

Carbon, Nitrogen Balances and Greenhouse Gas Emission during Cattle Feedlot Manure Composting

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ABSTRACT

Carbon and N losses reduce the agronomic value of compost and contribute to greenhouse gas (GHG) emissions. This study investigated GHG emissions during composting of straw-bedded manure (SBM) and wood chip-bedded manure (WBM). For SBM, dry matter (DM) loss was 301 kg Mg⁻¹, total carbon (TC) loss was 174 kg Mg⁻¹, and total nitrogen (TN) loss was 8.3 kg Mg⁻¹. These correspond to 30.1% of initial DM, 52.8% of initial TC, and 41.6% of initial TN. For WBM, DM loss was 268 kg Mg⁻¹, TC loss was 154 kg Mg⁻¹, and TN loss was 1.40 kg Mg⁻¹, corresponding to 26.5, 34.5, and 11.8% of initial amounts. Most C was lost as CO₂ with CH₄ accounting for <6%. However, the net contribution to greenhouse gas emissions was greater for CH₄ since it is 21 times more effective at trapping heat than CO₂. Nitrous oxide (N₂O) emissions were 0.077 kg N Mg⁻¹ for SBM and 0.084 kg N Mg⁻¹ for WBM, accounting for 1 to 6% of total N loss. Total GHG emissions as CO₂-C equivalent were not significantly different between SBM (368.4 ± 18.5 kg Mg⁻¹) and WBM (349.2 ± 24.3 kg Mg⁻¹). However, emission of 368.4 kg C Mg⁻¹ (CO₂-C equivalent) was greater than the initial TC content (330.5 kg Mg⁻¹) of SBM, raising the question of the net benefits of composting on C sequestration. Further study is needed to evaluate the impact of composting on overall GHG emissions and C sequestration and to fully investigate livestock manure management options.

GLOBALLY, CH₄ emissions from livestock manure represent 5 to 6% of total CH₄ emissions (Hogan et al., 1991), and N₂O represents 7% of total N₂O emissions (Khalil and Rasmussen, 1992). In Canada, livestock manure emits 240 000 Mg yr⁻¹ of CH₄ and 14 000 Mg yr⁻¹ of N₂O (Environment Canada, 2002). Sommer and Møller (2000) observed that current emission inventories for livestock manure are based on limited data. González-Avalos and Ruiz-Suárez (2001) found that CH₄ emission factors for cattle manure in Mexico were less than one-fifth of those proposed in the revised 1996 Intergovernmental Panel on Climate Change guidelines (IPCC, 1996). To better estimate GHG emission levels and develop techniques for emission reduction, more accurate knowledge about CH₄ and N₂O emissions during the handling of animal manure is needed.

In southern Alberta, Canada, cattle feedlots are increasing in size and in animal density with large amounts of solid manure produced on a relatively small land base. Therefore, composting is an attractive alternative to direct land application of fresh manure since both the weight and volume, and hence hauling cost, are reduced considerably (Larney et al., 2001), as are coli-

form bacteria (Larney et al., 2003) and weed seeds (Larney and Blackshaw, 2003).

Recently, the lumber industry has been promoting the use of wood residuals as an alternative bedding material to traditional cereal straw. Wood chip bedding is a mixture of bark, post peelings, and sawdust with a greater portion of materials <5 mm in size (Ward et al., 2000). In addition, wood chips offer greater water holding capacity (Ward et al., 2000), require less frequent addition as bedding (Ward et al., 2001), and keep animals cleaner (McAllister et al., 1998) compared with straw. Moreover, recent drought on the Canadian prairies has decreased the supply (and increased the cost) of straw for livestock bedding, making wood chips a more economically viable alternative.

Bedding materials also affect the physical and chemical properties of fresh feedlot manure and its composted end-product (Larney et al., 2001, 2002). While the effect of bedding materials on emissions of CO₂, N₂O, and CH₄ during composting of feedlot manure has not been widely investigated, GHG emissions are known to occur (Hao et al., 2001; Sommer and Møller, 2000). The amount and the proportion of these GHG emissions and the quality of the final compost product may be affected by several factors such as water content, C/N ratio (Shi et al., 1999; Al-Kanani et al., 1992), and chemical amendments (Swinker et al., 1998; Mahimairaja et al., 1994). The presence of wood chips increases convection of air through the compost windrow (Van Ginkel et al., 2002; Barrington et al., 2003), possibly increasing the supply of O₂, and hence increasing aerobic decomposition. In other words, increases in CO₂ and decreases in CH₄ and N₂O production might be possible. In this study, we investigated C and N balance and emissions of CO₂, CH₄, and N₂O from straw and wood chip-bedded cattle manure during open windrow composting in southern Alberta.

