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Fugitive methane emissions from an agricultural biodigester

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ABSTRACT

The use of agricultural biodigesters provides a strategy for reducing greenhouse gas (GHG) emissions while generating energy. The GHG reduction associated with a biodigester will be affected by fugitive emissions from the facility. The objective of this study was to measure fugitive methane (CH_4) emissions from a Canadian biodigester. The facility uses anaerobic digestion to produce biogas from cattle manure and other organic feedstock, which is burnt to generate electricity (1 MW capacity) and heat. An inverse dispersion technique was used to calculate emissions. Fugitive emissions were related to the operating state of the biodigester, and over four seasonal campaigns the emission rate averaged 3.2, 0.8, and 26.6 kg CH_4 hr^{-1} for normal operations, maintenance, and flaring periods, respectively. During normal operations the average fugitive emission rate corresponded to 3.1% of the CH_4 gas production rate.

1. Introduction

Agricultural biodigesters are seen as a viable means to reduce greenhouse gas (GHG) emissions while generating clean energy for on-farm consumption and to sell to power companies. Through anaerobic digestion, biodigesters reduce organic compounds in waste material to methane (CH_4) and carbon dioxide (CO_2). The subsequent capture and combustion of CH_4 can result in a reduction in GHG emissions compared to traditional waste management.

The net GHG reduction associated with a biodigester will be affected by fugitive (unintended) CH_4 emissions from the facility. Accounting for these emissions is an important part of the calculation of carbon credits for biodigestion offset protocols, such as the Organic Waste Digestion Project Protocol [1] in the United States and the Quantification protocol for the anaerobic decomposition of agricultural materials [2] in Alberta, Canada. However, measurements of the magnitude of fugitive emissions from biodigester facilities are lacking.

A draft report to U.S. Environmental Protection Agency on anaerobic digestion systems [3] recommended adoption of the 2008 California Climate Action Registry default value, where the fugitive emission rate is taken as 15% of the total CH_4 production rate, unless a lower value can be justified by supporting documentation. Methodologies developed under the Clean Development Mechanism [4] assume a fugitive emission rate of 15% for anaerobic digesters when calculating GHG reductions. The Intergovernmental Panel on Climate Change [5] assumes a default value of 10% for digesters. In a report on offset methodologies [6] it was recommended that one assume 0–5% fugitive emissions from covered anaerobic lagoons.

The objective of this project was to measure fugitive CH_4 emissions from a biodigester facility. The first portion of the

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paper describes the application of an inverse dispersion technique to measure emissions. We then present the results of four seasonal measurement campaigns at a modern biodigester in Alberta, Canada. This includes a brief discussion of how our measurements could affect GHG offset calculations.

2. IMUS Biodigester

The Integrated Manure Utilization System (IMUS) biodigester was developed by Highmark Renewables Inc. and the Alberta Research Council. Located near Vegreville, Alberta, it is one of the largest and most technologically advanced of its kind, and may become the model for additional biodigesters in Canada. It operates adjacent to a beef cattle feedlot (capacity of 36,000 head) and uses anaerobic digestion to produce biogas from the cattle manure, as well as other organic feedstock delivered to the site. The biogas is made up of about 55% CH_4, and is used to generate electricity and heat.

The electrical generating capacity of the facility is approximately 1 MW (future expansion will boost capacity to 5 MW). It consumes about 100 tonnes of manure daily, or about 20% of the feedlot output. The main steps in the IMUS operation are:

1. Dry manure is collected from feedlot pens and transported to a storage pad at the facility.
2. Manure (and other organic material) is loaded from the storage pad to a feedstock "hopper" by bucket loading tractor. Here the feedstock is mixed with warm water and recycled effluent (heated by waste heat) and pumped to one of two digester tanks.
3. Anaerobic digestion takes place in two insulated concrete tanks: 15 m in diameter and 11 m high, capped with a heavy rubberized cover. Internal temperature is maintained at

