

Chapter 22

Greenhouse Gas Emissions from Cattle Feedlot Manure Composting and Anaerobic Digestion as a Potential Mitigation Strategy

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Livestock production is a significant source of greenhouse gas (GHG) emissions, and the management of livestock manure is responsible for approximately 3% of CH₄ and 12% of total N₂O emissions in Canada. Composting is a manure management strategy that reduces the impact of land applied manure to the environment. However, nitrogen loss and GHG emissions are known to occur during the composting process. Anaerobic digestion is an alternative manure management strategy that can be utilized in conjunction with composting to limit nitrogen loss and reduce GHG emissions. A discussion focused on comparing different manure management strategies, the biological mechanisms at work in each, and their impact on GHG emissions will be undertaken in this chapter.

Introduction

Emission of greenhouse gases (GHG), such as CH₄ and N₂O, is a major contributing factor to global warming (1). Worldwide livestock numbers are growing as per capita meat consumption is projected to double from 1964-1966 levels by 2030 (2), increasing the significance of animal production systems on GHG emissions (3, 4). Currently, animal production systems are responsible for contributing up to 18% of total global GHG emissions (5), including approximately 5-30% of total CH₄ and 7-18% of total N₂O (6-10). Livestock manure management specifically contributes 30-50% of total agricultural N₂O

emissions (11), and 12-41% of total agricultural CH₄ emissions depending on the country considered (12).

The main greenhouse gases associated with livestock manure management are CH₄ and N₂O, which exhibit global warming potentials of 25 and 298 times that of CO₂ over a 100 year time horizon (or 72 and 289 times that of CO₂ over a 20 year time horizon) (13). In addition to its global warming potential, N₂O can also adversely affect the ozone layer. The photochemical decomposition of N₂O into NO results in depletion of stratospheric ozone (14), causing increased levels of ultraviolet (UV) radiation to reach the earth's surface (15). Methane oxidized by hydroxyl radicals in the troposphere increases tropospheric ozone levels, which enhances the greenhouse effect (15). Although significant amounts of CO₂ are produced during livestock manure management, this is considered GHG neutral because mineralization to CO₂ and H₂O is the optimal fate of all organic matter (16). As such, efforts to reduce GHG emissions from livestock manure are focused on CH₄ and N₂O.

Livestock manure management can be considered as a continuum, beginning with manure production from animals, proceeding to storage and treatment, and culminating in land application (12). Within this continuum, manure management can be approached in two ways: either the manure is immediately applied directly to a field, or more typically, it is stored in some way prior to land application. If fresh manure is applied directly to land, CH₄ production is negligible, particularly in comparison to manure storage, but emission of N₂O is prevalent (17). Fresh manure has a high moisture content, making transport away from intensive production areas costly. This in turn leads to over-application of raw manure to land immediately adjacent to production areas, which introduces problems with nutrient leaching and runoff into water resources (18). If manure is stored prior to land application, there are generally four different options available, each with its own advantages and disadvantages (Table I). The manure can be stockpiled, stored as slurry, composted, or anaerobically digested. For a review on the implications of different manure management strategies on GHG emissions, see Chadwick et al. (12). In the future, on farm disposal options for other types of agricultural wastes could also affect manure management practices. For example, disposal of animal mortalities (19–21), specified risk materials (22), or other agricultural wastes by co-composting with livestock manure or by anaerobic digestion (23, 24) would affect the GHG emission potential due to greater availability of readily degradable organic matter. In the last decade, composting has gained a foothold in the cattle feedlot industry in southern Alberta as an alternative to land spreading of fresh manure (25, 26). Composting decreases manure volume (27), and reduces or eliminates coliform bacteria (28), human parasites (29), plant pathogens (30), and weed seed viability (31). Despite its benefits, like most manure-handling or storage practices, composting is associated with N and C losses, some of which are in the form of GHG (32, 33). The focus of this discussion will be on GHG emission from composting livestock manure. Other manure management options such as anaerobic digestion and land application will also be briefly discussed.

Table I. Advantages and disadvantages of different manure management strategies with a focus on greenhouse gas emission potential

Manure Management Strategy	Advantages	Disadvantages
Direct Land Application	No CH ₄ emission Inexpensive Non-technical Enhanced sequestration of soil C Improve soil fertility	Time- manure storage is often necessary Nitrate leaching into groundwater N ₂ O emission Phosphate runoff into surface water Malodour No pathogen reduction Transportation costs
Manure Stockpiling	Inexpensive Non-technical	Highest GHG emissions Minimal pathogen reduction
Slurry	Automated Moderately expensive	Second highest GHG emissions Malodour High water content/low dry matter content Minimal pathogen reduction Semi-technical Water pollution
Compost	Inexpensive Organic fertilizer production Enhanced sequestration of soil C Improve soil fertility Malodour reduction Pathogen reduction Pest/weed reduction	Moderate CH ₄ and N ₂ O emission Semi-technical N loss due to NH ₃ emission Land requirement (compact earthen pad/catch basin) for compost site
Anaerobic Digestion	No GHG emission Renewable energy production Organic fertilizer production Malodour reduction Pathogen reduction	Expensive infrastructure Technical Economies of scale

Biological Mechanisms of GHG Production During Composting

Greenhouse gas production during composting is primarily the result of the microbiological breakdown of organic matter. Under aerobic conditions, organic matter is broken down into CO₂ and H₂O by the activities of a variable

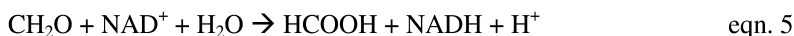
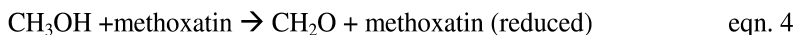
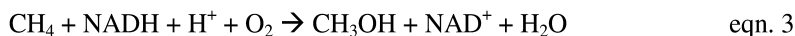
microbial consortium (including bacteria, archaea, and fungi) in a series of complex biological reactions. Part of the energy produced during these reactions is lost as heat and the remainder is used for cellular growth. Depending on the chemical composition of the substrate, complete mineralization to CO₂ and H₂O may not be possible, at least during a short enough time frame to be of practical use. For example, total degradation of complex polymers such as lignin, cellulose, or hemicellulose is unlikely to be complete even after several years, which contributes to carbon sequestration in soils over time (34).

