 6. The IFSM model simulated lower feed N intake and thus a higher efficiency in conversion of feed N to milk (25% vs. 21%). Feed intake recorded on farm was greater than that predicted by the model. Compared to expected feed intakes for a herd of these characteristics (NRC, 2001), this total feed N intake is high. However, many factors influence feed intake and the N concentration in feed, so the measured intake is plausible. The IFSM-simulated values for N excretion + bedding were similar to the on-farm estimated values, indicating good agreement in N excretion. The model predicted that total N lost in housing and manure storage was 17% greater than that estimated on farm, and the IFSM prediction for N leaving the storage facilities in manure (solid + liquid) was 24% lower. IFSM estimated that 69% of manure N (excreted plus bedding) was lost while 50% of fed N was lost, which was greater than that measured on farm. This rather large difference was most likely due to misrepresentation of the manure stacks by the model. This relatively new component in IFSM was extensively evaluated in simulating emissions from dynamic and static compost windrows (Bonifacio et al., 2017). Further evaluation and refinement of this component is needed for simulating this type of manure stack.

Simulated NH_3-N losses from the lagoon (2,294 kg) were similar to the losses determined on farm. The simulated NH_3-N emissions over time indicate that the greatest loss of NH_3 occurred during the summer months (fig. 12). This general trend is similar to that measured on farm (fig. 5); however, IFSM simulations indicated less NH_3-N emissions during April and May (days 100 to 150) and more during the summer months (days 200 to 250) than measured on farm. Further investigation is needed to explain this shift in emissions toward the spring period. The measured data indicate that high emissions can occur when the lagoon temperature begins to warm in the spring, and the processes involved are not full represented by the model. Simulated N loss was 67% of that entering the lagoon, similar to the 65% determined on farm. This close agreement indicates that the model appropriately predicted the annual emission despite the disagreement in the pattern throughout the year.

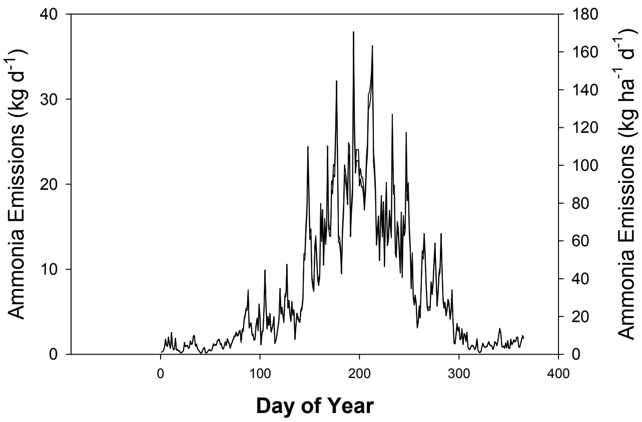

Figure 12. Simulated ammonia emissions from the lagoon system at D6 using the Integrated Farm System Model.

CONCLUSIONS

The average NH_3 emissions from lagoons in southern Idaho ranged from 12 to 43 kg ha^{-1} d^{-1} (5.4 to 85 kg d^{-1}). These emissions varied seasonally, with higher emissions during warmer periods of the year as well as temporary high fluxes in emissions associated with high wind events, freeze/thaw events, and agitation of the lagoons. Emissions were most closely associated with the amount of N in the lagoons (TAN or TKN) as well as the temperature, wind speed, and pH of the lagoon. The settling basin at one dairy contributed 29% of total lagoon system emissions, with higher emissions in summer than in winter. Thus, this source must be included when calculating annual emission estimates from manure lagoon facilities. Because emissions are variable over the year, data must be collected over time periods long enough to capture these variations in emissions for accurate estimates of annual emission factors. For one dairy, an N balance indicated that lagoon NH_3-N losses represented 9% of total N lost from the facility, 65% of the total N entering the lagoon, and 5% of lactating herd N intake. Use of a process-based model to estimate on-farm N flows produced similar values for N excretion and NH_3-N loss from the lagoon, but there was some disagreement in the distribution of emissions throughout the year. Future modeling efforts and emission factor estimates must capture the temporal variability in emissions and include the key variables driving NH_3 emissions, such as the N content and pH of lagoon manure, in addition to meteorological variables.

