Greenhouse Gas Emissions from Land Application of Manure

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Abstract

Greenhouse gas (GHG) emissions from agricultural activities such as land application of livestock manure cannot be ignored when assessing overall emissions from anthropogenic sources. The magnitude of these emissions will be influenced by management practices such as manure placement during land application. The objective of this work was to compare GHG fluxes resulting from the surface and subsurface application of liquid and solid manure. For this comparison, all measurements were made 24 hours after application. The results showed that subsurface application significantly increased carbon dioxide equivalent (CO_2-e) fluxes for both solid and liquid manure. The overall CO_2-e fluxes from the injected treatments were 3.2 times higher than CO_2-e fluxes from the surface applied plots, mainly due to a pronounced increase in N_2O fluxes which was likely caused by increased denitrification rates. The CO_2-e fluxes from the liquid manure applications were also higher than the CO_2-e fluxes from the solid manure applications, probably due to higher levels of ammonium available for nitrification and subsequent denitrification. For this particular study, the measured specific fluxes (total flux per kg N applied) remained relatively constant with application rate, indicating that GHG emissions from manure applications were approximately proportional to the amount of land applied manure.

Introduction

During the last 150 to 200 years, human activity has increased the atmosphere's content of carbon dioxide (CO_2) by 30%, of methane (CH_4) by 145%, and of nitrous oxide (N_2O) by 15% based on International Panel on Climate Change (IPCC) data (Greatorex, 2000). These gases contribute to the "greenhouse effect" of the atmosphere which is believed to play a major role in the global warming of Earth's climate (IPCC, 2007). The Kyoto Protocol, a multi-national agreement, was put in place at the end of the twentieth century with the goal of significantly reducing anthropogenic emissions of these greenhouse gases. Canada's commitment under the Kyoto Protocol was to reduce net greenhouse gas emissions by 6% relative to the 1990 levels of 608 Mt by 2008-2012 (Kebreab et al., 2006). This commitment has resulted in widespread research on emission reducing strategies and technologies that cover all aspects of society including manufacturing, transportation, industry, and agriculture. More recently, the Conference of the Parties (COP) meeting of the IPCC held in Copenhagen, Denmark has further emphasized the urgent need to limit anthropogenic GHG emissions, including those from agricultural sources.

It has been estimated that agricultural activities contribute to 20% of anthropogenic greenhouse gas (GHG) emissions (Lovanh et al., 2008) and more specifically to 60 to 80% of total N_2O emissions (Jarecki et al., 2008). Agricultural emissions include CO_2 from burning fossil fuels, CH_4 from enteric fermentation in ruminant animals, CO_2 and CH_4 from storage of livestock manure, and N_2O from fertilizer and manure application to land. The land application of manure and fertilizers contributes to 50% of Canadian agricultural emissions (Kebreab et al., 2006) and it is the main source of agricultural N_2O because fertilizer and manure applications significantly increase microbial production of N_2O from soils (Davidson, 2009). Nitrous oxide's high global warming potential (310 times that of CO_2 (UNFCCC, 2004)) makes it a large contributor to GHG budgets.

The majority of research to date on GHG emissions resulting from the land application of manure has focused on liquid manure, even though more than two thirds of land applied with manure in Canada receives solid or composted manure (Statistics Canada, 2006). Thus, there exists a distinct need for research on emissions from solid manure application. Another important element to consider is the impact of manure management systems, such as surface broadcasting or injection of manure, on GHG emissions. The injection or incorporation of manure into the soil has the potential to increase these GHG emissions from manure spreading, which is an important consideration when attempting to assess agriculture's contribution to a region's total GHG emissions. With new plans and strategies being put in place to reduce global GHG emissions, it is important to carefully analyze emissions that result from new technologies or practices. There are very few comprehensive studies that have addressed the effect of subsurface application on GHG emissions, particularly for solid manure. Therefore, the objective of this research was to compare GHG emissions between liquid and solid manure and surface and subsurface application.

Literature Review

There have been numerous laboratory, plot, and field scale studies examining the effects of application variables on GHG emissions resulting from the land application of manure. This review focuses on emissions of CO_2 and N_2O since several studies (Chadwick et al., 2000; Chadwick and Pain, 1997; Sherlock et al., 2002; Dittert et al., 2005) noted that methane emissions following manure spreading are typically short-lived (less than 24 hrs) because the majority of CH_4 flux from manure amended soils comes from the volatilization of CH_4 compounds in the manure. Additional information on the effects of manure vs. fertilizer application and application timing on GHG emissions, as well as diurnal variations, duration of emissions, time to peak, and correlation of GHG emissions with soil properties can be found in the literature and in the comprehensive overview completed by Agnew (2010).

Comparison of Emissions from Different Manure Types

Several studies have noted that GHG emissions from liquid manure applications differ from emissions from solid manure applications. In a laboratory scale study, it was found that applications of liquid manure resulted in immediate and intense denitrification while those of solid manure resulted in less intense but prolonged denitrification (Loro et al., 1997). Tenuta et al. (2000) also reported that solid manure applied to the soil provided a "more sustained release" of available C as the bedding material decomposed, promoting denitrification enzyme activity for longer periods. The majority of solid manure C and N is in the form of organic matter, but anaerobic conditions during storage of liquid manure results in high levels of easily decomposable C species and mineral N, resulting in higher emissions from liquid manure applications in the short term (Rochette et al., 2008). Solid manure application adds recalcitrant forms of C and N to the soil, suggesting that although their potential to stimulate nitrification and denitrification may be less than that of liquid manures, the stimulatory effect of solid manures may extend over longer periods (Lemke et al., 2009). Gregorich et al. (2005) also noted that short measurement periods (i.e.: one year) following application of solid manure may not fully account for all the total manure-induced emission of N_2O. Indeed, Mogge et al. (1999 in: Rochette et al., 2008) reported emission from soils with a long history (30 yr) of repeated application of solid manure were higher than emissions from liquid manure and concluded that nitrification was the major contributor to N_2O production.

