Nutr Cycl Agroecosyst (2011) 89:105–114
DOI 10.1007/s10705-010-9380-6

ORIGINAL ARTICLE

Greenhouse gas emissions when composting manure from cattle fed wheat dried distillers' grains with solubles

X. Hao · M. Benke · F. J. Larney · T. A. McAllister

Received: 1 May 2010 / Accepted: 19 June 2010 / Published online: 3 July 2010

Abstract Dried distillers' grains with solubles (DDGS) are a co-product of ethanol production that is increasingly available for use as a livestock feed. Including DDGS in diets could affect animal manure properties and impact manure management strategies. The objectives of this study were to investigate changes in the rate of greenhouse gas (GHG) emissions during composting and final properties of manure compost when DDGS is included in feedlot cattle diets. Treatments were: (1) Control; manure from cattle fed a typical finishing diet containing barley (*Hordeum vulgare* L.) grain and silage and (2) DDGS; manure from cattle fed a finishing diet with 60% DDGS from wheat (*Triticum aestivum* L.) in the dietary ration. Manure, consisting of feces, urine and wood shavings, was composted in open windrows. Samples were collected for analysis at initiation and completion of composting. Greenhouse gas surface fluxes were collected weekly during the first 4 weeks and every 2–3 weeks for the remainder of the composting period. The DDGS compost had lower total C, but similar total N (TN) content relative to Control, reflecting the initial manure conditions. The DDGS compost also had higher moisture, higher water-extractable NH_4^+ and NO_3^-, a greater fraction of TN in available form, and a lower pH than the Control. The O_2 consumption and N_2O emission from DDGS compost were higher, whereas CO_2 and CH_4 emissions were similar to Control. The higher N_2O emissions from DDGS compost were likely related to the high water-extractable N content in DDGS manure. Increased use of DDGS in feedlot diets may have environmental repercussions that include greater emissions of GHG (N_2O) during manure composting. From an end user perspective, enhanced availability of N could increase the nutrient value of the compost for crop production.

Keywords Methane · Nitrous oxide · Open windrow composting · Beef feedlot manure · Wheat dried distillers' grains · Carbon · Nitrogen

Introduction

In response to high oil prices, climate change and government bio-energy initiatives, ethanol production has increased in North America. This has led to increased availability and use of dried distillers' grains with solubles (DDGS) as a livestock feed (Berger and Good 2007; Klopfenstein et al. 2007). With the nitrogen content in DDGS approximately two to three times that of unprocessed grain (Spiehs et al. 2002; Widyaratne and Zijlstra 2007), adding DDGS to livestock diets could potentially change the manure pH, N form and content, and C/N ratio, given

---

X. Hao (✉) · M. Benke · F. J. Larney · T. A. McAllister
Agriculture and Agri-Food Canada, Lethbridge Research Centre, 5403-1st Avenue South, Lethbridge, AB T1J 4B1, Canada
e-mail: xiying.hao@agr.gc.ca

that these properties are influenced by diet (Hao et al. 2005b; Velthof et al. 2005; Yan et al. 2006; Maguire et al. 2007; Hao et al. 2009).

Hao et al. (2009) reported increases in manure pH, NH_4^+ content and NH_4^+/TN ratio as the level of wheat DDGS in cattle diets increased. These changes could potentially affect the dynamics of CH_4 and N_2O greenhouse gas (GHG) emissions during manure storage and composting, as GHG emissions are affected by manure properties including pH (Ekince et al. 2000; Hao et al. 2005a), N status (Szanto et al. 2007), C/N ratio (Eklind and Kirchmann 2000; Yamulki 2006), availability of NH_4^+ and NO_3^- (Hao et al. 2001) and degradability of C and N (Barrington et al. 2002; Paillat et al. 2005; Yamulki 2006). A positive relationship between dietary protein level and N_2O emission from solid dairy manure storage has been reported (Külling et al. 2001).

Because nitrification of NH_4^+ to NO_3^- is responsible for reducing pH during composting (Sánchez-Monedero et al. 2001), the elevated NH_4^+ content in DDGS manure could cause a greater change in pH during composting than when composting manure collected from traditional barley grain diets. This could potentially impact CH_4 emission as methanotrophs are more tolerant of alkaline pH than are methanogens (Dunfield et al. 1993; Semrau et al. 2008). Additionally, CH_4 formation can be inhibited by free NH_3, a factor that has been reported to be responsible for the lower CH_4 yield during anaerobic digestion (Hansen et al. 1998; Vedrenne et al. 2008). Reduced CH_4 emissions have also been reported from dairy slurry collected from cows fed high protein compared to low protein diets (Külling et al. 2001).

Assessing the impact of DDGS in diets on GHG emission from manure is important to the development of environmentally-sound manure management strategies. The objective of this study was to investigate the rates of GHG and NH_3 emissions during composting of manure collected from cattle fed wheat DDGS compared to a typical barley-based finishing diet.

Materials and methods

Experimental design

The study was conducted in summer-fall 2007 in a semiarid southern Alberta climate. The daily precipitation and mean air temperature data were obtained from a weather station less than 500 m from the research site. During the experimental period, the mean daily air temperatures varied considerably, with an average air temperature of 11.3°C over the study period. This value was 1.7°C higher than the long-term average. The total precipitation received during the experiment was 67.6 mm with a total evaporation of 704.4 mm, values which are 67 and 179%, respectively, of the long-term average.