MATERIALS AND METHODS

Compost Windrow Establishment

The study was conducted at the Agriculture and Agri-Food Canada Research Centre in Lethbridge, AB, Canada (49°43' N, 112°48' W). Fresh straw-bedded manure (SBM) or wood chip-bedded manure (WBM) was removed from feedlot pens on 12 July 2000 (Day 0) and formed into four windrows (two SBM, two WBM) on a concrete pad in an open-sided, roofed facility. Each windrow covered an area of about 33 m² (4.3 by 7.7 m) and was about 1.6 to 1.8 m high. Windrows were turned eight times (Days 8, 14, 21, 35, 49, 64, 78, and 99) after initial construction with a tractor-pull windrow turner (Fuel

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Table 1. Initial and final properties of materials used in the experiment.

Treatment	Moisture (wet wt.)	Total C†	TN	C/N	NO ₃ -N	NH ₄ ⁺ -N	pH
	g kg ⁻¹			mg kg ⁻¹			
Day 0							
SBM‡	601 (6)§	330.5 (4.7)	19.92 (0.24)	16.6 (0.1)	182 (55)	2090 (4)	8.35 (0.03)
WBM‡	596 (1)	447.2 (4.5)	12.40 (0.29)	36.1 (0.5)	244 (38)	2666 (207)	6.99 (0.10)
Day 99							
SBM	282 (9)	223.1 (8.0)	16.66 (0.32)	13.4 (0.2)	135 (7)	998 (95)	8.09 (0.03)
WBM	367 (11)	398.5 (6.1)	14.94 (0.20)	26.7 (0.1)	330 (36)	876 (2)	7.00 (0.01)

† All properties are presented on a dry wt. basis unless otherwise indicated.

‡ SBM is the cattle manure from feedlot pens bedded with cereal straw and WBM is cattle manure from feedlot pens bedded with wood chips.

§ Means with standard error in parentheses ($n = 24$).

Harvester Equipment Corp., Midland, TX). Day 99 (19 Oct. 2000) represented the end of thermophilic composting and the onset of the mesophilic or curing phase (when compost windrow temperature <40°C). Weather data were obtained from a meteorological station ~0.5 km from the composting facility.

Physical and Chemical Analyses and Mass Balance

At establishment and just before each turning, a series of measurements and samplings were taken from each windrow. The surface area and volume were calculated from circumference and radius measurements of the windrows, assuming the windrow cross-section was hyperbolic in shape. Bulk density was estimated by weighing an aluminum pail of known volume (0.064 m³) filled with manure-compost (Larney et al., 2001). Particle density was measured using a pycnometer method (Blake and Hartge, 1986) as modified and described by Hao et al. (2001). Total porosity in the windrow was calculated using bulk density and particle density. The water content of the manure-compost samples was determined by oven-drying at 60°C to constant weight. The air-filled pore space was calculated as the difference between total porosity and pores occupied by water.

Before turning, each windrow was cut perpendicular to its length with a skid-steer loader, exposing two vertical faces. Manure samples of approximately 1 kg were collected from each face at depths of 0 to 5, 5 to 15, 15 to 35, 35 to 60, 60 to 90, and 90 cm from the top of the windrow. This represented 24 samples from each bedding treatment. The 0- to 5-cm sampling depth included the windrow peak. Ammonium-N (NH₄⁺-N) and nitrate-N (NO₃⁻-N) concentrations were measured on 10 g of fresh (wet) sample from each depth. Each was shaken in 200 mL of 2 M KCl for 1 h, and filtered through KCl-washed filter paper (Whatman no. 42). Extracts were frozen at -15°C, and then analyzed on a Technicon AutoAnalyzer II. Subsamples were oven-dried at 60°C and finely ground (<150 μm) for total C (TC) and total N (TN) measurement in an automated elemental analyzer (Carlo Erba, Milan, Italy). The pH was measured using oven-dried materials by shaking 30 g (dry wt.) of manure-compost with 120 mL of deionized water for 1 h, then the pH values of the filtrate were determined with a pH meter (Accumet pH meter 50, Fisher Scientific). Characteristics of the SBM and WBM at the beginning of the experiment (fresh manure, Day 0) and on Day 99 (end of thermophilic composting) are shown in Table 1.

For mass balance calculations, the dry matter (DM) and mass losses of C and N during 99 d of composting were calculated by the difference between the amounts on Day 0 and Day 99, assuming that the DM mass loss was entirely due to organic matter loss and that organic matter was TC × 1.724.

Gas Profile Measurements within Compost Windrows

Gas profile measurements were collected two to three times per week for the first 3 wk and once per week thereafter for

the 99 d of composting. Two sets of gas samples were collected at the windrow surface (0 cm) and at 15, 40, 70, and 100 cm below the surface using a multilevel sampler (Hao et al., 2001). All gas samples were taken between 0800 and 0900 h and analyzed for CO₂, CH₄, N₂O, and O₂ on the same day using a gas chromatograph (Varian 3600, Varian Instruments, Walnut Creek, CA) equipped with an electron capture detector (ECD), flame ionization detector (FID), and thermal conductivity detector (TCD). The average concentrations of CO₂, CH₄, N₂O, and O₂ for each windrow were calculated based on the geometric mean of each sampling section, assuming the windrows were hyperbolic in shape. The geometric mean was used since the volume of manure represented by each sampling depth was not equal.