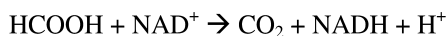
Under anaerobic conditions, the degradation of organic matter results in methanogenesis via the combined activity of three distinct groups of microbes (Figure 1) (35). Complex polymers are reduced to monomers via extracellular enzymes produced, and subsequently those monomers are fermented into organic acids, alcohols, H₂, and CO₂ by the activity of fermentative bacteria. Acetogenic bacteria convert larger organic acids to acetic acid and H₂, which in turn are converted to CH₄ by the action of either acetoclastic (eqn. 1) or hydrogenotrophic methanogenic archaea (eqn. 2) (36). Methylotrophic production of CH₄ from C1 compounds is also possible, but will not be discussed here (35). Methanogenesis does not occur, or at least is reduced, if alternate electron acceptors, such as sulphate, nitrate, Mn(IV), or Fe(III) are available (37).



Methanogens are slow growing, obligately anaerobic organisms with optimal growth at or near neutral pH. In compost, methanogenic activity is optimal within a pH range of 6.8-7.4 (38). The presence of oxygen or a pH < 6 is lethal to methanogens (39, 40). However, they are able to survive and remain active over a wide temperature range, from psychrophilic conditions (<20°C) to thermophilic (<70°C) (41, 42).

A group of obligately aerobic bacteria, comprised of 13 genera among the γ (Type I) and α (Type II) Proteobacteria and known collectively as methanotrophs (43), are able to utilize CH₄ as an energy source (44). Beginning with the activity of the methane monooxygenase enzyme, CH₄ is converted to methanol (eqn. 3). Formaldehyde produced from the oxidation of methanol (eqn. 4) can be fed into the ribulose monophosphate pathway (Type I) or the serine isocitrate lyase (Type II) pathway (Type II) (45). Otherwise, formaldehyde is converted to formic acid (eqn. 5), which is subsequently converted to CO₂ (eqn. 6).





eqn. 6

Methanotrophic bacteria also exist in soil and they are sensitive to the presence of a variety of chemicals, including herbicides and fertilizers (46, 47). In a study examining methanotrophic community dynamics in compost, Type I methanotrophs were predominant and species diversity was inversely correlated to compost temperature (45). It should be noted that low levels of CH₄ were produced in the Halet et al. (45) study, which favours Type I methanotrophs over Type II (48)(49), but that this may not be the situation in all compost piles. Methane oxidation can also be performed under anaerobic conditions by the actions of sulphate reducing bacteria and some archaea, although the importance of these processes is not yet well understood (50). Methane emission during composting is therefore not only a function of the rate of CH₄ production by methanogens but also the rate of CH₄ consumption by methanotrophs (51).

Nitrous oxide is a byproduct of nitrification (autotrophic or heterotrophic), denitrification, and nitrifier denitrification (52). In the case of autotrophic nitrification, NH₄⁺ is converted to NO₂⁻ in a two-step process by the joint activity of ammonia oxidizing bacteria (AOB) (e.g., *Nitrosomonas*) and nitrite oxidizing bacteria (NOB) (e.g., *Nitrobacter*) under aerobic conditions (Figure 2A). Ammonia oxidation by certain archaea (AOA) has also been reported, and their contribution may be greater than bacterial ammonia oxidizers within certain ecological niches (53, 54). In one study, the estimated population of AOA was greater than that of AOB, suggesting a strong role for AOA, particularly later in the composting process when temperatures and NH₄⁺ concentrations are reduced (55). Ammonia monooxygenase, which catalyzes the oxidation of NH₃ to NH₂OH, has a wide substrate specificity and can be inhibited by non-target substrates (e.g., acetylene) through competition or covalent binding to the active site of the enzyme. Hydroxylamine oxidoreductase catalyzes the oxidation of NH₂OH to NO₂⁻, and during this reaction N₂O can be produced as a byproduct by chemical decomposition (caused by organic or inorganic compounds, particularly at low pH) of NH₂OH or NO₂⁻ (56) or due to incomplete oxidation of NH₂OH (52, 57, 58). The oxidation of NO₂⁻ to NO₃⁻ is performed by NOB through the enzymatic activity of nitrite oxidoreductase. High concentrations of NH₃ are toxic to NOB such as *Nitrobacter* (52). In general, AOB and NOB are slow growing because of the poor energy return realized from using NH₃ and NO₂⁻, respectively, as substrates. To produce 1 g of dry microbial biomass, 30 g of NH₃ must be consumed (59), which indicates even a small nitrifier population can have significant impact on NH₃ conversion to NO₃⁻ and byproducts such as N₂O.

