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Use of AERMOD to determine a hydrogen sulfide emission factor for swine operations by inverse modeling

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ABSTRACT

This study was conducted to determine both optimal settings applied to the plume dispersion model, AERMOD, and a scalable emission factor for accurately determining the spatial distribution of hydrogen sulfide concentrations in the vicinity of swine concentrated animal feeding operations (CAFOs). These operations emit hydrogen sulfide from both housing structures and waste lagoons. With ambient measurements made at 4 stations within 1 km of large swine CAFOs in Iowa, an inverse-modeling approach applied to AERMOD was used to determine hydrogen sulfide emission rates. CAFO buildings were treated as volume sources whereas nearby lagoons were modeled as area sources. The robust highest concentration (RHC), calculated for both measured and modeled concentrations, was used as the metric for adjusting the emission rate until the ratio of the two RHC levels was unity. Utilizing this approach, an average emission flux rate of 0.57 $\mu g\ m^{-2}\ s^{-1}$ was determined for swine CAFO lagoons. Using the average total animal weight (kg) of each CAFO, an average emission factor of $6.06 \times 10^{-7}\ \mu g\ yr^{-1}\ m^{-2}\ kg^{-1}$ was calculated. From studies that measured either building or lagoon emission flux rates, building fluxes, on a floor area basis, were considered equal to lagoon flux rates. The emission factor was applied to all CAFOs surrounding the original 4 sites and surrounding an additional 6 sites in Iowa, producing an average modeled-to-measured RHC ratio of 1.24. When the emission factor was applied to AERMOD to simulate the spatial distribution of hydrogen sulfide around a hypothetical large swine CAFO (1 M kg), concentrations within 0.5 km from the CAFO exceeded 25 ppb and dropped to 2 ppb within 6 km of the CAFO. These values compare to a level of 30 ppb that has been determined by the State of Iowa as a threshold level for ambient hydrogen sulfide levels.

1. Introduction

A large increase in the number of intensive hog production facilities in the United States has been evident over the last two decades. The United States Environmental Protection Agency (USEPA) refers to these operations as "confined animal feeding operations" (CAFOs). These facilities are operated with a high animal density in large buildings that hold manure in subfloor pits that are periodically flushed into nearby lagoons or directly sprayed on local agricultural fields. Given their potential to be sources of ambient odor and gas concentrations from the manure slurry, several research efforts have noted a relationship between living in proximity to these facilities and the prevalence of adverse health effects (Wing and Wolf, 2000; Chrischilles et al., 2004; Merchant et al., 2005; Sigurdarson and Kline, 2006).

Given the difficulty and expense of obtaining ambient air pollutant measurements over a large region, plume dispersion modeling is an attractive alternative for the assessment of the impacts of CAFOs on air quality in surrounding areas. Plume dispersion models developed by the USEPA for Clean Air Act (CAA) compliance were initially developed for tall smoke stacks emitting a continuous pollutant stream. However, CAFOs are low-level sources occupying a large area, and may produce pollutants on an intermittent basis (Hoff et al., 2002). Regardless, a number of attempts have been made to use air dispersion models to estimate odor emissions from CAFOs. In these studies odor was typically measured via a human odor panel and were primarily conducted for the purpose of determining setback distances from nearby residences (Heber, 1997; Piringer and Schauberger, 1999; Jacobson et al., 2001; Zhu et al., 2000; Schauberger and Piringer, 2001; Guo et al., 2004; Hoff et al., 2008). Furthermore, Wing et al. (2008) found a significant correlation between odor complaints and hydrogen

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sulfide concentrations at low wind speeds as well as PM_{10} at high wind speeds. However, only one published study described the use of a dispersion model to estimate gaseous concentrations (ammonia and hydrogen sulfide) downwind from CAFOs (MPCA, 2003), and none were found that modeled particulate contaminants.

In 2004, a Technical Workgroup organized by the Iowa Department of Natural Resources (IDNR) determined that, of all dispersion models available, AERMOD (USEPA, 2004) was the most suitable for modeling the dispersion of airborne contaminants from CAFOs (IDNR, 2004). Like all dispersion models, AERMOD requires an emission rate for each pollutant emitted at each source in the model in order to calculate downwind concentrations. However, the USEPA currently does not provide AP-42 emission factors for agricultural sources that could be used to calculate an emission rate for use in AERMOD.

The goal of this study, therefore, was to determine an emission factor for hydrogen sulfide emitted from CAFOs scalable by total operation animal weight to derive a site-specific emission rate for use in AERMOD. This study focused on hydrogen sulfide because (1) it represents one of the most hazardous gases emitted from these facilities (Roth and Goodwin, 2003); (2) it can be measured accurately to the part-per-billion level by direct-reading instruments; (3) its primary source in rural areas containing swine CAFOs are the CAFOs; and (4) the Iowa State legislature adopted a limit on ambient hydrogen sulfide concentrations which are not to exceed 30 ppb more than seven times over a year (IAB, 2004).

1.1. CAFO emission rates

As mentioned by Bunton et al. (2007), studies involving the use of dispersion models applied to CAFOs are complicated by a lack of accurate information on the gaseous emission rates emanating from this source type. However, a number of studies have been conducted to either directly measure the emission rate of gases from swine CAFOs or estimate emission rates using an inverse-modeling approach.