Gas Fluxes from Windrow Surfaces

Gas emissions and O₂ consumption during composting were measured according to the same schedule as gas profile measurements, using a modified vented chamber technique (Hutchinson and Mosier, 1981). Two sets of gas samples were collected from each windrow. At each sampling time, a chamber (15.5 cm in diam. and 15 cm in height) was placed on the peak of the windrow. Ten mL of air was drawn with a plastic syringe from the chamber headspace 0, 5, 10, 20, and 30 min after chamber placement. Immediately after sampling, the syringe needle was placed in a rubber stopper to prevent gas exchange. As with profile measurements, all surface flux gas samples were taken between 0800 and 0900 h and analyzed for CO₂, CH₄, N₂O, and O₂ the same day.

Gas fluxes were calculated from concentrations by assuming a steady state gradient in the underlying windrows (Anthony et al., 1995) as described by Hao et al. (2001). Briefly, the concentration vs. time relationships for each chamber were fitted with a second order polynomial equation ($C = a + bx + cx^2$), where C is the concentration of gases and x is the time in minute(s) for each sampling time (SAS Inst., 2001). The flux at time 0 was calculated by taking derivatives of the second order polynomials ($dC/dx_{x=0} = b$). Cumulative emissions were approximated by assuming that daily fluxes, which were measured more frequently during early composting, represented the average for the whole week. To account for emissions during turning, it was assumed that these were equivalent to the difference between the average profile gas concentration before and after turning and the average air-filled pore space before the turning event. These amounts were added to the total cumulative emissions. The total GHG emissions during the composting period were expressed on an initial surface area (kg C m⁻² or kg N m⁻² of manure) and initial dry weight (kg C Mg⁻¹ manure or kg N Mg⁻¹ manure) basis.

Data Analysis

The O₂, CH₄, and CO₂ flux data were analyzed for two separate composting stages: Day 0 to 49 (early) and Day 50

Table 2. Mass, C, and N balance (dry wt. basis) of straw-bedded and wood chip-bedded cattle feedlot manure during open windrow composting.

Period	SBM†			WBM†		
	Mass	TC‡	TN‡	Mass	TC	TN
	kg Mg ⁻¹					
Initial	1000	330.5 (4.7)§	19.92 (0.24)	1000	447.2 (4.5)	12.40 (0.29)
Day 99	699 (3)	156.1 (6.2)	11.65 (0.26)	735 (50)	292.9 (24.3)	10.98 (0.89)
Total loss	301 (3)	174.4 (1.5)	8.26 (0.03)	265 (50)	154.3 (28.7)	1.42 (1.17)
	%					
Total loss	30.1 (0.3)	52.8 (0.5)	41.5 (0.1)	26.5 (5.0)	34.5 (6.4)	11.4 (9.5)

† SBM is the cattle manure from feedlot pens bedded with cereal straw and WBM is cattle manure from feedlot pens bedded with wood chips.

‡ TC or TN amount divided by mass in this table will be the TC and TN content in Table 1.

§ Means with standard error in parentheses ($n = 2$).

to 99 (late). Three stages (Day 0–14, Day 15–49, and Day 50–99) were used for N₂O due to its distinctive flux patterns. The average rate of O₂ consumption and GHG emission during different stages of composting was obtained by covariance analysis, treating time as the covariant using the PROC MIXED procedure (SAS Inst., 2001). When treatment effects were significant, mean flux among different compost stages and between two bedding treatments were tested using contrast analysis.

RESULTS AND DISCUSSION

Weather Conditions

During the first 49 d of composting (12 July–30 Aug. 2000), the mean daily air temperature ranged from 8.7 to 25.8°C and averaged 2.3°C above the long-term normal. For Day 50 to 99 (31 Aug.–19 Oct. 2000), mean daily air temperature ranged from –3.0 to 21.5°C and averaged 1.6°C above the long-term normal. The mean daily wind speeds ranged from 6 to 34 km h⁻¹. Total rainfall during the 99-d study was 78 mm, much less than the long-term average of 123 mm. Although the windrows were under a roof, which prevented addition of water via precipitation, they were exposed to evaporation. The warmer and drier conditions contributed to a higher evaporation potential (811 mm) than the long-term normal (565 mm) for the 99-d composting period.