Greenhouse gas emissions also vary with animal type due to different diets, feed conversions, and management of the manure (Chadwick et al., 2000). Chadwick et al. (2000) noted immediate emissions of N_2O from beef manure and pig slurry, likely due to rapid nitrification of NH_4 or denitrification of NO_3 already in manure (beef manure) and the high C content and moisture content (pig slurry). The N_2O emissions from dairy slurry, layer manure and pig manure were not significantly different from untreated control plots (Chadwick et al., 2000). Watanabe et al.

(1997) noted that CO_2 and N_2O-N fluxes were higher from swine excrement applications than from cattle excrement applications, but N contents were not normalized.

Manure treatments such as anaerobic digestion, slurry separation, slurry aeration, and straw covered manure storages may also affect GHG emissions after land application. For example, anaerobic digestion alters the availability of C in the substrate, affecting the potential N_2O production (Petersen, 1999). Amon et al. (2005) monitored N_2O emissions after application of dairy cattle slurry with several treatments (control, slurry separation, anaerobic digestion, slurry aeration and straw covered storage). The proportion of N_2O emissions from land application ("total" emissions are from storage and spreading) was highest for separated slurry followed by straw covered, untreated, aerated, and digested slurry (Amon et al., 2005).

Comparison of Emissions from Different Application Methods

The greater contact of injected slurry with soil can induce favourable conditions for N_2O formation because of restricted aeration in the vicinity of the injected manure (Wulf et al., 2002; Flessa and Beese, 2000). Many researchers have hypothesized that injection or sub-surface application of manure N will promote denitrification (Comfort et al., 1988; Wulf et al. 2002). However, Wulf et al. (2002) noted that literature results on the effect of injection and incorporation on GHG after manure application are contradictory as some show an increase in emissions due to injection and others show no differences. For example, in a laboratory scale study, Dendooven et al. (1998) found no difference in CO_2 and N_2O production within 15 days of injecting pig slurry versus surface application. Flessa and Beese (2000), however, did note significantly higher N_2O and CH_4 emissions from an injection treatment compared to a surface treatment, but the CO_2 flux was not affected by application method (Flessa and Beese, 2000). Lovanh et al. (2008) and Sistani et al. (2008) showed that surface application of swine slurry produced higher, but not significantly higher, fluxes of N_2O compared to row injection and aerway injection (surface application over artificially perforated or aerated soil). Weslien et al. (1998) reported slightly, but not significantly, higher N_2O emissions after banding+harrowing compared with trenching, shallow injection and band-spreading. Harrowing was thought to spread around the manure under the soil, creating more hot spots and partially anaerobic regions while injection resulted in complete denitrification, producing N_2 instead of N_2O (Weslien et al., 1998). However, Perala et al. (2006) showed that slurry injection produced higher cumulative N_2O emissions than slurry incorporation, but the difference was not significant. Wulf et al., (2002) compared GHG emissions from splash plate, trail hose, trail shoe and injection methods. Results indicated that trail hose application with immediate incorporation resulted in the lowest GHG emissions on arable land while trail shoe application had the smallest risk of high GHG emissions on grassland. Wulf et al. (2002) stated that, in terms of CO_2 equivalents, the increase in N_2O emissions after injection might be as high as the reduction of NH_3 losses or, as in the case of injection on grassland, might even increase overall GHG emissions. The flux patterns for different application techniques varied, but cumulative emissions showed injection increased overall emissions (Wulf et al., 2002). The authors attributed this result to the promotion of anaerobic sites and diffusion constraints that occur with the injection technique.

Effect of Application Rate on GHG Emissions

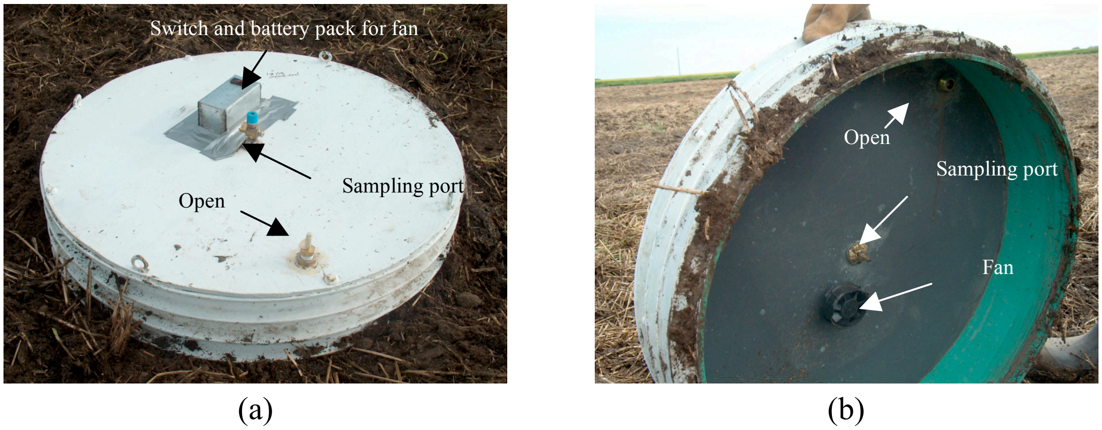
Nitrous oxide fluxes increased linearly with fertilizer application rate in the information compiled by Gregorich et al. (2005). Generally, for manure application, GHG emissions in the short term increase with application rate for both solid (Chang et al., 1998) and liquid (Paul et al., 1993) manure since any N not used by the plants is available for denitrification. However, other studies that measured cumulative N losses over longer periods found that rate of manure application had little effect on overall N_2O emissions (Hansen et al., 1993). Lessard et al. (1996) noted that application rate did not affect GHG flux but did affect NH_4-N and NO_3-N contents in soil profile. In Rochette et al. (2000b), the addition of the second 60 Mg/ha resulted in a greater incremental increase than the first 60 Mg/ha, suggesting a non-linear relationship between application rate and N_2O flux. Van Groenigen et al. (2004) also concluded that N_2O emissions were not linearly related to N application rates and the effect of application rate varied with type of fertilizer.