Heterotrophic nitrification is more prevalent in fungi than bacteria, but some heterotrophic bacteria have the ability to nitrify, and some can even denitrify under aerobic conditions (60). Although heterotrophic nitrifiers produce more N₂O per cell than autotrophic nitrifiers (61, 62), they are typically not considered a major source of N₂O. Under certain conditions (e.g., low pH, high O₂ concentration, plentiful organic C), the importance of heterotrophic nitrification could be increased (52).

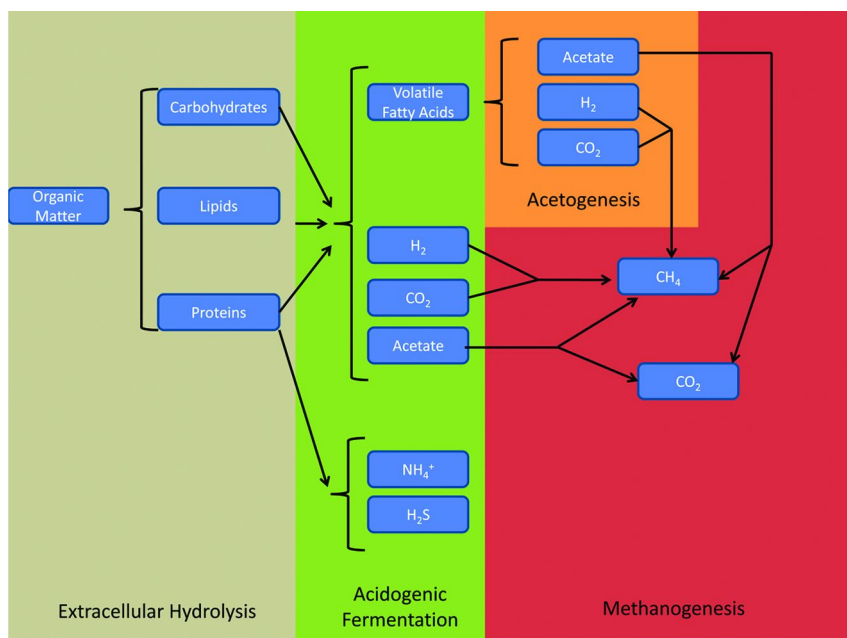


Figure 1. The anaerobic degradation of organic matter can be divided into four stages which are undertaken by three different groups of bacteria. Initial hydrolysis of complex organic matter (i.e., proteins degraded to amino acids, carbohydrates to sugars, lipids to fatty acids) and fermentation of monomeric components to produce volatile fatty acids (VFA), CO₂, H₂, acetate, NH₄⁺ and H₂S is performed by the first bacterial group. Acetogenesis is the conversion of VFA to acetate, H₂, and CO₂, and is performed by the second group. The final group of methanogens produces CH₄ either by splitting acetate or combining CO₂ with H₂.

Denitrification is the anaerobic reduction of NO₃⁻ to N₂, and the process is performed by a diverse group of bacteria (63). Dissimilatory denitrification resulting in N₂O formation can also be performed by certain fungi under anoxic conditions (64), but will not be discussed in detail here. Denitrifying bacteria are typically heterotrophic, facultative anaerobes. Nitrous oxide is a direct intermediate in the reduction pathway of NO₃⁻ (Figure 2B). Any disruption in the denitrification pathway can therefore lead to the emission of N₂O. For example, nitric oxide reductase is inhibited at low pH, which would allow accumulation and emission of N₂O (65). If the concentration of NO₃⁻ is high, reduction of N₂O is decreased as NO₃⁻ is the preferred electron acceptor (59). If oxygen is present in sufficient concentration, denitrification will stop. But under limited O₂ conditions, N₂O emission can be enhanced because nitrous oxide reductase is more easily inhibited by O₂ than the other enzymes in the pathway (52). This effectively makes N₂O the end product of denitrification. Although nitrification is

an aerobic process and denitrification is an anaerobic process, the two processes can be coupled in environments where an aerobic/anaerobic interface exists (66). In this situation, NO_3^- produced during nitrification is utilized by denitrifiers to produce N_2 .

In coupled nitrification/denitrification, three distinct bacterial groups are involved in the conversion of NH_3 to N_2 . However, in nitrifier denitrification, the entire process occurs in a single organism, which is an autotrophic NH_3 oxidizer (Figure 2C) (67). In contrast to coupled nitrification/denitrification, this process does not ever produce NO_3^- . The ecological niche for nitrifying denitrifiers likely requires low O_2 concentration, abundance of N, limited C, and possibly lower pH (52). The importance of nitrifier denitrification in emission of N_2O from soil environments varies widely in the literature, from insignificant (68) to 30% of total N_2O emission (69). Determining the ratio of N_2O emission caused by nitrification vs. denitrification vs. nitrifier denitrification is difficult due to the interdependence of the groups (70).