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APPENDIX

Table A1. Ammonia (NH_3) emission prediction models, associated Akaike information criterion (AIC), and square root of the mean square prediction error (RMSPE) obtained through cross-validation. Emissions are expressed in kg ha^{-1} d^{-1}.

Prediction Equation[a]	AIC	RMSPE[b] (kg ha^{-1} d^{-1})	RMSPE[b] (% of mean)
NH_3 = –28.0 + 0.096(TKN) + 3.386(Wind) + 0.498(T_m) – 0.752(pH)	1,669	11.8	47.4
NH_3 = 50.5 + 4.015(Wind) + 0.349(T_m) – 6.093(pH)	1,715	11.4	45.8
NH_3 = –34.0 + 0.142(TKN) + 0.124(T_m) + 0.497(pH)	1,749	17.5	70.3
NH_3 = 93.2 – 0.22(T_m) – 8.022(pH)	1,810	12.2	49.0
NH_3 = –23.5 + 0.075(TKN) + 2.828(Wind) + 1.077(pH)	1,689	10.1	40.6
NH_3 = 41.3 + 3.49(Wind) – 3.88(pH)	1,723	11.3	45.4
NH_3 = –33.2 + 0.135(TKN) + 1.021(pH)	1,748	16.6	66.7
NH_3 = 105 – 9.99(pH)	1,811	12.2	49.0
NH_3 = −34.7 + 0.098(TKN) + 3.38(Wind) + 0.492(T_m)	**1,667**	**11.9**	**47.9**
NH_3 = 3.07 + 4.08(Wind) + 0.264(T_m)	1,720	11.5	46.2
NH_3 = – 29.7 + 0.141(TKN) + 0.128(T_m)	1,747	17.4	69.9
NH_3 = 31.1 – 0.346(T_m)	1,815	12.5	50.2
NH_3 = –13.6 + 0.071(TKN) + 2.83(Wind)	1,687	10.0	40.2
NH_3 = 9.70 + 3.63(Wind)	1,723	11.4	45.8
NH_3 = –23.9 + 0.132(TKN)	1,746	16.3	65.4
NH_3 = −78.3 + 0.112(TAN) + 3.19(Wind) + 0.437(T_m) + 7.45(pH)	**1,650**	**12.0**	**48.0**
NH_3 = –100.6 + 0.158(TAN) + 0.058(T_m) + 11.7(pH)	1,726	16.0	64.2
NH_3 = –80.6 + 0.104(TAN) + 2.61(Wind) + 9.23(pH)	1,667	11.2	45.0
NH_3 = –100.5 + 0.155(TAN) + 11.9(pH)	1,724	15.7	63.0
NH_3 = –14.8 + 0.085(TAN) + 3.35(Wind) + 0.476(T_m)	1,654	10.7	43.0
NH_3 = 0.056 + 0.117(TAN) + 0.09(T_m)	1,737	13.1	52.6
NH_3 = –0.612 + 0.070(TAN) + 2.73(Wind)d	1,674	9.9	39.8
NH_3 = 2.71 + 0.112(TAN)	1,735	12.7	51.0

[a] Wind is the wind speed (m s^{-1}) ranging from 1.39 to 11.7, T_m is the mean air temperature (°C) ranging from 0.60 to 31.5, TKN is total Kjeldahl nitrogen (mg L^{-1}) ranging from 110 to 855, and TAN is the total ammoniacal nitrogen (mg L^{-1}) ranging from 18.2 to 676.

[b] With 10-fold cross-validation and using only fixed regression coefficients.