In terms of C fluxes from different rates of manure application, Rochette et al. (2000a) reported a linear response of C oxidation to the amount of liquid manure added, suggesting that there were no physical or chemical limitations to increased microbial activity with increased amount of liquid manure added. In contrast, Gregorich et al. (1998) reported that the CO_2 flux increased proportionately less for the second increment of manure added than for the first increment.

Materials and Methods

Description and Operation of Static Chamber

Since this study was concerned with comparisons among multiple treatments, the static (closed) chamber technique was selected to collect GHG flux data. Static chambers have been widely used in the past for similar research (Chadwick et al., 2000; Ginting et al., 2003; Lessard et al., 1996; Lovanh et al., 2008; Petersen, 1999; Rochette et al., 2000a, 2000b; Van Groenigen et al., 2004; Wulf et al., 2002; etc.). Two identical static chambers were constructed for assessing the GHG emissions from surfaces applied with manure and are depicted in Figure 1. The chambers were 0.60 m in diameter (0.2826 m^2 surface area) and 0.15 m high, made of corrugated PVC tube. The chambers were capped with 6.35 mm thick PVC plates. Small, battery powered (9 volt) computer fans were wired inside the chamber to facilitate good mixing of the sample gases. The cap also included a sampling port and septum and an open port (30 mm high, 10 mm diameter) for pressure equalization and depth measurements. The exterior of the chambers were painted white to minimize reflective heating inside the chamber during deployment. The internal headspace varied, depending on how deeply the chamber was inserted in the soil, but the average headspace was 0.040 m^3.



(a) (b)

Figure 1. Static chambers for greenhouse gas emission measurement a) exterior view and b) interior view.

Samples were collected for GHG flux determination approximately 24 hrs after application of manure. All GHG samples were collected in the morning between 0900 and 1200 to minimize the effects of diurnal variations. Samples were drawn from the sampling port at even intervals (5, 10, 15 minutes) after chamber deployment. Upwind ambient samples were collected periodically during the sampling session to represent the time = 0 sample. Four depth measurements were collected for each flux measurement for an accurate chamber volume calculation.

Gas concentrations were assessed using gas chromatography. The CO_2 concentration analysis utilized a Varian Micro GC CP-2003 with a Poraplot U column and was identified using a thermal conductivity detector (TCD) with helium carrier gas. The N_2O and CH_4 concentration analysis utilized a Varian CP-3800 gas chromatograph and were detected with one of 2 electron capture detectors (ECD's) with Poraplot Q coated plot fused silica columns.

The enclosure time of 15 minutes was considered to be short, minimizing the effect of the chamber on the concentration gradient between the soil surface and the headspace air. This eliminated the need for the complex model proposed by Hutchinson and Mosier (1981).

Therefore, the fluxes were calculated using Equation 1:

$$F = \rho \frac{V}{A}\frac{\Delta C}{\Delta t} \qquad (1)$$

where
F = surface gas flux (mg $m^{-2}s^{-1}$),
ρ = density of gas (kg/m^3),
V = volume of chamber (m^3),
A = area of chamber (m^2), and
$\Delta C/\Delta t$ = rate of change of gas concentration (ppm/s).

Other forms of this equation that account for the temperature and partial pressure of water vapour in the chamber have been used (Rochette and Hutchinson, 2005 in: Rochette et al. 2008; Ginting et al., 2003; Hutchinson and Livingston, 1993). The changes in temperature and partial pressure inside the chamber were assumed negligible due to the short enclosure time used in this study.

The rate of increase of gas concentration in the chamber headspace was evaluated on a case by case basis using linear or quadratic regression. If the regression model was insignificant ($P>0.15$), the flux was assumed to be zero. If the regression model was significant, it was differentiated with respect to time and evaluated at time = 0 to determine the rate of change of gas concentration at the instant the chamber was deployed.

Experimental Design for Data Collection

Greenhouse gas emissions from surface and subsurface application of liquid and solid manure were measured on a plot scale rather than full-scale field testing to control variables such as application rate and application method and type of manure. Liquid swine and dairy manure and solid swine, poultry and feedlot manure were surface applied and injected at three application rates with 3 repetitions using a randomized block design at three sites in Central Saskatchewan. A summary of the sites are shown in Table 1. Application rates were selected based on recommended agronomic rates defined by the nitrogen content of the manure. A recommended “one-year” application rate (1X) would be applied annually to supply enough nitrogen for one year of crop growth. Two and three year application rates (2X and 3X), where larger quantities of manure are applied every two or three years, are common in the Prairies and were also used in this study. Greenhouse gases from control plots (0X) where no manure was applied were also included. Table 2 summarizes the application rates used in the study while Table 3 includes selected manure properties. For the poultry manure, a prototype solid manure injection system was used to apply the manure. All other manure applications were simulated by hand. A detailed description of the solid injection prototype can be found in Laguë et al. (2006) and protocols for the simulated applications can be found in Agnew (2010).

Table 1. Summary of Odour Emission Experiments Conducted in Summer, 2007.

Location	Type of manure	Method of application
U of S Feedlot	Liquid dairy and solid feedlot	Simulated application
Saskatoon area	Liquid swine and solid swine	Simulated application
Humboldt area	Solid poultry	Solid injection prototype

* All experiments were randomized block designs and included surface and subsurface applications at four application rates (0X—control, 1X, 2X, and 3X).

Table 2. Liquid and Solid Manure Application Rates.

Rate	Solid (Mg/ha)	Liquid (m^3/ha)
1X	20.2	56.1
2X	40.4	84.2
3X	60.6	112.2

Table 3. Manure Chemical Properties.

	Total Solids (%)	Ammonia as N (kg/m^3)	Total N
Solid feedlot	38.2	n/a	8.3 kg/Mg
Liquid dairy	6.9	0.60(1)	2.5 kg/m^3
Solid swine	43.2	n/a	7.0 kg/Mg
Liquid swine	2.8	2.88	3.24 kg/m^3
Solid poultry	46.4	3.25	17.3 kg/Mg

(1) Liquid dairy manure was "generated" by taking fresh semi-solid manure directly from alley of barn and diluted with equal parts of water, and applied within 12 hours of mixing, resulting in little time for microbial activity and generation of NH_4-N.