Dry Matter, Carbon, and Nitrogen Changes during Composting

The initial water content of fresh SBM and WBM was similar (Table 1) and close to 60% (wet wt.). Water content decreased steadily to 28% (SBM) and 37% (WBM) by Day 99. Windrow turning and hot, dry weather accelerated evaporation and water loss. The DM mass loss during 99 d of composting was 30.1% of initial mass for SBM and 26.8% of initial mass for WBM (Table 2). Therefore, compost yield was 0.699 Mg Mg⁻¹ for SBM and 0.735 Mg Mg⁻¹ for WBM.

The TC levels were initially lower in the SBM (330 g kg⁻¹) than in the WBM (447 g kg⁻¹) treatment reflecting the higher TC content of the wood-chip bedding. The TC content dropped to 223 g kg⁻¹ (SBM) and 398 g kg⁻¹ (WBM) by Day 99. After accounting for the DM mass loss, this was equivalent to a TC loss of 174 kg Mg⁻¹ for SBM (52.8% of initial TC) and 154 kg Mg⁻¹ for WBM (34.3% initial TC, Table 2). Most of the C loss occurred during the first 20 d of composting (Fig.

1a). The difference in TC reductions reflects the characteristics of the bedding materials. Although initial TC content was greater for WBM than SBM, wood chips have more recalcitrant lignin and a less readily degradable hemicellulose than cereal straw (Eklind and Kirchmann, 2000a; Ward et al., 2000). Although woodchips had a greater proportion of smaller sized material than straw before addition to the feedlot pen, the large sized pieces remained relatively unchanged. In contrast, straw had a much longer length initially, but broke down to much smaller-sized particles in the pen. The combination of larger particle sizes and a higher initial C/N ratio (Table 1) contributed to the slower decomposition of

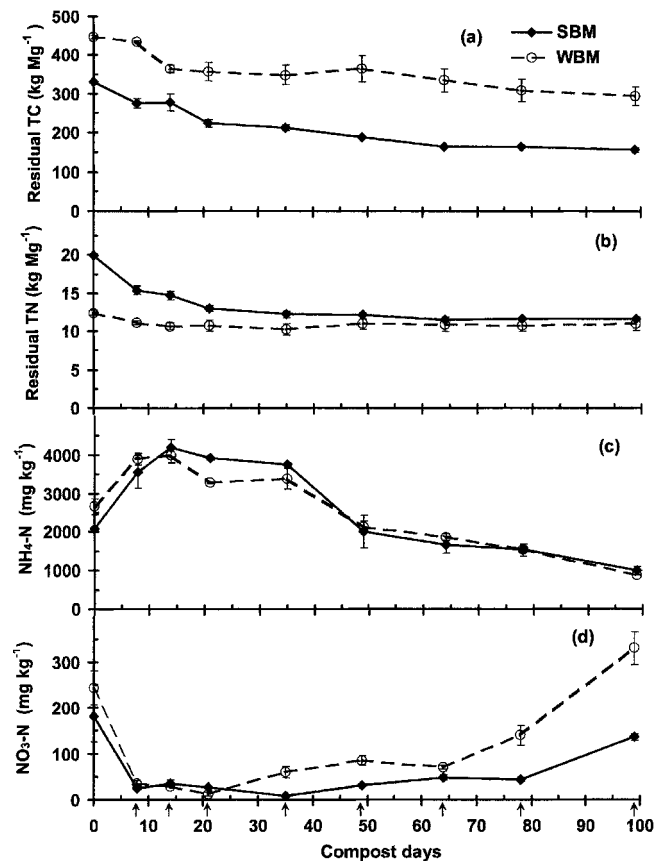


Fig. 1. Changes in (a) total carbon (TC), (b) total nitrogen (TN), (c) NH₄-N, and (d) NO₃-N during composting of straw-bedded manure (SBM) and wood chip-bedded manure (WBM). Vertical bars are standard errors, while vertical arrows on the x axis indicate windrow turning dates.

WBM during the 99 d of composting. Eklind and Kirchmann (2000a) also found that C in finished compost was positively correlated to the lignin content of initial materials.

Initial TN concentrations were 19.9 mg kg⁻¹ (SBM) and 12.4 mg kg⁻¹ (WBM). However, TN decreased to 16.7 mg kg⁻¹ for SBM, but increased to 14.9 g kg⁻¹ for WBM by Day 99 (Table 1). The increase in TN for WBM occurred because the rate of DM mass loss exceeded the rate of N loss. After accounting for the DM mass loss, the TN loss was 8.26 kg Mg⁻¹ for SBM (41.5% of initial TN) and 1.42 kg Mg⁻¹ for WBM (11.4% initial TN, Table 2). Similar to C, most of the N loss occurred during the early stages of composting (Fig. 1b) and the large difference in N loss between treatments reflects the characteristics of the bedding. The higher N loss for SBM could be due to an initially high TN content, a lower C/N ratio (Table 1), or possibly greater ammonia (NH₃) losses. Low C/N ratios generally result in high N losses (Al-Kanani et al., 1992). Price (2001) measured gaseous NH₃ losses from the windrows and found that losses from SBM were two to six times greater than those from WBM. The higher pH with SBM in part contributed to higher NH₃ emission loss since the volatilization of NH₃ increases with pH. Moisture content also played a role in the N loss. Treatments started with similar water content. However, WBM retained more water (Table 1) and potentially more NH₃, since NH₃ is highly soluble in water. The fraction of TN loss from the SBM treatment was similar to values reported by Garrison et al. (2001).