Typically the direct-measurement studies were conducted by detecting the interior concentration of a gas as well as the total exhaust flow rate exiting a CAFO from exhaust fans. The emission rate was then determined by multiplying the concentration by the flow rate (Blunden et al., 2008; Ni et al., 1999, 2000, 2002; Kim et al., 2008; Zhu et al., 2000; Heber et al., 1997; Hoff et al., 2005; Jerez et al., 2005; Jacobson et al., 2005; Koziel et al., 2005). An alternative approach involved using the decay rate of a tracer gas to estimate exhaust flow rate (Demmers et al., 1999; Gay et al., 2003). Studies of this type can be complicated when both the gas emissions from buildings and on-site waste lagoons are considered (Harper et al., 2000; Zahn et al., 2001; Aneja et al., 2001). The results from these studies vary considerably and none of the emission rates have been applied to a published study involving dispersion modeling.

The inverse-modeling method has been utilized to avoid the complications of the direct-measurement approaches, and to obtain a gas emission rate for an entire source, including both buildings and lagoons. Inverse-modeling is conducted by first applying an arbitrary emission rate to the model, Q_o; running the model to determine a concentration value at a receptor location, C_o; then comparing that value with a background-compensated measurement at the same location, $C - C_b$ (Flesch et al., 2005). The next iteration of this procedure will result in a perfect association between modeled and measured concentration by applying a new emission rate, Q, to the model that is the result of adjusting the original rate by the following ratio:

$$Q = Q_o(C - C_b)/C_o \quad (1)$$

This methodology has been applied to CAFOs and other low-level emission sources utilizing a variety of dispersion model types (McCulloch et al., 1998; Rege and Tock, 1996; Hensen et al., 2009). Flesch et al. (2005, 2007, 2009) has developed and verified the use of inverse modeling applied to a backward Lagrangian Stochastic (bLS) dispersion model to predict emission rates from agricultural facilities. This technique accurately inferred emission rates from sources that were far enough upwind of a receptor as to not be influenced by topographical features that could influence wind trajectories. Sanz et al. (2010) further verified this technique when applied to the emissions of ammonia from surface-applied swine slurry.

1.2. Study rationale

The present study was motivated by the desire to use AERMOD to predict ambient hydrogen sulfide concentrations produced by swine CAFOs over areas large enough to encompass affected nearby residences. An inverse-modeling approach to determine CAFO emission rates was motivated by the availability of a large data set of gas concentration measurements made with accurate instruments at point locations near large CAFOs over at least a year-long period by the IDNR (2009). This approach differs from that of other published inverse-modeling efforts in that it may not result in rates that accurately estimate the gaseous output of swine CAFOs because AERMOD has not been adequately validated for this purpose. Furthermore, AERMOD does not permit the incorporation of chemical reactions. For example, Hydrogen sulfide can react with hydroxyl radicals and ozone with a resulting residence time in the atmosphere of approximately 3 days (Warnek, 2000). However, given the large data set available, it can be assumed that the rates determined by this method will represent values that maximize the accuracy of AERMOD to predict large-scale (>100 m), downwind concentrations given a prescribed set of input options. Furthermore, it was our intention to constrain our determination of an emission rate based on the limited information that would typically be available when modeling a swine CAFO: the location, footprint area, and permitted animal weight capacity of the facility.

1.3. Evaluation criteria

Inverse modeling is relatively simplistic if a small set of measured downwind concentrations are to be compared to the associated modeled values. However, the comparison of modeled-to-actual concentrations at a receptor site over a time period long enough to include multiple changes in meteorological conditions can involve the assessment of thousands of observations. Therefore, an evaluation criteria is required that includes multiple pairs of modeled-versus-measured data. One approach is to perform a correlation analysis of the entire distribution of measured data obtained during the sampling period relative to the time-associated modeled values. The emission rate is then adjusted to provide the highest correlation possible. However, the USEPA decided that this approach was impracticable when using AERMOD (USEPA, 2005) because "estimates of concentrations that occur at a specific time and site are poorly correlated with actually observed concentrations." The USEPA recognized that this poor correlation was due to a variety of uncertainties such as deviations from an exact knowledge of the true wind direction where, for example, an error of 5–10° can lead to concentration errors of 20–70 percent.

The USEPA (2005) concluded that dispersion models are more reliable when estimating long time-averaged concentrations over simple topography, and when estimating the magnitude of the highest concentrations regardless of where or when they occur relative to a source location. This second criteria implies that

a comparison can be made to relate the magnitude and probability of occurrence of the highest concentrations produced by a model with that of measured values as they occurred over, say, a year-long period in any radial direction downwind from the source. This approach was taken by Perry et al. (2005) when validating AERMOD using multiple field locations in which both the emission rate was known and downwind concentrations were measured. The study group relied on quantile–quantile (Q–Q) plots that pair modeled and measured values after ranking both sets of data as a visual interpretation of the accuracy of the model. The plotted points were evaluated relative to lines placed a factor of 2 above and below a 1:1 line, representing bounds of model accuracy deemed reasonable by the USEPA (USEPA, 2005).