Soil Properties

All plots were located in wheat, flax or barley stubble and had no commercial fertilizer application after the crop was harvested the previous year. Soil samples were collected from each site on each day of emission sampling to provide data on basic soil characteristics. Samples were collected using a 10 cm soil probe from four locations immediately surrounding the plot site. Sub-samples were used for moisture content analysis by oven drying according to ASTM standards and the remaining sample was dried and frozen for nutrient and particle size analysis. A summary of the soil properties for the locations used in this study is presented in Table 4.

Table 4. Soil Properties for Data Collection Sites.

Site Location	Texture Class	Moisture Content Range (% d.b.)	Bulk Density (g/cm^3)	Nitrogen Content (% LECO-N)	Organic Carbon Content (%)	Organic Matter Content (%)
U of S Feedlot	Sandy loam	15.7 – 34.4	1.49	0.30	3.2	5.5
Saskatoon area	Loam	19.8 – 23.8	1.47	0.34	3.4	5.8
Humboldt	Clay loam	26.1 – 31.9	1.31	0.44	4.4	7.5

Statistical Analysis

The Kruskal Wallis non-parametric test was used to determine significance of treatment effects on N_2O, CO_2 and CO_2-e fluxes because the data were not normally distributed. Treatments were considered to have a significant effect on the flux when the P value was less than 0.05 to provide a high level of confidence (95%). All statistical analyses were performed using Minitab software (version 15).

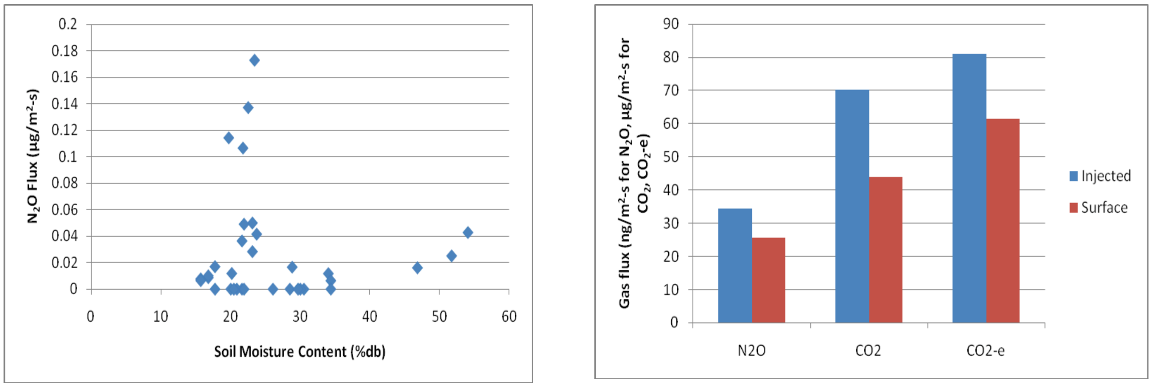
Results

Most of the plots produced statistically significant N_2O and CO_2 fluxes, but very few CH_4 fluxes had significant regressions for the rate of increase in gas concentration in the headspace. Furthermore, the significant CH_4 fluxes were very low and varied between positive (emission of CH_4) and negative (uptake or CH_4 oxidation) values. Additionally, there were no significant treatment effects on CH_4 flux, so their results are not discussed here. The complete results can be found in Agnew (2010). These low CH_4 fluxes measured 24 hrs after application were consistent with those reported by Chadwick et al. (2000) and others who stated that the majority of CH_4 emissions from manure spreading occur within 12 hrs of application. Therefore, the carbon dioxide equivalent (CO_2-e) calculation excluded the CH_4 fluxes and accounted for N_2O (with a global warming potential of 310) and CO_2 only.

Control Fluxes

Microbial activity in soil is highly dependent on soil moisture content, so it follows that GHG emissions may be dependent on soil moisture content. The scatterplots describing the relationship between soil moisture content and N_2O and CO_2-e emissions for the plot data are presented in Figure 2a. Based on these data, there is no apparent trend between moisture content and CO_2-e fluxes measured 24 hours after application, but the maximum fluxes appear to be confined to a small range of moisture contents (20-25% d.b.), which corresponded to a water filled pore space range of 44 to 51% for these soils (average wet bulk density 1.42 Mg/m^3).

Since the fluxes may have also been influenced by the amount of soil disturbance due to injection, the fluxes were compared between disturbed and undisturbed control plots. While emissions from the disturbed control plots tended to be higher than emissions from the undisturbed control plots, the difference was not significant for any of the gases measured ($P = 0.243$ for N_2O, 0.052 for CO_2, and 0.131 for CO_2-e, Fig. 2b). The low P value for CO_2 indicates that soil disturbance tended to increase CO_2 flux, likely due to increased soil respiration due to enhanced aeration and aggregate disruption that exposes soil organic matter to microbial decomposition.



(a) (b)