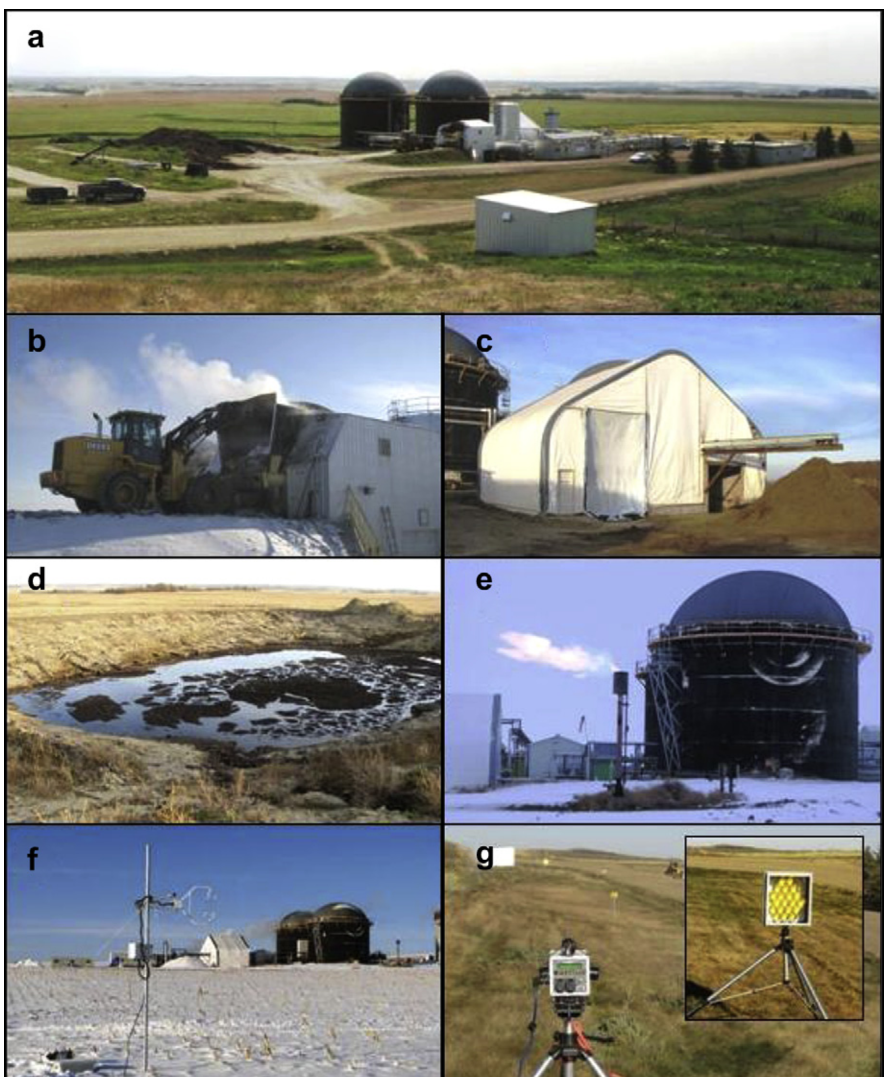

Fig. 1 – Photographs from the IMUS facility: a) distant view showing black digester tanks; b) feedstock hopper where feedstock enters the facility; c) fertilizer tent where solids are separated from the digestate; d) runoff pond; e) gas flare; f) sonic anemometer east of the facility; and g) methane laser and reflector.

55 °C to promote bacterial growth. Approximately 5% new manure is added to the tanks each day, and 5% of the digestate (slurry left after digestion) is removed.

4. Biogas is collected under the rubber cover of the digester tanks and is treated to reduce moisture and hydrogen sulfide before the gas is fed to the generator.
5. Digestate leaving the tanks flows through a separation process where solids are removed. The liquid is pumped to a lagoon at the feedlot (which also collects feedlot runoff). The lagoon water is re-used at the hopper stage, and to irrigate nearby fields. Separated solids are stockpiled and sold as fertilizer.

Photographs of the IMUS facility are shown in Fig. 1, and a map of the facility is given in Fig. 2.

3. Emission measurement technique

Measuring fugitive gas emissions from a biodigester facility is a challenge. There are many possible emission sites, such as flares, leaky pipes, ponds, open loading hoppers, etc. A measurement survey of all of these potential sites would be a time consuming task. In this study we use an alternative "inverse-dispersion" technique for measuring the totality of emissions, which promises economy and simplicity.

The inverse dispersion technique is a micrometeorological method that uses a downwind concentration measurement C to deduce the gas emission rate Q. The relationship between C and Q depends on the size and shape of the emission source, wind conditions, and the C sensor location. In principle, the relationship can be quantified by an atmospheric dispersion model. The model predicts the ratio of the downwind concentration (above the background level) to the emission rate, $(C/Q)_{sim}$, so that

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

where C_b is the background gas concentration. The advantage of the technique is the limited measurement requirements: only a single concentration sensor and basic wind information, with flexibility in the measurement location. The technique is well suited to situations where the wind can be described by simple statistical relationships (e.g., flat and homogeneous terrain), and where emission sources are spatially well defined [7].

The IMUS facility presents complications for an inverse dispersion calculation. Buildings and structures create wind