Given that AERMOD is most accurate when predicting the likelihood of high concentration events, a comparison could be made between the highest measured and modeled concentrations to determine the accuracy of the model. However, Perry et al. (2005) used a metric known as the robust highest concentration (RHC), first developed by Cox and Tikvart (1990), for this comparison. The equation used to calculate the RHC mitigates the potential for unusual events to influence the highest concentration by incorporating a set of the largest values from a distribution of measurements to determine the highest probable concentration. The ratio of the RHC developed from modeled values to that of measured values was then assessed for its proximity to unity. A similar, but inverted, approach was used during this study whereby an accurate emission rate applied to the model was identified by adjusting it until the RHC of the modeled results was identical to the RHC of the values measured at a downwind monitoring site over the entire time period as those modeled.

2. Methods

2.1. Study sites and instrumentation

The IDNR, in cooperation with the State Hygienic Lab at the University of Iowa, conducted a sampling assessment of hydrogen sulfide and ammonia in the vicinity of CAFOs at various locations in Iowa. Data collected during 2003 and 2004 at 10 sites was used for this study. Depending on the site, between 6 and 12 months of hourly data was available for analysis. Hydrogen sulfide was measured with a continuous-pulsed fluorescent analyzer (Model 450C, Thermo Electron Corp., Waltham, MA). This instrument was operated with a range of 0–200 ppb and averaging time of 300 s, which provided a lower detection limit (LDL) of 0.06 ppb. The instrument zero level was not auto-corrected but was manually re-zeroed when the drift exceeded 2 ppb. Given this procedure for offsetting instrument drift, a true background level of hydrogen sulfide could not be obtained from the IDNR data and was therefore considered to be 0 ppb. These monitors were operated continuously to obtain hourly observations at fixed sites located on residential properties adjacent to the CAFOs. The monitors were stored in temperature-controlled trailers. A sample line with entry probe at 4 m was positioned on top of the trailer.

Of the various sites measured by the IDNR, four separate sites (Table 1, sites A–D) that were primarily influenced by swine CAFOs were chosen for determining an average hydrogen sulfide emission factor. At each site, the nearest CAFO to a measuring station that had no other CAFO within 5 km in the same line as that between the CAFO and the monitoring station was selected to apply the inverse-modeling method. Sites A–C met this criterion, whereas site D contained a second large CAFO (1.6 M kg) 4.3 km away from the station, and within 5° of the line between the station and the closest CAFO. Therefore, the emission rate for the closest CAFO was determined with the more removed CAFO added as another source. To negate influence by other CAFOs in the area, the data set of concentrations applied to the inverse-modeling approach was constrained to times when winds were blowing within ±22° from the selected CAFO.

After an average emission factor was determined by inverse-modeling of a single CAFO at sites A–D, the factor was tested by applying it to all CAFOs within 7 km of the monitoring stations at sites A–D as well as those surrounding an additional 6 sites located in other areas of northern Iowa (Table 1). Of the 10 sites, 7 utilized meteorological data from Mason City, IA. The other three sites were widely separated and used meteorological data from Waterloo, IA, Sioux City, IA, and Omaha, NE.

2.2. Meteorological data

AERMOD requires hourly surface and upper-air meteorological data as inputs. Wind speed and direction were recorded at each sampling site. Additional meteorological data needed by AERMOD such as temperature, dew point, pressure, and solar radiation were not recorded. Therefore, all necessary ground-level data was obtained from the nearest National Weather Service (NWS) station located in Mason City, Iowa, located 40–60 km from the original 4 study sites. Upper-air data was recorded in Omaha, Nebraska, approximately 260 km from the study region and contains information on the vertical aspects of the atmosphere including pressure, temperature, and relative humidity. Both data sets were pre-processed and made available by the IDNR Air Quality Bureau

Table 1
Attributes of the swine CAFOs closest to the monitoring station at each site.

Site	Swine Type[a]	Permitted Weight kg	Dir. & Dist.[b] deg., m	Met Data[c] hours	Met Data[d] hours	Number of CAFOs[e]
A	Finishing	1,123,000	185°, 975	1206	8784	8
B	Finishing	1,048,000	170°, 940	973	4416	9
C	Finishing	1,048,000	20°, 1095	422	4416	9
D	Sow	810,000	200°, 710	1119	5880	8
E	Sow	663,000	27°, 680		5880	19
F	Sow	796,000	172°, 860		3768	17
G	Finishing	857,000	260°, 1360		5516	4
H	Finishing	1,123,000	288°, 1050		5136	22
I	Finishing	1,366,000	350°, 820		8765	3
J	Finishing	1,021,000	140°, 230		8760	26

[a] Finishing CAFOs are used to raise pigs to market weight (100 kg); Sow CAFOs are used to hold sows prior to and during pregnancy.
[b] Direction (degrees from true north) and distance (m) from monitoring station.
[c] Hours of meteorological data analyzed based on hours of the year during which wind was blowing from the site to the monitoring station ±22°.
[d] Hours of meteorological data analyzed when applying the overall emission factor to all sites.
[e] CAFOs within 7 km of a monitoring site to which the overall emission factor was applied.

for use in AERMOD. Meteorological data between 2003 and 2004 that coincided with the time periods of the IDNR measurements at each site were utilized. As explained further in Section 2.4, our goal was to develop a method for accurately modeling the highest concentrations from swine CAFOs regardless of when they occurred relative to their measured occurrence. Therefore, we assumed that the meteorological conditions actually occurring at the sampling station are represented in the meteorological data set obtained, with the understanding that they may not coincide temporally. Furthermore, in the part of the Midwestern US where the study took place there are no major terrain features to significantly change the meteorology of the sample site relative to that of the NWS station.