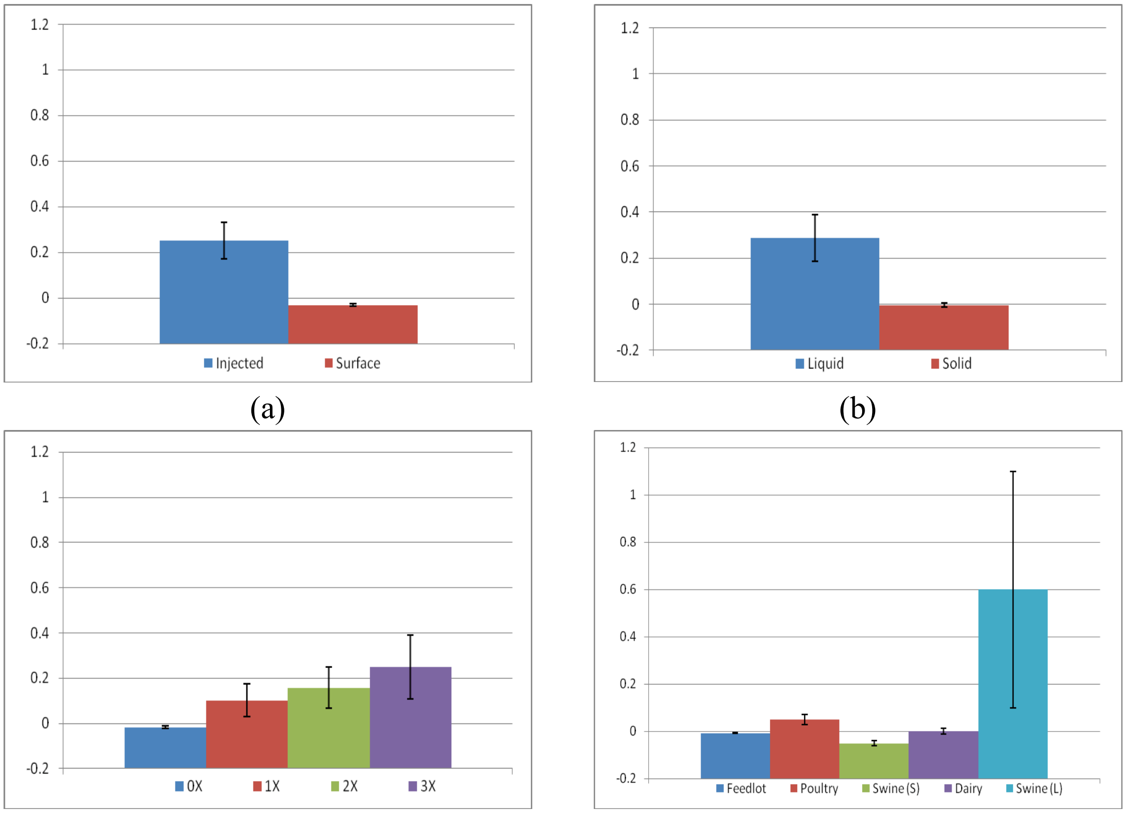
Figure 2. Control fluxes (a) scatter plot of N_2O emissions ($\mu g/m^2$-s) versus oven dry basis soil moisture content (%), (b) effect of soil disturbance on background fluxes.

For one randomized block experiment (liquid dairy), additional disturbed plots were applied with a 1X (56.1 m^3/ha) rate of water to investigate whether the application of liquid promoted the generation of significant GHG's. The emissions from these disturbed control plots were higher but not significantly different from the other control plots in that block ($P = 0.146$, data not shown), suggesting that the moisture applied when injecting manure did not affect the microbial population enough to alter the GHG emissions occurring one day after manure application.

The background N_2O fluxes varied significantly among locations (U of S Feedlot < Humboldt Area < Saskatoon Area, $P = 0.003$). This made it necessary to calculate a "manure induced" N_2O flux to account for the varying N_2O emitted from bare soil when analyzing the treatment effects on the N_2O flux. Since the background N_2O fluxes varied only with location, the data were pooled by location to determine overall background N_2O flux.

N_2O Fluxes

Because the background N_2O fluxes varied by location, the “manure induced” N_2O flux was calculated by subtracting the mean background flux of each location from the total fluxes obtained at that location. Analysis of these manure induced fluxes showed that injection significantly increased the N_2O from the manure (P=0.000) and the manure induced N_2O fluxes were higher from liquid manure applications than solid manure applications (P=0.025). Of note is that the mean manure induced N_2O flux from the surface applications and solid manure showed N_2O uptake by the soil while injected applications and liquid manure showed N_2O emission (Fig. 3). The solid feedlot, solid swine and liquid dairy applications had negative manure induced N_2O fluxes while the solid poultry and liquid swine had positive manure induced N_2O fluxes. The application rate did not affect manure induced N_2O fluxes (P=0.243).



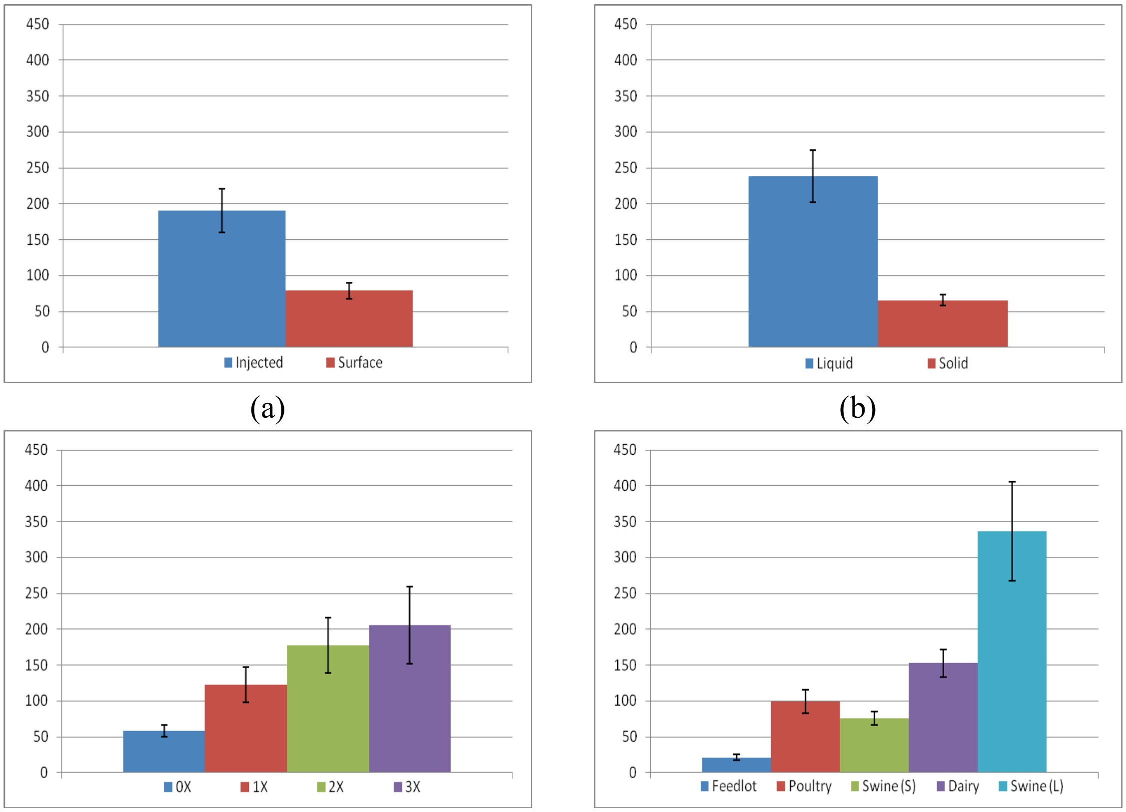
(a) (b)