Two types of manure were collected for the study: (1) Control; manure from cattle fed a typical finishing diet (DM basis) containing 85% barley grain, 10% barley silage and 5% supplement and (2) DDGS; manure from cattle fed a diet (DM basis) containing 60% wheat DDGS, 25% barley grain, 10% barley silage and 5% supplement. The feeding experiment was conducted in a research feedlot from 5 Dec. 2006 to 18 June 2007 with cattle penned by feeding treatment and bedded on wood chip shavings. Individual pens were cleaned in April 2007 and manure was stockpiled by treatment. Cattle continued to be fed their respective diets. Upon completion of the feeding trial (18 June 2007), cattle were moved out and the manure was left in the feedlot pens for an additional 45 days. Pens were cleaned by treatment on 2–3 August, just prior to construction of the windrows.

Windrow management

Within each treatment, stockpiled (from 18 June) and freshly cleaned (2–3 August) manure were thoroughly mixed by loading buckets of each into a tractor-mounted manure spreader (Model 225, Hydra–Spread manure spreader, N. E. Hagedorn & Sons, Paisley, Ontario, Canada) before being placed in two replicate compost windrows per treatment (four windrows in total). A trench was constructed along the peak of each windrow using the front-end loader, and sufficient water was added to the trench to raise the moisture content to 0.65 g g^{-1}. The windrows were turned with a tractor-pulled EarthSaver windrow turner (Fuel Harvesters Equipment Inc., Midland, TX) to ensure uniformity. This point was designated as the initiation of the composting period (Day 0). Compost windrows were subsequently turned on Days 7, 13, 20, 35, 49, and 70, and sampling was discontinued on Day 99 (15 Nov. 2007), by which point the temperature in the compost windrows had fallen below 40°C.

Windrow temperature

The temperature in each windrow was recorded every 20 min at nine locations using three thermocouple probes placed at the top, middle and bottom and linked to a data logger (Sciemetric, Nepean, ON). Probes were installed as soon as compost windrows were formed and were removed and reinstalled with each turning. Temperatures at each location were averaged and reported as mean daily values (Fig. 1).

Initial manure and final compost properties

At the beginning (8 Aug. 2007) and end of composting (15 Nov. 2007), samples were collected from each windrow (12 samples × 2 replicates = 24 samples for each treatment) and analyzed for chemical and physical properties. At each sampling time, compost materials were collected from the windrow peak (0 cm) and distances of 38, 76, 102, and 127 cm below the surface. Samples were divided into three portions. Water content (WC) was determined gravimetrically by drying 10 g in an oven at 60°C for 6 days.

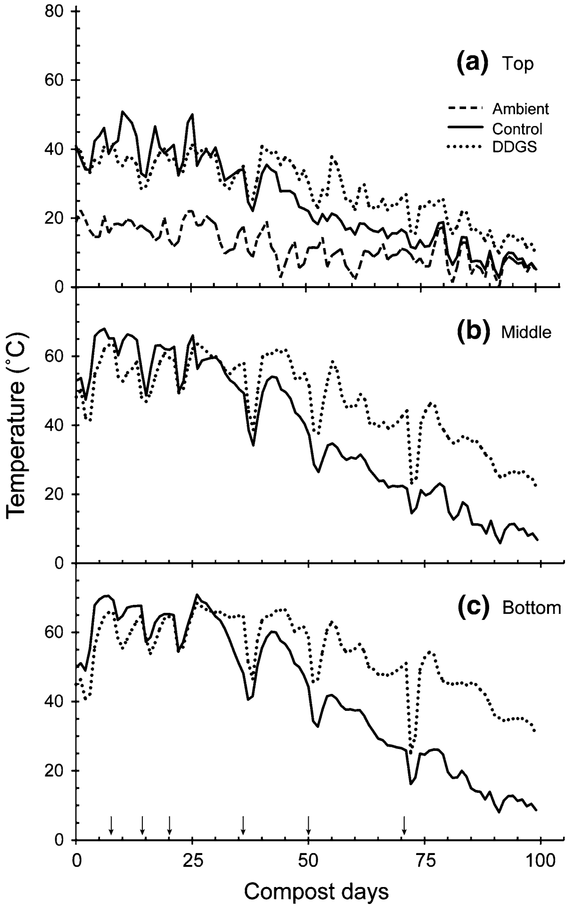

Fig. 1 Mean ambient and composting temperatures at three depths over the composting period (8 Aug. 2007 to 14 Nov. 2007; *Arrows* indicate windrow turning dates)

A 50-g portion was placed into a 500-ml bottle containing 250 ml de-ionized water and shaken on a horizontal reciprocal shaker for 1 h. After pH measurement (Model 290A pH meter, Orion, Boston, MA), samples were centrifuged at 16,300 *g* for 15 min and filtered. Filtrate was measured with an EC meter (Model 125A, Orion, Boston, MA). Filtrate NH_4^+ and NO_3^- content were determined using a Bran + Luebbe AutoAnalyzer III (Bran + Luebbe GmbH, Norderstedt, Germany) and a Dionex AS50 Ion Chromatograph (Dionex, Sunnyvale, CA).