2.3. Modeling options

A vendor-supplied modeling package (AERMOD View™ ver. 6.7.1, Lakes Environmental, Waterloo, Ontario) was used to execute AERMOD. All modeling was performed with the dispersion coefficient set as "rural" and terrain height option set to "elevated". The modeled pollutant type was sulfur dioxide given no option for hydrogen sulfide. All pollutant types available in AERMOD are treated as inert compounds except sulfur dioxide when utilizing the "urban" dispersion option, and therefore produce the same model output for a given emission rate. The IDNR monitoring sites were treated as receptor locations with a height of 4 m, coinciding to the height of the inlet of the sample probe.

The adjustable options for modeling CAFOs as sources in AERMOD were source type and emission height. Source type options include point, area, line, and volume. All sites selected for initial analysis contained multiple buildings and at least one lagoon, where both structures were considered sources of hydrogen sulfide. For example, the site-A CAFO consists of 10 buildings, each 88 m × 15 m, and two adjacent and adjoining lagoons with total dimensions of 130 m × 280 m (Fig. 1). Given the source type options available, volume sources were chosen to represent the CAFO buildings and rectangular area sources to represent the lagoons. The building downwash option is not available for these source types. The lack of this feature is also justified given that our intent was to model the dispersion of hydrogen sulfide over a wide radius around a CAFO (>100 m) where building wake effects will have little influence on ground-level concentrations.

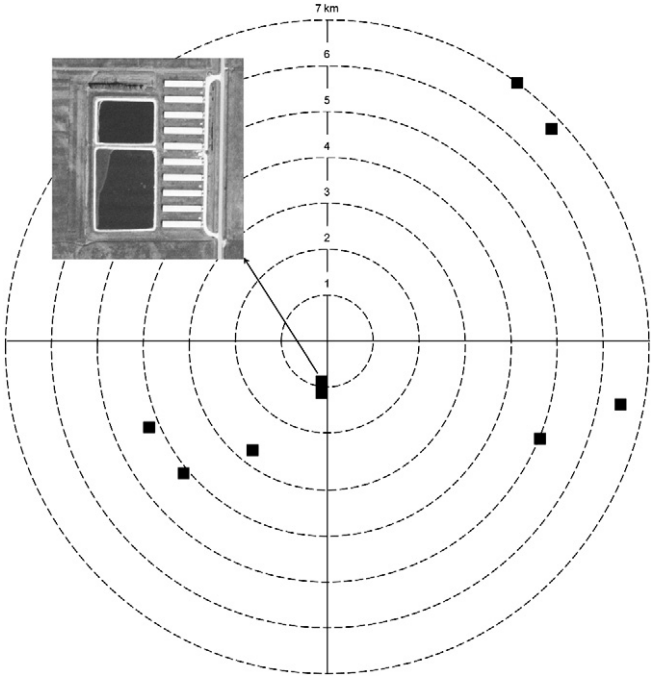

Fig. 1. Site A swine operations relative to the sampling station receptor location. A detail of the large CAFO used to develop an emission factor is also provided.

Volume sources are modeled as a virtual point source with an initial lateral and vertical dimension to the plume established at the point of release of the plume. These dimensions are scaled in AERMOD relative to the physical dimensions of the volume, which results in these "initial" dimensions being smaller than the physical dimensions (USEPA, 1995). Volume sources can only be applied in AERMOD with a square base. Therefore, a long, rectangular swine building was modeled with multiple, contiguous volumes with base length and width equal to the width of the building (typically 14 m) and height equal to the height of the building roof peak (6.5 m). In some cases the volumes were not contiguous in order to have an integer number of square volumes with one exactly on either end of the building floor area. The area source type used for lagoons can be applied to AERMOD given the length and width of the rectangular area of the lagoon as well as the degrees from north that the rectangle is tilted. Building and lagoon dimensions for each site were obtained from aerial photographs available on the internet.

The emission release height of volume and area sources can be altered in AERMOD. Likewise, an initial vertical dimension can be established for an area source that accounts for an initial loft in the plume from the source. A sensitivity analysis of release height and loft was performed to determine whether there was justification for values other than zero for both. An emission factor was determined for Site A (as described in Section 2.4) modeled as one large area source or as a combination of area and volume sources. For each scenario, the release height was varied between 0 and 6 m. Likewise, the loft was varied from 0 to 6 m when the Site-A CAFO was modeled as an area source and an emission factor determined for each set of options.