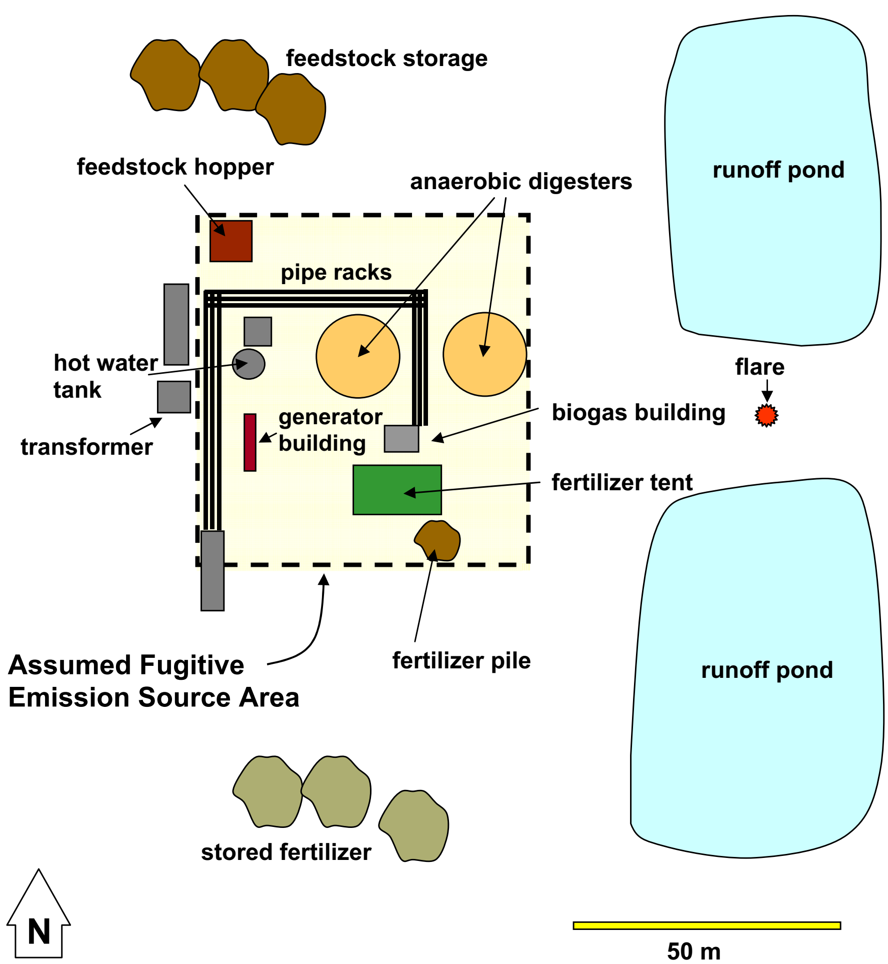

Fig. 2 – Map of the IMUS facility. The dashed square is the assumed source area for fugitive emissions (excluding the flare). The size of the runoff ponds, feedstock storage pile, and the fertilizer piles varied during the study.

complexity, and the actual location(s) of fugitive emissions is unknown. However, several studies have shown that if C is measured far enough downwind of the emission site there is insensitivity to these complications [8–10]. In these cases one can assume simplified wind conditions (i.e., an ambient wind measurement) and an imprecise designation of the source area when calculating $(C/Q)_{sim}$. How far downwind is sufficient? Flesch et al. [8] argue that one should be more than ten times the height of the largest wind obstacle h (in this study the digester tanks, h = 11 m) and roughly two times the maximum distance between potential sources x_s (in this study the distance between the feedstock hopper and the fertilizer pile exiting the separator tent, $x_s \approx 70$ m).

We use a backward Lagrangian stochastic (bLS) dispersion model [11] to calculate $(C/Q)_{sim}$ (this is described in more detail below). The "bLS technique" has been used to measure gas emissions from farms [12], fields [13], feedlots [14], ponds [15], and pastures [16]. Harper et al. [17] summarized several verification studies, conducted in a variety of settings, and concluded that with careful use the bLS technique has an expected accuracy of ± 10%.

3.1. *Concentration and wind observations*

Emissions were measured during autumn, winter, spring, and summer seasonal campaigns in 2008–2009, with each campaign lasting 6–7 days. Methane concentrations were measured with open-path lasers (GasFinder 2.0, Boreal Laser Inc., Edmonton, Canada; Spectra-1, PKL Lasers Inc., Edmonton, Canada). These sensors give the line-average concentration between the laser unit and a distant retroreflector (see picture in Fig. 1g).

Fig. 3 shows the laser lines used in winter and summer. Our main focus was on emissions within the facility boundary (identified in Fig. 2). The winter diagram in Fig. 3 illustrates the primary configuration, with laser lines positioned to give concentrations upwind and downwind of the facility. At any time the lines being used depended on the wind direction, and lines were switched manually as the wind direction changed. The primary lines were far enough from the facility to satisfy the placement criteria described above: more than 10 times the digester tank height, and about two hopper-to-fertilizer-pile distances downwind. At certain times we were concerned with emissions from secondary sources (e.g., the runoff ponds), and then we would temporarily position lasers downwind of these sources in order to estimate their emissions.

A 3-D sonic anemometer (CSAT-3, Campbell Sci., Logan, Utah) provided the wind measurements for our calculations, as described in Flesch et al. [7]. The anemometer was positioned to measure ambient winds in an open field approximately 200 m east of the facility (surface of corn stubble, snow, bare soil, or young corn plants depending on the season). The anemometer was placed at height $z_{son} = 1.85$ m (see Fig. 1f) to evaluate key wind parameters needed for the dispersion model: friction velocity u^*, Obukhov stability length L, surface roughness length z_0, and wind direction β.

3.2. *bLS Application details*

Field observations were prepared in time series of 15-min average CH_4 concentrations and wind parameters. The bLS software WindTrax (available at www.thunderbeach-scientific.com) was used to calculate $(C/Q)_{sim}$ for each 15-min period, with the total fugitive emission rate Q then given from Eqn (1). WindTrax is based on the bLS dispersion model described in Flesch et al. [7].