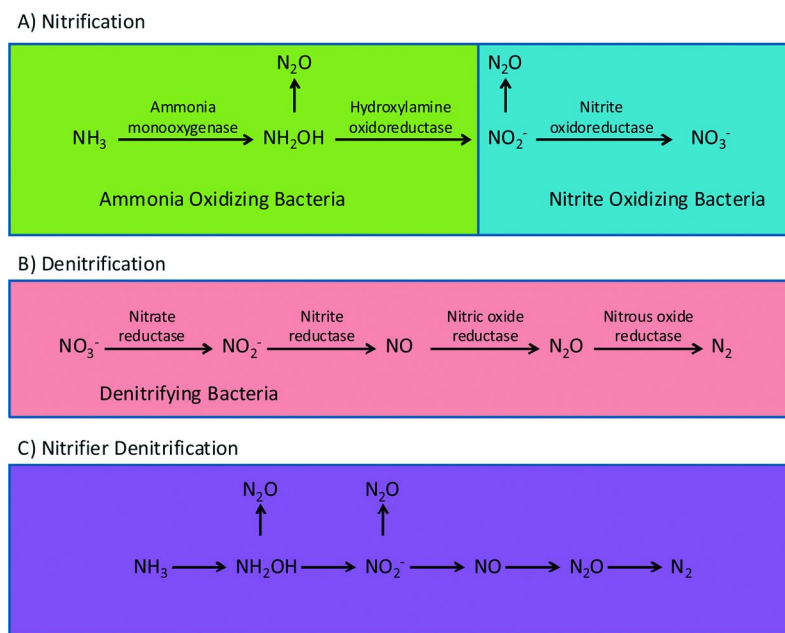


Figure 2. The processes of nitrification (A), denitrification (B), and nitrifier denitrification (C). Nitrification requires the activity of two different bacterial groups, represented by the separated boxes. At anoxic interfaces where the processes of nitrification and denitrification can both proceed, NO_3^- produced by nitrifiers can be consumed by denitrifiers. In nitrifier denitrification, both processes take place in a single organism.

Typical GHG Emission Patterns from Compost

The composting process has multiple phases which succeed each other over time (Figure 3). Substrate availability is highest at the start of composting, which leads to a rapid increase in microbial activity. Substrate degradation, CO₂ emission, and compost pile temperature are directly correlated (16, 71). Because heat is a byproduct of microbial metabolism, the compost pile temperature increases to thermophilic levels quickly (e.g., 70°C within 24 h of compost windrow establishment). Over time the amount of substrate available for metabolism decreases, causing a reduction in microbial activity and heat production. This gradual reduction in temperature shifts the compost pile from thermophilic to mesophilic. If the compost pile is turned to increase aeration, a temporary increase in temperature can be expected. The temporal changes in compost pile temperature and organic matter composition leads to microbial community succession (72, 73). Bacteria dominate the thermophilic phase while the prevalence of fungi increases as temperatures are reduced to mesophilic or ambient levels. The heating pattern of compost piles produces a “chimney effect” in which an efflux of heated air exits at the top of the pile, while fresh air is brought into the pile from the bottom and sides (16, 74). As such, most GHG emissions occur from the top of the pile and not the sides (75).

Another result of the high initial microbial activity is that O₂ is rapidly consumed, leading to the formation of anaerobic zones within the compost pile (Figure 4). Because O₂ is consumed during aerobic degradation of organic matter, O₂ concentration within the compost pile will decrease during the thermophilic phase of composting. In areas where little or no O₂ is present, anaerobic degradation of organic material occurs. Unlike aerobic degradation, which primarily yields CO₂ and H₂O, anaerobic degradation primarily results in CH₄ production by the activity of methanogens (Figure 1). As mentioned previously, methanogens are sensitive to O₂ and are only active under strictly anaerobic conditions. Such conditions occur temporally early in the composting process when microbial activity and O₂ consumption are at their peak, and spatially deep within the compost pile (32). Emission of CH₄ from compost piles is therefore found predominantly at the beginning of the composting process and is negligible as microbial activity decreases later on (Figure 3) (32, 76).

Methanotrophic activity reduces the amount of CH₄ emitted from compost, oxidizing CH₄ as it moves upward from anaerobic zones within the pile due to the “chimney effect” (74). Methanotrophic microbes have been detected in compost piles both near the surface and near the middle, suggesting their activity could have a broad spatial range within the pile (77). However, CH₄ emission can still occur for two main reasons. The first is that as the peak of CH₄ production occurs shortly after the initial start of the composting process, the methanotrophic community does not have adequate time to become established, allowing some CH₄ to escape out of the pile. The second reason is that methanotrophic oxidation of CH₄ is not completely efficient, so at peak CH₄ production some will be emitted.

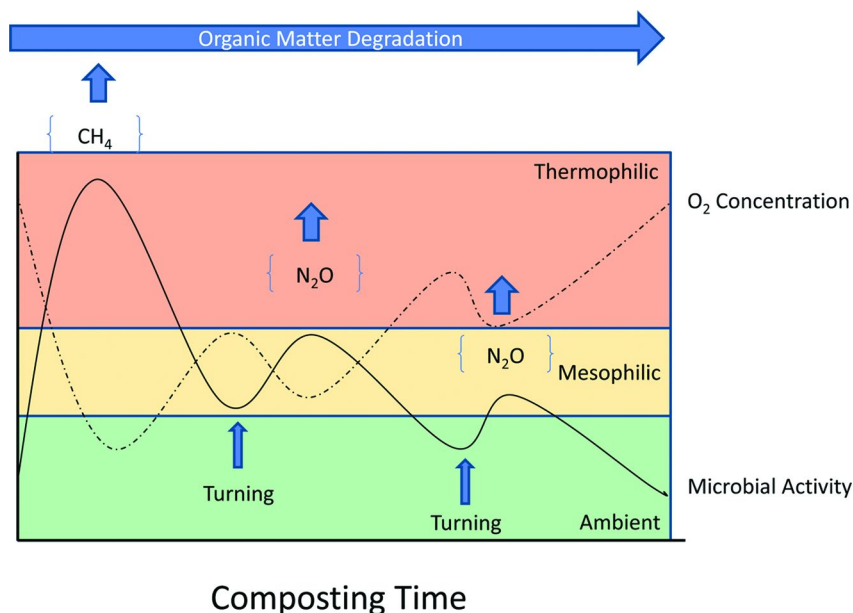
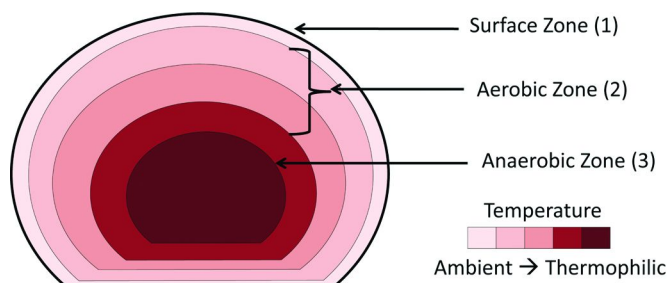