2.4. Emission factor determination

The four sites, A–D, were used to obtain an average hydrogen sulfide emission factor with units of $\mu g\ s^{-1}\ m^{-2}\ kg^{-1}$. To obtain this emission factor by inverse modeling, an arbitrary emission flux, F_o with units of $\mu g\ s^{-1}\ m^{-2}$, was chosen for application to the lagoons as area sources. A review was conducted of all publications in which the hydrogen sulfide emission flux for waste storage structures (Blunden and Aneja, 2008; Gay et al., 2003; Zahn et al., 2001; Lim et al., 2003; Jacobson et al., 1999) or building types (Blunden et al., 2008; Gay et al., 2003; Hoff et al., 2005; Jacobson et al., 2005; Jerez et al., 2005; Kim et al., 2008; Koziel et al., 2005; Ni et al., 1999, 2000, 2002; Zahn et al., 2001; Zhu et al., 2000) was reported. These studies revealed a geometric mean (GM) of storage flux rates of 3.18 $\mu g\ s^{-1}\ m^{-2}$ ($n = 14$) and a GM of building flux rates of 5.22 $\mu g\ s^{-1}\ m^{-2}$ ($n = 32$). A t-test on the log-transformed rates did not demonstrate a significant difference between the two source types ($p = 0.58$). Therefore the flux rate per building footprint area was treated as being equivalent to the flux rate of the lagoons. Furthermore, the volume sources representing the CAFO buildings require the input of a mass emission rate. Therefore, the emission rate applied to an individual volume source, Q_v, with square footprint area, A_v, was calculated as:

$$Q_V = F_o A_V \quad (2)$$

AERMOD was run with hourly meteorological data coincidental with the sampling time period to derive hourly-modeled

concentrations at the monitoring station location. These modeled values were ranked and the RHC was determined as given in Eq. (3):

$$\mathrm{RHC} = x_n + (\bar{x}_{n-1} - x_n)\ln((3n - 1/2)) \quad (3)$$

where x_n is the nth largest value, and $\bar{x}_{n-1}$ is the average of the n−1 largest values. An $n = 26$ was suggested by Cox and Tikvart (1990) as being adequate for obtaining a reliable RHC value when the total number ranked is large (>100). Likewise, the measured concentrations at the monitoring station were ranked and an RHC value obtained by application of Eq. (3). The ratio of the RHC value determined from the hydrogen sulfide sensor data, RHC_s, to the value determined from modeled concentrations, RHC_m, was used to correct the initial area flux in a way similar to that given in Eq. (1) except that the measured values were not background corrected:

$$F = F_o(\mathrm{RHC}_s/\mathrm{RHC}_m) \quad (4)$$

The resulting flux rate, F, was then scaled relative to the permitted weight of the facility to obtain an emission factor.

As explained previously, two CAFOs were added to the model as sources at Site D. This complicated the inverse modeling as a rate had to be determined for both sites during the same inverse-modeling iteration procedure. The assumption was made that the rate for the more removed CAFO was directly proportional to that of the closer CAFO and scaled relative to the ratio of their respective weights. Only the value obtained from the closest unit was used when determining an average emission factor for all sites.

2.5. Concentration values in a CAFO vicinity

After a hydrogen sulfide emission factor was determined it was applied to hypothetical CAFOs of varying sizes to estimate the influence of swine CAFOs on surrounding areas. The largest was given a live weight of 1 M kg, and was represented by 8 buildings, totaling 48 volume sources measuring 14 × 14 m, and two lagoons, represented by two area sources measuring 70 × 155 m and 155 × 1555 m. A 1 M kg CAFO represents the average of those nearest the monitoring sites evaluated in this study, which are among the largest such facilities in the Midwestern United States. A layout typical of large swine CAFOs was chosen. The model was run to determine hydrogen sulfide concentrations at each node of a uniform polar grid with 16 rings at 500-m intervals extending to 8000 m. The grid contained radial sections every 10° starting at 0°. Hourly meteorological data for an entire year (2004) was applied to the model to determine the first-highest values existing at all grid nodes for that year. Similar modeling was also performed on CAFOs with live weights of 0.25, 0.5 and 0.75 M kg, and footprint areas scaled to increase with animal weight. By comparison, for all 5048 swine operations for which the IDNR has a live weight on record, the median weight is 0.16 M kg and the upper quartile constitute operations >0.27 M kg. For this analysis, all first-highest values at a set radial distance away from the source were averaged for a number of radial distances to determine a general relationship between concentration and downwind distance.

3. Results

As shown in Fig. 2, release height with no vertical loft will affect the inverse-modeled emission flux where, in general, a greater height results in a greater emission flux. This was expected as a greater release height will cause the model to produce lower downwind receptor-level concentrations by enhanced dispersion. In order to compensate for lower concentrations, a higher emission flux is needed to match modeled with measured concentrations. This effect was more pronounced when modeling an entire CAFO as one area source than as a combination of area (lagoons) and volume (buildings) sources, and justifies the decision to model the CAFOs as a combination of area and volume sources. A similar trend is seen in Fig. 3 where both release height and vertical loft are applied to CAFOs as area sources. However, loft height had very little influence on the value of the inverse-modeled emission flux between lofts of 0–3 m. Given no physical rationale for either elevating these sources or applying an initial loft, those model options were set to zero.

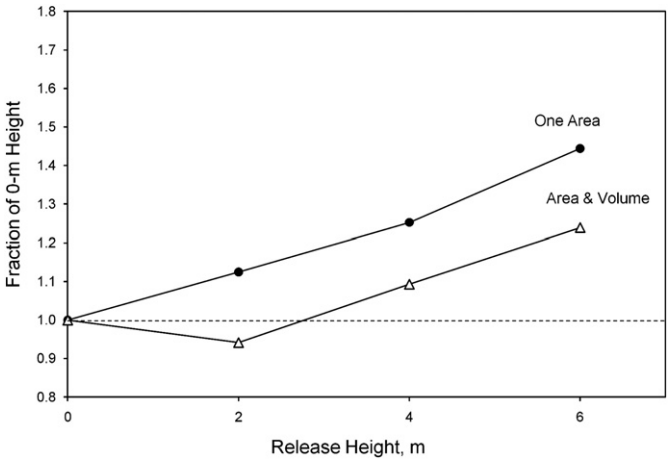

Fig. 2. The ratio of the inverse-modeled emission flux for a release height relative to the emission flux determined with a release height of 0 m for two different source configurations.