The NH₄⁺ concentration in the windrows increased during the first 35 d of composting (Fig. 1c), when the rate of NH₄⁺ production due to intense organic matter degradation (Sánchez-Monedero et al., 2001) exceeded the rate of NH₄⁺ loss through nitrification of NH₄⁺ to NO₃⁻ and gaseous NH₃ emission loss since NH₄⁺ and NH₃ are in equilibrium in compost solution. In addition, the higher rates of DM mass loss also contributed to increased NH₄⁺ concentration during early composting. The average concentration of NH₄⁺ increased from the initial values of 2090 mg kg⁻¹ (SBM) and 2666 mg kg⁻¹ (WBM) to around 3000 to 4000 mg kg⁻¹ for the first 35 d, then decreased steadily afterward reaching their lowest values of <1000 mg kg⁻¹ on Day 99 (Table 1 and Fig. 1c).

The NO₃⁻-N concentration decreased sharply from the initial 244 mg kg⁻¹ (SBM) and 182 mg kg⁻¹ (WBM) to <40 mg kg⁻¹ for both treatments on Day 8. This decrease continued, reaching a minimum of 10 mg kg⁻¹ on Day 21 (WBM) and on Day 35 (SBM). The NO₃⁻ content remained low for the next 40+ d, then increased again to 135 mg kg⁻¹ (SBM) and 330 mg kg⁻¹ (WBM) on Day 99 (Table 1 and Fig. 1d).

The initial NO₃⁻-N content of cattle manure in our study was much greater than values (<10 mg kg⁻¹) from previous studies in southern Alberta (Larney et al., 2001; Hao et al., 2001). Normally, cattle manure is removed from feedlot pens either in fall (October–November) or spring (May–June). In our study, manure removal was delayed until mid-July, which allowed an

extra 1 to 2 mo of decomposition in the feedlot pen at a time when average air temperatures were approaching their annual peak. Manure decomposition and nitrification of NH₄⁺ was enhanced during this period, which contributed to the higher initial NO₃⁻ concentration for the manure in our study.

Oxygen Consumption and Greenhouse Gas Emissions

The rates of O₂ consumption varied considerably during composting (Fig. 2a) and were significantly greater for SBM than WBM during early composting (Day 0–49). In contrast, no differences between the two treatments were found during late composting (Day 49–99) (Table 3). Contrast analysis also indicated that the O₂ consumption rate was significantly greater during the early stage, compared with the late stage of composting, for SBM but not for WBM (Table 3, Fig. 2a). The rate of O₂ consumption significantly decreased following windrow turning on Days 8 and 14 (Fig. 2a), mainly due to decreased windrow temperatures (temperature data not shown).

The rates of CO₂ emission also varied considerably during composting (Fig. 2b), and followed similar patterns to O₂ consumption. The CO₂ emission rate was significantly greater for SBM than WBM during early

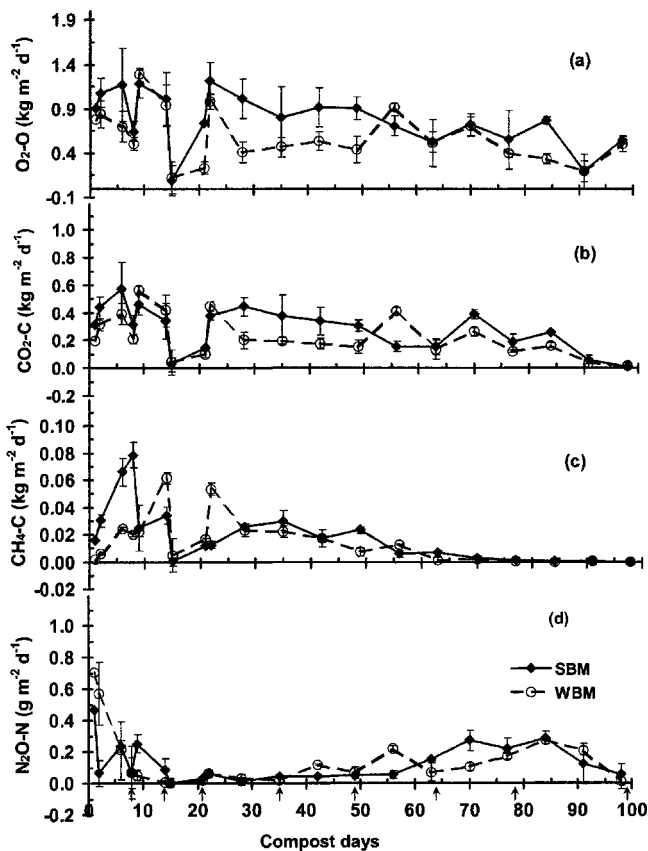


Fig. 2. The rates of (a) oxygen (O₂) consumption, (b) carbon dioxide (CO₂), (c) methane (CH₄), and (d) nitrous oxide (N₂O) emission during composting of straw-bedded manure (SBM) and wood chip-bedded manure (WBM). Vertical bars are standard errors, while vertical arrows on the x axis indicate windrow turning dates.