Figure 3. Typical pattern of microbial activity, temperature profile, oxygen concentration, and greenhouse gas emissions over time during composting. Thermophilic temperatures are rapidly achieved due to microbial degradation of organic matter which consumes oxygen and results in anaerobic zones where CH_4 production can occur via methanogenesis. Over time the temperature declines to mesophilic levels, enabling the growth of nitrifying bacteria which may cause N_2O emission. After turning events, NO_2^- accumulated in the surface layer of compost can be mixed down into anaerobic zones of the pile, causing N_2O emission due to denitrification.

Emission of N_2O from compost piles varies according to manure source and management strategies. When composting animal manures, N_2O emissions typically represent approximately 5% of total N loss (78, 79). Depending on compost management, N-loss due to N_2O can range from <1% to >6% (80). Emission of N_2O from compost is primarily due to two different processes. The first is nitrification, which occurs near the aerobic outer surface of the compost pile (Figure 2). Nitrification occurs near the compost surface, where O_2 concentrations are highest (79). The activity of autotrophic nitrifying bacteria is typically considered to be inhibited at temperatures above 40°C (81), so N_2O emissions due to their activity typically occur either immediately after the compost process starts, but before thermophilic phase, or after the temperature has receded to mesophilic levels (16, 79). The existence of thermophilic ammonia oxidizing bacteria has been reported in compost, though their importance in the nitrogen cycle within compost piles is not well studied (82). Ammonia oxidation increases rapidly during the first days of composting, reducing NH_3 emissions (83). The second primary source of N_2O emission occurs immediately after pile construction (if NO_2^- had accumulated in the manure to be composted during

storage) or after compost turning has occurred (84, 85). In this scenario, NO_3^- produced through nitrification near the compost surface is mixed down into the middle of the compost pile, which may be anaerobic. Anaerobic conditions foster denitrification, which can produce N_2O if not completed (see discussion above). Additionally, leaching of highly mobile NO_3^- from the top layer down into anaerobic zones can result in N_2O emissions from incomplete denitrification (22, 78). Of the two processes, nitrification near the surface of the compost pile is responsible for the majority of N_2O emissions from compost piles according to some studies, and this emission is tightly correlated to accumulation of NO_2^- (76, 86). However, other studies suggest that the majority of N_2O emission occurs as a result of denitrification immediately after turning events (70). Reduction of N_2O to N_2 is reported to occur sparsely throughout the composting process (70).



Process	Primary Products	Zone	Conditions
Aerobic Organic Matter Degradation	CO_2 , H_2O , NH_4^+	1,2	Requires O_2
Anaerobic Organic Matter Degradation	CH_4 , CO_2	2,3	Absence of O_2
Nitrification	NO_3^- , N_2O	1	Requires O_2 , NH_3
Denitrification	N_2 , N_2O	3	Requires NO_2^-
Nitrifier Denitrification	N_2 , N_2O	1	Low O_2 , high N, low C
Methanotrophy	CO_2	1,2	Requires O_2

Figure 4. Spatial profile of the oxygen gradient found in compost. At the center of the pile, oxygen concentration would be lowest, leading to the formation of anaerobic zones. Near the compost pile surface, aerobic conditions would exist. The area between the aerobic surface and the anaerobic center is a heterogeneous gradient. At anoxic interfaces, the activities of aerobic and anaerobic microbes may be in contact with one another. Although certain processes are typically associated with particular locations in the pile, the heterogeneity of compost makes exact spatial allocation of these activities imprecise.

Emission of GHGs from compost is highly variable, even when considering similar substrate types (e.g., cattle manure) (87). Given heterogeneity in the substrate itself as well as the impact of environmental factors, GHG emission patterns and total emissions can vary widely (Table II). The diet of animals affects the composition of their manure, which in turn can alter GHG emissions during composting. For example, beef cattle fed dried distillers' grains and solubles (DDGS) have higher concentrations of $\text{NH}_4^+\text{-N}$ in their manure, which subsequently leads to increased N_2O emissions during composting (88). The presence of trace chemicals, such as antibiotics, can alter the microbial community composition and potentially affect the compost process (87).

Managing Compost Conditions for Minimizing GHG Production

As discussed previously, the most important factor for limiting GHG emissions from compost is to limit the formation of anaerobic zones which are responsible for CH_4 production and denitrification-based N_2O production. Many factors can contribute to the formation of anaerobic areas, including moisture content, bulk density, manure composition, pH, compost pile size and dimensions, and turning frequency (71, 89). Over time organic matter degradation leads to compaction of the pile, reducing pore space and making aeration more difficult until the pile is turned (90). A variety of strategies can be employed to manage the conditions responsible for GHG emissions during composting. In general, compost can either greatly reduce (-900 kg CO_2 equivalent (eq.) tonne⁻¹ waste) or be a source (+300 kg CO_2 eq. tonne⁻¹ waste) of GHG depending on the efficacy of the strategies employed and the costs of various inputs (91).