(c) (d)

Figure 3. Graphical summary of manure induced N_2O fluxes. Vertical axes represent N_2O flux ($\mu g/m^2$-s). Solid bars and line error bars correspond to average values and standard errors of the means respectively. (a) Effect of application method, (b) effect of manure type, (c) effect of application rate, (d) effect of manure species.

CO_2 Fluxes

Since background CO_2 fluxes did not vary by location, the statistical analysis was performed on the overall pooled data. The overall analysis showed that injection significantly increased CO_2 flux ($P=0.003$, Fig. 4a) and fluxes from liquid manure were higher than from solid manure ($P=0.000$, Fig. 4b). The CO_2 flux increased with application rate. Although the differences among the 1X, 2X and 3X application rates were not significant, CO_2 fluxes from the manured plots were significantly higher than from the control plots ($P=0.021$, Fig. 4c). The poultry manure plots generated the highest fluxes of the solid manures while the liquid swine plots generated the highest CO_2 fluxes of the liquid manures (Fig. 4d).



(a) (b)

(c) (d)

Figure 4. Graphical summary of absolute CO_2 fluxes. Vertical axes represent absolute CO_2 flux ($\mu g/m^2$-s). Solid bars and line error bars correspond to average values and standard errors of the means respectively. (a) Effect of application method, (b) effect of manure type, (c) effect of application rate, (d) effect of manure species.

CO_2-e Fluxes

To account for the high global warming potential of N_2O (310 times that of CO_2), the carbon dioxide equivalent (CO_2-e) values were calculated. Since the CO_2 fluxes were more than double the carbon dioxide equivalent N_2O fluxes (overall mean CO_2 flux = 137.5 µg/m^2-s and overall mean carbon dioxide equivalent N_2O flux = 51.4 µg/m^2-s), the CO_2-e flux trends and treatment significances were very similar to the CO_2 flux trends (Fig. 4). The treatment effects on CO_2-e fluxes are reported in Agnew (2010).

When the effect of injection on CO_2-e fluxes was analyzed for each manure species, injection significantly increased CO_2-e fluxes from liquid swine and solid poultry manure (P=0.002, 0.017 respectively). Injection tended to increase CO_2-e fluxes from solid swine manure (P=0.074) while injection had no significant effect on feedlot and liquid dairy manure (P=0.621 and 0.312, respectively). A summary of the CO_2-e flux values for the overall data, solid manure and liquid manure is presented in Table 5.

Table 5. Summary of Absolute CO_2-e Flux Data (µg/m^2-s).

		N	P value	Mean	Std Err
Overall	Injected	61	0.001	279.6	54.4
	Surface	55		86.1	11.6
	Liquid	49	0.000	342.7	64.9
	Solid	67		74.7	9.1
	0X	32	0.054	71.8	10.4
	1X	29		163.4	45.9
	2X	29		240.2	63.3
	3X	26		299.6	99.0
Solid	Feedlot	23	0.000	23.0	4.6
	Poultry	21		109.7	20.9
	Swine (S)	23		94.3	12.3
	Injected	34	0.030	99.6	15.5
	Surface	33		49.0	7.2
	0X	17	0.578	49.4	6.2
	1X	17		60.7	15.3
	2X	17		104.9	24.9
	3X	16		84.2	19.7
Liquid	Dairy	26	0.034	157.6	20.6
	Swine (L)	23		552.0	124.0
	Injected	27	0.009	506.0	107.0
	Surface	22		22.3	22.3
	0X	15	0.002	97.3	19.4
	1X	12		308.9	95.9
	2X	12		432.0	133.0
	3X	10		644.0	120.0

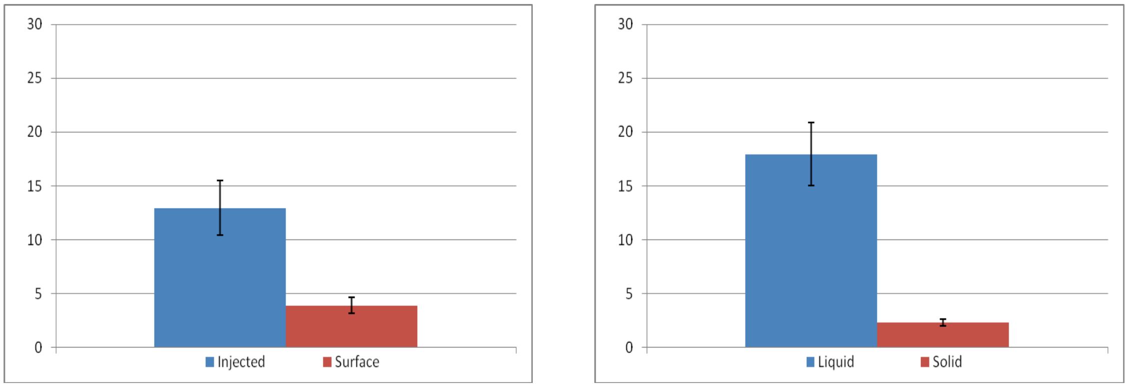
Specific Fluxes

Since the total N application rates were not the same for the different manure types and species, specific GHG flux rates were calculated by dividing the flux values by the total N application rates outlined in Table 6. Only the results of CO_2-e per kg total N are presented in Figure 5. The specific flux trends for N_2O and CO_2 can be found in Agnew (2010).