The IMUS facility is represented as a spatially uniform surface area source (Fig. 2) in the bLS calculation. The area includes what we believe are the potential emission sites. This treatment is undoubtedly wrong, as fugitive emissions will not occur uniformly over the designated ground area, and may occur at heights above ground. However, with C measured sufficiently far downwind of the facility, the (C/Q) ratio is assumed to be insensitive to these simplifications (as discussed in Flesch et al. [8]). We anticipate two complications. During gas flaring CH_4 can originate outside the designated source area. Flaring is not part of normal operations and

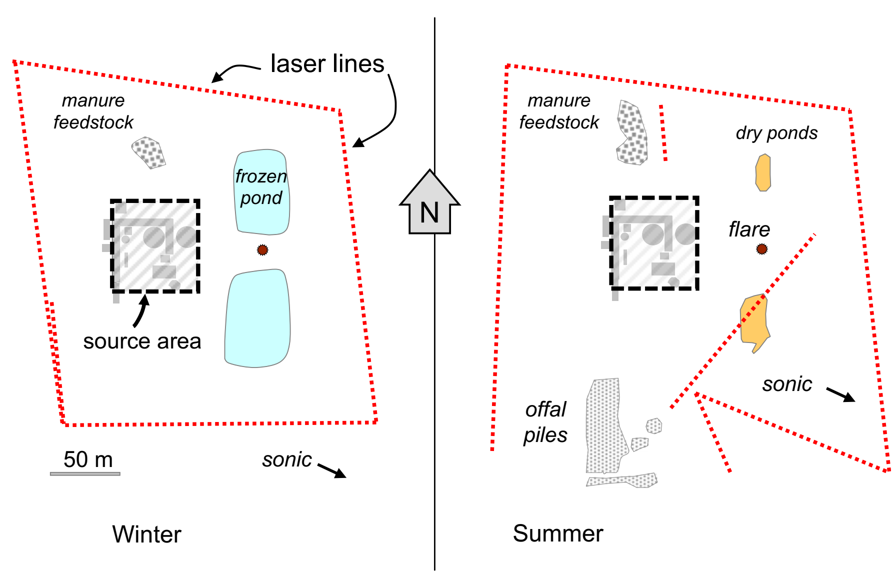

Fig. 3 – Location of the laser lines used in the winter and summer measurement campaigns.

when it occurs (known by the IMUS management) we redo our analysis and include the flare as an emission source, assuming the main facility continues to emit gas at the rate calculated during non-flaring periods. Another complication is the potential for CH_4 emissions from nearby secondary sources (e.g., runoff ponds, manure feedstock, etc.) to be falsely attributed to emissions from the IMUS facility. Our approach was to independently calculate these secondary emissions and include them in our dispersion analysis as known CH_4 sources, i.e., subtracting their calculated contribution to downwind concentration before calculating facility emissions.

Not all observation periods allow for good emission measurements. Dispersion models are known to be inaccurate in some conditions. Three criteria were used to remove periods of potential inaccuracy [8]:

1. low winds (when the friction velocity $u^* \leq 0.15$ m s^{-1});
2. strongly stable/unstable atmospheric stratification (the Obukhov length $|L| \leq 10$ m);
3. unrealistic wind parameters (surface roughness length $z_0 \geq 0.3$ m).

For some wind directions the fugitive emission plume only "glances" the path of the lasers, leading to uncertain Q estimates. To avoid these problems we:

4. removed periods when the laser measurement footprint (an output variable in WindTrax) does not cover 50% or more of the designated IMUS source area.

4. Results

4.1. *Autumn measurements*

Autumn measurements took place from 27 October to 1 November 2008. The average on-site air temperature during the period was 4.2 °C, with temperatures ranging from 16 °C to −2 °C. No significant precipitation was observed. In addition to periods of normal facility operations, there were periods of gas flaring and facility maintenance.

The potential for CH_4 emissions from secondary sources complicates our analysis. We judged the south runoff pond to be an emission source. This pond alone received runoff from the facility, and bubbles were observed rising to the surface at this pond. A laser line was positioned approximately 20 m north of the south pond and the bLS technique was used to estimate pond emissions during one afternoon having southerly winds. The calculated pond emission rate was 0.18 kg CH_4 hr^{-1} (this proves to be less than 5% of the emission rate from the IMUS facility). This pond was added as a known source in the analysis. We believe the north runoff pond was an insignificant CH_4 source. No measurable concentration rise was observed downwind of the feedstock or fertilizer piles, and we assume zero emissions from these areas.

Fig. 4 shows the time series of IMUS fugitive emissions during the autumn campaign (data gaps are due to winds not meeting the measurement criteria, or temporary laser misalignment). Measured emission rates range from near zero to just over 60 kg CH_4 hr^{-1}. There are two important features in the emission data. One is the high emission rates on 29 October. Management confirmed this was a day with biogas flaring, and some gas venting due to a malfunction of the flare igniter. The second feature is the low emission rates between 30 October and 1 November, a period of plant maintenance.