The third portion (about 1 kg) was freeze-dried and coarsely ground (<2 mm). Subsamples were further ground (0.150 mm) for total C (TC), and total organic N (TN) determination. The TC and TN contents were determined using a CN analyzer (NA 1500 Series 2, Carlo Erba Instruments, Milan, Italy). The initial manure and the final compost properties are presented in Table 1.

Gas collection and analysis

Greenhouse gas and O_2 content profiles inside the composting windrows and their surface fluxes were measured weekly during the first 4 weeks and every 2 weeks for the remainder of the experiment. Gas samples (10 ml) were collected at 0, 38, 76, 102 and 127 cm below the windrow peak surface with a depth sampler similar to that described by Hao et al. (2001). Surface flux gas samples were collected using a vented static chamber technique similar to Hutchinson and Mosier (1981) as modified by Hao et al. (2001). Two sets of surface flux samples were collected from each composting windrow (four replications for each treatment). Gas samples (10 ml) were collected from each chamber at 0, 5, 10, 20 and 30 min after the chambers were placed on the windrow surface. Each gas sample was extracted with an air-tight syringe and injected into a 5.9-ml, pre-evacuated, septum-stoppered vial (Exetainer; Labco Limited, Buckinghamshire, UK). The samples were analyzed for O_2, CO_2, CH_4 and N_2O content

Table 1 Characteristics of initial manure and final compost (Day 99)

Date	Treatment	Moisture g g^{-1}	pH	EC dS m^{-1}	TC g kg^{-1}	TN g kg^{-1}	C/N ratio	NH_4^+–N mg kg^{-1}	NO_3^-–N mg kg^{-1}	AN[a]/TN
Initial	Control	0.504 ± 0.037	8.94 ± 0.05	2.3 ± 0.1	429 ± 5	18.3 ± 0.5	26.3 ± 0.6	1,495 ± 190	148 ± 53	0.090 ± 0.005
	DDGS	0.551 ± 0.041	8.79 ± 0.07	3.8 ± 0.2	398 ± 5	20.3 ± 0.5	23.3 ± 0.3	2,708 ± 331	141 ± 48	0.140 ± 0.017
Final	Control	0.369 ± 0.018	7.76 ± 0.04	5.9 ± 0.3	407 ± 1	20.1 ± 0.2	20.8 ± 0.3	55 ± 23	471 ± 52	0.026 ± 0.010
	DDGS	0.417 ± 0.009	7.45 ± 0.05	8.7 ± 0.4	375 ± 4	21.0 ± 0.4	19.3 ± 0.3	708 ± 16	6,039 ± 296	0.333 ± 0.011

[a] AN is the sum of NH_4^+– and NO_3^-–N

using a gas chromatograph (Varian 3800; Varian Instruments, Walnut Creek, CA) equipped with an electron capture detector (ECD), a flame ionization detector (FID), and a thermal conductivity detector (TCD), and a micro-GC (Varian 4900) equipped with ECD and TCD. The content versus time relationships for each chamber were fitted with a second-order polynomial equation for each sampling time (SAS Institute Inc 2005), and the flux at time 0 was calculated by taking derivatives of the second-order polynomials (Hao et al. 2001). Cumulative emissions were approximated by assuming that daily fluxes represented the average for each period. The total GHG emissions over the composting period were expressed per initial unit surface area (kg C m^{-2} and kg N m^{-2} of manure) and initial unit dry weight (kg C Mg^{-1} and kg N Mg^{-1} manure).

Statistical analysis

All data were analyzed using Proc MIXED in SAS (SAS Institute Inc 2005). When there was a significant treatment effect, a means comparison was conducted using the Tukey test ($P < 0.05$).

Results and discussion

Manure and compost properties

The DDGS manure used in the composting experiment had higher ($P < 0.05$) EC and water-extractable NH_4^+ (including dissolved NH_3) levels, similar ($P > 0.05$) moisture, pH, TN, water extractable NO_3^- and C/N ratio, and a lower ($P < 0.05$) TC content than the Control manure (Table 1). Most available N (AN: sum of water-extractable N) was in the form of NH_4^+ (>90%). The higher EC and water-extractable NH_4^+-N in wheat DDGS than Control manure reflects greater N and salt excretion into manure originating from the DDGS diet, a result that is consistent with a previous assessment of manure arising from feedlot cattle in which cattle were fed increasing quantities of wheat DDGS (Hao et al. 2009). The high NH_4^+ content also contributed to a greater ratio of available N to TN in wheat DDGS (0.155) manure than in the Control manure (0.09) since the proportion of NH_4^+ to TN in manure is generally higher with higher protein diets (Canh et al. 1998).

Decomposition and loss of water, TC and TN while manure was stored in piles or in the pen reduced pH, TC, TN and NO_3^- content in the Control and wheat DDGS manure at the time of composting as compared to when it was collected immediately upon completion of the feeding experiment (Hao et al. 2009). This outcome would suggest that considerable NH_3 volatilization and organic matter decomposition occurred during the stockpiling period as well as the 45 days that the manure was stored in the feedlot pen prior to initiation of composting.