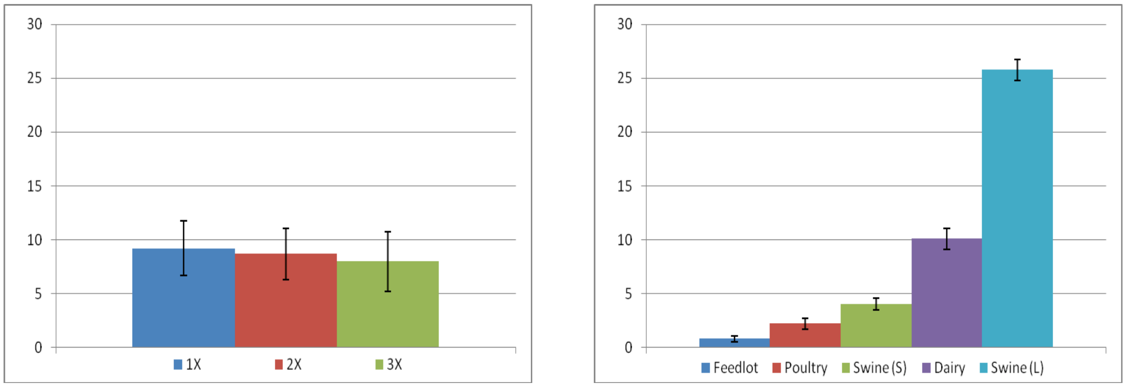
Table 6. Summary of Actual N Application Rates.

		kg total N/ha		
Manure	**Total N**	**1X**	**2X**	**3X**
Feedlot	8.3 kg/Mg	168	335	503
Swine (S)	7.0 kg/Mg	141	283	424
Poultry	17.3 kg/Mg	350	700	1050
Dairy	2.5 kg/m^3	140	211	281
Swine (L)	3.2 kg/m^3	182	273	364

Similar to the absolute CO_2 and CO_2-e flux analyses, specific CO_2-e fluxes were significantly higher from the injected plots (P=0.005) and from the liquid manure (P=0.000). Again, there was no statistical difference among the 1X, 2X and 3X application rates (P=0.428). However, unlike the absolute CO_2-e flux, the specific CO_2-e flux appeared to decrease with application rate (Fig. 5c), although the treatment effect is not significant. This suggests that the rate of increase of absolute GHG flux with application rate is proportional to the rate of increase of N applied. In terms of specific CO_2-e flux, the solid swine manure emitted the most GHG's of the solid manures while the liquid swine emitted the most GHG's of the liquid manures.



(a) (b)



(c) (d)

Figure 5. Treatment effects on mean specific CO_2-e fluxes (mg/kg N applied/s). Solid bars and line error bars correspond to average values and standard errors of the means respectively. (a) Effect of application method, (b) effect of manure type, (c) effect of application rate, and (d) effect of manure species.

Discussion

Effect of Application Method and Manure Type on GHG Emissions

Not unexpectedly, injection of manure increased overall CO_2-e emissions measured 24 hrs after application. The CO_2-e fluxes from the injected treatments were 3.2 times higher than CO_2-e fluxes from the surface treatments (specific CO_2-e flux was also 3.2 times higher from injected plots). While both CO_2 and N_2O significantly increased with injection, the increase in N_2O flux was more pronounced. The overall mean CO_2 flux from the injected plots was 2.5 times higher than from the surface plots while the overall mean N_2O flux from the injected plots was 13.5 times higher than from the surface plots (mean manure induced flux was 10 times higher). This suggests that the enhanced microbial decomposition and increased CO_2 respiration due to increased contact under the soil surface is not entirely responsible for the increased emissions due to injection. The soil aeration or oxygen status when manure is placed under the surface is likely to become partially or fully anaerobic due to reduced diffusion rates and rapid microbial activity that depletes the oxygen available very soon after application. Microbes that degrade organic material in anaerobic or partially anaerobic conditions then use nitrate as a terminal electron acceptor and produce more N_2O through denitrification than microbes that degrade organic material in aerobic conditions (on the surface).

While previous research has sometimes found few significant trends in the effect of manure application on GHG fluxes, most researchers have noted in previous research significantly higher fluxes from liquid manure applications than solid manure applications in the short term. Results from this study also indicate that GHG fluxes measured 24 hours after application from liquid manure were higher than from solid manure (CO_2-e fluxes were 4.5 times higher and specific

CO_2-e fluxes were 7.5 times higher). The manure induced N_2O flux was almost 100 times higher from liquid applications than from solid applications while the CO_2 and specific CO_2 fluxes were 3.5 and 7.5 times higher, respectively.

Because liquid manures are usually stored under anaerobic conditions, liquid manure contains higher levels of water-soluble carbon and nitrogen (Banham and Haugen-Kozyra, 2004; Moolecki et al., 2002), leading to increased rates of nitrification and denitrification after it is applied to the soil. In solid manure, nutrients are physically protected from the attack of decomposers by the solid matrix (Rochette et al., 2004). Additionally, the N and C in solid manures tend to be in organic forms that release available N very slowly (Qian and Schoenau, 2002). The low NH_4-N content in solid manure results in less nitrification to NO_3 and subsequent denitrification to N_2O. However, due to the inclusion of bedding material such as straw, solid manures tend to have high total C contents. In fact, due to this high C content, feedlot manure addition can actually initially immobilize inorganic N. These findings are consistent with the negative manure induced N_2O flux for solid manure observed in this study (fig. 3). The C in solid manure can also be mineralized to CO_2 over time, resulting in higher cumulative emissions from solid manure applications, as was observed in Loro et al. (1997). Since C is also used by microbes during nitrification and denitrification, many researchers have noted that available C content is as important as NO_3 and O_2 concentrations in driving the N transformation processes (Myrold and Tiedje, 1985; Hojberg et al., 1994 in: Rochette et al., 2000b).