composting (Day 0–49), but there was no significant difference during late composting (Day 50–99) (Table 3). For both bedding treatments, the rate of CO₂ emissions was also significantly greater early in the composting than later. High rates of O₂ consumption were positively correlated with high rates of CO₂ emission ($r = 0.88^{***}$ for SBM; $r = 0.94^{**}$ for WBM, $n = 32$; ** significant at the 0.01 level; *** significant at the 0.001 level) because organic matter decomposition consumes O₂ and releases CO₂.

The increased rate of O₂ consumption and CO₂ emission from SBM compared with WBM was probably caused by differences in manure characteristics. Cereal straw with less recalcitrant lignin and more readily degradable hemicellulose than wood chip materials (Ekland and Kirchmann, 2000a; Ward et al., 2000), combined with smaller-size particles of the straw compared with the larger wood particles and a lower initial C/N ratio of 17 (Table 1), favored microbial decomposition and hence SBM had higher O₂ consumption and CO₂ emissions. The higher O₂ consumption and CO₂ emission during early composting may be attributed to the greater degradation rate of aliphatics, hemicellulose, and proteins during early stages of composting (Veeken et al., 2001). As this easily degradable fraction of organic matter was depleted, O₂ consumption and CO₂ emission declined.

More than 50% of total CH₄ emission occurred during the first 28 d of composting, decreasing rapidly to near zero after Day 70 for both treatments (Fig. 2c). This is similar to the findings of Lopez-Real and Baptista (1996), Sommer and Møller (2000), and Hao et al. (2001). Initially, relatively high rates of O₂ consumption and CO₂ production (Fig. 2a and 2b) created anaerobic conditions that not only favored production of CH₄, but also increased CH₄ stability. Although correlation coefficients for the rate of O₂ consumption vs. CH₄ emissions ($r = 0.45^*$ for SBM; $r = 0.49^*$ for WBM, $n = 32$; * significant at the 0.05 level) were significant, they were lower than those obtained for O₂ consumption vs. CO₂ emission. This may be partly explained by factors affecting production and emission of CO₂ vs. CH₄. Besides the rate of O₂ supply, the rate of CO₂ production and emission directly depends on the C source in the com-

posting substrate. For CH₄, in addition to C availability in the substrate, other factors, such as the rate of O₂ consumption during the production of CO₂ and the stability of the CH₄ produced, affect the rate of CH₄ emission. For example, while production of CH₄ may be high at the bottom and center of the windrow profile where anaerobic conditions exist, this CH₄ could be oxidized while diffusing upward and outward before reaching the emitting surface (Hao et al., 2001).

The rates of CH₄ emission were not significantly different between SBM and WBM, so all data were pooled in a contrast analysis of the composting stage effect. The CH₄ emission rate (0.0219 kg m⁻² d⁻¹) during early composting (Day 0–49) was significantly greater than late composting (Day 50–99, Table 3).

The highest rates of N₂O emission were measured at the onset of composting (0–14 d), followed by a period of low emissions during midcomposting and a minor peak toward the end (Fig. 2d). In contrast, Martins and Dewes (1992), Lopez-Real and Baptista (1996), and He et al. (2001) reported that N₂O emission occurred only in the later stages of composting when CH₄ production had ceased. Our results suggest that the presence of NO₃⁻ in the manure led to the production and emission of N₂O during early composting when the emission of CH₄ was also largest. A significant correlation between the rate of N₂O emission and average NO₃⁻ concentration in the compost ($r = 0.79^{***}$ for SBM and $r = 0.80^{***}$ for WBM, $n = 16$) suggest that denitrification of NO₃⁻ rather than the nitrification of NH₄⁺ was responsible for N₂O emissions in our study.

Since there were three distinct periods of N₂O emission (Fig. 2d), the rates were analyzed separately for the initial major peak (Day 0–14), the midperiod of lower emission (Day 15–49), and the later minor peak (Day 50–99). There were no significant differences due to bedding in N₂O emission rates for Day 0 to 14 or Day 15 to 49 (Table 4). However, SBM had a significantly greater N₂O emission rate from Day 50 to 99 than WBM. For the SBM treatment, the average N₂O emission rate of 0.200 g m⁻² d⁻¹ during the first 14 d was significantly greater than the rate from Day 15 to 49 (0.027 g m⁻² d⁻¹), but not significantly different from the rate from Day 50 to 99 (0.144 g m⁻² d⁻¹, Table 4).