The overall size and dimensions of the compost pile can be manipulated to maximize O_2 concentration within the pile. In general a larger compost pile will emit more CH_4 than a smaller compost pile, as increasing pile size also increases the likelihood of anaerobic zones developing, spurring CH_4 formation (79, 92, 93). As the compost pile matures, anaerobic zones will decrease in size and eventually disappear, but in larger piles this zone will persist for a longer period of time (94). It should also be noted that increasing piles increases the heterogeneity of GHG emissions, making accurate measurements more difficult (93). Bench scale composters may not emit any CH_4 at all (95), but such small piles are not possible at industrial scale. Large piles emit more total N_2O , and do so over a longer period of time, than a smaller pile (79). Increasing the compost pile's ratio of surface area to volume will also reduce the formation of anaerobic zones. For conically shaped compost piles, the volume of anaerobic zones increases logarithmically against the volume of aerobic zones as pile size increases (79). Utilizing smaller compost piles in conjunction with more frequent turning can reduce GHG emissions by maintaining aerobic conditions (93).

Table II. Summary of greenhouse gas emissions from various composting studies

<i>Compost material</i>	<i>Method</i>	<i>CH₄ emissions</i>	<i>N₂O emissions</i>	<i>CO₂ emissions</i>	<i>Reference</i>
Beef manure	Windrow	0.742 g m ⁻² d ⁻¹	NA	23.0 g m ⁻² d ⁻¹	(117)
Beef manure + cattle mortality	Windrow	2.204 g m ⁻² d ⁻¹	NA	53.6 g m ⁻² d ⁻¹	(117)
Beef manure	Windrow	2.60 g m ⁻² d ⁻¹	0.098 g m ⁻² d ⁻¹	129.1 g m ⁻² d ⁻¹	(87)
Beef manure DDGS diet	Windrow	2.42 g m ⁻² d ⁻¹	0.213 g m ⁻² d ⁻¹	115.6 g m ⁻² d ⁻¹	(87)
Beef manure	Passive windrow	132.3 kg CO ₂ eq. Mg ⁻¹	34.1 kg CO ₂ eq. Mg ⁻¹	73.8 kg CO ₂ eq. Mg ⁻¹	(32)
Beef manure	Active windrow	170.1 kg CO ₂ eq. Mg ⁻¹	58.9 kg CO ₂ eq. Mg ⁻¹	168.0 kg CO ₂ eq. Mg ⁻¹	(32)
Beef manure	Windrow	322.6 kg CO ₂ eq. Mg ⁻¹	1.6 kg CO ₂ eq. Mg ⁻¹	93.7 kg CO ₂ eq. Mg ⁻¹	(108)
Beef manure + 30% phosphogypsum	Windrow	278.9 kg CO ₂ eq. Mg ⁻¹	1.95 kg CO ₂ eq. Mg ⁻¹	100.3 kg CO ₂ eq. Mg ⁻¹	(108)
Beef manure + mtraw + mattle mortality	Windrow	1.57 g d ⁻¹ m ⁻²	0.17 g d ⁻¹ m ⁻²	31.6 g d ⁻¹ m ⁻²	(22)
Beef manure + straw + SRM	Windrow	0.47 g d ⁻¹ m ⁻²	0.17 g d ⁻¹ m ⁻²	24.3 g d ⁻¹ m ⁻²	(22)
Beef manure	Windrow	0.053 kg Mg ⁻¹	0.004 kg Mg ⁻¹	10.8 kg Mg ⁻¹	(88)
Beef manure + TYL11 antibiotic	Windrow	0.074 kg Mg ⁻¹	0.032 kg Mg ⁻¹	34.4 kg Mg ⁻¹	(88)
Cattle and horse manure	Windrow	NA	0.5% of total N	NA	(84)
Swine manure + straw	Passive pile	0.2 % of total C	0.8 % of total N	8 % of total C	(74)
Swine manure + cardboard	In vessel	NA	0.1 % of total N	NA	(85)
Grass + green waste	In vessel	1.7 % of Total C	0.5 % of total N	81 % of total C	(99)

<i>Compost material</i>	<i>Method</i>	<i>CH₄ emissions</i>	<i>N₂O emissions</i>	<i>CO₂ emissions</i>	<i>Reference</i>
Swine manure	Small pile in controlled chamber	1.0 g kg ⁻¹ OM	37.2 g kg ⁻¹ total N	NA	(79)
Swine manure	Large pile in controlled chamber	1.9 g kg ⁻¹ OM	46.5 g kg ⁻¹ total N	NA	(79)
Swine manure	Pile in controlled chamber	NA	88.5 g kg ⁻¹ total N	NA	(114)
Swine manure + NOB	Pile in controlled chamber	NA	20.2 g kg ⁻¹ total N	NA	(114)