In the final compost, the DDGS treatment had higher ($P < 0.05$) moisture, EC, water extractable NH_4^+ and NO_3^- content, and AN/TN ratio, a similar ($P > 0.05$) TN and C/N ratio, but a lower ($P < 0.05$) pH and TC content than Control compost. The higher EC and NH_4^+, similar TN and lower TC content reflect the properties of manure at the initiation of composting. Reduced volatile loss of NH_3 at a lower pH likely contributed to the higher NH_4^+ content in wheat DDGS compost as compared to Control compost. Additionally, higher moisture content in wheat DDGS compost (Table 1) also likely reduced aeration of the compost, lowering NH_3 emissions (Abd El Kader et al. 2007).

The lower pH in wheat DDGS compost could be due to a higher initial manure NH_4^+ content, given the release of H^+ ions as NH_4^+ is converted to NO_3^- during nitrification (Sánchez-Monedero et al. 2001). This is consistent with the higher ($P < 0.05$) NO_3^- content of wheat DDGS compost as compared to Control compost. The slightly but significantly lower pH values in the DDGS compost (7.45) compared with Control (7.76) did not reflect the large differences in NO_3^- content in the final between DDGS (6,039 mg kg^{-1}) and Control (471 mg kg^{-1}) composts. The narrower pH differences between the two treatments could be due to increases in humic acid content during composting (Larré-Larrouy and Thuriès 2006; Spaccini and Piccolo 2009) and the ability of humic acid to buffer pH changes through physiosorption of H^+ to humic acid (Pertusatti and Prado 2007). Increases in NO_3^- content were largely responsible for the higher ($P < 0.05$) AN to TN ratio in wheat DDGS (0.333) as compared to Control (0.026) compost. A higher AN to TN ratio in DDGS compost increases its agronomic value as less additional N fertilizer would be required to meet crop requirements.

Compost windrow temperatures

There was considerable variation in windrow temperatures during open windrow composting (Fig. 1). Overall temperatures in DDGS compost relative to the Control compost were lower for the first 25 days, similar for the next 10 days and higher for the remainder of the composting period. Low temperatures in the DDGS windrows for the first 25 days were probably due to the high water extractable NH_4^+ (incl. dissolved NH_3) content in DDGS manure (Table 1) that may have negatively affected microbial activities. Combined with higher moisture content (Table 1), which reduced the amount of O_2 available in the windrow for aerobic composting, less heat was generated. Additionally, higher moisture content in DDGS manure also means more heat is needed to raise the compost temperature.

Windrow temperatures were higher in the DDGS than in the Control compost after Day 35, and this difference was greatest at the bottom location in the windrow (Fig. 1). The lower temperature near the surface reflects dissipation of heat to the surrounding air. Compost temperatures dropped after turning by as much as 25°C during early and at least 5°C during late composting as considerable heat from the pile was released during the turning process. Temperatures returned to pre-turning or slightly lower values after each turning. The temperature of both compost windrows decreased progressively as composting neared completion, reflecting a decline in microbial activity. Peak temperatures were higher and more prolonged in DDGS than in Control compost (Fig. 1). At the completion of composting, the average temperature in the DDGS treatment remained approx. 20°C higher than in the Control at the bottom location. The higher temperature in the DDGS compost compared with Control, was probably related to its higher moisture content, which improved heat retention, and a low air-filled pore space, which impeded heat loss. One other reason for the higher temperature in DDGS compost could have been higher microbial activity, as reflected in the lower O_2 content in DDGS compost in the second half of the composting period (Fig. 2).

O_2, CO_2, CH_4 and N_2O profiles in compost

The O_2 content was highest at the surface and lowest at the bottom of both DDGS and Control windrows

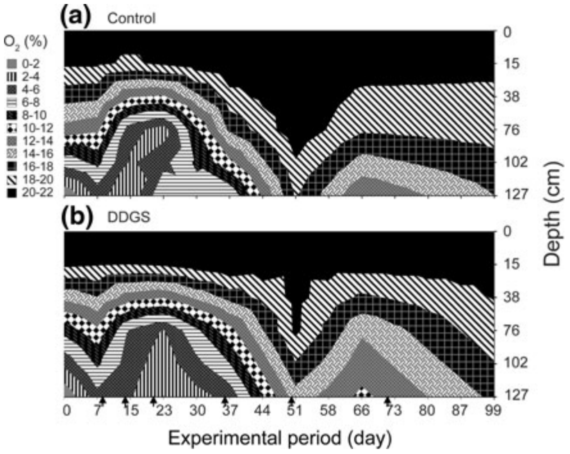

Fig. 2 Compost windrow O_2 concentration profiles (*Arrows* indicate windrow turning dates)