We define "normal" operations as periods without flaring or maintenance (i.e., the facility operates as intended). During normal operations the average autumn fugitive emission rate was 3.9 kg CH_4 hr^{-1} ($\sigma = 1.4$ kg hr^{-1}, $n = 109$ observations). There was substantial period-to-period variability in emissions, but no clear relationship with air temperature, windspeed, or time-of-day. During facility maintenance from 30 October to 1 November the emission rate fell to 0.5 kg CH_4 hr^{-1} ($\sigma = 0.9$ kg hr^{-1}, $n = 95$).

4.2. *Winter measurements*

Winter measurements were made from 11 to 16 December 2008. The average on-site air temperature during the period was −21 °C, with temperatures ranging from −1 °C to −38 °C. No significant precipitation was noticed during the observations. An advantage of the winter period was that secondary CH_4 sources were insignificant (e.g., the ponds were completely frozen and we observed no downwind concentration rise).

Fig. 4 shows winter emissions ranging from 0.9 to over 70 kg CH_4 hr^{-1}. The larger emission rates correspond to gas flaring. Roughly half of our good winter observations correspond to flaring periods. The emission rate averaged over all periods was 15.8 kg CH_4 hr^{-1} ($\sigma = 17.8$ kg hr^{-1}, $n = 158$ observations). This includes "normal" operations when emissions averaged 3.4 kg CH_4 hr^{-1} ($\sigma = 1.7$ kg hr^{-1}, $n = 91$), and flaring periods where emissions averaged 32.7 kg CH_4 hr^{-1} ($\sigma = 15.7$ kg hr^{-1}, $n = 67$). Period-to-period variability in emissions was poorly correlated with air temperature, windspeed, or time-of-day.

4.3. *Spring measurements*

Spring measurements were made from 11 to 17 May 2009. The average air temperature during the period was 7.2 °C, with temperatures ranging from −4 to 20 °C. There was light snow and rain during the period, but this did not impact our measurements. There were no reported flaring events, although there was a period of maintenance when the feedstock hopper was serviced.

We anticipated the south runoff pond would again be a CH_4 source, and on several days a laser was positioned to measure pond emissions. Over 33 observation periods the average pond emission rate was 0.24 kg CH_4 hr^{-1} (similar to the autumn value). A laser was also positioned downwind of the feedstock pile. From 40 observations the average feedstock emission rate was 0.20 kg CH_4 hr^{-1}. The pond and feedstock piles were included as known sources in our analysis.

Fig. 4 shows the emission time series during the spring. Emission rates range from near zero to almost 7 kg CH_4 hr^{-1}. The maximum spring emission rates are significantly smaller than the autumn or winter maxima due to the lack of flaring. Emissions were low during facility maintenance on 13–14 May. An interesting feature is the appearance of a strong diurnal emission cycle on 16 May (and suggested on 17 May),