Table 3. Effect of manure type and compost stage on O₂ consumption and CO₂, CH₄, and N₂O emissions from cattle feedlot manure.

Gases	Treatment	Flux			Contrast
		Early (0–49 d)	Late (50–99 d)	Whole (0–99 d)	Early vs. late
kg m ⁻² d ⁻¹					
O ₂					
Flux	SBM†	0.748	0.527	0.638	***
	WBM†	0.538	0.424	0.481	ns
Contrast	SBM-WBM	***	ns	*	
CO ₂					
Flux	SBM†	0.293	0.184	0.239	***
	WBM†	0.238	0.152	0.195	**
Contrast	SBM-WBM	*	ns	ns	
CH ₄ ‡	Both	0.0219	0.0024	0.0123	***

* Significant at the 0.05 probability level.

** Significant at the 0.01 probability level.

*** Significant at the 0.001 probability level.

† SBM cattle manure from feedlot pens bedded with cereal straw; WBM cattle manure from feedlot pens bedded with wood chips.

‡ There were no significant treatment effects so data for both bedding materials were pooled for the analysis.

Table 4. Effect of manure type and compost stage on the rate of N₂O emission from cattle feedlot manure.

Treatment	Flux				Contrast		
	E1 (0–14 d)	E2 (15–49 d)	Later (50–99 d)	Whole (0–99 d)	E1 vs. E2	E1 vs. Late	E2 vs. Late
	g m ⁻² d ⁻¹						
Flux							
SBM†	0.1995	0.0268	0.1445	0.1103	***	ns	***
WBM†	0.2248	0.0417	0.1113	0.1026	***	***	***
Contrast							
SBM-WBM	ns	ns	**	ns			

* Significant at the 0.05 probability level.

** Significant at the 0.01 probability level.

*** Significant at the 0.001 probability level.

† SBM is the cattle manure from feedlot pens bedded with cereal straw and WBM is cattle manure from feedlot pens bedded with wood chips.

For the WBM treatment, the average N₂O emission rate of 0.225 g m⁻² d⁻¹ during the first 14 d was significantly greater than rates for both Day 15 to 49 (0.042 g m⁻² d⁻¹) and Day 50 to 99 (0.111 g m⁻² d⁻¹, Table 4). The lower nonsignificant correlation coefficients between the rate of O₂ consumption and N₂O emission ($r = 0.18$ for SBM and $r = 0.26$ for WBM, $n = 16$) indicate the relationship between the two parameters was nonlinear and probably complex. While production of N₂O occurs via denitrification under anaerobic conditions, under extreme anaerobic conditions N₂O becomes unstable and is further reduced to N₂, perhaps explaining the poor correlation between N₂O emission and O₂ consumption.

Mass Balance for Greenhouse Gas Emissions and Carbon and Nitrogen

Total cumulative GHG emissions for SBM during the 99-d composting period were 23.27 kg C m⁻² from CO₂, 1.254 kg C m⁻² from CH₄, and 0.0109 kg N m⁻² from N₂O in terms of initial windrow surface area (Table 5). In terms of initial dry weight of manure, emission rates were 165.0 kg C Mg⁻¹ for CO₂, 8.92 kg C Mg⁻¹ for CH₄, and 0.0771 kg N Mg⁻¹ for N₂O (Table 5). For the WBM treatment, cumulative emissions on a surface area basis were 18.60 kg C m⁻² for CO₂, 1.141 kg C m⁻² for CH₄, and 0.0107 kg N m⁻² for N₂O. Expressed on an initial dry weight basis, emissions were 145.6 kg C Mg⁻¹ for CO₂, 8.93 kg C Mg⁻¹ for CH₄, and 0.0842 kg N Mg⁻¹ for N₂O.

For SBM, the measured TC loss of 173.9 kg C Mg⁻¹ as CO₂ and CH₄ emission was almost identical to TC loss (174.4 kg C Mg⁻¹), calculated using initial (Day 0) and final (Day 99) TC concentrations. Similarly, for WBM, the measured TC loss of 154.5 kg C Mg⁻¹ as CO₂ and CH₄ emissions was almost identical to the calculated loss of 154.3 kg C Mg⁻¹. The similarity between gaseous C

emission losses measured with the small static chamber method and C losses calculated using the TC content of the initial manure and final compost suggests that the simple static chamber was a reliable tool for estimating gaseous C emissions during composting, especially under a semicontrolled environment with no leaching or runoff losses due to a roof and concrete floor. Proportionately, C losses as CH₄ (8.9 kg Mg⁻¹ for both treatments) were small compared with those via CO₂, representing only 5.1% (SBM) and 5.8% (WBM) of TC loss or 2.7% (SBM) and 2.0% (WBM) of the initial TC.