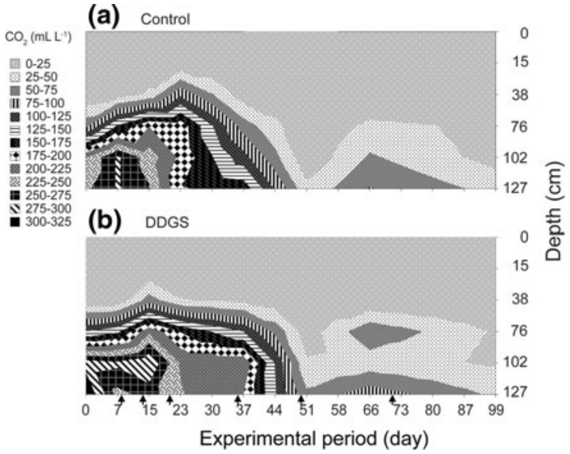

Fig. 3 Compost windrow CO_2 concentration profiles (*Arrows* indicate windrow turning dates)

throughout the composting period (Fig. 2). Additionally, O_2 content in the windrow was lowest at initiation of composting and progressively increased over time. Although microbial activity consumes O_2 as a result of aerobic respiration, a reduction in degradable C reduces the microbial demand for O_2 as composting proceeds. The large increases in O_2 content in the windrow profile around Day 51 reflected O_2 being introduced into the windrow profile by windrow turning and biological activities that consumed O_2 being decreased as composting progressed. The O_2 levels in the Control were lower during early composting but higher during later composting, a pattern that reflected differences in temperature between the compost types. Additionally, moisture loss, particularly in Control compost where it declined to <0.40 kg kg^{-1} (wet weight basis) on Day 51 (Hao, unpublished data), may have limited microbial activity, further lowering microbial O_2 consumption.

Throughout the composting period, CO_2 concentration was lowest at the surface and highest at the bottom of the windrow in both Control and DDGS compost (Fig. 3). The CO_2 content was higher during early composting and decreased over time, consistent with the consumption of O_2 and the microbial decomposition of organic matter (Fig. 2).

Elevated CH_4 content was observed only at ≥76 cm beneath the windrow peaks with maximum content occurring at the bottom of the windrow. Elevated CH_4 occurred only during the first 30 days of composting (Fig. 4) when O_2 consumption and CO_2 production were high. The maximum CH_4 concentration (39.4 ml l^{-1}) observed at the bottom of the DDGS compost windrow was much lower than in the Control windrow (66.8 ml l^{-1}), possibly due to higher NH_4^++ dissolved NH_3 in DDGS compost (Table 1) having inhibited activities of methanogens (Hansen et al. 1998; Külling et al. 2001).

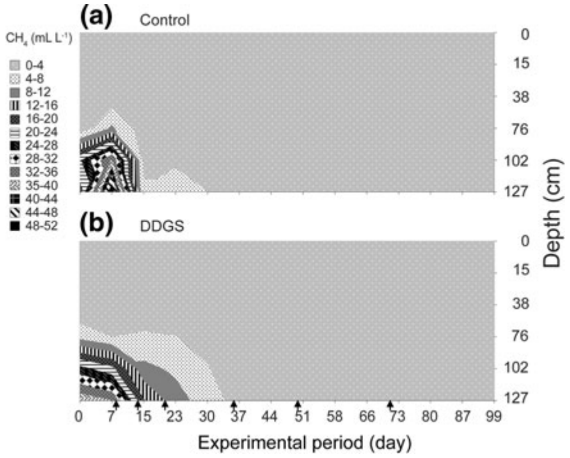

Fig. 4 Compost windrow CH_4 concentration profiles (*Arrows* indicate windrow turning dates)

Elevated N_2O levels were observed at shallower depths (38 cm below the windrow peak) than CH_4 throughout the composting period (Fig. 5). For all sampling dates, N_2O content in the compost windrow was consistently higher in the DDGS treatment than

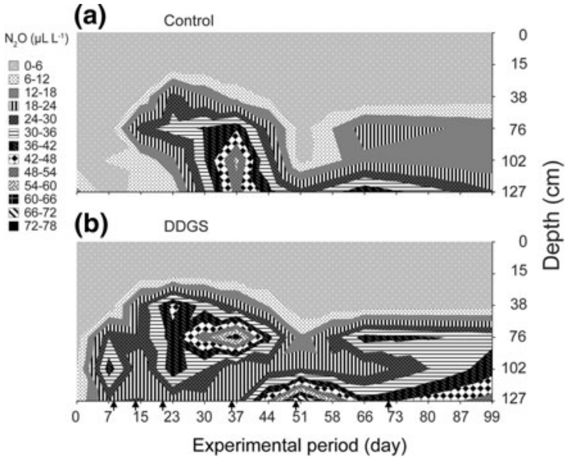

Fig. 5 Compost windrow N_2O concentration profiles (*Arrows* indicate windrow turning dates)

in the Control, likely a result of the higher available N contributing to N_2O production as a result of nitrification and denitrification (Table 1).

Surface fluxes of O_2, CO_2, CH_4 and N_2O

The rate of O_2–O consumption varied considerably, ranging from 157 to 1,116 g d^{-1} m^{-2} (Fig. 6a), whereas the rate of CO_2–C emission ranged from 9 to 339 g d^{-1} m^{-2} (Fig. 6b), and both were generally higher during earlier rather than later stages of composting. The average CO_2 emission rate was similar ($P > 0.05$) between Control and DDGS compost (Table 2), and was positively correlated to the O_2 consumption rate ($r = 0.694$*** for Control; $r = 0.377$* for DDGS; $n = 32$) as organic matter decomposition consumes O_2 and releases CO_2. The similar O_2 consumption and CO_2 emission rates between treatments probably reflect the similar initial manure properties between the Control and DDGS treatments (Table 1).