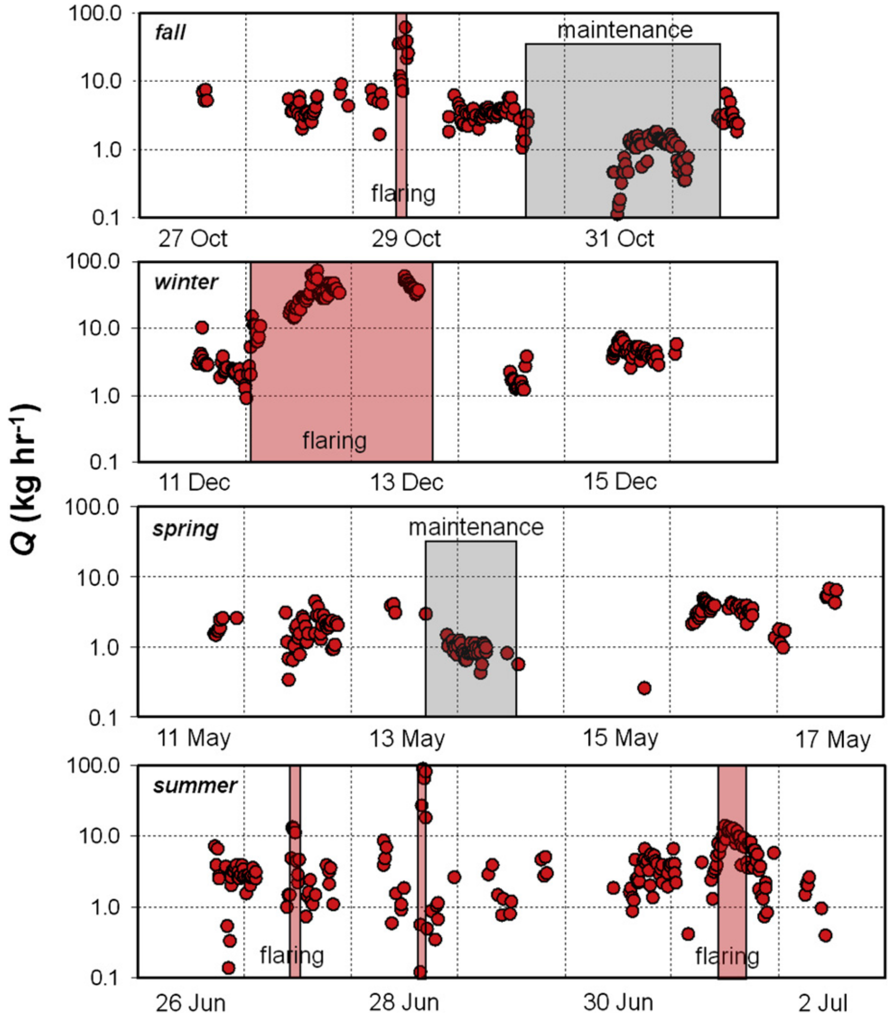

Fig. 4 – Timeseries of fugitive emissions (Q) from the biogas facility for the four seasonal campaigns. Periods of flaring and maintenance are identified. Each data point represents emissions over a 15-min observation period. Emission rates are plotted on a log scale (to de-emphasis the high rates during flaring).

with maximum emissions during the day and minimums at night. We hypothesize that this is due the daytime schedule of feedstock loading into the hopper. Why this is not seen on other days (or in other seasons) is unclear.

The overall fugitive emission rate during the spring was 2.1 kg CH_4 hr^{-1} ($\sigma = 1.3$ kg hr^{-1}, $n = 149$ observations), including "normal" operations where emissions averaged 2.5 kg CH_4 hr^{-1} ($\sigma = 1.3$ kg hr^{-1}, $n = 108$), and maintenance periods where emissions averaged 0.9 kg CH_4 hr^{-1} ($\sigma = 0.2$ kg hr^{-1}, $n = 41$). The lower spring emission rate during normal operations, compared with autumn or winter, may be related to the lower spring biogas production rate at the facility.

4.4. *Summer emissions*

Summer measurements took place from 26 June to 2 July 2009. The average on-site air temperature during the period was 14.4° C, with temperatures ranging from 3 to 21° C. The regional weather station reported 17 mm of rain during the period. There was flaring during our measurements, but no prolonged maintenance periods.

The runoff ponds were almost dry during summer, and we assume they were emission sources. A laser was positioned downwind of the feedstock pile, and from nine observations we estimate the feedstock emission rate was 0.4 kg CH_4 hr^{-1}. During the summer there was a large stockpile of "fresh" organic feedstock south of the facility (offal from a poultry rendering plant). For much of the measurement period a laser was positioned downwind of this stockpile (see Fig. 3), and from 72 observation periods we calculate an average emission rate of 5.2 kg CH_4 hr^{-1}. The feedstock and offal piles were included as known sources in the analysis.

Fig. 4 shows the time series of summer emissions. Values range from near zero to over 80 kg CH_4 hr^{-1}. The largest emissions occurred during three reported flaring events. The average summer emission rate was 4.7 kg CH_4 hr^{-1} ($\sigma = 9.3$ kg hr^{-1}, $n = 223$ observations), which includes "normal" operations where emissions average 2.9 kg CH_4 hr^{-1} ($\sigma = 2.1$ kg hr^{-1}, $n = 199$), and flaring/venting periods of 20.2 kg CH_4 hr^{-1} ($\sigma = 22.6$ kg hr^{-1}, $n = 24$). As in the previous seasons, the variability in emissions appeared uncorrelated with temperature, windspeed, or time-of-day.

5. Discussion

5.1. Emission summary

Table 1 summarizes the seasonal fugitive emission rates, categorized into periods of normal operations (when the IMUS facility operated as designed), flaring, and maintenance. Note the seasonal consistency in emissions during normal operations, with average rates ranging between 2.7 and 4.0 kg CH_4 hr^{-1}. There is also consistency during flaring and maintenance. Compared to normal operations, fugitive emissions increased by roughly a factor of 10 when flaring occurred, and fell to roughly 1/5 during maintenance.

Biodigester GHG offset protocols assume fugitive emissions are a percentage of the CH_4 production rate [4–6]. According to the IMUS management, the seasonal gas production at the facility was 131 (autumn and winter), 50 (spring), and 161 kg CH_4 hr^{-1} (summer). The low spring production was due to non-ideal feedstock material (protein content was too high for optimum decomposition). In Table 2 we express fugitive emissions as a percentage of these CH_4 production rates. During normal operations the emissions ranged from 1.7% in the summer to 5.2% in the spring. Over all four seasons the average is 3.1% of gas production. It is interesting that the seasonal range in absolute emissions (2.7–4.0 kg CH_4 hr^{-1}) is proportionally smaller than the range in percentage emissions (1.7–5.2% of gas production). This indicates that emissions were not highly dependent on the gas production rate, e.g., low spring production did not result in a proportional reduction in fugitive emissions.