An example of the general relationship between modeled and measured concentration data sets that have the same RHC value is given in Fig. 4. This plot was developed from Site-A data sorted by wind direction for which there was only one isolated CAFO almost directly to the south of the monitoring station (Fig. 1), hence the peak in hydrogen sulfide concentrations is associated with winds from 185°. As shown in the figure, the range and spread of concentrations modeled by AERMOD closely match the measured values.

Q–Q plots for all fours sites are given in Fig. 5. These plots show good agreement between the highest measured and modeled concentrations. However, the measured-to-modeled relationship curves downward for lower concentration values. This pattern is

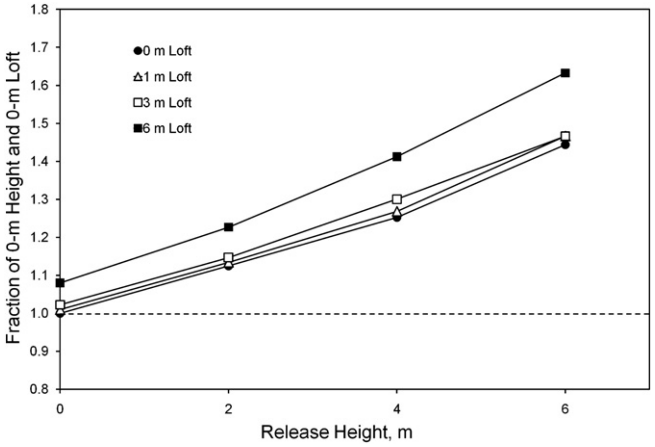

Fig. 3. The ratio of the inverse-modeled emission flux for a release height and loft relative to the emission flux determined with a release height of 0 m and loft of 0 m for a CAFO modeled as an area source.

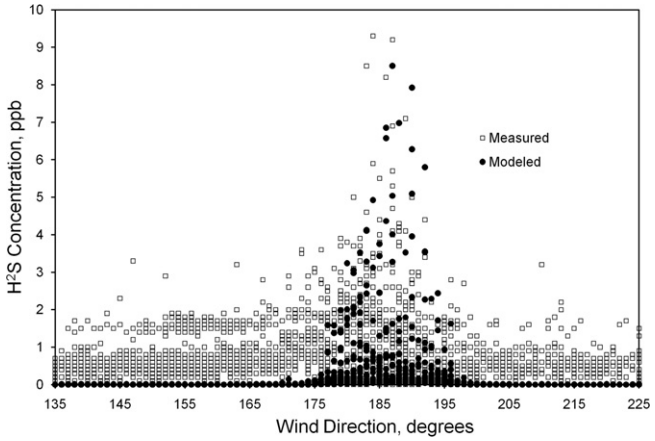

Fig. 4. Measured and modeled hydrogen sulfide concentrations relative to wind direction for Site A.

explained by the relative difference between the lowest modeled values, which go to 0, and the lowest measured values which retained levels up to 2 ppb because of instrument drift.

The emission rates and emission factors determined by inverse modeling for each site are given in Table 2. The average emission flux for lagoons was 0.57 $\mu g\ s^{-1}\ m^{-2}$. The average emission factor for lagoons was $6.06 \times 10^{-7}\ \mu g\ s^{-1}\ m^{-2}\ kg^{-1}$ with a standard deviation of $4.38 \times 10^{-7}\ \mu g\ s^{-1}\ m^{-2}\ kg^{-1}$. The average building emission rate was 111.7 $\mu g\ s^{-1}$ with a standard deviation of 61.6 $\mu g\ s^{-1}$.

The average emission factor was applied to 10 sites in northern Iowa including the 4 original sites used to determine that factor, but with additional sources applied to the model that included all CAFOs within 7 km of the monitoring site. An RHC ratio was calculated for each site to compare modeled-to-measured concentration values. The average RHC ratio was 1.24 with a range of 0.70–1.89. As a measure of uncertainty in the emission factor, the coefficient of variation (CV) of the RHC ratios was computed with a result of 29.3%. A one-sample *t*-test demonstrated that the average ratio was not significantly different from 1.00 at the 95% confidence level ($p = 0.064$). The relationship between RHC_s and RHC_m for each site is shown in Fig. 6; all values were within the EPA guideline that modeled values be within a factor of 2 of measured values (USEPA, 2005).

Fig. 7 was created to visualize the range and extent of hydrogen sulfide concentrations downwind of hypothetical swine CAFOs of varying sizes for which the average emission factor reported in Table 2 was applied. This figure gives an average of the first-highest concentrations at all nodes of a polar grid at set distances away from the source at the origin. Fig. 6 therefore demonstrates that worst-case hydrogen sulfide concentrations produced by large CAFOs (1 M kg) can extend to an average distance of 6.0 km from the CAFO before dropping below 2 ppb. Furthermore, concentrations can exceed 25 ppb within 0.5 km of a large CAFO. As when determining the emission factor, background concentrations were not included when developing Figs. 6 and 7.