Another strategy for limiting GHG emissions from compost piles is to modify the compost turning strategy. Compost turning introduces O_2 throughout the pile, restores pore space structure, and breaks up anaerobic zones. Compared to manure stockpiling, turning or forced aeration reduces CH_4 emissions significantly (96, 97). However, turning also transfers some NO_2^- from the aerobic surface zones into the middle of the compost pile (32). In only 12 h after a turning event, the O_2 levels can fall below 3%, potentially fostering denitrification of NO_2^- and subsequent N_2O emissions (32). In a study comparing static pig manure compost piles, monthly turning resulted in $\sim 4\times$ less N_2O emission than with no turning (90). Similarly, the amount of organic C degradation resulting in CH_4 formation was only 0.5% in turned piles, compared with over 12% in unturned piles. Frequent compost turning, or even forced aeration, does not completely prevent the formation of anaerobic zones (32, 90, 98–101), so this strategy is only partially effective in mitigating GHG emissions. Frequent turning has also been reported to increase N-loss via NH_3 (102). In some circumstances, frequent compost turning may actually increase GHG emissions compared to unturned static piles (32). In that study, frequent turning reduced CH_4 concentration but almost doubled N_2O emissions (Table II). Compost turning also requires the use of heavy machinery, so GHG emissions due to fossil fuel use must also be considered in the overall GHG emission calculations (33, 91). However, optimizing the composting process by balancing turning frequency with optimum water content and C/N ratio will shorten the time required to reach compost maturity and thus shorten the window of GHG emission during composting (89, 103).

Bulking agents such as wood chips and straw can be incorporated into the compost mix to reduce bulk density and enhance O_2 diffusion into the compost pile, limiting the formation of anaerobic zones (33, 104, 105). Increasing pile density results in greater N_2O emissions (106). Inclusion of straw when composting pig or cattle manure results in reduced bulk density and reduced GHG emissions (74, 105). Enhanced air circulation from straw inclusion reduces the development of anaerobic zones, but increased aeration can also enhance NH_3 emission (107). While this NH_3 loss reduces N_2O formation due to nitrification (74), it also lowers the N-content of the final compost product, reducing its value as a fertilizer (32). Kuroda et al. (85) found that little CH_4 was emitted when swine manure was composted in a mixture with cardboard in a lab scale active aeration system. However, some N_2O emission still occurred despite the maintenance of aerobic conditions, most likely due to oxidation of NH_3 to NO_2^- and subsequent conversion to N_2O . Moisture content also plays a role in the efficacy of O_2 diffusion within the compost pile. If there is too much water then pore spaces have less volume available for gas diffusion, increasing the likelihood of anaerobic zones forming.

Although pile turning and addition of bulking agents can reduce the prevalence of anaerobic zones, such strategies cannot completely prevent their formation in full scale compost windrows (32, 98). A different strategy for reducing GHG emissions involves manipulation of the microbial community structure through the addition of amendments to the compost pile. One such amendment is phosphogypsum (PG; $CaSO_4 \cdot 2H_2O$), which is a byproduct of the phosphate fertilizer and construction industry (108). Addition of PG to compost

increases the S content while decreasing pH. Lower pH has been associated with increased N_2O emissions from soil (109). When added in sufficient quantities (20 or 30% by dry weight), PG reduces CH_4 emissions while having no effect on CO_2 or N_2O emissions (108). The reduction in CH_4 emission is likely due to the activity of sulphate reducing bacteria (SRB), which utilize sulphate as an electron acceptor and out-compete methanogens for nutrients (37, 110). Additionally, a variety of sulphur compounds such as SO_4^{2-} , S^{2-} , and SO_3^{2-} adversely affect methanogen activity (111, 112).

Another compost amendment of interest is the addition of nitrite oxidizing bacteria (NOB). The activities of ammonia oxidizing bacteria (AOB) lead to NO_2^- formation, while NOB activity prevents NO_2^- accumulation (Figure 2A). Under typical compost conditions, AOB produce NO_2^- at a greater rate than NOB can consume it, leading to NO_2^- accumulation. This is due to NOB having a slower growth rate (113) and their sensitivity to free NH_3 and nitrous acid (81). Since accumulated NO_2^- can yield N_2O through chemodenitrification or nitrifier denitrification (Figure 2B, C), supplementing the initial NOB population can reduce N_2O emission (32, 114). If the total concentration of NOB can be increased to 10^6 cells g^{-1} compost, the activities of AOB and NOB can be balanced, reducing NO_2^- accumulation and in turn reducing N_2O emissions by up to 80% (114). Applying finished compost, which is rich in NOB, to mesophilic stage compost resulted in a 70% reduction in N_2O emission from swine manure in laboratory scale composters (115). Certain chemicals, which act as nitrification inhibitors that have been demonstrated to reduce N_2O emissions from soils by preventing NO_2^- , could potentially be applied to compost to suppress N_2O emissions (70).

Co-composting animal mortalities and SRM for control of infectious disease outbreaks has been investigated increasingly in recent years (19–22, 116). From a GHG emission perspective, inclusion of animal tissues increases organic N levels and additionally favors the development of anaerobic zones within the compost pile due to poor aeration and accelerated microbial activity. In general, mortality composting results in greater GHG emissions than basic manure composting (116, 117). Comparing GHG emissions from SRM vs. intact beef cattle carcasses, the latter emitted more CH_4 while CO_2 and N_2O levels did not significantly differ (22).

Composting, Soil C Sequestration, and Reduction in Chemical Inputs for Soil Fertility

Thus far the focus of the discussion has been on GHG emissions from active compost piles. Another important aspect to consider is the long term effect of composting on enhancing and maintaining agricultural soil fertility. As discussed previously, the majority of CO_2 emissions from compost occur during the active or thermophilic phase. Over time CO_2 emission decreases as organic matter degradation slows. Eventually the compost process is completed and the finished product can be applied to soils. Storage of finished compost does not significantly contribute to further GHG emissions, although it does increase the mineral N content which is advantageous from a fertilization perspective (118). Finished compost applied to soil will still slowly emit CO_2 as it is further degraded (103).