All of the previous research that reported increased emissions after subsurface application used liquid manure or slurry. The effect of injection on emissions from solid manure has not been investigated. When the results from the solid manure applications were analyzed separately from the liquid manure applications, the N_2O flux (mean flux, median specific flux and manure induced flux) were significantly higher from the injected plots for both manure types, but the magnitude of increase was much higher from the liquid plots. For example, the manure induced N_2O flux from the solid plots was 2.67 times higher due to injection while the manure induced N_2O flux from the liquid plots was 19 times higher due to injection. Therefore the liquid manure with more N in ammonium form coupled with the addition of liquid that will initially reduce water filled pore space is more likely to be affected by placement strategy as related to N_2O emissions.

The results from the different manure species indicated that injection significantly increased N_2O fluxes from the liquid swine and solid poultry manures, likely due to their higher NH_4 contents. The ammonium probably rapidly nitrified to NO_3 which was then susceptible to denitrification and transformation to N_2O. Since both the nitrification and denitrification processes are sources of N_2O (Firestone and Davidson, 1989), this rapid nitrification is a significant source of N_2O from those manures. Interestingly, injection significantly increased CO_2 fluxes from only the liquid swine and solid poultry manures as well, suggesting that the microbial activity and decomposition were higher in the soil after the application of those manures. The differences between surface fluxes and injected N_2O fluxes were too small and variable to determine significance for the other manure species. The recalcitrant nature of some cattle manures and composts (Qian and Schoenau, 2002) could explain a reduced effect of placement for the feedlot manure.

In order to fully assess the effect of application method and manure type on total GHG emissions, fluxes should be monitored over several weeks or months after application. Alternatively, mechanistic models that predict nutrient transformations may be used to simulate the effects of varying environmental conditions associated with different application techniques and manure types. This way, the entire effect of applying liquid or solid manure and the placement of the manure on total GHG emissions can be assessed. Indeed, part of the reason for variable results reported in the literature is due to the different time scales used in the assessment. Previous researchers have monitored fluxes anywhere from 72 hrs up to 6 months after application (Lovanh et al., 2008; Sistani et al., 2008; Weslien et al., 1998; Perala et al., 2006; Flessa and Beese, 2000; Wulf et al., 2002) and up to 1 year after application (Chang et al., 1998; Goodroad et al., 1984; Rochette et al., 2004). Since manure type and application method are likely to affect fluxes in the longer term, comparisons after only 24 or 72 hrs will not represent the full impact of the manure or application treatment. Similarly, measurements made only several weeks or months after application may miss important short-term pulses of GHG.

Effect of Application Rate on GHG Emissions

Generally, absolute fluxes of N_2O and CO_2 increased with application rate, although only the CO_2 fluxes from the manured plots were distinguishable from the control plots. Therefore, it appears that manure addition increased microbial populations and activity (and thus, CO_2 by respiration), but the onset of N transformations such as nitrification and denitrification may not have yet been sufficient to produce significant, measurable increases in N_2O flux with the different rates. The amount of manure applied (1X, 2X or 3X) did not affect CO_2 or CO_2-e flux in the short-term.

In line with findings of the current study, Hansen et al. (1993) also found no effect of manure application rate on N_2O flux. However, the authors noted that increasing levels of cattle slurry resulted in a reduction in N_2O flux per kg NH_4-N applied (Hansen et al., 1993). Gregorich et al. (1998) also found a non-proportional CO_2 flux response with increasing application rate suggesting that proportionately more manure C was retained in the soil with increasing levels of manure applied. This effect could be due to the fact that the microbial population has a finite capacity for respiration and activity. When the GHG fluxes from this study were expressed on a per kg N applied basis (i.e.: specific flux), CO_2-e fluxes decreased (but not significantly) with application rate. These results suggest that GHG emissions from manure application may be proportional to the amount of N applied, at least over the range of rates examined. These results agree with the IPCC assumption that N losses increase proportionally with the amount of N applied (IPCC, 1997; Penman et al., 2000).

Conclusions

The results of the absolute flux analysis showed that injection significantly increased CO_2-e fluxes for both solid and liquid manure. The overall CO_2-e fluxes from the injected treatments were 3.2 times higher than CO_2-e fluxes from the surface applied plots, mainly due to a pronounced increase in N_2O fluxes. This is explained by creating conditions with liquid injection that are highly conducive to the conversion of the available N and C to GHG, especially N to N_2O and N_2 by denitrification. The CO_2-e fluxes from the liquid manure applications were also higher than the CO_2-e fluxes from the solid manure applications. This was likely due to a high proportion of N in liquid manure in the ammonium form due to the anaerobic conditions during liquid manure storage. The solid manures used in this study had very little ammonium available for nitrification and denitrification. However, this comparison was made only 24 hrs after application. Solid manure generally has a higher C content, which will mineralize over time, likely providing for sustained denitrification if the conditions remain anaerobic. It is likely that conditions beneath the soil surface will remain anaerobic for long periods of time as the diffusion rate of oxygen into the topsoil is often lower than the rate of oxygen use by the increased microbial activity. This could result in prolonged denitrifier activity and N_2O emissions.

Doubling and tripling a one year agronomic application rate had no significant effect on the CO_2-e fluxes, although the absolute flux tended to increase with increased application rate. However, the specific flux (the flux per kg N applied) remained relatively constant with application rate. This indicates that GHG emissions from manure applications were approximately proportional to the amount of manure applied in this study.

When deciding whether or not to inject manure, producers must evaluate the overall environmental and economic impact of the technology. On one hand, subsurface application of livestock manure often constitutes an effective means to reduce odour emissions (Agnew et al. 2010). However, the need to limit odour complaints must be weighed against the potential economic and environmental costs associated to increased GHG emissions. Since it appears that subsurface application of both solid and liquid manure will increase total GHG emissions over a period of time after application, it may not be possible to reduce both odour and GHG emissions using that particular management practice. In addition, other environmental and economic issues related to subsurface manure application, such as increased soil compaction, increased energy requirements, soil disturbance, and the increased field area required to dispose of the manure at agronomic rates, must also be considered when assessing the overall impacts of manure injection compared to surface application.

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