The rates of CH_4 emission from the DDGS treatment were not different ($P > 0.05$) from the Control (Table 2) with most CH_4 emissions occurring during the first 14 days of composting, and decreasing rapidly to near zero after Day 37 for both treatments (Fig. 6c). This pattern is similar to that reported by others (Lopez-Real and Baptista 1996; Hao et al. 2001, 2004; Fukumoto et al. 2003). The higher O_2 consumption and CO_2 production rates (Fig. 6a, b) during the initial stages of composting created anaerobic conditions that were conducive to CH_4 production. Although the correlation of O_2 consumption with CH_4 emissions ($r = 0.377$*; $n = 34$) in Control compost was significant, it was lower than that obtained for the relationship between O_2 consumption and CO_2 emission. CH_4 and CO_2 emissions were strongly correlated ($r = 0.618$*** for Control; $r = 0.511$** for DDGS; $n = 32$) in both compost types. Most CH_4 was produced near the bottom of the windrow (Fig. 4) where anaerobic conditions existed. The rate of CO_2 emission may in part reflect conditions for CH_4 oxidation near the windrow surface. This CH_4 could be oxidized by methanotrophs while diffusing upwards and outwards before reaching the emitting surface. The sharp drop in CH_4 content near the windrow surface (Fig. 4) further confirms the oxidation near the emitting surface by methanotrophs of CH_4 produced near the bottom of the windrow.

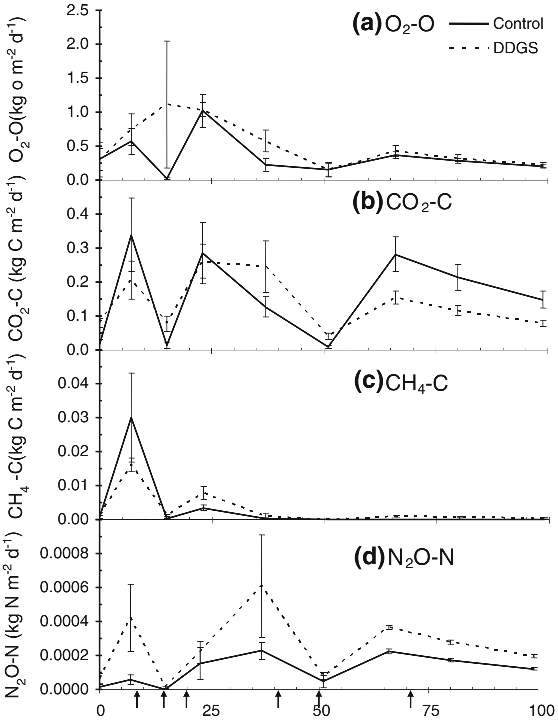

Fig. 6 Rate of greenhouse gas (CO_2, CH_4, and N_2O) emissions and O_2 consumption at the surface of compost windrows over the composting period (*Bars* indicate SEs, and *arrows* indicate windrow turning dates)

Consistent with results reported by Hao et al. (2004) and Mahimairaja et al. (1995), N_2O emissions were observed throughout the composting period

Table 2 Average rate and cumulative amount of greenhouse gas emissions during composting of cattle manure

Treatment	CO_2–C g $m^{-2}d^{-1}$	CH_4–C g $m^{-2}d^{-1}$	N_2O–N g $m^{-2}d^{-1}$	CO_2–C kg Mg	CH_4–C kg Mg	N_2O–N kg Mg	CO_2–C/TC[a] %	CH_4–C/TC[a] %	N_2O–N/TN[a] %
Control	129.1 a[b]	2.60 a	0.098 b	70.0 a	1.40 a	0.053 b	16.34 a	0.34 a	0.29 b
DDGS	115.6 a	2.42 a	0.213 a	63.0 a	1.31 a	0.115 a	15.76 a	0.35 a	0.57 a

[a] TC and TN refer to initial values in cattle manure (before composting)

[b] Within a column, data followed by different lowercase letters differ at 0.05 probability level

(Fig. 6d) due to the presence of NH_4^+ and NO_3^- in manure/compost material (Table 1). The average rate and cumulative amount of N_2O emissions (Table 2) were consistently higher for DDGS than for Control compost (Fig. 6d), reflecting greater N_2O production through nitrification and denitrification as the NH_4^+ and NO_3^- contents as well as moisture content were higher in DDGS compost throughout the experiment. As observed by Hao et al. (2004), the lower (Control $r = 0.385^*$; $n = 33$) and non-significant (DDGS $r = 0.128$; $n = 32$) correlation coefficients between the rate of O_2 consumption and N_2O emission indicate the relationships between the two parameters are complicated and may be non-linear. While production of N_2O occurs via denitrification under anaerobic conditions, under extreme anaerobic conditions N_2O becomes unstable and is further reduced to N_2, perhaps explaining the poor correlation between N_2O emission and O_2 consumption.