4. Discussion

The inverse-modeling approach was applied to four CAFOs in north-central Iowa represented as a combination of volume and area sources. The resulting lagoon emission flux rates and associated emission factors were within a half-order magnitude of each other. Three of the four sites produced very similar emission factors. This consistency may be attributed to their similar size and type of swine (finishing), and all were in the same meteorological region with regards to model input. The facility producing the highest emission factor was also the exception in terms of swine type (sows). Given the relative similarity of the four sites, it cannot be determined how accurately the average emission factor reported

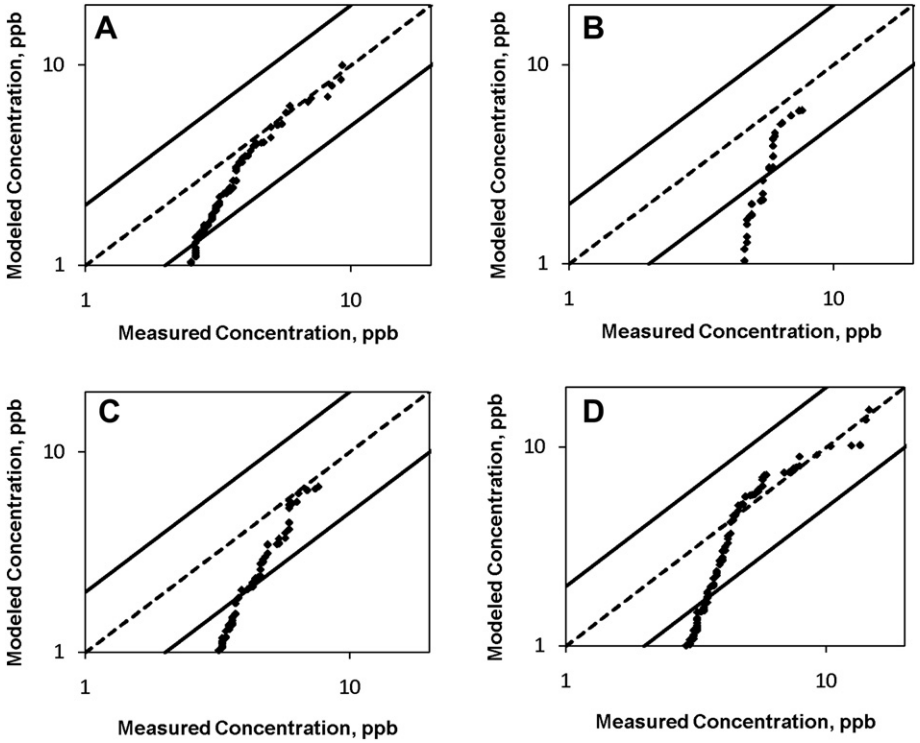

Fig. 5. Q–Q plots for all monitoring sites A–D. Solid lines represent values varying by a factor of 2 to either side of the dashed 1:1 line.

Table 2
CAFO emission rates and emission factors.

Site	Lagoon Emission Flux, μg m^{-2} s^{-1}	Building Emission Rate[a], μg s^{-1}	Lagoon Emission Factor, μg m^{-2} s^{-1} kg^{-1}
A	0.54	106.0	4.82E-07
B	0.46	89.2	4.34E-07
C	0.27	53.6	2.61E-07
D	1.01	198.0	1.25E-06
Mean	0.57	111.7	6.06E-07

[a] Rates applied to a volume source with 14 m × 14 m footprint.

here could be applied to CAFOs of sizes or regions different from those utilized in this study. However, the CAFOs analyzed represent those that are typical of any in the United States built and operated with modern technology and management practices. Regardless, this emission factor is most suitable for modeling large CAFOs located in the Midwestern United States.

An average emission factor was developed from results obtained from four monitoring sites. When this factor was applied to all CAFOs within 7 km of the monitoring station at those sites and an additional six sites, the average RHC ratio was 1.24; a value that indicates the modeled values over estimated nearby concentrations. Regardless, we do not recommend lowering the emission factor by that amount without further research to justify that approach, especially given that the average obtained was not significantly different from 1.0.

The meta-analysis of all available papers reporting hydrogen sulfide emission rates mentioned above suggests that measured emission flux rates (GM = 3.18 μg s^{-1} m^{-2} for lagoon storage units) are an order of magnitude higher than the average value determined here by inverse modeling (0.57 μg s^{-1} m^{-2} for lagoons). Therefore, a direct application of a measured flux rate to AERMOD, with the modeling options utilized for this study, could produce extremely over estimated downwind concentrations. However, the GM of storage flux rates incorporated results from one study (Blunden and Aneja, 2008) that reported seasonal lagoon flux rates which varied between 0.0 μg s^{-1} m^{-2} in the winter and 0.08 μg s^{-1} m^{-2} in the summer. This variability in measured emission rates contributes to the uncertainty when comparing the emission factor derived here with those from other published studies. For example, an analysis performed in 2002 on the variability in ammonia emission rates from swine lagoons resulted in a CV of 25% (NRC, 2002). Therefore, we cannot be certain whether the difference between the flux rate developed from this analysis and that reported in most other studies is a consequence of