Estimating the lifespan of soil organic C from compost applied to land is difficult based on the variability of many factors, such as soil type, temperature, and moisture, all of which affect the C turnover rate (119). However, it has been reported that 9% of organic C applied to soil as compost will remain after 100 years (120). This implies that sequestration of C in the soil is limited over a long time horizon, but application of compost derived organic C has many other GHG reduction benefits as well.

Application of compost to soil enhances soil fertility, stabilizes soil structure, enhances water holding capacity, and reduces the need for inputs such as chemical fertilizers and pesticides (34). Production of P_2O_5 fertilizer from rock phosphate requires approximately 243 kWh tonne⁻¹ (121). Applying 10 tonnes of compost ha⁻¹ would displace 190 kg of N fertilizer, providing an energy savings of 160-1590 kWh in fertilizer production costs alone. Over 20 years, the application of 10 tonnes compost ha⁻¹ would offset 60-600 kg of CO₂ eq. N₂O compared to using chemical fertilizer. In addition to the above benefit, N₂O emissions from chemical fertilizers are greater than from compost amendment (34). Application of compost to soils has a negligible impact on overall GHG emissions from soil while improving soil N and C (17). In addition to nutrients and degradable substrates, compost application to soil also includes active microorganisms (122), which indirectly alter the microbial community composition of soil (123). Because the final compost product is relatively dry, odorless, and free of pathogens, transport and application to land a greater distance away from areas of intensive manure production is more economical, reducing the potential for excessive nutrient loading into water resources.

Anaerobic Digestion for Pretreatment of Manure

Anaerobic digestion (AD) is an engineered, tightly controlled process that takes advantage of the activities of anaerobic microbes to convert organic waste into CH₄ and CO₂ (Figure 1). Anaerobic digestion can be applied at a variety of temperature ranges, operating conditions, and using different reactor configurations, which will not be discussed here (124). Typically the ratio of CH₄ to CO₂ produced during AD is 2:1 (by % concentration). The yield of CH₄ from AD of manures ranges from 0.2-0.4 L CH₄/g VS for swine, 0.2-0.3 L CH₄/g VS for beef, and 0.3-0.35 L CH₄/g VS for poultry (124). Instead of being emitted to the atmosphere, the CH₄ produced is combusted for energy production. The net result of AD coupled with combustion of CH₄ is that only CO₂ and H₂O are emitted, making the process GHG neutral. This assumption is predicated on the fact that there are no fugitive CH₄ emissions from the AD reactor or the long-term effluent storage tank for treated manure (124). Although NH₄⁺ is produced during AD, the absence of O₂ prevents nitrification (and subsequently denitrification), greatly decreasing the potential for N₂O formation during digestion (12).

One of the most desirable aspects of composting manure is that the final compost product is a good fertilizer and soil conditioner. The percentage of organic matter converted to gas during AD varies depending on the degradability of the substrate and the process conditions employed, but volatile solid degradation

rates of over 50% are typical. The remaining organic matter consists primarily of components that are too difficult to degrade within the time frame of the AD process (typically days vs. months for composting). These remaining solids can either be collected and composted or directly applied to land as slurry, and used as fertilizer and soil conditioner. Some care must be taken when applying AD slurry, as it contains high levels of NH_4^+ , which can be lost through volatilization if not injected into soil (125). Acidification of slurry may be a method to enhance stabilization of NH_4^+ and limit volatilization (126). Anaerobic digestion has additional benefits compared to land application of raw manure such as pathogen reduction, elimination of malodour, and prevention of nutrient leaching or runoff (5).

Slurry that has been processed by AD prior to land application may emit less N_2O than untreated slurry (127–130), but a significant reduction is not always observed, likely due to the complex nature of N_2O emission after land application (124). During storage anaerobically digested slurry produces 30–66% less CH_4 than non-digested slurry (128, 131). In a dairy manure system, implementing AD instead of slurry storage could result in CH_4 reduction of ~ 300 kg CO_2 eq. per livestock unit (132). The same authors found that implementing AD in that system would enhance N_2O emissions after field application, but that the reduced need for additional N-requirements helped offset this disadvantage. Solids that are subjected to AD and then composted emit significantly less GHGs when compared to composting raw solids (133).

In a study comparing GHG emissions from beef or dairy manure that was either stockpiled, stored as slurry, or composted, compost was found to produce the lowest CH_4 emissions but the highest N_2O (10). While the pattern was the same for both beef and dairy manure, CH_4 and N_2O emissions from dairy manure were higher than for beef. Comparisons of total GHG estimates revealed that compost produced the lowest emissions, followed by manure stockpiling, and finally slurry storage. Taking into account subsequent land application of the composted or stored manure, the emission hierarchy remained the same. If applied to Canada, switching all slurry and manure stockpiling to composting would result in a GHG reduction of 0.7 Tg CO_2 eq. In contrast, if all manure was stored as slurry and then processed by AD, a 1.08 Tg CO_2 eq. reduction in GHG would be possible.

Summary

Composting livestock manures offers a management strategy with advantages and disadvantages. The cost of GHG and ammonia emissions must be balanced against the other environmental and economic benefits, such as increased soil fertility and structure, reduced reliance on chemical fertilizers, and control of odors and nutrient displacement. Composting within enclosed facilities, potentially enabling recapture or removal of fugitive GHG and ammonia emissions, could further enhance the benefits of composting. Where feasible, anaerobic digestion can be used in place of, or in conjunction with, composting to maximize the benefits of nutrient recycling via manure management while reducing GHG emissions.

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