Our observations show a clear pattern of reduced emissions when facility maintenance halted feedstock loading. During these periods gas was still produced and burnt to generate electricity, albeit at a slowly decreasing rate. For some initial short period of time, the only substantive difference between maintenance and normal periods was feedstock loading. Thus the drop in maintenance-period emissions is evidence that the loading hopper was the main source of fugitive CH_4 emissions. This is not surprising. The hopper is where feedstock is handled and mixed with warm water, and this process is open to the atmosphere (see Fig. 1). Further evidence for the hopper as the dominant source was the observed reduction in summer emissions. Prior to the summer the hopper was modified to create a negative pressure environment in the hopper interior (with air pulled into the biodigester). This should reduce the "escape" of gas to the atmosphere. The fact that we indeed saw lower summer emissions – lower than autumn or winter in absolute terms, and lower than all seasons in terms of gas production rates – is evidence that the hopper was the main source of fugitive emissions, and that hopper modifications were effective.

5.2. Implications for carbon credit calculations

Common biodigester GHG protocols assume fugitive emissions rates are 5–15% of the total CH_4 gas production rate [4–6]. We found that during normal operations the fugitive emissions were only 3.1% of the gas production rate. The choice of a 15% rate of emissions, as opposed to the observed 3.1%, would have a profound impact on the calculation of carbon credits for the IMUS biodigester (t CO_2e, tonnes of carbon dioxide equivalents). For a CH_4 production rate of 1030 t y^{-1} (118 kg hr^{-1}), a 15% rate of fugitive emissions would result in calculated CH_4 losses of 155 t CH_4 y^{-1}, or 3250 t CO_2e y^{-1} assuming a 100-year global warming potential for CH_4 of 21. Compared to the measured fugitive emission rate of 3.1%, this represents an overestimation of 2580 t CO_2e, and a potential financial loss of $30,960 to $64,500 per year, assuming the introductory floor and ceiling carbon prices proposed in the 'American Power Act' of $12 and $25 per t CO_2e. The financial implications of this calculation become even greater as this facility expands and increases electrical generating capacity from 1 to 5 MW.

Our measurements suggest that modifications to the feedstock hopper may have resulted in even lower fugitive emission rates. Our summer measurements, made after hopper modifications, found fugitive emissions during normal operations had dropped to 1.7% of gas production. However, because the fugitive emission rate depended dramatically on the operating state of the biodigester, the actual emission rate over a prolonged period would depend on the frequency of flaring and maintenance.

5.3. Flare efficiency

Periods of gas flaring led to large increases in fugitive emissions. Flaring occurs when gas cannot be used for electrical generation (e.g., due to generator servicing, H_2S scrubber malfunction, etc.). With limited storage capacity, the buildup of unused gas must be released and burnt in the flare. Flare efficiency (η) is a measure of the efficiency of converting CH_4 in the vented gas stream to CO_2 during burning. This can be calculated from the CH_4 content of the vented biogas stream and the CH_4 content of the flare exhaust:

Table 1 – Average fugitive emission rates (kg CH_4 hr^{-1}) from the IMUS biogas facility for the four seasonal measurement campaigns. The standard deviation of the emission rates is given in parenthesis, along with the number of 15-min observation periods (n).

Operating State	Autumn	Winter	Spring	Summer	Average
Normal	3.8 (1.4) $n = 83$	3.5 (1.7) $n = 86$	2.6 (1.3) $n = 99$	2.8 (2.1) $n = 176$	3.2
Flaring/Venting	26.6 (16.8) $n = 11$	32.7 (15.7) $n = 67$	–	20.4 (22.5) $n = 24$	26.6
Maintenance	0.7 (0.7) $n = 74$	–	0.9 (0.2) $n = 37$	–	0.8
All periods	*3.8 (7.0) $n = 197$*	*16.3 (17.9) $n = 153$*	*2.2 (1.4) $n = 136$*	*4.9 (9.8) $n = 200$*	

Table 2 – Average seasonal fugitive emission rates from the IMUS biodigester as a percentage of seasonal biogas production rates. Emissions are categorized into periods of normal operations, flaring/venting, and maintenance.

		Autumn	Winter	Spring	Summer	Average
IMUS Gas Production kg CH_4 hr^{-1}		131	131	50	161	118
Fugitive emissions	Normal	2.9%	2.7%	5.2%	1.7%	3.1%
(% of gas production)	Flaring/Venting	20%	25%	–	13%	19%
	Maintenance	0.5%	–	1.8%	–	1.2%

$$\eta = 1 - \frac{CH_4 \text{ flowrate in exhaust}}{CH_4 \text{ flowrate in vented biogas}} \quad (2)$$

For periods of flaring we estimate efficiency using the measured CH_4 flare emission rates (Q_{flare}) and the seasonal CH_4 production rates (GP_{season}) for the facility:

$$\eta = 1 - \frac{Q_{flare}}{GP_{season}} \quad (3)$$

This assumes the CH_4 flow rate in the vented gas stream equals GP_{season}. In our analysis we assume that flared CH_4 is dispersed as a non-buoyant tracer from the stack (height z = 5 m). However, if partially burnt CH_4 is warm it can rise upon emission, and lead to errors in our Q_{flare} calculation. However, Johnson et al. [18] showed that "fuel stripping" occurs in flares, with unburnt fuel ejected on the underside of the plume, countering the effect of a rising plume. In view of these complications, one must consider our estimates of η to have large uncertainty.