Total GHG emissions

The average surface GHG emission rate over the 100-d composting period was 129.1 g C m^{-2} d^{-1} (CO_2), 2.60 g C m^{-2} d^{-1} (CH_4) and 0.098 g N m^{-2} d^{-1} (N_2O) for Control and 115.6 g C m^{-2} d^{-1} (CO_2), 2.42 g C m^{-2} d^{-1} (CH_4) and 0.213 g N m^{-2} d^{-1} (N_2O) for DDGS compost (Table 2). The cumulative emissions in terms of initial dry weight of manure were 70.0 kg C Mg^{-1} (CO_2), 1.40 kg C Mg^{-1} (CH_4) and 0.053 kg N Mg^{-1} (N_2O) for Control and 63.0 kg C Mg^{-1} (CO_2), 1.32 kg C Mg^{-1} (CH_4) and 0.115 kg N Mg^{-1} (N_2O) for DDGS compost (Table 2). Proportionately, C losses as CH_4 ($\leq$1.40 kg Mg^{-1} in both treatments) were small compared to CO_2, representing only 2% total C loss and <0.4% of TC initially present in manure. The N_2O loss was low (0.053 kg Mg^{-1}) and represented 0.29% of initial TN in the Control. For DDGS, the N_2O loss of 0.115 kg Mg^{-1} represented 0.57% of initial TN. Including wheat DDGS in the diet doubled the N_2O emission rate and increased the proportion of TN lost as N_2O during composting. With the global warming potential of N_2O at 296 times and CH_4 at 21 times that of CO_2, the impact of those two GHGs when composting DDGS manure would increase by 37% compared to the Control cattle manure.

During the 45-d manure storage (between the end of the cattle feeding trial and the start of the composting experiment), volatile loss of NH_3 was probably a major cause of N losses, consistent with the much lower water-extractable NH_4^+ and dissolved NH_3 at the start of composting (Table 1) than at the end of the feeding trial (Hao et al. 2009). Additionally, GHG emissions could potentially have occurred during manure storage, which was not investigated in our study. Published results from a study conducted in the same region on passive aeration windrow composting (Hao et al. 2001), which is similar to manure storage, suggest that GHG emissions occurring during manure storage could have been 30 to 50% greater than the values recorded for composting.

Currently, the most common manure management practice is direct land application without prior composting. One advantage of applying fresh manure is that CH_4 is not emitted from arable land under most field conditions (except for rice production) whereas it is unavoidable during manure composting. The N_2O emission from arable land occurs regardless of whether fresh or composted livestock manure was applied, given that both are N sources and N_2O emission is positively related to the amount of N added to the soil.

Application of compost to agricultural land could increase soil C content, and is regarded as a means of C sequestration (Beauchamp and Voroney 1994) while N in the compost is also a good source of N for crop production, especially the higher available N in

DDGS compost. The higher available N in compost means less commercial fertilizer is needed for crop production. However, higher available N in DDGS compost also poses a greater potential for N_2O emission than from Control compost. In summary, increased use of DDGS in livestock diets may have environmental repercussions that include greater N_2O emissions during manure composting. However, for end use, enhanced availability of N in the compost is beneficial for crop production.

Acknowledgments This project was funded by Agriculture and Agri-Food Canada (AAFC) and the AAFC—ABIP program. Technical assistance was provided by G. Travis, B. Hill, P. Caffyn, A. Olson and C. Gilbertson. This is LRC contribution number 38710008.

References

Abd El Kader N, Robing P, Paillat J-M, Leterme P (2007) Turning, compacting and the addition of water as factors affecting gaseous emissions in farm manure composting. Bioresour Technol 98:2619–2628

Barrington S, Choinière D, Trigui M, Knight W (2002) Effect of carbon source on compost nitrogen and carbon losses. Bioresour Technol 83:189–194

Beauchamp EG, Voroney RP (1994) Crop carbon contribution to the soil with different cropping and livestock systems. J Soil Water Conserv 49:205–209

Berger LL, Good DL (2007) Distillers dried grains plus solubles utilization by livestock and poultry, in corn-based ethanol in Illinois and the US: a report from the Department of Agricultural and Consumer Economics, University of Illinois. pp 97–111. http://www.farmdoc.uiuc.edu/policy/research_reports/ethanol_report/Ethanol%20Report%20-%20Ch%206.pdf. Accessed 29 June 2010

Canh TT, Aarnink AJA, Schutte JB, Sutton A, Langhout DJ, Werstegen MWA (1998) Dietary protein affects nitrogen excretion and ammonia emission from slurry of growing-finishing pigs. Livest Prod Sci 56:181–191

Dunfield P, Knowles R, Dumont R, Moore TR (1993) Methane production and consumption in temperature and subarctic peat soils–response to temperature and pH. Soil Biol Biochem 25:321–326

Ekince K, Keener HM, Elwell DL (2000) Composting short paper fiber with broiler litter and additives. Compost Sci Util 8:160–172

Eklind Y, Kirchmann H (2000) Composting and storage of organic household waste with different litter amendments. II: nitrogen turnover and losses. Bioresour Technol 74:125–133