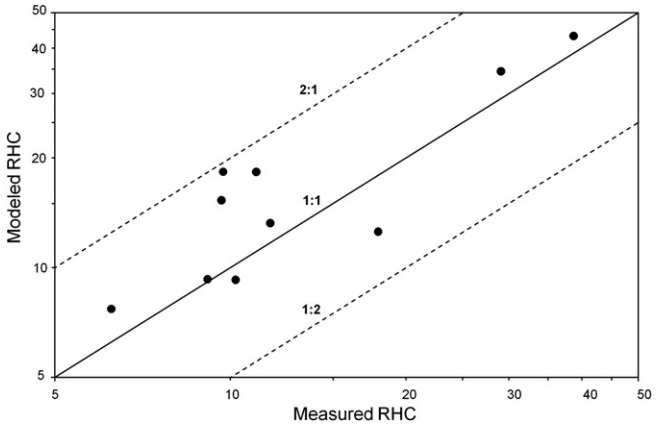

Fig. 6. Relationship between measured and corresponding modeled RHC values relative to factor-of-two ranges and the 1:1 line.

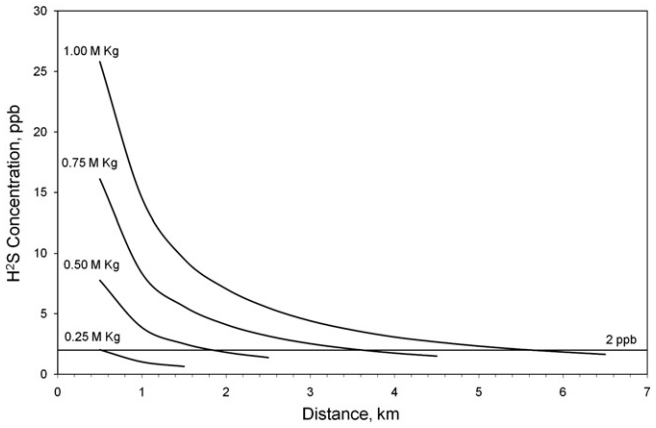

Fig. 7. Relationship between distance from CAFOs of various sizes (M-kg of pig weight) and first-highest hydrogen sulfide concentrations relative to a value (2 ppb) that represents instrument drift.

uncertainties inherent to AERMOD or that the CAFO types we analyzed represent those that emit less hydrogen sulfide than most of those reported elsewhere. These findings suggest that, if inverse-modeling is used to determine an emission rate, then that process should be performed with the same model to which the emission rate will be applied. However, a similar comparison of inverse-modeled emission rates to those measured by other means could not be found in the literature to substantiate this assertion.

If the flux rate discrepancy is due to issues inherent to the AERMOD code, then there are some possible causes. Although they evaluated the ISC model rather than AERMOD, the study by Rege and Tock (1996) determined that the standard Pasquill–Gifford dispersion coefficients under-represented the lateral dispersion of contaminants from a low-level source. If applied to the model, greater dispersion would reduce the modeled concentration at any receptor point but would likewise result in a higher emission rate by inverse modeling than for the more constrained lateral dispersion inherent to the ISC model. A similar assessment utilizing AERMOD could not be found in the literature. However, a comparable effect may be causing a reduction in the emission rate by inverse modeling with that dispersion model. There are also general aspects of the AERMOD code that contribute to statistical uncertainty in modeled concentration values. Hanna (2007) provides an excellent review of this issue that includes the relative influence of model inputs such as wind speed and mixing height on model uncertainty. In general he states that the ISC model and AERMOD can produce a relative error in concentrations below 50% but not if the sources have a time-varying release, are located in areas with large elevation differences, or are located far from the nearest meteorological station.

An application of the hydrogen sulfide emission factor found for swine CAFOs to hypothetical CAFOs of various sizes by animal weight revealed that the highest levels could exceed 30 ppb but only for the largest CAFO analyzed and for distances within 0.5 km. This evaluation suggests that setback distances applied to swine CAFOs over 0.5 km will result in hydrogen sulfide concentration levels that do not exceed 30 ppb on any given hour of the year.

5. Conclusions

With the use of inverse modeling applied to AERMOD, a mean hydrogen sulfide emission factor of 6.06 × 10^{-7} μg s^{-1} m^{-2} kg^{-1} was determined for large swine CAFOs. This factor was based on emission flux rates determined from inverse-modeling utilizing

AERMOD applied to four large swine CAFOs in north-central Iowa for which animal weights were supplied from a state permitting process and lagoon and building dimensions were determined by manual evaluation of aerial photographs. When these factors were applied to hypothetical swine CAFOs of varying animal weight, the highest concentrations modeled were less than 30 ppb beyond 0.5 km of the CAFO. The overall extent of hydrogen sulfide emitted from the largest CAFOs extended to 6.0 km from the source CAFO.

The emission rates determined by this method were lower than most other values reported from studies during which the emission rate was directly measured. Therefore the emission factor derived in this study represents a value that can be applied to determine ambient hydrogen sulfide concentrations produced by swine CAFOs in the Midwestern United States with the use of AERMOD, but should not be used to obtain an assessment of the actual emission rate. Furthermore, the emission factor was derived from the highest downwind concentrations and with non-local meteorological data applied to the model over a large range of climatological conditions. Therefore, the use of the emission factor in AERMOD applied to other sites should be performed with a large meteorological data set with the intent of estimating the highest concentrations at a receptor site.

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