The loss of N₂O was low, similar to findings reported from other composting studies (Martins and Dewes, 1992; Kuroda et al., 1996; Eklind and Kirchmann, 2000b; Hao et al., 2001; Sommer, 2001; Amon et al., 2001). The measured N₂O loss was 0.0771 kg Mg⁻¹ and represented only 0.9% of total N loss (8.26 kg Mg⁻¹) and 0.39% of initial TN (19.92 kg mg⁻¹) for SBM. For WBM, the N₂O loss of 0.0842 kg Mg⁻¹ represented about 5.9% of total N loss (1.42 kg N Mg⁻¹), and 0.68% of initial TN (12.40 kg Mg⁻¹). Since there was a roof and concrete floor, runoff and leaching losses of N did not occur. Most N was lost either as NH₃ or possibly N₂.

In addition to the direct GHG emissions, 5.1 kg C Mg⁻¹ (SBM) and 4.2 kg C Mg⁻¹ (WBM) CO₂ was also released from the diesel fuel used to turn and maintain the compost windrows. Using global warming potential factors of 1 for CO₂, 21 for CH₄, and 310 for N₂O, total emissions during composting expressed as CO₂-C equivalents were 368.4 kg C Mg⁻¹ manure for SBM, which was not significantly different from the total emission of 349.2 kg C Mg⁻¹ manure for the WBM treatment. Although most C is emitted as CO₂ (94% total C loss), the impact of CH₄ was greater since its global warming potential is 21 times more than CO₂ (Table 5). Although N₂O contributed <5% of total GHG emission (CO₂-C

Table 5. Cumulative CO₂, CH₄, and N₂O emission during composting of cattle feedlot manure.

Treatment	Emission rate†						CO ₂ -C equivalent‡				
	CO ₂ -C	CH ₄ -C	N ₂ O-N	CO ₂ -C	CH ₄ -C	N ₂ O-N	CO ₂ -C	CH ₄ -C	N ₂ O-N	Fuel§	Total
	kg m ⁻²			kg Mg ⁻¹			kg C Mg ⁻¹				
SBM†	23.27a	1.254a	0.0109a	165.0a	8.92a	0.0771a	165.0a	187.3a	10.2a	5.1a	368.4a
WBM†	18.60a	1.141a	0.0107a	145.6a	8.93a	0.0842a	145.6a	188.6a	11.2a	4.2a	349.2a

† Initial surface area of 45.285 m² windrow⁻¹ and weight of 6.384 Mg windrow⁻¹ for SBM treatment and initial surface area of 43.565 m² windrow⁻¹ and weight of 5.560 Mg windrow⁻¹ for the WBM treatment were used in converting the GHG emission from an initial surface area basis to an initial manure dry wt. basis.

‡ Using global warming potential of 1, 21, and 310 for CO₂, CH₄, and N₂O, respectively.

§ Based on fuel consumption of 0.266 and 0.220 L turn⁻¹ Mg⁻¹ manure for straw and wood chip bedding treatments, respectively (B.S. Freeze, personal communication, 2002), and on a CO₂-C emission rate of 2.73 kg C L⁻¹ diesel fuel.

equivalent), its emission is a concern because N₂O is important to the troposphere radiation balance and in stratospheric ozone chemistry.

Application of compost to agricultural land has been reported to increase soil C content, and is regarded as a means of C sequestration (Beauchamp and Voroney, 1994; Smith et al., 1998). However, for straw-bedded manure, total GHG emissions during composting in terms of CO₂-C equivalent (368.4 kg Mg⁻¹) were higher than the initial total C content (330.5 kg Mg⁻¹) in the manure, raising the question of the net benefit of composting in C sequestration. Crop production (photosynthesis) captures atmospheric CO₂, but composting releases a portion of this captured CO₂ as CH₄, a more harmful GHG. If livestock manure were directly applied to agricultural land, there would be no CH₄ emission loss from soil since soil (except for waterlogged soil, such as a rice paddy) acts as a sink for atmospheric CH₄ (Hütsch, 2001).

CONCLUSION

In summary, total cumulative GHG emissions during composting were not significantly different between bedding treatments. The use of wood chips as a bedding material in feedlot operations instead of traditional cereal straw may not affect GHG emissions but there was less overall N loss during composting. However, the emission of CH₄ during composting is a concern, raising the question of the net benefits of composting on C sequestration.

Our study is one component addressing the much larger and more complex question of mitigating GHG emissions using manure management technologies. Further study is needed to fully evaluate the impact of composting on overall GHG emissions between feedlot pen and field as well as emissions after compost is applied to soil. For example, GHG emissions from soils receiving fresh manure and compost should be compared and these differences factored into the overall GHG mass balance for these manure management options.

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