Fukumoto Y, Osada T, Hanajima D, Haga K (2003) Patterns and quantities of NH_3, N_2O and CH_4 emissions during swine manure composting without forced aeration–effect of composting pile scale. Bioresour Technol 89:109–114

Hansen KH, Angelidaki I, Ahring BK (1998) Anaerobic digestion of swine manure inhibition by ammonia. Water Res 32:5–12

Hao X, Chang C, Larney FJ, Travis GR (2001) Greenhouse gas emissions during cattle feedlot manure composting. J Environ Qual 30:376–386

Hao X, Chang C, Larney FJ (2004) Carbon and nitrogen balances and greenhouse gas emission during cattle manure composting. J Environ Qual 33:37–44

Hao X, Larney FJ, Chang C, Travis GR, Nichol C, Bremer E (2005a) The effect of phosphogypsum on greenhouse gas emissions during cattle manure composting. J Environ Qual 34:774–781

Hao X, Mir PS, Shah MA, Travis GR (2005b) Influence of canola and sunflower diet amendments on cattle feedlot manure. J Environ Qual 34:1439–1445

Hao X, Benke MB, Gibb DJ, Stronks A, Travis G, McAllister TA (2009) Effects of dried distillers' grains with solubles (wheat-based) in feedlot cattle diets on feces and manure composition. J Environ Qual 38:1709–1718

Hutchinson GL, Mosier AR (1981) Improved soil cover method for field measurement of nitrous oxide fluxes. Soil Sci Soc Am J 45:311–316

Klopfenstein TJ, Erickson GE, Bremer VR (2007) Feeding corn milling byproducts to feedlot cattle. Vet Clin N Am Food A 23:223–245

Külling DR, Menzi H, Kröber TF, Neftel AN, Sutter F, Lischer P, Kreuzer M (2001) Emission of ammonia, nitrous oxide and methane from different types of dairy manure during storage as affected by dietary protein content. J Agric Sci Camb 137:235–250

Larré-Larrouy M-C, Thuriès L (2006) Does the methoxyl group content of the humic acid-like fraction of composts provide a criterion to evaluate their maturity? Soil Biol Biochem 38:2976–2979

Lopez-Real J, Baptista M (1996) A preliminary comparative study of three manure composting systems and their influence on process parameters and methane emissions. Compost Sci Util 4:71–82

Maguire RO, Crouse DA, Hodges SC (2007) Diet modification to reduce phosphorus surpluses: a mass balance approach. J Environ Qual 36:1235–1240

Mahimairaja S, Bolan NS, Hedley MJ (1995) Denitrification losses of N from fresh and composted manures. Soil Biol Biochem 27:1223–1225

Paillat J-M, Robin P, Hassouna M, Leterme P (2005) Predicting ammonia and carbon dioxide emissions from carbon and nitrogen biodegradability during animal waste composting. Atmos Environ 39:6833–6842

Pertusatti J, Prado AGS (2007) Buffer capacity of humic acid: thermodynamic approach. J Colloid Interface Sci 314:484–489

Sánchez-Monedero MA, Roig A, Paredes C, Bernal MP (2001) Nitrogen transformation during organic waste composting by the Rutgers system and its effects on pH, EC, and maturity of the composting mixtures. Bioresour Technol 78:301–308

SAS Institute Inc (2005) SAS OnlineDoc 9.1.3. SAS Institute Inc., Cary, NC

Semrau JD, Dispirto AA, Murrell JC (2008) Life in the extreme: thermoacidophilic methoanotraphy. Trends Microbiol 16:190–193

Spaccini R, Piccolo A (2009) Molecular characteristics of humic acids extracted from compost at increasing maturity stages. Soil Biol Biochem 41:1164–1172

Spiehs MJ, Whitney MH, Shurson GC (2002) Nutrient database for distillers dried grains with solubles produced from new ethanol plants in Minnesota and South Dakota. J Anim Sci 80:2639–2645

Szanto GL, Hamelers HVM, Rulkens WH, Veeken AHM (2007) NH_3, N_2O and CH_4 emissions during passively aerated composting of straw-rich pig manure. Bioresour Technol 98:2659–2670

Vedrenne F, Béline F, Dabert P, Bernet N (2008) The effect of incubation conditions on the laboratory measurement of the methane producing capacity of livestock wastes. Bioresour Technol 99:146–155

Velthof GL, Nelemans JA, Oenema O, Kuikman PJ (2005) Gaseous nitrogen and carbon mosses from pig manure derived from different diets. J Environ Qual 34:689–706

Widyaratne GP, Zijlstra RT (2007) Nutritional value of wheat and corn distillers' dried grains with solubles; digestibility and digestible contents of energy, amino acids and phosphorus, nutrient excretion and growth performance of grower-finisher pigs. Can J Anim Sci 87:103–114

Yamulki S (2006) Effect of straw addition on nitrous oxide and methane emissions from stored farmyard manures. Agric Ecosyst Environ 112:140–145

Yan T, Frost JP, Agnew RE, Binnie RC, Mayne CS (2006) Relationships among manure nitrogen output and dietary and animal factors in lactating dairy cows. J Dairy Sci 89:3981–3991