For all of our observations (all seasons) η ranges from 0.48 to 0.99, with an average of 0.81 ($\sigma = 0.14$, $n = 99$). This is a lower figure than found in several studies (e.g., [19]). RIRDC [20] reports that flaring efficiencies for unshrouded biogas flames (such as used here) should be in the range of $\eta = 0.90$ to 0.95. However, a field study of flares [21] found η as low as 0.62. And for diluted biogas (55% CH_4), low flare exit velocities, and high wind conditions, the data of Johnson and Kostiuk [22] suggest η could fall well below 0.90.

5.4. Secondary CH_4 sources

The objective of this study was to determine fugitive emissions from the biodigester, which we defined as the engineered facility that includes the feedstock hopper, the biodigester, effluent separator, fertilizer output tent, generator, flare, and the piping that ties these components together. This is the area within the 60 × 60 m square identified in Fig. 2 (and including the flare). We were not directly interested in CH_4 sources outside this area, which included the feedstock piles, the runoff ponds, and the offal storage area. However, we did estimate emissions from these "secondary" sources in order to more accurately calculate emissions from the IMUS facility. These secondary emission rates may also be of interest when considering the total GHG implications of a biodigester system.

The secondary emissions were generally small compared to emissions from the IMUS facility (Fig. 5). The pond and feedstock pile each emit less than 10% of that from the IMUS facility during normal operations. The offal storage pile in summer was an exception in being a large source of CH_4. Emissions from the offal were almost twice that from the IMUS facility: 5.1 kg CH_4 hr^{-1} versus 2.8 kg CH_4 hr^{-1} (normal operations). This illustrates the potential for on-site storage of fresh organic material to be a large emission source. But this is unlikely to represent an increased GHG emission component of biodigester systems, as traditional waste management will have a similar period of waste storage prior to land application. For this reason emissions during waste storage are generally excluded in carbon credit offset protocols for biodigesters [1,2]. It is possible that the duration of waste storage is actually reduced in a biodigester system – as feedstock energetic value decreases over time, there is incentive to minimize storage.

The IMUS biodigester is integrated into a beef feedlot production system. As a part of this system, feedlot runoff is collected in a retention lagoon and is then pumped and mixed with manure at the IMUS hopper. Effluent from the digester is returned to the feedlot lagoon following solid/liquid separation. In this study we did not measure the emission contribution associated with effluent return to the lagoon. Such a transfer may alter the CH_4 emissions compared to an exclusive runoff lagoon. Adding biodigester effluent may increase lagoon emissions by introducing additional organic matter to the lagoon. However this may be offset by the reduction in organic matter in the feedlot runoff as a result of the removal of manure from the feedlot for biodigester feedstock. Additionally, Harper et al. [17] found a 50% decrease in CH_4 emissions from swine waste effluent after processing in anaerobic digester. So the returned effluent may be a weaker emission source than the original lagoon water.

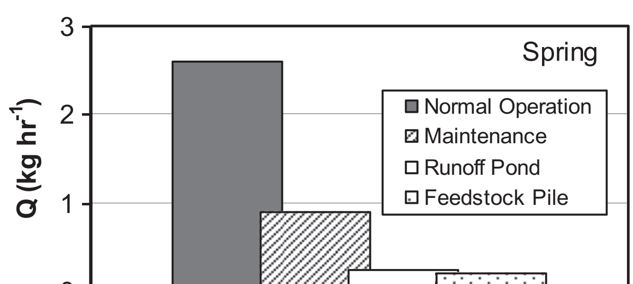

Fig. 5 – Spring fugitive emission rates (Q) from the biogas facility (during "Normal Operation" and "Maintenance") and from secondary sources (the "Runoff Pond" and the "Feedstock Pile").

6. Conclusions

Fugitive emissions of CH_4 from the IMUS biodigester facility were related to its operating state. Over four seasonal measurement campaigns the average emission rates were 0.8, 26.6, and 3.2 kg CH_4 hr^{-1} for maintenance, flaring, and normal operating periods, respectively. During normal operations the feedstock hopper appears to be the main source of emissions, although when flaring occurs the flare is an order-of-magnitude larger source.

Fugitive emissions were relatively small when expressed as a percentage of biogas production. During normal operations the fugitive emission rate was 3.1% of the CH_4 gas production rate. This is much lower than the default values of 5–15% assumed in GHG offset protocols [4–6], and this has large financial implications when calculating GHG offsets and carbon credits. However, the emission rate over any prolonged period will ultimately depend on the frequency of flaring and maintenance.

The bLS inverse dispersion technique proved well-suited to our study. With modest equipment and labor resources (one person) we were able to quickly setup and monitor emissions. Monitoring occurred continuously, with equipment left unattended except to swap batteries, move lasers to address wind changes, or focus on different sources (laser reconfiguration took only a few minutes). And because of the modest resource requirements it was possible to monitor emissions for prolonged periods, and capture the characteristics of a highly variable source.

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